Chad Herring, NC Farm Families
Public Hearing on Digester Permit
April 5, 2022

Good evening. My name is Chad Herring, and I am a third-generation family farmer from Mount Olive. I also serve as executive director of North Carolina Farm Families, and I'm proud to represent thousands of family farmers across our state.

Farmers like me believe in protecting our environment by taking care of our water, air and land. Generating renewable natural gas on pig farms is one way that we can contribute to reducing greenhouse gas emissions. Reducing those harmful emissions is a goal that we all share — the State of North Carolina and the Environmental Protection Agency have both highlighted the need to reduce greenhouse gases and promoted the benefits of generating biogas.

Generating renewable natural gas on farms is a positive step forward — for our farmers, for our community, and for our environment. It reduces emissions, reduces odors, and reduces the potential for flooding — and everyone can agree that those three things are beneficial.

These systems help make our farms more sustainable, and farmers who are thinking about investing in a digester system need a predictable and consistent permitting process.

The draft general permit takes appropriate measures to ensure that digester systems are tested and safe before beginning operation. Please do not add unreasonable and unnecessary conditions like the ones being proposed here tonight. Thank you.

# # #

Good evening. My name is Will Hendrick and I'm with the North Carolina Conservation Network.

First, I want to thank DEQ for heeding the procedural advice of the EJEAB by providing advance notice of hearings, holding hearings in person, holding them in impacted communities, providing interpretation & translation services, and extending the comment period.

However, I'm disappointed that DEQ, when drafting the permits themselves, ignored the substantive advice from the same Board. Environmental justice demands not only equitable inclusion in the decision making process, but also agency decisions that acknowledge and address the disparate impact of pollution on communities of color and low income. And, as Governor Cooper has stated in a recent executive order, it is imperative, even as we strive to reduce greenhouse gas emissions, that we do so in a way that incorporates environmental *justice and equity considerations.*

I also want to address two more things DEQ is ignoring. First, DEQ is ignoring the fact that there is technology–indeed even technology compatible with biogas production—that can better protect our environment and public health. For instance, covering a lagoon doesn't increase the nitrogen content in animal waste, but it changes the amount of nitrogen present in its inorganic form, elevating the threat to waters and neighbors posed by mismanagement of an even more potent waste stream. As such, I'm disappointed by the failure of the permit to respond to the increased threat of ammonium discharges and nitrate pollution of groundwaters posed by covering lagoons without requiring components like nitrification-denitrification basins to mitigate these impacts. DEQ should require technology that reduces threats, rather than blessing technology that increases them. The technology exists to continue farming, but stop harming, and DEQ should require its use.

I'm also discouraged that DEQ continues to ignore harm by requiring absolutely no monitoring of groundwater or surface waters absent a self-reported spill. The agency is at best assuming, and at worst willfully overlooking, the impacts, individually and cumulatively, of permitted operations. In court, you're arguing that there are no impacts, because the permit is supposed to prevent them and you conduct an annual inspection to ensure compliance. But that's like saying my car isn't leaking oil, or indeed that no car on the road is leaking oil, because everyone got their car inspected last year. *If you don't look, you don't know and you shouldn't claim* otherwise. I suppose that's an imperfect analogy though; no court has found that my car is harming property rights, but 5 juries have reached that conclusion about DEQ-permitted swine operations. And the EPA hasn't lamented the disparate impact of my driving on vulnerable North Carolinians, but the agency has expressed grave concern about the disparate impacts of permitted operations that you're continuing to ignore. This permit should require monitoring to ensure and evaluate compliance with the non-discharge requirement on which your unchecked assumptions rest.

After all, ultimately it doesn't matter if you enable community participation through these public hearings if, at the end of the day, you ignore what you hear and continue ignoring harm that you have the power to prevent.

Draft General Permits – Digester Systems
Public Meeting
April 19, 2022 6 p.m.
Statesville, N.C. Civic Center

My name is David White, I live at 320 Pennsylvania Ave. in Winston-Salem. I am a former board vice-president and board member of the Yadkin Riverkeeper.

I am a conservationist. I believe in the careful management of our natural resources in partnership and cooperation with government, land owners and the companies who reside here and provide jobs for our citizens.

I am also an active year-round paddler on the Yadkin River and a vocal advocate for keeping the river and its lakes clean for increasing recreational use and the economic benefits that brings.

However, when I see issues which potentially could affect the Yadkin River watershed, particularly drinking water and recreational uses of the river and lakes, I believe it is my responsibility to speak up and that's why I'm here today.

I am patently against creating a new general permit for farm digester systems. On the surface it sounds like a real solution to the problem of animal waste produced by farming, in particular dairy farms: turn manure into usable energy.

Unfortunately, as a solution for manure management, digester systems are hardly a simple or complete solution, but rather one step in an insufficient and fallible process. In the end, you still have a lagoon or cesspool full of manure.

For digester systems to be at all effective they require scale – or a large operation – lager than most of the current dairy farms in our watershed. In effect, this would encourage more Concentrated Animal Feeding Operations (CAFOs) which in turn would create more animal waste and potentially more nutrient and bacteria pollution in the South Yadkin River watershed.

The larger farms are more likely to build giant, unlined pits to store liquid manure. These giant cesspools release methane, carbon dioxide, and other noxious gases that can be fatal to humans and animals. They are also vulnerable to structural failures and extreme weather events, which can cause storage pits to overflow or leak. And "capping" these cesspools is simply not an acceptable management option either, particularly under a general permit as proposed tonight.

There are reasonable alternatives to farm digestive systems such as small-scale sewage/wastewater treatment plants. These systems can be cost-effective and address the issue of manure management.

I am NOT anti-farmer or anti-business or anti-growth. Quite the contrary. But I do welcome common-sense solutions that address the continuing issue of water quality. We simply can't sit back while we unwittingly create another ecological disaster like the Duke Energy coal ash ponds. Thank you.

Run Time: 2:50

My name is Alec Linton and I work for Brock Equipment Company located in Bailey and Calypso, NC. We are a family owned and operated business that, between these two stores, specializes in providing and servicing irrigation equipment to farmers across the state and is also a part of the RNG project.

The RNG project is an exciting opportunity for Brock Equipment since we get to partner with farmers to implement new and innovative technology right here in our state. Farmers are excited for innovation. If you don't believe me, go to your local farm show and watch them check out the latest equipment and technology. The innovation in RNG is no different. Since getting involved in RNG it has been an absolute joy seeing the excitement and hopefulness that these farmers show in the opportunity this project provides to better our farms and communities.

With that said, I believe in the work our farmers do on a daily basis. I believe in the technology and benefit of projects like RNG that help us manage our farms, protect the environment, and better our communities. It is frustrating to see people choose to bash and discredit farmer's efforts for progress. This is not only counterproductive, but it hurts family farms and family businesses like Brock Equipment. There are those that say they don't want to put anyone out of business, but that is exactly what they're doing.

Growing up, I lived next to a hog farm. I work on these farms every week and I see the hard work, passion, and dedication this industry has and I'm proud to be a part of it. At the end of the day, these communities are our communities. We live in them, work in them, and we care about them. I'm grateful to work in this industry and to live in these communities.

From all the employees and their families here at Brock Equipment, thank you for your time.

Comments, Tuesday, April 19, 2022

Joe Morris
8670 US Highway 601
Salisbury, NC 28147

1. My name is Joe Morris, resident of Rowan County, property owner on the South Yadkin River. Small acreage (18 acres) in the South Yadkin / Beaver Dam Creek sub basin, which flows into Second Creek, a short distance upstream of the confluence of the South Yadkin and the Yadkin Rivers – the location of the water intake facility for the Salisbury Rowan Utility Department on Hannah Ferry Road. I serve as a volunteer on the BOD of the Yadkin Riverkeeper.
    a. My farmstead is on a drinking water well. The well is 280' deep and located less than 300' from the South Yadkin River.
    b. On February 6, 2020, we experienced very high water that brough the water level within inches of a structure on my property and overtopped my neighbors well-head, setting off a process of disinfectant and a period without drinking water for his family.
    c. I am concerned about the level of unhealthy bacteria currently in the river and the frequency of e-coli measurements indicating conditions unsafe for recreational activities.
    d. I observe many fishermen across from our property who are catching fish for consumption, feeding their families with fish from these waters.
2. I am pro-farmer, pro-rural life-style and pro-river, these are not mutually exclusive, but for them to sustain, both independent from and dependent on one another, will require a balance of communication, cooperation and to some extent compromise. Of the three, we cannot afford to compromise the river any more. It is polluted. We are a rapidly growing population, which relies entirely on farmers for food and inherently requires rural land on which to produce food for human consumption.
    a. The river provides essential drinking water to innumerous communities in the Yadkin River Basin. It is imperative that we review, and fully understand the implications of the proposed bio-gas capture technology for what its promise might be (the laudable intent to reduce greenhouse gases), and the areas where the technology comes up short.
    b. Simply covering animal waste pools will not improve the issues at hand with water quality, especially the bacteria and nutrients finding their way into the river after every rain event.
    c. The notion of a single, system-wide permit process will alleviate pressure on farmers, which a noble pursuit, but will lump high-performing practitioners with those who may occasionally catch a bad break, or, sad to say, may just cut corners. I can imagine this will cause problems in the future.
3. Finally, I'd like to say that there are better ways to go about this.
    a. Increased monitoring of ground water and surface water in the basin is a beginning.
    b. Providing adequate resources to farmers to build and operate micro- waste water treatment plants to create affluent that is cleaner than the river is now exists as technology. Yes, it is expensive. But we cannot simply not afford not to do it.
    c. Thank you for the opportunity to comment.

RIVERPARK
at Cooleemee Falls
"The Bullhole"

RiverPark at Cooleemee Falls, colloquially known as "The Bullhole" is an 80-acre public park that sits astride the South Yadkin in northern Rowan County and in the town of Cooleemee. While we want you to clearly understand that RiverPark supports the farming community – as I myself own farmland – we have concerns that from a park perspective, and to safeguard the water quality and health of nearby residents, the permitting requirements should include surface and groundwater monitoring and limits for nutrients and bacteria. There is already excessive sedimentation, nutrient overload, and high fecal coliform bacteria (FCB) in the South Yadkin watershed from existing agriculture activities. Not doing so would harm economic development in the watershed around outdoor recreation, particularly in the case of RiverPark.

- The town of Cooleemee, as well as Rowan and Davie counties, has invested over one million dollars in redeveloping a historic mill site as a regional water recreation location. Continuing our improvements, we have just received a $250,000 appropriation from the State for further development.
- RiverPark at Cooleemee Falls has clearly become a regional destination park. Visitors regularly travel from Winston-Salem, Greensboro, High Point, Stanly County, Raleigh, Durham, Chapel Hill, Monroe, Lincolnton, Gastonia, Belmont, Fort Mill, and South Carolina. Out of state visitors are frequent.
- While I have been advised that the South Yadkin is classified as a Class-C or 'secondary contact recreation' watershed, Riverpark is actually Class-B 'primary contact recreation.' Visitors to the park can exceed 400 per day on a summer weekend. Most of these visitors find themselves in the river enjoying the falls, slides and tubing.
- In 2020, the Development Finance Initiative of the UNC School of Government presented to Davie County the outcomes of the feasibility study for the Cooleemee Mill and RiverPark. The bottom line of this study indicated that economic development in the Cooleemee area would not occur without the continued development of RiverPark.
- Yadkin Riverkeeper regularly tests our river water during the summer season. We have discovered that when the water reaches 3' above normal at the Foster Road measuring station located 8 miles upstream, we are guaranteed to have an e. coli problem at the park. This has become such a concern that we have requested grant funding for a method to warn park visitors of unsafe river conditions. No one wants to swim in a dirty river!
- The type of "Development" being discussed here does not support economic development around outdoor recreation and could possibly make it non-viable. Again, no one wants to swim in a dirty river. We should be looking to improve the situation, not implementing practices that can ~~increase~~ the risk.
- There is nothing else like RiverPark in our region. You really ought to come visit some time! CONTINUE

Respectfully submitted,

Addison Davis, President

Park: 645 Erwin Temple Ch. Rd. Woodleaf, NC 27054 – Mail: 135 S Salisbury St. Mocksville NC 27028
riverpark@cooleemee.org – www.cooleemee.org/riverpark

About RiverPark

- 80-acre nature park in northern Rowan County and the Town of Cooleemee that sits astride the South Yadkin River.
- *RP operates as a 501(c)(3) nonprofit, managed by a volunteer board of directors.*
- Opened in 2003, "The Bull Hole" earned its name some 200 years ago.
- The Visitor Experience includes swimming, hiking, picnicking, kayaking and canoeing, fishing and shelter rental.
- 2021 saw the implementation of full–time staff.

RIVERPARK
at Cooleemee Falls
"The Bullhole"

About RiverPark

RIVERPARK
at Cooleemee Falls
"The Bullhole"

RiverPark Board of Directors


Addison Davis
President
At Large Rowan


Paul Moore
Vice President
Davie Rec. & Parks


Carolyn McManamy
Treasurer
Davie Economic Dev.


Steve Allred
Secretary
At Large Rowan


Ron Bivins
Davie County
Commission


Carolyn Moser
Davie County
Tourism


Ashley Honbarrier
Rowan County
Parks & Rec.


Aaron Thies
Cooleemee
Town Clerk


John Peeler
At Large
Davie

Photo Pending
TBD
Rowan County
Commission

Photo Pending
TBD
Rowan Tourism
Development


Brian Williams
At Large
Davie

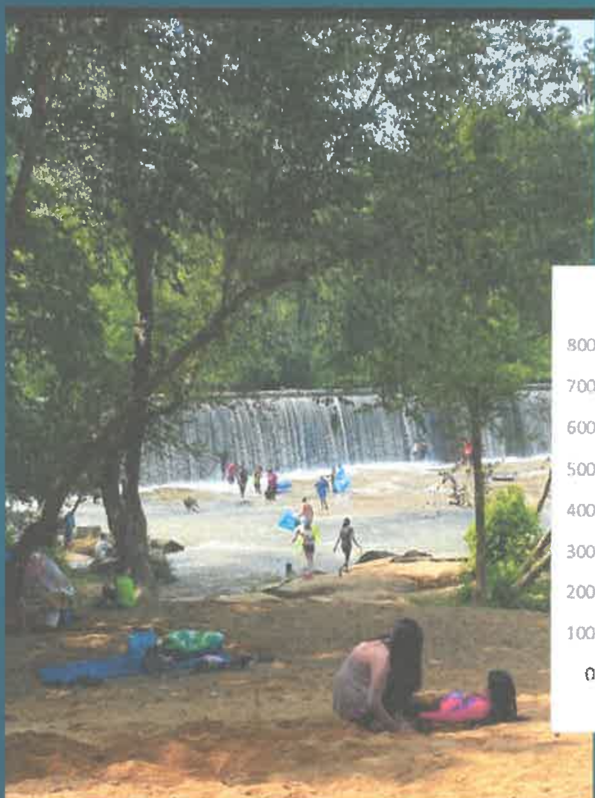
Photo Pending
TBD
Town of Cooleemee
Commission


Jonathan Vizard
RiverPark
Manager

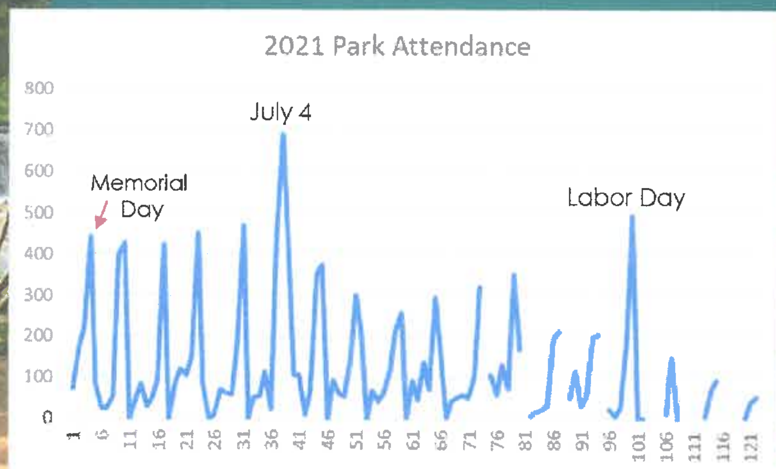
RIVERPARK
at Cooleemee Falls
"The Bullhole"

We continue to be discovered!

- Today, RiverPark hosts peak days of over 400 people in the park
- Visitors continue to come from throughout the region
- Growth is expected to continue with park improvements



Typical Summer Day at the Bullhole



INDEX OF EXHIBITS

Exhibit No.	Title
01	Letter from Blakely Hildebrand, SELC, to Ramesh Ravella, DEQ (Dec. 17, 2021)
02	Letter from Lilian Dorka, EPA, to William G. Ross, N.C. Dep't of Env't Quality, Letter of Concern (Jan. 12, 2017)
03	Letter from N.C. Env't Just. Equity Advisory Bd. to Elizabeth Biser, Sec'y, N.C. Dep't of Env't Quality (Oct. 22, 2021)
04	Letter from N.C. Env't Just. Advisory Bd. to Elizabeth Biser, Sec'y, N.C. Dep't of Env't Quality (Aug. 26, 2021)
05	Colleen N. Brown et al., *Tracing Nutrient Pollution from Industrialized Animal Production in a Large Coastal Watershed*, 192 ENV'T MONITORING & ASSESSMENT 515 (2000)
06	Michael A. Mallin & Lawrence B. Cahoon, *Industrialized Animal Production — A Major Source of Nutrient and Microbial Pollution to Aquatic Ecosystems*, 24 POPULATION & ENV'T 369 (2003)
07	JoAnn Burkholder et al., *Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality*, 115 ENV'T HEALTH PERSPS. 308 (2007)
08	Stephen L. Harden, U.S. Dep't of the Interior, U.S. Geological Surv., Surface-Water Quality in Agricultural Watersheds of the North Carolina Coastal Plain Associated with Concentrated Animal Feeding Operations (2015)
09	R.L. Huffman, *Seepage Evaluation of Older Swine Lagoons in North Carolina*, 47 TRANSACTIONS AM. SOC'Y AGRIC. ENG'RS 1507 (2004)
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INDEX OF EXHIBITS

16	Nina G.G. Domingo et al., County Level Data
17	N.C. Dep't Env't Quality, Technical Stakeholder Working Group Meeting Summary – Fall 2021
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20	Viney P. Aneja et al., *Characterizing Ammonia Emissions from Swine Farms in Eastern North Carolina: Part 2—Potential Environmentally Superior Technologies for Waste Treatment*, 58 J. Air & Waste Mgmt. 1145 (2008)
21	Viney P. Aneja, Summary of Expert Opinion Compiled by Dr. Viney P. Aneja, PhD (Sept. 3, 2021)
22	John T. Walker et al., *Atmospheric Transport and Wet Deposition of Ammonia in North Carolina*, 34 Atmospheric Env't 3407 (2000).
23	Michael Mallin et al., *Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution*, 226 J. Water Air Soil Pollution, 12, (2012)
24	*Cape Fear River Animal Feeding Operations Monitoring Study: Preliminary Report* N.C. Dep't Env't Quality (May 4, 2020)
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26	Viney P. Aneja et al., *Characterization of Atmospheric Ammonia Emissions from Swine Waste Storage and Treatment Lagoons*, 105 J. Geophysical Rsch. 11,535 (2000)
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28	Jennifer Costanza et al., *Potential Geographic Distribution of Atmospheric Nitrogen Deposition from Intensive Livestock Production in North Carolina, USA*, 398 Sci. Total Env't, 76 (2008)

INDEX OF EXHIBITS

29	Viney P. Aneja et al., *Agricultural Ammonia Emissions and Ammonium Concentrations Associated with Aerosols and Precipitation in the Southeast United States*, 108 J. Geophysical Rsch. 12-1 (2003)
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33	Kelley Donham et al., *Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations*, 115 Env't Health Persps. 317 (2007)
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37	Katherine Martin et al., *Terra Incognita: The Unknown Risks to Environmental Quality Posed by the Spatial Distribution and Abundance of Concentrated Animal Feeding Operations*, 642 Sci. Total Env't 887 (2018)
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INDEX OF EXHIBITS

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42	Lowry A. Harper et al., *Dinitrogen and methane gas production during the anaerobic/anoxic decomposition of animal manure*, 100 Nutrient Cycling in Agroecosystems 1 (2014)
43	Lowry A. Harper et al., *The Effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions*, 39 J. Env't Quality 1984 (2010)
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46	Alessandra Fusi et al., *Life Cycle Environmental Impacts of Electricity from Biogas Produced by Anaerobic Digestion*, Frontiers in Bioengineering & Biotechnology 1, Mar. 11, 2016
47	Emily Grubert, *At Scale, Renewable Natural Gas Systems Could be Climate Intensive: The Influence of Methane Feedstock and Leakage Rates*, 15 Env't Rsch. Letters, 4 Aug. 11, 2020
48	J. Liebetrau et al., *Analysis of Greenhouse Gas Emissions from 10 Biogas Plants within the Agricultural Sector*, 67 Water Sci. & Tech. 1370 (2013)
49	Charlotte Scheutz & Anders M. Fredenslund, *Total Methane Emission Rates and Losses From 23 Biogas Plants*, 97 Waste Mgmt. 38 (Sept. 2019)
50	Phoebe Gittelson et al., *The False Promises of Biogas: Why Biogas is an Environmental Justice Issue*, Env't Just., (2021)
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60	Kyoung S. Ro et al., *High-Rate Solid-Liquid Separation Coupled with Nitrogen and Phosphorous Treatment of Swine Manure: Effect on Ammonia Emission*, 2 Frontiers Sustainable Food Sys. (2018)
61	Deisi Cristina Tápparo, *Swine Manure Biogas Production Improvement Using Pre-Treatment Strategies: Lab-Scale Studies and Full-Scale Application*, 15 Bioresource Tech. Reps. 100716 (2021)
62	Marcelo Miele et al., *Effluent Treatment From Biogas Plants*, RushTranslate (2021)
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INDEX OF EXHIBITS

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ATTACHMENT 1

SOUTHERN ENVIRONMENTAL LAW CENTER

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Facsimile 919-929-9421

December 17, 2021

Ramesh Ravella
Department of Environmental Quality
1636 Mail Service Center
Raleigh, North Carolina 27699-1636
publiccommentsDWR@ncdenr.gov

Re: Comments on Upcoming Animal Waste Digester System General Permit

Dr. Ravella,

The Southern Environmental Law Center ("SELC") submits these preliminary comments on behalf of itself and Environmental Justice Community Action Network, Duplin County Branch of the North Carolina Conference of the NAACP, North Carolina Poor People's Campaign, Cape Fear River Watch, Rural Empowerment Association for Community Help, North Carolina Conservation Network, North Carolina Sierra Club, Sound Rivers, Winyah Rivers Alliance, Coastal Carolina Riverwatch, Haw River Assembly, Catawba Riverkeeper, Yadkin Riverkeeper, MountainTrue, Clean Water for North Carolina, Toxic Free NC, CleanAIRE NC, Center for Biological Diversity, Waterkeeper Alliance, and Appalachian Voices regarding the Department of Environmental Quality's ("DEQ") development of an animal waste digester system general permit ("biogas general permit") pursuant to the 2021 North Carolina Farm Act.[1] Thank you for the opportunity to provide comments on this important decision. It is critical that DEQ draft a permit that protects public health in communities living nearby hog operations covered under the biogas general permit, as well as the air, waterways, and drinking water these communities rely on.

In issuing the biogas general permit, DEQ is obligated under state law to require the waste treatment and disposal alternative with the least adverse impact on the environment and to evaluate and address the cumulative effects of its permitting decisions on waterways. N.C Gen. Stat. § 143-215.1(b)(2). In addition, to comply with federal civil rights law, DEQ must prevent the disproportionate impact of the biogas general permit on communities of color, which requires DEQ to do more than produce a basic report of the demographics of the communities that will be impacted by the biogas general permit and other pollution sources. 42 U.S.C. § 2000d (2018).

We urge DEQ to include the following provisions in the forthcoming biogas general permit:

- Adoption of cleaner technologies and practices that are compatible with biogas production and address the water and air pollution caused by the use of the digesters in conjunction with lagoons and sprayfields, including but not limited to the anticipated

[1] According to its statements at technical stakeholder meetings, DEQ will be developing a general permit for all animal operations that wish to install a farm digester system, including swine, dairy, and wet poultry. These comments are specific to swine waste to energy projects.

increase in ammonia emissions from the hog operations that will result from the use of covered digesters;

- Robust monitoring and reporting requirements, including monitoring of groundwater at the hog operations, surface waters near the hog operations, ammonia emissions, and monitoring of the waste going into and coming out of the digester or covered lagoon, with regular submission of all monitoring data to DEQ; and
- A requirement to regularly update nutrient management plans to account for the changes in land-applied digester waste, including increases in nitrate levels that are not detected by the current sampling requirements.

These recommendations are consistent with and build upon the recommendations made by the DEQ Secretary's Environmental Justice and Equity Advisory Board dated October 22, 2021 and attached as Exhibit 1. This Board also provided detailed recommendations regarding the public process that DEQ should follow when developing and seeking input on the draft biogas general permit, including, for example, opportunities to provide comments at an in-person hearing given the challenges of virtual platforms and limited broadband access for many impacted community members. The undersigned support those recommendations and urge DEQ to follow those recommendations moving forward.

I. Factual Background

The 2021 Farm Act tasks DEQ with developing a general permit for hog operations that will capture methane and other gases from hog waste lagoons. As discussed in more detail below, hog operations equipped with digesters, in addition to open lagoons and sprayfields, threaten the environment even more than existing industrial hog operations; among other concerns, these operations are expected to emit even more ammonia, which pollute the air, waterways, and soils and lead to sickness for people living nearby. The biogas general permit must account for these changes and require additional protections for nearby communities and the environment.

Over 2,000 industrial hog operations are covered under the 2019 swine general permit, a so-called "non-discharge" permit whose conditions are intended to prevent pollution to waterways. But as DEQ's and other peer-reviewed research has demonstrated, the 2019 swine general permit and its predecessor permits have not adequately protected waterways or groundwater from pollution from these operations. For decades, industrial hog operations have burdened neighbors with water and air pollution and odors, which have resulted in sickness and even death for many neighbors. Earlier this year, the Proceedings of the National Academy of Sciences found that the emissions from industrial animal operations in Sampson and Duplin Counties are responsible for a combined total of 178 premature deaths annually.[2] A disproportionate share of these neighbors are communities of color, making this a significant environmental justice issue.

[2] Nina G.G. Domingo et al., *Air quality-related health damages of food*, 118 PROCEEDINGS OF THE NAT'L ACAD. SCIS. 1 (May 18, 2021).

This is unacceptable, and DEQ must do more to protect communities and the environment from pollution.

A. New Digesters Will Rely on Lagoons and Sprayfields and Increase Ammonia Emissions and Risk of Pollution

The biogas general permit will allow industrial hog operations to install digesters—essentially covered lagoons—that increase the processes of anaerobic digestion and siphon off methane and other gases from hog waste. In most circumstances, these gases will be sent offsite through a network of new pipelines, processed, and ultimately used as fuel. These gases may also be used to generate electricity on-site. The only major biogas project proposed to date, the Align RNG (joint venture of Smithfield Foods and Dominion Energy) Grady Road Project, includes no additional treatment or other measures to control pollution from biogas digester waste; instead, as proposed, the waste from the digester will be transferred to uncovered secondary lagoons and eventually sprayed on nearby fields. This system fails to prevent pollution.

DEQ as well as numerous researchers have documented the pollution to rivers, streams, and groundwater caused by the use of the lagoon and sprayfield system. Hog waste stored in open lagoons emits ammonia into the environment. Pollution runs off from sprayfields; seeps into groundwater from lagoons and sprayfields; and reaches waterways through flooding and airborne ammonia deposits directly on waterways.[3] This pollution results in fish kills and algal blooms and other adverse impacts to water quality and contaminates drinking water.

Ammonia is a well-documented source of water pollution, air pollution, and health problems for people living nearby.[4] The process of siphoning off methane and other gases from untreated hog waste increases the proportion of inorganic nitrogen (e.g., ammonia/ammonium and nitrate) to organic nitrogen[5] in the waste leftover in the digester, which is transferred to open air lagoons and eventually sprayed on fields. The digester waste stored in uncovered secondary lagoons will release even more ammonia into the environment than a conventional hog waste lagoon. Spraying the digester waste from these open lagoons onto fields also releases ammonia and other pollutants into the environment.[6] Airborne ammonia can travel long distances from the source and deposit directly into waterways, leading to algal blooms and fish kills. Ammonia can

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[3] *See, e.g.,* Michael A. Mallin, *The Risk of Increased Cumulative Effects and Water Pollution from Swine Operations Producing Biogas in North Carolina* (Sept. 2021) (citing numerous peer-reviewed papers and data sets, including DEQ data and research, and describing various pathways for pollution from lagoons and sprayfields and the risk of increased pollution from hog operations using digesters without additional pollution control technologies) (attached as Exhibit 2); *see also, e.g.,* Steve Harden, et al., *Surface-Water Quality in Agricultural Watersheds of the North Carolina Coastal Plain Associated with Concentrated Animal Feeding Operations*, U.S. Dep't of Interior-U.S. Geological Survey (2015) (documenting water quality impacts from industrial hog operations).
[4] *See* Compilation of research by Dr. Viney P. Aneja, at 4-5 (Sept. 2021) (attached as Exhibit 3).
[5] *See id.*
[6] *See id.*; *see also* Summary of Expert Opinions of Dr. Shane W. Rogers (Sept. 2021) (attached as Exhibit 4).

also deposit on land, where it converts to nitrate, seeps into the soil, and contaminates drinking water resources for rural well users.[7]

Ammonia emissions are also a precursor to particulate matter pollution, which can penetrate deep into lungs and case serious cardiovascular and respiratory problems, and can even lead to death as documented by the Proceedings of the National Academy of Sciences' recent report.[8]

In addition, the use of digesters increases risk of groundwater and surface water quality problems by increasing the potential of nitrogen and phosphorus, among other pollutants, to move with water when digester waste is sprayed on fields.[9]

These environmental harms lead to adverse health outcomes for neighbors of hog operations, a disproportionate share of whom are Black, Latinx, and Native American. Peer-reviewed research has documented higher rates of asthma and other respiratory conditions, higher death rates from common diseases, and higher rates of depression and anxiety among people living nearby these hog operations.[10] Without stringent pollution controls, the biogas general permit could perpetuate the health problems caused by the primitive lagoon and sprayfield system and increase risk of sickness and death for nearby communities.

B. Cleaner Technology That is Compatible with Biogas Production is Available and Economical

Animal waste management technology has advanced dramatically in the last two decades. As described in more detail in Exhibit 5, there are numerous technologies and practices that are in use in North Carolina and beyond that are both compatible with biogas production *and* address ammonia emissions and other environmental and public health harms caused by the primitive lagoon and sprayfield system. These include nitrification-denitrification, ANNAMOX, alkaline precipitation, Struvite precipitation, Super Soils/Terra Blue, Sistrates, and membrane separation and solids recovery technologies, among others.

II. Legal Background

A. The Farm Act Requires DEQ to Create a General Permit for Farm Digesters

The Farm Act of 2021, Session Law 2021-78, requires DEQ to develop a permit to allow numerous animal operations to "construct and operate a farm digester system," *id.* at Sec. 11(b), which is defined as "a system, including all equipment and lagoon covers, by which gases are

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[7] *See* Aneja, *supra* note 4; *see also* Mallin, *supra* note 3, at 4.
[8] *See* Domingo, *supra* note 2, at 2.
[9] *See* Mallin, *supra* note 3, at 4-5.
[10] *See id.; see also* Julia Kravchenko et al., *Mortality and Health Outcomes in North Carolina Communities Located in Close Proximity to Hog Concentrated Animal Feeding Operations*, 79 N.C. MED. J. 278, 278 (2018), https://doi.org/10.18043/ncm.79.5.278 (finding higher mortality rates for people living near industrial hog operations); *see also* Steve Wing, et al., *Environmental Injustice in North Carolina's Hog Industry*, 108 Env't Health Perspectives 225 (2000), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1637958/ (documenting that hog operations are concentrated in areas with higher populations of people of color and low-wealth communities).

collected and processed from an animal waste management system for the digestion of animal biomass for use as a renewable energy resource." *Id.* Sec. 11(a). The legislature instructed DEQ to include conditions in the biogas general permit which are "required to describe and authorize the construction, monitoring, and proper operation" of the systems, *id*. Sec. 11(d), in addition to the conditions in the 2019 swine general permit.[11] The "proper operation" of the farm digester system includes provisions that ensure the new system "does not cause pollution in the waters of the State," N.C. Gen. Stat. §143-215.10C(b), which includes groundwater and surface waters, *id.* § 143-212(6). The requirements of the Farm Act likewise authorize DEQ to require additional treatment and other practicable alternatives to minimize adverse impacts on the environment. In other words, DEQ has the authority to include any conditions in the biogas general permit that are necessary to ensure that the facility covered by the permit does not pollute groundwater or surface waters. DEQ should draft the biogas general permit to provide for any additional pollution control measures that may be needed to address the effects of covered anaerobic digesters and should also account for the site-specific circumstances of individual facilities.

B. North Carolina's Water Pollution Control Statute Requires DEQ to Protect the Environment and Neighbors from Pollution from Hog Operations

The North Carolina Constitution provides that it is the policy of North Carolina "to conserve and protect its lands and waters for the benefit of all its citizenry … [and that] it shall be a proper function of the State … to control and limit the pollution of our air and water." N.C. Const. art. XIC, sec. 5. Moreover, the legislature declared "the public policy of the State to provide for consideration of its water and air resources," N.C. Gen. Stat. § 143-211(a), and "to maintain, protect, and enhance water quality within North Carolina," *id.* at (b). In issuing permits, DEQ must "prevent . . . any significant increase in pollution of the waters of the State," *Id.* § 143-215.1(b)(1), and ensure compliance with water quality standards, *Id.* § 143-215.10F. DEQ may conduct "any inquiry or investigation it considers necessary before acting on an application." *Id*. § 143-215.10C(c).

North Carolina's pollution control statute mandates that for *all* permits, DEQ require the "practicable waste treatment and disposal alternative with the last adverse impact on the environment." N.C. Gen. Stat. 143-215.1(b)(2); *see also* 15A N.C. Admin. Code 2T.0105(f). This provision of state law applies to the biogas general permit. As such, DEQ must require that any permittees under the biogas general permit use the waste treatment technology with the least adverse impact on the environment. The broad language of this requirement includes the obligation and authority to minimize ammonia pollution.

In addition, when issuing the biogas general permit DEQ must prevent water pollution due to the "cumulative effects" of its permitting decisions. N.C. Gen. Stat. 143-215.1(b)(2). This means that DEQ has an obligation to evaluate water pollution anticipated from the biogas general permit, as well as from other permitted operations in the area. This obligation is not limited to consideration of the effects of new operations or solely operations engaged in the

[11] The 2019 swine general permit is issued pursuant to the N.C. Gen. Stat. §143-215, *et seq*., and Subchapter 2T of Title 15A of the North Carolina Administrative Code.

production of biogas; for instance, DEQ must consider the impacts attributable to the collective effects of other permitted animal feeding operations, including deemed permitted poultry operations. DEQ must prevent pollution of groundwater and surface waters resulting from the combined effects of these operations. DEQ's obligation to prevent these water quality impacts includes ammonia pollution, a documented source of water quality impacts, as reflected in the statute's broad definition of "water pollution" as "the man-made or man-induced alteration of the chemical, physical, biological, or radiological integrity of the waters of the State, including, but specifically not limited to, alterations resulting from the concentration or increase of natural pollutants caused by man-related activities." *Id*. § 143-213(19).

To ensure compliance with water quality standards, state regulations expressly provide for DEQ's authority to "require monitoring and reporting requirements, including of groundwater, surface water or wetlands, waste, wastewater, residuals, soil, treatment processes, lagoon or storage ponds, and plant tissue, if necessary to determine the source, quantity, and quality of the waste and its effect upon the surface water, ground waters, or wetlands." 15A N.C. Admin. Code 02T .0108(c). A recent decision from the Ninth Circuit Court of Appeals reinforces the need for monitoring and reporting requirements for permits such as the biogas general permit:

> Without a requirement to monitor runoff from irrigated CAFO [concentrated animal feeding operation] fields, there is no way to ensure that a CAFO is complying with the Permit's dry weather no-discharge requirement for land-application areas. . . . Without a requirement that CAFOs monitor waste containment structures for underground discharges, there is no way to ensure that production areas comply with the Permit's zero-discharge requirement.

Food & Water Watch v. U.S. Env't Prot. Agency, No. 20-71554, 2021 WL 4203496, at *10 (9th Cir. Sept. 16, 2021).

In sum, the state water pollution control statute and implementing regulations give DEQ broad authority, and indeed an obligation, to draft permits in a way that protects surface water quality and groundwater.

C. Title VI of the Civil Rights Act of 1964 Requires DEQ to Ensure the Biogas General Permit Protects All Communities

As a recipient of federal funding, DEQ must comply with Title VI of the Civil Rights Act of 1964, which provides that:

> No person in the United States shall, on the ground of race, color, or national origin, be excluded from participation in, denied the benefits of, or be subjected to discrimination under any program or activity receiving Federal financial assistance.

42 U.S.C. § 2000d (2018). EPA regulations prohibit a recipient of federal funds from using criteria or methods of administering a program or activity which have the effect of subjecting individuals to discrimination. 40 C.F.R. § 7.35(b). "Title VI imposes on states an affirmative

obligation to include consideration of Title VI criteria in permitting decisions."[12] Therefore, DEQ—as a recipient of federal funds—is required to administer its permitting programs in a manner that does not have the effect of subjecting individuals to discrimination or else the agency risks losing those funds.

To date, DEQ has failed to comply with Title VI. For the first several hog operations participating in the Grady Road Project, DEQ collected demographic information showing that a disproportionate share of the people living nearby those hog operations are people of color and low-wealth households. Reporting who will be affected by permitting decisions is not enough; DEQ must also document the nature of the effects of the permitting decisions, the cumulative effects of other DEQ-permitted operations in the community, and the potential impact of pollution or adverse public health outcomes on vulnerable communities. For the previous biogas permit decisions, DEQ took no steps to evaluate or prevent the serious adverse impacts, such as increased health problems and death rates, associated with hog operation pollution, which are only going to increase with the addition of biogas production at these operations. Title VI requires more from DEQ.

III. Recommendations

To meet its obligations under state and federal laws and to address the anticipated increase in ammonia emissions and the resulting pollution to surface waters, groundwater, and human health, DEQ must include conditions in the biogas general permit that go above and beyond the existing 2019 swine general permit. But before DEQ issues the biogas general permit, the agency must conduct several analyses to comply with state and federal permitting requirements.

A. Procedural recommendations

Under the state water permitting statute, DEQ must conduct a thorough analysis of the combined effect of its decision to permit hog operations that will increase overall ammonia emissions *plus* the effect of other permitted operations on rivers and streams and groundwater. It is not enough for DEQ to acknowledge "atmospheric losses from lagoons" in a line in a public meeting report and move on.[13] DEQ cannot ignore this substantial source of pollution or its effects on water quality and public health. And data on cumulative effects are not hard to come by: DEQ itself has studied water quality in areas of high concentrations of animal feeding operations and found these waters have elevated levels of pollutants. DEQ must consider this information in its permitting.

In addition, DEQ should research and develop a list of waste treatment and disposal alternatives that are compatible with biogas production and reduce the risk of air and water pollution; permittees should have the option to choose the alternative that is most compatible with its operation. To aid DEQ in this evaluation, Exhibit 5 attached to these comments provides a summary of several waste treatment and disposal alternatives that are compatible with biogas

[12] *S. Camden Citizens in Action v. New Jersey Dept. of Envtl. Protection*, 145 F. Supp. 2d 446, 476 (D. N.J. 2001).
[13] *Swine Biogas Permit Modifications: Public Meeting Report and Recommendations*, N.C. Dep't of Env'tl Quality, 14, 18 (Mar. 2021), https://deq.nc.gov/media/21373/download.

production and result in less air and water pollution than the lagoon and sprayfield system equipped with a digester.

B. Substantive recommendations

To comply with federal civil rights laws, DEQ must also analyze the effect of the biogas general permit and other facilities and prevent harmful air and water pollution that have disproportionate impacts on the basis of race, color, or national origin. The hog operations that would be eligible for coverage under the biogas general permit are disproportionately located in Black, Latinx, and Native American communities which are already overburdened by pollution from countless polluting facilities in eastern North Carolina. These analyses must be more than paper exercises; results of these analyses must inform DEQ's permitting decisions, including its decision about whether a particular facility can be covered under the biogas general permit and what pollution control measures are needed.

In addition, the biogas general permit should acknowledge the effect of ammonia emissions on water quality and include specific provisions to control ammonia emissions from hog operations covered under the biogas general permit.

Because the evaluation of cumulative impacts is best conducted in the context of a specific geographic area, DEQ should put permit applicants on notice that the results of the agency's evaluations of cumulative effects on water quality and/or cumulative impacts on local residents may require that the agency issue an individual permit with site-specific conditions.

In addition, DEQ must include robust, frequent monitoring and reporting requirements to ensure compliance with surface water and groundwater standards and permit conditions. At a minimum, these requirements should include:

- Monthly groundwater monitoring with monitoring wells installed upgradient and downgradient of the digester and secondary lagoon(s);
- Monthly sampling and reporting of the influent and effluent of the digester for the following water quality parameters: Total Nitrogen, Total Kjeldahl Nitrogen, ammonium nitrogen, copper, sulfur, nitrate nitrogen, Total Phosphorus, zinc, and fecal coliform bacteria;
- Monthly sampling of the waste that is to be land-applied for the following parameters: Total Nitrogen, Total Kjeldahl Nitrogen, ammonium nitrogen, Total Phosphorus, zinc, copper, and fecal coliform bacteria;
- Monthly sampling and analysis of surface water, specifically tributaries previously impacted by operations and/or in the flow path of each site/lagoon for the following parameters: Total Nitrogen, Total Phosphorus, Total Suspended Solids, BOD, fecal coliform, *E.coli*, Total Kjeldahl Nitrogen, ammonia, and nitrates/nitrite;
- Quarterly soil sampling from land application fields at 2 and 10 inches and analyzed for phosphorus, mineral/heavy metals, Total Organic Carbon, total carbon, Total Nitrogen, Total Kjeldahl Nitrogen, pH, and nitrate; and
- Monthly air quality monitoring for ammonia emissions.

Permittees should report all results of sampling to DEQ on an annual basis.

Finally, to address the possibility of increased nitrate levels in the land-applied waste, DEQ should require *at least* annual updates to nutrient utilization plans and animal waste management plans for all operations that are based on the monthly digester waste sampling described above.

IV. Conclusion

SELC and the undersigned urge DEQ to adopt the recommendations herein and in the Environmental Justice & Equity Advisory Board's October 22, 2021 letter to DEQ Secretary Elizabeth Biser. We look forward to participating in the public comment process in early 2022, and will supplement these comments at that time.

Thank you for your consideration of these comments.

Sincerely,

Blakely E. Hildebrand

Blakely E. Hildebrand
Staff Attorney
Southern Environmental Law Center

Submitted on behalf of:
Environmental Justice Community Action Network
Cape Fear River Watch
Duplin County Branch of the North Carolina Conference of the NAACP
North Carolina Poor People's Campaign
Rural Empowerment Association for Community Help
North Carolina Conservation Network
North Carolina Sierra Club
Sound Rivers
Winyah Rivers Alliance
Coastal Carolina Riverwatch
Haw River Assembly
Catawba Riverkeeper
Yadkin Riverkeeper
MountainTrue
Clean Water for North Carolina
Toxic Free NC
CleanAIRE NC
Center for Biological Diversity
Waterkeeper Alliance
Appalachian Voices

Exhibit 1

SENT VIA EMAIL

October 22, 2021

Elizabeth S. Biser, Secretary
N.C. Department of Environmental Quality
217 W. Jones Street
Raleigh, NC 27603

Dear Secretary Biser:

The Environmental Justice and Equity (EJE) Advisory Board was chartered to assist the Department of Environmental Quality (DEQ) in ensuring fair and equal treatment and meaningful involvement of all North Carolinians, regardless of race, color, national origin, or income, in the development, implementation, and enforcement of environmental laws and policies. In this role, we strive to ensure access to clean air, clean water, and clean soil, and the opportunity to live in safe and healthy communities for all North Carolina families.

Today, we write to you about pollution from hog operations, a long-standing environmental justice issue that has affected thousands of North Carolina families for decades. We respectfully request that DEQ take steps to protect these families, their health, and the environment.

Under the 2021 North Carolina Farm Act, N.C. Sess. L. 2021-78, DEQ must develop a general permit for hog operations that will produce swine waste-to-energy (biogas) by July 2022. As we expressed in our August 26, 2021 letter to you, the EJE Advisory Board has significant concerns about the pollution and public health implications of this general permitting scheme.

In addition to the procedural recommendations we provided in our August 26, 2021 letter, we advise DEQ to ensure the new general permit include robust substantive protections against hog waste pollution and its disparate impacts on surrounding communities. Cleaner technologies and practices that reduce water and air pollution—and that are compatible with biogas production—are available and practicable. In fact, some of these technologies are used by Smithfield Foods, the nation's largest pork producer, in other states. North Carolinians deserve the same protections.

In North Carolina, biogas is produced by capturing methane from hog waste lagoons using covered anaerobic digesters. To date, DEQ has allowed hog operations to dispose of waste from these digesters by transferring the digester waste to open "secondary" lagoons, and spraying the digester waste on fields.[1] The biogas is sent off-site for processing and eventually used to produce energy. Biogas produced using the lagoon and sprayfield system is not a clean source of energy.

[1] *See, e.g.*, Permit No. AWI310039 Benson Farm (Mar. 31, 2021) (authorizing the use of a Waste-to-Energy system, which includes a covered anaerobic digester; a clay-lined lagoon; pumps, pipes, and other equipment to transfer waste; and sprayfields).

The lagoon and sprayfield waste management system used at industrial hog operations pollutes waterways,[2] contaminates drinking water,[3] and dirties the air people breathe.[4] This pollution and the resulting harms to human health have burdened neighbors—mainly people of color and low wealth communities--for decades.[5] As such, this is one of the most significant and well-studied environmental injustices in North Carolina; public health and environmental experts agree on the harm that this system causes for people and the environment.

Producing biogas from hog waste using anaerobic digesters, open secondary lagoons, and sprayfields does *not* address many of the longstanding, serious pollution problems of using open lagoons and sprayfields to store and dispose of hog waste. The use of digesters is likely to *increase* ammonia emissions when the digester waste is stored in open secondary lagoons and sprayed on fields.[6] Airborne ammonia from hog operations deposits in surrounding waterways, causing

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[2] Michael A. Mallin et al., *Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution*, 226 WATER, AIR, SOIL & POLLUTION 407 (2015), *available at* https://link.springer.com/article/10.1007/s11270-015-2669-y; Christopher D. Heaney et al., *Source Tracking Swine Fecal Waste in Surface Water Proximal to Swine Concentrated Animal Feeding Operations*, 511 SCI. TOTAL ENV'T 676 (2015), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4514616/; JoAnn M. Burkholder et al., *Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality*, 115 ENVT. HEALTH PERSP. 308 (2007), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1817674/.

[3] Wendee Nicole, *CAFOs and Environmental Justice: The Case of North Carolina*, 121 ENVT. HEALTH PERSP A182, A186 (2013), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC3672924/ ("Even without spills, ammonia and nitrates may seep into groundwater, especially in the coastal plain where the water table is near the surface."); M.E. Anderson & M.D. Sobsey, *Detection and Occurrence of Antimicrobially Resistant E. coli in Groundwater on or near Swine Farms in Eastern North Carolina*, 54 WATER SCI. & TECH. 211, 217 (2006), *available at* https://pubmed.ncbi.nlm.nih.gov/17037155/ ("Overall, the results of this study demonstrated that antibiotic-resistant E. coli were present in groundwaters associated with commercial swine farms that have anaerobic lagoons and land application systems for swine waste management."); Kenneth Rudo, *Groundwater Contamination of Private Drinking Well Water by Nitrates Adjacent to Intensive Livestock Operations (ILOs)*, N.C. DEP'T OF HEALTH AND HUMAN SERV., 414, 418 (June 1999).

[4] Nina G.G. Domingo et al., *Air quality-related health damages of food*, 118 PROCEEDINGS OF THE NAT'L ACAD. SCIS. 1 (May 2021), *available at* https://www.pnas.org/content/118/20/e2013637118; Leah Schinasi et al., *Air Pollution, Lung Function, and Physical Symptoms in Communities Near Concentrated Swine Feeding Operations*, 22 EPIDEMIOLOGY 208, 208 (2011), *available at* https://pubmed.ncbi.nlm.nih.gov/21228696/; Sacoby M. Wilson & Marc L. Serre, *Examination of Atmospheric Ammonia Levels Near Hog CAFOs, Homes, and Schools in Eastern North Carolina*, 41 ATMOSPHERIC ENV'T 4977, 4985 (2007), *available at* https://www.researchgate.net/publication/223777299_Examination_of_atmospheric_homes_and_schools_ammonia_levels_near_hog_CAMS_in_Eastern_North_Carolina

[5] Steve Wing & Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians* 2 (2014), *available at* https://www.ncpolicywatch.com/wp-content/uploads/2014/09/UNC-Report.pdf (finding that industrial hog operations are disproportionately located near communities of color and low-wealth communities in eastern North Carolina); Dana Cole et al., *Concentrated Swine Feeding Operations and Public Health: A Review of Occupational and Community Health Effects*, 108 ENVTL. HEALTH PERSPECTIVES 685 (2000), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1638284/; Kendall M. Thu, *Public Health Concerns for Neighbors of Large-Scale Swine Production*, 8 J. AGRIC. SAFETY & HEALTH 175, 176 (2002), *available at* https://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.410.1811&rep=rep1&type=pdf; Steve Wing & Susanne Wolf, *Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents*, 108 ENVTL. HEALTH PERSP. 233 (2000), *available at* https://www.jstor.org/stable/3454439.

[6] Baines, R. (Edited), *Reducing greenhouse gas emissions from livestock production*, Taylor & Francis Group, London, 145 (2021), *available at* https://www.taylorfrancis.com/books/edit/10.1201/9781003048213/reducing-greenhouse-gas-emissions-livestock-production-richard-baines (finding that the potential for ammonia emissions when storing digested hog waste increases); Viney Aneja, et. al, *Characterizing Ammonia Emissions from Swine Farms in North Carolina: Part 2—Potential Environmentally Superior Technologies for Waste Treatment*, 58 J. AIR

pollution that can lead to algae blooms and fish kills.[7] Airborne ammonia also deposits on the ground, where it can seep into the soil and cause nitrate pollution in drinking well water, which can harm infants and pregnant women.[8] Airborne ammonia also forms fine particulate pollution that causes serious health problems and premature deaths in surrounding communities.

In fact, a recent study published by the National Academy of Sciences attributes an astounding 95 premature deaths in Sampson County and 83 premature deaths in Duplin County to the emissions from hog operations every year.[9] This is already an unacceptable situation that must be stopped. And the prospect of *increasing* the rates of sickness or death, resulting from sending more ammonia and fine particulate pollution into the surrounding environment, is simply unacceptable. DEQ must not allow hog waste pollution to continue harming more people in our most vulnerable communities.

Communities in eastern North Carolina have been complaining about pollution from industrial hog operations for decades. Since DEQ began considering permits for the first large-scale biogas project almost two years ago, hundreds of people across eastern North Carolina and beyond participated in public hearings, submitted comments, and appealed to DEQ to protect their communities and the environment from pollution from lagoons and sprayfields. To date, DEQ has failed to heed these calls.

In developing the conditions of the biogas general permit, DEQ must address this environmental injustice and protect families and the environment in eastern North Carolina. To start, DEQ's environmental justice analysis must be more than a formality intended to inform agency outreach. Instead, DEQ must conduct a comprehensive environmental justice analysis that translates into substantive permit conditions to minimize disparate impacts from cumulative impacts of the general permit and other DEQ-permitted operations on surrounding communities, including

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& WASTE MGMT. ASS., 1145, 1156 tbl. 4 (2008), *available at* https://www.tandfonline.com/doi/pdf/10.3155/1047-3289.58.9.1145 (finding more than an 11 percent increase in ammonia emissions from an open secondary lagoon storing digester waste as compared to an open lagoon storing hog waste that has not been in a digester); Kupper et al., *Ammonia and greenhouse gas emissions from slurry storage—A Review*, 300 AGRICULTURE, ECOSYSTEMS, & ENV'T 1, 9 (2020) *available at* https://www.sciencedirect.com/science/article/pii/S0167880920301481; Lowry A. Harper et al, *The Effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions*, 39 J. ENVT. QUAL. 62 (2010), *available at* https://pubmed.ncbi.nlm.nih.gov/21284295/ (noting that because of the reduction of methanogenesis and its reduced effect on the chemical conversion of ammonium to dinitrogen gas, ammonia emissions from operations generating biogas increased by 46 percent compared to operations that did not produce biogas).

[7] Jennifer K. Costanza et al., *Potential geographic distribution of atmospheric nitrogen deposition from intensive livestock production in North Carolina, USA*, 398 SCI. OF TOTAL ENV'T 76, 77 (2008) http://jencostanza.com/docs/Costanza_et_al_2008_STOTEN.pdf; John T. Walker et al, *Atmospheric transport and wet deposition of ammonia in North Carolina*, 34 ATMOSPHERIC ENV'T, 3407, 3416 (2000), *available at* http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.557.3074&rep=rep1&type=pdf (detecting deposition of ammonia and ammonium upwards of 80 km from the source of that pollution).

[8] Mary Berg et al, *Nitrogen Behavior in the Environment*, N.D. AGR. EXTENSION SERV. 3 (2017), https://www.ag.ndsu.edu/publications/environment-natural-resources/nitrogen-behavior-in-the-environment; Dennis Keeney & Robert Olsen, *Sources of nitrate in groundwater*, 16 CRITICAL REVIEWS IN ENV'T SCI. & TECH. 257 (1986), https://www.tandfonline.com/doi/abs/10.1080/10643388609381748; Mary Ward, et al, *Drinking Water Nitrate and Human Health: An Updated Review*, 15 INT'L J. ENV'T RESEARCH & PUBLIC HEALTH 1 (July 23, 2018), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC6068531/.

[9] Nina G.G. Domingo et al., *Air quality-related health damages of food*, 118 PROCEEDINGS OF THE NAT'L ACAD. SCIS. 1 (May 2021), https://www.pnas.org/content/118/20/e2013637118.

communities of color and low-wealth communities that are already overburdened by pollution from multiple industries.[10] To be clear, it is not enough for DEQ to evaluate the cumulative effects of permitting decisions on water quality, as required under state environmental law; the agency, as a recipient of federal funding, also has obligations under Title VI of the Civil Rights Act of 1964, which require the agency to address harm to vulnerable North Carolinians.

In addition, as part of the general permit, we strongly advise DEQ to require the following:

- Cleaner technology and practices that are compatible with biogas production *and* address water and air pollution caused by the lagoon and sprayfield system, particularly given the increased ammonia pollution associated with open storage of biogas digester waste;

- Robust groundwater and surface water monitoring at every hog operation to identify pollution to rivers, streams, and groundwater, which is a source of drinking water for many rural residents;

- Updated nutrient management plans that account for the changes in the land-applied waste after digestion; and

- More protective freeboard requirements, such as automated lagoon/storage pond waste-level monitors and recorders, to reduce the likelihood that flooding or inundation of lagoons due to increasing frequent and severe storms will result in the discharge of the more harmful digester waste.

Thank you for your attention to this important matter.

Respectfully submitted by the EJE Advisory Board

James H. Johnson, Jr., Chair
Marian Johnson-Thompson, Vice Chair

[10] DEQ must scrutinize the environmental impact of the poultry industry as part of this analysis, as poultry operations with 30,000 or more birds are deemed permitted under state law, are often co-located in communities hosting swine operations, and have proliferated most rapidly in recent years in communities of color.

Exhibit 2

The Risk of Increased Cumulative Effects and Water Pollution from Swine Operations Producing Biogas in North Carolina

Summary compiled by Dr. Michael A. Mallin, Research Professor, Center for Marine Sciences, University of North Carolina Wilmington on behalf of Environmental Justice Community Action Network & Cape Fear River Watch

Environmental Justice Community Action Network & Cape Fear River Watch vs. N.C. Department of Environmental Quality – Division of Water Resources & Murphy-Brown, LLC 21 EHR 02068, 02069, 02070, 02071

I. Executive Summary

On March 31, 2021, N.C. Department of Environmental Quality ("DEQ") issued permits to four industrial hog operations in the lower Cape Fear River basin: AWI301135 (Waters Farm – M&M Rivenbark), AWI310039 (Benson Farm), AWI820466 (Farm 2037 and 2038/Goodson Farm), and AWS820005 (Kilpatrick Farm 1, 2, 4 & 5 & Merritt Farm) (together the "Hog Operations"). The new permits allow three of the Hog Operations to excavate and install new lagoons to act as digesters, and allow the fourth operation (Kilpatrick Farm) to cover an existing lagoon to produce swine waste-to-energy biogas. The permits also allow these operations to add new piping and pumps to transfer waste between digesters/covered lagoons, secondary (open-air) lagoons, and land application fields.

This summary addresses the cumulative effects of pollution from hog operations in the Cape Fear River basin. This report surveys the types of pollution and their effects and the means by which hog operations contribute to these cumulative effects on surface water and groundwater quality in the Cape Fear River basin.

This summary also applies this research to draw conclusions about how the waste management systems authorized by the four permits issued by DEQ in this case fail to prevent adverse impacts to water quality and may increase the cumulative impacts of pollution in the Cape Fear River basin. This opinion is based on the plans submitted for these facilities, the location of these facilities, the site conditions at each facility, and a long history of published research on the impacts of the swine waste disposal methods similar to those permitted for the four operations.

To summarize, the new permits for the Hog Operations in the Cape Fear River basin will use a lagoon and sprayfield waste management system to manage and dispose of hog waste by installing new equipment to transfer hog waste to anaerobic digesters/covered lagoons and then to earthen, open air lagoons and spraying this waste onto nearby sprayfields. Abundant published research indicates that nutrients and fecal bacteria in swine waste cannot be contained on-site when this waste management system is used, and this waste management system is likely to pollute offsite waters of the state, creating adverse impacts to water quality and increasing human health risk. Furthermore, research indicates that covered lagoons have more concentrated ammonia, phosphate and fecal coliform bacteria than open lagoons, which increases offsite pollution threats. Anaerobic digesters concentrate ammonium to very high levels, which will then be piped to an open lagoon where volatilization to the atmosphere will occur; the waste will be sprayed out, and further volatilization of ammonia will occur. Atmospheric ammonia from swine sprayfield operations travels well offsite and is later deposited onto public waters and private property, degrading water quality and entering other individuals' airspace. As these new covered digester lagoons will have very high ammonia concentrations, the permitted operations are likely to further pollute waters of the state.

II. Background

The vast majority of swine and poultry in North Carolina are raised in large, industrial-style mass production facilities known as concentrated animal feeding operations (CAFOs, also called AFOs, or ILOs, intensive livestock operations). The U.S. Environmental Protection Agency ("EPA") (2014) defines large CAFOs as containing $\geq$ 1,000 head of beef cattle, 2,500 swine > 25 kg, 10,000 swine < 25 kg, 125,000 chickens, 82,000 laying hens or 55,000 turkeys. While inventory and sales vary year-to-year, the 2017 Census of Agriculture showed North Carolina to be the second-largest producer of swine in the USA, only behind Iowa (USDA 2021). The Cape Fear River basin drains the largest concentration of swine CAFOs in North Carolina.

Swine CAFOs consist of hundreds to many thousands of head of swine living in close quarters (see photos), which generate vast volumes of animal manure. The floors of hog houses have slats, through which manure and urine falls or is rinsed. Swine waste is then drained, flushed, or pumped into outdoor pits called lagoons where some anaerobic digestion occurs; it can also be pumped into covered lagoons, or covered lagoon/anaerobic digesters where ammonia concentration occurs. At a conventional hog operation, the liquid waste is periodically pumped out of the uncovered lagoons and sprayed onto adjoining sprayfields (see photos), many with a cover crop of Bermudagrass. Under the new permits for the Hog Operations, the liquid waste will be transferred from the covered lagoon/digester to a secondary lagoon, and ultimately land applied on nearby fields. In North Carolina spraying of animal waste onto Bermudagrass fields is permitted between March 1 and September 30.

Explanation of photos: (left) Eastern North Carolina swine CAFOs showing hog houses and waste lagoons, (middle) Swine within hog house with slats in floor for manure removal, (right) Spraying swine lagoon waste out on a saturated sprayfield (indicating a high water table), with the waste ponding at the surface (photos by Dr. M.A. Mallin).

This section describes sources of cumulative effects on water quality from CAFOs in the Cape Fear River basin. Section IV below discusses how the waste management systems at the Hog Operations are likely to contribute to these issues.

a. Waste lagoon pollutants and their effects on aquatic systems

The type of swine waste management systems used at the Hog Operations have a variety of effects on water quality. There are several key pollutants in swine waste that are threats to water quality and human health. Foremost among them are excess nitrogen (primarily in the forms of ammonia and nitrate), phosphorus, fecal bacteria and viruses, and biochemical oxygen demand. These pollutants reach rivers and streams in a variety of ways, including via airborne deposition from swine waste spraying (i.e., ammonia, fecal bacteria, and other pollutants), surface water runoff from sprayfields during precipitation events (i.e., stormwater runoff of nitrate, phosphorus and fecal microbes), and groundwater contamination and subsequent sub-surface movement into streams (i.e., nitrate and fecal bacteria). Accidents of all kinds periodically occur that release swine waste into nearby streams; note that such releases have occurred at the Waters Farm in recent years.

Importantly, the distance of some CAFOs to public waters can be uncomfortably short, which increases the risk that pollution from these operations reaches surface waters, particularly during rain events. Martin et al. (2018) used a GIS mapping approach to determine that 19% of North Carolina CAFOs (primarily swine) were located within 100 meters of a stream, with some less than 15 meters from the stream's edge. FEMA flood maps provided by Murphy-Brown indicate that some sprayfields at Farm 2037 & 2038 (Goodson Farm) are located in the floodplain and a sprayfield at the Kilpatrick Farm appears to be close to stream waters.

The Hog Operations are located nearby several streams in the lower Cape Fear River Basin. The Kilpatrick Farm is located along a tributary to Stewart's Creek, which drains into Six Runs Creek, a major tributary of the Black River; the proposed digester is approximately 2,000 ft. from Stewart's Creek proper. The Waters Farm property appears to be drained by two different watersheds, with the proposed digester site located about equidistant between Stewart's Creek in the Six Runs Creek drainage, and Murphey's Creek in the Rockfish Creek drainage. Farm 2037 & 2038 (Goodson Farm) is located near the juncture of Turkey Creek and Six Runs Creek, with the proposed digester site approximately 1,000 ft. from Turkey Creek and roughly 3,000 ft. from Six Runs Creek. The Benson Farm is located near Goshen Swamp.

i. *Bacteria runs off into nearby rivers and streams*

Fecal coliform bacteria, which are an indicator bacteria for the presence of pathogenic bacteria, have been found (see Mallin et al. 2015; Mallin and McIver 2018; NCDEQ 2020) to be excessively high in streams that pass near sprayfields (at times many thousands of colony-forming units per 100 milliliters of water (CFU/100 mL)), well exceeding the North Carolina standard for Class C waters (geometric mean of no more than 200 CFU/100 mL from five sample collections within 30 days). Normal rain events occurring shortly after animal waste is land-applied to fields cause vertical and horizontal movement of microbes to nearby water bodies (Crane et al., 1983; Mawdsley et al., 1995).

ii. Nitrate and ammonia in excess are highly polluting

Nitrogen (N) in water exists in both organic and inorganic forms. Under anaerobic conditions, such as within waste lagoons, there is no dissolved oxygen, and ammonia is the major inorganic N form; data from five swine lagoons yielded an average ammonia concentration of 550 mg/L (Westerman et al. 1990); ammonia concentrations in swine waste lagoons in Utah, including biofuel digesters are even higher (Harper et al. 2010). To put this into perspective, average ammonia in raw human sewage is far lower, about 15 mg/L (Clark et al. 1977). When swine waste is sprayed onto fields where there is abundant oxygen, natural bacteria convert (oxidize) much of this ammonia to nitrate. Nitrate is highly mobile in groundwater as it is negatively charged, as is clay, and thus does not adhere to soil particles. As a result large quantities of nitrate readily move into groundwater below sprayfields and from there laterally to surface waters off-site and downstream. This movement off-site is enhanced when soils are porous and the groundwater table shallow, such as in much of the North Carolina Coastal Plain. Nitrate is also readily moved overland to the nearest stream by stormwater runoff.

Elevated nitrate concentrations have been reported to cause toxic effects on aquatic animals including fish. Mortality and developmental changes in fish and amphibians have been documented from exposure to concentrations of nitrate less than the 10 mg/L drinking water standard established by the U.S. Environmental Protection Agency (Guillette and Edwards 2005); such effects are very species-specific and largely impact early developmental stages rather than adults.

In addition, in Coastal Plain blackwater streams, inputs of inorganic nitrogen (nitrate or ammonium) stimulate the growth of algae (Mallin and Cahoon 2020), in some cases to excessive levels (algal blooms). Note that even small concentrations of nitrate or ammonium can stimulate this growth -- 0.50 mg/L or less of nitrate or ammonium or about one hundred times *less* than concentrations of inorganic nitrogen in swine waste lagoons. The levels of ammonium or nitrate concentrations sampled in streams near sprayfields or downstream from sprayfields far exceed 0.50 mg/L (Harden 2015; Mallin et al. 2015; Mallin and McIver 2018; NCDEQ 2020; Sousan et al. 2021).

Either consistent inputs of elevated levels of ammonium or nitrate such as above, or acute accidents releasing large quantities of nutrients from animal waste can cause algal blooms (Burkholder et al. 1997; Mallin et al. 1997; Mallin et al. 2015). Once algal blooms die, the organic remains are fed on by natural bacteria, which multiply greatly and use up dissolved oxygen in a process called respiration. This creates what is called a biochemical oxygen demand (BOD), which reduces dissolved oxygen in the water column. The State of North Carolina has dissolved oxygen requirements for most waters of 5.0 mg/L, and 4.0 mg/L for swamp waters. When dissolved oxygen levels are reduced by high BOD to low levels (< 2.0 mg/L) fish kills may occur; low dissolved oxygen is commonly cited as a reason for fish kills by North Carolina Division of Water Quality (NCDEQ website). DEQ also notes that lack of oxygen created by decomposing algal blooms sometimes results in fish kills and other aquatic life impacts.

iii. Phosphorus

Swine waste also contains high concentrations of phosphorus. A survey of five swine lagoons yielded an average total phosphorus (P) concentration of 100 mg/L (Westerman et al. 1990). Average total phosphorus in raw human sewage is about 10 mg/L, for comparison (Clark et al. 1977). Excess phosphorus (concentrations > 0.50 mg/L) in Coastal Plain stream waters are known to directly stimulate the growth of aquatic bacteria, thus increasing BOD, and further straining dissolved oxygen levels (Mallin and Cahoon 2020). The lower Cape Fear River is currently on the North Carolina list of impaired waters for low dissolved oxygen, meaning the levels of dissolved oxygen are persistently below state water quality standards. These conditions can lead to fish kills and make waters unsafe for recreation. The Cape Fear River system also hosts dwindling numbers of Atlantic sturgeon, which are federally-listed endangered species; a number of these were killed by low dissolved oxygen from high BOD from swine waste lagoon releases and human sewage releases following Hurricane Florence in 2018.

b. Ammonia volatilization from swine lagoons and sprayfields

Ammonia volatilizes from sprayfields and waste lagoons, and is transported downwind (McCulloch et al. 1998; Aneja et al. 2000; Walker et al. 2000). Walker et al. (2000) documented a trend of increasing ammonium deposition in the coastal region of North Carolina, which they attributed to animal production sources. The lower Cape Fear River watershed, draining the largest concentration of CAFOS in the state, showed a 30% increase in airborne ammonium from the period 1988 to 2003 (Willey et al. 2006), which was attributed to increases in swine industry airborne discharges. Research has demonstrated that ammonia aerosolized from lagoons and sprayfields comes back to earth within 60 miles of its point of origin, where many sensitive water bodies on the NC Coastal Plain are located (Costanza et al. 2008). This form of nitrogen loading can contribute to the eutrophication of streams, lakes, rivers and estuaries. Ammonia can also reduce dissolved oxygen in surface waters, thereby increasing BOD. This is because ammonium is a chemically reduced compound; when it enters a stream and encounters dissolved oxygen in the presence of nitrifying bacteria, it absorbs the dissolved oxygen and is converted to nitrate, an oxidized compound. The net result is that this process removes dissolved oxygen from the water, stressing aquatic life.

c. Groundwater contamination

The Hog Operations sit on generally sandy, permeable soils (Autryville soils are the most abundant; others present include Wagram, Norfolk, Johns and Lumbee), and there is a high water table (0-2 meters). Permeability of Autryville soils is classified as moderately rapid (2.0-6.0 inches/day) and the permeability of the others as generally moderate, or 0.6-2.0 inches/day (USDA-NRCS website). This combination makes downward movement of rainfall—and nitrate-containing liquids sprayed on fields—to the water table quite rapid, hours to a few days maximum. Nitrate has been documented to readily enter groundwater from agricultural sources (Liebhart et al. 1979; Keeney 1986; Ritter and Chirnside 1990), where it can impact drinking water. Elevated nitrate in well water is known to cause potentially-fatal methemoglobinemia, or blue-baby syndrome (Johnson and Koss 1990); as such the U.S. EPA and Canada established drinking water standards for nitrate at 10 mg/L. Additionally, there is a growing body of evidence suggesting that elevated nitrate levels are associated with other human health factors including various cancers (Temkin et al. 2019). A study by the North Carolina Department of Health and Human Services in the late 1990s found elevated nitrate (above 10 mg/L) in 10% of 1595 wells tested statewide. However, the percent of contaminated wells that were adjacent to swine CAFOs (called ILOs in the report) was *three times* the statewide average for nitrate contaminated wells elsewhere. Sampson and Duplin counties showed the highest percentage of contaminated wells, with leaking hog waste lagoons and sprayfields specifically noted as responsible for the exceedances in some instances (Rudo 1999).

On a chronic, long-term basis swine waste lagoons and sprayfield waste pollute area groundwater with nitrate, an oxidized form of dissolved nitrogen. Groundwater sampling wells near hog operations have been found to far exceed the nitrate drinking water standard of 10 mg/L, in some cases up to 70 mg/L (Westerman et al. 1987; Ritter and Chirnside 1990), and ammonium in groundwater exceeded 500 mg/L in some sampling sites near swine CAFOs (Westerman et al. 1995). To put this concentration into perspective, the ammonium concentration of raw human sewage is about 15 mg/L; (Clark et al. 1977) and the concentration of ammonium known to stimulate growth of algae in Coastal Plain streamwater is 0.50 mg/L or less (Mallin and Cahoon 2020). Many lagoons are built on soils of low permeability or soils which seal through biological action or sedimentation, but clay liners or synthetic liners are required where the soils on site are too permeable (Zering 2005). Unlined lagoons are clearly subject to leakage (Westerman et al. 1995); however, even clay liners crack and significant quantities of nitrate leak into the soil beneath lagoons where it will enter groundwater (Ritter and Chirnside 1990). The studies cited above found nitrate levels in groundwater below sprayfields to be many times the 10 mg/L drinking water standard, and an NC Health Department study demonstrated nitrate polluted drinking wells were most concentrated near swine CAFOs, to put groundwater pollution into perspective.

In a study I conducted along Stocking Head Creek, in Duplin County, N.C., Mallin et al. (2015), my colleagues and I found very high concentrations of fecal coliform bacteria, ammonium, and nitrate in a stream passing through a watershed with numerous swine and poultry CAFOs. There was no difference in fecal coliform, ammonium, and nitrate concentrations between wet periods (rain within 48 hours prior to sampling) and dry periods, indicating that waste spraying caused percolation of pollutants through the soil to the groundwater table, causing chronic groundwater contamination, which moved laterally into the stream from there.

d. Nitrate from swine CAFO traveling well offsite under normal operations

Chemists use isotopic ratio techniques (^{15}N) to trace the origins of nitrogen in the environment; for instance the ^{15}N signal of atmospheric nitrogen "fixed" by crop legumes or blue-green algae is approximately zero, nitrogen from chemical fertilizers is around 0-2, and nitrogen from swine waste is high, 10-20 or more. Using nitrogen isotopic (^{15}N) techniques Karr et al., (2001) traced nitrate generated from swine CAFOs through shallow groundwater into receiving stream waters, and at least 1.5 km downstream. A later and much broader ^{15}N study in eastern North Carolina by Harden (2015) also used ^{15}N isotopic techniques and found that CAFO-influenced streams were significantly more enriched in ^{15}N than control streams with no

CAFOs, and the ^{15}N values in the CAFO sites agreed with ^{15}N signals common to manure signals. A third more recent study found highly enriched ^{15}N signatures associated with streams draining swine CAFO-impacted areas (especially following the onset of spraying season), as well as two sites impacted by human wastewater discharges, whereas a near-pristine control stream and two sites near the ocean showed unenriched ^{15}N signatures (Brown et al. 2020). In that study high nitrate concentrations were positively correlated with enriched ^{15}N signatures, indicating that swine waste and sewage were the primary sources of high nitrate to the system rather than chemical fertilizers. On one occasion during spraying season when high river flow occurred, CAFO-generated ^{15}N was detected scores of miles downstream to the upper estuary (Brown et al. 2020).

e. Failures of animal waste lagoons

Animal waste lagoons are subject to wall failures (breaches) and overtoppings (spills), which can result in the release of untreated hog waste into waterways, polluting those waterways and degrading water quality. Severe weather conditions can lead to lagoon failures. This has been documented after numerous storms over the last few decades. The 1995 hurricane season resulted in several CAFO spills in North Carolina – New River (swine), 25,000,000 gallons and Harris Creek (swine) 1,000,000 gallons. More recently, in 2018, Hurricane Florence led to the spill of millions of gallons of hog waste polluting the Cape Fear River. A recent report by state regulators (NCDEQ 2020) found that several swine operations in the Stocking Head Creek watershed had their entire operations or their sprayfields inundated by floodwaters from Hurricane Florence in 2018. Please see photos below for examples of flooded CAFO operations as a result of hurricanes. According to numerous scientific papers in the past decade storms are getting stronger, and stronger storms are expected to increase in number. With stronger storms comes greater flooding; besides Florence, we recently witnessed extreme flooding along the Gulf of Mexico from Hurricane Ida. Thus, I expect further flooding of swine production facilities in Eastern North Carolina, whether on the flood plain or beyond it.

Structural failures or equipment malfunctions can also lead to spills at hog operations. In 2017, a pipe at the Waters Farm broke, leading to 900 gallons of hog waste into an unnamed tributary of Stewarts Creek. In 2019, a pipe broke at Waters Farm leading to approximately 800 gallons of wastewater reaching surface waters. Those accidents point out how close the Waters Farm operation is to waters of the state. These and other types of accidents causing water pollution continue to periodically occur in North Carolina swine waste management systems using lagoons.

Swine lagoon accidents have caused severe water quality damage in receiving streams, with excessive loading of nitrogen, phosphorus, fecal microbes, and BOD; these incidents have also caused very low dissolved oxygen and fish and crustacean kills in receiving waters (Burkholder et al. 1997; Mallin et al. 1997).

f. Vulnerability to flooding and lagoon failure

Whereas any individual CAFO may show localized pollution, the numerous swine CAFOs currently present, coupled with an ever-increasing number of poultry CAFOs (Patt

2017), pose major chronic, long-term pollution issues to Coastal Plain watersheds. Many CAFOs are located on river floodplains, which makes their lagoons or sprayfields more vulnerable to flooding by hurricane floodwaters (see photos below), as well as to lagoon breaches through weakened walls. A GIS-based analysis of areas flooded during Hurricane Floyd estimated that 241 CAFOs were located within flooded areas, as estimated by satellite photography (Wing et al. 2002). However, Hurricane Florence caused flooding far outside of the 100-year floodplain (even I-40 was flooded for days), so being located outside of the floodplain does not guarantee freedom from flooding.

Above: Flooded swine and poultry CAFOs after Hurricanes Floyd and Florence in Eastern North Carolina (photos courtesy of Rick Dove, Waterkeeper Alliance, and Walker Golder, Audubon).

According to FEMA maps provided by Murphy Brown, some sprayfields at Farm 2037 & 2038 (Goodson Farm) are located in the floodplain, and a sprayfield at Kilpatrick Farm is located close to the 100-year floodplain and nearby streams. These sprayfields are particularly vulnerable to flooding during major rain events.

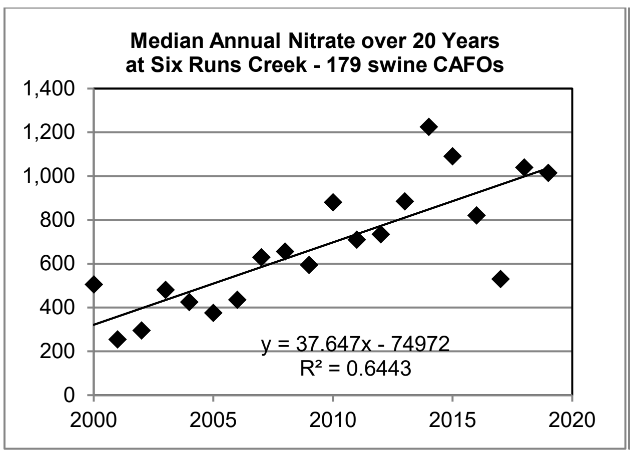
III. Hog CAFOs have a cumulative effect on water quality in the Cape Fear River basin.

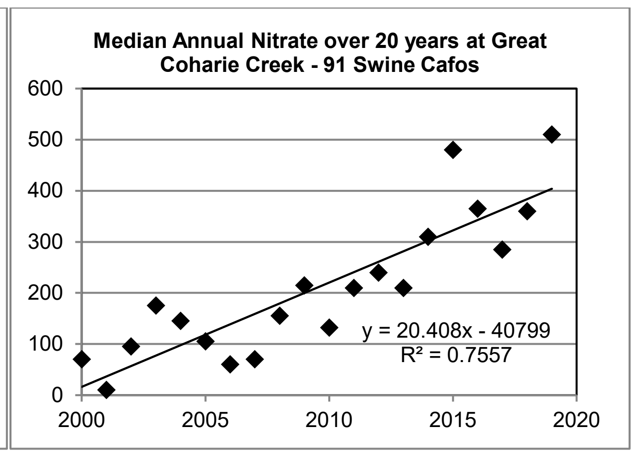
In my 2015 Stocking Head Creek Study, Mallin et al. (2015), my colleagues and I investigated chronic pollutant loading to a 2^{nd} order stream (Stocking Head Creek) with a watershed containing 13 swine and 11 poultry CAFOs. At a stream site adjoining a sprayfield, ammonium ranged up to 38 mg/L, and Total Phosphorus up to 11 mg/L, with fecal coliform counts up to 60,000 CFU/100 mL. Stream sites well downstream from sprayfields had average nitrate concentrations > 6 mg/L, ranging up to 13.6 mg/L. Stream BOD5 concentrations ranged up to 26 mg/L, and were strongly correlated with ammonium ($r = 0.666$, $p<0.0001$ and TP ($r = 0.626$, $p <0.0001$); BOD5 of unpolluted streams is typically 1 to 2 mg/L. Additionally, a broad-scale North Carolina USGS study found that streams whose watersheds contained swine and poultry CAFOs had significantly higher concentrations of ammonium, nitrate and total N than streams whose watersheds lacked CAFOs (Harden 2015). Several other studies have also demonstrated high nitrate concentrations occurring in CAFO-influenced streams (Weldon and Hornbuckle 2006; Hoorman et al. 2008; Mallin and McIver 2018; NCDEQ 2020).

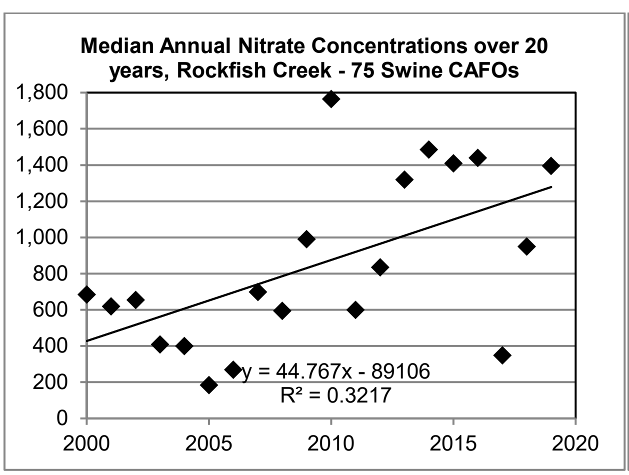
Initial sampling results by DEQ as part of the Cape Fear River Animal Operations Monitoring Study (DEQ 2020) indicate that Murphey's Creek downstream of the Waters Farm has elevated levels of nitrate/nitrite. And ongoing sampling efforts in Stewarts Creek, adjacent to Kilpatrick farm AWS820005, by Cape Fear River Watch likewise show high nitrate and fecal bacteria concentrations. Critically, it has been demonstrated (Mallin and McIver 2018) that

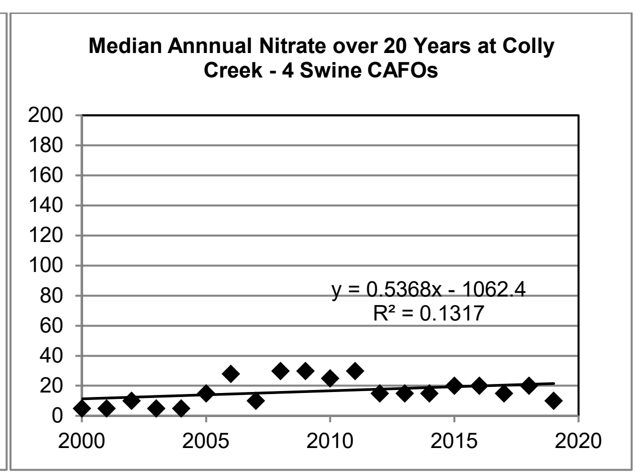
CAFO-influenced stream sites contain significantly higher concentrations of nitrate, total nitrogen, total organic carbon, and fecal coliform bacteria in mid-summer (spraying season) than in early March (spraying is only allowed March 1 – September 30). This indicates that spraying of waste leads to higher levels of off-site pollution than in non-spraying periods.

Several stream stations in the lower Cape Fear watershed have been sampled monthly since 1995 using state-certified techniques by the Lower Cape Fear River Program (LCFRP), which reports these data to DEQ. A number of long-term stream sampling sites collect from watersheds where copious swine and poultry CAFOs are located (Six Runs Creek, Great Coharie Creek, and Rockfish Creek for instance) and have shown increases in nitrate over time to quite high levels (see figures below), whereas Colly Creek, with very few watershed CAFOs, has low nitrate with no significant increase. Farm 2037 & 2038 (Goodson Farm,) is on Six Runs Creek, and is located upstream of the Six Runs Creek long-term LCFRP sampling site. Kilpatrick Farm and Waters Farm (in part) are also located upstream of the Six Runs Creek long-term LCFRP sampling site, which drains into Stewarts Creek, which drains into Six Runs Creek. Waters Farm is partially on Murphey's Creek, draining into Rockfish Creek, which is upstream of the long-term LCFRP sampling site.









Explanation of graphs: The median values (from the monthly nitrate concentrations) for each year over the 20-year period 2000-2019 were computed. Six Runs Creek, Great Coharie Creek and Rockfish Creek all drain watersheds with many swine CAFOs and strong increasing trends in nitrate are evident for all. In contrast, the Colly Creek watershed has very few CAFOs but abundant wetlands coverage and shows extremely low nitrate and no significant increase. Swine CAFO numbers are approximations.

IV. The permits will not prevent adverse impacts to water quality

If a swine waste lagoon is covered, as some lagoons at the Hog Operations will be, this will greatly reduce the volatilization of microscopic (< 2.5 μg) air pollutants from the lagoon (Sousan et al. 2021). However, covered waste lagoons can concentrate pollutants within. North Carolina researchers performed a study of covered versus uncovered waste lagoons and found that the liquid waste in the covered lagoon contained 2.3X the ammonium concentration of the liquid waste uncovered lagoon (Ducey et al. 2019). That same study found that inorganic phosphate in the liquid waste in the covered lagoon was 3.1X the concentration of the liquid waste in the uncovered lagoon (Ducey et al. 2019). Furthermore, that same study found that the concentrations of fecal coliform bacteria and *Escherichia coli* (*E. coli*) bacteria in the liquid waste in the covered lagoon were significantly higher than in the uncovered lagoon.

This means that when waste from covered lagoons are sprayed out on adjoining fields, the concentration of nitrogen, phosphorus and fecal bacteria are actually higher than that of waste sprayed from an uncovered lagoon. Another North Carolina study of a covered waste lagoon operation containing an anaerobic digester (Sousan et al. 2021) found that stream waters downstream of the sprayfield had 8X more ammonium plus nitrate than upstream of the sprayfield, and a seep discharging groundwater from the sprayfield yielded an average concentration of 32.0 mg/L of ammonium plus nitrate. On two incidents during and after spraying, Sousan et al. (2021) found that the ammonium plus nitrate reached the remarkably high concentrations of 140 mg/L and 83 mg/L downstream of the sprayfield, a condition that persisted for some 20 hours.

The new waste management systems at the Hog Operations are likely to increase pollution to waters of the state off-site and downstream of the sites. According to the literature noted above, waste from covered lagoons, as allowed under the permits, show higher ammonia concentrations than uncovered lagoons; the waste at the permitted operations will be pumped into open lagoons (which have very high nutrient and bacteria concentrations) and then the waste will be sprayed on sprayfields. Ammonia will volatilize from both the open lagoons and sprayfields. Numerous peer-reviewed publications and my research have demonstrated that spraying of swine waste onto fields, as will be done at the Hog Operations, causes nutrient and bacterial pollution of stream waters outside of the sprayfield. Movement overland to the nearest stream through stormwater runoff occurs on swine waste deposition fields, like the ones at the four permitted Hog Operations, as do operational accidents.

As noted above, the Hog Operations sit on Autryville and other sandy, permeable soils, and the water table in this area is generally high. This facilitates nitrate movement into the groundwater, from where it moves laterally to the nearest stream to become part of its base flow, from where it is carried far downstream from the facility's property.

Adding to the risk of water pollution are the locations of the Hog Operations: a sprayfield at Farm 2037 & 2038 (Goodson Farm) is located in the floodplain, and a sprayfield at the Kilpatrick Farm appears to be close to the floodplain. The proximity of these fields to streams increases the chances of flooding, and thus movement of pollutants in land-applied waste into waterways. This is particularly troubling in light of the increasing severity and frequency of rain events.

At least the Waters Farm has been previously cited for discharging waste to nearby streams; these types of spills increase threats to waterways because untreated hog waste contains harmful pollutants. Note that the discharge incidents also demonstrate the proximity of the facility to nearby streams; the closer a lagoon or sprayfield is to a stream the greater is the threat for off-site pollution. If the more concentrated waste from the digesters, secondary lagoons, or sprayfields from the Hog Operations reaches waterways as a result of an accident, equipment malfunction, flooding, over-application, or otherwise, this further threatens water quality nearby.

The permits also fail to protect water quality because they do not require regular sampling of nearby streams or of groundwater to monitor any impacts to water quality. As noted above, sampling waterways and groundwater near and at hog operations shows elevated levels of numerous pollutants, including bacteria, fecal coliform, several forms of nitrogen, and phosphorus and decreased dissolved oxygen levels. These conditions can cause algal blooms and fish kills and human health impacts.

Given the documented pollution problems caused by components of the waste management system authorized by these permits and the additional risks of increased pollution from more concentrated waste, these permits, as written, will not prevent off-site water pollution.

V. Conclusion

In summary, the Hog Operations are located in the North Carolina Coastal Plain. The current waste management system of holding highly concentrated waste in storage lagoons and spraying out on fields leads to chronic pollution of groundwater and surface water by high concentrations of nitrogen, phosphorus, and fecal bacteria. Vast amounts of ammonia are volatilized from open lagoons and sprayfields, travel downwind, and are deposited in sensitive coastal waters. This waste disposal method presents direct threats to natural ecosystems and biota, as well as human health threats to downwind and downstream residents. Research indicates that the waste management system allowed by these permits will change the composition of the waste that is stored in open lagoons and land-applied to fields. As a result, these permits will not prevent adverse impacts on the environment or cumulative effects on ground water or surface water quality in the river basin, and in fact, waste discharges from the Hog Operations as permitted are likely to increase off-site water pollution. In particular, the waste management system allowed by these permits is likely to increase nitrate concentrations in groundwater and reduce dissolved oxygen, increase BOD levels, and increase bacteria in surface waters, which could lead to harmful algal blooms, fish kills, and human health impacts.

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<u>Exhibit 3</u>

Compiled by Dr. Viney P. Aneja, PhD

September 3, 2021

There is abundant scientific evidence that swine concentrated animal feeding operations (CAFOs) operated in North Carolina have adverse impacts on the environment. One of these well-documented impacts are emissions of ammonia (NH_3) from hog waste lagoons, spraying and fields where the waste is land-applied. The ammonia emitted by these sources also deposits on the ecosystem, including surface waters where it can cause increased oxygen demand and eutrophication and can impact aquatic ecosystems and harm fish and other aquatic organisms. Airborne ammonia also deposits on land, where it can overload soil with nitrogen and increase nitrate leaching into groundwater, and make well water unsafe to drink. There is a concern that swine-waste biogas production could exacerbate these problems by emitting more reactive nitrogen, including ammonia, into the atmosphere. This report considers this risk and concludes that storing digestate in open lagoons and land-applying it to field may increase ammonia emissions from hog CAFOs and contribute to further degradation of local air and water quality.

I. Nitrogen cycle in North Carolina hog animal waste management systems

In North Carolina, Concentrated Animal Feeding Operations (CAFOs) are used extensively for meat production. Though the term Concentrated Animal Feeding Operation bears a technical definition under the Clean Water Act, here it is used to refer generally to a production model that raises large numbers of animals in confinement where they are fed and watered until they are slaughtered. Unlike traditional models of livestock husbandry, animals raised in CAFOs do not roam to forage and the feed is produced off-site. The CAFO model of production is used to produce beef, dairy, hogs, poultry, milk, and eggs. The majority of CAFOs in North Carolina produce are either broiler chickens or hogs.

North Carolina witnessed intense growth in its hog industry during the 1990s (Aneja et al., 2000). Due to the large number of animals raised in a concentrated location, CAFOs produce large volumes of waste (US EPA, 2004). In traditional animal production models, animal waste is deposited throughout the environment as the livestock forage. However, in the CAFO model of production, the waste accumulates within the barn. The most common system for disposing of this massive amount of waste is known as the lagoon and sprayfield system. In general, the floor of the swine barns is made of concrete with slats, allowing the urine and feces excreted by the hogs to fall into an underground storage pit below the barn. Depending on the design of the CAFO, the waste either remains in the pit for months before it is scraped out or is flushed out with lagoon water periodically. In North Carolina, there has been a significant shift towards flush systems that remove the waste from barns more frequently. At least three of the four hog operations at issue in this case use the flushing method of removing hog urine and feces from below the barn.[1]

In the conventional lagoon and sprayfield system, once waste is removed from the storage pit, it is transferred into an open-air retention pond or "lagoon" that stores millions of gallons of animal waste. This waste contains bacteria, nutrients such as phosphorus and nitrogen, and heavy metals including arsenic, cadmium, copper and zinc. The pH of waste in the lagoon is manipulated to favor bacteria that anaerobically digest the waste. The liquid waste rises to the top, and nutrient and elemental rich sludge forms at the bottom. The sludge is periodically removed and applied to land. The liquid waste is frequently applied as fertilizer to growing fields, known as spray-fields, via high-pressure sprayers. The waste may also be applied through other methods such as drag-hose application of waste to the surface of the land. Strategies such as injection, which incorporate the waste into the soil and help limit emissions, have not been widely adopted in North Carolina. The spray-fields grow crops such as hay and Bermuda grass in order to absorb the nutrients contained in the waste. In North Carolina, over 2,200 swine operations are permitted to use this kind of animal waste management system. Hog production in North Carolina is overwhelmingly centered in the Eastern Coastal Plain, particularly in Robeson, Columbus, Bladen, Sampson, Pender, Duplin, Onslow, Wayne, Lenoir, Greene, and Pitt counties.

One of the main drawbacks of managing waste in this manner is the effect on air and water quality. CAFOs are significant contributors to air pollution, which often disproportionately impacts low-income and minority communities (Wing et al., 2000). Hog barns, lagoons, land application (i.e. spraying) of animal waste, and land biogenic emissions all emit large quantities of ammonia and other pollutants into the atmosphere (Aneja et al. 2001, 2008, 2009).

Ammonia (NH3), a form of reactive nitrogen, is the most abundant gas-phase alkaline species in the atmosphere. Ammonia emissions from animal agriculture result from the degradation of urea by the ubiquitous enzyme urease, which results in ammonium (NH4+) formation Urea is mainly excreted in the animal urine and once it is hydrolyzed it is much more prone to ammonia emissions than organic nitrogen excreted in the feces.

Ammonia emitted by hog operations is transported and dispersed by wind and is deposited on surface waters or land through dry deposition or wet deposition (Figure 1) (Aneja et al., 2001). Multiple studies have modelled the dispersion patterns of ammonia from CAFOs in Eastern North Carolina (Walker et al., 2000; Costanza et al., 2008; Bajwa et al., 2008). These studies have established that ammonia produced by hog CAFOs deposits a significant amount of nitrogen into the Cape Fear River Basin.

When ammonia directly or indirectly deposits into surface waters it can cause algal blooms and eutrophication (Costanza et al., 2008; Aneja et al., 2001). These conditions in turn cause hypoxia—low oxygen levels—in rivers and streams that alters aquatic ecosystems and harms fish and other species (Costanza et al., 2008).

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[1] *See, e.g.,* Letter from Jeff P. Cappadona, Cavanaugh & Associates, to Christine Lawson, DEQ, Att. 1 Anaerobic Digester System O&M 3-4 (Jan. 30, 2020).

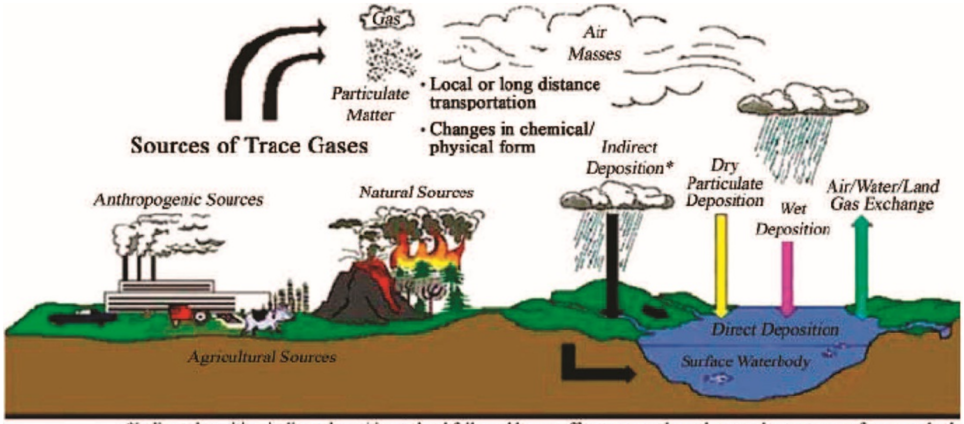

Fig. 1. Atmospheric emissions, transport, transformation and deposition of trace gases. (Aneja et al., 2001)

NH3 can neutralize sulfuric acid and nitric acid in the atmosphere to form fine particulate matter with an aerodynamic diameter <2.5 µm (PM2.5), which is closely linked to health and climatic effects. PM2.5 can penetrate deep into people's lungs and bloodstream and affect respiratory and cardiovascular health. PM2.5 exposure has been linked to a variety of health problems including heart attacks, irregular heartbeat, aggravated asthma, decreased lung function, premature death in people with heart or lung disease, and increased respiratory symptoms such as irritation of the airways, coughing, and difficulty breathing (EPA 2021). PM2.5 also has significant environmental effects, including formation of haze. PM2.5 can be carried over long distances by wind and settle on land or surface waters. When the PM2.5 containing ammonium/ammonia settles in surface waters it can increase the acidity or cause nutrient overloading, which leads to algal blooms and eutrophication.

High concentrations of ammonia, PM2.5, and other pollutants associated with hog CAFOs have a significant detrimental impact on the health and wellbeing of people living nearby. A study published by researchers from Duke University in 2018 found, after controlling for other factors, that for North Carolinians who live near hog CAFOs that use lagoons and sprayfields, mortality rates were substantially higher from causes such as anemia, kidney disease, tuberculosis, and lower birth rates than people who live further away from these operations (Kravchenko et al., 2018). Another recent study linked ammonia and particulate matter created by hog CAFOs to increased mortality rates in nearby communities (Domingo et al., 2021). CAFOs create areas of highly concentrated air pollution and odor that impairs the quality of life of nearby communities. Vulnerable populations are most at risk from the health impacts of CAFO-produced pollutants. Children, for instance, inhale 20-50% more air than adults, and air pollution can exacerbate existing health conditions in the elderly. In addition to affecting health, CAFO-produced pollution has substantial social impacts. Odor, for instance, may be detected several miles from CAFOs.

The waste treatment lagoons that CAFOs use to manage animal waste in North Carolina are a public hazard in times of extreme weather events e.g. hurricanes (Aneja et al., 2001). These events can cause lagoon overflows, which highly contaminate soil, surface water and groundwater used for wells with nutrients and pathogens impacting human health and the environment. Environmentally, these events cause extreme nutrient overload in waterways, which can have a negative impact on entire ecosystems by causing events like algae blooms.

II. Impact of retrofitting existing animal waste management systems to produce biogas on ammonia emissions

Anaerobic digestion (AD) is a method for converting biomass into bioenergy. Livestock manure is a commonly used biomass material for production of bioenergy.

Many livestock (hog and cattle) manure treatment systems rely on open lagoons where the CH_4, CO_2, NH_3 (ammonia) and other gases, such as reduced sulfur compounds, volatile organic compounds (VOCs) are emitted into the atmosphere. When these open systems are covered, gaseous emissions except ammonia are reduced, which results in the effluent leaving the anaerobic digester, known as digestate, with a modified chemical content (e.g. total solids, carbon, ammonia, ammonium (NH_4^+), and pH), relative to waste from a conventional open lagoon system. TAN content and pH in digested slurry are higher than in untreated slurry. Thus, potential for ammonia emissions during subsequent slurry storage are increased (Baines, 2021). The digestate contains more ammonium (NH_4^+) due to reduction in ammonia emissions from the anaerobic digester (i.e. covered lagoon) to the atmosphere, and has less degradable biomass carbon than the substrate in an open lagoon resulting in changes in GHG and NH_3 emissions (Baines, 2021).

The production of biogas through AD of livestock manure is a complex process. It involves a variety of physiological and biochemical metabolic pathways, the essence of which is the material and energy metabolism of microorganisms under anaerobic conditions. AD may be classified typically into three stages according to the utilization and transformation of organic matter (Baines, 2021):

1. Hydrolysis;

2. Acidogenesis; and

3. Methanogenesis.

In the hydrolysis step, macromolecular organic matters (fat, carbohydrate, protein, etc.) are hydrolyzed into small molecules such as monosaccharides, amino acids, fatty acids and so on by the action of extracellular enzymes. In the Acidogenesis step, the small molecule organic compounds are converted to a volatile organic acid, ethanol etc. by the acidified bacteria. H_2, CO_2 and acetic acid are then formed under the action of hydrogenic bacteria and acetogenic bacteria. Finally, in the Methanogenesis step, the methanogenic bacteria synthesize methane using acetic acid, H_2, CO_2 etc. in the methanogenesis stage.

In general, when the digestion process is complete, the pH of the digestate hovers between 7.5 and 8. The pH value in the course of anaerobic digestion is the result of the acid-alkali balance; the pH value decreasing with the increasing of organic acids, and increasing with the increases of Total Ammoniacal Nitrogen (TAN) (TAN = ammonium + ammonia), which is the product of the decomposition of nitrogenous organics (Lorimor and Sawyer, 2004; Grabow; Baines, 2021). The mass of total N is not significantly changed by anaerobic digestion, however the mass of organic nitrogen decreases and the mass of ammoniacal nitrogen increases. Organic N decreases as it is mineralized to TAN (Total Ammoniacal Nitrogen=NH4+N + NH3); i.e. NH3 N expressed as a percentage of TKN increases as manure is digested. Both the TAN and the mineral N (mineral N= TAN + NO3- N) increase. The impact of N transformations is to increase the fraction of total-N in the total ammoniacal form. This is important since the ammoniacal form is composed of both dissolved ammonium (NH4+) and ammonia (NH3) gas. Ammonia gas is emitted into the atmosphere from the open lagoon, i.e. secondary lagoon, during land application (i.e. spraying) of animal waste, and land biogenic emissions of the digested effluent. The amount of NH3 lost during land application of the digestate depends on multiple factors, including the method of application (generally high-pressure spray versus low pressure application or injection) and temperature. (Nyord et al. 2012; Dari et al., 2019).

Anaerobic digesters do not significantly change the nutrient quantity as nitrogen and phosphorus are retained, only the carbon is reduced through conversion and degassing of methane and carbon dioxide. The mass of organic nitrogen is decreased, and it is mineralized to TAN. Thus, ammonia and ammonium are found at higher concentrations in liquid digestates than raw manure (Nkoa, 2014). Therefore, anaerobic digestates stored in open secondary lagoons and land-applied to fields have higher NH3 emission potential than undigested animal manures and slurries (Aneja et al., 2008; Alburquerque et al., 2012; Nkoa, 2014) especially as the temperature increases. An increase in NH3 emissions during the summer from a secondary lagoon filled with digestate relative to conventional open lagoons was documented in North Carolina in 2008 as part of the North Carolina State University Animal and Poultry Waste Management Center's evaluation of potentially environmentally superior technologies. (Aneja 2008). Factors such as temperature and pH may alter the equilibrium between ammonia and ammonium. For example, increasing temperature and pH will enhance ammonia emissions. (Angelidaki et al., 2003; Weaver et al., 2012; Nkoa, 2014). Air movement across an open lagoon surface also enhances NH3 loss. (Dari et al., 2019).

Once digestate is removed from lagoons and applied to fields it has the potential to emit ammonia at a greater rate than conventional hog waste, as depicted in Figure 2. (Nyord et al., 2012). Most studies evaluating ammonia emissions from land-applied digestate have evaluated land-application methods such as drag-hose irrigation or injection. (Nyord et al. 2012; Chantigny et al. 2009). It is well-established that these methods of land-application produce substantially less ammonia emissions than high-pressure spray hoses, (Grabow 2007), which are the land-application method of choice on hog operations in North Carolina. Some studies evaluating ammonia emissions from land-applied digestate have also used data from much colder climates than North Carolina. (Chantigny et al. 2009). It is well-established that low temperatures inhibit NH3 volatilization. (Dari et al., 2019). Therefore, it is reasonable to believe that land-application of digestate effluent through spraying in North Carolina, particularly during the summer, is likely to cause greater ammonia emissions than documented in these studies.

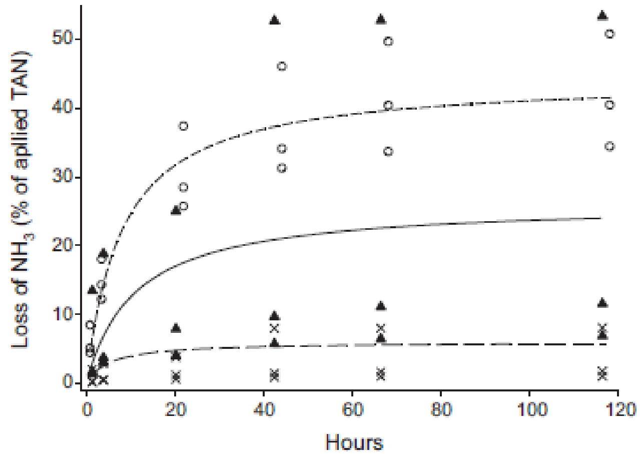

Fig. 3. Ammonia emissions 2008. Cumulative percentage loss of total applied TAN through emission over 116 h after application of pig slurry by trailing hoses. Regression lines and symbols represent cumulative losses of NH_3 emission from each measuring period; ---- and ○: digested slurry; — and ▲: untreated slurry; — — and ×: separated slurry.

Figure 2: Cumulative Ammonia loss by digestate, cattle, and pig slurry spread onto land under identical conditions (Nyord et al. 2012)

III. Impact of DEQ's issuance of permits to M-B to incorporate anaerobic digesters (covered lagoon) into the lagoon & sprayfield system

The permits issued by DEQ for the Waters, Kilpatrick, Benson, and Farm 2037/2038 hog operations authorize Murphy-Brown to modify its existing waste management systems to produce biogas, store digestate in open secondary lagoons, and apply digestate to fields. Instead of flushing hog urine and feces from barns into open lagoons the permits allow the hog operations to store the urine, feces, and water in covered lagoons where, as described in the previous section, biogas will be produced through the process of methanogenesis. The biogas along with other gases trapped under the lagoon cover will be siphoned off to be conditioned and then processed at a central plant.

The digestate will have a different composition from conventional waste, including a significantly higher concentration of NH3/NH4+. The digestate will be transferred to a secondary open lagoon. Ammonia emissions from the secondary lagoon are likely to exceed those expected from a similarly situated conventional lagoon system. (Aneja et al., 2008; Alburquerque et al., 2012; Nkoa, 2014). When the digestate effluent is land-applied, primarily through spraying, this higher rate of ammonia emissions is likely to continue. (Nyord et al., 2012).

The increased ammonia emissions from the secondary open lagoons and land-applied digestate effluent relative to a conventional lagoon and sprayfield system means that a technology intended to benefit the environment, biogas production and capture, may worsens air and water pollution coming from these four hog operations (Harper et al., 2010). The anticipated increase in ammonia emissions could cause more ammonia to be deposited onto nearby soil, where it can seep into groundwater or runoff into surface waters, and directly into surface waters, where it causes algal blooms, low oxygen conditions, and harms aquatic ecosystems.

IV. Availability of technology that mitigates ammonia emissions from lagoons & sprayfield system

Emissions of reactive nitrogen in animal waste can be reduced through various technologies and practices. (Aneja et al., 2009; Szogi et al., 2014). Farm-specific feed management practices cannot be enforced by regulators, but waste management technology requirements can. For example, in the Netherlands, Denmark, and UK, manure injection into soil, rather than spraying, has long been mandated to reduce ammonia emissions. However, technological fixes must be assessed for potential pollutant swapping, i.e., the increased emission of one pollutant resulting from abating another. One example of pollution swapping is increased nitrate leaching that may result from switching to manure injection without reducing the nitrogen application rate. (Aneja et al., 2009).

Over a decade ago, an engineered waste management system known as "Super Soils" was developed and tested in North Carolina. (Aneja et al., 2008). The system included a module that removed nitrogen from waste through a process known as nitrification-denitrification. The system reported a 73% reduction in ammonia emissions from hog operations. (Szogi et al., 2006). The Super Soils system also created an additional income stream for farmers by allowing them to sell concentrated nutrient byproducts of the system as fertilizer. The Super Soils technology has been improved upon twice. (Szogi et al., 2014). Though there is no lagoon in the Super Soil System for storage of the animal manure, the system's modules have been adapted elsewhere to work alongside covered lagoons producing biogas. (Schmidt, 2009).

V. Conclusion

In sum, hog waste lagoons and sprayfields are a significant source of ammonia emissions, which harm local air and water quality and enhance formation of PM2.5, which is harmful to human health. The system permitted by DEQ, which includes a covered lagoon anaerobic digester, secondary uncovered lagoons that store digestate, and land application of the digestate through spraying, is likely to increase those ammonia emissions. The process of anaerobic digestion alters the composition of hog waste such that the digestate that comes out of the covered lagoon has a higher concentration of total ammoniacal nitrogen (TAN) than conventional waste. This change in composition increases ammonia emissions from lagoons and sprayfields fertilized with digestate relative to conventional lagoon and spray waste management systems. The expected increase in ammonia emissions, may in turn exacerbate air and water quality impacts. There are technologies available that reduce the nitrogen content of digestate and can therefore decrease ammonia emissions, avoiding significant environmental and public health impacts.

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<u>Exhibit 4</u>

Summary of Expert Opinions
Dr. Shane W. Rogers
September 10, 2021

1. **The lagoon and spray swine waste manure management system used at swine concentrated animal feeding operations (CAFOs) in North Carolina leads to negative environmental and public health consequences.**[1]

   A. The U.S. Environmental Protection Agency defines an Animal Feeding Operation (AFO) as one in which animals have been, are, or will be stabled or confined and fed or maintained for a total of 45 days or more in any 12-month period, and crops, vegetation, forage growth, or post-harvest residues are not sustained in the normal growing season over any portion of the lot or facility."[2] Concentrated Animal Feeding Operations (CAFOs) are generally identified as the largest of the AFO operations owing their potential for significant pollution. Large swine CAFOs are those operations that confine at least 2,500 swine weighing over 55 pounds, or that confine more than 10,000 swine weighing less than 55 pounds.[3]

   B. It takes about 24 to 29 weeks for a hog to reach an average market weight of 283 pounds from birth, including about 3 weeks to wean, 6 weeks in a nursery stage, and then 16 to 20 weeks for the finishing stage.[4] A typical finishing operation for Smithfield may have 2.5 sellouts (rotation of hogs) per year. To get from birth to 283 pounds in 200 days means that the hogs must gain weight rapidly; industry reports reflect average daily weight gains of as much as 1.95 pounds as of the year 2001.[5] This requires significant food and water intake, and with that comes significant manure production. Based on reported manure production characteristics, hogs produce multiple times more waste than humans. As shown in Table 1, a 283-pound finishing hog can be expected to produce on average about 1.67 gallons (13.8 pounds) of manure each day.[6]

   C. Most swine CAFOs in North Carolina, including (at present) the subject swine CAFOs, use a lagoon and spray system of manure waste management. There are three primary

[1] Swine CAFOs may also be referred to as industrial hog operations in some scientific literature.
[2] See https://www.epa.gov/npdes/animal-feeding-operations-afos
[3] According to the U.S. EPA regulatory definitions of Large CAFOs, Medium CAFOs, and Small CAFOs, swine AFOs that are smaller than this size threshold may also be designated a CAFO owing a manmade ditch or pipe discharge of manure or wastewater to surface water, if there is direct contact of animals with surface water that passes through the confinement area, or if the permitting authority finds an operation to be a significant contributor of pollutants. See https://www.epa.gov/npdes/npdes-afos-policy-documents-0 for more details.
[4] United States Department of Agriculture, Economic Research Service, web page, Hogs and Pork, available at http://www.ers.usda.gov/topics/animal-products/hogs-pork/background.aspx (chart showing that it may take 2 to 3 weeks to wean a pig, then nursery stage for 6 weeks, then 16 to 20 weeks for the finishing stage = 24 to 29 weeks total = 168 to 203 days).
[5] National Hog Farmer, Tracking Progress in Grow-Finish, article dated October 15, 2002, available at http://nationalhogfarmer.com/mag/farming_tracking_progress_growfinish.
[6] Midwest Plan Service, 2004. "Manure Characteristics", Manure Management Systems Series, MWPS-18 Section 1, second edition, Iowa State University, Ames, IA.

sources of concern regarding pollution in this system that include the swine housing facilities, anaerobic lagoons, and spray application of swine wastes.

Table 1. Manure production characteristics of swine[7]

Life Stage	Animal size	Daily Manure Production		
	Pounds	Pounds	Gallons	Liters
Nursery	25	1.9	0.23	0.87
	40	3.0	0.37	1.40
Finishing	150	7.4	0.89	3.37
	180	8.9	1.07	4.05
	220	10.9	1.31	4.96
	260	12.8	1.55	5.87
	300	14.8	1.79	6.78
Gestating	300	6.8	0.82	3.10
	400	9.1	1.10	4.16
	500	11.4	1.37	5.19
Lactating	375	17.5	2.08	7.87
	500	23.4	2.78	10.5
	600	28.1	3.33	12.6

Animals housed in swine CAFOs continually defecate. As swine manure waste is produced in the swine houses, it falls, or is pushed through (by the animals), slots in the concrete floor into pits or flush lanes under the floor. Removal of manure waste from underfloor of the swine houses occurs 4-8 times per day (typical). This is accomplished by flushing the manure that accumulates under the slotted portion of the floors out to the lagoon using wastewater drawn from near the top of the anaerobic lagoon.

D. An anaerobic lagoon is an in-ground manure holding structure, commonly left uncovered and open to the air. The subject swine CAFO sites have historically relied on open lagoons to store manure. Aside from rain that may fall over the anaerobic lagoons, no additional water aside from that contained in swine manure is typically added to the lagoon - by design. The high waste load rapidly depletes oxygen, and thus decomposition of the waste occurs under anaerobic conditions. Anaerobic bacteria in well-functioning anaerobic lagoons break down and stabilize the organic fraction of materials. When this process is upset due to a number of potential factors such as overloading with wastes, solids accumulation and reduction of treatment volume, pH changes or other upsets, the stabilization process is greatly reduced or eliminated.

E. Anaerobic decomposition in anaerobic lagoons generates significant quantities of methane gas, carbon dioxide, ammonia, nitrous oxide, and other noxious gases that are emitted to the atmosphere along with other pollutants. Methane, a greenhouse gas, is a

[7] Source: Midwest Plan Service, 2004 Manure Characteristics, MWPS-18 Section 1, Second Edition, Jeff Lorimor, Wendy Powers, and Al Sutton (eds.). Manure Management Systems Series, Iowa State university, Ames, IA.

significant component of the biogas produced during anaerobic decomposition.[8] The quantity of ammonia nitrogen emitted is also large and of concern.[9] Its uncontrolled emission reduces the nitrogen content of the waste and poses a risk to the environment and public health. It has been estimated that 80-90% of the total ammonia emitted from livestock operations is redeposited uncontrolled within 10 km of the source, while the remainder is dispersed into the atmosphere contributing to the haze, acid rain, acidification of terrestrial and aquatic ecosystems, and eutrophication of surface water bodies.[10]

F. Swine manure wastes are held, exposed to the atmosphere, in anaerobic lagoons until such time they can be applied to land. Land application of manures is permitted in North Carolina so long as it is done in accordance with a certified animal waste management plan (CAWMP). The purpose of the CAWMP is to assure that manure is applied at a rate to meet, but not exceed, the nutrient requirements of the crop to receive the manure as a nutrient source. In general, CAWMPs consider application history and nutrients contents of the soil, nutrient content of manure, infiltration rates and application equipment used (effect on nutrient delivery to the plants), and land area over which the crop(s) will be grown. Like for most swine CAFOs in North Carolina, the subject swine CAFO sites are permitted to apply their manure waste at an agronomic rate for nitrogen. This does not assure that overapplication of phosphorus, an important nutrient that affects water quality, does not occur. It also does not assure that nitrogen and other pollutants in swine manure do not affect nearby water quality.

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[8] *See* The Humane Society of the United States, An HSUS Report: The Impact of Animal Agriculture on Global Warming and Climate Change. Available at http://www.hsi.org/assets/pdfs/farming_climate_impact.pdf (page 7 – "Storing and disposing vast quantities of manure can produce anthropogenic methane and nitrous oxide emissions. According to the Pew Center on Global Climate Change, farm animal manure management currently accounts for 25% of agricultural methane emissions in the United States and 6% of agricultural nitrous oxide emissions. As noted above, methane has 23 times the GWP of carbon dioxide, and its concentrations have increased by approximately 150% since 1750. Globally, farm animals are the most significant source of anthropogenic methane, responsible for 35-40% of methane emissions worldwide."

[9] See for example Szögi, A.A. and M. B. Vanotti (2007) Abatement of Ammonia Emissions from Swine Lagoons Using Polymer-Enhanced Solid-Liquid Separation, *Applied Engineering in Agriculture*, 23(6): 837-845; these authors measured for one anaerobic lagoon ammonia emissions of 13,633 kg /ha/yr. See also Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; these authors report from a sample of 40 reports of ammonia emissions measurements from swine waste anaerobic lagoons an average emission of 13,000 kg-N/ha/yr. See also Aneja, V.P., S.P. Arya, D.-S. Kim, I.C. Rumsey, H.L. Arkinson, H. Semunegus, K.S. Bajwa, D.A. Dickey, L.A. Stefanski, L.Todd, K.Motlus, W.P. Robarge, C.M. Williams (2008) Characterizing Ammonia Emissions from Swine Farms in Eastern North Carolina: Part 1 – Conventional Lagoon and Spray Technology for Waste Treatment, Journal of the Air & Waste Management Association, 58:1130-1144 ; these authors report ammonia emissions from three anaerobic lagoons were temperature dependent, ranging from as low as 2.2 kg-N/ha/d in the winter to as great as 57.8 kg-N/ha/d in the summer. See also Nkoa. R. (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review, *Agronomy for Sustainable Developmen*t, Springer Verlag/EDP Sciences/INRA, 34 (2):473-492. 10.1007/s13593-013-0196-z hal-01234816; on page 480 - "NH_3 emission inventories from several countries have shown that agriculture produces approximately 90% of the total emission of NH_3 to the atmosphere."

[10] See Nkoa. R. (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review, *Agronomy for Sustainable Developmen*t, Springer Verlag/EDP Sciences/INRA, 34 (2):473-492. 10.1007/s13593-013-0196-z hal-01234816, page 481.

G. When anaerobic manure lagoon effluents and solids are land-applied, nutrients, oxygen-demanding materials, metals, odors, particulates, endotoxins, bacteria, ammonia, and other harmful pollutants are released into the environment. Ammonia and other pollutants emitted to the atmosphere can move downwind or deposit uncontrolled onto land, physical structures, or waterbodies nearby. Land-applied pollutants can run off from fields in surface drainages, move via groundwater, or through tile drainage flows to affect water quality.[11] In a recent study of the U.S. Geological Survey, an overall measurable effect of swine CAFO waste manures on stream water quality in watersheds containing swine CAFOs was reported. Land application of waste manure at swine CAFOs influenced ion and nutrient chemistry in many of the North Carolina Coastal Plain streams that were studied, and the effect was directly related to higher swine barn densities and (or) higher total acres available for applying waste manure at swine CAFOs.[12]

H. In a lagoon and spray system, including the system permitted for the subject swine CAFOs, waste application is typically completed using methods of application that spray waste through the air, for example, using a pump and reel or center pivot spray applicator. Less frequently, operators may choose to use alternative application techniques such as low height or drag hose spreading or injection. The operators at the subject swine CAFOs use a combination of spraying waste and drag hose spreading.

I. Spraying swine manure through the air results in significant ammonia emissions. Direct injection of manure, such as can be accomplished with Aerway spreaders, is one alternative to spray irrigation that can reduce potential for off-site transport of harmful pollutants in manures or anaerobic digester effluents. Reduction in hydrogen sulfide emissions is estimated to be between 50-75%, but there may be a slight increase in greenhouse gas emissions (up to 10%) owing a slight increase in nitrous oxide emissions from decomposition by soil microbes. Injection also results in greater preservation of nutrients (up to 90% reduction in ammonia volatilization compared to spray application), which reduces the amount of anaerobic lagoon effluent that must be applied to land to meet crop nutrient requirements. Lower mass of anaerobic lagoon effluent applied per land area means less pollutant loading and emissions.

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[11] See, for example, (i) Harden, S.L., Rogers, S.W., Jahne, M.A., Shaffer, C.E., and Smith, D.G. 2012. Characterization of nutrients and fecal indicator bacteria at a concentrated swine feeding operation in Wake County, North Carolina, 2009–2011: U.S. Geological Survey Open-File Report 2012–1047, 31 p.; (ii) Rogers, S. and J. Haines. "Detecting and mitigating the environmental impact of fecal pathogens originating from confined animal feeding operations: review", U.S. EPA, NRMRL, EPA/600/R-06/021, September 2005. (iii) Harden, S.L. (2008) Microbial and Nutrient Concentration and Load Data During Stormwater Runoff at a Swine Concentrated Animal Feeding Operation in the North Carolina Coastal Plain, 2006-2007: U.S. Geological Survey Open-File Report 2008-1156, 22p., https://doi.org/10.3133/ofr20081156.
[12] Harden, S.L. (2015) Surface-water quality in agricultural watersheds of the North Carolina Coastal Plain associated with concentrated animal feeding operations: U.S. Geological Survey Scientific Investigation Report 2015-5080, 55 p., 7 apps., http://dx.doi.org/10.3133/sir20155080.

2. **Well-designed anaerobic digestion technologies for swine manure treatment can reduce select environmental pollution and public health risks of concern relative to anaerobic lagoons, but may exacerbate others.**

   A. Anaerobic digestion is a rapidly growing technology for farm waste management in the United States. The process of anaerobic digestion uses microbes to produce biogas, which is a mixture of methane, carbon dioxide, and trace gases. While anaerobic digestion occurs in an anaerobic lagoon, an anaerobic digester differs in that it is covered with an impermeable material rather than left open to the atmosphere and it is not exposed to diluting rainfall. Because anaerobic digesters are covered, the methane, a powerful greenhouse gas with high emissions from anaerobic lagoons, is captured rather than emitted to the atmosphere, thus reducing emissions that contribute to global warming. Owing the absence of diluting rainwater and other efficiencies, anaerobic digesters are more efficient at waste decomposition than anaerobic lagoons. Gas capture in anaerobic digesters may also reduce the amount of pathogens in the waste.

   B. Nutrients in manure are conserved during anaerobic digestion, but are converted to more readily available and mobile forms with higher potential to move with water.[13] Organic nitrogen is converted to ammoniacal nitrogen, which is not only inhibitory of the anaerobic digestion process, but can also result in higher ammonia emissions during subsequent storage if left uncovered, and during land application if not incorporated when applied.[14] For example, relative to anaerobic lagoons, ammonia emissions from anaerobically digested swine manure stored in open lagoons in one study increased by 46%. This was hypothesized to be caused by reduced methanogenesis and its reduced effect on the chemical conversion of ammonium to dinitrogen gas.[15]

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[13] See United States Department of Agriculture - Natural Resources Conservation Service, Conservation Practice Standard: Anaerobic Digester, Code 366, October 2017, 10 p. On page 6: "Consider the effects of digestion upon nutrient availability. Land application of digester effluent, compared with fresh manure, may have a higher risk for both ground and surface water quality problems. Compounds such as nitrogen, phosphorus, and other elements become more soluble due to anaerobic digestion and therefore have higher potential to move with water."

[14] See Westerman, P., M. Veal, J. Cheng, and K. Zering "Biogas Anaerobic Digester Considerations for Swine Farms in North Carolina" North Carolina State University, A&T University Cooperative Extension, 8 p.; See also Aguirre-Villegas, H., R.A. Larson, M.D. Ruark (2016) Dairy Anaerobic Digestion Systems and their Impact on Greenhouse Gas and Ammonia Emissions, Sustainable Dairy Fact Sheet Series, University of Wisconsin Extension, 5 p. See also Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; p. 1 "Anaerobically digested slurry shows higher emissions during storage for NH_3 while losses tend to be lower for CH_4 and little changes occur for N_20 and CO_2 compared to untreated slurry. All cover types are found to be efficient for emission mitigation of NH_3 from stores."

[15] Harper, L. T.K. Flesch, K.H. Weaver, J.D. Wilson (2010) The effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions, Journal of Environmental Quality, 39:1984-1992. In the abstract on pg 1984: "NH_3 emissions in the biofuel farms increased by 46% over the conventional farms. These studies show that what is considered an environmentally friendly technology had mixed results and that all components of a system should be studied when making changes to existing systems."

3. **Subject swine CAFO characteristics as currently operating and with modifications to the waste treatment system: minimal changes in the permits and certificates of coverage are not protective of air and water quality.**

A. Thousands of Smithfield hogs are being kept in each of the subject swine CAFOs as summarized in Table 2, along with the number and types of anaerobic lagoons at each site. Each of the subject swine CAFOs currently employ the lagoon and spray system of swine manure management. The subject swine CAFOs' irrigation records indicate that the predominant method of waste application is spraying using pump and reel and center pivots.[16] Murphy-Brown has indicated that there is also some limited use of drag hoses at these sites. Table 3 presents the basic characteristics of the proposed anaerobic digestion systems for each subject swine CAFO.

Table 2. Production characteristics and anaerobic lagoon characteristics of the subject swine CAFOs[17]

Facility	Operation type	Permitted hog counts	Number and type of anaerobic lagoons
Benson	Feeder to finish	6,120	One single stage
Goodson 2037/2038	Feeder to finish	20,992	One single stage and one 2-stage (primary and secondary)
Kilpatrick	Wean to Finish	13,336	One 2-stage (primary and secondary)
Merritt	Wean to Finish	5,083	One single stage
M&M	Wean to finish	12,308	Three single stage

Notes: Permitted hog count is the allowable annual average count calculated pursuant to the permit, not the number of hogs present at any particular time.

Table 3. Lagoon characteristics at each of the subject swine CAFOs[18]

[16] Data acquired from the most recent irrigation design plans provided including Benson: MB000788; Goodson 2037/2038: MB000127; Kilpatrick & Merritt: MB001143; M&M: MB001851. Document identification is by the first page number of the document.

[17] Data acquired from lagoon design information provided including Benson: MB000741; Goodson 2037/2038: MB000092 and MB000085; Kilpatrick & Merritt: MB001092 and MB001104; and M&M: MB001733, MB001741, and MB001748.

[18] Data acquired from the most recent nutrient utilization plans (NUPs), anaerobic digester permit applications, and permits or certificates of coverage issued by the DEQ provided for each of the subject swine CAFOs. These include (i) NUPs: Benson: MB000798; Goodson 2037/2038: MB000160 and MB000179; Kilpatrick: MB001154 and MB001184; M&M: MB001836; (ii) Permit / COC applications: Benson: MB000667; Goodson 2037/2038 MB000001; Kilpatrick: MB001016; M&M: MB001632; and (iii) DEQ Permits / COCs: Benson: MB000720; Goodson 2037/2038: MB000065; Kilpatrick: MB001090; M&M: MB001730 and MB001711. Note that the "MBXXXXXX" document identification is by the first page number of the document.

Facility	Proposed Anaerobic Digester (AD) Size *gallons*	New AD Construction or New anaerobic digester or Lagoon Cover	Proposed number of anaerobic digestate storage lagoons
Benson	1,693,156	New	One single stage, soil improved or clay lined
Goodson 2037/2038	5,044,114	New	One 2-stage, clay lined
Kilpatrick	9,121,716	Cover	One single stage, synthetically lined
Merritt	NA	NA	NA
M&M	2,672,790	New	One single stage synthetic, two single stage clay lined

NA = not applicable. Note that an anaerobic digester will not be installed to serve the Merritt farm, which will continue to be served by the existing single stage lagoon.

B. Reviewing the permit applications and permits issued by the North Carolina Department of Environmental Quality, it is evident that some discrepancies exist. For example, the DEQ issued permit for the Benson Farm indicates that one existing clay-lined lagoon will be used to store anaerobic digester effluent, whereas the permit application submitted by Cavanaugh & Associates on behalf of Smithfield Foods identifies the lagoon having a soil improved liner rather than a clay liner.[19] While investigation of historical satellite imagery catalogued on Google Earth could not reveal such detail, it does reveals an interesting feature of the lagoon, what looks to be a divide running through the center, the reduced volume for which is not reflected in the lagoon design information provided by Smithfield, and dated December 17, 2008.[20] Satellite imagery reported in Figure 1 dating from February 1998 through July 2018 clearly show a divide that is deep enough to affect water quality differences from one side of the lagoon to the other. Other discrepancies between permit applications and actual lagoon liner details for Goodson 2037/2038 and M&M Waters & Riverbank exist, but were corrected in the final permits.

[19] Compare the DEQ issued Permit no. AWI310039 dated March 31, 2021 (MB000720), page 3: "…consisting of a 1,693,156 gallon synthetically lined anaerobic digester with an 80 mil HDPE synthetic cover, one existing clay lined lagoon, one influent pump station…") to the permit application stamped MB000667 and dated 12-19-19, page 2, which identifies the existing lagoon having a soil improved liner.

[20] See the Benson Farm Anaerobic Waste Lagoon Design document dated 12/17/08 (starting on MB000741).

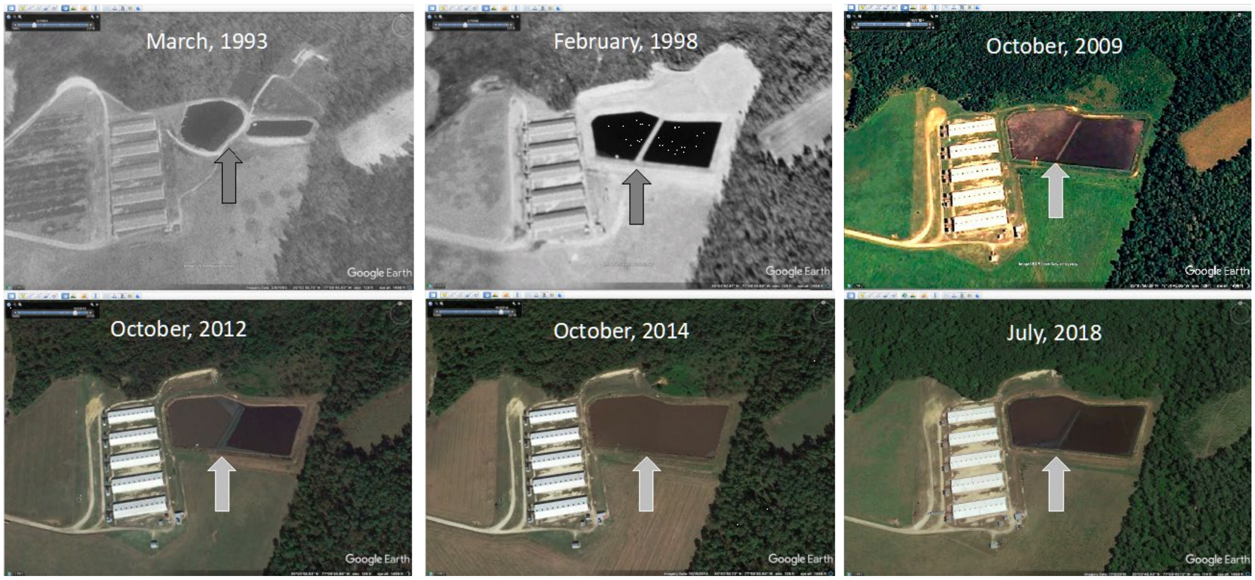

Figure 1. Google Earth historical satellite imagery of the Benson swine CAFO anaerobic lagoon showing a divide in the lagoon center (see February 1998) that roughly correlates to a historic drive (see March 1993), and which is large enough to also be visible and cause water quality differences between the two sides of the lagoon as seen in the subsequent images. This divide is not accounted or in the calculation of the lagoon design volume as reported in in documents provided by Smithfield (see Benson Lagoon Design, dated December 17, 2008 and identified MB000741 on page 1).

C. Permits in North Carolina prohibit siting of the swine CAFO housing units and waste treatment systems in the 100-year flood plain; the same is not true for swine manure application areas. These areas are subject to increased risk of leaching manure pollutants into the waters of North Carolina during and following rainfall. Figure 2 shows satellite imagery of the Goodson 2037 and 2038 swine CAFO and spray fields. As can be seen in the image to the left of this figure, the 100-year flood plain intersects portions of spray fields at this swine CAFO. Historical satellite images suggest that these spray irrigation fields are subject to flooding.

Figure 2. Satellite imagery of the Goodson 2037 / 2038 swine CAFO and manure spray fields. Left: overlay of the 100-year flood plain and the irrigation system map reveals spray irrigation of swine manure occurs in areas subject to flooding, increasing runoff potential of manure pollutants. Right: Historical satellite images from Google Earth reveal recurrent saturation of spray fields within the 100-year flood plain.

D. The permits issued by the North Carolina State Department of Environmental Quality authorize the waste treatment systems of the subject swine CAFOs to change; however, the permits included very few substantive changes to accommodate the new anaerobic digestion waste treatment systems, even though the new technology can alter the waste composition and nutrients as described above. The subject swine CAFOs must continue to monitor their manure wastes as usual and adjust application rates based upon either total nitrogen or total Kjeldahl nitrogen results. Notably, increased ammonia volatilization from lagoon storage of anaerobically digested manure may increase uncontrolled air emissions of ammonia nitrogen, reducing the nitrogen content of the waste to be applied, and allowing for increased concentration of waste application considering that crop nutrient requirements will not change. Owing to changes in the form of the nutrients, increased mobilization may occur. The permits issued for Goodson 2037 & 2038 and Benson swine CAFOs mandate two years of quarterly monitoring and reporting of influent and effluent total nitrogen, total Kjeldahl nitrogen, nitrate nitrogen, ammonium nitrogen, total phosphorus, copper, sulfur, zinc, and fecal coliform bacteria. The nutrient management plans must be modified as needed based on performance of the anaerobic digestion system, and if performance is not as predicted, immediate measures must be taken. Regardless, these measures will only monitor changes in composition at the source. Notably, permits for swine CAFOs in North Carolina, including the four at issue here, do not generally require monitoring of nearby groundwater, surface water, or air emissions to detect potentially increased pollutant mobilization, and thus associated air and water quality degradation by such changes will go undocumented and unaddressed.

E. Other changes in the permits or Certificates of Coverage are not clearly related to changes in the swine manure waste treatment technology. For example, changes in the Certificate of Coverage at the Kilpatrick & Merritt and the permit for Benson swine CAFOs include a directive to review the facilities' CAWMP with respect to land application areas in landscape positions that are in close proximity to public roads, dwellings, and wells, and provide within 180 days a report to the division to describe what, if any, additional BMPs are to be implemented in these areas to improve protections and further reduce risk of off-site impacts. Nothing more is required than a paper exercise. The permit for the Benson swine CAFO also includes a mandate to limit application on fields with a high phosphorus loss assessment rating to crop nutrient requirements for phosphorus, and restricts swine manure application on fields with very high phosphorus loss assessment rating.

4. Biogas-compatible technologies that could mitigate environmental impacts from the subject swine CAFOs by decreasing nutrients in land-applied waste and reducing ammonia emissions are available.

A. Considerable progress has been made in manure treatment technology development in the last 24 years. There are several technologies readily available and in development that can reduce significantly gaseous emissions from anaerobically digested swine CAFO manure. At a basic level, impermeable covers could be installed on the lagoons that will store anaerobic digestate and reduce ammonia volatile losses from the lagoon by nearly 90%.[21] Volatile losses of ammonia can be reduced 70-90% upon application by injection of the anaerobically digested swine manure waste into the ground rather than spray applying. Conservation of the nitrogen in the manure rather than emission into the atmosphere can reduce nearby effects of uncontrolled ammonia deposition as well as public health risks.

B. Other treatment technologies are also available and currently used at other swine CAFO operations that can remove or recover nutrients, significantly reducing environmental and public health risks. One high achieving set of developments stemmed from the Super Soils / Terra Blue technology developed in North Carolina under the Smithfield Agreement.[22] Since that time, advancements from the research group have included development of more efficient two-stage and then one-stage ANAMMOX bioreactors to replace nitrification/denitrification and gas permeable membranes to recover ammonia and phosphate minerals from swine wastewater.[23] An advanced waste treatment technology developed in a joint venture with Embrapa (Brazil) known as Sistrates has successfully implemented the Terra Blue technology with anaerobic digestion at a 9,500 head swine CAFO in Brazil.[24] Swine waste anaerobic digestion-compatible membrane separations technologies have also matured, such as that of Digested Organics LLC which can recover nutrients in higher value streams and simultaneously produce water of sufficient quality for livestock.[25]

5. Conclusion.

A. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that the anaerobic digestion systems to be implemented at the subject swine CAFOs are more likely than not to exacerbate ammonia emissions relative to the current anaerobic lagoon and spray system. Resulting unregulated emissions and

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[21] See Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; Table 11, page 10.

[22] Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, F.J. Humenik (2007) Development of environmentally superior treatment system to replace anaerobic swine lagoons in the USA, *Bioresource Technology*, 98(17):3184-94. doi: 10.1016/j.biortech.2006.07.009.

[23] See Vanotti, M.B., K.S. Ro, A.A. Szogi, J.H. Loughrin, P.D. Millner (2018) High-rate solid-liquid separation coupled with nitrogen and phosphorus treatment of swine manure: Effect on water quality, *Frontiers in Sustainable Food Systems* 2:49. https://doi.org/10.3389/fsufs.2018.00049; see also Vanotti, M.B., P.J. Dube, A.A. Szogi, M.C. Garcia (2017) Recovery of ammonia and phosphate minerals from swine wastewater using gas-permeable membranes. *Water Research*, 112:137-146.; see also Magri, A., M.B. Vanotti, A.A. Szogi (2012) Anammox sludge immobilized in polyvinyl alcohol (PVA) cryogel carriers, *Bioresource Technology*, 114(2):231-240.

[24] See Tápparo, D.C., D. Cândido, R.L. Radis Steinmetz, C. Etzkorn, A. Cestonarodo Amaral, F. Goldschmidt Antes, A. Kunz (2021) Swine manure biogas production improvement using pre-treatment strategies: Lab-scale studies and full-scale application, Bioresource Technology Reports, 15:100716 (8p.)

[25] For more details, see https://digestedorganics.com/manure-and-digestate-management/.

increased nutrient mobilization are more likely than not to negatively affect environmental quality and public health.

B. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that the subject permits authorizing swine CAFO waste management systems known to change the nature of the waste material in ways that may exacerbate water quality impacts from land application, and increase atmospheric ammonia emissions, fail to prevent cumulative effects on water quality and allow adverse environmental impacts.

C. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that technologies to abate these pollution sources are readily available.

D. All of my opinions are expressed to a reasonable degree of scientific certainty.

Exhibit 5

Dr. Shane Rogers
Rogers Environmental Consulting

December 15, 2021

Re: New general permits for biogas projects for confined animal feeding operations in North Carolina

To whom it may concern:

It is my understanding that the North Carolina Department of Environmental Quality is undergoing a process to prepare a new general permit that would authorize biogas projects for confined animal feeding operations, including the large number of industrial hog operations in the state.

The pollution from industrialized hog operations in the state of North Carolina is both an environmental problem and a social justice concern that has garnered significant global exposure for many years. The moratorium on the expansion of new hog operations in the state, particularly those employing lagoon and spray systems, has not resulted in the desired effect of significantly reducing environmental and public health burdens. More than 20 years later, waterways of the state are still polluted by the practices of these operations. Poor and non-white communities remain exposed to pollution and nuisances caused by these operations near their homes.

Anaerobic digestion systems may be a step in the right direction. However, while anaerobic digestion and associated biogas technologies can be designed and operated in such a way as to reduce these burdens,[1] their inappropriate or incomplete application can exacerbate selected environmental problems and significantly reduce their potential to resolve public health threats and nuisance issues.

Solutions exist that should be included in the permit

Undeniably, anaerobic digestion with biogas capture technologies will improve waste stabilization and reduce methane emissions over uncovered anaerobic lagoons. However, continued and new problems associated with biogas projects at confined animal feeding operations will occur if they are designed for maximum monetization of the biogas without appropriate consideration of needed pollution mitigation. For example, installation of a new anaerobic digester or conversion of an existing anaerobic lagoon to a covered lagoon without additional treatment of the waste to reduce pollutants or changes in other practices such as spray application of digested wastes will not resolve uncontrolled atmospheric emissions of ammonia.

The mass of nutrients in manure wastes remains largely unaffected by anaerobic digestion, but the nutrients are altered in form. Improved waste stabilization means greater production of ammonium and ammonia during anaerobic digestion versus anaerobic lagoons. Large amounts of nitrogen from manure wastes emitted through ammonia volatilization to the atmosphere are well documented from uncovered anaerobic lagoons and spray application of waste. Once in the

[1] For example, covered anaerobic digestate storage and proven technological solutions that work in concert with, or as an alternative to, biogas systems can be implemented that reduce meaningfully their air emissions, improve the conservation of resources, and reduce their potential to pollute the environment and cause nuisance issues. Working examples exist at large-scale confinement operations globally.

atmosphere, ammonia can react to form fine particulates that threaten air quality, public health, and the environment. Much of this ammonia deposits uncontrolled within a small radius of the farm where it can contribute to water quality problems as well.

Because nutrient mass is largely unaffected by anaerobic digestion, simply installing anaerobic digestors cannot resolve nutrient loading issues from hog manure application that affects North Carolina waters. Indeed, the conversion of nitrogen to more mobile forms may exacerbate water quality issues upon application.

Improved nutrient conservation practices and established nutrient removal and nutrient recovery technologies congruent with anaerobic digestion technology presently exist and can mitigate these issues. There exists a suite of anaerobic digestion-compatible technologies available for nutrient removal or recovery. Many of these well-understood and well-practiced technologies are already in use for treatment of livestock manures globally. For example:

- **Nitrification - denitrification technology** has been used in wastewater treatment for decades; working examples exist at hog operations in the United States, including at Smithfield/Murphy-Brown hog operations producing biogas in Missouri. The first step of this biological process converts ammonium nitrogen in waste to nitrate under aerobic conditions. In the second step, denitrifying bacteria respire nitrate to support maintenance energy production under anoxic conditions. In this process, they convert the nitrate into harmless dinitrogen (N_2) gas. The resulting treated waste has greatly reduced nitrogen content and improved characteristics that are less offensive than anaerobic digestate. Land application can then occur at greater application rates, lowering land area requirements.[2] Dinitrogen gas is odorless and comprises 78% of our atmosphere; it poses no threat to public health, to the environment, or to nuisance conditions on the part of neighbors.

- **Anaerobic ammonium oxidation** (ANAMMOX), a biological process that converts ammonium to dinitrogen gas in an anaerobic environment, has been implemented successfully with many anaerobic digesters globally. While limited anaerobic ammonium oxidation may occur naturally in anaerobic digesters or anaerobic lagoons for livestock wastes, engineered processes have been designed to operationalize this process in a meaningful way towards nitrogen removal from wastes with lower costs and space requirements than nitrification – denitrification technologies.

- **Alkaline precipitation** is a widely used chemical process in which hydrated lime is added to wastewater facilitating precipitation of calcium phosphate as solids that can be recovered for beneficial reuse. The technology has been used widely for wastewater treatment, and has been proven in full-scale hog waste treatment systems as described below. Solids in the wastewater (including pathogenic microorganisms) act as nucleation sites upon which precipitation can initiate. The combined effects of entrapment in solid precipitates and

[2] This technology can thus allow operations that would otherwise have to reduce stocking density by ammonia conservation practices such as covered storage for anaerobic digester effluent owing limited land available for manure application at nitrogen limited rates to continue to operate with the same livestock numbers. Added waste neutralization afforded by the technology such as lowered odor reduces its potential to cause nuisance conditions as well.

increased pH upon alkaline stabilization leads to significant decimation of pathogenic microorganisms as a co-benefit to application of this phosphorus removal technology.

- **Struvite precipitation** is a chemical treatment technology used to recover phosphorus from wastewater at several municipal wastewater treatment plants in saleable form. This technology has also been used safely at concentrated animal feeding operations. Struvite is a phosphorus-containing mineral that works as a slow-release fertilizer when used to fertilize land. It naturally precipitates in many anaerobic digesters, thus can be a nuisance to their operation. However, controlled struvite precipitation in concert with anaerobic digestion can reduce its potential to cause a nuisance in the anaerobic digestion process while producing a valuable phosphorus fertilizer.

- **Super Soils / Terra Blue**: Several of the above processes have been employed in conjunction with one another to treat industrialized livestock agriculture wastes to a high standard. Indeed, many high achieving developments stemmed from the Super Soils / Terra Blue technology first developed and ground-truthed over a decade ago in North Carolina.[3] This advanced technology was capable of achieving 97.6% removal of suspended solids, 99.7% removal of biochemical oxygen demand (BOD), 98.5% removal of total Kjeldahl nitrogen (TKN), 98.7% of soluble ammonia, 95% of total phosphorus, 98.7% of copper, and 99% of zinc. It also removed 97.9% of odor compounds in the liquid effluent and reduced pathogen indicator bacteria to non-detectable levels. The produced water was of sufficient quality for reuse to clean swine houses and for safe reuse for crop irrigation. Since that time, advancements from those developers have included ANAMMOX bioreactors such as those described in the prior paragraph for treatment of livestock wastes, and gas permeable membranes to recover ammonia and phosphate minerals from swine wastewater, such as those discussed below.[4]

- **Sistrates (Swine Effluent Treatment System)** is an advanced waste treatment system developed in a joint venture between Embrapa (Brazil) and the developers of the Terra Blue technology to successfully implement it with anaerobic digestion technology at a 9,500 head swine CAFO in Brazil. [5] This modular technology first removes solids from swine barn flush system water by a sieve and rotating brush. Separated solids are anaerobically digested in a high-rate digester, while liquids are treated in a covered anaerobic lagoon (ambient temperature digester). Anaerobic digestate is treated to remove

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3 Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, F.J. Humenik (2007) Development of environmentally superior treatment system to replace anaerobic swine lagoons in the USA, *Bioresource Technology*, 98(17):3184-94. doi: 10.1016/j.biortech.2006.07.009.

4 See Vanotti, M.B., K.S. Ro, A.A. Szogi, J.H. Loughrin, P.D. Millner (2018) High-rate solid-liquid separation coupled with nitrogen and phosphorus treatment of swine manure: Effect on water quality, *Frontiers in Sustainable Food Systems* 2:49. https://doi.org/10.3389/fsufs.2018.00049; see also Vanotti, M.B., P.J. Dube, A.A. Szogi, M.C. Garcia (2017) Recovery of ammonia and phosphate minerals from swine wastewater using gas-permeable membranes. *Water Research*, 112:137-146.; see also Magri, A., M.B. Vanotti, A.A. Szogi (2012) Anammox sludge immobilized in polyvinyl alcohol (PVA) cryogel carriers, *Bioresource Technology*, 114(2):231-240.

5 See Tápparo, D.C., D. Cândido, R.L. Radis Steinmetz, C. Etzkorn, A. Cestonarodo Amaral, F. Goldschmidt Antes, A. Kunz (2021) Swine manure biogas production improvement using pre-treatment strategies: Lab-scale studies and full-scale application, Bioresource Technology Reports, 15:100716 (8p.)

ammoniacal nitrogen through the process of nitrification-denitrification described above. Phosphorus is recovered for beneficial reuse from the waste as calcium phosphate by alkaline precipitation, as designed for the Super Soils / Terra Blue technology. Produced wastewater has greatly reduced nitrogen content, improving the ability of operators to custom blend recovered phosphorus with produced water and chemical nitrogen fertilizer to meet specifically crop nutrient requirements.

- Anaerobic digestion-compatible **membrane separation and solids recovery technologies** have also matured, such as that of Digested Organics LLC. This system can recover nutrients in higher value streams and simultaneously produce water of sufficient quality for reuse for livestock[6]. Briefly, when used in conjunction with anaerobic digestion, their modular system first uses a spiral brush screen filter recover large solids from anaerobic digestate prior to down-stream pressure-driven membrane processes. Phosphorus recovered with the dewatered solids amounts to approximately 15-30% that in the anaerobic digester effluent. In dairy manure digester applications, the recovered dewatered fibrous solids are of sufficient quality for reuse as bedding compost. Suspended solids removal by the spiral brush screen facilitates treatment of the permeate by stainless steel ultrafiltration. The retentate (concentrate) from ultrafiltration is a useful nitrogen and phosphorus rich fertilizer. It is much reduced in volume, thus facilitating smaller covered storage than otherwise would be needed to limit fugitive emissions. These first two steps of the process remove greater than 99.9% of the suspended solids and pathogens, greater than 95% of total phosphorus, and 30-40% of total nitrogen (predominantly organic nitrogen) from the waste stream. The high quality permeate (product water) stream produced from the ultrafilters are then passed through a two-step reverse osmosis system to recover much of the remaining total nitrogen and remove color, dissolved solids, and metals from the waste. The nitrogen-rich retentate from the two-step reverse osmosis process can be custom blended with the phosphorus-rich retentate from the stainless-steel ultrafiltration units to meet specifically the unique fertilization requirements of different crops, thus reducing risks of overapplication of phosphorus when meeting crop nitrogen requirements. The product water of the two-step reverse osmosis system is of high quality. Greater than 99% total solids, ammoniacal nitrogen, phosphorus, potassium, and several metals, 92% of TKN, and 86% of organic nitrogen can be removed from anaerobic digester effluent by these three combined processes. When product water is treated with their ultraviolet disinfection system, it is of sufficient quality for reuse as livestock drinking water or can be discharged safely to a waterway.

While the above focus on technological solutions for waste treatment and resource recovery, it is important to note that additional options exist to reduce environmental, public health, and nuisance issues while conserving valuable nutrients in hog manure wastes.

- **Nutrient conservation:** Most simply, mandating that secondary lagoons used for anerobic digestate storage be covered with impermeable covers to capture fugitive methane and other volatile emissions[7] and limiting application practices that encourage volatile

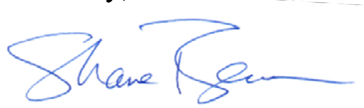
[6] For more details, see https://digestedorganics.com/manure-and-digestate-management/

[7] Impermeable covers on the lagoons used to store anaerobic digestate can reduce ammonia volatile losses considerably. See: Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020)

pollutant losses (e.g., spray gun application) in lieu of low height spreading or injection[8] can help hog operations with methane capture to achieve its greater potential for pollution reduction. The nutrient conservation (lowered ammonia volatile losses) will lead to greater fertilization potential and lower manure application rates. This will reduce co-pollutant loading rates to land that can runoff and emissions of nuisance odors that can improve conditions for neighbors of industrial hog operations. Nutrient conservation through covered anaerobic digestate storage will benefit those operations that presently supplement their crop fertilization requirements with chemical fertilizers, reducing their reliance upon them. Further, covered storage will improve timing and frequency of application, being less subject to wet weather conditions and risks of freeboard violations that plague uncovered lagoons in North Carolina -- especially during the hurricane the season.

Considering the above, there are myriad options to facilitate the implementation of biogas systems at concentrated animal feeding operations in North Carolina in ways that will reduce, rather than exacerbate, the pollution problems associated with these projects. New general permits that include rules to assure that technology interventions are fully reflective of modern practices and capabilities globally, and address meaningfully the environmental emissions, public health concerns, and nuisance issues that surround pollution emanating from CAFO operations, will better protect North Carolina's waterways and communities.

Sincerely,



Dr. Shane Rogers
Rogers Environmental Consulting

Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; Table 11, page 10.

8 Volatile losses of ammonia can be reduced 70-90% upon application by injection of the anaerobically digested swine manure waste into the ground rather than spray applying. Conservation of the nitrogen in the manure rather than emission into the atmosphere can reduce nearby effects of uncontrolled ammonia deposition as well as public health risks.

ATTACHMENT 2

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

EXTERNAL CIVIL RIGHT COMPLIANCE OFFICE
OFFICE OF GENERAL COUNSEL

January 12, 2017

Return Receipt Requested
Certified Mail# 7015 3010 0001 1267 5133

In Reply Refer to:
EPA File No. 11R-14-R4

William G. Ross, Jr.
Acting Secretary
North Carolina Department of Environmental Quality
1601 Mail Service Center
Raleigh, NC 27699-1611

Re: Letter of Concern

Dear Acting Secretary Ross:

We are writing to you to provide the North Carolina Department of Environmental Quality (NC DEQ) preliminary information related to the United States Environmental Protection Agency's (EPA) External Civil Rights Compliance Office's[1] (ECRCO) investigation into alleged discriminatory impacts from NC DEQ's operation of the Swine Waste Management System General Permit (Swine Waste General Permit). ECRCO has not concluded its investigation of EPA File No. 11R-14-R4 (Complaint) or reached final conclusions of fact or law. However, in light of the preliminary information gathered, ECRCO has deep concern about the possibility that African Americans, Latinos, and Native Americans have been subjected to discrimination as the result of NC DEQ's operation of the Swine Waste General Permit program, including the 2014 renewal of the Swine Waste General Permit.

EPA recognizes that there is new leadership at NC DEQ who were not involved in the events and correspondence described below relating to this Complaint and who will understandably need to come up to speed. ECRCO looks forward to sitting down with NC DEQ's new leadership in the next few weeks to provide any necessary background on NC DEQ's obligations under the federal nondiscrimination statutes and to discuss issues raised by ECRCO's investigation to date; any additional information NC DEQ may have relevant to the issues under investigation; the recommendations ECRCO has made below; and how to move forward on a constructive path to informally resolve the Complaint in the near future and ensure NC DEQ is in compliance with the applicable nondiscrimination statutes and regulations.

[1] Formerly the Office of Civil Rights. To eliminate confusion, except where quoting another document, this letter will use the Office's current name, rather than its name at the time of any particular action or correspondence.

Procedural Background of Complaint

On September 3, 2014, Earthjustice filed a complaint under Title VI of the Civil Rights Act of 1964, as amended, 42 U.S.C. §§ 2000d et seq., and the EPA's nondiscrimination regulations found at 40 C.F.R. Part 7, on behalf of the North Carolina Environmental Justice Network, Rural Empowerment Association for Community Help (REACH), and the Waterkeeper Alliance, Inc. alleging discrimination based on race and national origin by NC DEQ. The complaint alleged that NC DEQ's 2014 renewal of the Swine Waste General Permit without adequate measures to control, dispose of, and monitor animal waste from industrial swine feeding operations subjects African Americans, Latinos, and Native Americans to discriminatory impacts (*e.g.*, health issues, noxious odors, nuisances, increased expenses, social and psychological harms, declining property values).

On February 20, 2015, EPA opened an investigation into:

> Whether the North Carolina Department of Environmental Quality's (NC DEQ) regulation of swine feeding operations discriminates against African Americans, Latinos, and Native Americans on the basis of race and national origin in neighboring communities and violates Title VI of the Civil Rights Act of 1964 and the Environmental Protection Agency's implementing regulation.

On March 6, 2015, the Complainants and NC DEQ entered into an Alternative Dispute Resolution (ADR) process funded by EPA. ECRCO placed the investigation on hold pending the outcome of the ADR process. On March 7, 2016, the Complainants informed ECRCO that they were withdrawing from the ADR process.

On May 5, 2016, ECRCO informed NC DEQ that the ADR process between NC DEQ and the Complainants concluded without resolution; therefore, consistent with ECRCO procedures, ECRCO's investigation was reinitiated. On July 11, 2016, the Complainants filed an additional complaint alleging NC DEQ violated EPA's regulation prohibiting retaliation, intimidation, and harassment of Complainants (40 C.F.R. § 7.100). Among other events, Complainants point to events involving the Pork Councils' attempt to intervene at the January 2016 ADR session.

On August 1, 2016, NC DEQ submitted a response to the retaliation allegations. On August 2, 2016, ECRCO informed NC DEQ that it will also investigate:

> Whether NCDEQ's actions or inactions, including those associated with the presence and activities of the Pork Councils related to the January 2016 mediation session, violated 40 C.F.R § 7.100 which prohibits intimidating, threatening, coercing, or engaging in other discriminatory conduct against any individual or group because of actions taken and/or participation in an action to secure rights protected by the non-discrimination statutes OCR enforces.

On September 2, 2016, NC DEQ requested that ECRCO dismiss the original complaint. After reviewing the information provided by NC DEQ, ECRCO notified NC DEQ that the Complaint would not be dismissed. On December 5, 2016, NC DEQ submitted a response to the Complaint.

ECRCO's investigation is being conducted under the authority of Title VI of the Civil Act of 1964, and EPA's nondiscrimination regulations (40 C.F.R. Part 7), and consistent with ECRCO's Interim Case Resolution Manual (https://www.epa.gov/ocr/case-resolution-manual) Title VI provides that "[n]o person in the United States shall, on the ground of race, color, or national origin, be excluded from participation in, be denied the benefits of, or be subjected to discrimination under any program or activity receiving Federal financial assistance." 42 U.S.C. § 2000d. As implemented by EPA's regulation, these prohibitions include intentional discrimination as well as practices that have a discriminatory effect on the bases of race, color, or national origin. See 40 C.F.R. §§7.35(a), 7.35(b). The EPA regulation at §7.35 (b) prohibits a recipient from using criteria or methods of administering its program or activity which have the effect of subjecting individuals to discrimination. The EPA regulation also prohibits intimidation and retaliation against any individual or group for the purpose of interfering with a right protected by Title VI or because the individual has "filed a complaint, or has testified, assisted or participated in any way in an investigation, proceeding, or hearing" under EPA's regulation or has opposed any practice prohibited by the regulation. See 40 C.F.R. § 7.100.

ECRCO's investigation thus far has included an on-site visit to interview residents; reviewing information submitted by the Complainants including declarations prepared by residents and other witnesses; reviewing scientific and other literature and interviewing the authors when appropriate, and, a review of NC DEQ's responses to the Complaint dated August 1, 2016, September 2, 2016, and December 5, 2016. NC DEQ's December 5, 2016, letter requested that ECRCO "...provide any relevant information in its possession on the issue of discrimination by the State's regulation of Swine feeding operations." Below is a summary of information gathered thus far through ECRCO's investigation.

Adverse Impacts from Industrial Swine Operations on Communities of Color

On-Site Interviews and Declarations

ECRCO conducted an on-site visit to North Carolina (November 13-15, 2016) and interviewed over 60 residents living near industrial swine operations permitted under the Swine Waste General Permit. ECRCO's interviews were conducted mostly in Duplin and Sampson counties which have the highest concentration of industrial swine operations. To investigate the effects of the permitting program more broadly throughout the state, ECRCO also conducted interviews in other counties including Northampton County on the Virginia border and Pender County near the South Carolina border.

Some of the people interviewed had previously submitted declarations to ECRCO as exhibits to the Title VI complaint, some had not. The issues raised in the declarations and the impacts they discussed are similar or identical to those heard during the interviews. Many of those interviewed who had previously provided declarations provided updates to their declarations. ECRCO found credible all those interviewed thus far in the investigation. So far, ECRCO has heard in writing and/or orally from 85 witnesses in North Carolina who live near and described problems caused by their proximity to the industrial hog operations.

Residents, many of whom have lived in these communities for generations, described problems caused by their proximity to the industrial hog operations that have negatively changed their lives and communities, including those impacts described in studies referenced or discussed below. The residents described an overpowering stench, pests -- including a constant large number of flies, and the truck traffic all associated with the hog operations have forced residents to keep doors and windows closed and significantly limit any outdoor activity. Residents said the stench permeates homes, cars, and clothing. Some residents said the strength of the odor can be so strong it causes gagging, nausea and/or vomiting. For some residents who live near large numbers of industrial swine operations, they said stench is a weekly event lasting several days. They also stated they have no warning of when confinement house fans, spraying of the hog waste, or trucks transporting live or dead hogs will again bring the stench and actual waste onto their homes, property or themselves. Some described feeling as though they are prisoners in their own homes.

Residents described the loss of community that has occurred since the industrial hog farms began operating. They reported that young adults leave and do not return because of the odors, fear of health impacts from the air and drinking water, and other impacts. Prior to the arrival of the industrial hog operations, many of their family, community, and church gatherings had been held outdoors. Now they said those events are rarely held outdoors or if attempted outdoors, they are marred or forced to end early due to odors, flies, and other impacts.

Residents described increases in cases and severity of asthma and other respiratory illnesses, nausea, headaches and other health conditions. They stated these impacts have been compounded by the increase in industrial poultry operations, as well as the operation of landfills and waste disposal sites for hog sludge and carcasses. Those who had hunted and fished for both food and enjoyment, said they no longer do so because of the odors and fear of the contamination of wild sources of food such as fish. Residents stated they no longer keep gardens or grow their own vegetables for fear of contamination. Some residents are still on well water and are concerned about the safety of their drinking water. Some residents would prefer to use their private wells rather than public drinking water, but said they have either been told not to drink it or are afraid that it is contaminated. Residents also discussed increased expenses from buying and using public water, bottled water, clothes dryers, air fresheners, pesticides, air conditioning units, and food.

When asked whether they had filed complaints with NC DEQ or local governments about the odors, pests, and waste sprayed on them or their property, some residents said they did not know how or where to file complaints. ECRCO was told the filing of complaints with NC DEQ would be pointless and has resulted in retaliation, threats, intimidation, and harassment by swine facility operators and pork industry representatives. Several residents said that for more than 15 years, the government has been well aware of the conditions they have to live with, but has done nothing to help, so complaining to NC DEQ would be futile.

ECRCO also interviewed residents who live near an industrial swine operation that began operation using the lagoon and spray field method under a Certificate of Coverage under the Swine Waste General Permit. When discussing the impacts that occurred while the facility operated under the General Permit, the residents described the same impacts as those currently living near facilities operating under the General Permit, including nausea; headaches; odors that permeated their homes and prevented them from enjoying their yards and the outdoors; concerns about impacts to groundwater and surface water; and increased numbers of flies and other pests.

After several years of operation, the operator installed innovative technologies and practices to reduce the odor and other impacts from his operation including covering the waste lagoons; not spraying in the evenings and on weekends; and not using dead boxes. When asked to describe the current impacts from that industrial swine operation, for the most part the residents who live nearby said they rarely notice intense odors and that the number of flies has been greatly reduced. The exception was that one bordering neighbor said that the smell was unbearable during spraying. Other neighbors pointed out that the smell could also be from other industrial swine operations near that resident's property that still employ open lagoons and spray fields. ECRCO and other EPA staff who toured the operation including the confinement houses and the edge of the lagoons were surprised at how little odor there was given the number of swine housed there at that time and the presence of more than one waste lagoon.

Other Information on Adverse Impacts of Industrial Swine Operations on Nearby Residents

EPA recognizes that industrial hog operations have a negative impact on nearby residents, particularly with respect to objectionable odors and other nuisance problems that can affect their quality of life. (EPA, *Animal Feeding Operations Consent Agreement and Final Order*, 70 FR 4957, 4959 (Jan. 31, 2005)). The adverse impacts of offensive odors from North Carolina's industrial hog operations have been a known issue for more than 20 years. In 1994, the North Carolina legislature established a Blue Ribbon Panel, the Swine Odors Task Force, to study "the problem of swine odors and how to reduce them."[2] The report of the Swine Odors Task Force stated protests had been "numerous and well publicized."[3]

In part to protect North Carolina's travel and tourism industry and allow time for the completion of the studies of odor and other problems associated with swine operations, the state legislature implemented a moratorium effective March 1, 1997, on the construction or expansion of swine operations[4] that use "an anaerobic lagoon as the primary method of treatment and land application of waste by means of a spray field as the primary method of waste disposal."[5] Any new or expanding swine operations were required to eliminate or reduce a number of impacts from the lagoon and spray field methods including <u>substantially eliminating</u> ammonia emissions and odors detectable beyond the boundaries of the swine operation.[6] The moratorium was made permanent in 2007. However, the industrial hog operations using the lagoon spray field configuration already operating at the time of the moratorium were allowed to continue to operate under the Swine Waste General Permit without the requirements to <u>substantially eliminate</u> ammonia emissions and odors. Today, in a handful of counties mostly in eastern North

[2] Dr. Johnny C. Wynne, et al., *Options for Managing Odor... a report from the Swine Odor Task Force (March 1, 1995)*, NORTH CAROLINA AGRICULTURAL RESEARCH SERVICE, N.C. STATE UNIV., *available at* http://www.mtcnet.net/~jdhogg/ozone/odor/swineodr.html#notsimple

[3] *Id.*

[4] S.L. 1997-458, H.B. 515, § 1.1(a).

[5] G.S. § 143-215.10I.

[6] G.S. § 143-215.10I (b)(2)-(3).

Carolina there is a total of more than 9 million hogs allowed in the more than 2000 industrial hog facilities operating under the Swine Waste General Permit.

North Carolina established a rule specifically to control objectionable odors from industrial swine operations.[7] "Objectionable odor" means any odor present in the ambient air that by itself, or in combination with other odors, is or may be harmful or injurious to human health or welfare, or may unreasonably interfere with the comfortable use and enjoyment of life or property. Odors are harmful or injurious to human health if they tend to lessen human food and water intake, interfere with sleep, upset appetite, produce irritation of the upper respiratory tract, or cause symptoms of nausea, or if their chemical or physical nature is, or may be, detrimental or dangerous to human health."[8] North Carolina's definition of "objectionable odor" encompasses the panoply of negative effects experienced by North Carolina residents, as told to ECRCO and discussed above.

Review of Reports and Studies

The adverse impacts on nearby residents from the lagoon spray field method of treatment and disposal of waste from industrial swine operations are documented in numerous peer reviewed scientific studies, including more than thirty conducted in North Carolina.[9] At ECRCO's request, EPA's Office of Research and Development (ORD) recently reviewed seven reports published by or with federal agencies.[10] ORD stated that the reports provide consistent support for the occurrence of potential health hazards (*e.g.*, eye, nose, and throat irritation; headaches; respiratory effects including asthma exacerbation; waterborne disease) at industrial swine operations and in their waste. Even while there is significant uncertainty regarding the levels of exposure in nearby communities to the identified contaminants and the risk of health effects attributable to those exposures, the risk for specific health effects in communities near industrial swine operations is a concern.

North Carolina's 1994 Swine Odors Task Force stated "it is not surprising to learn that living near a swine operation can affect mental health" when discussing a Duke University study of "the moods of people exposed to odors from commercial swine operations in North Carolina. Forty-four neighbors of hog operations . . . had less vigor and were significantly more tense, depressed, angry, fatigued, and confused."[11]

Additionally, ECRCO considered the findings of the analysis prepared by Drs. Steve Wing and Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians*, (revised October 19, 2015) (*Complainants' Disproportionate Impact Analysis*). While the *Complainants' Disproportionate Impact Analysis* has not undergone peer review, it uses a study protocol and methodology that are substantially similar to peer reviewed studies by Wing et al. and Johnston et al.[12] The

[7] 15A NCAC 02D.1802(a).
[8] 15A NCAC 02D.1801(9).
[9] *See* Attachment A. Additional studies can be made available.
[10] *See* Attachment B.
[11] *See* Wynne, et al., *supra* note 3.
[12] *See* Steve Wing, et al., *Environmental Injustice in North Carolina's Hog Industry* (Mar. 2000), ENVIRONMENTAL HEALTH PERSPECTIVES, https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1637958/; Jill E. Johnston, et al.,

Complainants' Disproportionate Impact Analysis concludes that the impacts from the manner in which waste is disposed of, and other impacts tied to industrial swine facilities operating under the Swine Waste General Permit, detrimentally affect those who live in neighboring properties and communities.

The *Complainants' Disproportionate Impact Analysis* also concluded, when examining those neighboring properties that African-Americans, Hispanics, and Native Americans are more likely than Whites to live near industrial swine operations granted COCs. Specifically, the *Complainants' Disproportionate Impact Analysis* concluded that, both state-wide, and in only-rural areas, African-Americans, Hispanics, and Native Americans living in North Carolina are more likely than Whites to live within 3 miles of an industrial swine operations granted COCs, and therefore suffer those detrimental effects. The *Complainants' Disproportionate Impact Analysis* looked at the steady state live weight (SSLW) of hogs within 3 miles of the center of census blocks. The SSLW was used as an indicator of density of hogs/the amount of swine feces and urine produced by the hogs in that 3-mile area. The *Complainants' Disproportionate Impact Analysis* found that for each 10 percent increase in the combined African-American, Hispanic, and Native American population and for each 10 percent increase in population for each of those census categories individually, the SSLW of hogs within 3 miles increases by anywhere from 47,000 to 165,000 pounds. This analysis concludes that there is a linear relationship between race/ethnicity and the SSLW or density of hogs.

Intimidation/Retaliation

The Complainants raised allegations of intimidation and harassment in their written submissions to ECRCO and during interviews related to the appearance of and actions by national and local representatives of the pork industry at what was to be a confidential ECRCO-sponsored alternative dispute resolution mediation session between NC DEQ and the Complainants. Complainants claim that, although NC DEQ representatives knew that complainants did not want representatives from the National Pork Producers Council and North Carolina Pork Council (Pork Councils) at this confidential meeting, NC DEQ representatives appeared to encourage their attendance and participation.

NC DEQ's responses in its letters dated August 1, 2016, and December 5, 2016, in part question whether complainants felt intimated by the Pork Council representatives' presence at the January 2016 mediation session. NC DEQ stated in both letters that "it strains credulity that these individuals were intimidated by the fact that they would be identified by representatives of organizations whom these individuals routinely criticized at public forums."[13] The following information provided to ECRCO may assist NC DEQ to better understand in part the context within which the Complainants have raised concerns about harassment, retaliation, and intimidation.

Wastewater Disposal Wells, Fracking, and Environmental Injustice in Southern Texas (Mar. 2016), AMERICAN JOURNAL OF PUBLIC HEALTH (2016), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4816143/.

[13] Letter to Lilian S. Dorka, Acting Director, EPA Office of Civil Rights from Sam M. Hayes, General Counsel, North Carolina Department of Environmental Quality (Dec. 5, 2016), at 8-9.

During interviews, residents including REACH members, and current and former Riverkeepers working in the eastern North Carolina rivers recounted first hand incidents of harassment, intimidation, and retaliatory behavior, including physical and verbal threats, by swine facility owners and/or operators and their employees. The accounts ranged from sustained tailgating; driving back and forth in front of the houses of residents who have complained; filming or photographing residents who are taking photos or videos of spraying; being yelled at; confronted in parking lots and at intersections; and threatened with guns and other physical violence.

Those interviewed stated that these are regular events, rather than an exception, creating a climate where residents believe that if they file an environmental complaint with NC DEQ, they will likely be retaliated against by neighboring swine facility operators or employees. The Riverkeepers stated that they are subjected to this type of harassment and intimidation two or three times every couple of weeks. Particularly egregious instances brought to ECRCO's attention include a local industrial swine facility operator entering the home of an elderly African American woman and shaking the chair she sat in while threatening her and her family with physical violence if they continued to complain about the odors and spray; the firing of a gun in the air when an African American REACH member tried to speak to a person sitting on their porch; and a truck that sped up and swerved toward a Riverkeeper who was standing on the side of a public road teaching a group of volunteers how to sample water from public ditches. Those interviewed believe that the NC DEQ's lack of response to their complaints lends to the hostile environment and emboldens local facility owners and operators to act in a threatening and intimidating manner.

ECRCO has grave concerns about these reports indicating a potential hostile and intimidating environment for anyone seeking to provide relevant information to NC DEQ or EPA. Also, ECRCO is concerned about the circumstances surrounding the attendance by pork industry representatives during the mediation session.

Under certain circumstances, Title VI's prohibition on retaliation extends to third parties, which may include lower-level recipient employees, program beneficiaries or participants, organizations with a relationship to the recipient such as contractors, and others. EPA Title VI regulations provide that "[n]o recipient or *other* person" may retaliate. 7 C.F.R. § 7.100. Recipients themselves have two key obligations related to third party retaliation: first, to protect individuals from potential retaliation, recipients are obligated to keep the identity of complainants confidential except to the extent necessary to carry out the purposes of the Title VI regulations, including conducting investigations, hearings, or judicial proceedings; and second, recipients must investigate and respond when a third party engages in retaliatory conduct that Title VI prohibits. As with other types of third party conduct, such as harassment, the extent of the recipient's obligation is tied to the level of control it has over the bad actor and the environment in which the bad acts occurred. *See Davis v. Monroe Cty. Bd. of Educ.*, 526 U.S. 629, 644 (1999). EPA makes these determinations on a case-by-case basis in light of the facts and totality of circumstances in a particular case.

NC DEQ's Response to the Complaint[14]

In part, NC DEQ's December 5, 2016 letter responding to the Complaint reiterated arguments in favor of dismissal previously submitted to ECRCO by letter dated September 2, 2016. ECRCO has considered the information in both letters concerning the alleged reduction of adverse impacts by the permit renewal. As ECRCO pointed out previously, NC DEQ itself explained that the majority of the changes from the 2009 permit to the "current permit are structural and grammatical in nature"[15] and "do not make the Permit more stringent, costly or burdensome."[16] NC DEQ's responses did not state or explain how the 2014 Permit will reduce adverse impact from the source, significantly or otherwise; therefore, as stated in ECRCO's letter dated October 5, 2016, ECRCO does not believe that the argument constitutes a proper basis for dismissal of the complaint.

Similarly, NC DEQ's December 5, 2016 letter raised again a concern that the Complainants did not pursue an administrative appeal of the Swine Waste General Permit. NC DEQ explained that the permit appeal process under state law is an appropriate forum for "those who believe that the terms of the General Permit failed to comply with state law, or if a more effective means of pollution control should have been incorporated into the General Permit."[17] The allegation raised to EPA by the Complainants is that the Swine Waste General Permit fails to comply with federal law, namely Title VI of the Civil Rights Act of 1964. For almost 30 years, recipients of EPA financial assistance have been required under EPA's Title VI implementing regulation to have in place a grievance process. 40 C.F. R. § 7.90. The NC DEQ administrative forum to investigate and resolve exactly the issues of discrimination alleged in the Complaint that should have been available to Complainants did not exist when they filed their Complaint with EPA. Regardless, as NC DEQ previously acknowledged in its October 5, 2016 letter, there is no requirement under Title VI that Complainants exhaust administrative remedies before filing a discrimination complaint with EPA.

NC DEQ's response did not deny or refute the allegation that the industrial hog facilities operating under the Swine Waste General Permit were creating discriminatory impacts, rather, NC DEQ points to siting decisions by operators and changing demographics as reasons why certain communities may be more impacted than others. The impacts of concern in this investigation flow from the operation of the facilities. While an industrial swine facility operator may apply for an individual permit or certificate of coverage to operate in a particular location, it is NC DEQ that determines whether that facility will be allowed to operate and under what type of permit and its conditions.

NC DEQ also pointed out that the population has grown and the demographics have changed in areas of high concentration of industrial swine facilities since the first Swine Waste General

[14] ECRCO is not specifically addressing all of the points NC DEQ raised in its December 5, 2016 letter, but this should not be interpreted as ECRCO accepting those arguments.
[15]State of North Carolina, Department of Environmental and Natural Resources, *Report of the Proceedings on the Proposed Renewal of the State General Permits for Animal Feeding Operations, Public Meeting, November 12, 2013, Statesville, North Carolina, Public Meeting, November 14, 2013, Kenansville, North Carolina,*" p. 4.
[16] *Id.*, at pp. 5 and 8.
[17] Letter to Lilian S. Dorka, Acting Director, EPA Office of Civil Rights from Sam M. Hayes, General Counsel, North Carolina Department of Environmental Quality (Dec. 5, 2016), at 2.

Permit was issued. NC DEQ discussed the population growth in Duplin and Sampson Counties, highlighting in particular the rapid growth of the Latino population, and speculated that the population growth may have been due to jobs created by the industrial hog industry. The reasons for an increase in the minority population in the past 20 plus years in Duplin and Sampson Counties does not change NC DEQ's obligation to ensure that its current programs and activities do not have the effect of subjecting individuals to discrimination based on race, color or national origin.

NC DEQ requested information ECRCO has on anecdotal as well as systemic concerns relative to the Swine Waste General Permit Program. With regard to concerns about individual facilities, based on interviews of Riverkeepers working in eastern North Carolina, it is our understanding that for more than a decade they have provided NC DEQ documentation of hundreds of instances of waste spray or drift entering play areas; landing on people in their gardens, on their cars, and on their houses; runoff of hog waste entering streams and ditches; and improper spraying of waste after issuance of a National Weather Service Flood Watch. Riverkeepers stated they have provided NC DEQ the information through eyewitness accounts, photographs with time, date, and GPS coordinates embedded in the metadata, and/or video. Some of this information has been shared with ECRCO as well. Witnesses have stated that, to their knowledge, very few of these reports have received any mitigating action or resulted in enforcement action by NC DEQ. The temporal and geographic breadth of the anecdotal instances documented by Riverkeepers, points to systemic issues about which NC DEQ is aware.

Nondiscrimination Procedural Safeguards

At the time the Complaint was filed, NC DEQ was not in compliance with EPA's longstanding requirements under 40 C.F.R. Part 7, Subpart D which form the foundational elements of a recipient's program to implement the federal non-discrimination statutes.[18] These regulatory requirements include a continuing notice of non-discrimination under 40 C.F.R. § 7.95, grievance procedures available to the public, and the designation of at least one person to coordinate its efforts to comply with its non-discrimination obligations under 40 C.F.R. § 7.85(g).

At some point during the summer of 2016, NC DEQ appears to have begun the process of establishing its non-discrimination program. However, it is unclear whether NC DEQ has put in place the foundational elements of a properly functioning nondiscrimination program. ECRCO has attached a Procedural Safeguards Checklist (Attachment C) to assist NC DEQ in evaluating whether it has in place the appropriate foundational elements to ensure that it will meet its obligations under the federal nondiscrimination statutes.

Mitigation

As NC DEQ's December 5, 2016 letter noted, the study of the feasibility of environmentally superior swine waste technologies to the lagoon and spray field method began back in 2000.

[18] Title VI of the Civil Rights Act of 1964, Section 504 of the Rehabilitation Act of 1973, the Age Discrimination Act of 1975, Section 13 of Federal Water Pollution Control Act of 1972, and Title IX of the Education Amendments of 1972 (hereinafter referred to collectively as the federal non-discrimination statutes).

Some reviews of particular technologies were concluded more than a decade ago. According to the designee who made the decisions regarding the economic and environmental feasibility of the technologies, "[S]ubsequent research has focused on improving the economics of targeted technologies while maintaining the environmental performance."[19] EPA's ORD found in its review of reports discussed above that a number of risk management options are available to reduce potential health risks to nearby communities. EPA, USDA, and academia have continued to work on new processes, methods, and technologies to reduce impacts from industrial swine operations and waste since the review of available technology mentioned in NC DEQ's letter was completed.

Recommendations

ECRCO has not concluded its investigation and this letter does not contain ultimate findings of facts or law. Rather, ECRCO has summarized some of the information gathered to date to explain why ECRCO continues to be concerned about possible discriminatory impacts.

The totality of the information ECRCO has collected to date in its investigation, including NC DEQ's response to the complaint, indicates that the types of adverse impacts described above are being felt by large segments of the communities of color and are potential evidence of systemic concerns, not purely anecdotal claims. The information raises a concern that Swine Waste General Permit Program may run afoul of Title VI and EPA's Title VI regulations. NC DEQ's responses thus far have not provided a reason to dismiss the complaint or halt the investigation, nor has the information or arguments provided served to diminish ECRCO's level of concern.

On this basis, EPA makes a series of preliminary recommendations and requests a meeting to explore informal resolution. The recommendations are designed to focus an inquiry that will help them determine whether the problems are being caused by: (1) structural problems with the General Permit Program; (2) a lack of enforcement of the requirements of the permit (for example, no odors, no discharges, no spray beyond borders); or, (3) both.

ECRCO recommends that NC DEQ:

- Conduct an assessment of current Swine Waste General Permit to determine what changes to the Permit should be made in order to substantially mitigate adverse impacts to nearby residents. Determine which changes are currently within NC DEQ's authority to make and develop a timetable for adopting them. For Permit changes necessary to substantially mitigate the adverse impacts that NC DEQ cannot adopt, determine the source of the impediment to their adoption.
- Conduct an assessment of current regulations applicable to facilities operating under the Swine Waste General Permit to determine what if any changes to the regulations would be required to substantially mitigate adverse impacts to nearby residents. Determine which changes are currently within NC DEQ's authority to make and develop a timetable

[19] Williams, C.M. Williams, "*C.M. "Mike" Williams: Waste economics.*" The News & Observer, 8 June 2015. http://www.newsobserver.com/opinion/letters-to-the-editor/article23534074.html.

to adopt them. For regulatory changes necessary to substantially mitigate the adverse impacts that NC DEQ cannot adopt, determine the source of the impediment to their adoption.

- Evaluate the feasibility of risk management options available to reduce adverse impacts to nearby communities, including covering the waste lagoons; not spraying in the evenings and on weekends; not using dead boxes; and others described in this letter.
- Conduct an assessment of current mitigation technologies that would satisfy NC DEQ's performance criteria for new or expanding industrial swine operations and what if any impediments exist to adopting those technologies.
- Conduct a self-evaluation of the sufficiency of NC DEQ's enforcement and compliance efforts for existing rules governing the operation of its Swine Waste Management Program, including its response to odor and adverse health effects complaints, to determine whether implementation of any corrective measures are necessary including those to ensure a prompt and appropriate response to odor and other complaints. Determine which corrective measures are currently within NC DEQ's authority to make and develop a timetable for adopting them.
- Conduct an evaluation of NC DEQ's current policies and adjust them as appropriate to ensure the protection of confidentiality and identities of residents who provide information to NC DEQ about either environmental or civil rights complaints.
- Conduct a self-evaluation of its new non-discrimination program using the attached Procedural Safeguards Checklist to determine whether it has in place all the foundational elements listed to ensure that NC DEQ will meet its obligations under the federal nondiscrimination statutes. If any of the elements are not in place, NC DEQ should correct those deficiencies.

ECRCO looks forward to working with NC DEQ and would like to discuss with NC DEQ as soon as possible: the concerns previously outlined regarding the impacts on residents and communities; any additional information NC DEQ believes is relevant to the issue of whether Title VI has been violated; ECRCO's preliminary recommendations; NC DEQ's non-discrimination program; and the potential for informal resolution of this Complaint.

I will be contacting you in the next day or so to schedule a meeting to occur in the next two weeks. It is our goal to be able to reach informal resolution as soon as possible. We believe that through productive conversations over the next 60 days, we can negotiate an Informal Resolution Agreement that would address the concerns discussed in this letter and that would include a plan for moving forward on the above recommendations. If after discussion with NC DEQ, ECRCO does not believe informal resolution is possible, ECRCO will move forward to conclude its investigation and issue formal findings. If you have any questions, please feel free to contact me at (202) 564-9649, by e-mail at dorka.lilian@epa.gov, or U.S. mail at U.S. EPA, Office of General Counsel, External Civil Rights Compliance Office (Mail Code 2310A), 1200 Pennsylvania Avenue, N.W., Washington, D.C., 20460.

Sincerely,

Lilian S. Dorka
Director
External Civil Rights Compliance Office
Office of General Counsel

Cc:

Elise B. Packard
Associate General Counsel for Civil Rights and Finance
U.S. EPA Office of General Counsel

Kenneth Lapierrre
Assistant Regional Administrator
U.S. EPA Region 4

Attachments

ATTACHMENT A

1. Wing, Steve, et al., *Environmental Injustice in North Carolina's Hog Industry*, 108 Envtl. Health Perspectives 225 (2000)
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16. Costanza, Jennifer K., et al., *Potential Geographic Distribution of Atmospheric Nitrogen Deposition from Intensive Livestock Production in North Carolina, USA*, 398 Sci. Total Env't 76 (2008)
17. Schiffman, Susan S., et al., *Symptomatic Effects of Exposure to Diluted Air Sampled from a Swine Confinement Atmosphere on Healthy Human Subjects*, 113 Envtl. Health Perspectives 567 (2005)

18. Sacoby, M. Wilson & Serre, Marc L., *Use of Passive Samplers to Measure Atmospheric Ammonia Levels in a High-density Industrial Hog Farm Area of Eastern North Carolina*, 41 Atmospheric Env't 6,074 (2007)
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21. Burkholder, JoAnn M., et al., *Impacts of Waste from CAFOs on Water Quality*, 115 Envtl. Health Perspectives 308 (2007)
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ATTACHMENT B

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2. EPA, 2004. Risk Assessment Evaluation for Concentrated Animal Feeding Operations (No. EPA/600/R-04/042). EPA, Washington, DC.
3. GAO, 2008. *Concentrated Animal Feeding Operations: EPA Needs More Information and a Clearly Defined Strategy to Protect Air and Water Quality from Pollutants of Concern* (No. GAO-08-944). GAO, Washington, DC.
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6. Hutchins, S.R., White, M.V., Mravik, S.C., 2012. *Case Studies on the Impact of Concentrated Animal Feeding Operations (CAFOs) on Ground Water Quality* (No. EPA/600/R-12/052). EPA, Ada, Oklahoma.
7. NRC, 2003. *Air Emissions from Animal Feeding Operations: Current Knowledge, Future Needs.* National Academies Press, Washington, D.C.

ATTACHMENT C

CHECK LIST FOR PROCEDURAL SAFEGUARDS FOR RECIPIENTS FEDERAL NON-DISCRIMINATION OBLIGATIONS

Federal Non-Discrimination Statutes: Title VI of the Civil Rights Act of 1964, Section 504 of the Rehabilitation Act of 1973, the Age Discrimination Act of 1975, Section 13 of Federal Water Pollution Control Act of 1972, and Title IX of the Education Amendments of 1972.[20]

Item	Yes & Supporting Documentation	Not Yet	Checking
Notice of Non-Discrimination under the Federal Non-Discrimination Statutes[21]			
See attachment for recommended text of notice			
The non-discrimination notice is posted:			
• in a prominent location in your offices and facilities			
• prominently on your website			
• in any publications			
Grievance Procedures for Complaints filed under the Federal Non-Discrimination Statutes[22]			
A grievance procedure that:			
• Clearly identifies the Non-Discrimination Coordinator, including contact information			
• Explains the role of the Non-Discrimination Coordinator relative to the coordination and oversight of the grievance procedures			
• States who may file a complaint under the procedures			
• Describes which formal and informal processes are available, and the options for complainants in pursuing either			

[20] 40 C.F.R. § 7.85(g)
[21] 40 C.F.R. § 7.95(a).
[22] 40 C.F.R. § 7.90.

Item	Yes & Supporting Documentation	Not Yet	Checking
• Explains that an appropriate, prompt and impartial investigation of any allegations filed under federal non-discrimination statutes will be conducted			
• States that the preponderance of the evidence standards will be applied during the analysis of the complaint			
• Contains assurances that retaliation is prohibited[23] and that claims of retaliation will be handled promptly if it occurs			
• States that written notice will be promptly provided about the outcome of the investigation, including whether discrimination is found and the description of the investigation process[24]			
• Is published in print in general publications distributed to the public			
Non-Discrimination Coordinator[25]			
At least one Non-Discrimination Coordinator to ensure compliance with the federal non-discrimination statutes			
Non-Discrimination Coordinator or other individual responsible for:			
• Providing information internally and externally regarding rights to services, aids, benefits, and participation without regard to race, national origin, color, sex, disability, age or prior opposition to discrimination			
• Providing notice of your Agency's formal and informal grievance processes and the ability to file a discrimination complaint			
• Establishing grievance policies and procedures or mechanisms (e.g., an investigation manual)			
• Tracking all complaints filed with your Agency under federal non-discrimination statutes including any patterns or systemic problems			

[23] 40 C.F.R. § 7.100.
[24] Whether OCR considers complaint investigations and resolutions to be "prompt" will vary depending on the complexity of the investigation and the severity and extent of the alleged discrimination. For example, the investigation and resolution of a complaint involving multiple allegations and multiple complainants likely would take longer than one involving a single allegation of discrimination and a single complainant.
[25] 40 C.F.R. § 7.85(g).

Item	Yes & Supporting Documentation	Not Yet	Checking
• Semiannual reviews of all complaints filed with your Agency under federal non-discrimination statutes in order to identify and address any patterns or systemic problems			
• Appropriate training for your Agency's employees on your Agency's non-discrimination policies and procedures and obligations to comply with federal non-discrimination statutes			
• Updating complainants on the progress of their complaints filed with your Agency under federal non-discrimination statutes and any determinations made			
• Periodic evaluations of the efficacy of your Agency's efforts to provide services, aids, benefits, and participation in any of your Agency's programs or activities without regard to race, national origin, color, sex, disability, age or prior opposition to discrimination			
Public Participation			
Written and published public participation process/procedures that provide that when your Agency prepares a public participation plan for a specific action, it will include:			
• A description of the community (including demographics, history, and background)			
• A contact list of Agency officials with phone numbers and email addresses to allow the public to communicate via phone or internet			
• A list of past and present community concerns (including any complaints filed under the federal non-discrimination statutes)			
• A detailed plan of action (outreach activities) your Agency will take to address concerns			
• A contingency plan for unexpected events			
• Location(s) where public meetings will be held (consider the availability and schedules of public transportation			

Item	Yes & Supporting Documentation	Not Yet	Checking
• Contact names for obtaining language assistance services for limited-English proficient persons, including, translation of documents and/or interpreters for meetings			
• Appropriate local media contacts (based on the culture and linguistic needs of the community			
• Location of the information repository			
Access To Programs And Activities by Persons with Limited English Proficiency			
Has your Agency conducted an appropriate analysis described in OCR's LEP Guidance found at 69 FR 35602 (June 25, 2004) and http://www.lep.gov to determine what language services it may need to provide to ensure that individuals with limited-English proficiency can meaningfully participate in the process?			
Has your Agency developed a language access plan consistent with the details found in OCR's training module for LEP. http://www.epa.gov/civilrights/lepaccess.htm?			
Does your Agency have written and published procedures that:			
• Ensure meaningful access to all of your Agency's programs and activities to persons with limited-English proficiency and individuals with disabilities			
• Make communities you serve aware that foreign language services are available			
• Translate standardized documents			
• Provide for simultaneous oral interpretation of live proceedings such as town hall meetings or public hearings			
Access To Programs And Activities by Persons with Disabilities			
Does your Agency have written and published procedures to ensure to provide access to your programs, services, and activities for individuals with disabilities that:			
• Provides at no cost appropriate auxiliary aids and services including, for example, qualified interpreters to individuals who are deaf or			

Item	Yes & Supporting Documentation	Not Yet	Checking
hard of hearing, and to other individuals as necessary to ensure effective communication or an equal opportunity to participate fully in the benefits, activities, programs and services provided by your Agency in a timely manner and in such a way as to protect the privacy and independence of the individual			
• Ensures that your Agency's facilities and non-Agency facilities utilized by your Agency (*e.g.*, if your Agency holds a public hearing at a school) are physically accessible for individuals with disabilities			
• Makes communities you serve aware that services for individuals with disabilities are available			

NOTICE OF NON-DISCRIMINATION RECOMMENDED TEXT

[Agency Name] does not discriminate on the basis of race, color, national origin, disability, age, or sex in the administration of its programs or activities, as required by applicable laws and regulations.

[Insert name and title of Non-Discrimination Coordinator] is responsible for coordination of compliance efforts and receipt of inquiries concerning non-discrimination requirements implemented by 40 C.F.R. Part 7 (Non-discrimination in Programs or Activities Receiving Federal Assistance from the Environmental Protection Agency), including Title VI of the Civil Rights Act of 1964, as amended; Section 504 of the Rehabilitation Act of 1973; the Age Discrimination Act of 1975, Title IX of the Education Amendments of 1972, and Section 13 of the Federal Water Pollution Control Act Amendments of 1972.

If you have any questions about this notice or any of [Agency Name]'s non-discrimination programs, policies or procedures, you may contact:

> [Insert name and title of Non-Discrimination Coordinator]
> [Insert Agency Name and Address]
> [Insert phone number of Non-Discrimination Coordinator]
> [Insert email address of Non-Discrimination Coordinator]

If you believe that you have been discriminated against with respect to a [Agency Name] program or activity, you may contact the [insert title of Non-Discrimination Coordinator] identified above or visit our website at [insert] to learn how and where to file a complaint of discrimination.

ATTACHMENT 3

SENT VIA EMAIL

October 22, 2021

Elizabeth S. Biser, Secretary
N.C. Department of Environmental Quality
217 W. Jones Street
Raleigh, NC 27603

Dear Secretary Biser:

The Environmental Justice and Equity (EJE) Advisory Board was chartered to assist the Department of Environmental Quality (DEQ) in ensuring fair and equal treatment and meaningful involvement of all North Carolinians, regardless of race, color, national origin, or income, in the development, implementation, and enforcement of environmental laws and policies. In this role, we strive to ensure access to clean air, clean water, and clean soil, and the opportunity to live in safe and healthy communities for all North Carolina families.

Today, we write to you about pollution from hog operations, a long-standing environmental justice issue that has affected thousands of North Carolina families for decades. We respectfully request that DEQ take steps to protect these families, their health, and the environment.

Under the 2021 North Carolina Farm Act, N.C. Sess. L. 2021-78, DEQ must develop a general permit for hog operations that will produce swine waste-to-energy (biogas) by July 2022. As we expressed in our August 26, 2021 letter to you, the EJE Advisory Board has significant concerns about the pollution and public health implications of this general permitting scheme.

In addition to the procedural recommendations we provided in our August 26, 2021 letter, we advise DEQ to ensure the new general permit include robust substantive protections against hog waste pollution and its disparate impacts on surrounding communities. Cleaner technologies and practices that reduce water and air pollution—and that are compatible with biogas production—are available and practicable. In fact, some of these technologies are used by Smithfield Foods, the nation's largest pork producer, in other states. North Carolinians deserve the same protections.

In North Carolina, biogas is produced by capturing methane from hog waste lagoons using covered anaerobic digesters. To date, DEQ has allowed hog operations to dispose of waste from these digesters by transferring the digester waste to open "secondary" lagoons, and spraying the digester waste on fields.[1] The biogas is sent off-site for processing and eventually used to produce energy. Biogas produced using the lagoon and sprayfield system is not a clean source of energy.

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[1] *See, e.g.*, Permit No. AWI310039 Benson Farm (Mar. 31, 2021) (authorizing the use of a Waste-to-Energy system, which includes a covered anaerobic digester; a clay-lined lagoon; pumps, pipes, and other equipment to transfer waste; and sprayfields).

The lagoon and sprayfield waste management system used at industrial hog operations pollutes waterways,[2] contaminates drinking water,[3] and dirties the air people breathe.[4] This pollution and the resulting harms to human health have burdened neighbors—mainly people of color and low wealth communities--for decades.[5] As such, this is one of the most significant and well-studied environmental injustices in North Carolina; public health and environmental experts agree on the harm that this system causes for people and the environment.

Producing biogas from hog waste using anaerobic digesters, open secondary lagoons, and sprayfields does *not* address many of the longstanding, serious pollution problems of using open lagoons and sprayfields to store and dispose of hog waste. The use of digesters is likely to *increase* ammonia emissions when the digester waste is stored in open secondary lagoons and sprayed on fields.[6] Airborne ammonia from hog operations deposits in surrounding waterways, causing

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[2] Michael A. Mallin et al., *Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution*, 226 WATER, AIR, SOIL & POLLUTION 407 (2015), *available at* https://link.springer.com/article/10.1007/s11270-015-2669-y; Christopher D. Heaney et al., *Source Tracking Swine Fecal Waste in Surface Water Proximal to Swine Concentrated Animal Feeding Operations*, 511 SCI. TOTAL ENV'T 676 (2015), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4514616/; JoAnn M. Burkholder et al., *Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality*, 115 ENVT. HEALTH PERSP. 308 (2007), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1817674/.

[3] Wendee Nicole, *CAFOs and Environmental Justice: The Case of North Carolina*, 121 ENVT. HEALTH PERSP A182, A186 (2013), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC3672924/ ("Even without spills, ammonia and nitrates may seep into groundwater, especially in the coastal plain where the water table is near the surface."); M.E. Anderson & M.D. Sobsey, *Detection and Occurrence of Antimicrobially Resistant E. coli in Groundwater on or near Swine Farms in Eastern North Carolina*, 54 WATER SCI. & TECH. 211, 217 (2006), *available at* https://pubmed.ncbi.nlm.nih.gov/17037155/ ("Overall, the results of this study demonstrated that antibiotic-resistant E. coli were present in groundwaters associated with commercial swine farms that have anaerobic lagoons and land application systems for swine waste management."); Kenneth Rudo, *Groundwater Contamination of Private Drinking Well Water by Nitrates Adjacent to Intensive Livestock Operations (ILOs)*, N.C. DEP'T OF HEALTH AND HUMAN SERV., 414, 418 (June 1999).

[4] Nina G.G. Domingo et al., *Air quality-related health damages of food*, 118 PROCEEDINGS OF THE NAT'L ACAD. SCIS. 1 (May 2021), *available at* https://www.pnas.org/content/118/20/e2013637118; Leah Schinasi et al., *Air Pollution, Lung Function, and Physical Symptoms in Communities Near Concentrated Swine Feeding Operations*, 22 EPIDEMIOLOGY 208, 208 (2011), *available at* https://pubmed.ncbi.nlm.nih.gov/21228696/; Sacoby M. Wilson & Marc L. Serre, *Examination of Atmospheric Ammonia Levels Near Hog CAFOs, Homes, and Schools in Eastern North Carolina*, 41 ATMOSPHERIC ENV'T 4977, 4985 (2007), *available at* https://www.researchgate.net/publication/223777299_Examination_of_atmospheric_homes_and_schools_ammonia_levels_near_hog_CAMS_in_Eastern_North_Carolina

[5] Steve Wing & Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians* 2 (2014), *available at* https://www.ncpolicywatch.com/wp-content/uploads/2014/09/UNC-Report.pdf (finding that industrial hog operations are disproportionately located near communities of color and low-wealth communities in eastern North Carolina); Dana Cole et al., *Concentrated Swine Feeding Operations and Public Health: A Review of Occupational and Community Health Effects*, 108 ENVTL. HEALTH PERSPECTIVES 685 (2000), *available at* https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1638284/; Kendall M. Thu, *Public Health Concerns for Neighbors of Large-Scale Swine Production*, 8 J. AGRIC. SAFETY & HEALTH 175, 176 (2002), *available at* https://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.410.1811&rep=rep1&type=pdf; Steve Wing & Susanne Wolf, *Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents*, 108 ENVTL. HEALTH PERSP. 233 (2000), *available at* https://www.jstor.org/stable/3454439.

[6] Baines, R. (Edited), *Reducing greenhouse gas emissions from livestock production*, Taylor & Francis Group, London, 145 (2021), *available at* https://www.taylorfrancis.com/books/edit/10.1201/9781003048213/reducing-greenhouse-gas-emissions-livestock-production-richard-baines (finding that the potential for ammonia emissions when storing digested hog waste increases); Viney Aneja, et. al, *Characterizing Ammonia Emissions from Swine Farms in North Carolina: Part 2—Potential Environmentally Superior Technologies for Waste Treatment*, 58 J. AIR

pollution that can lead to algae blooms and fish kills.[7] Airborne ammonia also deposits on the ground, where it can seep into the soil and cause nitrate pollution in drinking well water, which can harm infants and pregnant women.[8] Airborne ammonia also forms fine particulate pollution that causes serious health problems and premature deaths in surrounding communities.

In fact, a recent study published by the National Academy of Sciences attributes an astounding 95 premature deaths in Sampson County and 83 premature deaths in Duplin County to the emissions from hog operations every year.[9] This is already an unacceptable situation that must be stopped. And the prospect of *increasing* the rates of sickness or death, resulting from sending more ammonia and fine particulate pollution into the surrounding environment, is simply unacceptable. DEQ must not allow hog waste pollution to continue harming more people in our most vulnerable communities.

Communities in eastern North Carolina have been complaining about pollution from industrial hog operations for decades. Since DEQ began considering permits for the first large-scale biogas project almost two years ago, hundreds of people across eastern North Carolina and beyond participated in public hearings, submitted comments, and appealed to DEQ to protect their communities and the environment from pollution from lagoons and sprayfields. To date, DEQ has failed to heed these calls.

In developing the conditions of the biogas general permit, DEQ must address this environmental injustice and protect families and the environment in eastern North Carolina. To start, DEQ's environmental justice analysis must be more than a formality intended to inform agency outreach. Instead, DEQ must conduct a comprehensive environmental justice analysis that translates into substantive permit conditions to minimize disparate impacts from cumulative impacts of the general permit and other DEQ-permitted operations on surrounding communities, including

& WASTE MGMT. ASS., 1145, 1156 tbl. 4 (2008), *available at* https://www.tandfonline.com/doi/pdf/10.3155/1047-3289.58.9.1145 (finding more than an 11 percent increase in ammonia emissions from an open secondary lagoon storing digester waste as compared to an open lagoon storing hog waste that has not been in a digester); Kupper et al., *Ammonia and greenhouse gas emissions from slurry storage—A Review*, 300 AGRICULTURE, ECOSYSTEMS, & ENV'T 1, 9 (2020) *available at* https://www.sciencedirect.com/science/article/pii/S0167880920301481; Lowry A. Harper et al, *The Effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions*, 39 J. ENVT. QUAL. 62 (2010), *available at* https://pubmed.ncbi.nlm.nih.gov/21284295/ (noting that because of the reduction of methanogenesis and its reduced effect on the chemical conversion of ammonium to dinitrogen gas, ammonia emissions from operations generating biogas increased by 46 percent compared to operations that did not produce biogas).

[7] Jennifer K. Costanza et al., *Potential geographic distribution of atmospheric nitrogen deposition from intensive livestock production in North Carolina, USA*, 398 SCI. OF TOTAL ENV'T 76, 77 (2008) http://jencostanza.com/docs/Costanza_et_al_2008_STOTEN.pdf; John T. Walker et al, *Atmospheric transport and wet deposition of ammonia in North Carolina*, 34 ATMOSPHERIC ENV'T, 3407, 3416 (2000), *available at* http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.557.3074&rep=rep1&type=pdf (detecting deposition of ammonia and ammonium upwards of 80 km from the source of that pollution).

[8] Mary Berg et al, *Nitrogen Behavior in the Environment*, N.D. AGR. EXTENSION SERV. 3 (2017), https://www.ag.ndsu.edu/publications/environment-natural-resources/nitrogen-behavior-in-the-environment; Dennis Keeney & Robert Olsen, *Sources of nitrate in groundwater*, 16 CRITICAL REVIEWS IN ENV'T SCI. & TECH. 257 (1986), https://www.tandfonline.com/doi/abs/10.1080/10643388609381748; Mary Ward, et al, *Drinking Water Nitrate and Human Health: An Updated Review*, 15 INT'L J. ENV'T RESEARCH & PUBLIC HEALTH 1 (July 23, 2018), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC6068531/.

[9] Nina G.G. Domingo et al., *Air quality-related health damages of food*, 118 PROCEEDINGS OF THE NAT'L ACAD. SCIS. 1 (May 2021), https://www.pnas.org/content/118/20/e2013637118.

communities of color and low-wealth communities that are already overburdened by pollution from multiple industries.[10] To be clear, it is not enough for DEQ to evaluate the cumulative effects of permitting decisions on water quality, as required under state environmental law; the agency, as a recipient of federal funding, also has obligations under Title VI of the Civil Rights Act of 1964, which require the agency to address harm to vulnerable North Carolinians.

In addition, as part of the general permit, we strongly advise DEQ to require the following:

- Cleaner technology and practices that are compatible with biogas production *and* address water and air pollution caused by the lagoon and sprayfield system, particularly given the increased ammonia pollution associated with open storage of biogas digester waste;

- Robust groundwater and surface water monitoring at every hog operation to identify pollution to rivers, streams, and groundwater, which is a source of drinking water for many rural residents;

- Updated nutrient management plans that account for the changes in the land-applied waste after digestion; and

- More protective freeboard requirements, such as automated lagoon/storage pond waste-level monitors and recorders, to reduce the likelihood that flooding or inundation of lagoons due to increasing frequent and severe storms will result in the discharge of the more harmful digester waste.

Thank you for your attention to this important matter.

Respectfully submitted by the EJE Advisory Board

James H. Johnson, Jr., Chair
Marian Johnson-Thompson, Vice Chair

[10] DEQ must scrutinize the environmental impact of the poultry industry as part of this analysis, as poultry operations with 30,000 or more birds are deemed permitted under state law, are often co-located in communities hosting swine operations, and have proliferated most rapidly in recent years in communities of color.

ATTACHMENT 4

SENT VIA EMAIL

August 26, 2021

Elizabeth Biser, Secretary
N.C. Department of Environmental Quality
217 W. Jones Street
Raleigh, N.C. 27603

Dear Secretary Biser:

The Environmental Justice and Equity Advisory Board (EJE) was created to bridge the gap between the Department of Environmental Quality (DEQ) and the communities of North Carolina. The EJE Advisory Board's primary charge is to advise you as the Secretary and assist DEQ in achieving fair and equal treatment as well as meaningful involvement of all North Carolinians--regardless of race, color, national origin, or income--in the development, implementation and enforcement of environmental laws, regulations and policies. Our shared goal is to mitigate adverse impacts of environmental policymaking on communities burdened disproportionately by environmental harms.

For some time now the EJE Advisory Board has been aware of the environmental concerns regarding concentrated animal feeding operations (CAFOs). With the enactment of the 2021 North Carolina Farm Act (SL 2021-78), the EJE Advisory Board has become more aware of the fact that placement of anaerobic digester systems will further exacerbate conditions for residents living near swine CAFOs. This is a critical environmental justice concern because the harm caused by swine waste mismanagement is disproportionately borne by Black, Latino, and Native American residents of North Carolina.[1, 2] Multiple studies have shown that living near CAFOs adversely affects the health and quality of life of fellow North Carolinians.[3, 4, 5, 6] We, as the EJE Advisory Board, cannot remain silent considering these facts.

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1 Steve Wing & Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians* (2014), https://www.ncpolicywatch.com/wp-content/uploads/2014/09/UNC-Report.pdf.q

2 Paul B. Stretesky et al., *Environmental Inequity: An Analysis of Large-Scale Hog Operations in 17 States*, 1982-1997, 68 Rural Soc. 231 (2003) (finding that between 1982 and 1997 large-scale hog operations in North Carolina were more likely to be sited in areas with disproportionate number of black residents),

3 Steve Wing et al., *Environmental Injustice in North Carolina's Hog Industry*, 108 Envtl. Health Perspectives 225, 228 (2000), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1637958/pdf/envhper00304-0081.pdf.

4 Steve Wing & S. Wolf, *Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents*, 108 Env't Health Perspectives 233, 233 (Mar. 2000), https://www.ncbi.nlm.gov/pmc/articles/PMC1637983.

5 Michael Greger & Gowni Koneswaran, *The Public Health Impacts of Concentrated Animal Feeding Operations on Local Communities*, 33 Family & Community Health 11, 13 (2010), https://www.humanesociety.org/sites/default/files/docs/public-impacts-factory-farms-on-communities.pdf.

6 Wendee Nicole, *CAFOs and Environmental Justice: The Case of North* Carolina, 121 Envtl. Health Perspectives A182, A 186 (2013), ("Even without spills, ammonia and nitrates may seep into groundwater, especially in the

Further, we have heard and shared the public's concerns about biogas and the development of a general permit that will allow CAFOs to install anaerobic digesters under one-size-fits all conditions.[7] Production of biogas must not be allowed to exacerbate known harms caused by lagoon and sprayfield systems, which are disproportionately borne by vulnerable North Carolinians.[8] And the permitting process must take into account local realities such as community demographics, environmental and health risks, and the cumulative impacts of other DEQ-permitted activities in the vicinity.

Moreover, with regard to Biogas systems, such as those recently proposed and permitted, far better waste treatment and disposal alternatives exist, which do far less harm to the environment. Other states have required the industry to use such superior waste management technology. DEQ should do the same to protect its citizens.

DEQ is now tasked with developing permit conditions for a new general permit that would authorize the construction and operation of farm digester systems to generate biogas. While DEQ has advised members of the General Assembly that it has the authority to further scrutinize any applicant/application perceived to present a danger to community, it has failed thus far to exercise that authority to protect impacted North Carolinians. For instance, the four individual biogas permits approved earlier this year do not require any regular air or water monitoring to assess the nature or volume of any emissions or discharges or the impact thereof on public health or the local environment. DEQ must take the steps necessary, during the process of developing and issuing coverage under this general permit, to assure that there is adequate and meaningful public participation such that the issues raised here are appropriately understood, evaluated, and addressed by the agency.

DEQ cannot assure that the protection afforded under the permit is adequate if it fails to provide sufficient opportunities for participation of affected communities to inform the permit conditions. DEQ staff may inspect permitted facilities once a year, but neighbors bear witness to permitted operations on a daily basis. This lived experience can help inform permit conditions and the scrutiny of permit applications, but only if community input is solicited and considered by DEQ permitting staff. Past efforts have failed to be inclusive of all affected community members. That should not be the case in this instance; DEQ has ample time to develop and implement an inclusive process. As recognized in DEQ's public participation plan and limited English proficiency plan, where additional steps are needed to assure participation, then DEQ should take them to give the public meaningful opportunity to comment before the agency develops the proposed permit.

The hope of the EJE Advisory Board is that no affected community is left out of the conversation due to a lack of access to dependable broadband service, especially in light of the digital divide

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coastal plain where the water table is near the surface."), https://www.ncbi.nlm.nih.gov/pmc/articles/PMC3672924/.

[7] Phoebe Gittelson, et al., *The False Promises of Biogas: Why Biogas is an Environmental Justice Issue*, Environmental Justice 3 (2021), https://www.liebertpub.com/doi/pdf/10.1089/env.2021.0025.

[8] DEQ Environmental Justice Report, *Swine Farm Modifications 3* (Dec. 22, 2020) (noting that, of four biogas projects under consideration, one would cover a single lagoon but retain uncovered lagoons, while 3 would excavate new lagoons and still retain/use existing uncovered lagoons to manage waste).

acknowledged by the Cooper Administration in the creation of the new Office of Digital Equity and Literacy. Considering the limited access by impacted community members to online fora, meaningful engagement can only be accomplished by conducting face-to-face meetings. Using WebEx or some other virtual platform is not an adequate substitute.

In developing this permit, DEQ should consider both the procedural and the substantive process. This advisory statement is intended to address the concerns that the Board has about the process for engaging the impacted communities. That process should be open and transparent at all times. It should be inclusive, and communities should have adequate notice of any and all actions of DEQ in this matter. The Board will conduct a special meeting to solicit additional input to inform a future advisory statement regarding the substance of the general permit.

To accomplish meaningful engagement of affected communities the EJE Advisory Board unanimously consents to offer the following recommendations to DEQ.

1. Conduct at least four public meetings. At least two should be public face-to-face meetings in the counties most affected – one in Duplin County and one in Sampson County. All meetings should be held in accordance with CDC COVID-19 guidelines to protect the health and safety of all attendees.
2. Alternatively, if it is not feasible to conduct a scheduled face-to-face meeting, DEQ should extend the date for any scheduled meetings until it is safe to conduct in-person meetings to receive public input during the development of the general permit.
3. Provide at least a 60-day notice of any planned public meeting to allow impacted communities to plan their participation.
4. Extend the period for public comment on the draft permit for at least 90 days to allow adequate time for local community members to provide comments for consideration by DEQ.
5. Provide Spanish interpretation services for participants with limited English proficiency. Demographic data reveal Hispanic residents are 1.39 times more likely to live near CAFOs than their white counterparts.[9]
6. Provide Spanish translation of draft permits, EJ analyses and related notices, as well as interpretation of hearing dialogue.
7. Engage an independent consultant to facilitate dialogue between stakeholders and agency staff at all meetings.
8. Respond in writing to community concerns expressed during the permitting process so that agency decision-making is transparent and reflects consideration, not merely invitation, of public input.
9. Consult with the NC Department of Health and Human Services to evaluate the health impacts of existing swine CAFOs including but not limited to those employing directed biogas technology.

Thank you in advance for your consideration in this matter.

[9] *See* Wing, supra note 1.

Respectfully submitted by the EJE Advisory Board Leadership Team,

James H. Johnson, Jr., Chair
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ATTACHMENT 5

Environ Monit Assess (2020) 192: 515
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Tracing nutrient pollution from industrialized animal production in a large coastal watershed

Colleen N. Brown · Michael A. Mallin · Ai Ning Loh

Received: 28 January 2020 / Accepted: 18 June 2020 / Published online: 14 July 2020

Abstract One of the highest concentrations of swine and poultry concentrated animal feeding operations (CAFOs) in North America is located on the Coastal Plain of North Carolina, in which the Cape Fear River basin is located. The CAFOs produce vast amounts of manure causing loading of nutrients and other pollutants to receiving waters. With the Cape Fear River basin vulnerable to nutrient pollution, as are many other watersheds with CAFOs, $\delta^{13}C$ and $\delta^{15}N$ stable isotopic signatures were identified from water samples collected within the Northeast Cape Fear, Black, and lower Cape Fear River watersheds to trace nutrient sources and their distribution downstream. The spatial and temporal variability of nutrients and isotopic signatures were also identified to understand water quality impacts of animal waste spraying season and proximity to CAFOs. Our results showed that significantly enriched $\delta^{15}N$ signatures characterized sites in close proximity to CAFOs as well as point-source wastewater discharge areas, while the opposite was true for an unimpacted control stream and two estuarine sites. Additionally, the impacted sites yielded significantly ($p < 0.05$) higher nitrate concentrations than control and estuarine sites. Statistical analyses demonstrated that nitrate concentrations were positively correlated with heavier $\delta^{15}N$ signatures, suggesting that animal waste, as well as human wastewater, are relatively more important sources of N to this large watershed than fertilizers from traditional row crop agriculture. Our results also suggested that during appropriate hydrological conditions CAFO-derived N can be detected many kilometers downstream from freshwater sources areas to the estuary.

Keywords Nutrient pollution · CAFO · Agriculture · Stable isotope · Carbon · Nitrogen

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Introduction

Industrial agriculture has become a norm for sustaining food production demands in recent decades and has been particularly congregated in certain rural communities in the USA. Industrialized livestock farms, recognized as concentrated animal feeding operations or CAFOs, have largely replaced the traditional agricultural practices of farrow-to-finish farms (McBride and Key 2013). Large corporations own swine, cattle, or poultry, and contract animals out to individual landowners to raise hundreds to thousands of livestock and poultry in close confinement. This technique allows for mass production with lower costs. In the USA, North Carolina has the third highest industrial animal production concentration by state with 1222 registered CAFOs, only exceeded by Minnesota with 1300 and Iowa with 3588 (USEPA 2017). The environmental and water quality impacts of such industrialized production techniques have been documented in the literature (Burkholder et al. 2007; Mallin et al. 2015), but the impacts of such production on waterways farther downstream are less well-defined, and thus assessed in this study.

Swine and poultry CAFOs

The Cape Fear River watershed, the largest watershed in North Carolina, has the highest concentration of swine CAFOs in the world as well as numerous poultry CAFOs (Cahoon et al. 1999; Mallin and Cahoon 2003). The magnitude of livestock and poultry in this area increases the demand for large quantities of animal feed, mainly comprised of soybeans and cornmeal (Choi 2007). These primary feed grains are imported to Mid-Atlantic states from outside states that specialize in corn and soybean agriculture because facilities cannot grow enough of their own feed to meet the demand themselves, and feed importation is a low-cost option. CAFOs located within the Cape Fear River basin in North Carolina import over 90% of soybeans for animal feed (Cahoon et al. 1999). While low-cost feed imports have facilitated the growth of CAFOs in North Carolina, they have also increased external, "new," nitrogen (N) and phosphorus (P) influxes to affected watersheds. With the uneven distribution of swine CAFOs across the USA, 13% concentrated in North Carolina, the Cape Fear River watershed is biogenically stressed with excess nutrient and fecal microbial influxes (Kellogg 2000; Harden 2015).

One of the leading environmental concerns brought about by the industrial agriculture system is the quantity and management of waste produced (USEPA 2004). The high-density animal confinement of CAFOs yield industrial levels of waste that have surpassed human waste production 13-fold (Burkholder et al. 2007). CAFO manure specifically threatens water quality degradation with nutrient contamination (USEPA 2004). Concentrated swine waste management and storage practices are generally characterized by the disposal and storage of waste water, comprised of swine feces and urine, in outdoor lagoons which are seasonally pumped out and sprayed on surrounding fields, termed "spray fields" (Mallin et al. 2004; Mallin et al. 2015). Within the common swine CAFO, waste accumulates in each individual stall and throughout the containment facility. The floors are comprised of slats that have slits allowing for manure to be hosed down through and drop into an underground conveyance below for storage, prior to being flushed out into the waste lagoons (Mallin et al. 2015). Anaerobic lagoons act as a treatment of wastewaters, converting organic material into stable compounds such as carbon dioxide and methane that are volatized (USEPA 2002). After the treatment period, in which dense organic sludge settles out and separates from supernatant liquid wastewater, the supernatant is sprayed on nearby fields containing a cover crop, usually Bermudagrass; spraying is permissible March through September (Mallin and McIver 2018). The nutrients from the concentrated waste are assumed to be absorbed by the cover crops on the swine waste spray fields. In traditional, farrow-to-finish farms, this method was viable because manageable waste amounts could be upcycled into fertilizer for their own swine forage (Hribar 2010). However, when this common practice is transitioned to CAFOs, the manure quantity exceeds land and/or soil nutrient capacity (Haines and Staley 2004).

Poultry CAFOs are also abundant in North Carolina, though the exact number remains unknown as dry litter poultry operations are not required to obtain permits. There are currently 19 wet-litter poultry CAFOs with permits in North Carolina (egg-laying facilities) while the vast majority of poultry CAFOs utilize dry litter waste disposal. Dry litter poultry farms are defined in that manner as the poultry waste (i.e., litter) is dried and spread on surrounding fields or shipped to other sites to be applied as nutrient fertilizer (Harden 2015). Overall, environmental contamination from swine and poultry CAFOs results from oversaturated spray field soils leaching into ground water during routine spraying, runoff from over capacitated lagoons or freshly sprayed fields during precipitation events and hurricanes, or leakage from poor lagoon construction (Amini et al. 2017; Burkholder et al. 2007). The magnitude of environmental impacts that CAFO wastewater imposes depends on the contaminants involved, soil properties, and proximity to waterways (Huddleston 1996; Burkholder et al. 2007). Close proximity of numerous CAFOs to streams and floodplains, apparent in North Carolina, facilitates vulnerability to wastewater runoff contaminants during storms and lagoon overflow (Wing et al. 2002; Martin et al. 2018).

Cape Fear River watershed

The Southeastern region of North Carolina, in which the Cape Fear River watershed is located, is physiographically characterized as Coastal Plain (NCDC 2018). The Coastal Plain is comprised of two main sub-areas, tidewater and interior portion (NCDC 2018). Tidewater areas, closest to the ocean, are characteristically flat, swampy, and poorly drained. The interior region has a gentle slope with good drainage. Overall, the Coastal Plain soil composition is soft sediment with generally no hard rock substrate beneath the surface (NCDC 2018). Wetlands dominate the Coastal Plain, particularly the tidewater areas, and have demonstrated water quality benefits through sediment and nutrient sequestration (Robinson 2005). The central region of North Carolina is described as Piedmont, located between the Coastal Plain and Mountain regions. The Cape Fear River watershed originates within the Piedmont and flows southeast crossing the fall line into the Coastal Plain and eventually emptying into the Atlantic Ocean. The lower 40 km of the river comprises the Cape Fear Estuary.

Watersheds with swine CAFOs have been found to contain streams with considerably higher inorganic and organic N concentrations than unimpacted streams (Harden 2015; Mallin et al. 2006, 2015). Anaerobic waste lagoons with large quantities of organic N produce high concentrations of ammonium-N through deamination (Mallin and Cahoon 2003), some of which is nitrified following discharge into the environment. The ability for nitrate to leech into groundwater depends on underlying soil composition and groundwater depth. High permeability soil (i.e., gravel, sandy gravel, and coarse sand) and relatively low aquifer depth are associated with high nitrate concentrations in groundwater (USEPA 2004). The North Carolina Coastal Plain generally has porous soils and a high water table, around 0–2 m below the surface, which create vulnerability to nutrient leeching (Mallin et al. 2015; USGS 2019). Thus, underlying geological and hydrological conditions exacerbate off-site transport of nutrients to downstream water bodies. Tracing of off-site nutrient loading can be accomplished through stable isotope studies.

The Cape Fear River region and surrounding coastal areas have been ranked most vulnerable to manure nutrient pollution within the USA (Kellogg 2000). Over 50% of North Carolina's swine production is located in the Cape Fear River Basin, where lagoon effluent is able to enter the blackwater streams that dominate such watersheds (Mallin et al. 2015; Martin et al. 2018). Unimpacted blackwater systems are characteristically inorganically nutrient poor and have low dissolved oxygen levels in comparison to anthropogenically induced streams (Smock and Gilinsky 1992). Phytoplankton production in these systems is stimulated by N, rather than P; however, P inputs directly stimulate bacterial growth; thus these systems are susceptible to biochemical oxygen demand (BOD)-induced oxygen depletion from loading of either nutrient (Mallin et al. 2004, 2006). CAFO-dominated watersheds of blackwater streams have the potential to drain substantial nutrient loads into the two major 5th-order tributaries (Northeast Cape Fear River and Black River) of the Cape Fear River, which subsequently enters the Atlantic Ocean through its estuary (Fig. 1). It is essential to assess nutrient concentrations and sources within the watershed to understand and mitigate nutrient pollution to the larger rivers and estuary.

Particulate organic matter ($\delta^{15}N$ and $\delta^{13}C$)

Particulate organic matter (POM) is characterized as particles that range from 0.054 to 2.0 mm, and physically cannot pass through a 7-μm filter pore (Cambardella and Elliott 1992; Nebbioso and Piccolo 2013). Organic carbon (C) and nitrogen (N) stable isotope ratios have been identified as natural tracers for organic matter origins and seasonal influxes (Thornton and McManus 1994). These isotopic ratios are assumed to reflect a distinct end-member that correlates with the biogeochemical process that alters their composition. Organic matter pools within river sediments are a conglomerate of biogenic processes, and isotopic mixing models estimate contributions from the assortment of various organic sources.

As mentioned previously, the application and storage of manure is associated with ammonia volatilization. When ammonia volatizes during wastewater degradation, the lighter $\delta^{14}N$ is removed leaving a majority of heavier $\delta^{15}N$ isotopes in the soil or waste ponds (Macko and Ostrom 1994). Bacterial nitrification in soil produces nitrate from ammonium, still rich in $\delta^{15}N$. POM $\delta^{15}N$ signatures have been distinguished for various nitrogen

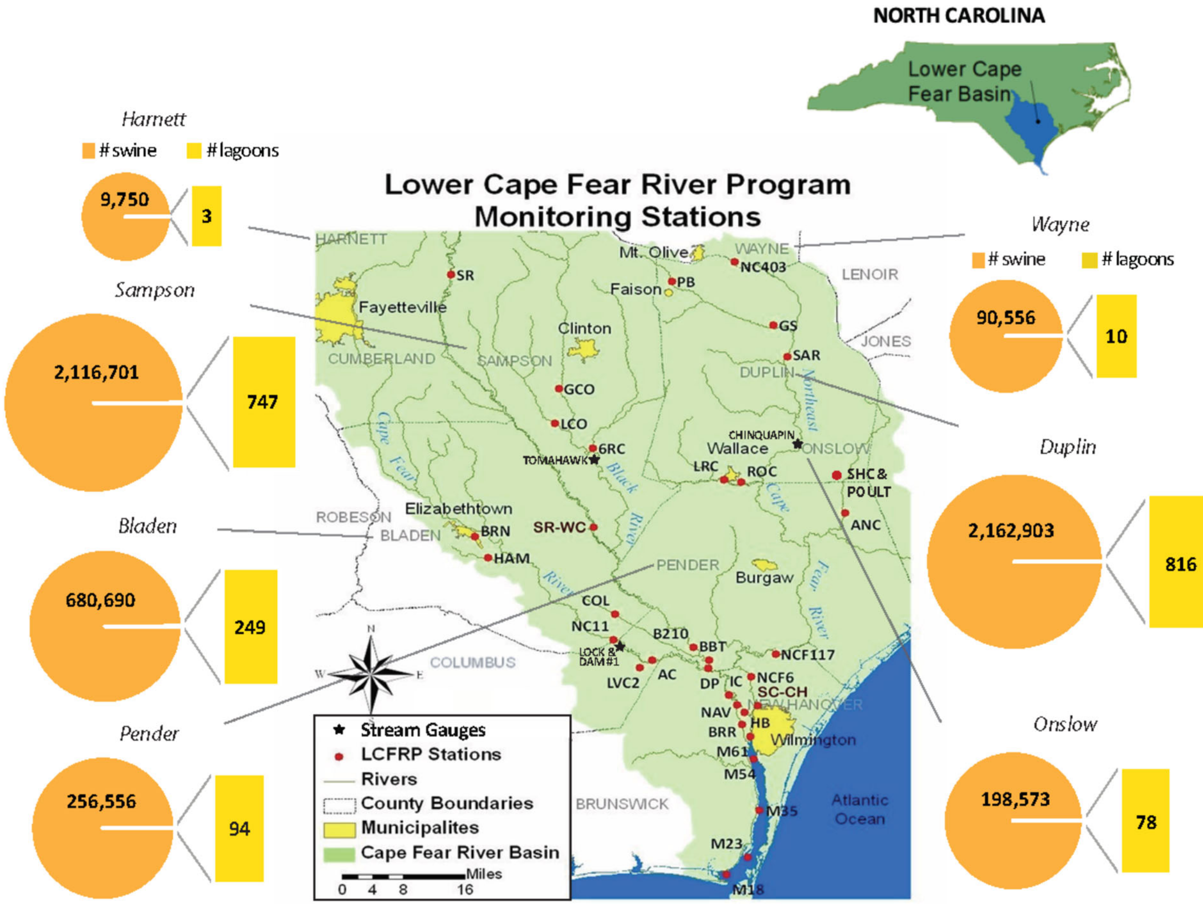


Fig. 1 Map of the Lower Cape Fear River Program monitoring stations showing sampling locations for this study. Inset shows location of sampling area within North Carolina, USA. Additional information includes standing crop of swine and number of waste lagoons per county (NCDEQ 2016)

source inputs (Table 1). Fixation of atmospheric N produces δ^{15}N signatures near zero (Kendall and McDonnell 1998), and atmospheric deposition of N produced signatures around − 3.0‰ (Heaton 1986). Synthetic fertilizers are generally in the − 2.0 to + 2.0‰ range (Bateman and Kelly 2007). Signatures of both human and animal waste are generally in the + 10.0 to > + 20.0‰ range, depending on degree of treatment or amount of microbial conditioning in the soil (Heaton 1986; Costanzo et al. 2001). The poultry waste signature generally ranges between + 8.0 and + 16.0‰, centered around + 8.0 to + 9.0 (Wassenaar 1995); although poultry manure pellets have displayed δ^{15}N signature as low as + 5.0‰ (Bateman and Kelly 2007). Thus, for human and animal waste, there is a considerable overlap, and distinguishing specific sources of N pollution can be aided by knowledge of the watershed land use and pollution sources (Lapointe et al. 2017). With deposition from atmospheric nitrogen, groundwater δ^{15}N typically ranges from + 2 to + 8‰ (Macko and Ostrom 1994). Marine aquatic sources have a broader range of δ^{15}N, from − 15 to + 20‰ (Fernandes et al. 2016).

In addition to δ^{15}N, the δ^{13}C signatures identify plant characteristics of specific water types (i.e., freshwater, estuarine, or seawater) to better understand the origin of carbon throughout the watershed. Freshwater plankton typically exhibit δ^{13}C signatures from − 30 to − 25‰, while C3 and C4 estuarine plants have heavier δ^{13}C signatures ranged − 27‰ to − 13‰ (Fry 2006). Carbon source nutrients from livestock and poultry waste can be identified through δ^{13}C signatures that reflect the plant material they eat, generally within the range of − 27‰ to − 22‰ (North et al. 2004). Overall, isotope tracing holds the potential to determine sources of environmental impact related to CAFO pollution distributed throughout a watershed.

Table 1 Overview of $\delta^{15}N$ signatures from various sources (from Heaton 1986; Wassenaar 1995; Costanzo et al. 211; Bateman and Kelly 2007; Lapointe et al. 2017)

Source	Range
Natural N-fixation	0‰
Atmospheric N	− 3‰ to + 1‰
Synthetic fertilizer	− 2‰ to + 2‰
Human wastewater (depends on degree of treatment)	+ 3‰ to + 19‰
Livestock waste	+ 10‰ to > + 20 ‰
Poultry waste (depends on poultry diet)	+ 5‰ to + 16 ‰

The CAFO-rich Cape Fear River watershed is vulnerable to the spread of nutrient pollution throughout the system (Mallin et al. 2004, 2006), and thus the primary objective of this study was to track and assess the presence of swine and poultry CAFO nutrient pollutants in the Cape Fear River watershed downstream through the Cape Fear River Estuary using $\delta^{15}N$ and $\delta^{13}C$ isotopic tracers. Additionally, the wide distribution of CAFOs throughout the watershed and periodic spraying of manure suggests the presence of seasonal and geographic distributions in water quality. Thus, the secondary objective was to assess spatial and temporal variability of nutrients (nitrogen and phosphorus), and isotopic signatures of POM in various streams and tributaries within the Cape Fear River watershed in relation to CAFO influence.

Materials and methods

Study sites

The lower Cape Fear River Basin is comprised of a series of rivers, streams, and tributaries that are subject to water quality fluctuations resulting from traditional agriculture, the numerous CAFOs present in the basin, and human and industrial point-source waste discharges permitted by the National Pollution Discharge Elimination system (NPDES). Ten sites were selected for sampling based on CAFO or point-source influence or lack thereof (Table 2). These sites are a subset of a greater and ongoing (since 1995) monitoring effort called the Lower Cape Fear River Program (Fig. 1). Two fifth-order blackwater streams, the Black River, and the Northeast Cape Fear River, together house approximately 5,000,000 head of swine and numerous poultry operations and flow downstream to connect with the main Cape Fear River upstream of the port city, Wilmington, NC (Fig. 1). The Black River is fed by tributaries, such as Six Runs Creek (6RC) that mainly drains watersheds used for CAFOs and agriculture, and Colly Creek (COL) which drains primarily undeveloped forest area and is considered a control site (Mallin et al. 2004, 2006). Station NC403 is a site in the headwaters of the Northeast Cape Fear River and receives influence from one NPDES point-source discharge (total 1.4 MGD), several CAFOs, and a cattle farm. Panther Branch (PB) and Stockinghead Creek (SHC) are tributaries of the Northeast Cape Fear River. PB is primarily impacted by an NPDES wastewater treatment plant outfall (0.5 MGD). The SHC watershed contains approximately 13 swine and 11 poultry CAFOs (Mallin and McIver 2018). A large poultry CAFO (POULT) was sampled in a perennial ditch that drains the facility and enters public waters. The main Cape Fear River watershed includes two tributaries selected for this study. One is the Browns Creek (BRN) watershed which contains three swine CAFOs but also encompasses 3500 resident Elizabethtown. The other is Hammond Creek (HAM) watershed which contains four poultry CAFOs and 13 swine CAFOs, including one very large CAFO (2500 or more swine heads). The watersheds of both creeks also are used for crop agricultural purposes (tobacco, sweet potatoes, corn, hay). The mainstem of the Cape Fear River flows downstream through Wilmington, and eventually discharges into the Atlantic Ocean in Southport, NC. Channel Marker 61 (M61) in downtown Wilmington and Channel Marker 18 (M18) in Southport, NC were sampled to investigate the influence of upstream watershed drainage to the Atlantic Ocean.

Sample collection

Of the ten sites within the Cape Fear watershed (Fig. 1, Table 2), eight were sampled monthly from February to August 2018; all sampling was suspended in September due to the approach of Hurricane Florence. Two sites, SHC and POULT were sampled in March (to represent low sprayfield influence) and in June (2 months after the onset of permissible swine waste spraying).At each site, a YSI Professional Plus unit with Quatro Multiparameter probe was used to measure temperature, salinity, dissolved oxygen (DO), turbidity, pH, and specific conductance (not presented in this paper). Water was collected at the surface using a bucket and rope, as the sites

Table 2 Site locations for sampling including coordinates and description (see also Fig. 1; EWG 2016; Waterkeeper Alliance; site locations on Lower Cape Fear River Program website https://uncw.edu/cms/aelab/lcfrp/)

Site	Coordinates	Description
COL	34.4641, − 78.2569	Colly Creek, 2nd order, control site located in a largely pristine blackwater wetland area; a tributary of the Black River; four swine CAFOs
NC403	35.1784, − 77.9807	Northeast Cape Fear River headwaters, 1st order, drains a watershed that hosts nine swine CAFOs, traditional agriculture; grazing cattle; an NPDES point-source waste water discharge (1 MGD)
PB	35.1345, − 78.1363	Panther Branch, 1st-order tributary of Northeast Cape Fear River, receives a NPDES point-source waste water discharge (0.5 MGD); one poultry CAFO in watershed
6RC	34.7933, − 78.3113	Six Runs Creek, 3rd-order tributary of the Black River, high influence of CAFOs (179 swine and 107 poultry CAFOs)
BRN	34.6136, − 78.5848	Browns Creek, 2nd-order tributary of the Cape Fear River, presence of three swine CAFOs and traditional agriculture; drains Elizabethtown (3500 residents)
HAM	34.5685, − 78.5155	Hammond Creek, 2nd-order tributary of the Cape Fear River; 13 swine CAFOs and four poultry CAFOs, traditional agriculture
M61	34.1938, − 77.9573	6th order, Cape Fear River at the State Port downtown Wilmington- estuarine system that receives water from Cape Fear River and Northeast Cape Fear River confluence
M18	33.913, − 78.017	6th order, Cape Fear River at South Port, lower end of the estuary that opens into the Atlantic Ocean
SHC	34.50305, − 77.51554	Stocking Head Creek, 2nd-order stream, site located along a swine CAFO spray field; 13 swine and 11 poultry CAFOs
POULT	34.50308, − 77.51548	1st-order, perennial drainage ditch with direct poultry CAFO influence; one large poultry CAFO

are located under bridges and are approximately 1–2 m in depth. Water samples were collected in duplicate 2 L, acid-washed (10% HCl) Nalgene bottles. Sample water was filtered on site through baked (550 °C for 4 h) 24-mm Whatman GF/F glass fiber filters (0.7 mm pore size) using a syringe and collected in acid-washed test tubes and baked clear glass vials. Once collected, all tubes and vials were placed on ice and transported back to the lab where they were frozen at − 20 °C until analyses for nutrients and dissolved organic matter.

Upon return to the lab, water samples in the 2-L Nalgene bottles were filtered through baked 47-mm Whatman GF/F filters. At least 500 mL of sample water from each bottle were filtered through an individual filter, and duplicates were produced for each site. The volume of water pumped through each filter was recorded to the nearest 0.1 mL. Once the filter was sufficiently full of organic material that water could no longer pass through, the filter was removed from the tower, wrapped in foil and frozen (− 80 °C) until analyses for $\delta^{13}C$ and $\delta^{15}N$ isotope signatures.

Nutrient analysis

All water samples that were filtered were analyzed for dissolved inorganic and organic nutrients. Dissolved inorganic nitrogen (DIN), as nitrite NO_{2-} nitrate NO_{3-}, and ammonium NH_4+, and total dissolved nitrogen (TDN), as well as dissolved inorganic phosphorus (DIP as orthophosphate, PO_4^{3-}), and total dissolved phosphorus (TDP) were analyzed using the Bran+Luebbe AutoAnalyzer 3. When nitrate is referenced henceforth, it will refer to the concentration of nitrate plus nitrite. TDN was determined using Koroleff's wet alkaline persulfate oxidation analysis where water samples were prepared with recrystallized persulfate solution and autoclaved. Samples with higher organic content were diluted 5:1 or 6:1 deionized water to sample ratios prior to analysis. The samples were digested to nitrate (NO_{3-}) and phosphate during analysis of TDN and TDP. The following equations were used to determine the concentration of dissolved organic nitrogen (DON) and dissolved organic phosphorus (DOP) from TDN, DIN, TDP, and DIP (Koroleff 1983):

$$[DON] = [TDN] - [DIN]$$

$$[DOP] = [TDP] - [DIP]$$

A variety of ancillary physical, chemical, and biological samples are collected monthly at most of the sampled sites and the state-certified data following QA/QC procedures are published on the Lower Cape Fear River Program website: https://uncw.edu/cms/aelab/lcfrp.

POM $\delta^{13}C$ and $\delta^{15}N$ isotopic analysis

For $\delta^{13}C$ and $\delta^{15}N$ isotopic analyses, POM filter samples were removed from the freezer and 15 mm diameter cores were punched from each filter. The glass fiber filter cores were weighed, then dried at 60 °C for 3–5 h and weighed again. Samples were then fumed in a desiccator with concentrated HCl to remove remaining inorganic carbonates present. Once dried again, samples were packed into tin capsules and placed into a sample tray for analysis on the ThermoDelta V Plus/Costech 4010 Elemental Analyzer to determine isotopic signatures of $\delta^{13}C$ and $\delta^{15}N$. The isotopic signatures of carbon and nitrogen were recorded as del notations and calculated using the isotope of the samples and standards as shown below:

$$\delta X = \left(\frac{R_{\text{sample}}}{R_{\text{standard}}} - 1\right) \times 1000$$

Vienna Pee Dee Belemnite was used as the stable carbon isotope standard, and atmospheric nitrogen (N_2) was the measure for the stable nitrogen isotope standard.

Statistical analyses

All data collected were recorded, archived, and organized using Microsoft Excel 2008. Statistical analyses were conducted using SAS statistical software (Schlotzhauer 2009). Nutrient, isotopic and hydrological data were first tested for normality using the Shapiro-Wilk test, with nitrate, ammonium, river discharge, and rainfall requiring log-transformation to achieve normality. Correlation analysis was used to test for associations among chemical and hydrological variables. Analysis of variance (ANOVA) was used to test for differences in mean nutrient concentrations and isotopic signatures among sites. Where significant differences occurred, sites were ranked using the least significant difference (LSD) procedure (Day and Quinn 1989). Statistical tests were considered significant if $P < 0.05$.

River discharge was used as a proxy for streamflow, since most individual streams in southeastern NC are not gauged. Data were used from USGS sites 02105769 (mainstream Cape Fear River at Lock and Dam #1), 02106500 (Black River near Tomahawk, NC), and 02108000 (Northeast Cape Fear River near Chinquapin, NC). The average discharge of the 7 days prior to sampling was used for statistical purposes, with data from Lock and Dam #1 used for BRN, HAM, M61, and M18; Tomahawk data were used for COL and 6RC, and Chinquapin data were used for NC403 and PB. Rainfall data for the day of sampling plus the preceding 48 h was used for statistical purposes, with the nearest official NC rain gauges (either from USGS or the NC State Climate Office) to the site in question used as sources.

Results

Dissolved nitrogen concentrations

Nitrate ($NO_{3-} + NO_{2-}$), exhibited the highest mean concentrations at station NC403 (68.8 μmol/L), followed by station 6RC (47.6 μmol/L) (Figs. 2a and 3d, f). The peak individual nitrate concentration occurred in March at NC403 with 124.8 μmol/L and the lowest nitrate concentrations were found from March to August at the unimpacted site COL (Figs. 2a and 3a). Nitrate concentrations at NC403 were significantly ($p < 0.05$) higher than all other sites except for 6RC, which in turn was significantly higher than all sites except PB and NC403. COL also had the lowest ($p < 0.05$) average concentrations of nitrate (2.24 μmol/L) among sites except for M18, and ammonium levels frequently exceeded that of nitrate more than any other station. March and July at M18 (nearest the ocean) had the lowest concentrations of ammonium (0.99 and 1.58 μmol/L), followed by June and August at 6RC (2.10 and 2.04 μmol/L). The highest values of ammonium occurred at PB in May (35.4 μmol/L) and HAM in February at 19.24 μmol/L. Average ammonium at PB was greater ($p < 0.05$) than 6RC, BRN, M61, and M18, with NC403 > 6RC and M18. Thus, highest inorganic N concentrations occurred at PB, N403, and 6RC. There was a general trend of higher nitrate concentrations during the start of spring (February, March, and April). COL exhibited the highest mean DON concentrations ($p < 0.05$), yet station 6RC had the highest individual DON overall in April (Figs. 2b and 3a, d).

At SHC, nitrate levels increased from March to June with a concentration of 18.6 to 26.5 μmol L^{-1} (Fig. 4a). Ammonium concentrations nearly doubled from March to June at SHC with concentrations of 1.8 to

3.2 μmol L^{-1} (Fig. 4a). DON concentrations for SHC also increased from March to June at 43.9 to 51.3 μmol L^{-1} (Fig. 4a).

At POULT, nitrate concentrations were only 1.05 μmol L^{-1} in March but increased to 28.8 μmol L^{-1} in June (Fig. 4b). Ammonium concentrations at POULT were temporally similar to SHC with 2.04 μmol L^{-1} in March and 3.43 μmol L^{-1} in June (Fig. 4b). There was a small increase in DON concentrations from March with 55.5 μmol L^{-1} to June with 60.7 μmol L^{-1} at POULT (Fig. 4b). Overall, both SHC and POULT showed a considerable increase in nitrate and DON concentrations and a relative increase in ammonium concentrations from March to June. All other stations, except M18, had a general decrease in nitrogen concentrations from March to June.

Dissolved phosphorus concentrations

DIP concentrations ranged from undetectable to 1.87 μmol L^{-1} (Fig. 5a–h). DOP concentrations ranged from 0.26 μmol L^{-1} up to 2.66 μmol L^{-1} (Fig. 5a–h). DIP concentrations were overall highest at COL with the maximum at 2.66 μmol L^{-1}, while DOP concentrations were lowest at COL with only one detectable data point at 0.20 μmol L^{-1} (Fig. 5a). Generally, lowest DIP and DOP concentrations occurred in April and May, and beginning in June there was an increase in both DIP and DOP concentrations that remained higher through August at each station (Fig. 5a–h). PB, BRN, HAM, and 6RC showed the highest concentrations of DP (Fig. 5b–e). The estuarine stations M61 and M18 had the lowest DOP concentrations ranging from 0.00 to 0.85 μmol L (Fig. 5g, h). Lower DIP concentrations were found at stations PB and BRN, with averages of 1.12 and 0.86 μmol L^{-1}. Overall, M18 had the lowest average DIP and DOP, and COL had nearly zero DOP concentrations, yet the highest average DIP. Average DIP concentrations at both COL and NC403 were significantly ($p < 0.05$) higher than DIP at PB, BRN, and M18. Overall, none of the DIP samples was particularly high.

DIP concentrations at SHC doubled from March with 1.05 μmol L^{-1} to June at 2.5 μmol L^{-1} (Fig. 4c). DOP increased substantially from 0.14 μmol L^{-1} in March to 1.49 μmol L^{-1} in June at SHC (Fig. 4c). At POULT, the DIP concentration increased 15× from 0.27 μmol L^{-1} in March to 4.1 μmol L^{-1} in June (Fig. 4d). DOP concentrations at POULT increased temporally from 0.40 μmol L^{-1} in March to 1.67 μmol L^{-1} in June (Fig. 4d).

Isotopic signatures

δ^{15}N POM isotopic signatures ranged from − 3.60 to + 21.7‰ (Table 3; Figs. 2e, 6, and 7). Isotopic signatures at COL never reached above δ^{15}N + 10‰, except for one outlier of δ^{15}N + 10.5‰ in October 2017 (Table 3; Figs. 2e, 6, and 7). M18 δ^{15}N signatures also remained below + 10‰ (Figs. 2e and 6). Station M61 had one δ^{15}N isotopic signature that was + 9.9‰ (Figs. 2e and 6). Stations that showed several instances of isotopic signatures above δ^{15}N + 10‰ include PB, NC403, BRN, HAM, and 6RC (Figs. 2e, 6, and 7). CAFO site 6RC and point-source site PB had particularly elevated mean δ^{15}N signatures at + 10.6 and + 10.5‰ (Table 3). Average δ^{15}N signatures at both point-source sites PB and NC403 were significantly ($p < 0.05$) higher than those of COL, M61, and M18; δ^{15}N signatures at CAFO sites 6RC and HAM were significantly higher than COL. The highest individual δ^{15}N signature was station PB in April at + 21.7‰ (Figs. 2e, 6, and 7). BRN closely followed at a maximum δ^{15}N of + 21.6‰ in April (Table 3). HAM and 6RC also had δ^{15}N signatures above + 20‰ at + 20.1‰ in August and + 20.9‰ in April (Figs. 2e, 6, and 7). Point-source site NC403 also showed heavy δ^{15}N signatures at + 12.8 and + 12.3‰ in May and June (Table 3). Overall, δ^{15}N signatures above + 10‰ generally had the highest values in April and occurred either in the summer months of May, June, or August (Figs. 2e, 6, and 7).

POM signatures of δ^{13}C ranged from − 32.7 to − 20.2‰ (Table 3). The control site COL generally had a narrow δ^{13}C isotopic range of − 30 to − 29‰ (Table 3; Figs. 2f, 6, and 7). M18 had isotopic signatures that ranged from δ^{13}C − 30.1 to − 20.2‰ (Table 3). HAM also exhibited a broad range of δ^{13}C signatures from − 31.3 to − 22.7‰. PB, BRN, 6RC, COL, and M61 all had isotopic signatures ranging from δ^{13}C − 31 to − 27‰, with one outlier point at δ^{13}C − 31.3‰ (Figs. 2f, 6, and 7). Overall, COL had the most specific δ^{13}C signature and the other stations varied temporally and spatially.

As mentioned, normalized data were subject to correlation analyses (n for all correlations below was either 53 or 54). The overall δ^{15}N signature was positively correlated with nitrate concentration ($r = 0.34$, $p = 0.01$) and DIN concentration ($r = 0.29$, $p = 0.04$), but was uncorrelated with ammonium concentration (Table 4). δ^{15}N

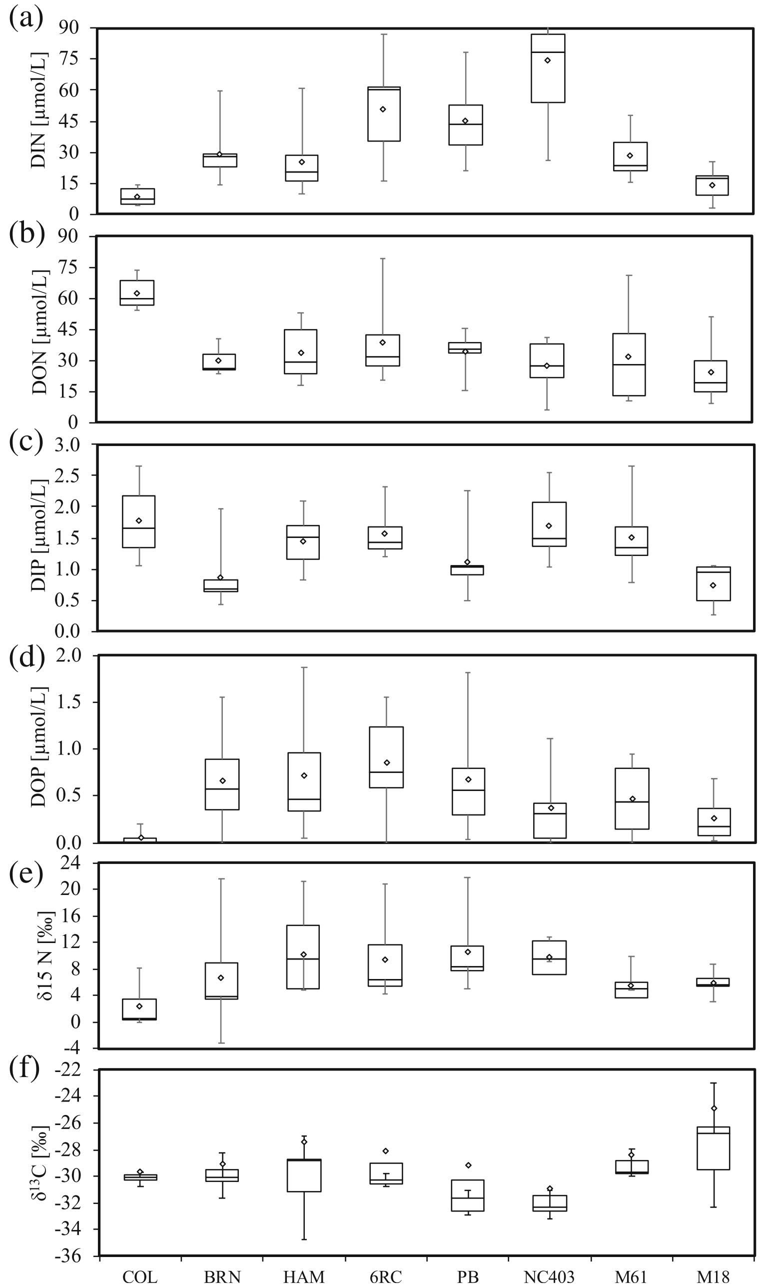


Fig. 2 Box-Whisker plots (minimum, maximum, 75th, 25th percentile, average (open circle), and median) **a** dissolved inorganic nitrogen (DIN), **b** dissolved organic nitrogen (DON), **c** dissolved inorganic phosphorus (DIP), **d** dissolved organic phosphorus (DOP), **e** stable nitrogen, and **f** stable carbon isotopic signatures in POM for the study duration (February through August, 2018)

signatures were not correlated with rainfall or discharge. Among nutrient concentrations, nitrate was negatively correlated with DON ($r = -0.44$, $p = < 0.01$). Ammonium was not significantly related to organic nutrients, but was positively correlated with DIP ($r = 0.36$, $p < 0.01$). Rainfall was not significantly correlated with inorganic nutrients, but was negatively related to DON ($r = -0.38$, $p < 0.01$). River discharge was negatively related to DIP ($r = -0.46$, $p < 0.01$).

Discussion

Spatial nutrient variability

Nutrient concentrations (TN, TP, DON, DOP, $\delta^{13}C$, $\delta^{15}N$) of streams throughout the Cape Fear River Basin vary by CAFO influence. COL drains largely undeveloped wetlands with little CAFO influence from the watershed and was used as a baseline comparison against

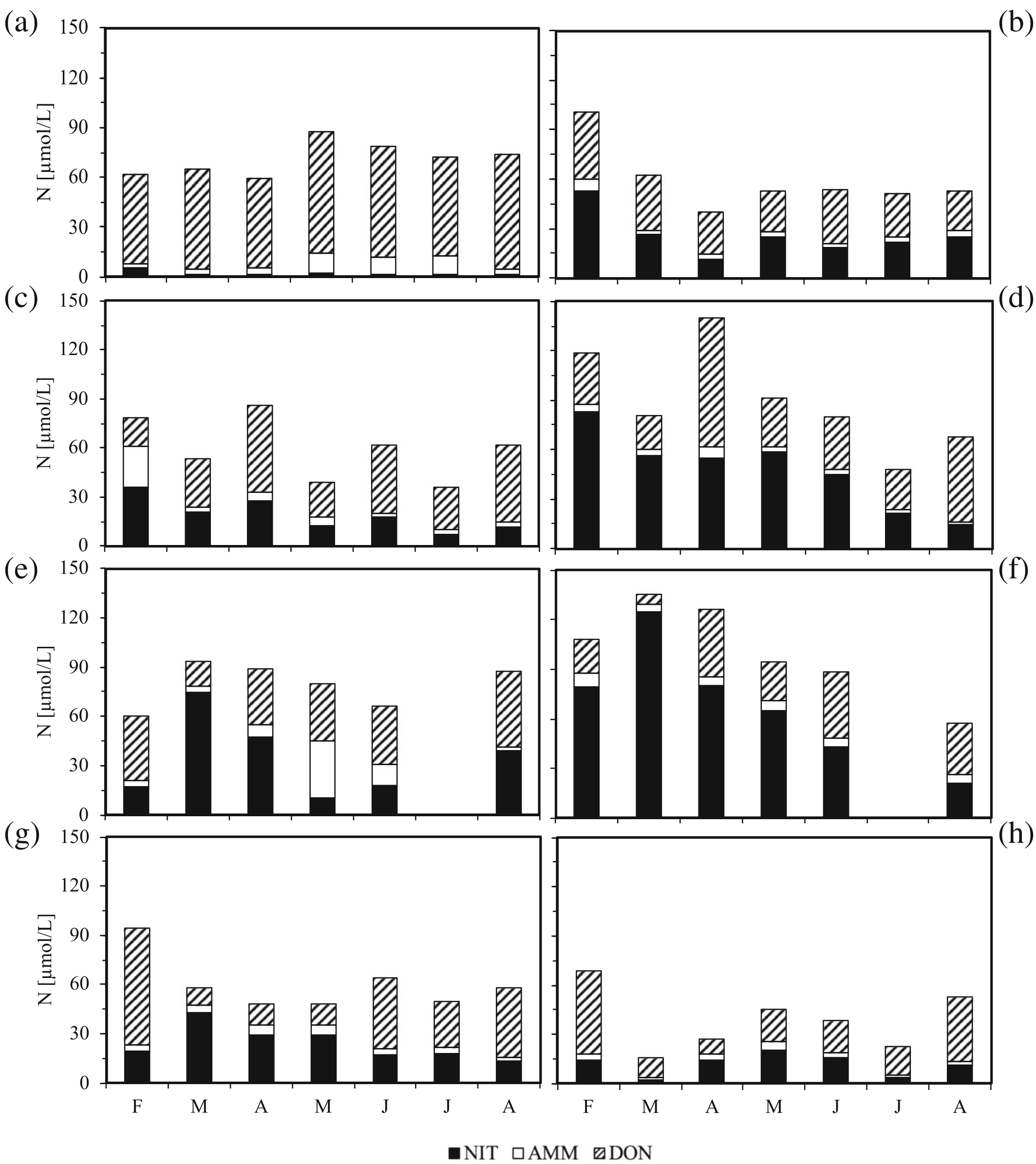


Fig. 3 Total dissolved nitrogen (TDN) concentrations are comprised of dissolved inorganic nitrogen (DIN), in the form of nitrate (NIT; $NO_{3-} + NO_{2-}$) and ammonium (AMM; NH_4+), and dissolved organic nitrogen (DON). The bars in this figure depict the concentrations of DIN and DON for comparison at each station, **a** COL, **b** BRN, **c** HAM, **d** 6RC, **e** PB, **f** NC403, **g** M61, and **h** M18, for the study duration, February through August of 2018. No samples were collected in July from PB and NC403

anthropogenically influenced sites. Though COL has one less CAFO in its direct vicinity than BRN, it is considered unimpacted because BRN's watershed drains the residential and commercial area of Elizabethtown. The consistently low nitrate concentrations (2.24 ± 1.33 μmol L^{-1}) at COL exemplify unimpacted black water systems that have characteristically low nitrogen concentrations as nutrients are not preserved in the floodplain and soils have low N concentrations (Smock and Gilinsky 1992; Mallin et al. 2006). The wetland-rich watershed would be conducive to enhanced denitrification. Total nitrogen values of unimpacted blackwater systems are predominately DON, which was demonstrated at COL with DON concentrations that significantly exceeded concentrations at all other sites (Figs. 2b and 3a–h). The data set as a whole demonstrated a

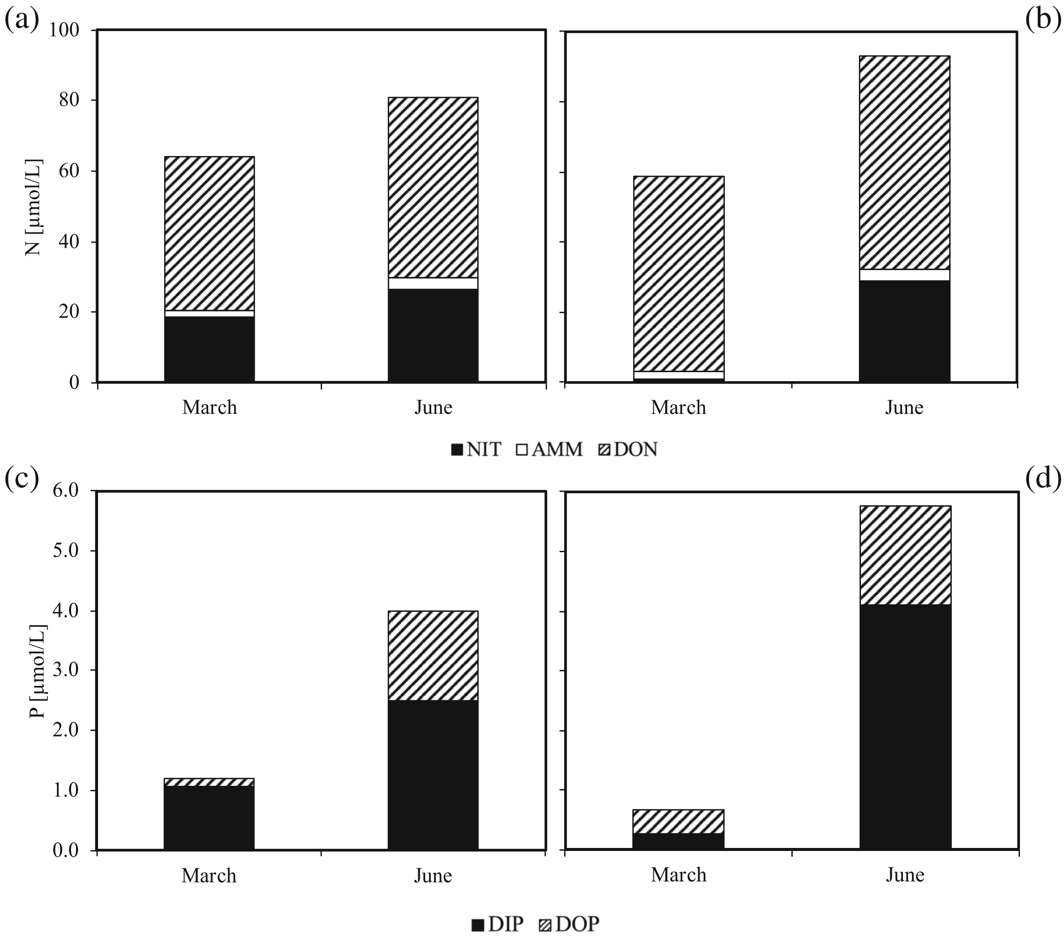


Fig. 4 Nutrient concentrations (N and P) are plotted for CAFO sites, Stocking Head Creek (**a**, **c**) and Poultry (**b**, **d**), prior to lagoon waste being sprayed (March) and after spray (June) of 2018

negative correlation between nitrate and DON; note that nitrate is anthropogenically sourced, while DON can be a product of wetland processes. The majority of total phosphorus concentrations were as DIP (orthophosphate), as DOP levels were nearly undetectable at COL with only one sample detectable at 0.20 μmol L^{-1} (Figs. 2c, d, and 5). Isotopic signatures at COL commonly did not show any influence of animal effluent ($\delta^{15}N < +10‰$), which further exemplified the near-unimpacted condition of this stream. Feral swine have been photographed by the research team in the vicinity of this sampling site and the single abnormally high $\delta^{15}N$ could be attributed to these swine. Additionally, the $\delta^{13}C$ values were consistently ~ 29.5‰, which is indicative of the C3 plants that inhabit a freshwater system such as COL (Fry 2006).

The other freshwater sites displayed inorganic N concentrations and isotopic signatures suggestive of animal effluent and/or human wastewater. CAFO-rich 6RC had elevated DIN and DOP (Fig. 2a, d) an enriched mean $\delta^{15}N$ of + 10.6‰ and a maximum of + 20.9‰ (Table 3). High DIN concentrations also characterized NC403 (Fig. 2a) which had a mean $\delta^{15}N$ of + 9.7‰ and maximum of + 12.8‰ (Table 3). This station drained a watershed containing several CAFOs and a human NPDES point-source wastewater discharge (Table 2) The low variability among $\delta^{15}N$ signatures at NC403 (Table 3) argues for strong watershed human influence (unfortunately the Cape Fear Basin has no TN or TP discharge limits—only the ammonium discharges are limited). The other NPDES site, PB, had elevated DIN and heavy $\delta^{15}N$ signatures (Table 3). Thus, both sites with human wastewater influence showed both elevated DIN and heavy $\delta^{15}N$ signatures, as did CAFO-rich 6RC.

The other site showing heavy $\delta^{15}N$ mean, median, and maximum values was HAM, which has a considerable number of swine and poultry CAFOs in its

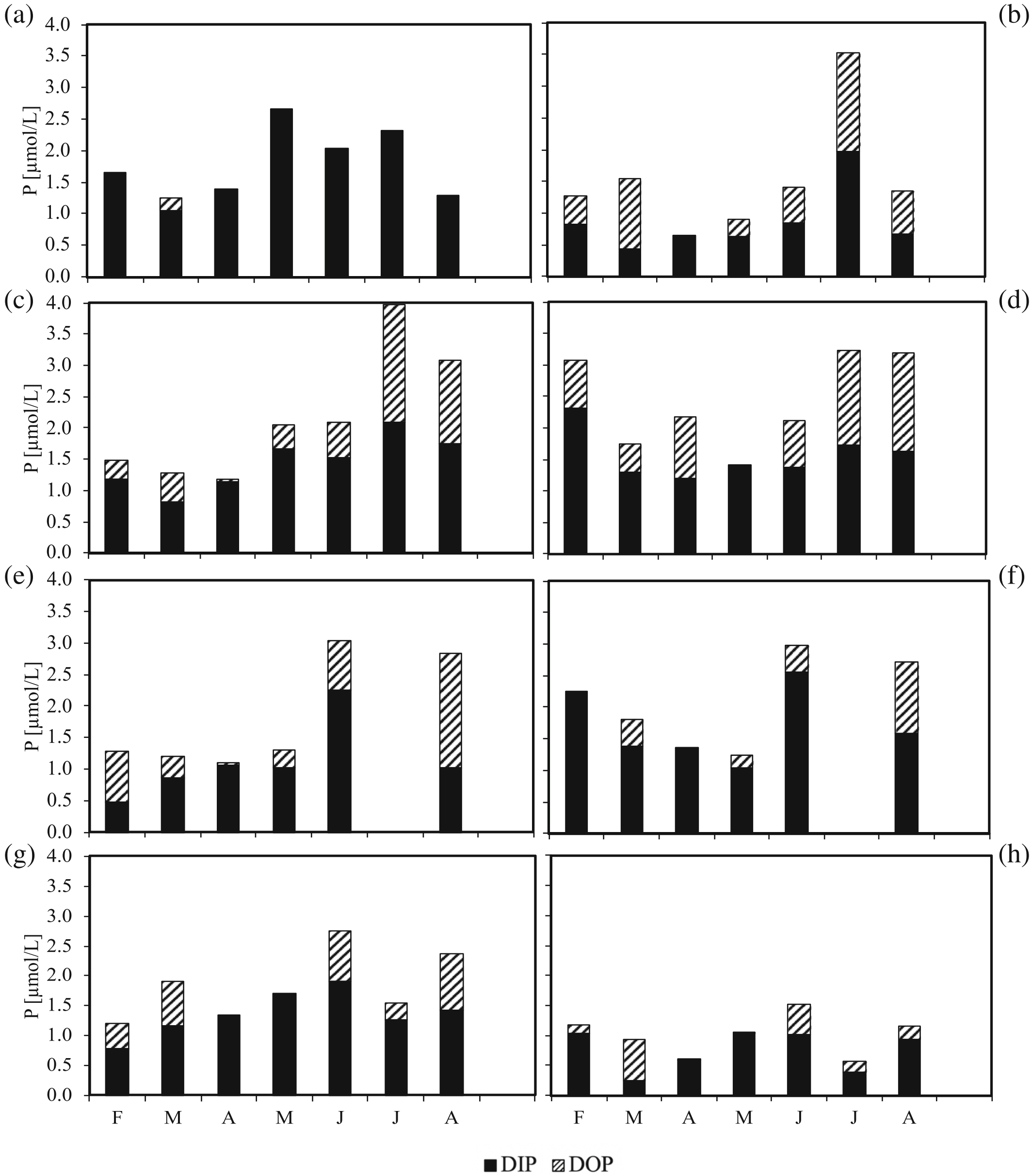


Fig. 5 Total dissolved phosphorus (TDP) concentrations, in the forms of inorganic (DIP) and organic phosphorus (DOP) are shown for each site **a** COL, **b** BRN, **c** HAM, **d** 6RC, **e** PB, **f** NC403, **g** M61, and **h** M18. The TDP concentrations are plotted for each sampling month (February through March, 2018) in the study duration. The bars in each table represent DIP as orthophosphate (PO_4^{3-}) and DIP, to represent TDP. No samples were collected in July from PB and NC403

watershed (Table 2). In contrast, BRN had widely variable $\delta^{15}N$ signatures, with a low median (3.9) and a moderate mean $\delta^{15}N$ signature of 6.7 (Table 3). This watershed contained three CAFOs but also drains the town of Elizabethtown, presumably introducing urban and suburban stormwater runoff into Browns Creek. Note that the town's wastewater treatment plant (1.2 MGD, secondarily treated) outfall discharges directly into the Cape Fear River. Urban stormwater runoff will contain a mixture of fertilizer N, atmospheric N, and nutrients from pet waste. The two estuarine sites showed low to moderate DIN concentrations (Fig. 2a) and lighter $\delta^{15}N$ signatures (except for April—see below).

CAFO source sites, SHC and POULT, showed nutrient concentrations and isotopic signatures that distinguished them from the other streams in the Northeast

Table 3 Isotopic signatures (‰) of the eight primary sampling sites (as mean ± standard deviation/median, range), $n = 8$ months

Site	$\delta^{13}C$ signature	$\delta^{15}N$ signature
COL	− 29.7 ± 0.4	3.3 ± 4.1
	− 29.7, − 30.3 to − 29.0	1.6, − 0.3 to 10.5
NC403	− 30.9 ± 1.2	9.7 ± 2.8
	− 30.6, − 32.7 to − 29.7	9.4, 6.9 to 12.8
PB	− 29.2 ± 1.4	10.5 ± 6.0
	− 29.0, − 30.9 to − 27.8	8.3, 4.6 to 21.7
BRN	− 29.1 ± 1.2	6.7 ± 7.9
	− 29.0, − 31.4 to − 27.4	3.9, −3.6 to 21.6
HAM	− 27.4 ± 2.7	10.0 ± 7.3
	− 28.7, − 30.6 to − 22.7	9.5, 0.4 to 21.1
6RC	− 28.2 ± 0.9	10.6 ± 6.8
	− 27.7, − 29.4 to − 27.2	6.3, 3.3 to 20.9
M61	− 28.4 ± 1.1	5.4 ± 2.3
	− 27.9, − 30.6 to − 27.6	5.1, 3.5 to 9.9
M18	− 25.0 ± 3.2	5.9 ± 1.9
	− 25.7, − 30.1 to 20.2	5.5, 2.9 to 8.8

Cape Fear River basin. Swine and poultry-influenced SHC had a $\delta^{15}N$ signature characteristic of swine effluent (+ 18.2) in March and a mixed animal waste signature of + 7.4‰ in June, and POULT showed poultry waste signatures (+ 6.6–7.0‰) in March and June (Wassenaar 1995). Sprayfields and lagoons for swine CAFOs were adjacent to SHC, while POULT was sampled at a perennial ditch draining a large poultry CAFO. The waste disposal system at poultry farms utilize dry litter applied on fields that pose different environmental impacts than hog farms as there is no liquid waste spraying unless the poultry CAFO is an egg-laying facility; there are no NPDES requirements for dry-disposal poultry CAFOs. Dry poultry litter can impose adverse environmental effects from field runoff in rain events or light litter particles carried by wind into adjacent waterways, which increase nutrient concentrations.

A broad-based USGS study found substantially higher nitrogen concentrations (nitrate, ammonium, and TN) in swine and poultry-intensive watersheds compared to those without influence from these agricultural facilities (Harden 2015). The Cape Fear River watershed has significant spatial variability, where stations with high-density CAFO influence (as well as sites with NPDES point-source discharges) had higher nutrient concentrations and $\delta^{15}N$ isotopic signatures indicative of waste effluent than that of the relatively unimpacted blackwater system. While the Cape Fear watershed has considerable acreage under crop agriculture, the positive correlation between nitrate concentrations and heavier $\delta^{15}N$ signatures suggests that animal waste as well as human wastewater play a stronger role in nitrogen pollution than crop fertilizers. We also note that under conditions of elevated streamflow (see next section) swine waste effluent from upstream regions of the watershed can influence the water quality in downstream estuarine regions that have no direct influence of CAFOs.

Seasonal variability

CAFOs follow a seasonal schedule for waste disposal (spraying), and the transitions through the seasons are shown in each sample site's nutrient concentration and isotopic composition, dependent on the proximity to

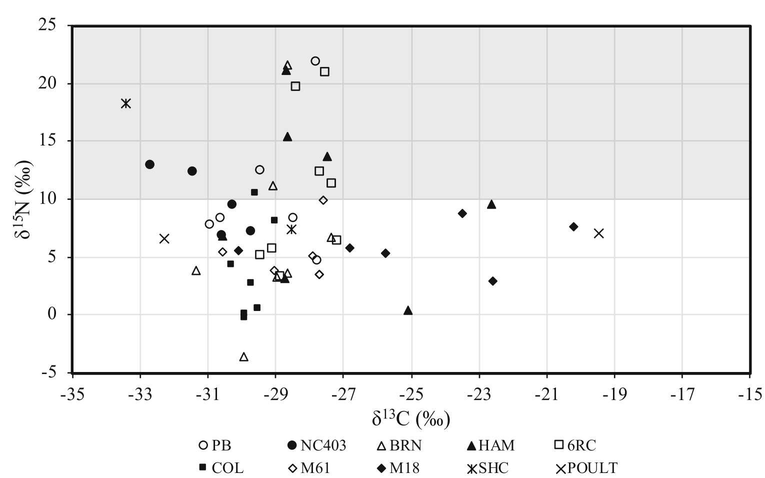
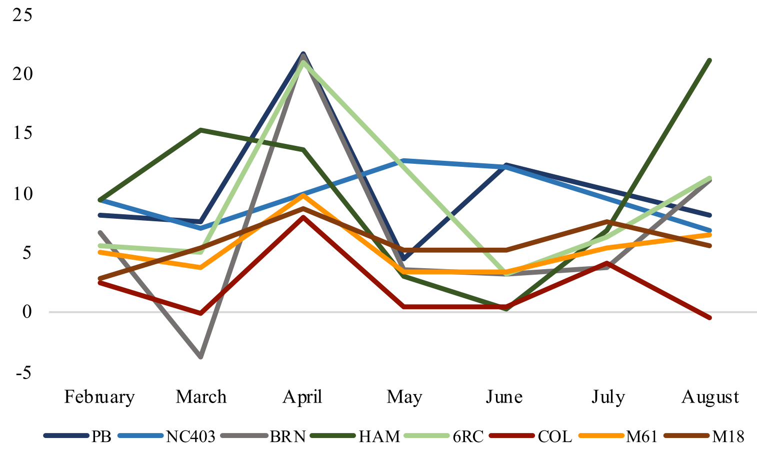


Fig. 6 Particulate organic matter (POM) isotopic signatures plotted for $\delta^{13}C$ and $\delta^{15}N$ at each sampling station from February to August, 2018

Fig. 7 Monthly variability of isotopic $\delta^{15}N$ signatures for each sampling station



CAFOs. The spraying season for swine farms to relieve waste lagoons onto surrounding Bermudagrass fields occurs during the growing season, from March to September. It was previously demonstrated that nitrate, total nitrogen, and fecal bacteria concentrations in CAFO-rich stream stations rose considerably after the onset of spray season in relation to before (Mallin and McIver 2018). Thus, it was expected for months following the start of spraying season in March to exhibit elevated nutrient concentrations and isotopic signatures characteristic of swine effluent, compared to the fall and winter months exclusive of spraying. The Bermudagrass fields, where waste is sprayed on, are intended to sequester the excess nutrients introduced by swine effluent. Bermudagrass has one of the highest capabilities to uptake nutrients, and thus was chosen as a cover crop on which to spray lagoon waste. However, as shown in this study and others (Harden 2015; Mallin et al. 2015; Mallin and McIver 2018), high concentrations of nutrients still enter adjacent waterways during the spraying season, although in the present study 6RC and NC403 showed decreased concentrations after the initial spraying. The maximum practical nitrogen application rate for Bermudagrass in coastal areas is suggested to be 300 pounds per acre per year, divided into three application times (Conrad-Acuña et al. 2019). This application rate is solely for the maximum growth of Bermudagrass and does not assess the environmental factors of N leaching into the surrounding environment. Overapplication of waste or leaching from applying the maximum N application rate may explain the increased nutrient concentrations. The threshold of nutrient sequestration was likely met in the Bermudagrass field, and excess nutrients leach into groundwater or run off over land into surrounding waterways.

Table 4 All significant correlations between measured parameters ($\delta^{15}N$, DIN, DON ammonium, nitrate, DIP, discharge, and rainfall)

	$\delta^{15}N$	DON	Ammonium	Discharge
Nitrate	$r = 0.34$ $p = 0.01$	$r = -0.44$ $p = < 0.01$		
DIP			$r = 0.36$ $p < 0.01$	$r = -0.46$ $p < 0.01$
DIN	$r = 0.29$ $p = 0.04$			
Rainfall		$r = -0.38$ $p < 0.01$		

Isotopic signatures were not notably characteristic of swine effluent in March; however, in April, there was a large and widespread increase as $\delta^{15}N$ signatures amplified at some sites to + 21.0‰, which is highly indicative of manure-sourced effluent. 6RC, a CAFO-dominated tributary with 179 swine CAFOs, also exhibited its heaviest $\delta^{15}N$ signature in April at + 20.9‰ (Fig. 7). In fact, average $\delta^{15}N$ signatures for the entire sampling set were significantly ($p < 0.05$) higher in April compared to all other months sampled. Nitrogen signatures may not have been expressed in March due to the lag time of leachate to enter waterways. Enriched $\delta^{15}N$ signatures in April may be explained by the demand for farmers to rid an entire winter's worth of waste from the lagoons in March. In April, hydrological conditions changed that favored distribution of swine waste-derived nitrogen well downstream. Local rainfall was elevated; for instance, the total April rainfall at the Wilmington airport was 13.6 cm, which was 6.5 cm over the long-term April average. In addition, monthly Cape Fear River discharge at Lock and Dam #1 (Fig. 1) rose to 2170 CMS, the highest discharge of 2018 until August (2562 CMS). The presence of accumulated spray field nutrients, elevated rainfall, and high river discharge carried swine-sourced N well downstream into the estuary.

Enriched $\delta^{15}N$ signatures at BRN were shown in April (+ 21.6‰) and again in August (+ 11.2‰). Additionally, HAM showed enriched $\delta^{15}N$ signatures in March, April, and August (Figs. 2e, 6, and 7). A lack of effluent signature mid-spraying season (May through July) suggests that waste lagoons were pumped down at the beginning of the spraying season and again later in the season (note the elevated rainfall in August would lead to increased pump-out). It was interesting to note that March $\delta^{15}N$ isotopic signatures at BRN were characteristic of synthetic fertilizer at − 3.6‰, the lightest signature occurring in the data set. As a predominantly rural region, inputs could have originated from surrounding traditional agricultural farms; also note that stormwater runoff from Elizabethtown would be factor as well. The nitrogen concentrations (nitrate, ammonium, and DON) at BRN, HAM, 6RC, and NC403 increased in February. Maximum nitrate concentrations are typical in winter/early spring months in watersheds containing crop agriculture from runoff, whereas summer nitrate concentrations in CAFO areas are generally significantly higher due to waste application on sprayfields (Mallin and McIver 2018).

Previous studies have traced nutrient sources with isotopic techniques and documented limited downstream nutrient transport and (Karr et al. 2001). The present study traced the potential presence of upstream CAFO-derived nitrogen downstream from CAFO-influenced areas to estuarine station M61 at least on one occasion. As previously noted, the stream and river sites had nitrogen signatures characteristic of swine effluent in April (+ 10 to +2 0‰), and estuarine sites M61 and M18 also exhibited unusually elevated signatures ($\delta^{15}N$ + 9.9‰ and + 8.8‰, respectively) in this month of elevated river discharge (Fig. 7). A clear regulatory application from these data is that waste application from lagoons in the spraying season have shown significant month-to-month variability within the basin, which indicates that a singular month of sampling to uphold state and federal water quality regulations would not be truly suggestive of potential environmental threats.

Regulations and impact of findings

Swine-derived isotopic signatures throughout the Cape Fear River basin support the findings of previous studies that showed elevated nutrient concentrations related to CAFO-impacted streams in the Cape Fear River basin. Elevated nitrogen concentrations and isotopic signatures characteristic of swine effluent have entered surface waters throughout the Cape Fear River watershed, which under the Clean Water Act G.S. 143-215.10E deems discharge of waste in surface waters illegal. CAFO facilities that intend to discharge waste into waterways are issued NPDES permits to mitigate and control waste discharge. However, only 14 of 1222 registered CAFOs claim to discharge waste into waterways and have obtained these permits. Presence of waste-derived nutrients throughout the watershed suggests that more than 14 CAFO facilities have waste that is discharged into surrounding waterways that have not obtained permitting. Nevertheless, violations of this regulation suggest that there is an enforcement issue with NPDES permits. A solution to this is mandated NPDES permits for all facilities which will aim to alleviate nutrient pollution in waterways and adverse human health impacts. In addition, a current permit monitoring only includes sampling lagoon waste to uphold regulatory standards; however, adjacent stream quality assessments should be required with permits to ensure waste is not illegally leeching into surrounding waterways. The permitting requirements for animal waste discharges vary considerably across states and suffer from lack of required stream water sampling (Rosov et al. 2020). Finally, the two point-source influenced stations in the data set displayed both elevated DIN concentrations and enriched $\delta^{15}N$ signatures in stream sites well downstream of the facilities, demonstrating that wastewater treatment in generally rural watersheds contributes to problematic nutrient loading as well as animal waste discharges.

Conclusions

Nutrient concentrations, isotopic signatures, and correlation analyses indicated that nitrogen inputs in the Northeast Cape Fear River, Black River, and the Cape Fear River are largely derived from CAFO swine effluent, with additional contributions from human discharges. CAFO-derived nutrients from watershed tributaries, under appropriate hydrological conditions (i.e., periods of elevated rainfall during swine waste spray season), can be traced downstream as far as the Cape Fear estuary near the City of Wilmington, NC. Spatial variability was shown throughout the basin as the unimpacted blackwater and estuarine systems revealed significantly lower N concentrations and lighter isotopic waste signals than animal waste and human wastewater-impacted waterways upstream. Significant seasonal variability was identified as months with waste application to spray fields (March through August) had maximum nitrate concentrations and isotopic signatures indicative of waste effluent. Adjustments to current regulations enforcing pollution permits or monitoring of adjacent streams could aid in management and overall health of basin's waterways and the community that uses it.

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ATTACHMENT 6

Industrialized Animal Production: A Major Source of Nutrient and Microbial Pollution to Aquatic Ecosystems

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Industrialized Animal Production—A Major Source of Nutrient and Microbial Pollution to Aquatic Ecosystems

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Livestock production has undergone massive industrialization in recent decades. Nationwide, millions of swine, poultry, and cattle are raised and fed in concentrated animal feeding operations (CAFOs) owned by large, vertically integrated producer corporations. The amount of nutrients (nitrogen and phosphorus) in animal manure produced by CAFOs is enormous. For example, on the North Carolina Coastal Plain alone an estimated 124,000 metric tons of nitrogen and 29,000 metric tons of phosphorus are generated annually by livestock. CAFO wastes are largely either spread on fields as dry litter or pumped into waste lagoons and sprayed as liquid onto fields. Large amounts of nitrogen and phosphorus enter the environment through runoff, percolation into groundwater, and volatilization of ammonia. Many CAFOs are located in nutrient-sensitive watersheds where the wastes contribute to the eutrophication of streams, rivers, and estuaries. There is as yet no comprehensive Federal policy in place to protect the environment and human health from CAFO generated pollutants.

KEY WORDS: swine; poultry; nutrients; pathogens; eutrophication.

INTRODUCTION

Humans first domesticated a number of animal species in several regions of the world ca. 4–6,000 years ago (Diamond, 1997). Early domes-

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tication of animals allowed humans to exploit their abilities to convert otherwise inaccessible resources into useful products and services. Animal production was necessarily resource-limited and since production was tightly coupled to the productivity of the landscape, animal waste production would seldom have exceeded the assimilation capacity of the landscape.

In recent decades livestock production, particularly that of swine, cattle and poultry, has undergone a major change toward industrialization. The industrialization of the cattle and poultry industries began in the late 1950s while industrialization of swine production began in the 1970s (Thu & Durrenberger, 1998). Industrialization of livestock production basically consists of moving animals from pastures and lots into large buildings, where they are confined and fed throughout their lives until they are ready for market. Adoption of confined feeding techniques, together with the availability of large quantities of feedstuffs and efficient transportation systems, now allow animal producers to circumvent the ecological constraints otherwise imposed by the landscape. As a consequence, animal waste production often exceeds the assimilatory capacity of the landscape both locally and regionally.

Individual concentrated animal feeding operations (CAFOs) now house hundreds to thousands of animals in each confinement structure, and vast amounts of animal waste are generated by these facilities. Swine waste is deposited on the floor of the structures by the animals, where it is periodically washed between slats in the floor into a system of trenches and pipes beneath the buildings. From there it is conveyed outside and into a cesspit called a "waste lagoon." Some anaerobic treatment occurs in the lagoon and the liquid waste is periodically applied on surrounding fields by surface spraying, surface spreading, or in some cases subsurface injection. Crops planted on the fields, such as Bermuda grass, cotton, corn, and soy take up some of the plant nutrients in the waste material. Some poultry CAFOs utilize the lagoon system, but the majority of poultry CAFOs dispose of dry litter on the fields (Williams et al., 1999). In any case, concentrated waste material is spread onto fields, from where it can enter the environment through surface runoff or groundwater infiltration (Edwards & Daniel, 1992; Mallin, 2000). Thus, individual CAFOs represent an ecologically anomalous concentration of animals whose waste production can easily exceed the assimilatory capacity of the local landscape.

Regional concentrations of CAFOs create circumstances in which very large imbalances of waste production versus waste assimilation capacity can arise (Barker and Zublena, 1995; Jackson et al., 2000). The use of carefully formulated feeds, the need for large amounts of these feeds, and transportation cost considerations have led to the regional concentration of CAFOs around feed mills and meat packing facilities (C. Wright, personal communication). Swine CAFOs are abundant on the North Carolina Coastal Plain, and in Midwestern states such as Iowa, Minnesota, Michigan, and Indiana, and are moving into western areas such as Utah and Colorado (Thu & Durrenberger, 1998). Poultry CAFOs are abundant in Iowa, Arkansas, Georgia, Maryland, Virginia, Delaware, North Carolina, California, and Mississippi (Edwards & Daniel, 1992). Cattle CAFOs are rare on the east coast but common in Texas and several midwestern states. The environmental challenge of regional concentration has been recognized explicitly for some time, e.g., in legislation introduced by Sen. Harkin (D-Iowa) in 1997 (the Animal Agriculture Reform Act, S.B. 1223). Sen. Harkin (1997) cited the Department of Agriculture as reporting: "The continued intensification of animal production systems without regard to the adequacy of the available land base for manure recycling presents a serious policy problem."

CAFOs have also had many acute pollution problems with their waste disposal systems, including lagoon ruptures and major leaks caused by mismanagement or weather (Mallin, 2000). For example, 25 million gallons of liquid swine waste entered North Carolina's New River and its estuary following a waste lagoon rupture in 1995, polluting 22 miles of the river and much of the upper estuary. The pollution load caused freshwater and estuarine fish kills and algal blooms, and polluted the river and its sediments with fecal bacteria for months (Burkholder et al., 1997). That same year a poultry lagoon breach and a large swine waste lagoon leak also caused algal blooms, fish kills, and microbial contamination in North Carolina's Cape Fear River basin (Mallin et al., 1997). In all of these cases large quantities of nutrients (nitrogen and phosphorus) entered downstream water bodies from the CAFO sites. Major CAFO accidents have also occurred in Iowa, Maryland, and Missouri (Thu & Durrenberger, 1998; Mallin, 2000). While the acute pollution caused by CAFOs is well documented, the sheer magnitude of their distribution and abundance merits an examination of the chronic effects that these facilities may have on our water resources.

North Carolina presents an excellent example of the effects of rapidly increasing industrialized livestock production, particularly that of swine. Industrialization of North Carolina's swine production began in the 1980s, and continued rapidly until the mid to late 1990s (Burkholder et al., 1997). The lagoon waste disposal system was deployed with little foresight for the environmental consequences, and CAFOs were constructed with little regulation until lagoon construction standards, siting regulations, and waste management plans were legally required in 1993 (Burkholder et al., 1997).

A moratorium on new CAFO production was begun in 1997; however, this did not take full effect until nearly 10,000,000 head of swine were present in eastern North Carolina, the vast majority in CAFOs (Burkholder et al., 1997; Mallin, 2000).

This large number of swine (currently exceeding the North Carolina human population of 7,900,000), as well as poultry and cattle, requires vast amounts of animal feed, which contains nitrogen (N) and phosphorus (P), nutrients that can lead to the eutrophication of water bodies (Carpenter et al., 1998; Correll, 1998; Cahoon et al., 1999; Glasgow & Burkholder, 2000; Mallin, 2000). Cahoon et al. (1999) noted that as of 1995 the animal production industry in North Carolina's Cape Fear River basin produced some 82,700 metric tons of N and 26,000 metric tons of P as waste in this watershed. Glasgow and Burkholder (2000) computed that in 1998 North Carolina's Neuse River watershed received 41,000 metric tons of N and 16,000 metric tons of P from CAFOs in that basin. Since the vast majority of feed for swine and poultry is shipped into these watersheds from midwestern states (Thu & Durrenberger, 1998; Cahoon et al., 1999), most of the nutrients added to the watershed through animal manures are considered "new" nutrients, imported into the system rather than recycled within it. The purpose of this paper is to describe the magnitude of industrialized animal production in a large region of the North Carolina Coastal Plain (see Figure 1), assess the potential contribution of nutrients and microbial pollution to this region, and describe the realized and potential effects of this pollutant load.

METHODS

An assessment of animal waste contributions to pollutant loads on the North Carolina Coastal Plain required computation of livestock numbers by animal category in the region, and estimates of the amount of N, P, and bacteria excreted by each species of livestock on an annual basis. The Coastal Plain contains over 90% of the State's swine population, the vast majority of its turkeys, and about 30% of the chicken population. For each of the 38 counties in the region, the most recent available data on annual production of several types of livestock (swine, broiler chickens, other chickens, turkeys, and cattle) were obtained from the website of the North Carolina Department of Agriculture (NCDA, http://www.agr.state.nc.us/stats/cntysumm). On an annual basis, there are approximately 2.9 turkey generations (cohorts) and 6.5 broiler chicken generations produced. Thus, the turkey and broiler production figures provided on the NCDA website for each

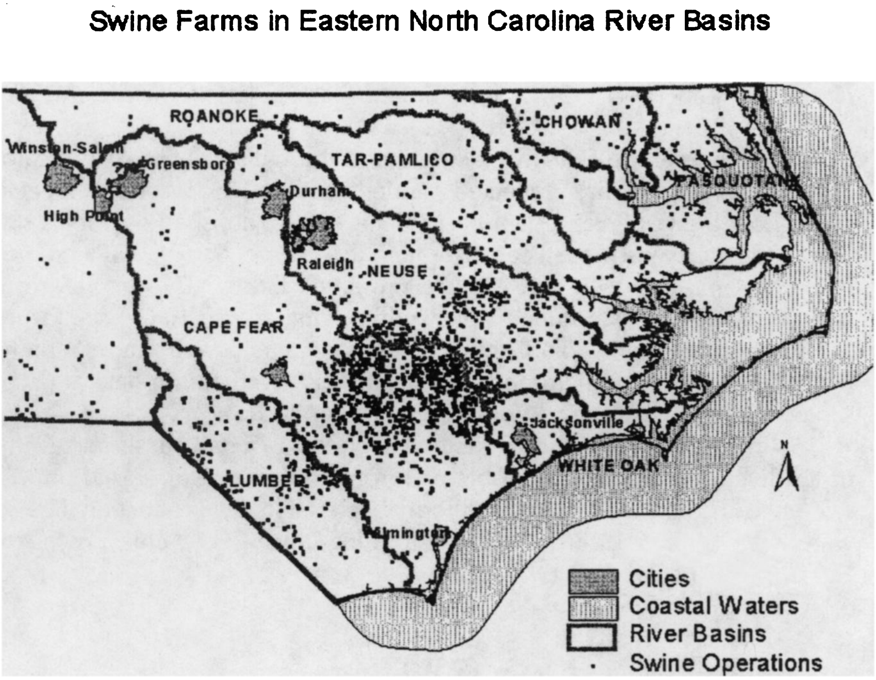

FIGURE 1. Location of swine CAFOs (operations with 250 or more head) on the North Carolina Coastal Plain by river basin.

county were divided by these numbers to yield average annual standing stock (total animals present at any one time), and subsequent annual manure production.

Animal waste N and P production rates were calculated using recent published information or data from industry sources. Swine waste N and P contents were calculated using data supplied by T. van Kempen (North Carolina State University): 15.9 kg N/yr and 5.3 kg P/yr for sows, 11.1 kg N/yr and 2.3 kg P/yr for grower-finisher pigs, and are similar to those reported elsewhere, e.g., Powers and Van Horn (1998). Total swine N and P excretion rates were then calculated using the proportion of sows and grower-finisher pigs (0.103 and 0.897, respectively in 1998 (NCDA, 1999). Turkey and broiler chicken N excretion were calculated using data from Powers and Van Horn (1998); they report N excretion as 0.395 kg N/turkey produced and 0.017 kg N/broiler produced. Using N:P ratios of 3.57:1 for turkey waste and 3.23:1 for broiler chicken waste (NRCS, 1996, Chapter

4), P excretion was calculated as 0.11 kg P/turkey produced and 0.0053 kg P/broiler produced. Annual N and P excretion rates for cattle were calculated as in Cahoon (1999), using estimates of 46.8 kg N/cow and 11.7 kg P/cow.

The Lower Cape Fear River Program at the University of North Carolina at Wilmington has collected nutrient data at 35 locations located throughout the Cape Fear River basin since 1995. Published data for a station in the Northeast Cape Fear River near the town of Sarecta (GPS coordinates N34 43.365, W77 51.752) are presented below. These data are of interest because of that station's proximity to numerous CAFOs (see Figure 1). Since ammonium volatilization is most active during warm months (NCDAQ, 1997), summertime (May–September) ammonium data are presented for a six-year period from 1996 through 2001.

Estimates of fecal coliform bacteria excreted on a daily basis for several of the livestock species were obtained from Sobsey (1996). Based on this reference the following fecal coliform bacterial daily production figures were used for pigs (1.2×10^{10} colony-forming units (CFU)), chickens (1.4×10^{8} CFU), and cows (6.0×10^{9} CFU).

RESULTS

The North Carolina Coastal Plain produces large numbers of swine, broiler chickens, and turkeys, and smaller but significant numbers of other chickens and cattle (Table 1). Swine production in North Carolina is second

TABLE 1

Population of Livestock by Category on the North Carolina Coastal Plain, 2000–2001 (About 6.5 generations of broilers and 2.9 turkey generations are produced per year. Dividing broiler chicken and turkey production by these factors provides standing stock, or numbers present at any one time on the Coastal Plain.)

Animal Category	Numbers Used in Nutrient Calculations
Swine	8,700,000 (standing stock)
Broiler chickens	210,000,000 (produced)
Other Chickens	3,480,000 (produced)
Turkeys	31,800,000 (produced)
Cattle	149,000 (standing tock)

in the United States only to Iowa (Burkholder et al., 1997; USNASS, 1997). North Carolina ranks fourth in the United States in broiler chickens sold, and first in the United States in turkeys sold (USNASS, 1997). The vast majority of the swine and poultry are in CAFOs, whereas many of the cattle are grazed on open lands.

Our computations show that swine and turkey production contribute the greatest amount of N and P in the annual waste stream (Table 2). Swine alone generate 101,000 metric tons of N and turkeys 12,600 metric tons. Swine also generate 22,700 tons of P and turkeys 3,500 metric tons. Thus, swine are by far the largest producers of nutrients in comparison with other livestock on the Coastal Plain, and the manner of their waste disposition deserves attention. Swine waste from CAFOs is invariably pumped into lagoons, some of which are located on river floodplains. In North Carolina liquid waste from the lagoons is typically then sprayed out on adjoining fields, from which surface drainage to waterways or subsurface drainage to groundwaters can occur. The nutrients produced by poultry CAFOs as manure are largely spread as dry litter on fields, with some pumped into waste lagoons, from which they are sprayed as liquid waste onto fields. Secondary treatment of livestock waste for nutrient removal is seldom practiced.

This analysis does not take into account nutrients produced by the decomposition of dead animals. Following Hurricane Floyd in October 1999, the news media published numerous photographs of drowned swine and poultry from CAFOs in areas inundated by floodwaters. The numbers of drowned livestock may have been very large, as Wing et al. (2002) determined that 241 CAFOs were within the geographical coordinates of the areas inundated by post-Floyd floodwaters according to satellite imagery.

TABLE 2

Estimated Amounts of Nitrogen and Phosphorus (metric tons) Excreted Annually by Various Livestock Categories on the North Carolina Coastal Plain, 2000–2001

Animal Category	Nitrogen	Phosphorus
Swine	101,000	22,700
Broiler chickens	3,570	1,110
Other Chickens	60	20
Turkeys	12,600	3,500
Cattle	7,000	1,750
Grand Total	124,230	29,080

N.C. Department of Agriculture statistics report over 1 million swine mortalities per year as of 1998, not counting piglets lost (N.C. D.A., 1999); thus, animal carcasses are likely another significant source of nutrients to the environment.

Data published by the Lower Cape Fear River Program (available at the website http://www.uncwil.edu/cmsr/aquaticecology/laboratory/lcfrp) demonstrate that there was a statistically significant increase in ammonium levels at a Northeast Cape Fear River station (Sarecta) during the period 1996–2001 (see Figure 2). Ammonium comprises the largest portion of total N in swine and poultry liquid waste (Burkholder et al., 1997; Mallin et al., 1997; Williams et al., 1999). Along with transport of ammonium in runoff or subsoil movement, it can be volatilized and transported in the gaseous ammonia form (Edwards & Daniel, 1992; Williams et al., 1999; Mallin, 2000). The station at Sarecta has 344 swine CAFOs within a 20 km radius, and 587 swine CAFOs within a 30 km radius (we have no data on poultry CAFOs). This station likely receives ammonium inputs from overland runoff and lateral groundwater flow, and airborne deposition. The implications of nutrient increases to downstream waters are discussed below.

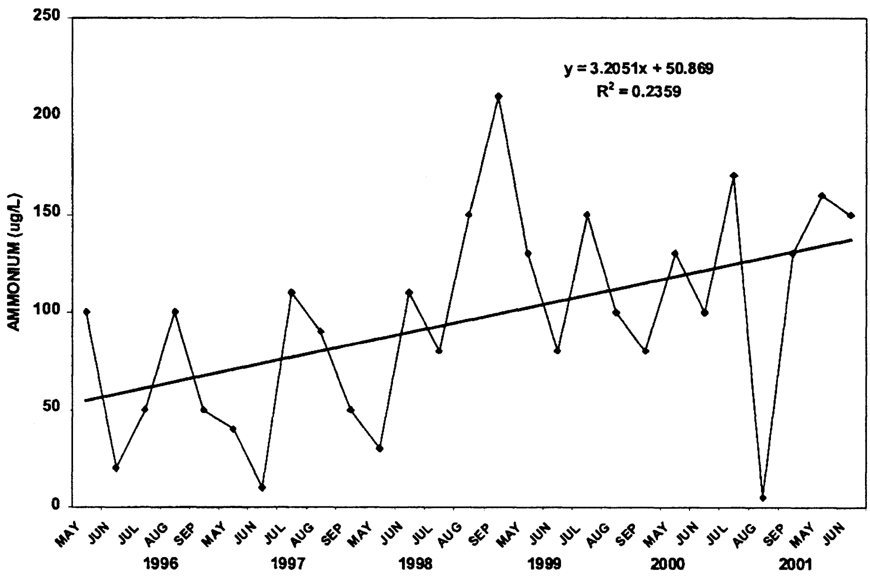

FIGURE 2. Summer ammonium concentrations at Sarecta, a water quality station on the Northeast Cape Fear River in a location near numerous CAFOs, data from 1996 to 2001.

Applying Sobsey's (1996) conversion factors figures to livestock populations on North Carolina's Coastal Plain yields estimated annual excretion of fecal coliform bacteria of 3.8×10^{18} from swine, 1.7×10^{18} CFU from broilers, 1.8×10^{17} from other chickens, and 3.3×10^{17} from cattle.

DISCUSSION

Fate of Excreted Nutrients

As mentioned earlier, major storms and accidents are documented mechanisms by which large amounts of nutrients have been abruptly transported from CAFOs to receiving waters (Burkholder et al., 1997; Mallin et al., 1997; 1999; Mallin, 2000). However, CAFOs also chronically export nutrients to water resources through several means. Normal rain events carry nutrients from swine sprayfields to nearby streams through surface and subsurface runoff (Evans et al., 1984; Westerman et al., 1987) where these inputs have caused stream nitrate-N to rise above 5 mg N/L and P above 1 mg P/L (Stone et al., 1995; Gilliam et al., 1996). Nutrients, mainly nitrate and ammonium, also leach downwards into groundwater from animal waste lagoons, sprayfields, and litter fields. In a set of 11 North Carolina swine lagoons, Huffman and Westerman (1995) found average inorganic (ammonium and nitrate) N concentrations of 143 mg/L in nearby groundwater, and found that through leakage the lagoons exported on average 4.7 kg N/day to groundwater. Also in North Carolina Westerman et al. (1995) found average concentrations of ammonium in downslope well fields that exceeded 50 mg N/L, compared with upslope wells that were less than 1 mg N/L. The nitrate form of N is especially mobile in soils and can pass readily through soils to contaminate groundwater. Liebhardt et al. (1979) found high levels of nitrate in soil groundwater beneath Delaware cornfields where poultry waste was applied as the sole fertilizer, with evidence that the nitrate moved laterally toward a nearby stream. Using nitrogen isotopic techniques Karr et al., (2001) have traced nitrate generated from swine waste spray fields through shallow groundwater into receiving stream waters, and at least 1.5 km downstream. Phosphorus is much less mobile, and binds readily to soil particles. However, when the P content of soils is built up dramatically through excessive manure application, both surface export and subsurface loss of P occurs (Sharpley et al., 1999).

Anaerobic treatment of swine wastes with high concentrations of organic N promotes deamination, resulting in high concentrations of ammonium-N in lagoon liquid. Liming is used to maintain a pH above about 7, favoring ammonia formation. Ammonia volatilizes from sprayfields and waste lagoons, and is transported downwind (McCulloch et al., 1998; Aneja et al., 2000; Walker et al., 2000). The North Carolina Department of Air Quality estimates that 70–80% of all swine waste N and a somewhat lesser percentage of poultry waste N is thus volatilized (N.C. D.A.Q., 1997). It is notable that the Neuse River watershed, which contains approximately 25% of North Carolina's swine population and numerous poultry production facilities and is downwind of a large concentration of CAFOs in the Cape Fear watershed, registered a 14% increase in total N and a 34% increase in nitrate over the seven year period 1990–1997 (Glasgow & Burkholder, 2000). While other anthropogenic sources of N undoubtedly contributed to this loading, the large recent rise in CAFOs in those watersheds would suggest that animal production is a significant cause of these nutrient inputs. Walker et al. (2000) and Mallin (2000) have documented a trend of increasing ammonium deposition in the coastal region of North Carolina, which they attribute to animal production sources. At Sarecta on the Northeast Cape Fear River a steady rise in river ammonium concentrations from 1996–2001 is evident (see Figure 2). There are no new or large wastewater treatment facilities in that area that can account for this increase. The single major land use change in that area has been the rapid proliferation of CAFOs during the 1980s and 1990s (see Figure 1).

Potential Impacts on Water Resources

Kellogg (2000) prioritized U.S. watersheds in terms of vulnerability to manure nutrient contamination based on a number of factors, including soil percolation, soil runoff potential, soil erosion potential, and amount of animal nutrients applied to soils. Much of the North Carolina Coastal Plain, especially the Albemarle-Pamlico and Cape Fear watersheds, ranked highest in the nation in vulnerability. Many of the surface water supplies downstream of CAFO-dense areas on the North Carolina Coastal Plain (Figure 1) are sensitive to N and/or P loading, and will respond by formation of algal blooms (Rudek et al., 1991; Paerl et al., 1990; Glasgow & Burkholder, 2000). This is especially true in the Neuse, Pamlico, and New Rivers and their estuaries (Dame et al., 2000; Mallin et al., 2000). Algal blooms can build up high concentrations of biomass, and eventually die and become a source of labile organic material. Bacteria feed on this biomass and multiply, creating high biochemical oxygen demand (BOD) that will at times lower dissolved oxygen concentrations to levels that can kill sessile bottom organisms and create areas in which finfish cannot survive—a loss of usable habitat. Another impact of increased nutrient loading on estuaries is to stimulate growth of the toxic dinoflagellates *Pfiesteria piscicida* and *P. shumwayae*, which have bloomed downstream of CAFO areas in the Neuse, Pamlico, and New River Estuaries of North Carolina and the Chesapeake Bay in Maryland (Burkholder et al., 1995; Burkholder & Glasgow, 1997; Glasgow et al., 2001). Growth of *P. piscicida* is more stimulated by P loading whereas *P. shumwayae* appears to be more stimulated by N inputs (Glasgow et al., 2001). Both species of *Pfiesteria* have caused many fish kills in North Carolina and some in Maryland, as well as human health problems to researchers and watermen exposed to its toxins (Burkholder et al., 1995; Burkholder & Glasgow, 1997; Burkholder & Glasgow, 2001). Blooms of these organisms and consequent fish kills have led to closures of areas in the Chesapeake Bay region and the Albemarle-Pamlico estuarine region in North Carolina to commercial fishing, due to health concerns over the consumption of affected fish and exposure to airborne *Pfiesteria* toxins when on the water (Burkholder & Glasgow, 2001).

In the Cape Fear River basin, which produces 50% of North Carolina's swine and vast numbers of poultry, most of the CAFOs are in watersheds drained by blackwater streams. These are streams that drain lowland forests and riverine swamps, and in pristine condition are naturally nutrient poor. Recent experiments have been conducted on the response of blackwater streams to increased nutrient loading (Mallin et al., 2001). These experiments showed that N inputs of 1 mg/L led to spring and summer algal blooms in test waters, while P levels of 1 mg/L caused significant production of heterotrophic microbes and increased biochemical oxygen demand (Mallin, 2000; Mallin et al., 2000; Mallin et al., 2001). Since recent assessments (Figure 2) show a steady increase in ammonium in certain downstream locations in the Cape Fear basin, this loading has the potential for degrading water quality in areas receiving nutrient inputs.

Seagrass beds are an important coastal habitat for many species of finfish and shellfish. Historically, important seagrass habitat has been located downstream of CAFO-rich areas in the Albemarle-Pamlico estuarine system in North Carolina as well as the Chesapeake Bay. Much of that habitat disappeared in the mid-to-late 1900s. A number of factors can cause losses of seagrass, including reduced photosynthesis from increased turbidity (Dennison et al, 1993). However, the most important seagrass species on the mid-Atlantic seaboard (eelgrass—*Zostera marina*) has been shown to be sensitive to nitrate loading, and can die under prolonged exposure to nitrate concentrations of 50 to 100 μg N/L or higher (Burkholder et al., 1992; Burkholder et al., 1994). Some coastal North Carolina waters can periodically receive extended inputs of nitrate from upstream freshwater sources that exceed these critical levels (Mallin et al., 1993; Paerl et al., 1995; Mallin et al., 1999; Glasgow & Burkholder, 2000) thus providing a habitat stressful to eelgrass survival or re-establishment.

Animal Pathogens and Humans

Livestock are known to excrete many of the same pathogenic bacteria, viruses, and protozoans that can afflict humans. These organisms include pathogenic bacteria such as *Escherichia coli*, *Salmonella* spp., and *Streptococcus* spp., pathogenic protozoans such as *Giardia lamblia* and *Cryptosporidium parvum*, and a number of viruses (Mawdsley et al., 1995). The way animal waste is treated will affect pathogen survival and potential transmission to humans. Composting of manure raises temperatures high enough to kill most microbes, but animal waste slurries do not reach lethal temperatures (Mawdsley et al., 1995). Microbes in animal waste slurries such as lagoon liquid can survive for extended periods; *E. coli* has been known to survive up to 11 weeks in such an environment (Mawdsley et al., 1995). If waste is applied to the land surface survival time is cut to a matter of days, particularly under conditions of bright sunlight (Crane et al., 1983; Mawdsley et al., 1995). However, rain events occurring shortly after animal waste is surface-applied to fields cause vertical and horizontal movement of microbes to nearby water bodies (Crane et al., 1983; Mawdsley et al., 1995; Mallin, 2000). Large-scale microbial disease outbreaks have been traced to livestock vectors. In 1999 and 2000 the news media reported incidents in Albany, New York (MMWR 1999) and Walkerton, Ontario of mass illnesses and some deaths to humans that were exposed to pathogenic *E. coli* in water sources contaminated by runoff from cattle husbandry areas.

As indicated above, livestock on the Coastal Plain excrete large amounts of fecal bacteria in manure. Unlike human waste, microbes generated by CAFOs are not exposed to secondary treatment or chlorination to disinfect the material. When applied to fields in manure the vast majority of these microbes are likely deactivated by ultraviolet radiation, microbial competition and predation, or other means (Crane et al., 1983). However, because of the sheer volume of microbes deposited, there still remains a significant pollution potential from this material entering surface or groundwaters that humans will contact. If CAFO-generated microbes enter the sediments of water bodies, organisms such as *E. coli* can find a favorable environment where they can remain viable for over two months (Davies et al., 1995). For example, following a large swine waste lagoon spill in the New River, North Carolina, Burkholder et al. (1997) found fecal coliform bacterial counts ranging from 1,000,000 to 3,000,000 per 100 ml of river water several km downstream from the spill site. These very high concentrations declined to the range of 1,000 to 5,000 per 100 ml after 14 days, and to less than 1,000 per 100 ml in 61 days. However, further sampling indicated that the river sediments maintained concentrations of fecal bacteria up to 5,000 per 100 ml for 61 days. The risk of large quantities of fecal microbes entering the environment is thus high following acute CAFO mishaps; although the risk of human exposure to these microbes chronically through normal operations is yet undetermined.

Regulation

Point source discharges from municipal or industrial wastewater treatment plants are regulated under the National Pollution Discharge Elimination System (NPDES) enacted by the National Environmental Policy Act of 1971 (NEPA). This process authorizes the US Environmental Protection Agency or individual states to license and inspect dischargers, and set maximum pollutant discharge concentrations. However, CAFOs have been considered to be non-point source dischargers, and were thus exempt from this process. As such, regulation of pollutant discharges from them has been piecemeal and varies from state to state. Current legislated and regulatory controls on the environmental effects of CAFOs have generally followed demonstration of negative environmental impacts, rather than preventing them, e.g., Maryland's Water Quality Improvement Act of 1998. Laws and regulations in many states define CAFOs as farms and tacitly assume that CAFOs manage nutrients and other wastes as do conventional farms, when in fact CAFO operations depart significantly from the ecological relationships that control farm productivity (Jackson et al., 2000). Moreover, most laws and regulations address CAFOs as individual operations, thus neglecting the considerable effects of concentration of many CAFOs in relatively small regions.

Although some Federal legislators have shown concern for the environmental impacts of CAFOs (Harkin, 1997), comprehensive legislation designed to regulate CAFO-generated pollution has not yet occurred on the Federal level. Federal regulations have only recently recognized the need to limit P over-application in animal wastes; the U. S. Department of Agriculture's Natural Resource Conservation Service mandates soil P management in its most recent version of the Nutrient Management Standard 590 (Sharpley & Tunney, 2000). Implementation by the states is not uniform, however, as they utilize different soil test procedures, different risk assessment methods, and different remediation responses. North Carolina has just developed a Phosphorus Loss Assessment Tool (PLAT), which has not yet been fully implemented. However, these new regulations address only one aspect of the larger set of environmental challenges posed by CAFOs, and fail to address the consequences of regional concentration of CAFOs at all. Consequently, CAFOs present a major challenge to the current system of environmental law and regulations in the United States.

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ATTACHMENT 7

Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality

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Waste from agricultural livestock operations has been a long-standing concern with respect to contamination of water resources, particularly in terms of nutrient pollution. However, the recent growth of concentrated animal feeding operations (CAFOs) presents a greater risk to water quality because of both the increased volume of waste and to contaminants that may be present (e.g., antibiotics and other veterinary drugs) that may have both environmental and public health importance. Based on available data, generally accepted livestock waste management practices do not adequately or effectively protect water resources from contamination with excessive nutrients, microbial pathogens, and pharmaceuticals present in the waste. Impacts on surface water sources and wildlife have been documented in many agricultural areas in the United States. Potential impacts on human and environmental health from long-term inadvertent exposure to water contaminated with pharmaceuticals and other compounds are a growing public concern. This workgroup, which is part of the Conference on Environmental Health Impacts of Concentrated Animal Feeding Operations: Anticipating Hazards—Searching for Solutions, identified needs for rigorous ecosystem monitoring in the vicinity of CAFOs and for improved characterization of major toxicants affecting the environment and human health. Last, there is a need to promote and enforce best practices to minimize inputs of nutrients and toxicants from CAFOs into freshwater and marine ecosystems. *Key words:* ecology, human health, poultry, swine, water contaminants, wildlife. *Environ Health Perspect* 115:308–312 (2007). doi:10.1289/ehp.8839 available via *http://dx.doi.org/* [Online 14 November 2006]

Background and Recent Developments

Concentrated animal feed operations and water quality. Animal cultivation in the United States produces 133 million tons of manure per year (on a dry weight basis) representing 13-fold more solid waste than human sanitary waste production [U.S. Environmental Protection Agency (U.S. EPA) 1998]. Since the 1950s (poultry) and the 1970s–1980s (cattle, swine), most animals are now produced for human consumption in concentrated animal feeding operations (CAFOs). In these industrialized operations, the animals are held throughout their lives at high densities in indoor stalls until they are transported to processing plants for slaughter. There is substantial documentation of major, ongoing impacts on aquatic resources from CAFOs, but many gaps in understanding remain.

Contaminants detected in waste and risk of water contamination. Contaminants from animal wastes can enter the environment through pathways such as through leakage from poorly constructed manure lagoons, or during major precipitation events resulting in either overflow of lagoons and runoff from recent applications of waste to farm fields, or atmospheric deposition followed by dry or wet fallout (Aneja 2003). The magnitude and direction of transport depend on factors such as soil properties, contaminant properties, hydraulic loading characteristics, and crop management practices (Huddleston 1996). Many contaminants are present in livestock wastes, including nutrients (Jongbloed and Lenis 1998), pathogens (Gerba and Smith 2005; Schets et al. 2005), veterinary pharmaceuticals (Boxall et al. 2003; Campagnolo et al. 2002; Meyer 2004), heavy metals [especially zinc and copper; e.g., Barker and Zublena (1995); University of Iowa and Iowa State Study Group (2002)], and naturally excreted hormones (Hanselman et al. 2003; Raman et al. 2004). Antibiotics are used extensively not only to treat or prevent microbial infection in animals (Kummerer 2004), but are also commonly used to promote more rapid growth in livestock (Cromwell 2002; Gaskins et al. 2002; Liu et al. 2005). In addition, pesticides such as dithiocarbamates are applied to sprayfields (Extension Toxicology Network 2003). Although anaerobic digestion of wastes in surface storage lagoons can effectively reduce or destroy many pathogens, substantial remaining densities of microbial pathogens in waste spills and seepage can contaminate receiving surface- and groundwaters (e.g., Burkholder et al. 1997; Mallin 2000). Pharmaceuticals can remain present as parent compounds or degradates in manure and leachates even during prolonged storage. Improper disposal of animal carcasses and abandoned livestock facilities can also contribute to water quality problems. Siting of livestock operations in areas prone to flooding or where there is a shallow water table increases the potential for environmental contamination.

The nutrient content of the wastes can be a desirable factor for land application as fertilizer for row crops, but overapplication of livestock wastes can overload soils with both macronutrients such as nitrogen (N) and phosphorous (P), and heavy metals added to feed as micronutrients (e.g., Barker and Zublena 1995). Overapplication of animal wastes or application of animal wastes to saturated soils can also cause contaminants to move into receiving waters through runoff and to leach through permeable soils to vulnerable aquifers. Importantly, this may happen even at recommended application rates. As examples, Westerman et al. (1995) found 3–6 mg nitrate (NO_3)/L in surface runoff from sprayfields that received swine effluent at recommended rates; Stone et al. (1995) measured 6–8 mg total inorganic N/L and 0.7–1.3 mg P/L in a stream adjacent to swine effluent sprayfields. Evans et al. (1984) reported 7–30 mg NO_3/L in subsurface flow draining a sprayfield for swine wastes, applied at recommended rates. Ham and DeSutter (2000) described export rates of up to 0.52 kg ammonium m^{-2} $year^{-1}$ from lagoon seepage; Huffman and Westerman (1995) reported that groundwater near swine waste lagoons averaged 143 mg inorganic N/L, and estimated export rates at 4.5 kg inorganic N/day. Thus, nutrient losses into receiving waters can be excessive relative to levels (~ 100–200 μg inorganic N or P/L)

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known to support noxious algal blooms (Mallin 2000). In addition to contaminant chemical properties, soil properties and climatic conditions can affect transport of contaminants. For example, sandy, well-drained soils are most vulnerable to transport of nutrients to underlying groundwater (Mueller et al. 1995). Nutrients can also readily move through soils under wet conditions (McGechan et al. 2005).

Presence of contaminants in water sources. The presence of many contaminants from livestock waste has been documented in both surface water and groundwater supplies in agricultural areas within the United States (e.g., Campagnolo et al. 2002; Kolpin et al. 2002; Meyer 2004). Urban wastewater streams also contain these contaminants, and efforts to accurately determine sources of contamination are under way (Barnes et al. 2004; Cordy et al. 2004; Kolpin DW, unpublished data). The U.S. Geological Survey (USGS) began pilot surveillance programs for organic wastewater contaminants in 1999 and expanded that effort to a national scale over the past 5 years (Kolpin et al. 2002). Recent USGS efforts have focused specifically on water quality in agricultural locations (Kolpin DW, unpublished data). Nutrient levels have been detected in high parts per million (milligrams per liter) levels; pharmaceuticals and other compounds are generally measured in low levels (ppb [micrograms per liter]). In Europe, surveillance efforts conducted in Germany documented the presence of veterinary pharmaceuticals in water resources (Hirsch et al. 1999).

Animal wastes are also rich in organics and high in biochemical oxygen-demanding materials (BOD); for example, treated human sewage contains 20–60 mg BOD/L, raw sewage contains 300–400 mg BOD/L, and swine waste slurry contains 20,000–30,000 mg BOD/L (Webb and Archer 1994). Animal wastes also carry parasites, viruses, and bacteria as high as 1 billion/g (U.S. EPA 1998). Swine wastes contain > 100 microbial pathogens that can cause human illness and disease [see review in Burkholder et al. (1997)]. About one-third of the antibiotics used in the United States each year is routinely added to animal feed to increase growth (Mellon et al. 2001). This practice is promoting increased antibiotic resistance among the microbial populations present and, potentially, increased resistance of naturally occurring pathogens in surface waters that receive a portion of the wastes.

Contaminant impacts. Some contaminants pose risks for adverse health impacts in wildlife or humans. The effects of numerous waterborne pathogens on humans are well known, although little is known about potential impacts of such microorganisms on aquatic life. With respect to nutrients, excessive phosphorus levels can contribute to algal blooms and cyanobacterial growth in surface waters used for recreation and as sources of drinking water. Research is beginning to investigate the environmental effects, including endocrine disruption and antibiotic resistance issues (Burnison et al. 2003; Delepee et al. 2004; Fernandez et al. 2004; Halling-Sorensen et al. 2003; Sengelov et al. 2003; Soto et al. 2004; Wollenberger et al. 2000). However, knowledge is limited in several crucial areas. These areas include information on metabolites or environmental degradates of some parent compounds; the environmental persistence, fate, and transport and toxicity of metabolites or degradates (Boxall et al. 2004); the potential synergistic effects of various mixtures of contaminants on target organisms (Sumpter and Johnson 2005); and the potential transport and effects from natural and synthetic hormones (Hanselman et al. 2003; Soto et al. 2004). Further, limited monitoring has been conducted of ecosystem health in proximity to CAFOs, including monitoring the effects on habitats from lagoon spills during catastrophic flooding (Burkholder et al. 1997; Mallin et al. 1997; Mallin et al. 2000).

Ecologic and wildlife impacts. Anoxic conditions and extremely high concentrations of ammonium, total phosphorus, suspended solids, and fecal coliform bacteria throughout the water column for approximately 30 km downstream from the point of entry have been documented as impacts of waste effluent spills from CAFOs (Burkholder et al. 1997; Mallin et al. 2000). Pathogenic microorganisms such as *Clostridium perfringens* have been documented at high densities in receiving surface waters following CAFO waste spills (Burkholder et al. 1997). These degraded conditions, especially the associated hypoxia/anoxia and high ammonia, have caused major kills of freshwater fish of all species in the affected areas, from minnows and gar to largemouth bass, and estuarine fish, including striped bass and flounder (Burkholder et al. 1997). Waste effluent spills also stimulated blooms of toxic and noxious algae. In freshwaters, these blooms include toxic and noxious cyanobacteria while in estuaries, harmful haptophytes and toxic dinoflagellates arise. Most states monitor only water-column fecal coliform densities to assess whether waterways are safe for human contact. World Health Organization (WHO) guidelines for cyanobacteria in recreational water are 20,000 cyanobacterial cells/mL, which indicates low probability of adverse health effects, and 100,000 cyanobacterial cells/mL, which indicates moderate probability of adverse health effects (WHO 2003). Yet fecal bacteria and other pathogenic microorganisms typically settle out to the sediments where they can thrive at high densities for weeks to months following CAFO waste effluent spills (Burkholder et al. 1997).

The impacts from CAFO pollutant loadings to direct runoff are more substantial after such major effluent spills or when CAFOs are flooded and in direct contact with surface waters (Wing et al. 2002). Although the acute impacts are often clearly visible—dead fish floating on the water surface, or algal overgrowth and rotting biomass—the chronic, insidious, long-term impacts of commonly accepted practices of CAFO waste management on receiving aquatic ecosystems are also significant (U.S. EPA 1998). One purpose of manure storage basins is to reduce the N content of the manure through volatilization of ammonia and other N-containing molecules. Many studies have shown, for example, that high nutrient concentrations (e.g., ammonia from swine CAFOs, or ammonia oxidized to NO_3, or phosphorus from poultry CAFOs) commonly move off-site to contaminate the overlying air and/or adjacent surface and subsurface waters (Aneja et al. 2003; Evans et al. 1984; Sharpe and Harper 1997; Sharpley and Moyer 2000; Stone et al. 1995; U.S. EPA 1998; Webb and Archer 1994; Westerman et al. 1995; Zahn et al. 1997). Inorganic N forms are added to the atmosphere during spray practices, and both ammonia and phosphate can also adsorb to fine particles (dust) that can be airborne. The atmospheric depositions are noteworthy, considering that a significant proportion of the total ammonium from uncovered swine effluent lagoons and effluent spraying (an accepted practice in some states) reenters surface waters as local precipitation or through dry fallout (Aneja et al. 2003; U.S. EPA 1998, 2000). The contributed nutrient concentrations from the effluent greatly exceed the minimal levels that have been shown to promote noxious algal blooms (Mallin 2000) and depress the growth of desirable aquatic habitat species (Burkholder et al. 1992). The resulting chronically degraded conditions of nutrient overenrichment, while not as extreme as during a major waste spill, stimulate algal blooms and long-term shifts in phytoplankton community structure from desirable species (e.g., diatoms) to noxious species.

A summary of the findings from a national workshop on environmental impacts of CAFOs a decade ago stated that there was "a surprising lack of information about environmental impacts of CAFOs to adjacent lands and receiving waters" (Thu K, Donham K, unpublished data). Although the knowledge base has expanded since that time, especially regarding adverse effects of inorganic N and P overenrichment and anoxia, impacts of many CAFO pollutants on receiving aquatic ecosystems remain poorly understood. As examples, there is poor understanding of the impacts of fecal bacteria and other microbial pathogens from CAFO waste effluent contamination on aquatic communities; impacts of antibiotic-resistant bacteria created from CAFO wastes on aquatic life; impacts of organic nutrient forms preferred by certain noxious plankton; impacts from the contributed pesticides and heavy metals; and impacts from these pollutants acting in concert, additively or synergistically. This lack of information represents a critical gap in our present ability to assess the full extent of CAFO impacts on aquatic natural resources.

Despite their widespread use, antibiotics have only recently received attention as environmental contaminants. Most antibiotics are designed to be quickly excreted from the treated organism. Thus, it is not surprising that antibiotics are commonly found in human and animal waste (Christian et al. 2003; Dietze et al. 2005; Glassmeyer et al. 2005; Meyer 2004) and in water resources affected by sources of waste (Glassmeyer et al. 2005; Kolpin et al. 2002). Although some research has been conducted on the environmental effects from antibiotics (e.g., Brain et al. 2005; Jensen et al. 2003), much is yet to be understood pertaining to long-term exposures to low levels of antibiotics (both individually and as part of complex mixtures of organic contaminants in the environment). The greatest risks appear to be related to antibiotic resistance (Khachatourians 1998; Kummerer 2004) and natural ecosystem functions such as soil microbial activity and bacterial denitrification (Costanzo et al. 2005; Thiele-Bruhn and Beck 2005).

Human health impacts. Exposure to waterborne contaminants can result from both recreational use of affected surface water and from ingestion of drinking water derived from either contaminated surface water or groundwater. High-risk populations are generally the very young, the elderly, pregnant women, and immunocompromised individuals. Recreational exposures and illnesses include accidental ingestion of contaminated water that may result in diarrhea or other gastrointestinal tract distress from waterborne pathogens, and dermal contact during swimming that may cause skin, eye, or ear infections. Drinking water exposures to pathogens could occur in vulnerable private wells; under normal circumstances community water utilities disinfect water sufficiently before distribution to customers. Cyanobacteria (blue–green algae) in surface water can produce toxins (e.g., microcystins) that are known neurotoxins and hepatotoxins. Acute and chronic health impacts from these toxins can occur from exposures to both raw water and treated water (Carmichael et al. 2001; Rao et al. 2002). Removal of cyanotoxins during drinking water treatment is a high priority for the drinking water industry (Hitzfield et al. 2000; Rapala et al. 2002). The WHO has set a provisional drinking water guideline of 1 µg microcystin-LR/L (Chorus and Bartram 1999). While there are no drinking water standards in the United States for cyanobacteria, they are on the U.S. EPA Unregulated Contaminant Monitoring Rule List 3 (U.S. EPA 2006).

Exposure to chemical contaminants can occur in both private wells and community water supplies, and may present health risks. High nitrate levels in water used in mixing infant formula have been associated with risk for methemoglobinemia (blue-baby syndrome) in infants under 6 months of age, although other health factors such as diarrhea and respiratory disease have also been implicated (Ward et al. 2005). The U.S. EPA drinking water standard of 10 mg/L NO_3–N and the WHO guideline of 11 mg/L NO_3–N were set because of concerns about methemoglobinemia. (Note: "nitrate" refers to nitrate–nitrogen). Epidemiologic studies of noncancer health outcomes and high nitrate levels in drinking water have reported an increased risk of hyperthyroidism (Seffner 1995) from long-term exposure to levels between 11–61 mg/L (Tajtakova et al. 2006). Drinking water nitrate at levels < 10 mg/L has been associated with insulin-dependent diabetes (IDDM; Kostraba et al. 1992), whereas other studies have shown an association with IDDM at nitrate levels > 15 mg/L (Parslow et al. 1997) and > 25 mg/L (van Maanen et al. 2000). Increased risks for adverse reproductive outcomes, including central nervous system malformations (Arbuckle et al. 1988) and neural tube defects (Brender et al. 2004; Croen et al. 2001), have been reported for drinking water nitrate levels < 10 mg/L.

Anecdotal reports of reproductive effects of nitrate in drinking water include a case study of spontaneous abortions in women consuming high nitrate water (19–26 mg/L) from private wells (Morbidity and Mortality Weekly Report 1996).

While amassing experimental data suggest a role for nitrate in the formation of carcinogenic *N*-nitroso compounds, clear epidemiologic findings are lacking on the possible association of nitrate in drinking water with cancer risk. Ecologic studies have reported mixed results for cancers of the stomach, bladder, and esophagus (Barrett et al. 1998; Cantor 1997; Eicholzer and Gutzwiller 1990; Morales-Suarez-Varela et al. 1993, 1995) and non-Hodgkin lymphoma (Jensen 1982; Weisenburger 1993), positive findings for cancers of the nasopharynx (Cantor 1997), prostate (Cantor 1997), uterus (Jensen 1982; Thouez et al. 1981), and brain (Barrett et al. 1998), and negative findings for ovarian cancer (Jensen 1982; Thouez et al. 1981). Positive findings have generally been for long-term exposures at > 10 mg/L nitrate. Case–control studies have reported mixed results for stomach cancer (Cuello et al. 1976; Rademacher et al. 1992; Yang et al. 1998); positive results for non-Hodgkin lymphoma at > 4 mg/L nitrate (Ward et al. 1996) and colon cancer at > 5 mg/L (De Roos et al. 2003); and negative results for cancers of the brain (Mueller et al. 2001; Steindorf et al. 1994), bladder (Ward et al. 2003), and rectum (De Roos et al. 2003), all at < 10 mg/L. Cohort studies have reported no association between nitrate in drinking water and stomach cancer (Van Loon et al. 1998); positive associations with cancers of the bladder and ovary at long-term exposures > 2.5 mg/L (Weyer et al. 2001); and inverse associations with cancers of the rectum and uterus, again at > 2.5 mg/L (Weyer et al. 2001).

Exposure to low levels of antibiotics and other pharmaceuticals in drinking water (generally at micrograms per liter or nanograms per liter) represent unintentional doses of substances generally used for medical purposes to treat active disease or prevent disease. The concern is more related to possible cumulative effects of long-term low-dose exposures than on acute health effects (Daughton and Ternes 1999). A recent study conducted in Germany found that the margin between indirect daily exposure via drinking water and daily therapeutic dose was at least three orders of magnitude, concluding that exposure to pharmaceuticals via drinking water is not a major health concern (Webb et al. 2003). It should be noted that when prescribing medications, providers ensure patients are not taking incompatible drugs, but exposure via drinking water is beyond their control.

Endocrine-disrupting compounds are chemicals that exhibit biological hormonal activity, either by mimicking natural estrogens, by canceling or blocking hormonal actions, or by altering how natural hormones and their protein receptors are made (McLachlan and Korach 1995). Although very low levels of estrogenic compounds can stimulate cell activity, the potential for human health effects, such as breast and prostate cancers, and reproductive effects from exposure to endocrine disruptors, is in debate (Weyer and Riley 2001).

Workshop Recommendations

Priority research needs.

- Ecosystems monitoring: Systematic sustained studies of ecosystem health in proximity to large CAFOs are needed, including effects of input spikes during spills or flooding events.
- Toxicologic assessment of contaminants: Identification and prioritization of contaminants are needed to identify those that are most significant to environmental and public health. Toxicity studies need to be conducted to identify and quantify contaminants

(including metabolites), and to investigate interactions (synergistic, additive, and antagonistic effects).

- Fate and transport: Studies of parent compounds and metabolites in soil and water must be conducted, and the role of sediment as a carrier and reservoir of contaminants must be evaluated.
- Surveillance programs: Programs should be instituted to assess private well water quality in high-risk areas. Biomonitoring programs should be designed and implemented to assess actual dose from environmental exposures.

Translation of science to policy.

- Wastewater and drinking water treatment: Processes for water treatment must be monitored to ensure adequate removal or inactivation of emerging contaminants.
- Pollution prevention: Best management practices should be implemented to prevent or minimize release of contaminants into the environment.
- Education: Educational materials should be continued to be developed and distributed to agricultural producers.

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ATTACHMENT 8

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Prepared in cooperation with the North Carolina Department of Environment and Natural Resources, Division of Water Resources

Surface-Water Quality in Agricultural Watersheds of the North Carolina Coastal Plain Associated with Concentrated Animal Feeding Operations

Scientific Investigations Report 2015–5080

U.S. Department of the Interior
U.S. Geological Survey

Front and back covers: Sandy Run tributary to Middle Swamp, Greene County, North Carolina. Photographs by Stephen Harden, U.S. Geological Survey.

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U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
Suzette M. Kimball, Acting Director

U.S. Geological Survey, Reston, Virginia: 2015

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Contents

Figures

Tables

Appendixes (available at http://pubs.usgs.gov/sir/2015/5080/)

Conversion Factors

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
million gallons (Mgal)	3,785	cubic meter (m^3)
	Flow rate	
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
	Mass	
ounce, avoirdupois (oz)	28.35	gram (g)
pound avoirdupois (lb)	0.4536	kilogram (kg)
ton, short (2,000 lb)	0.9072	megagram (Mg)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as
$°F = (1.8 \times °C) + 32$.

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter (µg/L).

Datum

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Abbreviations

AFO	animal feeding operation
ANOVA	analysis of variance
BK sites	background watersheds with no active CAFOs
CAFO	concentrated animal feeding operation
DO	dissolved oxygen
DWR	North Carolina Division of Water Resources
EPA	U.S. Environmental Protection Agency
GIS	geographic information system
GMWL	global meteoric water line
HSG	hydrologic soil group
lidar	light detection and ranging
LMWL	local meteoric water line
N	nitrogen
NLCD	National Land Cover Database
NPDES	National Pollutant Discharge Elimination System
NPS	nonpoint source
NWIS	USGS National Water Information System
NWQL	USGS National Water Quality Laboratory
ortho-P	orthophosphate
P	phosphorus
PAN	plant available nitrogen
RL	reporting level
RPD	relative percent difference
RSIL	USGS Reston Stable Isotope Laboratory
SP sites	watersheds with at least one active swine CAFO and one active poultry CAFO
SSLW	steady state live weight
SW sites	watersheds with one or more active swine CAFOs but no poultry CAFOs
USGS	U.S. Geological Survey

Surface-Water Quality in Agricultural Watersheds of the North Carolina Coastal Plain Associated with Concentrated Animal Feeding Operations

By Stephen L. Harden

Abstract

The effects of concentrated animal feeding operations (CAFOs) on water quality were investigated at 54 agricultural stream sites throughout the North Carolina Coastal Plain during 2012 and 2013. Three general watershed land-use types were examined during the study, including 18 background watersheds with no active CAFOs (BK sites), 18 watersheds with one or more active swine CAFOs but no poultry CAFOs (SW sites), and 18 watersheds with at least one active swine CAFO and one active dry-litter poultry CAFO (SP sites). The watershed drainage areas for these 54 stream sites ranged from 1.2 to 17.5 square miles. Conventional fertilizers used for crop production are the primary source of nutrients at the BK sites. Animal-waste manures represent an additional source of nutrients at the SW and SP study sites.

Land cover, soil drainage, and CAFO attributes were compiled for each watershed. Water-quality field measurements were made and samples were collected at the 54 primary sites during 6 bimonthly sampling periods from June 2012 to April 2013. An additional 23 secondary sites were sampled once during April 2013 to provide supplemental data at stream locations directly adjacent or in close proximity to swine CAFOs and (or) background agricultural areas within 9 of the primary watersheds. The watershed drainage areas for the 23 secondary sites ranged from 0.2 to 8.9 square miles. Water temperature, specific conductance, dissolved-oxygen concentration, and pH were measured directly in the streams. Water samples were analyzed for major ions, nutrients, and stable isotopes, including delta hydrogen-2 (δ^2H) and delta oxygen-18 (δ^{18}O) of water and delta nitrogen-15 (δ^{15}N) and δ^{18}O of dissolved nitrate plus nitrite.

Most of the water-quality properties and constituents varied significantly among the six sampling periods, changing both seasonally and in response to hydrologic conditions. The differences noted among the sampling periods indicate that the interactions between seasonal climatic differences, streamflow conditions, and instream biotic and abiotic processes are complex and their integrated effects can have varying degrees of influence on individual nutrients.

Water-quality differences were noted for the SW and SP land-use groups relative to the BK group. Median values of specific conductance, several major ions (magnesium, sodium, potassium, and chloride), and nitrogen fractions (ammonia plus organic nitrogen, ammonia, nitrate plus nitrite, total nitrogen, and δ^{15}N of nitrate plus nitrite) were higher for the SW and SP groups compared to the BK group. No significant differences in water temperature, dissolved oxygen, calcium, total organic nitrogen, orthophosphate, total phosphorus, or δ^{18}O of nitrate plus nitrite were noted among the land-use groups. When compared on the basis of land-use type, there was an overall measurable effect of CAFO waste manures on stream water quality for the SW and SP watershed groups.

Some individual sites within the SW and SP groups showed no measurable CAFO effects on water quality despite having CAFOs present upstream. An evaluation of sodium plus potassium concentrations coupled with δ^{15}N values of nitrate plus nitrite proved valuable for distinguishing which SW and SP sites had a water-quality signature indicative of CAFO waste manures. Sites with CAFO manure effects were characterized by higher sodium plus potassium concentrations (commonly between 11 and 33 milligrams per liter) and δ^{15}N values of nitrate plus nitrite (commonly between 11 and 26 parts per thousand) relative to sites reflecting background agricultural conditions, which commonly had sodium plus potassium concentrations between 6 and 14 milligrams per liter and δ^{15}N values of nitrate plus nitrite between 6 and 15 parts per thousand. On the basis of the results of this study, land applications of waste manure at swine CAFOs influenced ion and nutrient chemistry in many of the North Carolina Coastal Plain streams that were studied.

A classification tree model was developed to examine relations of watershed environmental attributes among the study sites with and without CAFO manure effects. Model results indicated that variations in swine barn density, percentage of wetlands, and total acres available for applying swine-waste manures had an important influence on those watersheds where CAFO effects on water quality were either evident or mitigated. Measurable effects of CAFO waste manures on stream water quality were most evident in those SW and SP watersheds having lower percentages of wetlands combined

with higher swine barn densities and (or) higher total acres available for applying waste manure at the swine CAFOs. Stream water quality was similar to background agricultural conditions in SW and SP watersheds with lower swine barn densities coupled with higher percentages of wetlands or lower acres available for swine manure applications. The model provides a useful tool for exploring and identifying similar, unmonitored watersheds in the North Carolina Coastal Plain with potential CAFO manure influences on water quality that might warrant further examination.

Introduction

The U.S. Environmental Protection Agency's (EPA) National Water Quality Inventory Report to Congress (U.S. Environmental Protection Agency, 2010) lists pathogens, sediment, organic enrichment and oxygen depletion, and nutrients as several leading causes of impairment of rivers and streams in the United States. Agriculture, including crop and animal production, was cited as the most probable source of impairments in the assessed rivers and streams. Nonpoint-source (NPS) pollution from agricultural activities is of particular concern in eastern North Carolina because nutrient over-enrichment in surface waters has contributed to water-quality problems in the Tar-Pamlico, Neuse, and Cape Fear River Basins, particularly in the estuaries (Spruill and others, 1998; Luettich and others, 2000; Burkholder and others, 2006). Excessive inputs of nitrogen (N) and phosphorus (P) to nutrient-sensitive waters can contribute to eutrophication, excess algal blooms, fish kills, and outbreaks of toxic dinoflagellates (Burkholder and others, 1995; Burkholder and Glasgow, 1997; Stow and others, 2001; Paerl and others, 2004). Animal feeding operations (AFOs) are recognized as important NPS contributors of N and P to streams in the North Carolina Coastal Plain physiographic province (Glasgow and Burkholder, 2000; Mallin and Cahoon, 2003; Burkholder and others, 2006; Rothenberger and others, 2009). Large amounts of land-applied animal manures in watersheds with high densities of AFOs can lead to nutrient surpluses that exceed the assimilative capacity of the watershed to absorb excess nutrients without having deleterious effects on water quality (Stone and others, 1998; Mallin and Cahoon, 2003; Hubbard and others, 2004; Sims and others, 2005; Copeland, 2010).

North Carolina is one of the Nation's leading animal producers, ranking second in the production of both swine and turkeys and fourth in the production of broiler chickens (North Carolina Department of Agriculture and Consumer Services, 2012). In North Carolina, AFOs are regulated and permitted as non-discharge facilities by the Animal Feeding Operations Program within the North Carolina Department of Environment and Natural Resources Division of Water Resources (DWR). As of January 2013, there were 2,356 individually permitted AFOs in North Carolina (North Carolina Division of Water Resources, 2013), with about 90 percent of the facilities consisting of swine AFOs (total of 2,132) and the remaining 10 percent consisting primarily of cattle (total of 199) and wet poultry (total of 21) AFOs. The majority of the swine AFOs (2,006) are located in the Coastal Plain (fig. 1). Most poultry AFOs in North Carolina consist of dry-litter operations that are exempt from permitting by the State. The number of dry-litter poultry AFOs in the Coastal Plain is likely similar to the number of swine AFOs (Keith Larick, North Carolina Division of Water Resources, oral commun., June 2013).

It is of note that the terms AFO and concentrated animal feeding operation (CAFO) often are used interchangeably within the literature; however, there are technical distinctions between them as defined by the EPA (40 CFR §122.23). The EPA generally defines AFOs as "operations where animals have been, are, or will be stabled or confined and fed or maintained for a total of 45 days or more in any 12-month period and where vegetation is not sustained in the confinement area during the normal growing season" (U.S. Environmental Protection Agency, 2012). An AFO may be further designated as a CAFO on the basis of the number of animals confined and specific criteria concerning the discharge of pollutants to adjacent surface waters, which if so designated makes the CAFO subject to National Pollutant Discharge Elimination Systems (NPDES) permitting requirements (40 CFR §122.23). In this report, swine and poultry feeding operations are collectively referred to as CAFOs even though they may not all technically meet the regulatory definitions.

At a typical swine CAFO, waste materials are flushed from the swine houses to one or more holding lagoons for temporary storage. Wastewater effluent from the lagoon(s) periodically is applied to nearby fields, commonly through surface spraying, in accordance with the permitted facility's Certified Animal Waste Management Plan such that the total N applied can be used during crop growth to avoid runoff or excessive leaching (Keith Larick, North Carolina Division of Water Resources, oral commun., June 2013); however, problems can result from adverse weather conditions or application rates that exceed crop uptake (Evans and others, 1984; Smith and Evans, 1998). At the poultry CAFOs, dry litter commonly is applied to cropland at the individual facilities if sufficient acreage is available, or the litter can be transported offsite and applied as a source of nutrients to other agricultural fields (Crouse and Shaffer, 2011).

Previous studies have examined the effects of swine and poultry CAFOs on groundwater and surface-water quality, especially regarding N and P, in the North Carolina Coastal Plain. Huffman (2004) found that seepage from swine-waste lagoons built before 1993, without clay liners, increased shallow groundwater concentrations of mineral N (ammonia N plus nitrate N) by 10 to 40 milligrams per liter (mg/L) as N at 11 sites and more than 40 mg/L as N at 16 sites. Various investigators have noted nitrate concentrations commonly between 10 and 30 mg/L, and in some cases between 50 and 150 mg/L, in groundwater collected beneath or adjacent to application fields receiving swine-lagoon effluent or

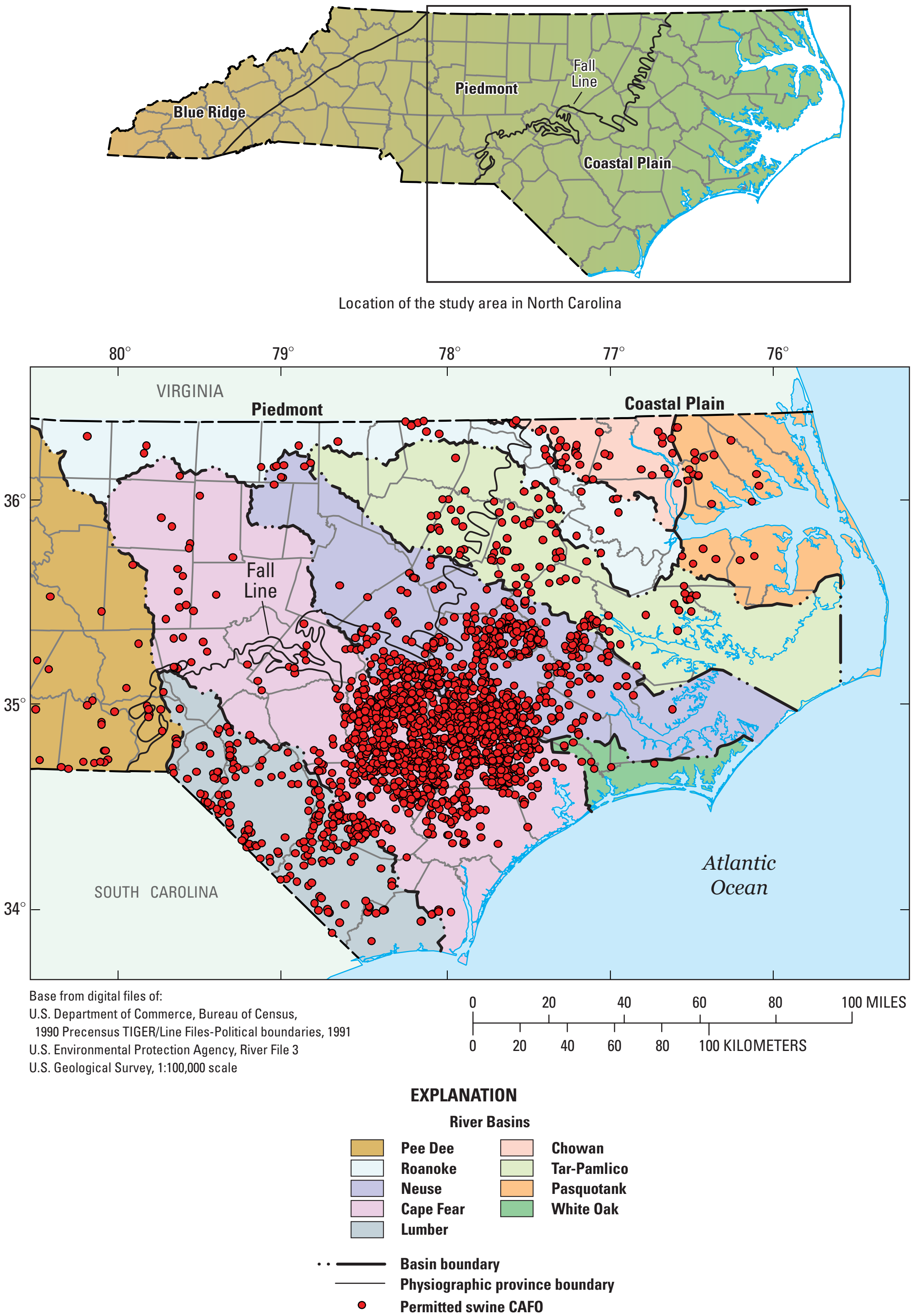

Figure 1. Locations of permitted swine concentrated animal feeding operations (CAFOs) in eastern North Carolina (swine CAFO locations obtained from North Carolina Division of Water Resources, 2013).

poultry litter (Hunt and others, 1995; Stone and others, 1998; Karr and others, 2001; Spruill and others, 2002; Israel and others, 2005; Dukes and Evans, 2006; Harden and Spruill, 2008). In addition to nitrate, increased concentrations of calcium, magnesium, sodium, potassium, and chloride have been observed in groundwater beneath swine CAFO spray fields (Karr and others, 2001; Spruill and others, 2005). The transport of P from agricultural fields to surface water typically occurs through overland runoff; however, repeated applications of swine-waste manure to fields can lead to excess accumulations of P in soil and subsequent leaching to groundwater for possible offsite transport to receiving streams (Novak and others, 2000; Nelson and others, 2005).

Elevated nutrient concentrations also have been observed in streams receiving overland runoff, groundwater discharge, and subsurface tile drainage derived from CAFOs (Stone and others, 1995; Karr and others, 2001; Spruill and others, 2005; Dukes and Evans, 2006; Harden and Spruill, 2008). Stone and others (1995) noted that a stream with intensive swine and poultry operations had nutrient concentrations during both stormflow and baseflow conditions that were several times higher than those in an adjacent background stream with no animal operations. In the stream influenced by the CAFOs, mean concentrations were 5.6 mg/L as N for nitrate, 0.74 mg/L as N for ammonia, and 0.68 mg/L for orthophosphate during baseflow conditions, and mean concentrations were 5.4 mg/L as N for nitrate, 2.28 mg/L as N for ammonia, and 1.3 mg/L for orthophosphate during stormflow conditions. Surface-water samples collected by Karr and others (2001) in a stream adjacent to two swine CAFOs had a median nitrate concentration of 6.7 mg/L as N. Harden and Spruill (2008) observed elevated levels of nitrate (median of 6.1 and range of 2.0 to 10.7 mg/L as N), ammonia (median of 0.76 and range of 0.09 to 2.38 mg/L as N), and dissolved P (median of 0.05 and range of 0.01 to 0.29 mg/L) in 28 surface-water samples collected in 2006 during stormflow and baseflow conditions from a stream next to waste-manure application fields at a swine CAFO. Elevated nitrate concentrations in this stream are considered to be strongly influenced by water discharged through a tile drain located in one of the adjacent spray fields (Spruill and others, 2005, Harden and Spruill, 2008). In 2006, water discharging from the tile drain to the stream had nitrate concentrations ranging from about 22 to 45 mg/L as N (Harden, 2008).

The practice of applying waste manure to fields at swine CAFOs is common in many watersheds throughout the Coastal Plain so there is substantial interest in understanding their influence on stream water quality. Many of the studies conducted to evaluate water-quality conditions related to CAFOs in the Coastal Plain have been limited in geographic extent, either focusing on individual farm sites or several streams within a particular watershed. The lack of stream water-quality data from a more representative number of watersheds makes it difficult for DWR to assess the extent to which effects of swine CAFOs on surface-water quality can be measured and how well existing CAFO regulations protect the waters of the State or to recommend effective changes to regulations or procedures. In 2011, DWR (formerly named the Division of Water Quality) and the U.S. Geological Survey (USGS) initiated a collaborative study to document whether swine CAFOs located in various Coastal Plain watersheds have a measurable effect on stream water quality. The study results presented in this report provide needed information from a large number of sites over a broader geographic area to better understand relations between swine CAFOs and stream water quality in eastern North Carolina.

Purpose and Scope

The primary purpose of this report is to summarize and synthesize chemical data collected from 54 agricultural watershed study sites throughout the North Carolina Coastal Plain to characterize water-quality conditions in streams receiving inputs from swine CAFOs compared to streams that receive inputs primarily from inorganic fertilizers. The scope of work included field measurements of water-quality properties and collection of surface-water samples for laboratory analysis of nutrients, major ions, and stable isotopes. Six rounds of bimonthly samples were collected from June 2012 to April 2013 at 54 primary watershed study sites. The last sampling round in April 2013 included collection and analysis of samples from 23 additional sites located within 9 of the 54 primary watersheds. Results were used to evaluate differences in stream water quality among watersheds with no CAFOs, watersheds with swine CAFOs, and watersheds with both swine and poultry CAFOs. Land cover, soil drainage class, and CAFO attributes (such as number of facilities, animal barns, swine animals, and total weight of swine) were used to examine potential relations between watershed environmental variables and water-quality conditions among the primary study sites. The main study objectives were to (1) assess water-quality differences among streams draining watersheds with and without land-applied CAFO waste manures, (2) examine the use of multiple chemical constituents for identifying effects of CAFOs on stream water quality, and (3) examine relations of environmental variables among watersheds with and without measurable CAFO manure effects. The study results are intended to assist water-resource managers and policy makers in their efforts to protect and improve stream water quality throughout North Carolina.

Description of the Study Area

The watershed sites examined in the Coastal Plain study area have drainage areas less than 20 square miles (mi^2) with land cover composed predominantly of cropland, forests, and wetlands. Most of the watersheds typically feature low-gradient blackwater streams and swamps with slow streamflow velocities. Varying degrees of submerged and floating aquatic vegetation and organic debris are present within and along the stream channels. These types of streams often have naturally low dissolved oxygen (DO) that can be depleted further as a result of nutrient and organic inputs from agricultural activities.

When examining stream water quality at the agricultural watershed sites in this study, it is important to understand that different processes influence fate and transport of nutrient inputs from agricultural fields to receiving streams. Nutrients applied to agricultural fields that percolate through the soils to the underlying surficial aquifer can be transported with groundwater as it discharges to receiving streams. Hydrograph separations performed on streamflow data during previous investigations indicate that groundwater, thought to be derived mostly from shallow aquifer systems, commonly contributes about 50 to 60 percent of the average annual streamflow to streams in the North Carolina Coastal Plain (McMahon and Lloyd, 1995; Spruill and others, 2005; Harden and others, 2013). Therefore, groundwater is potentially a major contributor of water and agriculturally derived chemical constituents to the stream study sites, particularly when there is minimal overland runoff from precipitation.

Various environmental, hydrogeologic, and geochemical factors that influence nitrate transport along groundwater flow paths beneath agricultural fields to receiving streams in the North Carolina Coastal Plain are discussed by Spruill and others (2005) and Harden and Spruill (2008). These factors include depth to water and saturated thickness of the surficial aquifer (Tesoriero and others, 2000; Tesoriero and others, 2005), groundwater residence times (Puckett, 2004; Tesoriero and others, 2005; Seitzinger and others, 2006), availability of organic carbon to drive denitrification reactions (Korom, 1992), and presence of riparian buffers (Speiran and others, 1998; Spruill, 2000; Puckett, 2004; Seitzinger and others, 2006). In evaluating changes in nitrate concentrations along groundwater flow paths at five study sites in the Coastal Plain, Harden and Spruill (2008) determined that denitrification was the most influential factor responsible for observed decreases in groundwater nitrate along the flow paths. Although some denitrification of groundwater nitrate occurred beneath the agricultural fields, nitrate reduction along the groundwater flow paths was most prevalent in the downgradient riparian buffer zone and hyporheic zone at the streams, where highly reduced conditions associated with organic-rich deposits enhanced the overall amount of denitrification.

The nitrate-reducing capacity of the buffer zone combined with that of the hyporheic zone can substantially lower the amount of groundwater nitrate discharged to streams in agricultural settings of the Coastal Plain (Spruill, 2000; Harden and Spruill, 2008). Depending on hydrogeologic and geochemical conditions, relatively young groundwater may move quickly along shallow flow paths beneath the riparian buffer and outpace the time needed for complete reduction of nitrate before discharging to a stream. Groundwater discharge along shallow flow paths may occur along seeps or channel walls that bypass the highly organic fluvial material in the hyporheic zone. If this water contains nitrate that has passed through the riparian buffer, the water can affect the nitrate concentration in the receiving stream.

In addition to groundwater transport, overland flow of water that occurs through field-drainage ditches is another important pathway that conveys nutrients from agricultural fields to receiving streams. Field-drainage ditches and subsurface tile drains commonly are used in the North Carolina Coastal Plain for improving drainage in agricultural fields with poorly drained soils (Evans and others, 1991; Gilliam and others, 1997). Water conveyed through the field ditches to the streams includes surface runoff from the fields, when rainfall amounts are greater than the infiltration capacity of soils, and subsurface inputs of shallow groundwater from beneath the fields. Lateral inflows of shallow groundwater through the banks and bottom of the ditches can occur during parts of the year when high water-table conditions are present beneath the fields. In fields with subsurface tile drains, shallow groundwater intercepted and collected by the tiles at the top of the water table is discharged through outlets directly to the ditches.

These drainage improvements lower the water table beneath agricultural fields, which increases the amount of land available for cultivation; however, the process of redirecting shallow groundwater beneath agricultural fields through tile drains and ditches can increase nutrient transport, particularly nitrate, in drainage water exiting the fields to receiving streams (David and others, 1997; Jaynes and others, 2001; Randall and Mulla, 2001; Harden and Spruill, 2004). As previously discussed, elevated nitrate concentrations in shallow groundwater beneath agricultural fields have commonly been observed in the Coastal Plain, especially at fields receiving land applications of animal-waste manures. A study by Harden and Spruill (2004) on the quality of drainage water from field ditches and tile drains in a North Carolina Coastal Plain watershed found that median concentrations of nitrate as N were significantly higher in water exiting field ditches (8.2 mg/L) and tile drains (32.0 mg/L) at fields receiving applications of swine-waste manures as compared to field ditches (2.7 mg/L) and tile drains (6.8 mg/L) at fields receiving applications of commercial fertilizers.

Because field ditches and tile drains are used to expedite the drawdown of the water table, they can allow groundwater with elevated nitrate levels in the upper part of the surficial aquifer beneath agricultural fields to bypass natural organic-rich aquifer sediments in the riparian buffer and hyporheic zones that normally would reduce the amount of nitrate in groundwater discharging to the streams (Spruill, 2000; Harden and Spruill, 2008). Considering that most watersheds examined for this study have substantial riparian buffer zones and organic-rich floodplain deposits and, hence, a high degree of denitrification potential prior to groundwater discharge, it is probable that overland inputs of water through field drainage ditches contribute much of the nitrate delivered to the stream sites. Overland transport through the field ditches can occur anytime there is excessive runoff from storm events but is most common during sustained periods of high water-table conditions, which typically occur during the colder winter and early spring months, generally from December to April, when evapotranspiration is lowest.

Methods

This section provides a discussion of the network design and watershed attributes compiled for the study sites, and the sampling and analytical methods used for generating the water-quality dataset. Statistical methods used during data analysis also are discussed.

Network Design and Watershed Attributes

An integrated approach was used for establishing the network of surface-water sampling sites for the study. Three general watershed land-use types, or groups, were included: watersheds with no active CAFOs (referred to as background (BK) sites); watersheds with one or more active swine CAFOs but no poultry CAFOs (referred to as SW sites); and watersheds with at least one active swine CAFO and one active poultry CAFO (referred to as SP sites). Although the initial study intent was to evaluate potential influences of swine CAFOs, it was difficult to find swine only watersheds across the study area that did not also contain poultry CAFOs. Therefore, the SP sites were included to provide data for additional watersheds containing swine CAFOs, as well as for examining potential differences between swine only sites and sites with both swine and poultry. Watersheds that contained only poultry CAFOs were not considered because it was outside the scope of work for this study.

The stream sites selected for study include an equal number (18) representing each of the BK, SW, and SP watershed land-use types (table 1; fig. 2) that also had similar distributions in watershed characteristics such as drainage areas and land cover. These 54 watershed sites are referred to as primary study sites because they were the primary focus of data-collection activities for the 6 bimonthly sampling periods from June 2012 to April 2013. The April 2013 sampling period included collection of surface-water samples from 23 additional sites, referred to as secondary sites, located within 9 of the primary watershed sites (table 1). One or more secondary sites were sampled upstream from the primary sites to provide additional water-quality data for stream sites located close or adjacent to swine CAFOs and (or) in subwatershed areas with no swine CAFOs. The study network spanned six river basins throughout the Coastal Plain in eastern North Carolina (table 1; fig. 2). Individual maps for the primary and secondary sites are provided in appendix A1 (figs. A1-1 through A1-54).

All study watersheds have than less than 10 percent developed (urban) lands, and none contain permitted NPDES wastewater-discharge facilities. Therefore, agricultural activities represent the most likely source of nutrients to the streams. The watersheds without CAFOs (BK sites) and with CAFOs (SW and SP sites) all contain agricultural lands where commercial fertilizers are used during the production of crops. The water-quality constituents analyzed in stream samples collected during the study include those essential primary nutrients (N, P, and potassium) and secondary nutrients (calcium, magnesium, and sulfur) found in commercial fertilizer materials commonly used in North Carolina for growing crops (Zublena and others, 1991; Tucker, 1999). These same essential plant nutrients, as well as sodium and chloride, are found in swine and poultry organic waste manures (Zublena and others, 1991, 1997a, 1997b; Barker and others, 1994; Osmond and Kang, 2008). Land applications of swine-waste manure and poultry litter represent an additional source of these constituents to agricultural fields in the SW and SP watersheds. Because watershed characteristics are similar among the three site groups, with the exception of the presence or absence of CAFOs, differences in stream concentrations of nutrients and (or) major ions observed at the SW and SP sites relative to the BK sites likely reflect inputs derived from swine and (or) poultry animal-waste manures.

Watershed boundaries and contributing drainage areas for the study sites were determined using the USGS StreamStats application developed for North Carolina (http://water.usgs.gov/osw/streamstats/north_carolina.html; Weaver and others, 2012). These features were calculated within StreamStats using a 30-foot (ft) by 30-ft lidar-derived digital elevation model (North Carolina Floodplain Mapping Program, 2012). Watershed drainage areas range from 1.2 to 17.5 mi^2 for the 54 primary sites and 0.2 to 8.9 mi^2 for the 23 secondary sites.

Data were compiled for selected watershed attributes to characterize environmental conditions at the study sites. Physical (land cover and soil drainage) and anthropogenic features (point-source dischargers, non-discharge land application sites, and CAFOs) were compiled using geographic information system (GIS) processes. The 54 primary sites were chosen to avoid or minimize potential influences of wastewater-discharge facilities, non-discharge facilities, and developed lands in order to facilitate water-quality interpretations between the watersheds with and without CAFOs.

Table 1. Study network, including primary and associated secondary sites, monitored for water quality in the North Carolina Coastal Plain.

[ID, identification; HUC, hydrologic unit code; USGS, U.S. Geological Survey; NC, North Carolina; HWY, highway; SR, secondary road; mi², square miles]

Primary study ID (see fig. 2)	Secondary study ID associated with primary sites (see appendix A1)	River basin	USGS station number	USGS station name	Decimal latitude	Decimal longitude	Drainage area (mi²)
BK-01		Roanoke	0208102325	Blue Hole Swamp at NC HWY 11/42 near Cahaba, NC	36.01654	–77.21197	14.9
BK-02		Roanoke	02081065	Smithwick Creek near Bear Grass, NC	35.76589	–77.05184	12.5
BK-03		Roanoke	02081040	Etheridge Swamp at SR 1326 near Oak City, NC	35.98837	–77.34820	3.9
BK-04		Roanoke	0208103875	Conoho Creek at SR 1336 near Oak City, NC	36.01207	–77.29780	10.0
BK-05		Roanoke	0208105040	Conoho Creek tributary at SR 1002 at Hassell, NC	35.91971	–77.27077	10.8
BK-06		Chowan	0205309110	Kirbys Creek tributary at SR 1356 near Pendleton, NC	36.49604	–77.17341	5.9
BK-07		Tar-Pamlico	02083583	Williamson Branch at SR 1128 near St. Lewis, NC	35.79453	–77.72893	4.5
BK-08		Tar-Pamlico	02083889	Tyson Creek at SR 1245 at Kings Crossroads, NC	35.65818	–77.55068	3.8
BK-09		Tar-Pamlico	02084212	Hunting Run near Pactolus, NC	35.66947	–77.26106	5.9
BK-10		Tar-Pamlico	0208451810	Beaverdam Swamp at SR 1520 near Alligoods, NC	35.55525	–76.92182	5.5
BK-11		Neuse	02090770	Whiteoak Swamp at SR 1514 near Holdens Crossroads, NC	35.70709	–77.75435	5.6
BK-12		Neuse	0209096970	Moccasin Run near Patetown, NC	35.47927	–77.90992	3.1
BK-13		Neuse	02091623	Langs Mill Run at SR 1242 near Fountain, NC	35.64908	–77.60427	5.9
BK-14		Neuse	02091712	Middle Swamp near Marlboro, NC	35.56626	–77.59853	14.7
BK-15		Cape Fear	0210682145	Big Creek at SR 1006 at Bethany Crossroads, NC	35.05978	–78.70102	6.1
BK-16		Cape Fear	0210591785	Sevenmile Swamp at US HWY 13 at Rosin Hill, NC	35.20431	–78.43143	9.2
BK-17		Cape Fear	0210754615	White Oak Branch at SR 1209 near Ivanhoe, NC	34.61149	–78.18248	3.9
BK-18		Lumber	0213453011	Horse Swamp at SR 2435 near Fairmont, NC	34.52107	–79.17844	5.4
SW-01		Roanoke	02081016	Steptoe Run near Scotland Neck, NC	36.10934	–77.37070	5.4
SW-02		Tar-Pamlico	02083686	Kitten Creek at SR 1251 near Sharp Point, NC	35.70728	–77.56920	9.0
SW-03		Tar-Pamlico	0208368850	Unnamed tributary to Otter Creek at SR 1615 near Sharp Point, NC	35.73388	–77.57359	4.8
SW-04		Neuse	02089225	Little Marsh Run at SR 1714 at Parkstown, NC	35.37789	–77.82240	1.2
	SW-04A	Neuse	0208922490	Little Marsh Run headwaters near Parkstown, NC	35.38754	–77.83183	0.4
	SW-04B	Neuse	0208922495	Little Marsh Run at St. Delight Ch. Road at Parkstown, NC	35.38270	–77.82576	1.0
SW-05		Neuse	02089584	Hornpipe Branch at SR 1130 near Deep Run, NC	35.14308	–77.66903	3.9
	SW-05A	Neuse	0208958380	Hornpipe Branch at SR 1137 near Deep Run, NC	35.13115	–77.66361	0.8
	SW-05B	Neuse	0208958385	Hornpipe Branch tributary at SR 1137 near Deep Run, NC	35.13326	–77.65996	0.5
	SW-05C	Neuse	0208958390	Hornpipe Branch tributary at SR 1130 near Deep Run, NC	35.13682	–77.66893	0.9
SW-06		Neuse	02091960	Creeping Swamp near Calico, NC	35.42944	–77.18974	11.2
SW-07		Neuse	02090793	Whiteoak Swamp tributary at SR 1514 at Drivers Store, NC	35.70027	–77.81418	1.3
SW-08		Neuse	02091725	Sandy Run at US HWY 13/258 at Lizzie, NC	35.51625	–77.61542	15.8
	SW-08A	Neuse	0209172000	Sandy Run at SR 1301 near Castoria, NC	35.53175	–77.65237	8.9
	SW-08B	Neuse	0209172150	Drainage ditch to Sandy Run at SR 1326 near Lizzie, NC	35.51573	–77.65001	1.2
	SW-08C	Neuse	02091722	Unnamed tributary to Sandy Run at SR 1301 near Lizzie, NC	35.52024	–77.64036	2.8
	SW-08D	Neuse	02091724	Unnamed tributary to Sandy Run at SR 1301 at Lizzie, NC	35.51052	–77.62631	1.2

Table 1. Study network, including primary and associated secondary sites, monitored for water quality in the North Carolina Coastal Plain.—Continued

[ID, identification; HUC, hydrologic unit code; USGS, U.S. Geological Survey; NC, North Carolina; HWY, highway; SR, secondary road; mi², square miles]

Primary study ID (see fig. 2)	Secondary study ID associated with primary sites (see appendix A1)	River basin	USGS station number	USGS station name	Decimal latitude	Decimal longitude	Drainage area (mi^2)
SW-09		Cape Fear	0210596803	Hornet Swamp at SR 242 near Piney Green, NC	35.11474	–78.47670	4.0
SW-10		Cape Fear	0210592050	Ward Swamp at SR 1711 near Monks Crossroads, NC	35.19976	–78.30362	1.3
SW-11		Cape Fear	0210770367	Youngs Swamp at SR 1725 near Giddensville, NC	35.16676	–78.21747	2.1
SW-12		Cape Fear	0210778920	Big Branch at SR 1301 at Bowdens, NC	35.06026	–78.10009	3.2
SW-13		Cape Fear	0210782015	King Branch at SR 1305 at Friendship, NC	35.06047	–78.04184	1.9
	SW-13A	Cape Fear	0210782010	King Branch Headwaters near Friendship, NC	35.06601	–78.06513	0.8
	SW-13B	Cape Fear	0210782013	King Branch Headwaters at Friendship, NC	35.06814	–78.05202	1.2
SW-14		Lumber	0213449620	Rattlesnake Branch at SR 1516 at Lennons Crossroads, NC	34.47430	–78.85823	3.1
SW-15		Lumber	0213453155	Aaron Swamp at SR 2455 near McDonald, NC	34.51163	–79.20262	12.1
SW-16		Lumber	0210899420	Little Whites Creek at SR 1700 near Bluefield, NC	34.54721	–78.61481	3.6
SW-17		Lumber	0210899878	Horseshoe Swamp at SR 1713 near Lisbon, NC	34.50059	–78.53169	9.4
SW-18		Lumber	0210910290	Butler Branch at US HWY 701 near Wootens Crossroads, NC	34.44726	–78.72026	3.7
SP-01		Tar-Pamlico	02084148	Chicod Creek at SR 1565 near Grimesland, NC	35.53304	–77.18784	17.5
	SP-01A	Tar-Pamlico	0208414580	Chicod Creek tributary at SR 1782 at Boyds Crossroads, NC	35.51606	–77.19316	1.6
	SP-01B	Tar-Pamlico	0208414590	Chicod Creek tributary south of SR 1780 at Boyds Crossroads, NC	35.52571	–77.18306	2.0
	SP-01C	Tar-Pamlico	0208414750	Chicod Creek tributary north of SR 1780 at Boyds Crossroads, NC	35.53302	–77.18058	0.5
SP-02		Neuse	0208813655	White Oak Branch at SR 1144 near Strickland Crossroads, NC	35.34614	–78.37521	5.3
SP-03		Neuse	02088285	Thoroughfare Swamp near Dobbersville, NC	35.23844	–78.15107	14.3
SP-04		Neuse	0208831520	Falling Creek at SR 1102 near Dobbersville, NC	35.27517	–78.27242	3.7
	SP-04A	Neuse	0208831504	Falling Creek tributary at SR 1201 near Newton Grove, NC	35.28633	–78.29202	0.4
	SP-04B	Neuse	0208831510	Falling Creek tributary at US HWY 13 near Newton Grove, NC	35.27540	–78.28327	1.5
SP-05		Neuse	02089598	Unnamed tributary to Southwest Creek at NC HWY 11 near Albrittons, NC	35.18177	–77.67071	1.4
	SP-05A	Neuse	0208959780	Southwest Creek tributary 2 at SR 1159 near Albrittons, NC	35.18384	–77.67951	0.5
	SP-05B	Neuse	0208959790	Southwest Creek tributary at SR 1159 near Albrittons, NC	35.17731	–77.67791	0.4
SP-06		Cape Fear	02105702	Davis Creek at SR 1713 near Lisbon, NC	34.54040	–78.50994	2.3
SP-07		Cape Fear	0210564590	Hammonds Creek at SR 1709 near Elizabethtown, NC	34.57002	–78.56049	12.0
SP-08		Cape Fear	0210687150	Big Swamp at SR 1441 near Clement, NC	35.08855	–78.59019	3.6
SP-09		Cape Fear	02107005	Cypress Creek at SR 1503 near Ammon, NC	34.78778	–78.50896	7.6
	SP-09A	Cape Fear	344734078312901	Drainage ditch to Cypress Creek near Ammon, NC	34.79279	–78.52442	6.9
SP-10		Cape Fear	02106011	Unnamed tributary to Bearskin Swamp at SR 1240 at Concord, NC	34.98793	–78.43314	1.5

Table 1. Study network, including primary and associated secondary sites, monitored for water quality in the North Carolina Coastal Plain.—Continued

[ID, identification; HUC, hydrologic unit code; USGS, U.S. Geological Survey; NC, North Carolina; HWY, highway; SR, secondary road; mi², square miles]

Primary study ID (see fig. 2)	Secondary study ID associated with primary sites (see appendix A1)	River basin	USGS station number	USGS station name	Decimal latitude	Decimal longitude	Drainage area (mi²)
SP-11		Cape Fear	0210608620	Six Runs Creek at SR 1742 near Giddensville, NC	35.14064	–78.25847	5.6
	SP-11A	Cape Fear	0210608603	Six Runs Creek at SR 1736 near Hobbton, NC	35.16458	–78.27822	0.7
	SP-11B	Cape Fear	0210608607	Six Runs Creek near Hobbton, NC	35.15719	–78.26996	1.2
	SP-11C	Cape Fear	0210608610	Unnamed tributary to Six Runs Creek near Giddensville, NC	35.15619	–78.26846	0.2
	SP-11D	Cape Fear	0210608612	Six Runs Creek near Giddensville, NC	35.15041	–78.26580	2.3
SP-12		Cape Fear	0210778820	Bear Swamp at SR 1301 at Bowdens, NC	35.05736	–78.13150	3.3
SP-13		Cape Fear	0210782005	Nahunga Creek at SR 1301 near Warsaw, NC	35.02692	–78.01086	8.2
SP-14		Cape Fear	0210760950	Poley Branch at SR 1534 at Outlaws Bridge, NC	35.15245	–77.85116	4.6
SP-15		Cape Fear	0210760860	Buck Marsh Branch at SR 1753 near Hines Crossroads, NC	35.18423	–77.87220	4.5
SP-16		Cape Fear	0210798920	Stephens Swamp at SR 1807 at Quinns Store, NC	34.88644	–77.72953	2.8
SP-17		Cape Fear	0210858154	Tenmile Swamp at SR 1207 near Cypress Creek, NC	34.76237	–77.66882	6.0
SP-18		Cape Fear	0210850250	Doctors Creek at SR 1129 near Shanghai, NC	34.75101	–78.16391	6.6

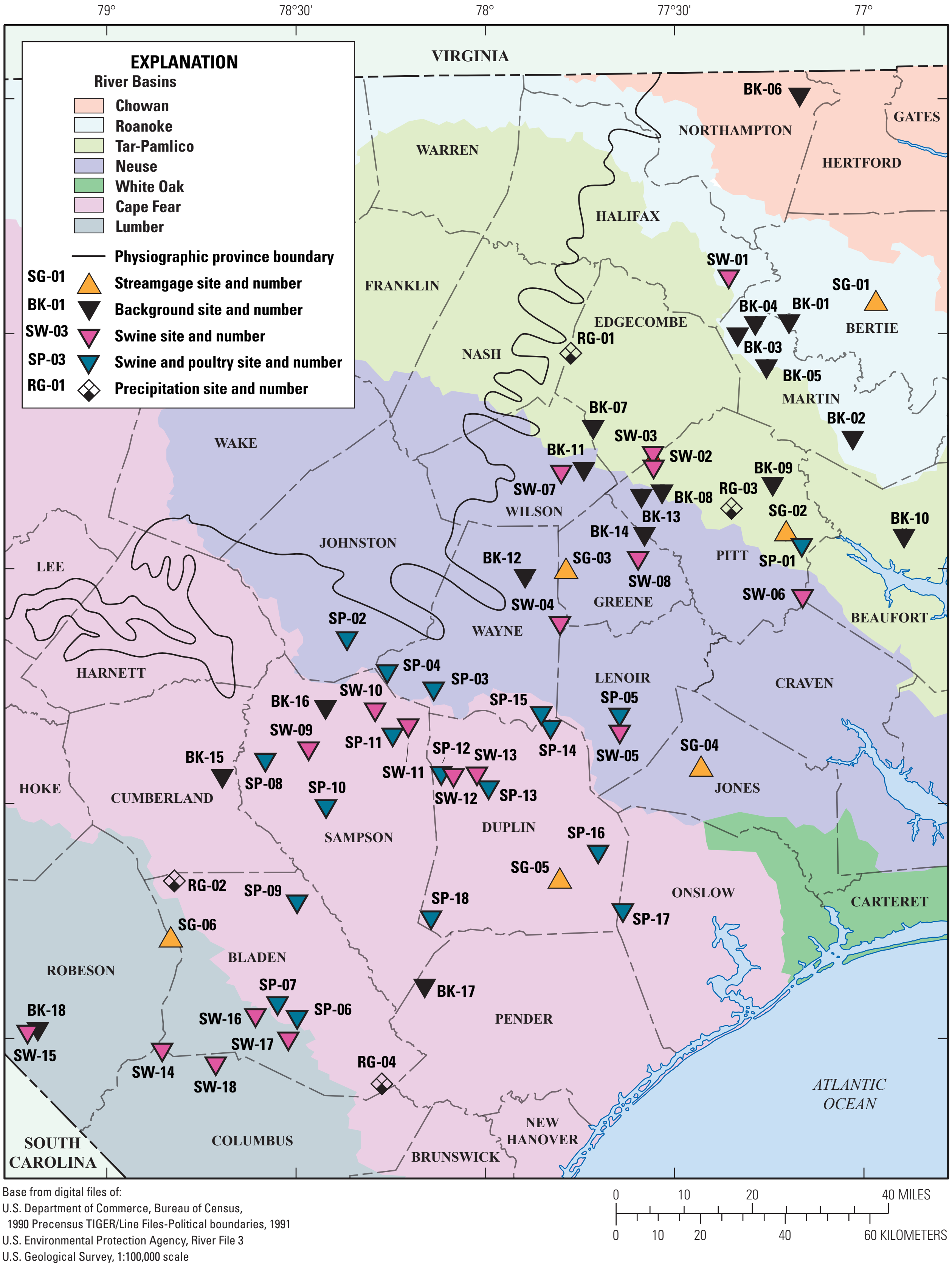

Figure 2. Locations of background, swine, and swine and poultry study sites, streamgage sites, and precipitation sites in the North Carolina Coastal Plain study area.

Land Cover and Hydrologic Soil Groups

Watershed attributes for land cover and hydrologic soil groups (HSGs) were compiled using StreamStats. Land-cover information was derived from the 2006 National Land Cover Database (NLCD) (Fry and others, 2011), which includes 15 individual land-cover classes. These 15 individual land-cover classes were aggregated into 8 principal land-cover categories (developed, forested, shrub, crops, grassland, wetlands, barren, and water), which were summarized for each watershed (appendix A2-1).

The study sites contain HSGs with varying degrees of soil drainage capacity. Data used to characterize the distribution of HSGs within the study sites were obtained through the U.S. Department of Agriculture Soil Survey Geographic Database (Soil Survey Staff, Natural Resources Conservation Service, n.d.). The areal extent and relative percentage for the four major HSGs (A, B, C, and D) and three dually classified HSGs (A/D, B/D, and C/D) were determined within the watershed of each site (appendix A2-2). Soils in HSGs A and B have low to moderately low runoff potential when thoroughly wet. Soils in HSGs C and D have moderately high to high runoff potential when thoroughly wet. Thus, soils in HSGs A and B have a higher degree of drainage, or water infiltration, as compared to soils in HSGs C and D, which are more poorly drained.

The dual hydrologic groups represent wet soils that were naturally classified as very poorly drained (HSG D) because of the presence of a water table within 2 ft of the land surface (U.S. Department of Agriculture, Natural Resources Conservation Service, 2009). If enhanced drainage measures, such as field ditches and subsurface tile drains, are used to maintain the seasonal high water table at least 2 ft below the surface, then the soils are characterized by the first letter of the dual groups (A/D, B/D, or C/D) on the basis of their saturated hydraulic conductivity and depth of the water table when drained (U.S. Department of Agriculture, Natural Resources Conservation Service, 2009). For this study, the data compiled for dual HSGs A/D, B/D, and C/D are assumed to represent drained soil conditions and were summed with their respective major HSGs to yield HSG total A, HSG total B, and HSG total C (appendix A2-2).

Wastewater Discharge Facilities and Non-Discharge Facilities

Information on NPDES-permitted wastewater-discharge facilities and permitted non-discharge facilities was provided by DWR (Michael Tutwiler, North Carolina Division of Water Resources, written commun., April 2012). Wastewater-discharge facilities that were considered included NPDES-permitted major municipal, minor municipal, major industrial/commercial, and 100 percent domestic discharge facilities. Harden and others (2013) previously indicated that point-source contributions of nutrients from wastewater-discharge facilities can have a significant influence on watershed nutrient yields in North Carolina. GIS analyses were used to map the locations of the discharge facilities in the Coastal Plain study area and to verify that none of the sites selected for study contained permitted dischargers.

GIS analyses also were performed to determine whether any permitted non-discharge facilities, which include wastewater irrigation, infiltration, or reclamation systems and land application of residual solids, were associated with the study sites. Only 2 of the 54 sites (SW-07 and SP-09) were found to have associated non-discharge facilities (appendix A3-1). Site SW-07 (appendix fig. A1-25) contains one residual solids land-application field, and site SP-09 (appendix fig. A1-45) contains two residual solids land-application fields. Any potential effects of these residual solids application fields on the water-quality results obtained at sites SW-07 and SP-09 are considered minimal and are not discussed in this report.

CAFOs

Available information on permitted CAFOs, including swine, cattle, and wet-poultry operations, was provided by DWR (Keith Larick, North Carolina Division of Water Resources, written commun., April 2012). All permitted CAFOs located in the 54 primary watersheds were mapped using GIS processes. The subgroups of the BK, SW, and SP study sites were operationally defined on the basis of the absence or presence of permitted active swine CAFOs located within the watersheds. None of the sites contained permitted cattle or wet-poultry CAFOs. Dry-litter poultry CAFOs, which are not required to have permits, were present in the SP watersheds.

Swine CAFO Attributes

Attribute data for the swine CAFOs were based on available information for facilities having either an active or inactive State of North Carolina permit. Swine CAFOs with active permits represent those facilities with ongoing swine production and field applications of swine-waste manure from the storage lagoons. Swine CAFOs with inactive permits represent former swine production facilities that are no longer operational. The inactive facilities currently have no swine animals or ongoing disposal of waste manure in application fields; however, remnant infrastructure, including barns and (or) inactive lagoons, may still be located at some of these facilities. The GIS analyses indicated that 10 of the study sites have 1 or 2 inactive-swine permits (appendix A3-2). Other than the permit numbers and locations, no other data were available for these inactive CAFOs. The active CAFOs, with ongoing waste-manure applications, are considered to have a more pronounced influence than the inactive CAFOs on water-quality conditions at the sites. Given the lack of information available for the inactive CAFOs, data evaluations conducted during the study focused on the permitted active swine CAFOs; the permitted inactive swine CAFOs were not considered further.

Several steps were taken in compiling attribute data for the active swine CAFOs. All active swine CAFOs within or along the boundaries of the 18 SW and 18 SP watershed sites were identified. Data provided by DWR for each active swine CAFO included information on the regulated swine activity, number of available acres for applying manure, amount of allowable plant available nitrogen (PAN), amount of generated PAN, and whether tile drains have been documented at the CAFO (appendix A3-3).

The regulated swine activity includes the type of swine production at the facility as well as the maximum annual average number of swine that can be produced. Seven types of swine production are associated with the CAFOs (Keith Larick, North Carolina Division of Water Resources, written commun., April 2012; table 2). Although multiple swine production activities are noted for some CAFOs, most produce only one type of swine. The average weight of swine produced and, consequently, the amount of waste manure generated by the swine population at a given CAFO depend on the type(s) of swine production at the facility. The maximum annual average number of swine (appendix A3-3) was multiplied by its respective average swine weight (table 2) to compute a total swine weight by production type. The number of swine and swine weights for all production types were summed to yield the total swine and total swine steady state live weight (SSLW) for each active CAFO.

The number of available acres listed for each active CAFO represents the total field acreage available at the facility for applying swine-waste manure (appendix A3-3). For a given facility, the amount of field acreage used for waste-manure applications during a given year may be lower than available. No information on the frequency and timing of applications or individual fields used was readily available for the CAFOs. The reported values for allowable PAN represent the maximum permitted amount of PAN that can be field applied annually at each CAFO. The reported values for generated PAN represent the calculated amount of PAN generated in waste manure that was field applied during 2012 at each CAFO (Keith Larick, North Carolina Division of Water Resources, written commun., July 2013). Ideally, the amount of generated PAN will be less than its allowable PAN on an annual basis such that the facility is not applying more PAN than allowed based on its permit.

Table 2. Swine production type and average swine weight associated with concentrated animal feeding operations in the study area.

Swine production type	Average weight of swine by production type (pounds)
Gilts	150
Wean to feeder	30
Wean to finish	115
Feeder to finish	135
Farrow to wean	433
Farrow to feeder	522
Farrow to finish	1,417

Qualitative information on the documented presence of tile drains at the CAFOs (appendix A3-3) was based on those either reported by the facility operator or identified by DWR facility inspectors; however, no specific information was available on the number or locations of documented tile drains at the facilities. Although there are no documented tile drains for some CAFOs, this may not be completely accurate because there are likely tile drains located at some facilities, the existence of which is unknown, and these would have gone unreported. The tile drain data are provided for informational purposes and are not considered to accurately reflect the extent to which subsurface tile drains may or may not be associated with the swine CAFO waste-manure application fields in the SW and SP study sites.

Available orthoimagery in Google Earth (http://www.google.com/earth/; accessed May 2012) was visually examined to identify the total number of lagoons and swine barns associated with each active swine CAFO and, of these, how many of the lagoons and barns were located within the watershed boundaries (appendix A3-3). Some of the CAFOs were located along the watershed drainage boundaries and, under these circumstances, overland runoff and groundwater flow from those facilities may be transported toward receiving streams both within and outside of the study watersheds. In these cases, the permit attribute data associated with CAFOs situated along the drainage boundaries were adjusted with a correction factor to allocate that fraction of the data deemed to be associated within the study sites (appendix A3-3). Where needed, the correction factor used to adjust the attribute data generally was taken as the ratio of swine barns located within the watershed to the total swine barns associated with the CAFO.

Attributes for the individual swine CAFOs, which reflect adjustments applied for total swine, total swine weight, available acres, PAN allowed, and PAN generated, are provided in appendix A3-4. This information was used to compute the total number of active swine CAFOs, lagoons, swine barns, swine animals and weight, available acres, allowable PAN, and generated PAN within each of the SW and SP watershed sites (appendix A3-5). Total watershed densities per square mile of swine barns, swine animals, swine weight (in tons), and available acres were determined as additional parameters for each site for use in evaluating the water-quality data.

Poultry CAFO Attributes

Available orthoimagery in Google Earth (http://www.google.com/earth/; accessed May 2012) was visually examined to identify apparent dry-litter poultry CAFOs and their associated number of poultry barns located within each watershed of the study sites. The SP sites were the only study sites determined to have one or more apparent dry-litter poultry CAFOs; these sites also contain one or more permitted active swine CAFOs. The apparent dry-litter poultry CAFOs were visually distinguished from the documented swine CAFOs on the basis of the presence of waste-storage lagoons at the permitted swine facilities and the absence of any waste-storage lagoons at the dry-litter poultry facilities. For verification purposes, a list of the apparent dry-litter poultry CAFOs identified for the 18 SP sites was provided to DWR for subsequent review by the North Carolina Department of Agriculture and Consumer Services, which indicated that the apparent dry-litter poultry CAFOs identified during this study were indeed active poultry facilities (Keith Larick, North Carolina Division of Water Resources, written commun., November 2012). No specific information on the operational characteristics (such as types and numbers of poultry raised, manure applications, or years of operation) for the dry-litter poultry CAFOs was publicly available for use in this study. Hereafter, the dry-litter poultry CAFOs at the study sites will be referred to as poultry CAFOs.

For this study, each cluster of poultry barns identified at the SP sites was considered to represent an individual poultry CAFO. Spatial coordinates and number of barns for the poultry CAFOs are provided in appendix A3-6. Each poultry CAFO was assigned a unique identifier, or field number, for use in this study. In some cases, adjacent poultry barn clusters may actually be part of the same operation. Similar to the process described previously for the swine CAFOs, in those cases where a poultry CAFO was located along the watershed drainage boundary, a prorated number of poultry barns was assigned to the CAFO to represent that fraction of the facility deemed to be within the watershed. The compiled information for the individual poultry CAFOs (appendix A3-6) was used to compute the total number of poultry CAFOs and poultry barns, as well as poultry barn density (barns per square mile), for each SP study site (appendix A3-7).

Data Collection

This section outlines procedures that were used to compile precipitation and streamflow monitoring data for examining hydrologic conditions in the study area. Sample collection procedures, laboratory analyses, and data quality-assurance practices are described for the water-quality data.

Precipitation and Streamflow

Precipitation data were obtained from four active USGS raingage monitoring stations (sites RG-01 through RG-04; table 3) in the Coastal Plain study area (fig. 2). Precipitation was measured at each site by using a tipping-bucket raingage that recorded precipitation at 15-minute intervals. Calibration checks were conducted semiannually on the raingages to ensure the accuracy of recorded data (U.S. Geological Survey, 2006). Precipitation data for sites RG-01, RG-02, RG-03, and RG-04 (table 3) are available from the USGS National Water Information System (NWIS) database (http://waterdata.usgs.gov/nc/nwis).

The precipitation data were used to better understand the extent to which each sampling date during the surface-water sampling periods was preceded by relatively wet or dry climatic conditions. For each raingage site, a cumulative total precipitation was computed for the 7-day period immediately preceding each date that samples were collected. Minimum, maximum, and mean values of the cumulative 7-day precipitation totals for the four raingage sites were determined for each sampling date for use in data analysis.

Ideally, instantaneous stream discharge would be measured to document streamflow conditions at the time water-quality samples are collected. However, the typical site conditions encountered during this study included low streamflow velocity coupled with varying degrees of submerged and floating aquatic vegetation within and along the stream channel. These conditions made it impractical to measure stream discharge during sample collections. Therefore, streamflow data were obtained from six active USGS streamgaging stations (sites SG-01 through SG-06; table 4) in the Coastal Plain study area (fig. 2) to describe regional hydrologic conditions during sampling periods. Streamflow data for the streamgage sites (table 4) are available from the USGS NWIS database (http://waterdata.usgs.gov/nc/nwis).

Table 3. Raingage monitoring sites in the North Carolina Coastal Plain study area used for collecting precipitation data.

[ID, identification; USGS, U.S. Geological Survey; NC, North Carolina]

Study site ID (see fig. 2)	USGS station number	USGS station name	Decimal latitude	Decimal longitude	Type of data collected
RG-01	355719077471345	Raingage at Tar River at NC 97 at Rocky Mount, NC	35.95536	–77.78683	Precipitation water quality
	02082585	Tar River at NC 97 at Rocky Mount, NC	35.95472	–77.78722	Continuous rainfall
RG-02	345006078493145	Raingage at Cape Fear River at Lock 3 near Tarheel, NC	34.83503	–78.82525	Precipitation water quality
	02105500	Cape Fear River at Wilm O Huske Lock near Tarheel, NC	34.83556	–78.82361	Continuous rainfall
RG-03	02084000	Tar River at Greenville, NC	35.61667	–77.37278	Continuous rainfall
RG-04	02105769	Cape Fear River at Lock 1 near Kelly, NC	34.40444	–78.29361	Continuous rainfall

Table 4. Streamgage monitoring sites in the North Carolina Coastal Plain study area used for compiling streamflow data.

[ID, identification; USGS, U.S. Geological Survey; NC, North Carolina; mi^2, square mile]

Study site ID (see fig. 2)	USGS station number	USGS station name	Decimal latitude	Decimal longitude	Drainage area (mi^2)
SG-01	0208111310	Cashie River at SR 1257 near Windsor, NC	36.04778	–76.98417	108
SG-02	02084160	Chicod Creek at SR 1760 near Simpson, NC	35.56167	–77.23083	45
SG-03	02091000	Nahunta Swamp near Shine, NC	35.48889	–77.80611	80.4
SG-04	02092500	Trent River near Trenton, NC	35.06417	–77.46139	168
SG-05	02108000	Northeast Cape Fear River near Chinquapin, NC	34.82889	–77.83222	599
SG-06	02134480	Big Swamp near Tarheel, NC	34.71028	–78.83639	229

Water-Quality Samples

Water-quality data compiled for the study include the analytical results for precipitation samples and surface-water samples. Precipitation samples were collected at raingage monitoring sites RG-01 and RG-02 from late July 2012 to early April 2013 for laboratory analyses. In this study, separate USGS station numbers are used for the precipitation water-quality data and the continuous rainfall data collected at monitoring stations RG-01 and RG-02 (table 3). The precipitation collectors were deployed for periods ranging from 2 days to 2 weeks to capture one or more rainfall events. The length of each deployment was based on the frequency and magnitude of rainfall events and the overall amount of rain that could be captured without overfilling the collection container. Clean sampling equipment was used for each deployment. Samplers were not deployed during periods of extreme cold to avoid freezing, which could compromise the analytical results.

Surface-water samples were collected at the 54 primary and 23 secondary study sites (table 1) for laboratory analyses. Samples at the primary sites were collected during six rounds of bimonthly sampling, during June, August, October, and December 2012, and February and April 2013. Samples were collected at the secondary sites once during the April 2013 sampling round. The number of days needed to collect samples during each round ranged from 3 to 6.

Water temperature, specific conductance, pH, DO, and barometric pressure were measured in the field during sample collections using instruments that were calibrated daily prior to sampling. Established, documented protocols were followed for collecting and processing samples for chemical analyses (U.S. Geological Survey, variously dated). Non-isokinetic methods were used for collecting samples because streamflow velocities generally were low. Samples were collected at the mid-depth of the water column at one or more points across the stream, depending on the stream width and type of road crossing (bridge or culverts). Subsamples collected from multiple points were composited into a single sample, representing the stream cross section.

Field equipment was cleaned between sampling sites (U.S. Geological Survey, variously dated). Samples were filtered and preserved in the field. A disposable 0.45-micron (µm) pore size capsule filter was used to process samples for major ions and filtered nutrient fractions. Samples collected for the determination of nitrogen-15/nitrogen-14 ($^{15}N/^{14}N$) and oxygen-18/oxygen-16 ($^{18}O/^{16}O$) isotopic ratios of nitrate plus (+) nitrite were filtered twice, first with a 0.45 µm capsule filter followed by a 0.20 µm disc filter, and subsequently frozen to prevent microbial degradation prior to laboratory analysis.

Nutrients and Major Ions

Surface-water samples were shipped to the USGS National Water Quality Laboratory (NWQL) in Lakewood, Colorado, for chemical analysis of nutrients and major ions. Methods and reporting levels (RL) for each measured analyte (table 5) remained consistent for all samples analyzed during the study. Unfiltered samples were analyzed for concentrations of total ammonia+organic N and total P. Filtered samples were analyzed for concentrations of dissolved ammonia, dissolved nitrate+nitrite, and dissolved orthophosphate (ortho-P). Filtered samples also were analyzed to determine concentrations of dissolved calcium, chloride, magnesium, potassium, sodium, and sulfate.

The water-quality data for the surface-water samples are presented in appendix A4-1. One dataset includes water-quality results for all samples collected at the primary sites. The second dataset includes results for samples collected during the April 2013 sampling at the 9 primary sites and their 23 secondary sites. Analytical concentrations for the nitrogen species are reported in milligrams per liter as N and concentrations for ortho-P and total P are reported in milligrams per liter as P. The water-quality data also are available from the USGS NWIS database (http://waterdata.usgs.gov/nc/nwis).

Values for total organic N and total N (appendix A4-1) were computed from three directly measured nitrogen fractions (table 5). Total organic N was computed by subtracting dissolved ammonia from total ammonia+organic N. Total N was computed by summing total ammonia+organic N and dissolved nitrate+nitrite. If one of the underlying constituents used in computing total organic N or total N had a left-censored (<) value, then the < remark code was carried forward with the computed value. Although the < remark codes were carried forward with the total organic N and total N, they were ignored for the purpose of data evaluations in this study because the censoring levels associated with dissolved ammonia (RL = 0.010 mg/L) and dissolved nitrate+nitrite (0.04 mg/L) have minimal influence on the calculated values for total organic N and total N, respectively. Thus, examinations of the total organic N and total N data were based on the concentrations as reported in appendix A4-1 without regard to any < remark codes associated with the computed values. It is of note that, by default, total organic N and total N concentrations retrieved from the NWIS database retain the < remark code if one of the underlying constituents is left-censored. The handling of censored data is left to the discretion of data users.

Table 5. Nutrients and major ions measured in surface-water samples.

[N, nitrogen; P, phosphorus; mg/L, milligram per liter; EPA, U.S. Environmental Protection Agency; APHA, American Public Health Association]

Analyte	Reporting level, in mg/L	Analytical reference
Nutrients		
Ammonia as N, dissolved	0.010	Fishman (1993)
Ammonia + organic nitrogen as N, total	0.07	Patton and Truitt (2000)
Nitrate + nitrite as N, dissolved	0.04	Patton and Kryskalla (2011)
Orthophosphate as P, dissolved	0.004	Fishman (1993)
Phosphorus as P, total	0.004	USEPA (1993)
Major ions		
Calcium, dissolved	0.022	Fishman (1993)
Chloride, dissolved	0.06	Fishman and Friedman (1989)
Magnesium, dissolved	0.011	Fishman (1993)
Potassium, dissolved	0.03	APHA (1998)
Sodium, dissolved	0.06	Fishman (1993)
Sulfate, dissolved	0.09	Fishman and Friedman (1989)

Stable Isotopes

Surface-water and precipitation samples were shipped to the USGS Reston Stable Isotope Laboratory (RSIL) in Reston, Virginia, for analysis of stable isotopes by using a continuous flow isotope-ratio mass spectrometer. Surface-water samples were analyzed for stable isotope ratios of water (hydrogen-2/hydrogen-1 [$^2H/^1H$] and $^{18}O/^{16}O$) and (or) stable isotope ratios of dissolved nitrate+nitrite ($^{15}N/^{14}N$ and $^{18}O/^{16}O$). Precipitation samples were analyzed for stable isotope ratios of water ($^2H/^1H$ and $^{18}O/^{16}O$).

Stable isotope ratios are reported using the delta (δ) notation in units of parts per thousand (denoted as per mil or ‰) relative to a standard of known composition according to the following equation:

$$\delta\ (‰) = (R_{samp}/R_{stand} - 1) * 1{,}000 \qquad (1)$$

where R_{samp} and R_{stand} are the ratios of the heavy to light isotope ($^2H/^1H$, $^{18}O/^{16}O$, or $^{15}N/^{14}N$) in the sample and standard, respectively.

Stable isotopes of water (δ^2H and $\delta^{18}O$) were analyzed in surface-water samples collected at the primary sites (appendix A4-1) and in precipitation samples collected at sites RG-01 and RG-02 (appendix A4-2) following methods outlined in Révész and Coplen (2008a, b). Results for δ^2H and $\delta^{18}O$ of water are reported with a 2-sigma (σ) uncertainty of ±2 ‰ and ±0.2 ‰, respectively. Analysis of stable isotopes of dissolved nitrate+nitrite ($\delta^{15}N$ and $\delta^{18}O$) in surface-water samples was based on the microbial denitrifier method (Sigman and others, 2001; Casciotti and others, 2002; Coplen and others, 2012). Measurements of $\delta^{15}N$ and $\delta^{18}O$ of dissolved nitrate+nitrite generally were performed on samples for the primary and secondary study sites with nitrate+nitrite concentrations greater than or equal to the RL of 0.04 mg/L (appendix A4-1). The $\delta^{15}N$ and $\delta^{18}O$ results are reported with 2-σ uncertainties of ±0.5 ‰ and ±1.0 ‰, respectively, when analyzed samples had nitrate+nitrite concentrations greater than or equal to 0.06 mg/L as N; the uncertainties are doubled for samples with nitrate+nitrite concentrations less than 0.06 mg/L as N.

An important issue to note regarding $\delta^{18}O$ analyses with the denitrifier method is that the $\delta^{18}O$ values generated for combined nitrate+nitrite may be underestimated if samples contain appreciable amounts of nitrite, yet the nitrite contributions to the $\delta^{18}O$ results are not taken into account (Casciotti and others, 2007). When available, measured concentrations of nitrite are used to make applicable corrections to the $\delta^{18}O$ results (Casciotti and McIlvin, 2007; Casciotti and others, 2007). In this study, however, samples were analyzed for combined nitrate+nitrite concentrations rather than individual concentrations of nitrate and nitrite. Therefore, the $\delta^{18}O$ values of nitrate+nitrite reported in appendix A4-1 may underestimate actual values. The extent to which the results may have been biased by unaccounted-for nitrite in the samples is unknown.

Although nitrite concentrations were not determined for samples collected during this study, nitrite typically constitutes a relatively small amount (<10 percent) of the overall nitrate+nitrite observed in streams in the North Carolina Coastal Plain. With nitrite likely representing less than 10 percent of the measured nitrate+nitrite in the study samples, the potential low bias associated with the $\delta^{18}O$ values determined for nitrate+nitrite should be relatively muted. The presence of unrecognized nitrite in samples with the lowest concentrations of nitrate+nitrite (near the analytical RL of 0.04 mg/L) would likely have the most pronounced bias on the nitrate+nitrite $\delta^{18}O$ results. Therefore, evaluations of the stable isotope data ($\delta^{15}N$ and $\delta^{18}O$) for dissolved nitrate+nitrite in this study were focused on those samples having nitrate+nitrite concentrations greater than or equal to 0.100 mg/L in an effort to reduce the potential uncertainties associated with the nitrate+nitrite $\delta^{18}O$ results.

Quality Assurance

Quality-control samples, including field blanks and replicate samples, were collected to document potential bias and variability in data that may result during the collection, processing, shipping, and handling of environmental samples (U.S. Geological Survey, variously dated). Field blanks were collected using inorganic-free water processed in the field with the same equipment used for the environmental samples. Field blanks help to identify contamination resulting from improperly cleaned equipment, field sampling activities and exposure, and laboratory practices. Overall, the results of the field blanks did not indicate any systematic or substantial quality-assurance issues with the environmental data. Replicate samples were collected to help document the variability in data results associated with sample collection, processing, and laboratory analysis. No quality-assurance problems were identified for the environmental dataset based on the replicate samples.

A total of 26 field blanks (appendix A4-3) and 26 replicate samples (appendix A4-4) were collected during surface-water sampling. One replicate sample was obtained during the collection of precipitation samples at site RG-02. Approximately 13 percent of the total number of samples collected during the study were quality-control samples. All surface-water blank and replicate samples were analyzed for nutrients and major ions. Stable isotopes of water (δ^2H and $\delta^{18}O$) were measured in replicate samples collected at the primary study sites and in the one precipitation replicate. Stable isotopes of nitrate+nitrite ($\delta^{15}N$ and $\delta^{18}O$) were measured in most surface-water replicate samples having detectable concentrations of nitrate+nitrite above the RL of 0.04 mg/L.

Most constituents were below analytical RLs in the field blanks (appendix A4-3). Magnesium, sodium, potassium, and sulfate were not detected in any blank samples. Concentrations of calcium and chloride in one blank sample (0.037 and 0.11 mg/L, respectively) were an order of magnitude lower than calcium and chloride concentrations measured in environmental samples (appendix A4-1). For nutrients, ortho-P was not detected in any blanks. Nitrate+nitrite was detected in one blank sample at a concentration (0.070 mg/L) just above the RL of 0.040 mg/L. Total phosphorus was also detected in one blank sample at a concentration (0.005 mg/L) just above the RL of 0.004 mg/L. Ammonia+organic N was detected in about 12 percent of the blank samples (3 of 26) at concentrations of 0.08 to 0.14 mg/L; however, there was no indication of systematic bias that would affect the environmental results. All ammonia+organic N concentrations measured for the environmental samples (appendix A4-1) exceeded the greatest concentration of 0.14 mg/L detected in the blank samples (appendix A4-3).

Ammonia was detected in about 27 percent of the blank samples (7 of 26) at concentrations of 0.011 to 0.020 mg/L. Blank samples frequently may become contaminated with ammonia when exposed to the atmosphere—both in the field and laboratory (Fishman, 1993). This is especially apparent when blanks are analyzed using low-level techniques, as was done in this study. Although some low-level contamination of ammonia may have occurred, any effects on the environmental data are considered minimal. Of the 344 total environmental samples, 319 had concentrations of ammonia above the analytical RL of 0.010 mg/L (appendix A4-1). Approximately 89 percent of these samples (283 of 319) had ammonia concentrations that exceeded the highest ammonia concentration of 0.020 mg/L detected in the blank samples (appendix A4-3). In addition, 75 percent of the samples (241 of 319) had ammonia concentrations greater than 0.040 mg/L, more than twice the highest concentration of 0.020 mg/L detected in the blanks.

Replicate samples were used to assess the overall precision of the entire sample collection, handling, and analysis approach. A statistical summary of the relative percent difference (RPD) determined for each analyte for all paired environmental and replicate samples is provided in table 6. The RPDs in analyte concentrations rarely exceeded 15 percent. Exceedances above 15 percent were limited to one or two replicate sample pairs for sulfate, nitrate+nitrite, total P, and $\delta^{18}O$ of nitrate+nitrite. The mean and median RPDs were less than about 5 percent for all the measured constituents (table 6), which indicates very good agreement between the environmental and replicate samples.

Prior to data analysis, the water-quality data (appendix A4-1) were reviewed to identify any obvious outliers or potential issues in the sample results. Site SW-02 was noted to have the highest measured values for specific conductance and the major ions, by up to an order of magnitude, among any of the study sites (appendix A4-1). Nutrient results for site SW-02 were similar to the other study sites. Site SW-02 contains both one small swine CAFO (1 barn with 4,330 swine) and a granite quarry in the headwater area of the watershed (appendix fig. A1-20). The very high ion concentrations for site SW-02 are suspected of being influenced by mining activities associated with the quarry; therefore, the results for specific conductance, calcium, magnesium, sodium, potassium, chloride, and sulfate for this site were excluded from data analyses in this report. Results for the August 26, 2012, sample collected at site BK-01 (appendix A4-1) were excluded from data evaluations because they were considered to be influenced by backwater conditions from the adjacent Roanoke River (appendix fig. A1-1) when storm runoff increased river levels by about 8 ft between August 25–26, 2012. In addition, the $\delta^{2}H$ and $\delta^{18}O$ isotopic results for sites BK-17 (appendix fig. A1-17) and SW-11 (appendix fig. A1-29), which were influenced by upstream impoundments, were considered atypical and also were excluded from the data evaluations.

Table 6. Statistical summary of relative percent differences in analyte concentrations for the environmental and replicate sample sets.

[RPD, relative percent difference; %, percent; N, nitrogen; P, phosphorus; δ, delta]

Analyte	Number of paired replicate samples[1]	Statistical measure: Minimum RPD (%)	Maximum RPD (%)	Mean RPD (%)	Median RPD (%)
Calcium, dissolved	26	0.0	5.6	1.4	1.0
Magnesium, dissolved	26	0.0	5.7	1.3	1.2
Sodium, dissolved	26	0.0	4.6	2.0	1.9
Potassium, dissolved	26	0.0	8.3	2.7	2.2
Chloride, dissolved	26	0.0	1.8	0.3	0.0
Sulfate, dissolved	26	0.0	16.6	1.2	0.4
Ammonia + organic nitrogen as N, total	26	0.0	10.7	2.6	1.4
Ammonia as N, dissolved	22	0.0	5.6	1.8	1.1
Nitrate + nitrite as N, dissolved	19	0.0	18.6	5.3	1.9
Orthophosphate as P, dissolved	21	0.0	14.0	2.8	1.4
Phosphorus as P, total	26	0.0	35.0	4.1	1.4
δ Hydrogen-2 of water, dissolved	25	0.0	6.2	2.7	2.6
δ Oxygen-18 of water, dissolved	25	0.0	2.4	0.8	0.7
δ Nitrogen-15 of nitrate + nitrite, dissolved	18	0.2	10.8	1.6	0.7
δ Oxygen-18 of nitrate + nitrite, dissolved	18	0.0	28.8	3.8	1.5

[1]Relative percent differences were computed when both samples in a pair had concentrations above analytical reporting levels.

Statistical Analyses

Statistical evaluations of the study data included the use of analysis of variance (ANOVA) tests and pair-wise multiple-comparison tests (Helsel and Hirsch, 1992). One-way ANOVA tests were used to test for significant differences in watershed attributes, such as basin drainage area, among the three watershed land-use types (BK, SW, and SP). Two-way, or multifactor, ANOVA tests were used to test for significant differences in surface-water constituents on the basis of sampling period and (or) land-use type. Because most of the study data are non-normally distributed, a non-parametric approach was used in which the ANOVA tests were performed on rank-transformed data to assess differences between groups. The use of statistical analyses that rely on data ranks, rather than actual data values, also is appropriate for examining water-quality data containing left-censored "<" values when the < values for a given constituent are censored to the same analytical RL (Bonn, 2008). Left-censored values reported for ammonia, nitrate+nitrite, and ortho-P in surface-water samples (appendix A4-1) were set equal to their respective RLs prior to ranking the data for use in statistical analyses.

Constituent concentrations were ranked for all samples collected from the 54 primary study sites during the 6 sampling periods. A two-way ANOVA test was then performed on the ranks of the concentration data to test for differences based on the grouping (or explanatory) variables of sampling period (June, August, October, and December in 2012, and February and April in 2013) and land-use type (including the 18 BK, 18 SW, and 18 SP sites). By evaluating sampling period and land-use type simultaneously, the effect of one explanatory variable can be measured while compensating for the other. The test compares the mean ranks of the constituent concentrations in the treatment groups to the overall mean rank for the entire dataset and determines whether there is an influential effect based on sampling period, land-use type, and (or) the combined interaction between sampling period and land-use type.

The ANOVA results for a given constituent may indicate that a statistically significant difference in the mean ranks of the concentrations exists among a particular treatment group (such as land-use type); however, it does not specify which of the group treatments (such as BK, SW, and SP site types) are different. Those constituents with significant differences identified by the ANOVA tests were analyzed further with Tukey pair-wise multiple-comparison tests to identify which sampling period comparison pairs and (or) land-use type comparison pairs had statistically different means in their ranked values. The ANOVA and pair-wise multiple-comparison analyses, which were tested at the 95 percent confidence level ($P=0.05$), were conducted using the S-Plus software suite (by TIBCO Software Inc.).

Relations of environmental variables among study sites identified as either being influenced or not influenced by CAFO waste manures were modeled using classification tree analyses (Breiman and others, 1984). Classification tree-based modeling is an exploratory technique for uncovering structure in the data. The classification tree models evaluate the response variable, or defined category (such as sites without CAFO effects and sites with CAFO effects), and the associated predictor variables (such as environmental attributes) to identify the predictor variables that best partition, or split, the response variable into increasingly homogeneous subsets. The resulting classification tree is simplified (pruned) by removing splits that do not contribute to a reduction in model error. The classification tree analyses were conducted using the S-Plus software suite (by TIBCO Software Inc.).

Characterization of Watershed Settings and Hydrologic Conditions

Information compiled on land cover, hydrologic soil groups (HSGs), and CAFO attributes was used to examine watershed settings among the study sites. Regional information on precipitation and streamflows and measurements of stable isotopes of water in collected samples were used to characterize general hydrologic conditions during the six water-quality sampling periods.

Watershed Settings

Land cover, HSGs, and CAFO attributes (appendixes A2-1, A2-2, A3-5 and A3-7) for the primary study sites were evaluated to identify similarities or differences in watershed settings among the BK, SW, and SP site groups. Land cover and HSGs were examined among all three site groups. Attributes for swine CAFOs were examined only for the SW and SP groups. A statistical summary of watershed attributes in each site group is provided in table 7.

The overall results of the statistical analyses indicate that the general watershed settings of the study sites are comparable among the BK, SW, and SP site groups. The primary difference between the land-use groups is that the BK sites contain no CAFOs, the SW sites contain swine CAFOs, and the SP sites contain both swine and poultry CAFOs. ANOVA tests indicated few statistical differences in land cover and HSGs among the BK, SW, and SP site groups (table 8). Shrub land cover, HSG total A, and HSG D were the only watershed attributes that were significantly different ($P<0.05$) between some site groups. In addition, the ANOVA tests also did not identify any statistically significant differences ($P<0.05$) in any of the swine CAFO attributes examined between the SW and SP site groups (table 8). In other words, the SW and SP groups are similar with respect to swine CAFO attributes in the watersheds but differ in that poultry CAFOs also are present only in the SP watersheds (table 7).

Table 7. Statistical summary of watershed attributes by land-use type.

[n, number; mi^2, square mile; %, percent; CAFO, concentrated animal feeding operation; PAN, plant available nitrogen; SSLW, steady state live weight; na, not applicable]

Watershed attribute (unit)	Background (BK) sites (n = 18)			Swine (SW) sites (n = 18)			Swine and poultry (SP) sites (n = 18)		
	Minimum	Median	Maximum	Minimum	Median	Maximum	Minimum	Median	Maximum
Land cover and hydrologic soil groups									
Drainage area (mi^2)	3.1	5.9	14.9	1.2	3.8	15.8	1.4	5.0	17.5
Developed (%)	0.6	4.6	10.0	1.2	4.3	9.1	1.0	4.0	6.4
Forested (%)	9.4	27.7	50.2	8.7	23.0	44.7	9.9	22.6	48.5
Shrubs (%)	2.7	6.8	17.0	4.1	10.5	23.5	6.4	11.5	16.8
Crops (%)	16.8	38.6	64.4	18.4	43.0	69.8	17.1	44.2	70.0
Grassland (%)	0.2	3.4	12.3	0.2	1.9	9.9	0.7	1.3	11.8
Wetlands (%)	4.3	15.6	55.0	6.3	13.3	27.3	3.7	12.8	21.2
Hydrologic soil group total A (%)	0.0	3.5	32.8	0.0	7.2	30.9	0.6	16.2	55.5
Hydrologic soil group total B (%)	12.6	58.0	88.3	27.9	52.6	87.6	13.8	54.0	86.0
Hydrologic soil group total C (%)	0.0	14.4	33.2	1.2	23.5	52.8	0.3	17.2	56.1
Hydrologic soil group D (%)	1.1	13.5	58.0	1.2	7.2	29.5	0.0	6.5	64.1
CAFO attributes									
Permitted active swine CAFOs (total)	na	na	na	1.0	1.5	12	1.0	3.0	10
Total allowable PAN (pounds)	na	na	na	2,347	38,760	132,355	2,743	36,239	253,906
Total generated PAN (pounds)	na	na	na	1,472	21,779	74,319	1,870	19,144	114,271
Swine lagoons (total)	na	na	na	1	3	18	1	5	15
Swine barns (total)	na	na	na	1	13	45	4	15	59
Swine animals (total)	na	na	na	1,200	9,225	65,532	550	9,928	67,797
Total swine SSLW (tons)	na	na	na	65.0	956	3,067	74.3	847	4,719
Available swine acres (total)	na	na	na	7.2	156	610	10.0	150	1,413
Swine barn density (barn/mi^2)	na	na	na	0.1	2.4	13.5	0.9	2.9	9.6
Swine animal density (animal/mi^2)	na	na	na	370	2,448	10,388	242	2,394	9,139
Swine weight density (ton/mi^2)	na	na	na	7.3	180	701	16.3	146	625
Swine acre density (acre/mi^2)	na	na	na	0.8	39	176	2.2	27	187
Active poultry CAFOs (total)	na	na	na	na	na	na	1.0	1.0	8
Poultry barns (total)	na	na	na	na	na	na	1.0	4.0	35
Poultry barn density (barn/mi^2)	na	na	na	na	na	na	0.2	0.9	5.7

Table 8. Summary results of the ANOVA and Tukey multiple-comparison tests of watershed attributes by land-use type.

[The null hypothesis was that the mean ranks of each distribution were the same. ANOVA, analysis of variance; *, indicates significant difference ($P < 0.05$); ns, no significant differences between site types based on ANOVA test; BK, background site type; SW, swine site type; SP, swine and poultry site type; CAFO, concentrated animal feeding operation; PAN, plant available nitrogen; SSLW, steady state live weight]

Watershed attribute	ANOVA test	Tukey multiple-comparison test
	p-value	Site-type comparison pairs significant at $\alpha = 0.05$
Land cover and hydrologic soil groups		
Drainage area	0.0901	ns
Developed	0.7661	ns
Forested	0.3564	ns
Shrub	0.0008*	BK-SW and BK-SP
Crops	0.2529	ns
Grassland	0.0920	ns
Wetlands	0.3126	ns
Hydrologic soil group total A	0.0005*	BK-SP and SW-SP
Hydrologic soil group total B	0.4401	ns
Hydrologic soil group total C	0.6864	ns
Hydrologic soil group D	0.0358*	BK-SP
Swine CAFO attributes		
Permitted active swine CAFOs	0.0768	ns
Total allowable PAN	0.7332	ns
Total generated PAN	0.5980	ns
Swine lagoons	0.2239	ns
Swine barns	0.2530	ns
Swine animals	0.3183	ns
Total swine SSLW	0.6870	ns
Available swine acres	0.8770	ns
Swine barn density	0.4008	ns
Swine animal density	0.9014	ns
Swine weight density	0.8043	ns
Swine acre density	0.6198	ns

Hydrologic Conditions During Sampling

Typical site conditions during sampling at most of the study sites included low streamflow velocity coupled with varying degrees of submerged and floating aquatic vegetation within and along the stream channel. Because of these conditions, it was not feasible to measure stream discharge at the study sites during sampling. Therefore, regional precipitation and streamflow data collected at active USGS monitoring stations (tables 3, 4; fig. 2), as well as δ^2H and $\delta^{18}O$ isotopic results for precipitation and stream samples, were used to assess general hydrologic conditions in the study area during the six sampling periods (June, August, October, and December in 2012, and February and April in 2013).

Precipitation

Regional precipitation measured during the study at the raingage monitoring sites (table 3; fig. 2) was slightly below normal levels. The annual precipitation recorded from May 1, 2012, through April 30, 2013, at raingage sites RG-01 (35.77 inches [in.]), RG-02 (40.49 in.), RG-03 (47.98 in.), and RG-04 (48.34 in.) has an average value of 43.14 in. Note that the annual values for RG-01 and RG-03 represent a lower limit because these sites had 17 days and 3 days, respectively, of missing data where precipitation was not recorded. The average annual precipitation is 45.60 in. if site RG-01 is excluded. Normal average annual precipitation in the study area, based on the 30-year period 1971–2000, ranges from about 46 to 52 in. (State Climate Office of North Carolina, n.d.).

Mean 7-day precipitation totals were used to document the differences in the amount of rainfall in the study area among the water-quality sampling periods (table 9; fig. 3). Overall, antecedent field conditions for the sampling periods were wetter for August and February, intermediate for June and April, and drier for October and December. It is important to note that for a given sampling event, there may have been considerable local differences in precipitation amounts among the study sites. For example, scattered thunderstorms occurred throughout the study area for the August period. The uneven distribution of precipitation is reflected by the higher standard deviations associated with the mean 7-day precipitation totals for August relative to the other sampling periods (table 9). The February sampling dates had mean 7-day precipitation totals similar to the August sampling dates, yet the lower standard deviations suggest that precipitation was more uniform across the study area during the February sampling event.

Table 9. Summary of the cumulative 7-day precipitation totals preceding each sample collection date based on raingage monitoring sites RG-01, RG-02, RG-03, and RG-04 (site locations in figure 2 and table 3).

Sample date	Number of primary study sites sampled	7-day precipitation total (inches)			
		Minimum	Maximum	Mean	Standard deviation
06/13/12	10	0.20	0.83	0.51	0.32
06/14/12	12	0.20	0.83	0.51	0.32
06/15/12	8	0.20	0.83	0.51	0.32
06/18/12	12	0.20	0.83	0.51	0.32
06/19/12	12	0.11	0.46	0.23	0.20
08/26/12	22	1.10	3.18	2.01	0.89
08/27/12	23	1.13	2.39	1.80	0.52
08/28/12	8	1.04	2.33	1.72	0.53
10/21/12	14	0.12	0.18	0.16	0.03
10/22/12	17	0.12	0.18	0.16	0.03
10/23/12	17	0.00	0.08	0.04	0.04
10/24/12	4	0.00	0.08	0.03	0.04
12/09/12	13	0.01	0.17	0.07	0.07
12/10/12	23	0.01	0.17	0.08	0.07
12/11/12	14	0.01	0.17	0.08	0.07
12/12/12	4	0.01	0.25	0.10	0.10
02/11/13	19	1.51	1.88	1.70	0.19
02/12/13	24	1.57	2.11	1.84	0.24
02/13/13	11	1.57	2.25	1.91	0.28
04/17/13	2	0.67	0.76	0.73	0.04
04/18/13	7	0.67	0.76	0.73	0.04
04/19/13	2	0.67	0.76	0.73	0.04
04/21/13	9	0.76	0.94	0.84	0.08
04/22/13	21	0.76	0.94	0.84	0.08
04/23/13	13	0.59	1.13	0.81	0.23

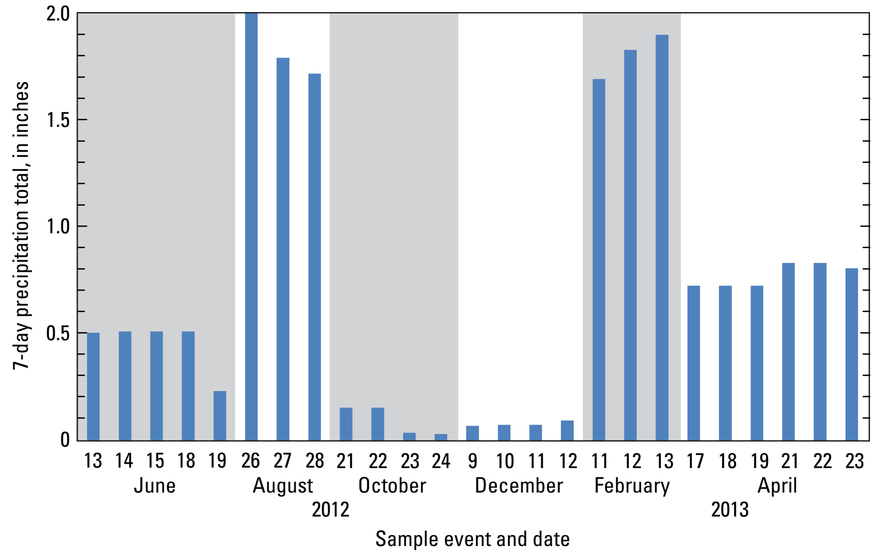

Figure 3. Mean cumulative 7-day precipitation totals preceding each sample collection date based on raingage monitoring sites RG-01, RG-02, RG-03, and RG-04 (site locations in figure 2 and table 3).

Streamflow

Relative differences in regional streamflow conditions during the water-quality sampling periods were inferred from streamflow records from six streamgage sites distributed throughout the study area (figs. 2, 4). The streamgage sites represent basin drainage areas ranging from 45 to 599 mi^2. Drainage areas for the primary study sites are considerably smaller, ranging from 1.2 to 17.5 mi^2. Although the magnitude of streamflow and the duration and timing of peak streamflows likely differ between the streamgage sites and the study sites, the hydrographs are useful indicators of relative streamflow trends throughout the study area during the sampling periods and the entire study period.

Streamflow conditions during most of the sampling periods were similar to or higher than historical streamflow conditions in the study area. Daily mean streamflows at the six streamgage sites during the study period (May 2012 through April 2013) are shown relative to long-term median daily mean streamflows for the 25-year period from May 1988 through April 2013 (fig. 4). In general, streamflows for the June, October, and April sampling periods were fairly similar to the long-term median values. Streamflows for the August and February periods tended to be substantially higher, and streamflows for the December period tended to be substantially lower relative to historical conditions.

Streamflow conditions varied among the six sampling periods (fig. 4). Compared to other sampling periods, streamflow conditions were relatively higher during the August and February sampling periods when precipitation amounts in the study area were higher (fig. 3) and overland transport of water to the streams was greater. The intermediate to lower streamflow conditions for the June, October, December, and April sampling periods reflect less precipitation and overland transport of water to the streams and a larger component of streamflow derived from groundwater compared to the August and February periods. The typically higher and more sustained stream-baseflow conditions (fig. 4) observed during the winter and early spring months (generally January to April) reflect greater groundwater discharge and likely higher inputs from field drainage ditches when the water table in the surficial aquifers is high. Variations in stream water quality at the study sites among sampling periods with higher versus lower relative streamflows may reflect relative differences in source contributions of water-quality constituents delivered through groundwater discharge and overland runoff.

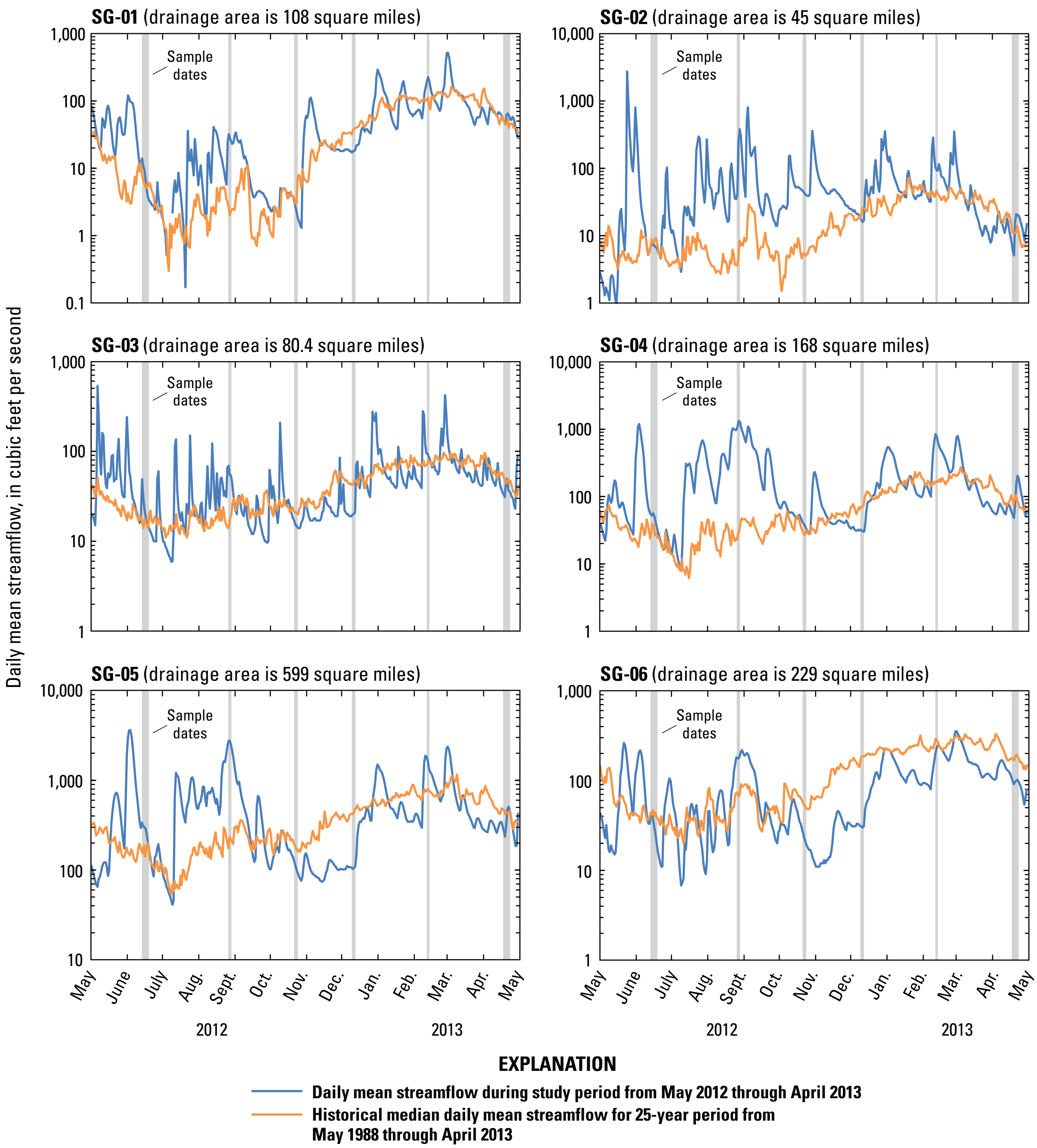

Figure 4. Streamflow hydrographs at sites SG-01, SG-02, SG-03, SG-04, SG-05, and SG-06 showing dates water-quality samples were collected during the study and historical median daily mean streamflows (site locations in figure 2 and table 4).

Water Stable Isotopes

Stable isotopes of water (δ^2H and $\delta^{18}O$) in precipitation and stream samples also were used to characterize general hydrologic conditions during the sampling periods. The δ^2H and $\delta^{18}O$ data for precipitation samples collected from July 2012 to April 2013 at rainfall monitoring sites RG-01 and RG-02 (fig. 2; appendix A4-2) were used to create a local meteoric water line (LMWL) for the Coastal Plain study area (fig. 5). The LMWL is represented by the linear relation between the δ^2H and $\delta^{18}O$ isotopic compositions in the precipitation samples:

$$\delta^2H = 8.33 * \delta^{18}O + 16.75 \quad (2)$$

The slope of 8.33 for the LMWL determined in this study is similar to the meteoric water line (MWL) equation ($\delta^2H = 8.29 * \delta^{18}O + 10.94$) determined by Kendall and Coplen (2001) using average values of surface-water samples obtained from 391 sites throughout the United States and Puerto Rico.

The δ^2H and $\delta^{18}O$ isotopic compositions of the samples collected at the primary sites (appendix A4-1) were compared to the LMWL to examine general differences in stream hydrologic conditions during the sampling periods (fig. 6). In general, surface-water samples with δ^2H and $\delta^{18}O$ values that correspond to the LMWL indicate that water in the streams reflects more recent inputs of precipitation to the land surface, which ultimately reaches the streams through runoff and groundwater discharge, that has undergone little fractionation. Samples with δ^2H and $\delta^{18}O$ values that plot along a line with a slope lower than the LMWL can be an indication that post-rainfall processes, commonly evaporation, altered the isotopic composition of the stream water prior to sample collection (Kendall and Coplen, 2001). As surface water evaporates, there is a preferential release of the lighter ^1H and ^{16}O isotopes to the atmosphere, which increases the δ^2H and $\delta^{18}O$ values of the remaining stream water; the values become increasingly more positive as evaporation proceeds.

During the six sampling periods, the δ^2H and $\delta^{18}O$ values for the February 2013 stream samples corresponded most closely to the LMWL (fig. 6*E*), reflecting the recent inputs of overland runoff when evaporation was least likely to have occurred (figs. 3, 4). The regression line for the February 2013 samples, with a slope of 6.97, almost paralleled the LMWL. For reference purposes, the regression line for the February 2013 data was superimposed on each of the δ^2H and $\delta^{18}O$ isotopic plots for the other five periods (fig. 6) to relate the isotopic compositions for those periods to the February period. The δ^2H and $\delta^{18}O$ values for the August 2012 samples plotted along a line with a slope of 6.08 (fig. 6*B*) that was just below the slope of 6.97 for the February 2013 period. The August samples had the largest observed range in δ^2H values (–12.3 to –37.3 ‰) and $\delta^{18}O$ values (–2.3 to –6.5 ‰). The August samples in the lower part of the regression line had isotopic signatures similar to the LMWL, indicating that stream water at some of the sites had received recent inputs of overland runoff and was minimally influenced by evaporation. August samples in the upper part of the regression line had more positive isotope δ^2H and $\delta^{18}O$ values that diverged to the right of the LMWL (fig. 6*B*), reflecting increased effects of evaporation and a lack of recent runoff at some of the sites sampled during August.

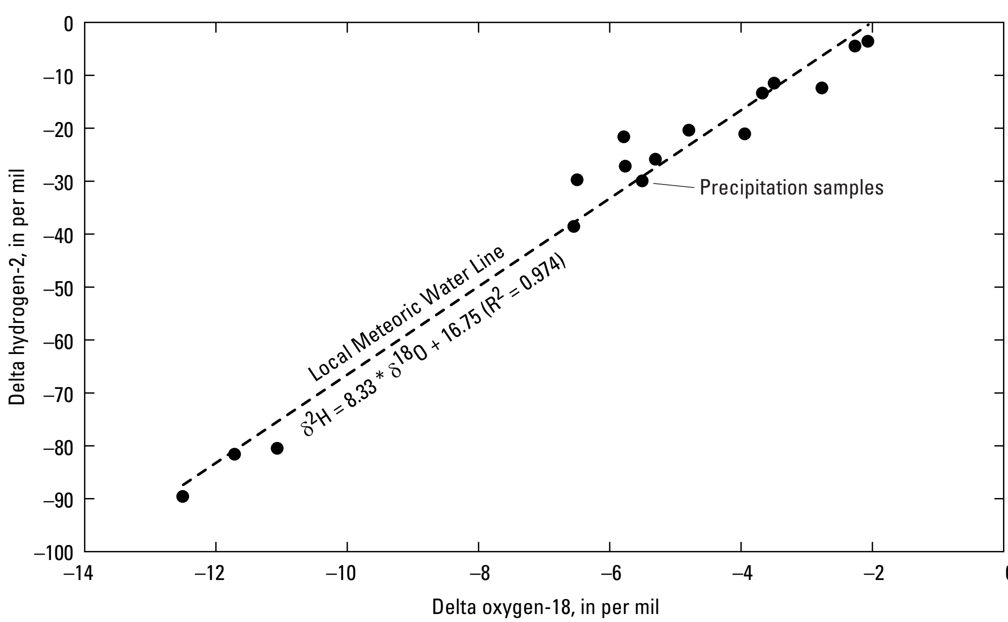

Figure 5. Comparison of delta oxygen-18 to delta hydrogen-2 isotope values in precipitation samples collected from July 2012 to April 2013 at raingage sites RG-01 and RG-02 in the Coastal Plain study area.

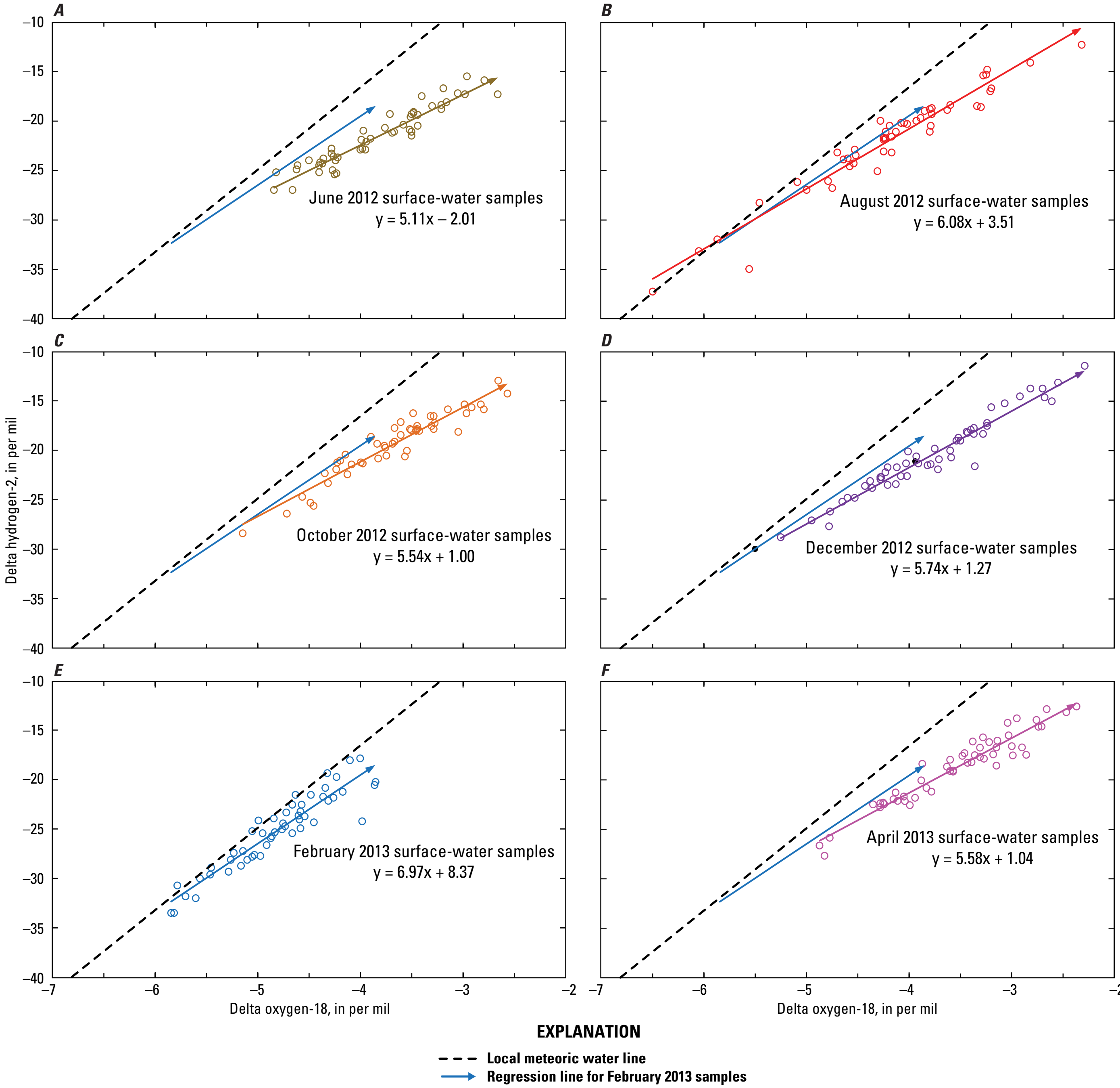

Figure 6. Comparisons of delta oxygen-18 to delta hydrogen-2 isotope values of surface-water samples for the (*A*) June 2012, (*B*) August 2012, (*C*) October 2012, (*D*) December 2012, (*E*) February 2013, and (*F*) April 2013 sampling periods relative to the local meteoric water line.

More pronounced effects of evaporation on the isotopic compositions at the stream sites were noted for the June, October, and December 2012 periods and the April 2013 period where the δ^2H and $\delta^{18}O$ values, with regression line slopes ranging from 5.11 to 5.74, plotted farthest away from the LMWL (fig. 6). These results support the previous discussion of the precipitation and streamflow data, which implied that streamflow conditions were relatively higher during the August and February periods as a result of increased rainfall and overland runoff (figs. 3, 4). Evaporation appeared to have a more influential effect on the surface-water δ^2H and $\delta^{18}O$ compositions during the June, October, December, and April periods. These periods were characterized by intermediate to lower streamflow conditions when there was less rainfall runoff to the streams and proportionally more input from discharging groundwater.

Comparison of Water-Quality Data by Sampling Period and Land-Use Type

Two-way ANOVA and multiple-comparison statistical tests were performed to characterize differences in stream water quality among the sampling periods (June, August, October, and December in 2012, and February and April in 2013) and watershed land-use types (BK, SW, and SP). Many of the water-quality properties and constituents were significantly influenced (ANOVA $P<0.05$) by one or both of the explanatory variables (sampling period and (or) land-use type) but there were no effects due to their combined interaction (sampling period:land-use type) (table 10). The lack of interaction indicates that the effects of sampling period and land-use type for a given constituent are independent; in other words, the effect of sampling period is the same across all land-use types and the effect of land-use type is the same across all sampling periods.

Seasonal and Flow-Related Water-Quality Differences

All of the water-quality properties and constituents, except calcium and the nitrate+nitrite isotopes ($\delta^{15}N$ and $\delta^{18}O$), had significant (ANOVA, $P<0.05$) differences among the sampling periods (table 10) based on data collected at the 54 primary sites. Differences reflected seasonal and hydrologic variations, as well as instream processes. Statistical summaries, by sampling period, of the original (non-ranked transformed) water-quality data are provided in tabular (table 11) and graphical formats (fig. 7) to aid the discussion. Figure 7 contains box plots for properties and constituents with significant differences (ANOVA $P<0.05$) among sampling periods; results of the multiple-comparison tests among the periods are denoted along the top of the plots. Rather than scrutinizing individual comparison pairs, the following discussion focuses on patterns among the sampling periods that reflect seasonal and hydrologic influences on water quality. Although ANOVA indicated a significant ($P=0.039$) difference for magnesium among sampling period (table 10), the multiple-comparison test did not identify any comparison pairs that were considered ($P<0.05$) different.

Water temperature followed an expected seasonal progression (fig. 7*A*). Specific conductance values were relatively lower during the August and February periods when rainfall was greatest, and higher for the October and December periods, when rainfall was least, although the difference was significant only for the December period (fig. 7*B*). Specific conductance in streams commonly is lower during high streamflows through dilution from overland runoff, and higher during low streamflows when baseflow, or groundwater discharge, is a larger component of the overall streamflow. Sodium (fig. 7*E*), potassium (fig. 7*F*), and chloride concentrations (fig. 7*G*) had distributions similar to specific conductance (fig. 7*B*) with highest concentrations during the drier December period.

In well-mixed, open flowing streams, DO concentrations typically are higher at cold temperatures and lower at warm temperatures. This is a result of higher solubility of dissolved gases in water at low temperatures. Although water temperatures (fig. 7*A*) followed expected seasonal patterns among the six sampling periods, there was no apparent relation between water temperature and DO (fig. 7*C*), with the exception of the February period. The streams examined in this study typically are slow moving and enriched with organic matter; low levels of DO are common in these stream settings. The variations in DO concentrations observed among the sampling periods likely reflect the integrated effects of hydrologic differences, such as the influx of oxygenated water from precipitation and overland runoff, and seasonal differences in the consumption of DO by microbial degradation of organic matter. The higher flow conditions for the February and August periods and intermediate flow conditions for the April period indicate more recent stream influxes of precipitation and runoff and, hence oxygenated water, were associated with these periods relative to the June, October, and December periods. The twofold difference in median DO concentrations between the February (8.0 mg/L) and August (3.6 mg/L) periods with the highest flow conditions appears to reflect seasonal differences in the microbial consumption of oxygen for degrading organic matter, which proceeds more quickly under warmer conditions and more slowly under cooler conditions. Although water temperatures were lower for October and December relative to August, the similarly low median DO concentrations for the drier October (2.4 mg/L) and December (2.1 mg/L) periods suggest that a substantial amount of microbial oxygen consumption occurred during the more sluggish streamflow conditions.

Concentrations of nutrients also differed among the sampling periods (table 10; fig. 7). Many biological, chemical, and physical processes can influence the forms and instream concentrations of the N and P constituents,

Table 10. Summary results of the two-way ANOVA tests on the ranked values of the water-quality properties and constituents based on sampling period and land-use type.

[The null hypothesis was that the mean ranks of each distribution were the same. *, indicates significant difference (P < 0.05); <, less than; N, nitrogen; P, phosphorus; δ, delta]

Explanatory grouping variable	p-values for water-quality properties				p-values for major ions					
	Water temperature	Specific conductance	Dissolved oxygen	pH	Calcium	Magnesium	Sodium	Potassium	Chloride	Sulfate
Sampling period	<0.001*	0.001*	<0.001*	0.015*	0.220	0.039*	<0.001*	<0.001*	<0.001*	<0.001*
Land-use type	0.254	<0.001*	0.157	<0.001*	0.084	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*
Sampling period:Land-use type	0.224	0.936	0.751	0.977	0.996	0.980	0.921	0.800	0.367	0.778

Explanatory grouping variable	p-values for nutrients							p-values for isotopes	
	Ammonia + organic N	Ammonia	Total organic N	Nitrate + nitrite	Total N	Orthophosphate	Total P	δ Nitrogen-15 of nitrate + nitrite	δ Oxygen-18 of nitrate + nitrite
Sampling period	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	<0.001*	0.625	0.484
Land-use type	0.007*	<0.001*	0.166	<0.001*	<0.001*	0.533	0.106	<0.001*	0.221
Sampling period:Land-use type	0.322	0.405	0.335	0.906	0.457	0.755	0.726	0.954	0.721

Table 11. Statistical summary of water-quality properties and constituents by sampling period.

[diss., dissolved; mg/L, milligrams per liter; <, less than; µS/cm, microsiemens per centimeter; °C, degrees Celsius, N, nitrogen; P, phosphorus; O, oxygen; ‰, per mil]

Chemical constituent or property (unit)	June 2012				August 2012				October 2012			
	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum
Water-quality properties												
Temperature, water (°C)	54	18.5	21.3	26.2	52	20.6	23.1	27.3	52	12.1	13.9	17.8
Specific conductance (µS/cm at 25 °C)	53	48	121	318	51	49	107	318	51	51	133	440
Oxygen, diss. (mg/L)	54	0.03	1.9	8.1	52	0.04	3.6	6.9	52	0.02	2.4	9.2
pH (standard units)	53	4.9	6.1	7.0	52	4.7	6.1	7.2	52	5.1	6.2	7.0
Major ions												
Calcium, diss. (mg/L)	53	2.01	8.41	43.9	51	1.94	6.29	27.2	51	1.94	7.63	35.6
Magnesium, diss. (mg/L)	53	0.78	3.38	7.85	51	0.76	2.52	6.85	51	0.80	3.42	7.81
Sodium, diss. (mg/L)	53	3.74	5.99	15.1	51	2.17	5.24	16.2	51	3.04	6.79	36.0
Potassium, diss. (mg/L)	53	0.90	4.73	17.4	51	1.49	5.27	24.2	51	2.18	5.72	46.2
Chloride, diss. (mg/L)	53	7.60	15.0	34.8	51	5.06	12.7	35.1	51	7.05	17.6	65.3
Sulfate, diss. (mg/L)	53	0.19	3.91	33.5	51	0.14	5.36	29.3	51	0.14	4.34	43.0
Nutrients												
Ammonia + organic N, total (mg/L as N)	54	0.16	1.0	2.9	52	0.60	1.0	6.3	52	0.22	0.83	7.4
Ammonia, diss. (mg/L as N)	54	0.013	0.140	0.932	52	<0.010	0.060	4.05	52	<0.010	0.044	4.70
Total organic N (mg/L as N)	54	0.12	0.88	2.7	52	0.59	0.96	2.3	52	0.21	0.75	2.7
Nitrate + nitrite, diss. (mg/L as N)	54	<0.040	0.066	5.97	52	<0.040	0.123	4.28	52	<0.040	0.049	6.66
Total N (mg/L as N)	54	0.20	1.3	6.8	52	0.71	1.2	7.4	52	0.34	1.0	14.0
Orthophosphate, diss. (mg/L as P)	54	<0.004	0.039	0.461	52	<0.004	0.042	0.399	52	<0.004	0.029	0.466
Total P (mg/L as P)	54	0.020	0.140	0.981	52	0.013	0.141	0.702	52	0.012	0.101	0.860
Isotopes												
$\delta^{15}N$ of nitrate + nitrite (‰)	24	5.34	13.33	39.21	27	5.12	12.98	48.88	22	6.24	15.42	39.48
$\delta^{18}O$ of nitrate + nitrite (‰)	24	–1.39	7.86	19.89	27	0.67	9.46	22.98	22	2.37	8.66	19.63

Table 11. Statistical summary of water-quality properties and constituents by sampling period.—Continued

[diss., dissolved; mg/L, milligrams per liter; <, less than; µS/cm, microsiemens per centimeter; °C, degrees Celsius, N, nitrogen; P, phosphorus; O, oxygen; ‰, per mil]

Chemical constituent or property (unit)	December 2012				February 2013				April 2013			
	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum
Water-quality properties												
Temperature, water (°C)	54	8.9	12.7	17.1	54	7.2	11.1	14.8	54	11.6	14.3	21.1
Specific conductance (µS/cm at 25 °C)	53	49	141	465	53	56	114	328	53	52	120	271
Oxygen, diss. (mg/L)	54	0.01	2.1	7.4	54	1.9	8.0	10.5	54	0.02	5.0	10.1
pH (standard units)	54	5.1	6.0	7.0	54	4.2	6.0	6.7	54	4.7	6.3	7.0
Major ions												
Calcium, diss. (mg/L)	53	1.92	8.58	37.8	53	2.01	6.37	18.2	53	1.73	6.99	21.4
Magnesium, diss. (mg/L)	53	0.80	3.56	11.3	53	1.00	2.94	7.74	53	0.81	2.90	6.22
Sodium, diss. (mg/L)	53	3.26	7.33	24.2	53	3.73	5.89	16.7	53	3.78	6.75	17.4
Potassium, diss. (mg/L)	53	1.58	6.44	27.2	53	1.54	4.94	24.9	53	0.60	4.75	19.4
Chloride, diss. (mg/L)	53	7.62	20.0	59.1	53	7.89	14.7	37.5	53	8.84	15.4	34.4
Sulfate, diss. (mg/L)	53	0.21	3.53	46.7	53	2.43	10.8	28.6	53	0.31	4.37	15.7
Nutrients												
Ammonia + organic N, total (mg/L as N)	54	0.18	0.81	2.0	54	0.32	0.66	1.5	54	0.52	1.1	4.8
Ammonia, diss. (mg/L as N)	54	<0.010	0.056	0.761	54	<0.010	0.030	0.284	54	<0.010	0.182	3.42
Total organic N (mg/L as N)	54	0.18	0.70	1.4	54	0.30	0.56	1.4	54	0.48	0.85	2.0
Nitrate + nitrite, diss. (mg/L as N)	54	<0.040	<0.040	7.94	54	<0.040	0.993	15.9	54	<0.040	0.153	5.04
Total N (mg/L as N)	54	0.22	0.94	9.1	54	0.36	1.6	17.0	54	0.56	1.3	6.4
Orthophosphate, diss. (mg/L as P)	54	<0.004	0.034	0.713	54	<0.004	0.009	0.052	54	<0.004	0.034	0.347
Total P (mg/L as P)	54	0.011	0.128	1.14	54	0.009	0.044	0.525	54	0.013	0.132	0.859
Isotopes												
$\delta^{15}N$ of nitrate + nitrite (‰)	19	6.09	15.33	38.64	46	6.08	11.33	22.87	32	4.92	13.22	30.65
$\delta^{18}O$ of nitrate + nitrite (‰)	19	5.36	8.60	21.33	46	5.18	9.31	14.01	32	3.46	8.87	16.60

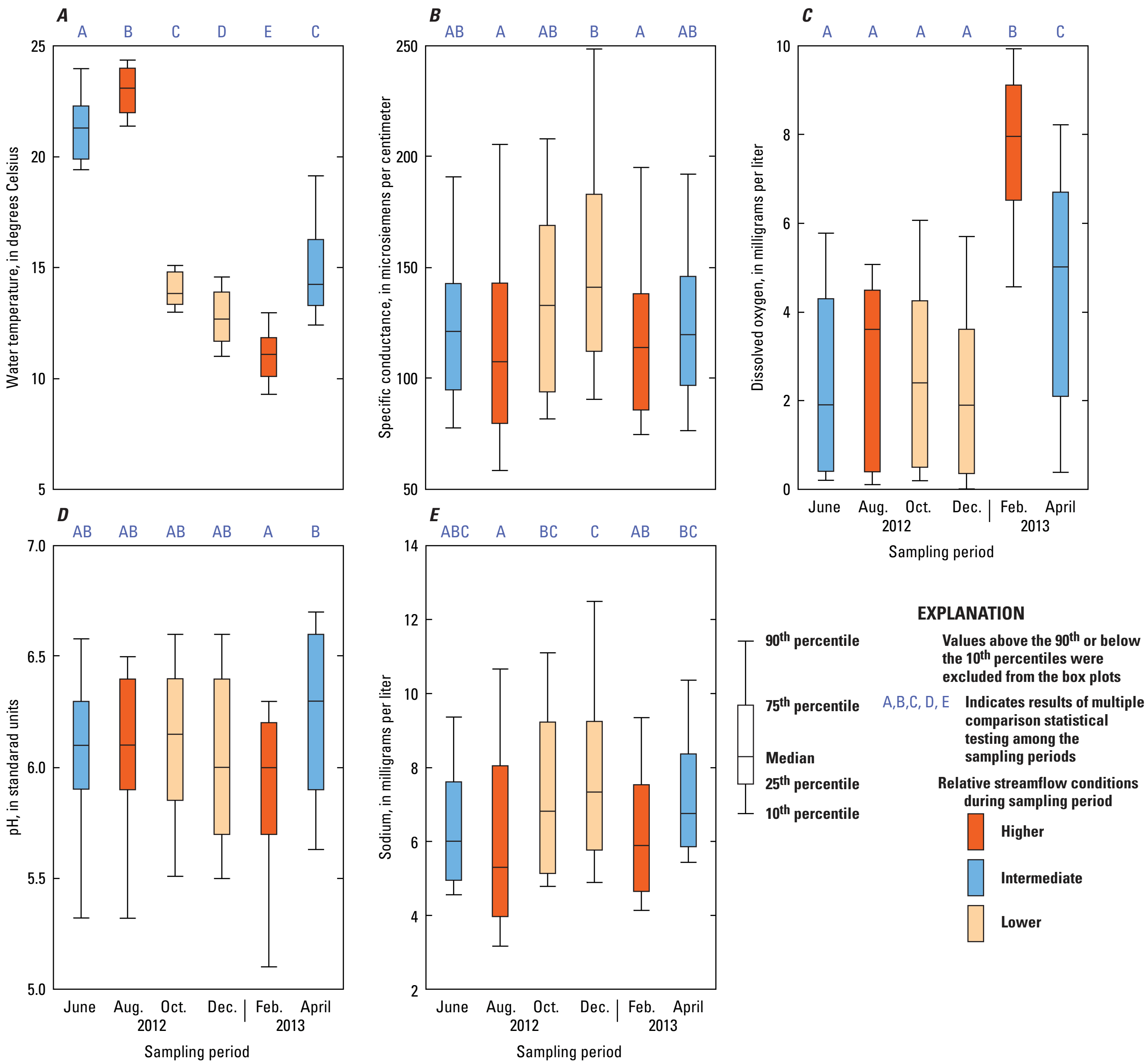

Figure 7. Distributions of (*A*) temperature, (*B*) specific conductance, (*C*) dissolved oxygen, (*D*) pH, (*E*) sodium, (*F*) potassium, (*G*) chloride, (*H*) sulfate, (*I*) ammonia plus organic nitrogen, (*J*) ammonia, (*K*) total organic nitrogen, (*L*) nitrate plus nitrite, (*M*) total nitrogen, (*N*) orthophosphate, and (*O*) total phosphorus for all study sites based on sampling period (for a given constituent, if a sampling period contains the same letter above it as another sampling period, there is no statistical difference between them at the 95 percent confidence level).

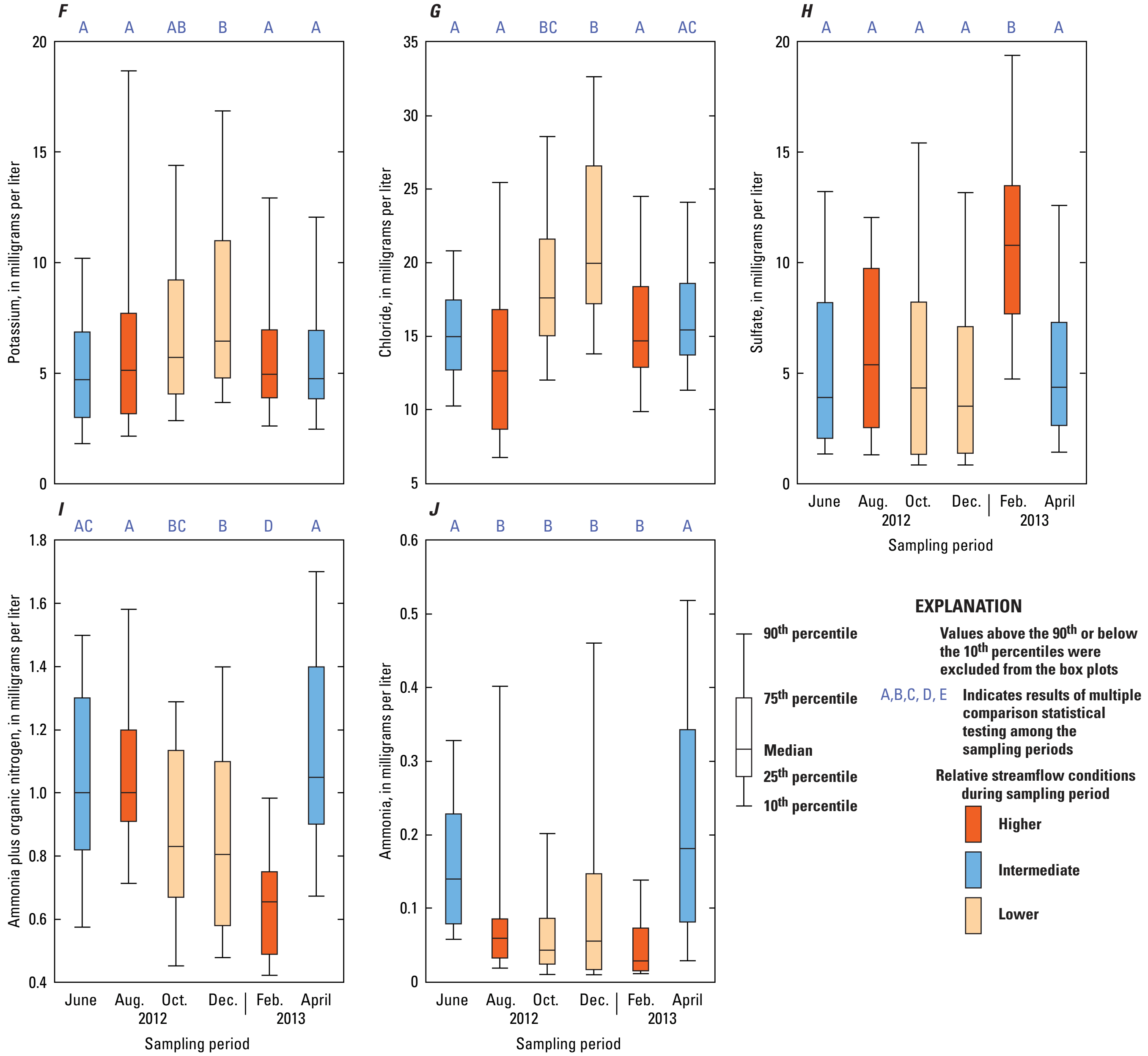

Figure 7. Distributions of (*A*) temperature, (*B*) specific conductance, (*C*) dissolved oxygen, (*D*) pH, (*E*) sodium, (*F*) potassium, (*G*) chloride, (*H*) sulfate, (*I*) ammonia plus organic nitrogen, (*J*) ammonia, (*K*) total organic nitrogen, (*L*) nitrate plus nitrite, (*M*) total nitrogen, (*N*) orthophosphate, and (*O*) total phosphorus for all study sites based on sampling period (for a given constituent, if a sampling period contains the same letter above it as another sampling period, there is no statistical difference between them at the 95 percent confidence level).—Continued

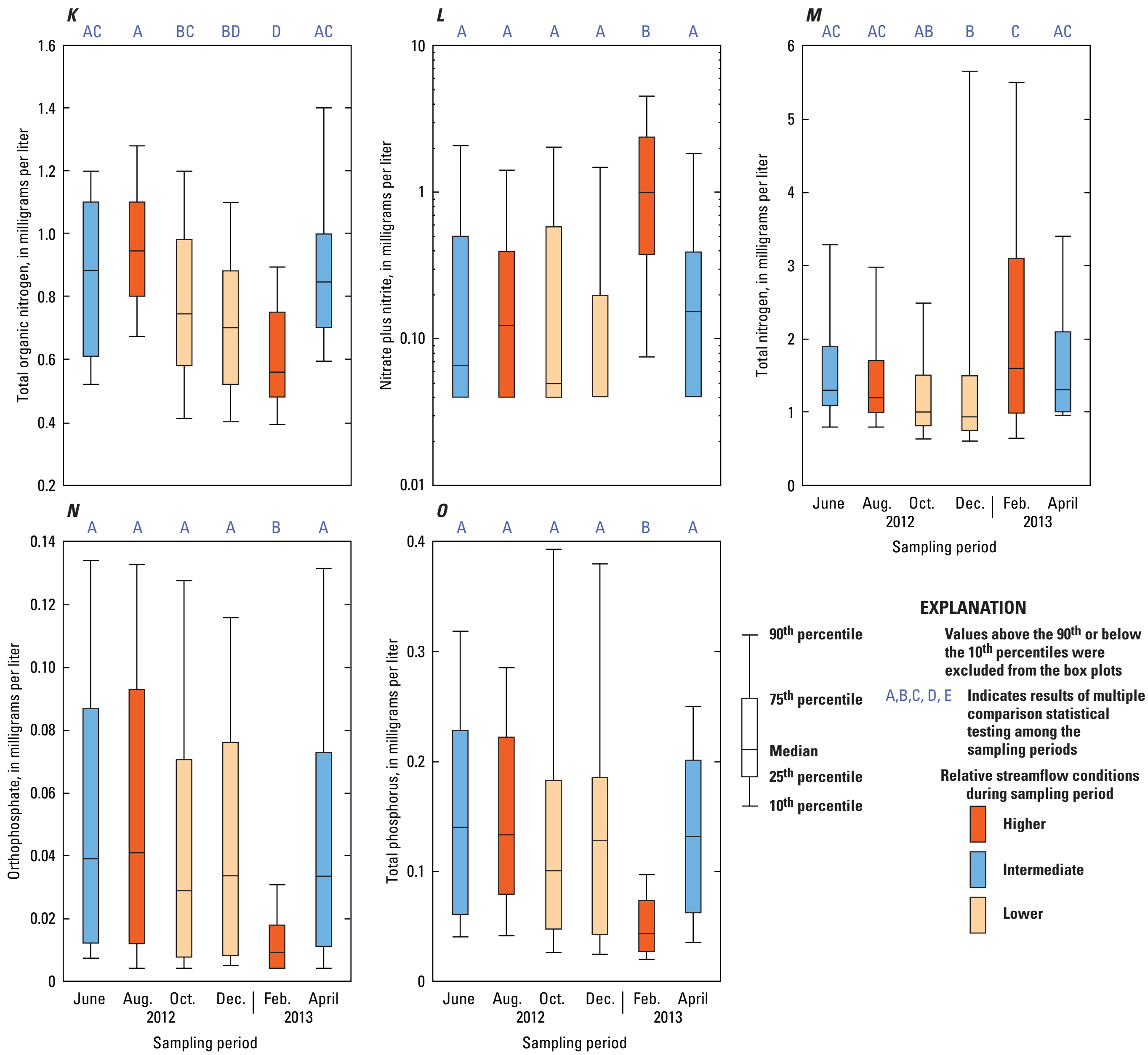

Figure 7. Distributions of (*A*) temperature, (*B*) specific conductance, (*C*) dissolved oxygen, (*D*) pH, (*E*) sodium, (*F*) potassium, (*G*) chloride, (*H*) sulfate, (*I*) ammonia plus organic nitrogen, (*J*) ammonia, (*K*) total organic nitrogen, (*L*) nitrate plus nitrite, (*M*) total nitrogen, (*N*) orthophosphate, and (*O*) total phosphorus for all study sites based on sampling period (for a given constituent, if a sampling period contains the same letter above it as another sampling period, there is no statistical difference between them at the 95 percent confidence level).—Continued

including assimilation and release by algae and aquatic plants; microbially mediated reactions like denitrification; adsorption and desorption processes; and exchange between streambed sediment and the overlying water column (Mulholland, 1992; McMahon and Böhlke, 1996; Mulholland and Hill, 1997; Mainstone and Parr, 2002; Dunne and Reddy, 2005). Interestingly, geochemically reducing conditions present in the buffer and hyporheic zones that help mitigate the amount of nitrate in groundwater discharged to the streams are the same conditions that can promote the mobilization and release of sorbed P from streambed deposits, including sediment derived from upland areas and decaying organic matter, into overlying stream water (Spruill, 2000; Spruill and others, 2005).

The results for nitrate+nitrite (fig. 7*L*) were notably different than the results for ammonia (fig. 7*J*) and organic N (fig. 7*K*). Nitrate+nitrite concentrations were substantially influenced by microbial denitrification, a process that reduces nitrate during anaerobic decomposition of organic matter. The median nitrate+nitrite concentration of 0.993 mg/L observed for February was substantially higher than the median concentrations for the other sampling periods, which ranged from <0.040 to 0.153 mg/L (table 11). The higher nitrate+nitrite concentrations for February coincided with higher streamflows and DO concentrations, and thus appear to reflect more overland contributions of nitrate in water from upstream field-drainage ditches to the streams, as well as less denitrification, for that period. These conditions are most likely to occur in the winter when the water table is high and the nitrate that is contributed to field ditches (from runoff, lateral groundwater inflows, and tile drainage) is likely to bypass the otherwise anoxic zones in near stream areas. Nitrate in the field ditches is rapidly carried to the main stem of the streams during high flows and is subject to less instream processing, including denitrification and uptake by plants and algae, when stream water temperatures are cold (fig. 7*A*) and DO concentrations are elevated (fig. 7*C*), as noted for the February sampling period. The lower nitrate+nitrite concentrations that occurred under the more reduced DO conditions during the June, August, October, and December sampling periods reflect a higher amount of denitrification. The highest median total N concentration of 1.6 mg/L also was observed for February (fig. 7*M*), reflecting the larger contribution from nitrate+nitrite compared to organic N, which constituted the more dominant fraction of total N among the other sampling periods.

Interestingly, sulfate (fig. 7*H*) had a similar distribution among the sampling periods as did both DO (fig. 7*C*) and nitrate+nitrite (fig. 7*L*). Sulfate concentrations were significantly higher during the February period. During the other periods with lower DO concentrations, sulfate apparently was reduced to other forms of sulfur.

In contrast to nitrate+nitrite, the median concentrations of ammonia (0.030 mg/L) and total organic N (0.56 mg/L) were lowest for the February period (fig. 7*J*, *K*; table 11). Similar to the seasonal pattern observed for water temperature (fig. 7*A*), median organic N concentrations were highest during the warm, growing-season months (June, August, and April) and steadily decreased through the fall and winter periods (October, December, and February). Organic N in streams occurs in both the dissolved form, such as urea, amino acids, and humic substances, and the particulate form, such as phytoplankton, zooplankton, microorganisms, and organic detritus. In this study, the dissolved organic N fraction was not measured. Therefore, the extent to which dissolved or particulate substances contributed to the organic nitrogen pool is not known. The observed pattern for total organic N is possibly influenced by algal and aquatic plant production, which likely would be higher during spring and summer and lower during the more dormant winter months.

Interesting differences among sampling periods also were noted for ortho-P (fig. 7*N*) and total P (fig. 7*O*). Overall concentrations for ortho-P (median of 0.009 mg/L) and total P (median 0.044 mg/L) were lowest in the February sampling period, the same period when the highest concentrations of nitrate+nitrite (fig. 7*L*) observed in the streams were attributed to increased overland transport of water through upstream field-drainage ditches. Concentrations of ortho-P and total P during the August period with higher flow conditions were not significantly different from the intermediate- or lower-flow sampling periods. In free-flowing streams with no point-source inputs, higher P concentrations in surface water tend to occur during higher streamflows in association with increased sediment inputs from overland runoff. In contrast, P patterns observed at the swampy, sluggish streams in this study area suggest that instream processes play a dominant role in P cycling. These processes may include adsorption/desorption processes and assimilation by aquatic plants, algae, and microbes in both the bed material and water column (Mainstone and Parr, 2002; Dunne and Reddy, 2005). The higher P concentrations observed during the more reduced DO conditions for the June, August, October, December, and April sampling periods possibly reflect higher amounts of algal biomass and (or) P releases into the water column from microbial degradation of organic matter and (or) desorption from organic substrates or anoxic bed sediments.

In summary, seasonal and hydrologic factors influenced water quality in these Coastal Plain agricultural watersheds. The differences noted among the sampling periods indicate that the interactions between seasonal climatic differences, streamflow conditions, and instream biotic and abiotic processes are complex and their integrated effects can have varying degrees of influence on individual nutrients. These findings are important to consider when developing studies to assess stream nutrient conditions in similar Coastal Plain settings and can inform the choice of specific objectives, nutrients to be examined, and overall timeline and frequency of sampling needed to capture seasonal and (or) hydrologic variability in the data.

Water-Quality Differences Related to Watershed Land-Use Type

Many of the water-quality properties and constituents were significantly influenced (ANOVA $P<0.05$) by watershed land-use type (table 10) on the basis of the results for all six sampling periods. Water-quality differences among the three land-use types, or groups (18 BK sites, 18 SW sites, and 18 SP sites), were examined to better understand potential CAFO influences. Statistical summaries, by land-use group, of the original (non-ranked transformed) water-quality data are provided in tabular (table 12) and graphical formats (fig. 8) to aid the discussion. Figure 8 includes box plots for properties and constituents with significant differences (ANOVA $P<0.05$) among land-use groups; results of the multiple-comparison tests among the groups are denoted along the top of the plots. No significant differences in water temperature, DO, calcium, total organic N, ortho-P, total P, and $\delta^{18}O$ of nitrate+nitrite were noted among the land-use types.

Significant differences were noted in specific conductance, pH, and all of the major ions, except calcium, among the land-use groups (table 10). Specific conductance, pH, magnesium, sodium, potassium, and chloride were significantly different between the BK and SW sites and the BK and SP sites, but not between the SW and SP sites (fig. 8*A–F*). Median specific conductance values for the SW and SP sites were higher than the BK sites, which reflects the higher median concentrations of dissolved magnesium, sodium, potassium, and chloride also noted at the SW and SP sites. Median pH values also were higher for the SW and SP sites relative to the BK sites. Sulfate (fig. 8*G*) for the SP sites was significantly different than both the BK and SW sites.

Median concentrations of ammonia+organic N, ammonia, and total N were higher at the SW and SP sites than at the BK sites (fig. 8*H*, *I*, and *K*; table 12). No significant difference in total organic N was noted among the land-use groups, suggesting that the differences in ammonia+organic N between the BK and SW sites and the BK and SP sites are associated with the ammonia fraction. Nitrate+nitrite was the only constituent found to be significantly different between all three land-use groups (fig. 8*J*). Median nitrate+nitrite concentrations progressively increase from the BK to the SW to the SP sites. Interestingly, no significant differences were identified for the P nutrients (ortho-P or total P) on the basis of land-use type (table 10).

Similar to the N constituents, median $\delta^{15}N$ values of nitrate+nitrite for the SW and SP sites were higher, or more positive, than the BK sites (fig. 8*L*), indicating that nitrate+nitrite at the SW and SP sites was more enriched in 15N. The higher median $\delta^{15}N$ values of nitrate+nitrite likely indicate that N inputs to streams at the SW and SP sites were more influenced by animal-manure sources; however, it is important to note that other processes, such as denitrification and assimilation by algae, also may have influenced the observed $\delta^{15}N$ values of nitrate+nitrite.

These results indicate that waste-manure storage and (or) field applications at the CAFOs have increased surface-water concentrations of selected constituents at the SW and SP sites above those noted for the BK sites, which do not contain any active CAFOs. Although the various types and amounts of commercial fertilizer products used in the watersheds of the individual study sites are unknown, it is considered unlikely that the significant differences noted in the water-quality constituents would only occur between the BK group of sites and both CAFO site groups (SW and SP) and not between the SW and SP site groups if related solely to differences in commercial fertilizer use. Most of the statistically significant differences for major ions (magnesium, sodium, potassium, and chloride) and nutrients (ammonia+organic N, ammonia, nitrate+nitrite, and total N) occurred between the BK and SW sites and the BK and SP sites (fig. 8). The median concentrations of these constituents were all higher at the SW and SP sites relative to the BK sites.

It is unclear whether the lack of detectable differences in P among the land-use groups indicates that stream inputs of P were the same among the study watersheds with and without animal-waste manure applications or whether other environmental processes (like sediment deposition, adsorption/desorption, and assimilation) have obscured differences in source inputs of P derived from commercial fertilizer and (or) animal-waste manure.

Phosphorus, which is relatively immobile in soil, typically is transported to streams in particulate form during overland runoff. The more soluble N constituents, such as ammonia and nitrate+nitrite, are prone to leaching in soils and may be transported to streams through both groundwater discharge and overland runoff. The disparity between N and P response among the sites may reflect differences in transport pathways or instream processing that influenced instream concentrations of these two classes of nutrients.

Table 12. Statistical summary of water-quality properties and constituents by land-use type.

[diss., dissolved; mg/L, milligrams per liter; %, percent; µS/cm, microsiemens per centimeter; °C, degrees Celsius, N, nitrogen; P, phosphorus; O, oxygen; δ, delta; <, less than; ‰, per mil]

Chemical property or constituent (unit)	Background (BK) sites				Swine (SW) sites				Swine and poultry (SP) sites			
	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum	Number of samples	Minimum	Median	Maximum
Water-quality properties												
Temperature, water (°C)	106	7.2	14.7	27.3	108	8.0	14.2	26.2	106	8.0	14.6	24.4
Specific conductance (µS/cm at 25 °C)	106	49	98	264	102	48	132	328	106	50	138	440
Oxygen, diss. (mg/L)	106	0.01	3.2	10.4	108	0.01	3.4	10.1	106	0.01	4.3	10.5
pH (standard units)	105	4.2	6.0	6.8	108	4.7	6.2	6.9	106	4.3	6.2	7.2
Major ions												
Calcium, diss. (mg/L)	106	1.73	6.92	15.9	102	1.94	8.52	19.7	106	2.34	7.16	43.9
Magnesium, diss. (mg/L)	106	1.45	2.64	4.61	102	0.76	3.34	7.74	106	0.92	3.76	11.3
Sodium, diss. (mg/L)	106	2.17	5.41	24.2	102	3.67	6.90	16.0	106	3.15	6.80	36.0
Potassium, diss. (mg/L)	106	0.60	3.90	15.6	102	0.90	6.84	24.9	106	1.41	6.58	46.2
Chloride, diss. (mg/L)	106	5.06	14.0	53.2	102	7.84	17.3	37.7	106	6.01	17.1	65.3
Sulfate, diss. (mg/L)	106	0.14	3.84	46.7	102	0.14	5.14	28.6	106	0.64	6.92	28.4
Nutrients												
Ammonia + organic N, total (mg/L as N)	106	0.36	0.83	2.3	108	0.32	0.94	4.8	106	0.16	0.96	7.4
Ammonia, diss. (mg/L as N)	106	<0.010	0.048	0.932	108	<0.010	0.102	3.42	106	<0.010	0.072	4.7
Total organic N (mg/L as N)	106	0.23	0.76	1.7	108	0.27	0.82	2.0	106	0.12	0.80	2.7
Nitrate + nitrite, diss. (mg/L as N)	106	<0.040	0.048	1.51	108	<0.04	0.173	15.9	106	<0.040	0.352	10.8
Total N (mg/L as N)	106	0.42	1.0	2.3	108	0.36	1.5	17.0	106	0.20	1.3	14.0
Ortho-phosphate, diss. (mg/L as P)	106	<0.004	0.026	0.713	108	<0.004	0.030	0.534	106	<0.004	0.026	0.466
Total P (mg/L as P)	106	0.015	0.098	1.14	108	0.009	0.122	0.981	106	0.012	0.100	0.860
Isotopes												
$\delta^{15}N$ of nitrate + nitrite (‰)	40	4.92	9.39	16.99	61	5.66	13.57	48.88	69	6.52	15.33	39.97
$\delta^{18}O$ of nitrate + nitrite (‰)	40	5.18	9.43	16.27	61	–1.39	8.48	22.98	69	0.29	9.04	21.33

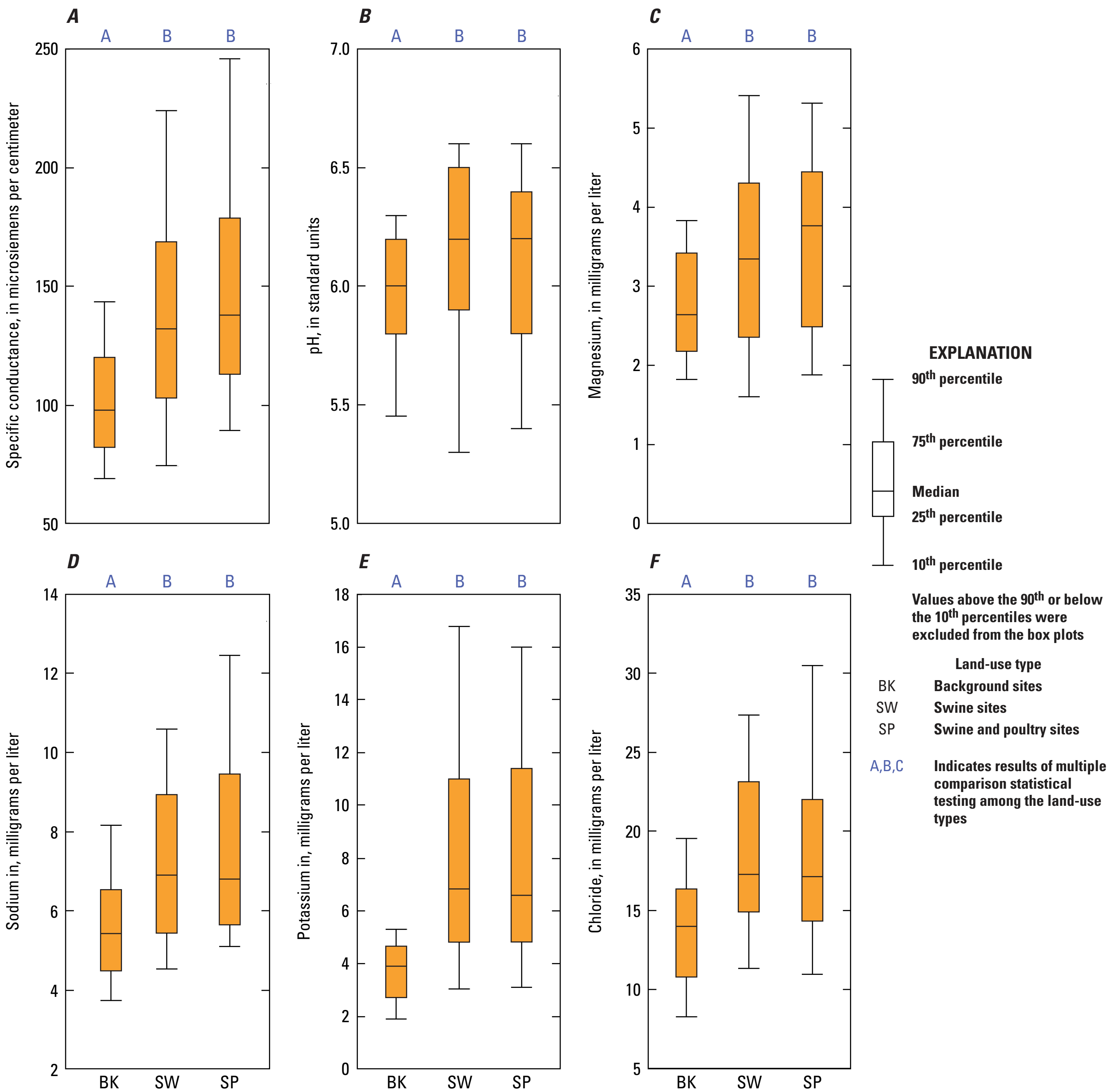

Figure 8. Distributions of (*A*) specific conductance, (*B*) pH, (*C*) magnesium, (*D*) sodium, (*E*) potassium, (*F*) chloride, (*G*) sulfate, (*H*) ammonia plus organic nitrogen, (*I*) ammonia, (*J*) nitrate plus nitrite, (*K*) total nitrogen, and (*L*) delta nitrogen-15 of nitrate plus nitrite for all sampling periods based on watershed land-use type (for a given constituent, if a land-use type contains the same letter above it as another land-use type, there is no statistical difference between them at the 95 percent confidence level).

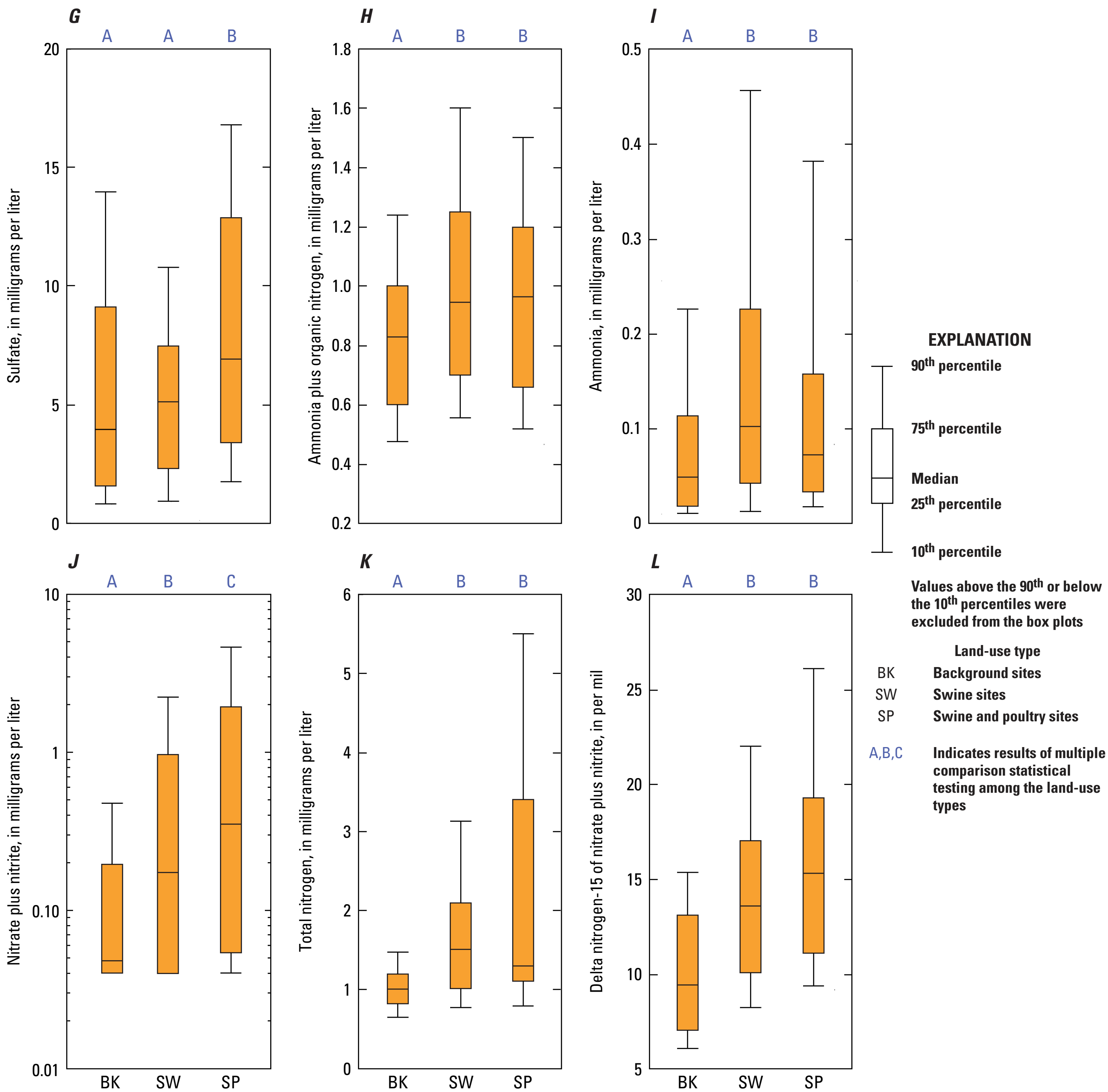

Figure 8. Distributions of (*A*) specific conductance, (*B*) pH, (*C*) magnesium, (*D*) sodium, (*E*) potassium, (*F*) chloride, (*G*) sulfate, (*H*) ammonia plus organic nitrogen, (*I*) ammonia, (*J*) nitrate plus nitrite, (*K*) total nitrogen, and (*L*) delta nitrogen-15 of nitrate plus nitrite for all sampling periods based on watershed land-use type (for a given constituent, if a land-use type contains the same letter above it as another land-use type, there is no statistical difference between them at the 95 percent confidence level)—Continued

Multi-Analyte Approach for Differentiating Sites With Water-Quality Effects From CAFOs

The statistical evaluations discussed previously indicated that when all 54 primary study sites were examined collectively on the basis of their land-use type (BK, SW, and SP), several water-quality differences related to animal-waste manures were identified for the SW and SP site groups. Interestingly, some individual SW and SP sites did not appear to be affected by animal-waste manures. Data were further evaluated to better understand distinctions among selected water-quality constituents at sites with and without CAFOs to aid identification of those SW and SP watersheds with measurable CAFO manure effects on water quality.

Insights Based on Multi-Site Reconnaissance Sampling Within Selected Watersheds During April 2013

During April 2013, samples were collected once at 23 secondary sites within 9 of the primary watersheds to obtain water-quality data from upstream reaches. These secondary sites were located in proximity to either swine CAFOs and spray fields or to background agricultural fields. Nutrient and ion concentrations and the nitrate+nitrite stable isotope data were evaluated to distinguish sites where CAFO waste manures did or did not have a measurable effect on surface-water quality.

Stable isotopes ($\delta^{15}N$ and $\delta^{18}O$) of nitrate are often used in water-quality studies as environmental tracers for investigating anthropogenic sources of nitrogen (such as atmospheric deposition, commercial inorganic fertilizers, and organic animal manures and septic wastes). Kendall and others (2007) diagrammed common ranges, or fields, of nitrate $\delta^{15}N$ and $\delta^{18}O$ values derived or nitrified from various N sources (fig. 9). The $\delta^{18}O$ values tend to be more useful for separating nitrate derived from atmospheric deposition or synthetic nitrate fertilizers from other sources. The $\delta^{15}N$ values tend to be more useful for distinguishing nitrate derived from microbial nitrification of ammonium and (or) organic N in fertilizer, precipitation, soil, and animal manure or human septic waste because these sources have overlapping $\delta^{18}O$ values, commonly between –10 and +15 ‰ (Kendall and others, 2007; Xue and others, 2009).

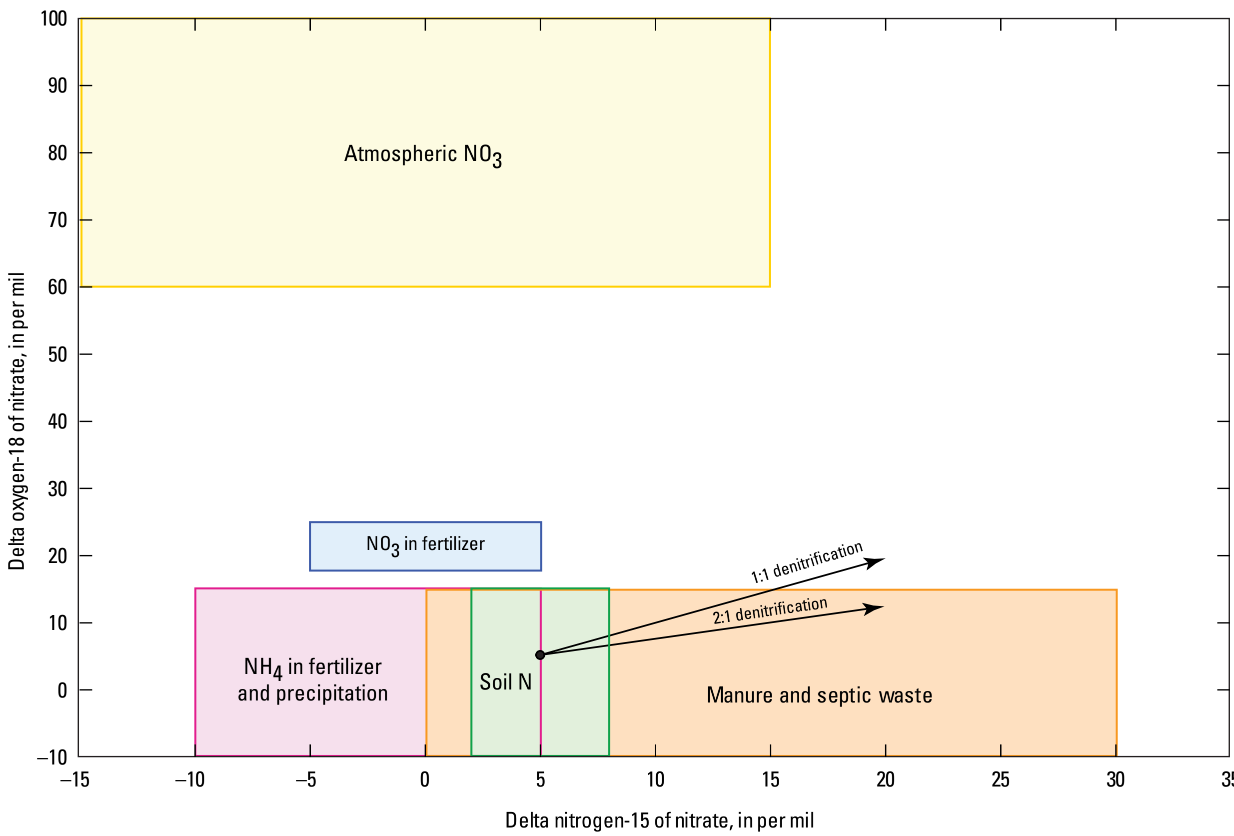

Figure 9. Common ranges in values of delta nitrogen-15 and delta oxygen-18 of nitrate derived from various nitrogen sources (modified from Kendall and others, 2007).

Inorganic fertilizers and animal-waste manures, which are the main sources of N in the agricultural watersheds in this study, generally have distinct $\delta^{15}N$ nitrate values (Kendall, 1998). The $\delta^{15}N$ values of nitrate originating from inorganic fertilizers typically are lower, about –5 to +5 ‰, than those from animal manures, which typically are higher and have a wider range of compositions, about 0 to +30 ‰ (Fogg and others, 1998; Kendall and others, 2007; Xue and others, 2009). Note that nitrate derived from human septic wastes generally has $\delta^{15}N$ values of about +5 to +20 ‰ that are indistinguishable from animal manures (Fogg and others, 1998; Xue and others, 2009); however, human-derived wastes are not considered to be a substantial contributor of N to streams in the study watersheds. Although the $\delta^{15}N$ values of soil nitrate derived from inorganic fertilizers tend to overlap those derived from the mineralization of natural soil organic N, about 0 to +8 ‰, they are often distinguishable from the higher nitrate $\delta^{15}N$ values associated with animal-waste manures (Fogg and others, 1998; Kendall and others, 2007; Xue and others, 2009).

Comparing measured nitrate $\delta^{15}N$ and $\delta^{18}O$ values in samples against the general source boxes depicted in figure 9 may be useful for assessing potential sources if the original source signal of the nitrate has not been substantially altered. Complications arise if the isotopic composition reflects a mixture of two or more nitrate sources and (or) has been influenced by biogeochemical processes, such as assimilation or denitrification, that transform N, which can cause the altered $\delta^{15}N$ and $\delta^{18}O$ values to resemble those of other sources (Kendall and others, 2007). During the process of denitrification, microbes preferentially use the lighter ^{14}N and ^{16}O isotopes, which enrich the remaining or residual nitrate pool with the heavier ^{15}N and ^{18}O isotopes, resulting in more positive nitrate $\delta^{15}N$ and $\delta^{18}O$ values. Denitrification causes coupled increases in the $\delta^{15}N$ and $\delta^{18}O$ values of the residual nitrate by an approximate 1:1 to 2:1 ratio (Böttcher and others, 1990; Kendall and others, 2007).

The effects of denitrification are illustrated using an example of assumed nitrate having an initial $\delta^{15}N$ value of 5 ‰ and $\delta^{18}O$ value of 5 ‰ similar to that derived from ammonium fertilizer or soil organic N (fig. 9). The two arrows indicate how the process of denitrification for nitrate with this initial isotopic signature produces residual nitrate $\delta^{15}N$ to $\delta^{18}O$ values that progressively increase along either a 1:1 denitrification line (having a slope of 1) or 2:1 denitrification line (having a slope of 0.5). As the $\delta^{15}N$ and $\delta^{18}O$ values of the initial nitrate reflecting an ammonium fertilizer or soil organic N source become increasingly more positive during denitrification, they become more similar to those expected for nitrate derived from animal-waste manures, thereby confounding interpretations of the nitrate sources.

These types of issues can make it complicated or impractical to identify nitrate sources solely on the basis of the nitrate isotopic compositions. It is beneficial to examine other chemical constituents in combination with the nitrate stable isotope data for differentiating sources of nitrate contamination in water (Spruill and others, 2002; Kendall and others, 2007; Xue and others, 2009). In the North Carolina Coastal Plain, Karr and others (2001) and Spruill and others (2002) used $\delta^{15}N$ data in combination with major ion data to examine sources of nitrate in groundwater. Karr and others (2001) used $\delta^{15}N$, potassium, and chloride data to examine swine-manure contamination in groundwater from a waste lagoon and spray field. Spruill and others (2002) evaluated the results of nitrate $\delta^{15}N$, nutrients (nitrate and ammonia) and major ions (calcium, magnesium, sodium, and potassium) with classification tree models to identify sources of groundwater nitrate derived from inorganic fertilizers, swine manure, poultry litter, and septic-system wastes. Ratios of selected ion concentrations (calcium to magnesium and sodium to potassium) and summed concentrations of sodium+potassium were found to be useful indicators for distinguishing the different nitrate sources.

The examination of the April 2013 water-quality data for the primary and secondary study sites primarily focused on evaluating nitrate+nitrite and sodium+potassium concentrations in combination with the nitrate+nitrite $\delta^{15}N$ values for differentiating those sites with measurable effects of CAFO manure on water quality (table 13). Comments on whether the surface-water samples that were collected had the potential to be influenced by one or more CAFOs upstream from the sites are noted in table 13. Detailed evaluations of the data for each group of associated sites are provided separately as appendix A5. Insights based on the evaluations of the April 2013 dataset (appendix A5) are discussed below.

In six of the nine watersheds that were examined, measured effects of swine CAFO manure on surface water at one or more upstream secondary sites also were noted further downstream at the primary site locations (table 13). The extent to which influences of CAFO manure may be identified in surface water at downstream watershed locations likely varies depending on the particular watershed setting, including such things as basin size, density of CAFOs and their locations, the presence or absence of tile drains and field ditches, stream morphology, and streamflow conditions. Many of the secondary sites that were located next to or downstream from swine CAFOs were found to be influenced by swine manure in terms of nitrate+nitrite and sodium+potassium concentrations and nitrate+nitrite $\delta^{15}N$ values. Conversely, no water-quality effect was noted at some of the sites (table 13), which suggests that all CAFOs do not necessarily have a measurable effect on these water-quality constituents in adjacent sections of streams.

The combined use of the nitrate+nitrite, sodium+potassium, and $\delta^{15}N$ of nitrate+nitrite data proved valuable for identifying those 9 primary and 23 secondary sites either having or not having a measurable water-quality effect associated with CAFO waste manures (appendix A5). Of the 32 sites, 18 had measurable manure influence, 11 had no measurable manure influence (including the 4 background agricultural sites), and 3 had unclear results (table 13). Distinctions among the results are illustrated in figure 10 for the sites with, without, or unclear CAFO manure influences. Boundaries delineating the general distribution in the

Table 13. Water-quality results for the April 2013 sample period used to examine waste-manure influences at the primary and secondary study sites.

[CAFO, concentrated animal feeding operation; mg/L, milligram per liter; $\delta^{15}N$, delta nitrogen-15; ‰, per mil; <, less than; na, not analyzed]

Study site (site maps in appendix A1)	Potential to be influenced by CAFOs	Dissolved oxygen (mg/L)	Nitrate + nitrite (mg/L)	Sodium + potassium (mg/L)	$\delta^{15}N$ of nitrate + nitrite (‰)	$\delta^{18}O$ of nitrate + nitrite (‰)	Are the results interpreted to reflect CAFO waste manure influences at the site? (see appendix A5)
SW-04A	Yes, near upgradient edge of swine spray field	6.3	0.307	7.96	15.80	11.09	Unclear
SW-04B	Yes, 1 swine CAFO	7.4	3.31	16.10	19.37	10.34	Yes
SW-04	Yes, 1 swine CAFO	3.4	1.09	16.66	22.16	10.62	Yes
SW-05A	Yes, 1 swine CAFO	0.08	0.052	10.01	na	na	No
SW-05B	No, background agricultural fields	4.2	1.70	7.28	9.66	8.43	No
SW-05C	Yes, 1 swine CAFO	5.4	3.40	19.16	21.68	10.78	Yes
SW-05	Yes, 4 swine CAFOs	2.9	0.795	12.42	17.05	8.87	Yes
SW-08A	Yes, 5 active and 1 inactive swine CAFOs	0.1	<0.040	16.41	na	na	Unclear
SW-08B	Yes, 1 swine CAFO	0.8	0.681	12.67	7.42	7.89	No
SW-08C	Yes, 3 swine CAFOs	4.0	1.22	16.40	24.56	10.05	Yes
SW-08D	No, background agricultural fields	6.3	2.74	9.95	5.44	6.27	No
SW-08	Yes, 12 active and 2 inactive swine CAFOs	0.02	<0.040	16.70	na	na	Unclear
SW-13A	Yes, 1 swine CAFO	5.9	35.4	65.70	18.92	9.95	Yes
SW-13B	Yes, 2 swine CAFOs	7.0	27.5	51.80	19.98	10.42	Yes
SW-13	Yes, 3 swine CAFOs	3.0	0.390	33.10	22.04	9.16	Yes
SP-01A	No, background agricultural fields	9.3	<0.040	5.19	na	na	No
SP-01B	Yes, 1 swine and 1 poultry CAFOs	10.6	<0.040	5.93	na	na	No
SP-01C	Yes, 2 swine CAFOs	11.8	0.592	31.10	27.99	9.74	Yes
SP-01	Yes, 6 swine and 1 poultry CAFOs	10.1	0.103	10.63	8.94	4.96	No
SP-04A	No, background agricultural fields	2.3	0.877	9.25	12.52	10.79	No
SP-04B	Yes, 2 swine and 1 poultry CAFOs	4.2	1.86	22.74	22.54	10.58	Yes
SP-04	Yes, 4 swine and 1 poultry CAFOs	2.1	0.110	21.24	17.01	9.58	Yes
SP-05A	Yes, 1 swine CAFO	7.1	3.50	12.06	7.93	5.20	No
SP-05B	Yes, 1 swine and 1 poultry CAFOs	9.2	2.62	12.16	8.75	6.91	No
SP-05	Yes, 1 swine and 3 poultry CAFOs	5.9	4.13	11.84	8.00	6.75	No
SP-09A	Yes, 3 swine and 1 poultry CAFOs	5.9	3.20	43.60	23.02	14.21	Yes
SP-09	Yes, 3 swine and 1 poultry CAFOs	5.4	1.94	33.70	23.13	14.72	Yes
SP-11A	Yes, 2 swine CAFOs	3.7	1.11	32.60	25.57	13.32	Yes
SP-11B	Yes, 4 swine CAFOs	1.4	1.73	32.50	28.96	9.67	Yes
SP-11C	Yes, 1 swine CAFO	9.5	2.98	12.66	11.91	8.63	Yes
SP-11D	Yes, 6 swine CAFOs	4.8	1.01	31.10	24.21	6.69	Yes
SP-11	Yes, 9 swine and 1 poultry CAFOs	0.3	<0.040	22.80	na	na	Yes

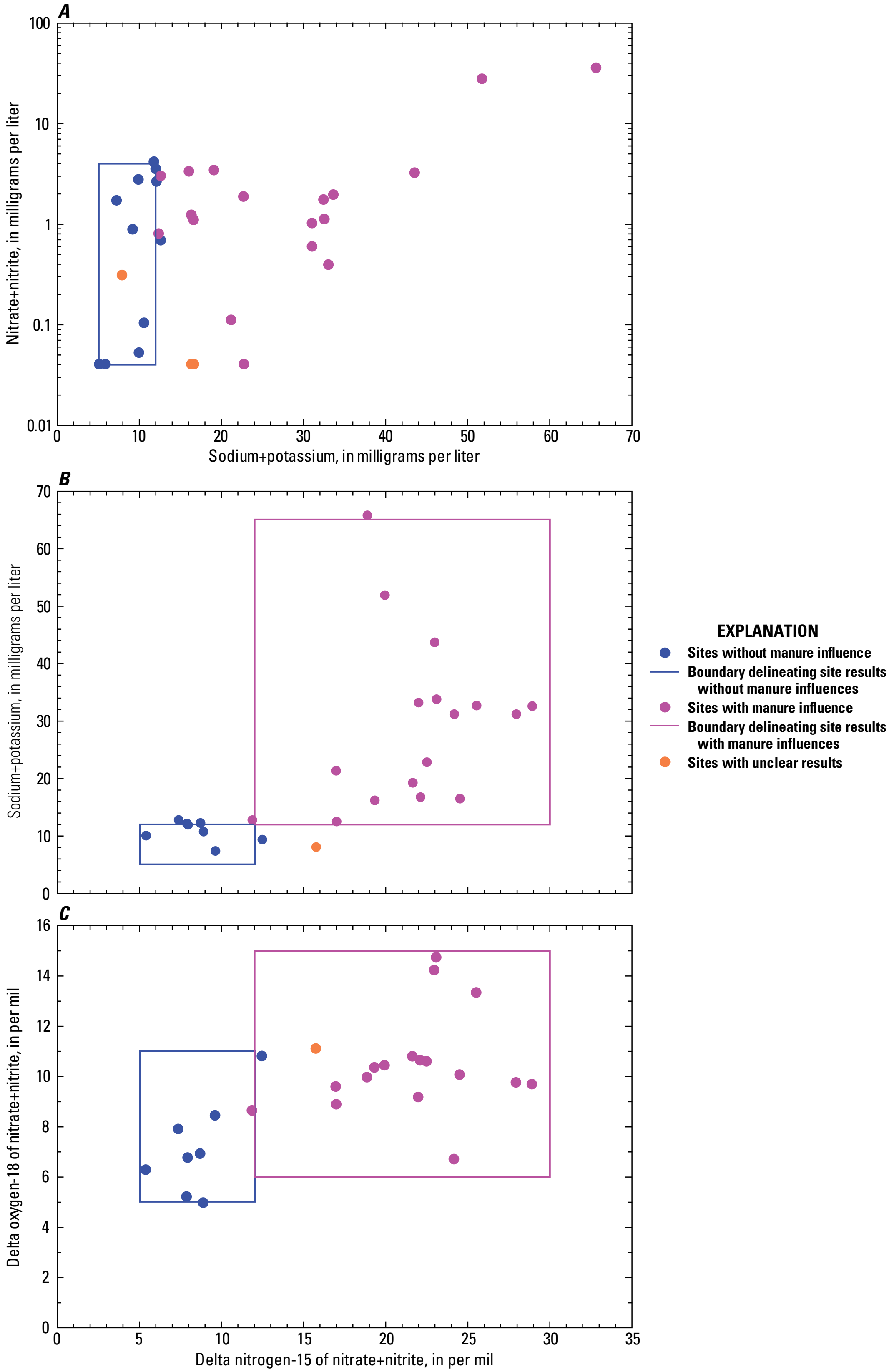

Figure 10. Graphs showing data comparisons of (*A*) sodium plus potassium to nitrate plus nitrite, (*B*) delta nitrogen-15 of nitrate plus nitrite to sodium plus potassium, and (*C*) delta nitrogen-15 to delta oxygen-18 of nitrate plus nitrite for sites with and without CAFO manure influences and sites with unclear results based on the April 2013 dataset.

sodium+potassium and nitrate+nitrite data for the sites without manure influences are shown in figure 10*A*. Boundaries delineating the general distributions in the nitrate+nitrite $\delta^{15}N$ and sodium+potassium data (fig. 10*B*) and the nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ data (fig. 10*C*) are shown for both the sites without and with manure influences. The nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ values for the sites without manure effects (fig. 10*C*) agree with the common $\delta^{15}N$ and $\delta^{18}O$ values of nitrate derived from ammonium fertilizer or natural soil organic N displayed in figure 9. The nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ values for the sites with manure effects (fig. 10*C*) also agree with the $\delta^{15}N$ and $\delta^{18}O$ values of nitrate commonly derived from animal manure sources (fig. 9).

The overall range of nitrate+nitrite concentrations was fairly similar for the sites with and without manure influences; however, sodium+potassium concentrations were higher for the sites with a manure influence than those without an influence (fig. 10*A*). Better separation among the sites is noted in the nitrate+nitrite $\delta^{15}N$ and sodium+potassium data (fig. 10*B*). The sites without manure influences had lower $\delta^{15}N$ values (about 5 to 12 ‰) and sodium+potassium concentrations (about 5 to 12 mg/L) than the manure influenced sites, which are characterized by higher $\delta^{15}N$ values (about 12 to 30 ‰) and sodium+potassium concentrations (about 12 to 65 mg/L). Comparison of the nitrate+nitrite $\delta^{15}N$ to $\delta^{18}O$ data (fig. 10*C*) indicates that although the $\delta^{15}N$ values appear to segregate, the sites without and with manure influences tend to have overlapping $\delta^{18}O$ values of about 5 to 11 ‰ and 6 to 15 ‰, respectively. For several sites, limited or inconsistent results made it difficult to determine whether water quality reflected background agricultural conditions or waste-manure effects. For example, the unclear results shown for some sites included a sodium+potassium concentration within the range of sites without manure influences (fig. 10*A*, *B*) but the elevated $\delta^{15}N$ value (fig. 10*B*, *C*) could be indicative of either a manure signature or denitrification effects on soil nitrate derived from inorganic fertilizer or natural organic N.

Identification of Study Watersheds Having Measurable CAFO Effects on Water Quality

On the basis of the insights gained from the above evaluation of the April 2013 dataset, nitrate+nitrite and sodium+potassium concentrations and the nitrate+nitrite isotopic values ($\delta^{15}N$ and $\delta^{18}O$) for all 6 sampling periods at the 54 primary study sites (appendix A6) were evaluated to determine which of the 18 SW and 18 SP sites had apparent CAFO waste-manure effects on stream water quality. Results for the 18 BK study sites first were plotted to serve as a baseline, or background, dataset (fig. 11) against which the SW and SP site data could be compared. The reference boundaries determined for sites without and sites with measurable manure influences using the April 2013 dataset (fig. 10) also were included in figure 11 to aid examination of the results.

Overall, the baseline results for the BK sites fall within fairly well-defined clusters (fig. 11). Most of the nitrate+nitrite and sodium+potassium concentrations for the BK sites fall within the reference boundary for sites without waste-manure effects. Note that many of the BK sites had nitrate+nitrite concentrations less than the RL of 0.04 mg/L. As previously discussed, denitrification is one of the important factors known to influence nitrate+nitrite concentrations at the study sites. The effects of denitrification are evident in the background nitrate+nitrite $\delta^{15}N$ results. The BK sites had nitrate+nitrite $\delta^{15}N$ values, up to about 17 ‰, that extended beyond the upper limit of about 12 ‰ for the reference boundary for sites without manure influences (fig. 11*B*). The nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ values for the BK sites plot along a best-fit regression line having a slope of 0.48 (fig. 11*C*), which is indicative of denitrification that causes coupled increases in the $\delta^{15}N$ to $\delta^{18}O$ values by a 2:1 ratio. Increased isotopic values resulting from denitrification explains why some of the BK sites, with no waste-manure influences, had nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ values within the reference boundary reflecting manure influence.

Data for each of the SW and SP sites were plotted and compared against the figure 11 boundaries respresenting the BK site baseline data, as well as the sites without and with measurable manure influences, to categorize those SW and SP sites with results that (1) were similar to background conditions, or (2) had distinct differences indicating CAFO manure effects. It was impractical to include all of the comparison plots in the report. Therefore, for illustrative purposes, representative plots for selected sites with results similar to background conditions are shown in figure 12, and selected sites with results indicating manure influences are shown in figure 13.

Sites SW-14, SW-16, SP-05, and SP-15 had results similar to background conditions based on comparisons of their sodium+potassium to nitrate+nitrite concentrations (fig. 12*A*), nitrate+nitrite $\delta^{15}N$ values to sodium+potassium concentrations (fig. 12*B*), and nitrate+nitrite $\delta^{15}N$ to $\delta^{18}O$ values (fig. 12*C*). The effects of denitrification can also be seen in the $\delta^{15}N$ results for site SP-15.

The effects of CAFO waste manures are indicated in some or all of the results for sites SW-04, SW-05, SP-12, and SP-16 as compared to the reference boundaries (fig. 13). Sites SW-05 and SP-16 had samples with results overlapping background conditions as well as manure influences. These site results likely reflect different instream mixtures of groundwater and overland runoff from areas with and without CAFOs where at times manure influences on water quality were not always evident. CAFO manure effects were evident in all of the sample results for sites SW-04 and SP-12 (fig. 13). Site SP-12, located immediately downstream from multiple swine CAFO waste-manure lagoons and application fields (appendix fig. A1-48), had high nitrate+nitrite $\delta^{15}N$ and $\delta^{18}O$ values. The isotopic signatures of nitrate+nitrite derived from waste manures at this site possibly reflect the effects of different fractionation processes, such as ammonia volatilization and denitrification, that occurred before, during, and (or) after the applications of waste manures from the storage lagoons to the spray fields.

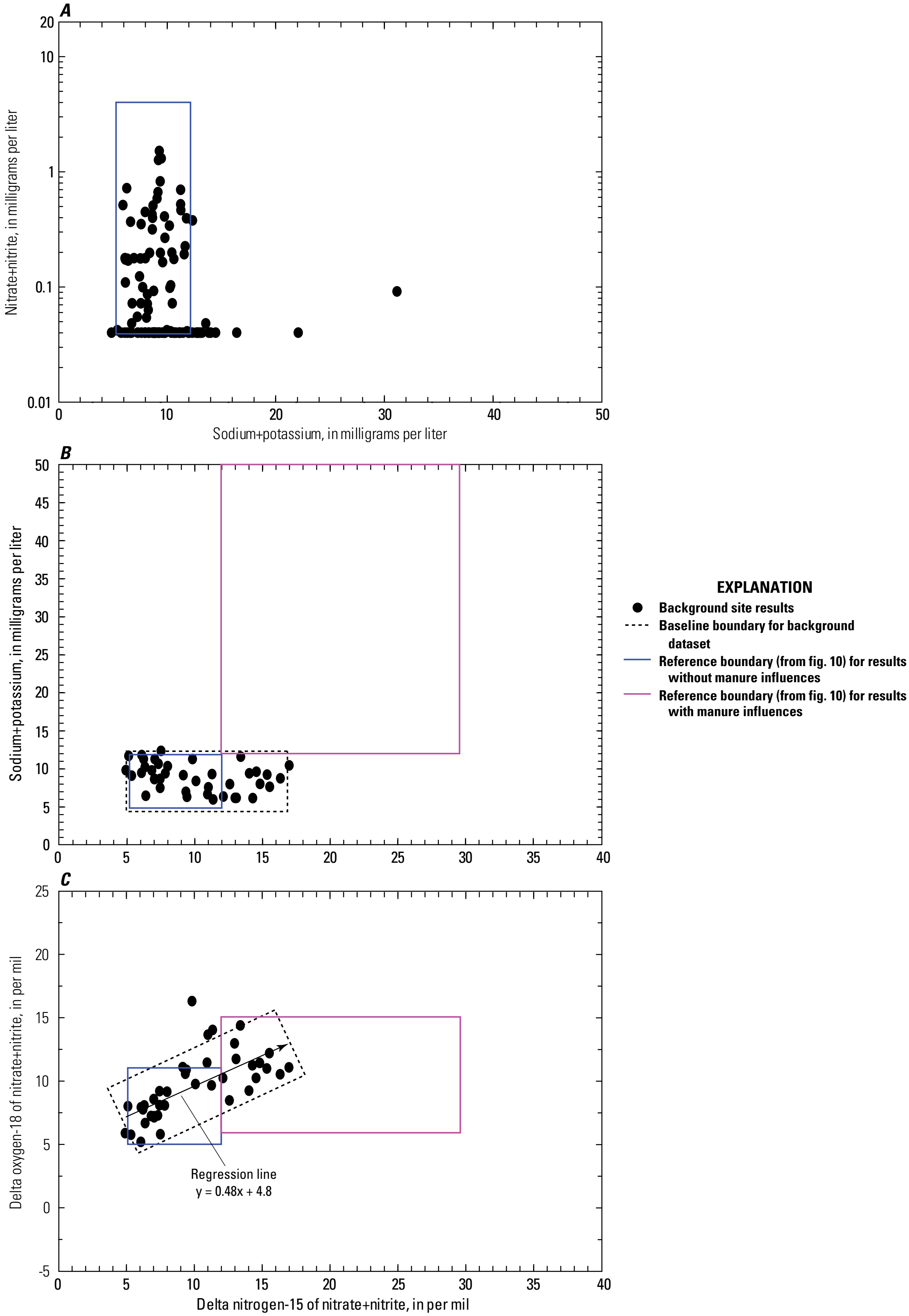

Figure 11. Graphs showing data comparisons of (*A*) sodium plus potassium to nitrate plus nitrite, (*B*) delta nitrogen-15 of nitrate plus nitrite to sodium plus potassium, and (*C*) delta nitrogen-15 to delta oxygen-18 of nitrate plus nitrite for the background sites.

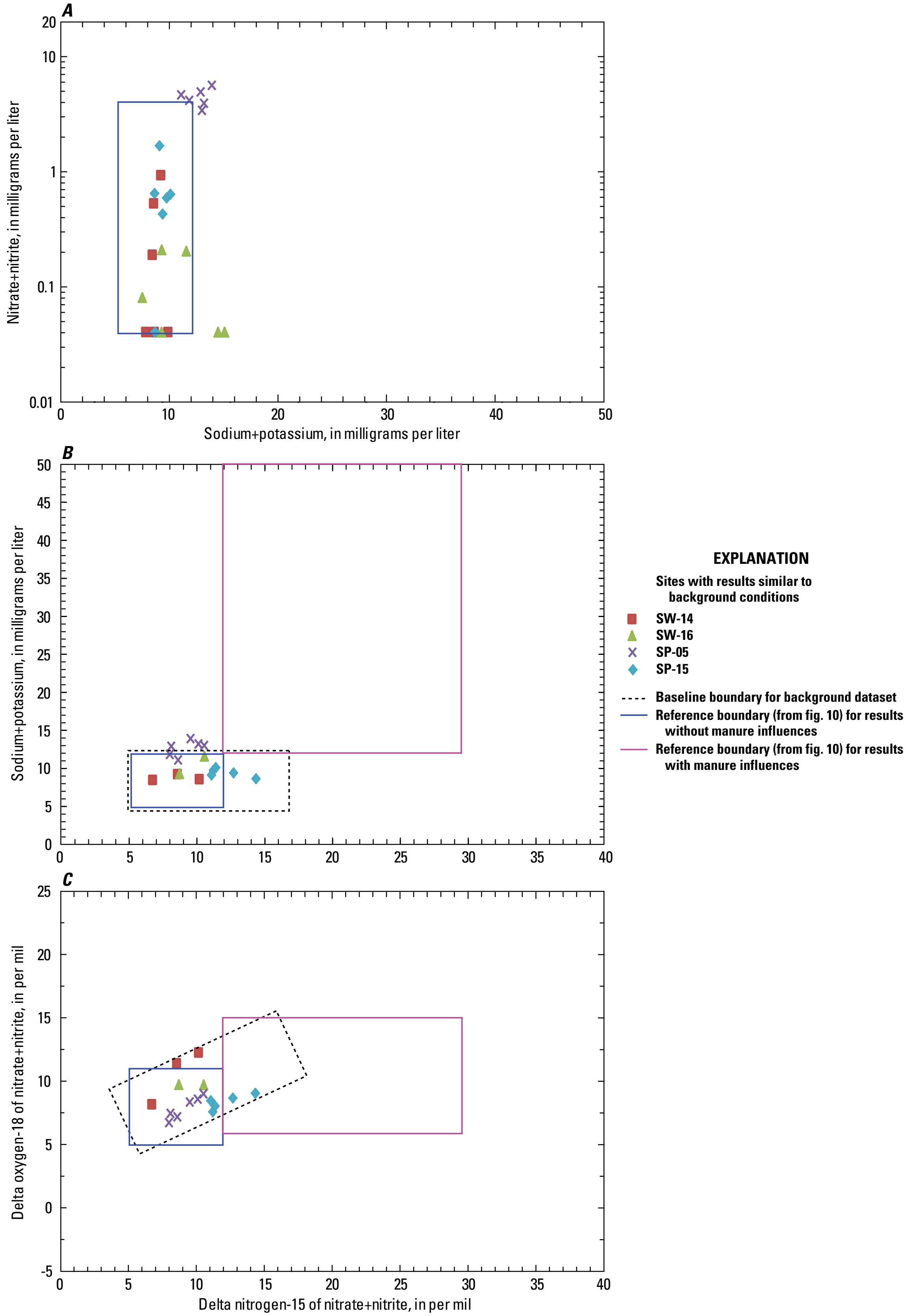

Figure 12. Graphs showing data comparisons of (*A*) sodium plus potassium to nitrate plus nitrite, (*B*) delta nitrogen-15 of nitrate plus nitrite to sodium plus potassium, and (*C*) delta nitrogen-15 to delta oxygen-18 of nitrate plus nitrite at four representative sites (SW-14, SW-16, SP-05, and SP-15) with results similar to background conditions.

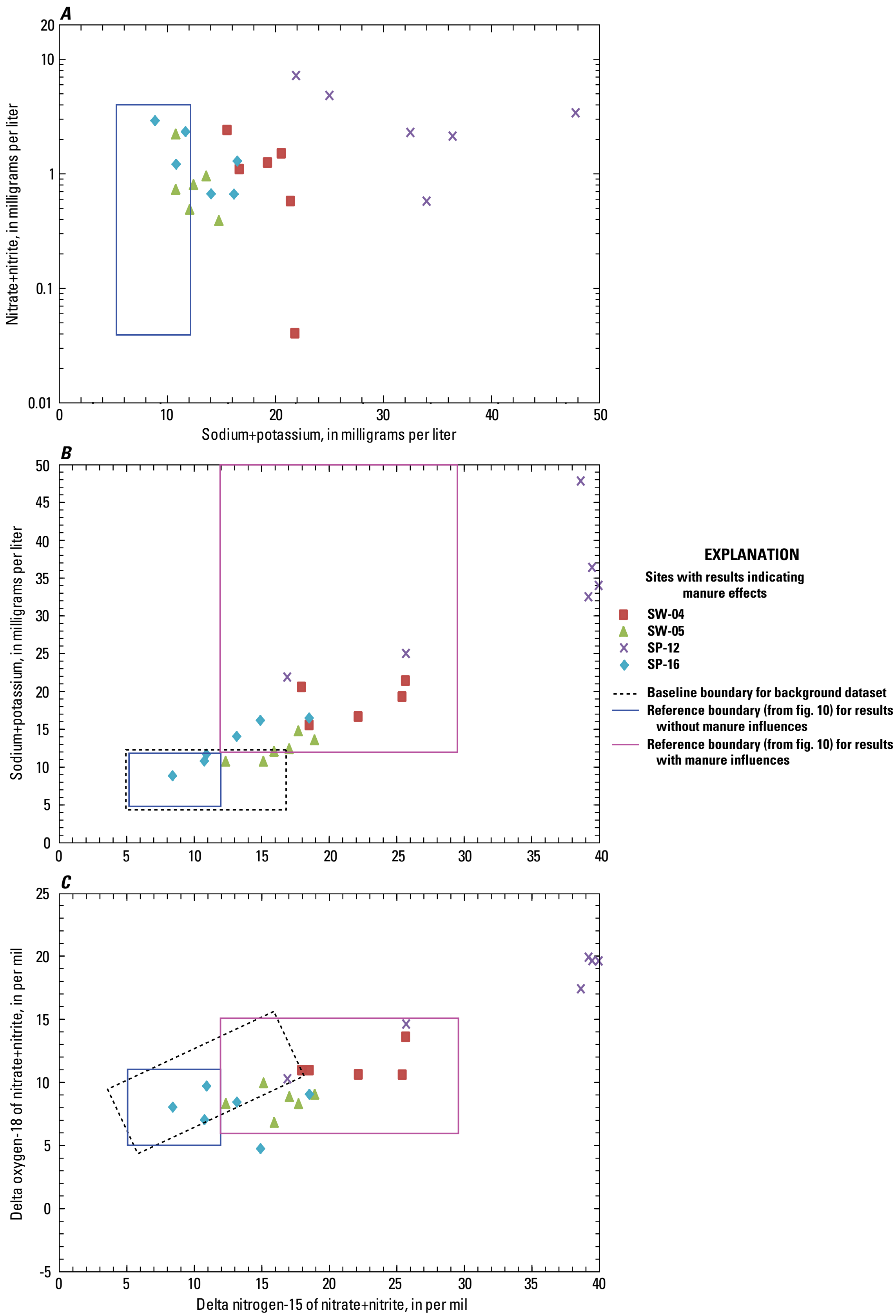

Figure 13. Graphs showing data comparisons of (*A*) sodium plus potassium to nitrate plus nitrite, (*B*) delta nitrogen-15 of nitrate plus nitrite to sodium plus potassium, and (*C*) delta nitrogen-15 to delta oxygen-18 of nitrate plus nitrite at four representative sites (SW-04, SW-05, SP-12, and SP-16) with results indicating manure effects.

On the basis of the comparisons of sodium+potassium concentrations, nitrate+nitrite concentrations, and the $\delta^{15}N$ and $\delta^{18}O$ of nitrate+nitrite values, 10 of the 36 CAFO sites (28 percent) had results similar to background conditions, and 21 of the sites (58 percent) had results with measurable CAFO manure effects (table 14). Note that the identification of those SW or SP watersheds as being similar to background conditions does not necessarily imply that CAFOs in those watersheds have no local influence on water quality, only that no distinction was noted at the watershed sampling location for the constituents that were examined. Three of the SW sites (SW-03, SW-08, and SW-15) and two of the SP sites (SP-03 and SP-08) had limited or indeterminate results for determining whether they were similar to background or manure influenced; these sites with unclear results were excluded from further evaluation.

The manure-influenced group of sites tended to have distinctly higher sodium+potassium concentrations (commonly between 11 and 33 mg/L) and $\delta^{15}N$ values of nitrate+nitrite (commonly between 11 and 26 ‰) relative to both the background and similar to background groups of sites, which commonly had sodium+potassium concentrations between 6 and 14 mg/L and $\delta^{15}N$ values of nitrate+nitrite between 6 and 15 ‰ (table 14; appendix A6). Based on the six sampling periods from June 2012 to April 2013, sodium+potassium concentrations and $\delta^{15}N$ values of nitrate+nitrite appear to be useful water-quality indicators for differentiating streams with measurable CAFO manure effects. It would be beneficial to base future similar analyses on a larger number of samples that more fully reflect hydrologic and seasonal variability in water-quality conditions among sites of interest.

Table 14. Statistical summary of selected water-quality constituents for the background sites, CAFO sites with results similar to background conditions, and CAFO sites with results reflecting manure influences.

[diss., dissolved; mg/L, milligrams per liter; N, nitrogen; O, oxygen; +, plus; <, less than; δ, delta; ‰, per mil]

Chemical constituent (unit)	Background sites[1]				Similar to background sites[2]				Manure-influenced sites[3]			
	Number of samples	10th percentile	Median	90th percentile	Number of samples	10th percentile	Median	90th percentile	Number of samples	10th percentile	Median	90th percentile
Sodium + potassium, diss. (mg/L)	106	6.35	9.23	12.9	54	6.48	9.57	14.5	124	10.8	16.66	32.7
Nitrate + nitrite, diss. (mg/L as N)	106	<0.040	0.048	0.505	60	<0.040	0.074	3.41	124	<0.040	0.692	4.27
$\delta^{15}N$ of nitrate + nitrite (‰)	40	6.08	9.39	15.10	27	7.33	6.74	12.42	95	10.80	16.28	25.70
$\delta^{18}O$ of nitrate + nitrite (‰)	40	6.26	9.43	13.29	27	4.96	2.54	11.42	95	6.50	9.16	14.62

[1]The background, or baseline, dataset includes the results of all 18 BK sites (BK-01 through BK-18).

[2]The sites with results deemed to be similar to background conditions include 6 SW sites (SW-02, 06, 07, 10, 14, and 16) and 4 SP sites (SP-01, 05, 15, and 17).

[3]The sites with results deemed to reflect manure influences include 9 SW sites (SW-01, 04, 05, 09, 11, 12, 13, 17, and 18) and 12 SP sites (SP-02, 04, 06, 07, 09, 10, 11, 12, 13, 14, 16, and 18).

Watershed Attributes Associated With CAFO Water-Quality Effects

Watershed environmental attributes were compared among the study sites with and without CAFO manure influences (see previous section). The five sites (SW-03, 08 and 15, and SP-03 and 08) with indeterminate results were not included in this analysis. The remaining 49 sites were grouped into three response categories: 18 background sites; 10 similar to background CAFO sites, and 21 manure-influenced CAFO sites. A classification tree model was developed to examine relations between selected watershed environmental variables and the three response categories (appendix A7).

The main intent in this analysis was to identify key differences in watershed characteristics associated with sites either having or not having measurable CAFO manure effects. Watershed characteristics analyzed as predictor (independent) variables in the model included drainage area size, land cover (percentages of forested land, cropland, grassland, and wetlands), soil drainage (percentages of HSGs total A, total B, total C, and D), swine CAFO attributes, and poultry CAFO attributes (appendix A7). The swine CAFO attributes included the total number of permitted active swine CAFOs, total swine barns and barn density, total swine and swine density, total swine weight and weight density, total acres available for applying swine-waste manure and acre density, and total generated PAN for each watershed site. The poultry CAFO attributes available for examination with the classification tree analysis were limited to the total number of identified poultry CAFOs, total poultry barns, and poultry barn density for each site. Results of the classification tree analysis, including the splits in the tree model, the selected environmental variable and value defining each split, and the response category with the number of sites classified in each category, are illustrated in figure 14 and summarized in table 15.

Table 15. Classification tree model results for the 49 study sites.

[#, number; <, less than; ≥, greater than or equal to; >, greater than; mi^2, square mile; %, percent; na; not applicable]

Split	Predictor variable and split value	Response category (# of sites)	Number of misclassified sites	Identity of misclassified sites (actual category)
1	Total active swine CAFOs < 1	Background group (18)	0 of 18	na
1 2	Total active swine CAFOs ≥ 1 Swine barn density > 2.9 barns/mi^2	Manure-influenced group 1 (15)	0 of 15	na
1 2 3	Total active swine CAFOs ≥ 1 Swine barn density < 2.9 barns/mi^2 Wetlands > 14.4 %	Similar to background group 1 (7)	0 of 7	na
1 2 3 4	Total active swine CAFOs ≥ 1 Swine barn density < 2.9 barns/mi^2 Wetlands < 14.4 % Total acres available for applying swine-waste manure > 52.4	Manure-influenced group 2 (5)	0 of 5	na
1 2 3 4	Total active swine CAFOs ≥ 1 Swine barn density < 2.9 barns/mi^2 Wetlands < 14.4 % Total acres available for applying swine-waste manure < 52.4	Similar to background group 2 (4)	1 of 4	SP-10 (Manure influenced)

The tree model selected the presence/absence of active swine CAFOs, swine barn density, percentage of wetlands, and acres available for applying swine-waste manure as the best discriminators, or predictor variables, for classifying the study sites among the background, similar to background, and manure-influenced response categories or groups (fig. 14; table 15). The model was highly successful in accurately classifying the sites into the appropriate response categories. Only 1 of the 49 sites was misclassified (table 15). The first, or primary, split in the tree model was based on the presence/absence of active swine CAFOs in the watersheds (fig. 14). All 18 of the BK sites were placed in the background group because none of the BK sites contain any active swine CAFOs.

Interestingly, the 15 SW sites and 16 SP sites, which all had at least 1 active swine CAFO, were further differentiated into two groups for the manure-influenced category (referred to as manure-influenced groups 1 and 2) and two groups for the similar to background category (referred to as similar to background groups 1 and 2) on the basis of subsequent splits in swine barn density, percentage of wetlands, and total acres available for applying swine-waste manure (fig. 14; table 15). The splits among these four groups indicate how variations in these particular swine CAFO and land-cover variables may inhibit or promote the ability of the watersheds to mitigate manure effects on water quality in streams receiving inputs from swine CAFO application fields.

When swine barn density in the watersheds was greater than 2.9 barns/mi^2, 15 sites (7 SW and 8 SP sites) with measurable CAFO manure effects on water quality were correctly placed in manure-influenced group 1 (fig. 14). The SW and SP sites in manure-influenced group 2 and similar to background groups 1 and 2 all had swine barn densities that were less than 2.9 barns/mi^2 (fig. 14; table 15). Seven sites (4 SW and 3 SP sites) without measurable CAFO manure effects on water quality were correctly placed in similar to background group 1 when the amount of wetlands in the watershed was greater than 14.4 percent. In comparing manure-influenced group 1 to similar to background group 1 (fig. 14), the SW and SP sites with measurable CAFO manure effects had higher swine barn densities (median of 4.8 barns/mi^2), more acres available for applying swine manure (median of 243.7 acres), and less wetlands (median of 12.1 percent) relative to the SW and SP sites without measurable CAFO manure effects. Similar to background group 1 had lower swine barn densities (median of 1.2 barns/mi^2), fewer acres available for applying swine manure (median of 66.9 acres), and more wetlands (median of 20.8 percent).

When both swine barn density was less than 2.9 barns/mi^2 and wetlands was less than 14.4 percent, the SW and SP sites with or without measurable CAFO manure effects were separated on the basis of the total acres available for applying swine-waste manure in the watersheds (fig. 14; table 15). Five sites (2 SW and 3 SP sites) were correctly placed in manure-influenced group 2 when total acres available were greater than 52.4; four sites (2 SW and 2 SP sites) were placed in similar to background group 2 when total acres available were less than 52.4 (fig. 14). Similar to background group 2 contained misclassified site SP-10, which actually belongs to the manure-influenced category (table 15). Site SP-10 had a swine barn density of 2.7 barns/mi^2, just below the split value of 2.9 barns/mi^2, wetlands of 8.7 percent, and total available acres of 39.2, which resulted in its placement in similar to background group 2. The sites in manure-influenced group 2 and similar to background group 2 had comparable median values of swine barn density (2.2 and 2.5 barns/mi^2, respectively) and wetlands (11.7 and 8.4 percent, respectively). The primary distinction between these groups is that the total available acres for applying swine manure for the sites in manure-influenced group 2 (median of 164.1 acres) were about 5 times higher than the total available acres for the sites in similar to background group 2 (median of 34.0 acres).

The classification tree analysis, as well as the other data evaluations in this report, indicate that land-applications of waste manure at swine CAFOs had an effect on water-quality conditions in streams at many, but not all, of the SW and SP study sites. Measurable effects of CAFO waste manures on stream water quality were most evident in those SW and SP watershed study sites having lower percentages of wetlands combined with higher swine barn densities and (or) higher total acres available for applying waste manure at the swine CAFOs. Conversely, the SW and SP watersheds with stream water quality similar to background agricultural conditions were associated with lower swine barn densities combined with higher percentages of wetlands or lower total acres available for applying waste manure at the swine CAFOs.

None of the poultry CAFO attributes examined with the tree model were selected as predictor variables for identifying differences between the sites with and without CAFO manure effects. This should not be misconstrued to indicate that poultry CAFO manures do not have an influence on stream water quality but rather may be a function of the limited poultry CAFO attribute data that were available for examination, as well as the nature of the watershed sites selected for this study, which had a primary emphasis on swine CAFOs. Thirteen of the 16 SP study sites included in the classification tree analysis (appendix A7) had substantially more swine barns (ranging from 4 to 59) than poultry barns (ranging from 1 to 8) in the watersheds. These watersheds likely received larger proportions of land-applied swine manure relative to poultry litter. Additional water-quality data, as well as more detailed information on poultry CAFO attributes (such as the types and numbers of poultry raised), from watersheds only containing poultry CAFOs would allow further comparisons to swine-only watersheds to better understand whether swine manure and poultry litter have similar or different effects on water quality.

The classification tree model provides a useful approach for exploring potential CAFO manure effects in similar, small (1 to 18 mi^2) Coastal Plain watersheds where water-quality data are lacking. Potential sites could be screened on the basis of the influential watershed attributes (swine barn density, acres available for applying swine manure, and percentage of wetlands) identified by the model. Results could help water-resource managers and researchers identify streams with high potential for manure influences on water quality in order to prioritize them for further investigation and (or) targeted best management practices. The classification tree model can be refined as additional CAFO attribute information and water-quality data become available, both for existing study sites as well as new locations. The inclusion of data on specific manure-disposal practices at both swine and poultry CAFOs (including specific application fields and the frequency, timing, and amounts of applied manures) would enhance understanding of the effects of swine and poultry waste manures on stream water quality in different agricultural settings of the North Carolina Coastal Plain.

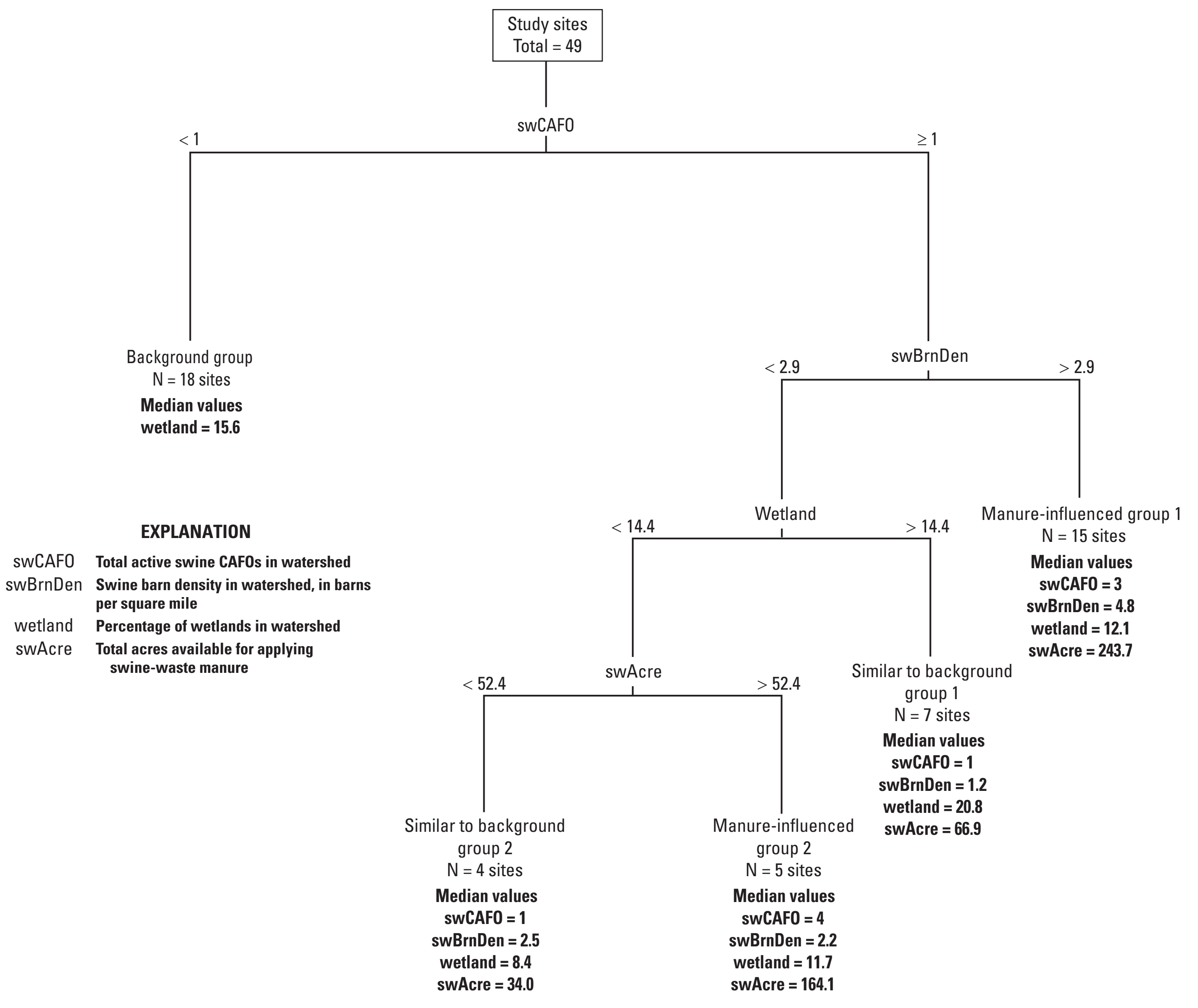

Figure 14. Classification tree model identifying the environmental predictor variables that best classified the 49 examined sites among the background, similar to background, and manure-influenced response categories.

Summary and Conclusions

Water quality was evaluated at 54 agricultural stream sites in the North Carolina Coastal Plain for the period June 2012 through April 2013. Water-quality data and detailed watershed attributes were collected, compiled, and statistically analyzed to determine differences among streams draining watersheds with and without land-applied CAFO waste manures. Three general watershed land-use types, or groups, were examined during the study, including 18 background watersheds with no active CAFOs (BK sites), 18 watersheds with one or more active swine CAFOs but no poultry CAFOs (SW sites), and 18 watersheds with at least one active swine CAFO and one active dry-litter poultry CAFO (SP sites). The watersheds had drainage areas ranging from 1.2 to 17.5 mi^2 and land cover was composed predominantly of cropland, forests, and wetlands. Most watersheds had low gradient, swampy floodplain streams that were typically characterized by slow velocities, high organic matter, and relatively low dissolved oxygen. None of the watersheds contained permitted point-source discharge facilities, cattle CAFOs, or wet-poultry CAFOs. Conventional fertilizers used for crop production were the primary source of nutrients at the BK sites. Animal-waste manures applied to agricultural fields associated with the swine or poultry CAFOs represented additional sources of nutrients at the SW and SP study sites.

Water-quality data included field measurements of water temperature, specific conductance, pH, and dissolved oxygen, and laboratory analyses of major ions, nutrients, and stable isotopes. Samples were collected at the 54 primary sites during 6 bimonthly sampling periods from June 2012 to April 2013. An additional 23 secondary sites within 9 of the primary watershed sites were sampled once during April 2013 to provide additional data at stream sites directly adjacent or in close proximity to swine CAFOs and (or) background agricultural areas. Regional precipitation and streamflow data, along with δ^2H and $\delta^{18}O$ isotopic results for precipitation and stream samples, were used to assess general hydrologic conditions during the sampling periods.

ANOVA and multiple-comparison statistical tests were performed to characterize differences in stream water quality among the six sampling periods and the three (BK, SW, and SP) watershed land-use types. Most of the water-quality properties and constituents varied significantly among sampling periods, changing both seasonally and in response to hydrologic conditions. Nutrient differences among the sampling periods indicate that the relations between seasonal climatic differences, streamflow conditions, and instream biotic and abiotic processes are complex, and their integrated effects can have varying degrees of influence on individual nutrients in different watersheds. These findings are important to consider when developing approaches to assess stream nutrient conditions in similar Coastal Plain settings and can inform the development of sampling strategies that capture seasonal and (or) hydrologic variability. For example, the highest median concentrations of dissolved oxygen and nitrate+nitrite were observed during February 2013, when higher streamflows appeared to reflect more overland contributions of nitrate from upstream field-drainage ditches. Nitrate in the field ditches is carried to the main stem of the streams during higher flows and is subject to less instream processing, including denitrification and assimilation, when stream water temperatures are colder and dissolved oxygen concentrations are elevated. Nitrate+nitrite tended to be lowest during warm and dry sampling periods, when conditions were favorable for denitrification. In contrast, median concentrations of ammonia, total organic N, ortho-P, and total P were lowest during February. Environmental factors that likely influenced the various forms and instream concentrations of the N and P constituents include assimilation and release by algae and aquatic plants, redox conditions, microbially mediated reactions, adsorption and desorption processes, and biogeochemical exchange between streambed sediment and the overlying water column.

Water quality also varied significantly among the three watershed land-use types. Median values of specific conductance, several major ions (magnesium, sodium, potassium, and chloride), and nitrogen fractions (ammonia+organic N, ammonia, nitrate+nitrite, total N, and $\delta^{15}N$ of nitrate+nitrite) were higher for the SW and SP land-use groups as compared to the BK group, which have no active CAFOs. The higher concentrations of these constituents reflect the influence of swine-waste manure storage or applications at the SW sites and swine- and (or) poultry-waste manure storage or applications at the SP sites. No significant differences in water temperature, dissolved oxygen, calcium, total organic N, ortho-P, total P, or $\delta^{18}O$ of nitrate+nitrite were noted among the land-use groups. The disparity observed between N and P response among the site groups may reflect differences in transport pathways or instream processing that influenced instream concentrations of these two classes of nutrients. When comparing the land-use groups, there was an overall measurable effect of animal-waste manures on stream water quality for the SW and SP watersheds relative to the BK watersheds; however, this does not mean that CAFO waste manures had an observable effect on water-quality conditions at every SW and SP site. Additional evaluations were performed on the water-quality data to distinguish those SW and SP sites where effects of CAFO waste manures were evident.

At the majority of individual SW and SP watersheds, measurable CAFO effects on water quality were clearly distinguished. At other sites, effects were less evident. Elevated concentrations of nitrate+nitrite did not necessarily indicate a CAFO effect; conversely, low nitrate+nitrite concentrations did not necessarily indicate the absence of a CAFO effect. An integrated evaluation of nitrate+nitrite concentrations, sodium+potassium concentrations, and stable isotopes ($\delta^{15}N$ and $\delta^{18}O$) of nitrate+nitrite was used to differentiate which SW and SP sites did or did not have a CAFO waste-manure signature.

Streams with CAFO manure effects typically had higher sodium+potassium concentrations (commonly between 11 and 33 mg/L) and $\delta^{15}N$ values of nitrate+nitrite (commonly between 11 and 26 ‰) relative to streams reflecting background agricultural conditions, which commonly had sodium+potassium concentrations between 6 and 14 mg/L and $\delta^{15}N$ values of nitrate+nitrite between 6 and 15 ‰. Denitrification affected the $\delta^{15}N$ and $\delta^{18}O$ signatures of nitrate+nitrite at some sites and must be accounted for during interpretations of nutrient sources.

As part of the evaluation, individual SW and SP sites were differentiated into two groups, including (1) those with results that were similar to background conditions, and (2) those with results reflecting CAFO waste-manure effects. Ten of the 36 SW and SP sites (28 percent) had water quality similar to background conditions. Twenty-one of the SW and SP sites (58 percent) had distinct water-quality differences, reflecting swine- and (or) poultry CAFO manure effects. Five of the SW and SP sites (14 percent) had limited or indeterminate results for determining whether they were similar to background or manure influenced; these sites were omitted from further evaluation. On the basis of the results of this study, it is apparent that land-applications of waste manure at swine CAFOs influenced ion and nutrient chemistry in many of the North Carolina Coastal Plain streams that were studied. In particular, sodium+potassium concentrations coupled with $\delta^{15}N$ values of nitrate+nitrite were useful water-quality indicators for distinguishing sites with measurable CAFO manure effects.

Relations in watershed environmental attributes among the similar to background and manure-influenced site groups were examined through classification tree analysis. The classification tree model identified swine barn density, percentage of wetlands, and total acres available for applying swine-waste manures as the best discriminators, or predictor variables, for classifying sites among the similar to background and manure-influenced groups. Variations in these particular attributes appeared to influence those watersheds where CAFO effects on water quality were either evident or mitigated. Measurable effects of CAFO waste manures on stream water quality were most evident in those SW and SP watersheds having lower percentages of wetlands combined with higher swine barn densities and (or) higher total acres available for applying waste manure at the swine CAFOs. Stream water quality was similar to background agricultural conditions in SW and SP watersheds with lower swine barn densities coupled with higher percentages of wetlands or lower acres available for swine manure applications.

The classification tree model provides a useful approach for examining potential CAFO manure effects on stream water quality among similar Coastal Plain watersheds, including those where water-quality data are lacking. The model can serve as an exploratory tool to identify watersheds that might warrant further examination and (or) targeted best management practices. The study model can be refined as additional watershed attribute information and water-quality data become available. Additional water-quality data, poultry CAFO attribute data, and information on manure disposal practices at both swine and poultry CAFOs would enhance scientific understanding of the effects of swine and poultry waste manures on stream water quality under different agricultural settings.

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ATTACHMENT 9

SEEPAGE EVALUATION OF OLDER SWINE LAGOONS IN NORTH CAROLINA

R. L. Huffman

ABSTRACT. *Thirty−four swine waste lagoon systems in North Carolina were examined for evidence of seepage losses to the shallow groundwater. All were constructed prior to the state's January 1993 adoption of stricter construction standards. Mineral nitrogen concentrations (ammoniacal plus nitrate nitrogen) were used as the primary indicators of seepage impacts. Total mineral concentrations were compared to the U.S. EPA drinking water standard for nitrate−N of 10 mg/L. The shallow groundwater on approximately one−third of the 34 systems met the EPA standard at a distance of 38 m (125 ft) downgradient from the lagoon(s).*

Keywords. *Groundwater, Lagoons, Seepage, Water quality.*

Swine production in North Carolina grew dramatically in the late 1980s and early 1990s, nearly quadrupling during that period. Virtually all of the growth involved intensive confinement systems that use wet waste handling and anaerobic lagoons. As of the beginning of 1993, state regulation required lagoon construction to meet the design standards recommended by the USDA Soil Conservation Service (SCS, changed in 1994 to Natural Resources Conservation Service, NRCS) in Appendix 10D of the *Agricultural Waste Management Field Handbook* (SCS, 1992). As of that date, well over 2000 swine waste lagoons were already in service in North Carolina (NCDENR, 1997). Many of those were sited and designed with NRCS assistance, but did not specifically include clay (or equivalent) liners as specified in the 1993 regulations.

Waste lagoons were expected to develop a seal at the liquid−soil interface that would impede seepage. Mechanisms for the phenomenon include physical clogging of pores by particulates or microbial gums and sludge (Chang et al., 1974; Barrington et al., 1987; Barrington and Madramootoo, 1989; Davis et al., 1973; Maulé et al., 2000). Many early studies concluded that the sealing effect would limit seepage to rates that would not significantly contaminate the shallow groundwater, even with sandy soils (e.g., Hills, 1976; Ritter et al., 1984; Miller et al., 1985; Rowsell et al., 1985). Some later studies found that self−sealing did not adequately control seepage on coarser materials (Ritter and Chirnside, 1990; Korom and Jeppson, 1994). Recent investigations of older, unlined lagoons in North Carolina found that about half of the lagoons studied had seepage losses high enough to exceed the drinking water standard for nitrate−nitrogen (Huffman and Westerman, 1995; Westerman et al., 1995). Those studies found that seepage problems were often localized, suggesting problems in construction where sandy lenses were not properly excavated and replaced with better materials. They reported concentrations of mineral nitrogen (nitrogen as both nitrate and ammonia) as high as 470 mg/L. Ham (2002) reported on ammonia and organic nitrogen distributions under lagoons in Kansas, showing concentrations in excess of 1000 mg/kg within the first meter and tapering off with depth. He attributed this to adsorption of the ammonia onto clays. In the stratified systems of the Atlantic Coastal Plain, where most North Carolina swine production is concentrated, nitrate from fertilizers and waste applications is very rarely observed below the first significant clay layer (Gilliam et al., 1996).

Concern for the environmental impacts of seepage have centered on excessive concentrations of nitrate−nitrogen in the groundwater. Although lagoon wastewater contains very little nitrate, the nitrogen present in seepage as ammonia, ammonium, or organic nitrogen can be converted to nitrate in the groundwater. The concern for possible groundwater contamination from lagoons prompted the North Carolina General Assembly to fund a large−scale survey of the older swine waste lagoons in the state. The survey was to extend the knowledge base by investigating a relatively large number of lagoons to see what contaminant concentrations were present in the shallow groundwater. The survey was to focus on systems built before the regulatory adoption of construction standards in 1993.

The primary objective of this survey was to determine the proportion of older (i.e., pre−1993) swine waste lagoons that pose a threat to local groundwater quality. A secondary objective was to determine whether soil textures of the upper profile (like those reported in soil surveys) could predict lagoon performance with respect to seepage losses.

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METHODS

The project was planned in consultation with the Groundwater Section of the Division of Water Quality of the North Carolina Department of Environment and Natural Resources. Since the focus was on systems that had been in operation for at least five years, it was assumed that seepage

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plumes within the area of interest would be well developed. It was therefore agreed that a one−time snapshot, as opposed to monitoring over some period of time, would be sufficient to indicate the presence and approximate severity of seepage contamination. Estimates of time and cost per site led to choosing a target of 40 sites. With the intent of providing a statistically valid sampling of lagoons within the state, a goal was set to identify 100 candidate sites from which 40 could be randomly selected.

Site Selection

Over 2600 letters were mailed to North Carolina swine producers to solicit cooperation in this project. To be included in the project, the lagoons had to predate 1993 and have reasonable access to a distance of at least 38 m (125 ft) from the lagoon to permit sampling operations. Only 136 responses were received. Fifty−two of those volunteered to cooperate in the study. Each of those sites was visited to assess its usability. The primary considerations were accessibility for sampling operations and absence of confounding factors such as mortality burial pits. Of the 52 sites, 22 had clear access, 8 had somewhat limited access, 9 had more limited access but could be used, 5 had very limited access, and 8 had no access. Limitations to access included buildings, fences, property boundaries, and woods.

In general, owners were not able to provide information regarding the specifics of lagoon construction, so it could not be determined whether the lagoons had clay liners. Common practice during that period did not include use of clay liners, so it is reasonable to assume that the construction materials were the soils on site.

The 40 candidate sites deemed most usable were scheduled for inclusion in the study. As the work progressed, some of those sites were lost for various reasons, such as changes in ownership. Two others were volunteered that had not responded to the original solicitation. The final number of sites included in the study was 36. Most of these sites were located in the Coastal Plain physiographic province of North Carolina. Three of the sites were in the Piedmont province. At each site, a topographic survey was conducted to determine the relative positions of the lagoons, buildings, nearby surface waters, and the general shape of the land.

Groundwater Sampling

Many of the sites had systems of two or more lagoons. For sites where the sampling procedure could not isolate a particular lagoon within a system, the lagoon complex was treated as a single unit. Where one or more of the lagoons on a site was constructed under the new regulations, the study focused only on the pre−1993 lagoon(s).

In North Carolina, standard permits for industrial waste sites define a review boundary at 38 m (125 ft) from a containment structure and a compliance boundary at 76 m (250 ft). Prior to 1993, lagoons were not individually permitted, but rather deemed “permitted by regulation” if the systems met the operational criteria in the North Carolina Administrative Code (NCAC, 2004). Although review and compliance boundaries were not defined under that arrangement, 38 m (125 ft) was selected, in consultation with the Groundwater Section, as a standard distance from the lagoons for groundwater sampling.

Three or more exploratory borings were made with 114 mm (4.5 in.) augers at approximately 38 m (125 ft) from

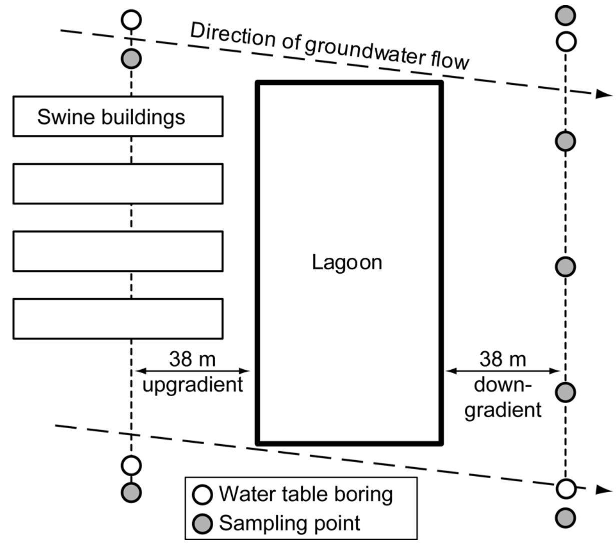

Figure 1. Layout of sampling points on a typical lagoon site.

the lagoon. Distances were measured from the interior top of the bank. The textures of cuttings were determined in the field, by feel, and recorded. Relative elevations of the water table in the boreholes were used to determine the most likely direction of groundwater flow. Once the direction of the local gradient was determined, sampling points were selected 38 m (125 ft) upgradient and 38 m (125 ft) downgradient of the lagoon. If the gradient was not well defined and the area was accessible, sample points were selected all around the lagoon(s). Buildings or other obstructions sometimes made it impossible to sample completely around the lagoons. In general, at least two sampling points were selected in upgradient positions, and four or more were selected in downgradient positions. Figure 1 shows a typical layout and how the points were chosen. Most systems were constructed on sloping land with the buildings situated upslope from the lagoon(s) to allow wastes to flow by gravity from the buildings into the lagoon(s). Since shallow groundwater generally flows in the direction of the surface slope, the gradients were usually similar to those shown in figure 1.

One of two methods was used for collecting groundwater samples, depending on the penetrability of the soils on the site. On most sites, it was possible to drive direct−push sampling probes (Diedrich Drill, Inc., La Porte, Ind.) into the

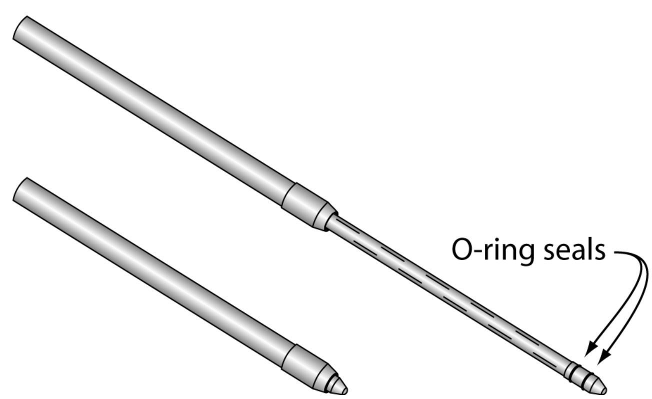

Figure 2. Direct−push tooling used for groundwater sampling. Top: slotted inner rod extended for water collection. Bottom: slotted inner rod retracted for driving to sampling depth.

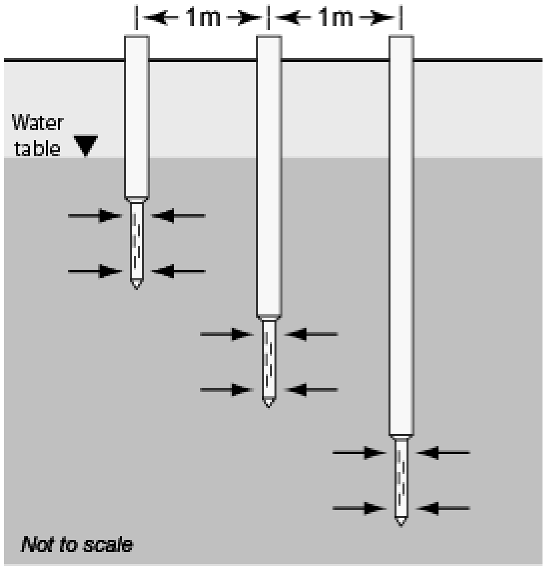

Figure 3. Profile sampling using direct−push tooling. Sample is bailed from slotted inner rod.

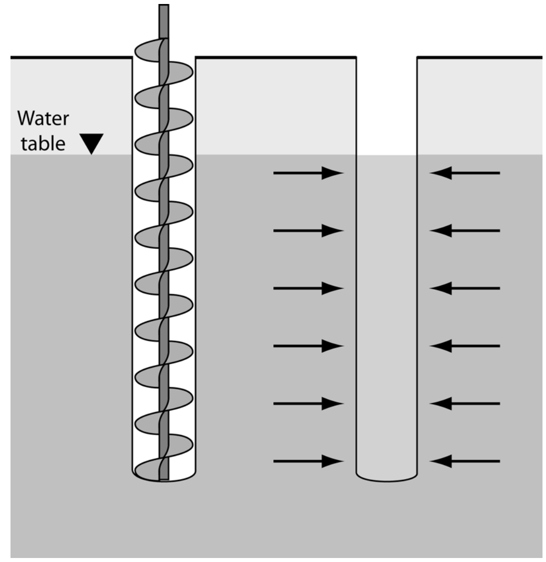

Figure 4. Auger−hole method for water sampling. Sample is bailed from open auger hole.

soil. The probes consist of a 51 mm (2.0 in.) diameter outer casing and a 29 mm (1.125 in.) diameter inner rod in 1.22 m (4 ft) sections, with drive points, shoes, and adapters. The probe assembly is illustrated in figure 2. Figure 3 illustrates sampling at multiple depths with the direct−push tooling. To avoid disturbance or interference between probes, multiple probes at one location were spaced approximately 1 m apart. The inner rod and outer casing were sealed together by two O−rings while the probe was driven to the desired depth. The inner rod was then pushed an additional 0.6 m (2 ft) to expose the slotted inner rod section. The slots were 51 mm (2.0 in.) long and 0.25 mm (0.010 in.) wide. A 250 mL sample was collected using a 12.7 mm (0.50 in.) diameter stainless steel bailer. The first flows into the sampler were collected for analysis. Tooling was cleaned between uses. The shallowest sample at a location was collected just below the water table. Additional samples were taken at 2.4 m (8.0 ft) intervals until either a restrictive layer was encountered or the capability of the drill rig was reached.

Table 1. Analytical methods.

	Method
TKN	Persulfate digestion and ammonia−salicylate method for automated analysis. Method 351.2 (EPA, 1979) with slight modifications including dialysis.
NH_3−N (ammonia)	Ammonia−salicylate method for automated analysis. Method 351.2 (EPA, 1979) or Standard Methods 418.F (APHA, 1981) with slight modifications including dialysis.
NO_3−N + NO_2−N (nitrate + nitrite nitrogen)	Cadmium reduction method for automated analysis. Method 353.2 (EPA, 1979), Technicon Industrial Method No. 100−70W (1973), or Standard Methods 418.F (APHA, 1981) with slight modification including dialysis.
Cl (chloride)	Ferricyanide method for automated analysis. Method 325.2 (EPA, 1979) or Standard Methods 407 D (APHA, 1981) with slight modifications including dialysis.
pH	Electrometric method. Method 150.1 (EPA, 1979) or Standard Methods 205 Conductivity (APHA, 1981).

The second method was used at eight sites because the resistance of the soil to the probes was so great that they could not be driven to the desired depths. In those cases, a hole was augered to about 1.5 m (5 ft) below the water table and a water sample was bailed from the open hole (fig. 4). This method limited sampling to only one sample for each location rather than the vertical profiles possible with the direct−push probes. Samples were collected as soon as there was sufficient water in the holes to permit bailing. Although it was less informative, sampling from auger holes was sufficient to indicate whether seepage was present in the shallow groundwater.

All samples were immediately placed on ice and delivered to the Environmental Analysis Laboratory in the Department of Biological and Agricultural Engineering at North Carolina State University. The analytical methods used are listed in table 1.

Seepage Classification

Anaerobic swine lagoons are strong sources of ammoniacal nitrogen. Most of the nitrogen present in the wastewater is in an ammoniacal form (NH_3 or NH_4^+). A minor fraction (typically about 20%) is in organic forms. Nitrate−N is not present in anaerobic lagoons in significant concentrations (Westerman et al., 1990; Barker et al., 2001). Where seepage rates were high, ammoniacal nitrogen was usually present in the shallow groundwater at concentrations well above the typical background concentrations, which are less than 1 mg/L (Spruill et al., 1996). Where ammoniacal nitrogen dominates, conditions are not conducive to complete conver−

Table 2. Classification criteria for evidence of seepage.

Class	Increase in Mineral−N Concentration from Upgradient to Downgradient Positions (mg/L)
None	<2
Weak	2−10
Moderate	10−40
Strong	40−100
Very strong	>100

Table 3. Operation/site characteristics and seepage summary for lagoon systems. Concentrations are sums of ammoniacal and nitrate nitrogen.

Site	Operation[a] Type	Operation[a] Size (Mg)	Operation[a] Age (yr)	Lagoons No.	Lagoons Surface Area (ha)	Adjacent Land Use[b] Up−gradient	Adjacent Land Use[b] Down−gradient	Soil Textures, 0−1.5 m	Avg. Water Table Depth (m)	Sample Method[c]	Upgradient Samples n[d]	Upgradient Samples Mean (mg/L)	Upgradient Samples Range (mg/L)	Downgradient Samples n[d]	Downgradient Samples Mean (mg/L)	Downgradient Samples Range (mg/L)	Seepage Class
1111	Fe−Fi	162	6	1	0.7	F	W	sc	2.1	P	3	2.1	0.1−3.4	8	10.0	1.0−22.6	Moderate
1116	We−Fe	71	7	1	0.2	W	W	fsl	1.8	P	6	0.0	0.0−0.1	5	0.1	0.0−0.3	None
1410	Fe−Fi	166	8	1	0.7	F	F	s, sl	2.1	P	4	3.6	0.1−13.7	9	62.2	4.2−253.8	V. Strong
1954	Fe−Fi	245	10	2	1.0	F	W	s	1.7	P	2	17.3	3.3−31.3	17	93.6	16.7−472.8	V. Strong
1961	Fe−Fi	228	7	2	1.1	F	W	sc	4.0	P	6	4.2	0.0−16.4	17	0.6	0.0−7.4	Weak
2757	Fe−Fi	365	15	1	0.3	F	W	sl	1.2	P	4	35.1	16.2−46.9	11	13.8	0.1−51.1	Moderate
3200	Fa−Fi	64	17	1	0.3	F	W	cl	5.0	A	2	11.1	10.4−11.7	7	29.6	7.7−80.0	Strong
3311	We−Fe	35	7	1	0.2	W&F	W	cl	1.6	P	3	1.1	0.4−1.6	4	3.7	0.2−13.8	Moderate
3373	Fe−Fi	150	6	1	0.9	W	F	c, cl	3.4	P	3	0.5	0.0−1.2	12	4.6	0.0−13.3	Moderate
3914	Fe−Fi	180	6	1	0.7	F	F	sl	1.7	P	3	9.4	2.2−15	9	11.7	0.5−32.8	Moderate
3933	Fe−Fi	566	7	1	0.7	F	F	sc	5.9	A	2	20.0	9.2−30.8	5	116.8	3.3−332.0	V. Strong
4444	Fe−Fi	187	20	2	0.3	W	F	cl, scl	1.8	P	1	2.9	−−	9	1.8	0.6−3.7	None
4547	Fe−Fi	1669	24	2	1.4	F	F	scl	2.0	P	4	10.5	5.9−13.8	13	58.6	0.5−182.1	V. Strong
4551	We−Fe	36	6	1	0.2	W&F	F	ls	1.9	P	3	7.3	4.0−9.7	7	74.1	0.4−391	V. Strong
4589	Fe−Fi	103	20	3	0.5	F	F	sl	2.5	P	2	17.8	14.6−21.1	9	29.6	0.3−129.1	V. Strong
4649	Fe−Fi	76	14	1	0.5	F	F	scl	1.8	P	5	9.1	2.5−26.8	15	26.2	0.1−79.1	Strong
4777	Fe−Fi	225	29	2	1.0	W	W	cl, scl	1.5	P	2	0.1	0.0−0.3	5	0.2	0.0−0.5	None
5437	We−Fe	35	8	1	0.3	W	F	s, ls	1.2	P	3	0.6	0.2−0.9	10	12.0	0.1−33.4	Moderate
5555	Fa−Fi	193	23	2	0.2	F	W	cl, sl	1.9	A	1	0.6	−−	4	15.0	8.7−20.7	Moderate
5827	Fe−Fi	270	6	2	1.1	W&F	F	cl	0.9	P	2	10.3	7.9−12.8	21	29.4	5.5−79.9	Strong
6603	Fe−Fi	255	11	4	2.0	F	F	sl	3.2	P	5	9.9	0.5−25.1	7	19.3	2.2−55.1	Strong
6633	We−Fe	35	7	1	0.2	F	W	sicl	2.0	P	2	1.0	0.2−1.8	5	0.5	0.1−1.7	None
6653	Fa−We	236	6	1	0.9	F	F	l, cl	4.2	P	2	5.3	2.8−7.7	12	37.1	4.3−329.1	V. Strong
6726	Fe−Fi	92	6	2	0.3	W	W	ls	1.8	P	2	0.8	0.8−0.9	4	86.2	32.0−232	V. Strong
7144	Fa−We	192	20	1	0.5	F	F	cl	2.8	P	2	3.1	2.9−3.4	10	11.9	0.3−24.7	Moderate
7225	Fe−Fi	22	19	2	0.4	F	F	ls	1.2	P	14	2.8	0.2−19.5	11	53.3	3.2−279.0	V. Strong
7286	Fe−Fi	294	7	1	1.1	F	W	cl	1.1	P	2	4.3	1.9−6.7	12	10.0	0.2−44.3	Strong
7674	Fa−Fi	76	7	1	0.5	W&F	F	sic	7.7	A	1	4.8	−−	3	1.3	0.0−3.3	None
7777	Fa−We	189	na[e]	1	0.9	W&F	W	cl	2.2	A	1	2.9	−−	5	3.3	0.9−7.7	Weak
7940	We−Fe	393	8	2	2.1	F	W&F	sc, cl, c	1.6	A	5	7.8	0.2−14.7	8	18.8	0.1−68.9	Strong
8158	Fe−Fi	35	7	1	0.3	W	F	l, c	1.2	P	4	0.4	0.3−0.6	16	5.3	0.2−35.8	Moderate
8829	Fa− Fe	721	10	1	0.9	W&F	F	sic, cl, c	2.1	A	1	1.2	−−	7	17.1	10.4−28.6	Moderate
8971	Fa− We	250	7	2	1.2	F	F	scl	1.7	P	4	19.1	6.3−37.0	8	20.2	0.7−89.9	Strong
9087	Fa−Fi	153	9	2	1.3	F	W	sl, c	5.6	A	2	9.1	6.7−11.5	5	6.1	1.5−17.5	Moderate

[a] Fa = farrow; We = wean; Fe = feeder; Fi = finish. “Size” refers to design steady−state live weight. Source: NCDENR (1997).
[b] W = woods, F = fields, and W&F = mixture of woods and fields.
[c] P = direct push probe, and A = auger.
[d] n = number of samples analyzed from a site−location combination.
[e] Estimated at 15 years.

sion to nitrate (NO_3^-). This is probably due to a combination of high seepage and limited oxygen availability in the saturated sediments. High ammonia concentrations may also be toxic to nitrifying bacteria (Westerman et al., 1995). Where nitrate is the dominant form, the oxygen supply is sufficient to permit nitrification of the ammoniacal nitrogen in the seepage.

Assessments of seepage were based on the sum of ammoniacal nitrogen and nitrate nitrogen concentrations. These two forms represent the major forms of mineral nitrogen present in the groundwater. Samples were also analyzed for total Kjeldahl nitrogen (TKN), which represents the sum of ammoniacal nitrogen and organic nitrogen. In anaerobic decomposition, such as occurs in lagoons, organic nitrogen is converted to ammoniacal nitrogen. In the presence of sufficient oxygen, ammoniacal nitrogen can be converted to nitrate nitrogen, which is the form that is most mobile in groundwater. Concentrations of either mineral form of nitrogen in natural groundwater are typically less than 1 mg/L. Most of the mineral nitrogen applied to agricultural fields, whether by manure, wastewater, or commercial fertilizers, appears in the nitrate form after a short period of time. Concentrations of 5 to 15 mg/L nitrate−N are common in the shallow groundwater under agricultural fields (Krider, 1986). Chloride (Cl^-) concentration is sometimes used as a seepage indicator (Ritter et al., 1984; Westerman et al., 1995; Ham, 2002) and was also evaluated. Since the chloride−based indications were very similar to the nitrogen−based indications, only the nitrogen−based results are presented here.

The U.S. Environmental Protection Agency’s drinking water standard for nitrate−N is 10 mg/L. There is no standard for ammoniacal nitrogen, but since it is readily converted to nitrate−N under favorable conditions, it was included with nitrate−N in these assessments.

Each site rating was based on the difference between the minimum upgradient concentration and the maximum downgradient concentration of mineral−N. This method gives the most severe rating because it does not average in samples that may have been taken above, below, or to the side of a seepage plume. Concentration differences were used to isolate the nitrogen contributed by lagoon systems from nitrogen contributed by upgradient sources, such as agricultural fields. The concentration ranges for the various classes for seepage evidence are given in table 2.

Results

The goal of randomly selecting 40 study sites from a large pool of candidates could not be met because the number of volunteered sites was too small. However, the 36 sites included in the study provide a reasonable representation of the types of conditions and performance of older lagoons in the state. The conditions and performance observed in this study were very similar to those found in earlier work (Huffman and Westerman, 1995; Westerman et al., 1995).

Shallow rock was encountered at two of the 36 sites, both of which were located in the Piedmont. Attempts to drill into the rock with the available equipment were unsuccessful. Since groundwater samples could not be collected, it was not possible to assess the seepage characteristics of those lagoons. Seepage assessments were therefore completed on only 34 sites.

Table 3 presents physical and operational characteristics of the sites, along with the results of the groundwater analyses. For each of the 34 sites, the means and ranges of mineral nitrogen concentrations are shown for both upgradient and downgradient locations. On 16 sites, the maximum upgradient nitrogen concentration exceeded 10 mg/L. One of those had a mixture of woods and fields upgradient. The remaining 15 had agricultural fields upgradient.

A statistical analysis of the results shown in table 3, using the maximum difference between upgradient and downgradient mineral nitrogen concentrations as the response variable, did not find any significant predictors among the variables Type of Operation, Size of Operation, Age of Operation, Lagoon Surface Area, or Average Water Table Depth.

The seepage classes at 38 m (125 ft) downgradient are summarized in table 4. Seven of 34 sites (21%) ranked as None or Weak. Eleven of 34 (32%) ranked as Moderate. Sixteen of 34 (47%) ranked as Strong or Very Strong.

County soil surveys provide descriptions of the upper 1.5 to 2.4 m (5 to 8 ft) of the soil profile. It would be useful if the appropriateness of a site for lagoon construction could be predicted on the basis of the soil type. To examine the relationship between soil texture and seepage classes, textures in the upper 1.5 m (5 ft) of the soil profile and seepage classes at 38 m (125 ft) downgradient were compared, as presented in table 5. As can be seen from the wide ranges of textures in the various classes, the textures in the upper 1.5 m of the soil profile do not provide a good indication of seepage containment.

Table 4. Summary of seepage classes at 38 m (125 ft).

Seepage Class	Number of Lagoons	Percent
None	5	15
Weak	2	6
Moderate	11	32
Strong	7	21
Very Strong	9	26
Total	34	100

Discussion

Sites ranked as either None or Weak were judged very unlikely to present any contamination problem. They contributed less than 10 mg/L mineral−N to the shallow groundwater. Sites ranked Moderate or higher could present problems if the shallow groundwater was extracted for use.

While sites ranked Moderate or higher would not meet EPA drinking water standards, they do not necessarily require corrective action. All of the lagoons in this survey were positioned on the landscape such that the seepage plumes moved toward areas where direct use of the shallow groundwater was very unlikely. Many are positioned immediately upgradient from woodlands or swamps that can assimilate modest nutrient loads. In some cases, there were streams nearby where the plumes would discharge with the natural flow of the shallow groundwater and be greatly diluted. Before requiring remedial action, consideration should be given to the affected area and whether adverse impacts actually exist. In a case where seepage losses are excessive and the plume has an actual adverse impact, corrective action should be taken. If a plume does not have an actual adverse impact at present, but the owner can foresee the possibility of future problems, corrective action would be prudent.

While inspection of tables 3 and 5 suggests a general trend toward higher mineral nitrogen concentrations on sites where sandy soils (s, sl, ls) were observed (1410, 1954, 4551, 4589, 6726, and 7225), there were lagoon systems on locations with sandy soils (fsl, sl) that showed little evidence of seepage (1116, 2757, and 3914). Conversely, some systems on locations with clayey soils (scl, cl, sc, c) had strong indications of seepage (3200, 3933, 4547, 4649, 5827, 7286, 7940, and 8971). The weakness of the relationship between seepage performance and textures in the upper profile can be explained in part by the fact that lagoons were often constructed in and with soil materials that were deeper than those that would be described in soil surveys. Clayey horizons that typically underlay sandy surface horizons in the Coastal Plain could have been used in lagoon construction. Soil textures may also vary dramatically, both vertically and horizontally, within the range of distances that are typical of lagoon dimensions.

On some sites, high concentrations were observed across much of the downgradient area, giving evidence of a broad plume. On other sites, high concentrations were found in only a few samples, suggesting a narrow plume that may have been the result of transport through sandy lenses that were not properly excavated and patched during lagoon construction.

Table 5. Soil textures observed in upper 1.5 m (5 ft) on sites in each seepage class.

Seepage Class	Soil Textures in Upper 1.5 m of the Soil Profile
None	Fine sandy loam, sandy clay loam, clay loam, silty clay loam, silty clay.
Weak	Sandy clay, clay loam.
Moderate	Sand, loamy sand, sandy loam, loam, clay loam, sandy clay, silty clay, clay.
Strong	Sandy loam, sandy clay loam, clay loam, sandy clay, clay.
Very strong	Sand, loamy sand, sandy loam, loam, sandy clay loam, clay loam, sandy clay.

Detailed reports for all sites, including site layout, sample locations, and analytical results, are in Huffman (1999).

CONCLUSIONS

Approximately one−fifth of pre−1993 swine waste lagoon systems in this survey contributed less than 10 mg/L mineral−N to the shallow groundwater at a distance of 38 m (125 ft) downgradient from the lagoons. Although four−fifths of the systems showed heavier loadings to the shallow groundwater, none of the systems in this survey were positioned in the landscape such that the seepage plumes represented an immediate hazard to groundwater users.

Lagoons constructed on sites with coarse−textured soils were expected to have higher seepage losses than those constructed on sites with fine−textured soils. Although the data suggest a very weak correlation between soil texture in the upper 1.5 m and seepage losses, soil textures in the upper 1.5 m of the profile were not good predictors of lagoon seepage performance.

A follow−up study was initiated to examine the variability of plume strength with distance from the source. It will be reported in a later article.

ACKNOWLEDGEMENTS

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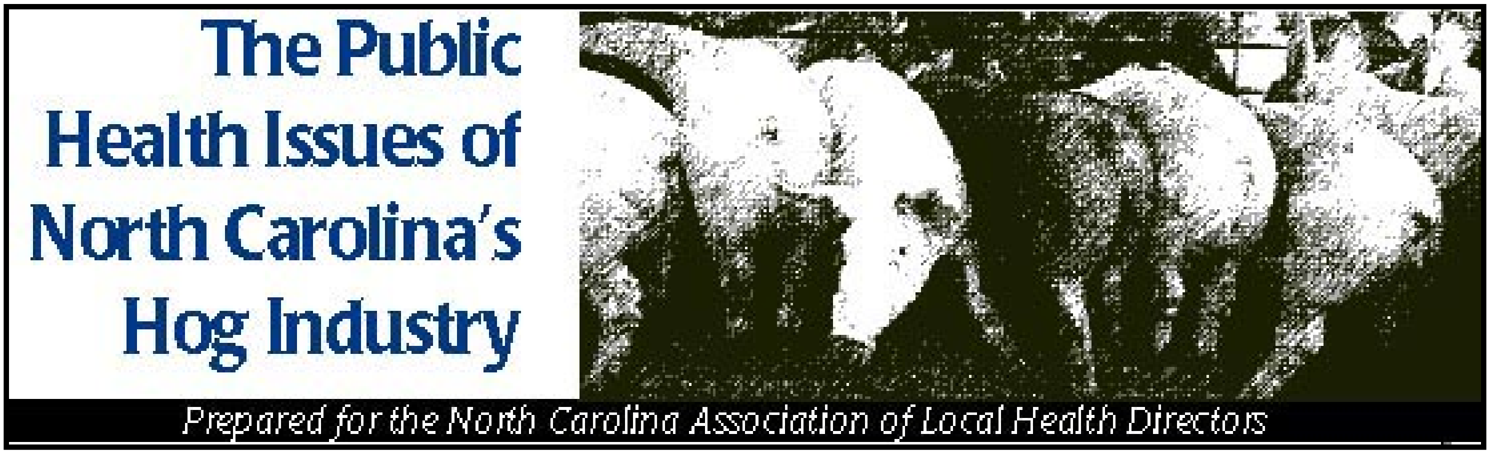
ATTACHMENT 10



Human Health Issues Associated with the Hog Industry

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Disclaimer

Contents

Introduction

The swine population in North Carolina increased from approximately 2.5 million in 1990 to 10 million in 1998 and went from being raised on small independent farms to large intensive livestock contract operations. More hog intensive livestock operations (ILOs) result in more hog waste concentrated in a specific area. The waste is first stored in a confinement or growing house and then flushed and pumped into a nearby lagoon. The wastewater is periodically sprayed onto fields. Odors from such operations emanate from the confinement houses, the lagoons, and the spraying fields. Generally, the odors coming from the confinement houses and some phases of the spraying operations are the most intense. The growth of the hog population is concentrated in the southeastern part of the state. Two counties, Duplin and Sampson, account for approximately four million hogs or forty percent of the state's hog population and rank as the number one and two hog producing counties in the United States. The sudden growth of swine intensive livestock operations (ILOs) resulted in increased community complaints particularly about the problems associated with hog waste. Citizens were concerned with the industry's impact on the health of nearby residents, the environment, and the overall quality of life in the community.

Neighbors of hog intensive livestock operations, especially those affected by the odors, have been especially vociferous at public meetings (Schiffman, 1998). Concerns about human health include exposure to odors; waste; resulting flies; poor air quality; and the contamination of drinking water supplies. Environmental concerns include groundwater and surface water contamination, air pollution, and the overloading of nutrients and heavy metals on soils where hog waste is applied. Citizens comment before boards of health that their health is negatively impacted and their quality of life is degraded. Dr. Hervy B. Kornegay, Sr., MD, who serves as Chair of the Duplin County Board of Health, reports, "Such a correlation is medically difficult to prove and document, but it is certainly perceived." (Kornegay, 1998) Citizen concerns in the counties with major swine livestock operations, such as Duplin and Sampson counties, and major processing plants, as in Bladen County, include increased truck traffic especially at night, dead animals sitting in the hot sun awaiting pick-up and then exploding when dumped in the hauling truck, and resulting animal waste on the road from alive and dead hogs. Community health issues associated with the hog industry embody concerns about the workforce, and pre-existing health problems they have, who are drawn to the area due to employment opportunities at the operations and the slaughterhouses. There is a high turnover rate of the workforce (Cooper, 1997). Many who work at such facilities are immigrant workers who suffer high rates of tuberculosis and HIV infection, especially among young Hispanics (Kornegay, 1998).

Researchers are challenged to determine what the impacts of the hog industry are, especially as it pertains to human health. Few studies have been conducted to determine the health effects for residents living near swine intensive livestock operations (ILOs). Most research has focused on the more direct consequences that workers face in the confinement houses and processing plants. Two recent preliminary studies have looked at the impact on the physical and mental health of nearby residents. These studies need to be conducted with larger study populations to validate their preliminary findings. Additionally, research is needed to see if some of the anecdotal findings as reported in the media, such as the effect of hog odors on asthmatics, can be substantiated through rigorous health studies. This paper summarizes the issues pertinent to human health that are raised by people and pigs living in proximity to each other. Section One includes research findings and information related to air issues. This section contains specific concerns for the health of workers employed in the hog industry; the effect on the physical and mental health of neighbors to hog ILOs; and some preliminary concerns for asthmatics living in proximity to such operations. Section Two covers the groundwater issues associated with hog ILOs and the related health concerns of consuming water contaminated with nitrates. Section Three includes some of the surface water issues that appear to be related to high nutrient waters. Section Four contains information about infectious disease concerns for workers and neighbors to such hog operations. In Section Five the findings are summarized and recommendations made.

Section One: Air Quality

Livestock air emissions and the resulting odors are not new to eastern North Carolina. The countryside is dotted with hog, turkey, and poultry operations. In 1998, the state ranked number 2 in the nation in hog production, number 1 in turkeys, and number 4 in poultry. The number 1 and 2 hog producing counties in the nation, Duplin and Sampson counties, are also number 1 and 2 in the nation in turkey production. The two counties have a combined hog population of 4 million and 22.4 million turkeys (NC Dept. of Agri., 1998). Although all livestock operations generate smell, odors are most intense from the large hog intensive livestock operations. These operations are wet-based waste management systems with water used to flush the waste periodically from the growing houses. Poultry and turkey are dry-based waste management systems. The dry litter helps absorb some of the volatile organic compounds (VOCs) which result in less VOCs transported to the atmosphere.

1.1 Physical Health Effects for Workers in Swine Intensive Livestock Operations

Thousands of gases and/or particles are emitted from swine intensive livestock operations. These gaseous emissions are of particular concern for workers at such facilities. The health issues are well-documented (Donham, et al., '84, '85, '90, '93). They include: scratchy throat, morning phlegm, cough, burning eyes, wheezing, shortness of breath and chronic bronchitis. Additionally, workers in swine confinement houses are found to experience increased organic dust syndrome (Rylander et al., 1990). The primary gases and particles emitted by such operations are: ammonia, carbon monoxide and dioxide, hydrogen sulfide, methane, dust, organic dust, and endotoxins (Donham et al., 1985). Bacteria are present in the hog dust that consists mostly of hog epithelium. Endotoxins are the primary lipid component of the outer membrane of Gram-negative bacteria (Donham, 1986) and are the cause of chronic respiratory symptoms of workers employed in swine containment facilities. The odors are generated by a mixture of fresh and decomposing feces, urine, feed, the animals themselves, and dead hog carcasses.

Worker health issues have become more pronounced as hog production has increased in scale and moved to large, closed confinement growing houses, a trend which started in the 1960s in the United States (Donham, 1993). Compared with the older conventional livestock houses, confinement buildings are more enclosed and tightly constructed, which results in the trapping and recirculation of air. The ILOs house a larger number of animals per house than in the older hog houses and typically confine from 800-1200 in each house in North Carolina. New operations constructed in the West, such as Utah, by North Carolina integrators can house from 3000-4000 hogs in one structure. The animals are grown in small tightly packed areas within the house, where they are in constant contact with each other. One of the reasons for housing the hogs in multiple structures is when there is a disease outbreak, it can hopefully be contained in the one structure and not spread to all the animals at the ILO.

Hogs are kept in such structures 24 hours a day for the duration of their life, approximately six months, until they are ready for shipment to a slaughterhouse. Such buildings are usually heated and ventilated and animal waste is disposed of through a mechanized system. Animals stand on cement floors with openings. The excrement of the animal is pushed through the openings after the animal lays on top of it. The waste sits in storage areas below the animals and is emptied periodically- sometimes only every few days. The air environments of such closed hog growing structures with up to 1200 animals located in a single house are more contaminated than in houses which are more open and where less animals are kept.

Poor air quality in the confinement structures is of concern to worker exposure, but likewise to animal health and productivity. Some growers may be more inclined to make structural changes to the houses knowing that their hogs will produce more efficiently. Swine grow faster and are therefor more productive in confinement structures with better quality air (Donham, 1990, 1993). Dr. Donham stated, "Advising this fact to a swine producer may be the most expedient way to create environmental improvement that would help the person as well as the animals in the building."(Donham, 1993)

The first study of worker health issues resulting from employment in hog confinement houses was published in 1977 by a team of Iowa researchers (Rylander et al., 1989). Workers employed in such hog growing structures are exposed to both dust and gases which can be harmful to human health (Donham, 1990). Of the off gases in such facilities, ammonia is most likely to exceed the Threshold Limit Value. Most gases emitted are in small amounts, well below Threshold Limit Values. The mixture of over 400 gaseous compounds (Schiffman, 1996) represents a potential health threat for the 400,000 workers employed in swine confinement buildings. The gases can on occasion reach acutely toxic levels during manure agitation, ventilation failure, or the malfunction of heating units.

The nature of the gases and their intensity depend on multiple factors: the age of the animals, time of year, management practices, ventilation, how and what animals are fed, how well the facility is managed, and most importantly how the animal waste is handled including frequency of wash (Donham, 1993). William Smith, the Health Director in Robeson County, North Carolina, described the importance of the waste options available to hog producers and the consequent impact on workers who spend hours a day in such structures but also the odor impact on nearby residents living downwind. Mr. Smith said a facility that does not frequently flush the waste out of the building is like a large family where everyone uses the same toilet that is only flushed once a day, or even every few days. He compared this to a family whose members flush the toilet after each use. An equivalent swine waste management system, developed by such companies as Awash, involves a continuous flush of hog feces and urine into a waste lagoon and thereby cuts down on odors and gases that might build up in the confinement houses.

When studying worker health response to employment in hog ILOs, it was found that cough and phlegm were the most common symptoms and were experienced by 12-55% of the workers (Donham, 1990) depending on the particular ILO facility studied. Chest tightness, coughing, nasal and eye symptoms can occur within 30 minutes of entering the confinement structures but typically require two or more hours of exposure. Symptoms usually disappear after one to two days, however, they can persist for long-term employees. Worker symptoms in response to employment in swine growing houses are more frequent and severe among smokers and by those who work in the larger swine operations. Health effects are also greater among those with pre-existing respiratory problems, such as hay fever and bronchitis, and among those with heart trouble or allergies.

A small percentage of the cases of workers experiencing symptoms are thought to be specific allergic-mediated illnesses, such as asthma, and the rest fall into chronic inflammatory reactions. In a Swedish study, hog farmers indicated reported symptoms that included throat irritation (28%), eye irritation (25%), and nose irritation (25%) (Rylander et al., 1990). Of particular concern is worker exposure in the confinement houses to hydrogen sulfide. At high concentrations the gas has toxic properties and can result in sudden collapse and associated respiratory paralysis and pulmonary edema and even death. Research conducted by an international team of scientists involved in clinical and epidemiological investigations of 2000 workers in five countries found symptoms of acute and chronic airway inflammation were common in addition to organic dust toxic syndrome.

In addition to immediate symptoms, workers can experience delayed reactions up to six-hours after working in the confinement buildings. This is true after exposure to especially dusty operations, involving the handling, moving or sorting of animals. Called organic dust toxic syndrome (ODTS), symptoms can include fever, malaise, muscle aches and pains, headache, cough, and tightness of chest. Some moderate pulmonary function changes were found. (Rylander et al., 1989)

Chronic health effects, such as bronchitis, is experienced by 25% of all swine confinement workers (Donham, 1993). Chronic bronchitis is found more than twice as frequently in workers in confinement buildings as those who work in conventional swine growing units. Symptoms related to chronic bronchitis include chronic cough, excess production of phlegm, and sometimes chronic wheezing.

Some workers, once removed from working in the confinement buildings, are still symptomatic two or more years later. However, most, especially nonsmokers, become asymptomatic after a few months. Long-term lung damage may be occurring as seen in lower flow rates. Pulmonary function decreases during the workday. The severity of chronic bronchitis increases with workers with a longer history in confinement units. The air in the confinement units does not cause asthma, however, exposure to the dust particles exacerbates asthmatic symptoms. Although the health effects for workers in swine ILOs has been well documented, the impact on neighbors has only been studied recently and so is less well understood.

1.2 Physical Health Effects for Neighbors to Hog ILOs

Neighbors of hog intensive operations have reacted to the noxious and unaesthetic air quality generated by the ammonia and hydrogen sulfide associated with the waste. Given the low concentrations of the gases, some believed that there was no significant risk to health and the odors represented more of a degradation of quality of life (Swinker, 1998) than a present health threat, even though those two can be hard to separate out. Swinker points out that it is difficult to distinguish health effects possibly caused by swine ILOs versus other human or agricultural activities. All possible contributing factors should be considered before describing and ascribing a causal relationship associated with swine ILOs. Further research is needed to determine the exact impact of the industry. A preliminary Iowa study described below (Thu et al., 1997), however, found that there are health effects associated with living adjacent to such operations.

A recently completed preliminary health study in Iowa (Thu et al., 1997) suggested that there were physical health effects for neighbors of hog intensive livestock operations. Ammonia, dust, and endotoxin were found in air samples taken downwind of a large-scale swine operation. The health study indicated that symptoms for near-by residents are similar to those found for workers in the confinement houses only less severe and less frequent (Thu et al., 1997). Residents in Iowa who lived within two miles of a 4000 sow production facility were interviewed in their homes. This facility was one of the largest sow operations in Iowa. Neighbors were asked to complete a survey concerning their physical and mental health status. They were also asked open-ended questions about their impressions of the impacts of the hog industry on the community and quality of life. Standard socio-demographic information was collected.

The particular operation was selected for study based on its size and the fact that some neighbors had expressed environmental and health concerns. Responses of residents living near the operation were then compared to a control group. The selection of a controversial operation for study may have biased the results. The study of neighbors of the largest sow operation in the state would give results that may represent the worse case scenario. Results indicate that residents living within two-miles of the operation reported significantly higher rates of four types of respiratory tract problems, which represent toxic or inflammatory effects. In 14 of 18 symptoms the study population reported higher frequencies than the control group. The symptoms more frequently reported are those that are extensively experienced by swine confinement workers. Thu and Donham's 1997 study found increased symptoms in four interconnected symptom clusters. The symptoms correlated well with those reported in the open-ended questions queried earlier in the interviews.

CLUSTER	SYMPTOMS
Cluster One	Sputum, Cough, Breath Shortness, Chest Tightness, Wheezing
Cluster Two	Nausea, Dizziness, Weakness, Fainting
Cluster Three	Headaches and Plugged Ears
Cluster Four	Runny Nose, Scratchy Throat, Burning Eyes

The first cluster shows interrelated symptoms that indicate inflammation of the bronchi and bronchioles, or chronic bronchitis and hyperactive airways. The particular kind of bronchitis experienced is often associated with environmental exposures. Figure 1 below shows that for most symptoms, the neighbors of the sow operation indicated more health problems than the control group. In the first cluster of inter-related symptoms, increased sputum was most pronounced in the study group.

The inter-related symptoms included in the second cluster are those that are commonly found in employees of swine operations. Residents living near the studied operation reported significantly higher rates for nausea, dizziness, weakness, and fainting, than the control group. Further study is needed to determine the effect of long-term exposure to lower than toxic levels of Endotoxins and hydrogen sulfide. The component gases associated with the air emissions from swine ILOs are individually well below federal standards. However, over 400 compounds have been found in the air, manure, and lagoons on hog farms. The compounds include acids, alcohols, aldehydes, amides, amines, aromatics, esters, ethers, inorganic gases, hydrocarbons, halogenated hydrocarbons, ketones,

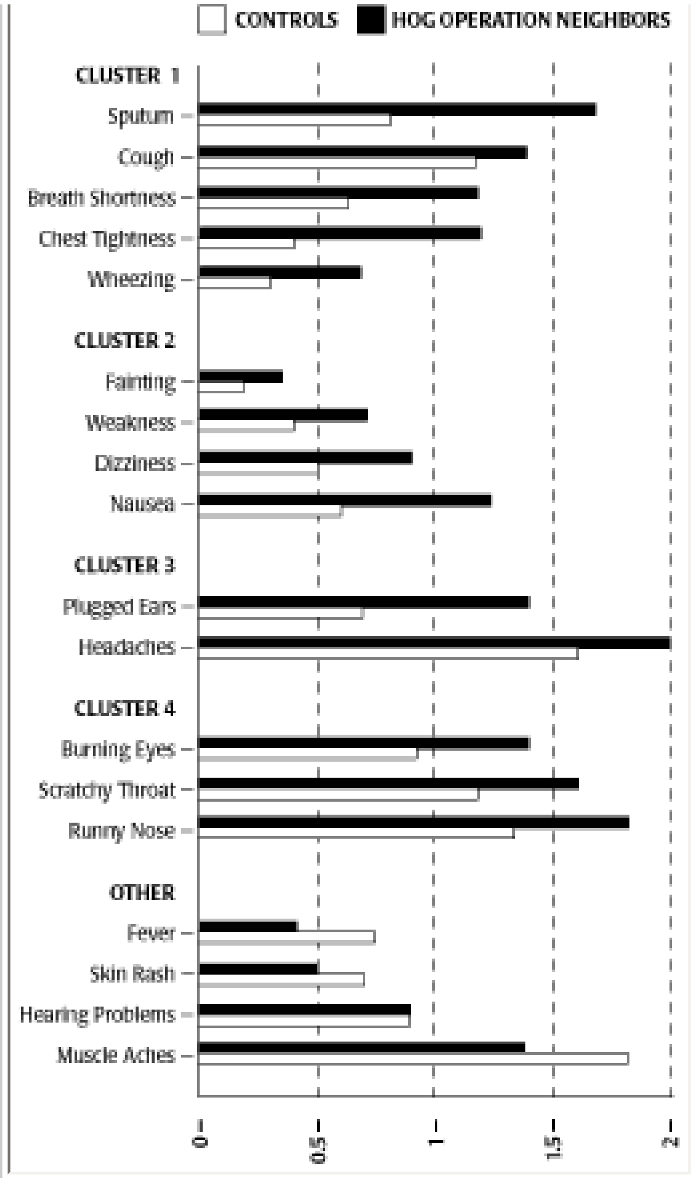


Fig.1. Frequency of physical symptoms experienced by rural residents*(comparison of mean scores, 0=Never, 4=Very Often).*

nitriles, nitrogen heterocycles, phenols, sulfides, mercaptans, and steroids (Schiffman, 1996). The compounds found both in North Carolina and elsewhere tended to be standard volatile organic compounds (VOCs) as associated previously with animal waste. Sixty percent of the compounds found in NC were not found elsewhere. More VOCs were found in NC than reported in other states. This can be explained by: the sensitivity of the gas chromatography testing methods used, temperature conditions in North Carolina, and chemical sprays mixed with the air and lagoon samples were included in the test.

Federal and state standards are only set for one gas at a time and standards have only been determined for a few hundred gases. One hundred and five air toxics are covered by state regulations and 189 by federal regulations, with an overlap between the two of approximately 85 air toxins. Researchers, such as Dr. Lori Todd and Dr. Susan Schiffman, suggest that a synergistic effect of the component parts working together may be of importance and further study is indicated.

The third symptom cluster, which includes headaches and plugged ears, is also often noted by workers in the swine industry. These symptoms are frequently associated with chronic sinusitis, and are found in approximately one-fourth of all active swine producers (Donham, 1993). The final cluster of symptoms noted, which showed the least significance, included symptoms such as: burning eyes, runny nose and scratchy throat. Among swine confinement workers, these symptoms are related to mucous membrane irritation caused by irritation from gases and particulates inside the swine confinement buildings. Nasal irritation has been shown to reduce respiratory volume (Warren, et al., 1994).

The Iowa study found residents to be no more or less depressed or anxious than the control group. These findings may have been influenced by the fact that someone was interviewing the respondents which can bias toward the under reporting of mental health effects.

One of the findings of the Iowa study was, regardless of whether respondents were experiencing any physical health symptoms, that the presence of the swine facility was creating social and political problems and divisions within the community. Respondents saw the facility as a violation of core rural values of what it means to be a "good neighbor". Such values include principles of egalitarian relationships, reciprocal exchanges of neighbors helping each other in times of need, mutual respect, and sharing of information (Thu et al., 1997). Neighbors felt that the facility threatened their sense of control over their land, homes, families, and quality of life. The concerns raised by neighbors encompassed more than physical and mental health issues but were intertwined with personal, environmental, economic, and social health matters. The hog industry is seen to impact both the quality of life and way of life for rural residents living near swine intensive livestock operations. These findings are consistent with citizen testimonials at county meetings in North Carolina. In Duplin County a citizen lamented that marshalls were needed at Board of Health meetings to assure there was no violence among attendees with opposing views.

1.3 Hog ILO Odors

Odors associated with swine ILOs emanate from the confinement houses where the animals are kept, the waste storage areas including the lagoons, and the land application area. The liquid wastewater in the lagoons is periodically sprayed onto fields via spraying systems that eject the wastewater as much as 100 yards. The odors result from a combination of fresh and decomposing feces, urine, and feed. Of these, the more offensive odors emanate from the decomposition of the feces. Emissions from livestock manure include volatile organic acids, alcohols, aldehydes, amines, fixed gases, carbonyls, esters, sulfides, disulfides, mercaptans, and nitrogen heterocycles.

The inhalation of volatile organic compounds (VOCs) causes smell sensations in humans. There are four primary ways in which these odors can affect human health:

- the VOCs can produce toxicological effects;
- the odorant compounds can cause irritations in the eye, nose, and throat;
- the VOCs can stimulate sensory nerves that can cause potentially harmful health effects; and
- the exposure to perceived unpleasant odors can stimulate negative cognitive and emotional responses based on previous experiences with such odors (Schiffman, 1998).

Levels of VOCs in breath have been measured and correlate with the individual's personal air. The body is subjected to some burden as a result of non-occupational exposure to VOCs (Raymer et al., 1991). Low concentrations of multiple VOCs can result in health consequences that a single low-level VOC would not cause (Schiffman, 1998). The volatile organic compounds, which cause odors, can be absorbed directly via gas exchange in the lungs. These VOCs can reach blood and adipose tissue. For hours afterwards, people can still detect the odor as it is expelled from their bodies.

1.3.1 North Carolina Studies of Mental Health Effect

The psychological impact on neighbors of swine intensive livestock operations living in odor concentrated areas was documented by Dr. Susan Schiffman, a medical psychologist with the Duke University Department of Psychiatry (Schiffman et al., 1995). In this preliminary study of North Carolina residents, 44 people living downwind and in the odor affected area of swine operations were compared with a control group of rural people not so affected. The study of mood was considered important not only to better understand the psychological impacts but additionally because negative mood can depress the body's immune response. Unpleasant odors can thus influence physical health. Brain structures broadly involved in smell can affect immune responses. Some studies suggest that sensory stimulation of the limbic forebrain, hypothalamus, and other odor projection brain areas can directly alter immune status (Schiffman et al., 1995).

In Schiffman's study a significant difference was found (at $p < .0001$). The experimental subjects that lived within the hog odor affected area expressed significantly more tension, depression, anger, less vigor, more fatigue, and more confusion.

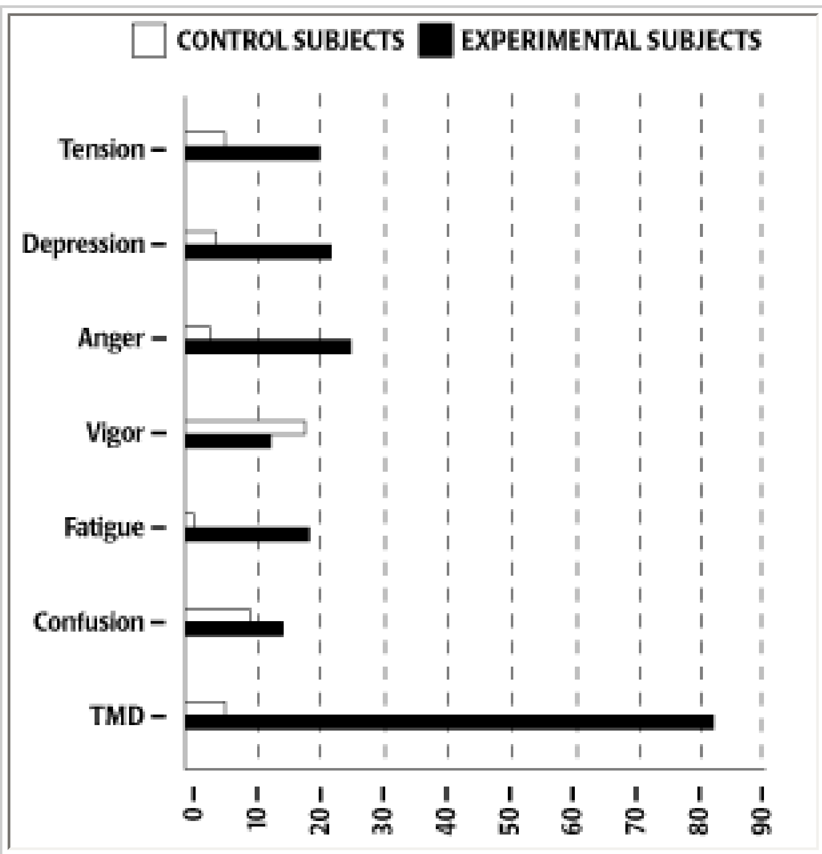


Fig. 2. Mean POMS scores of each factor and the total mood disturbance score (TMD) for experimental and control subjects.

Experimental males had significantly higher scores for anger and confusion than experimental females or control men and women. Scores for experimental females were significantly higher than control males and females.

Odors can have a lingering effect in the body. Volatile organic compounds (VOCs) are absorbed directly by the body into the bloodstream and fatty tissues by way of gas exchange in the lungs. Once absorbed these odorants are slowly released from the bloodstream via air expired from the body. When expired, the olfactory receptors are activated. Some of the compounds found in the waste plume, when absorbed in the body, can be transmitted to the brain through the nasal route (Monath et al., 1983). Some people, who are already experiencing other olfactory problems, may be more sensitive to the smell. Odors affect both the quality of outdoor and indoor air. Clothing, curtains, and building materials absorb the smell. Over time, the odor molecules are slowly released resulting in a prolonged effect from the odor even once the plume has passed over.

The effect of exposure to odors on respiratory responses has not been a subject of research interest. In part, this is due to the fact that animal studies indicate that changes in breathing patterns may require far higher chemical concentrations than are needed for detection. Recent work, however, shows that in human subjects, as opposed to animal studies, that the sensitivity of respiratory measures may not lag far behind that of perceptual measures of irritation (Warren et al., 1994).

People respond differently to the hog waste smell depending on their association with the odor. Some swine operators say the smell does not bother them and that they associate it with providing for their family. Many come from multi-generational hog farming families and see the operation and its smells as a way of life. At public hearings, owners of hog operations often say the odor smells like money to them.

But for the neighbors, often those not directly economically benefiting from the activity, they may associate the smell with unpleasant thoughts and lack of control. Strong livestock odors in one's home are considered inappropriate by most. One community activist tells the story of a birthday party for her child, when one of the children came to her and asked if she could please go home because the stink was so bad.

For others, the smell may result in environmental concerns, fear of loss of use and value of property, or an interference with the use of ones property. Some neighbors in Duplin County report being surrounded by hog confinement buildings that result in the odors from such operations being regularly present. Other neighbors report a frequency of one out of every three days that the odor affects their home with the early morning and evening as the time when the smell is usually strongest and most regular. Others may consider the smell to be a taboo odor and something that they shouldn't have to endure (Schiffman et al., 1995). As the public has learned more about the health consequences of breathing in the malodorous smell of second hand cigarette smoke, heightened consciousness can be applied to concerns about living in areas with regular exposure to the odors and air emissions from the hog operations.

Some people living in proximity to hog ILOs find conditions practicably unlivable. In a lawsuit, Parker v. Barefoot (No. COA97-713) against a Johnston County hog operation holding approximately 2,880 hogs in four hog houses and one open pit lagoon, 27 neighbors reported that fumes from the hog ILO were so noxious that at times it burned their eyes and noses, making it difficult for them to see and breathe. The stench from the lagoon was described as 'unbearable'. Plaintiffs sought injunctive and monetary relief alleging that the swine facility constituted a nuisance. In this case, the NC Court of Appeals on July 7, 1998 overturned the lower court's decision and granted plaintiffs' right to a new trial. In the earlier trial, the plaintiffs had submitted a written request for instructions to the jury that the law did not recognize as a defense to a claim of nuisance that defendants used the best technical knowledge or "state-of-the-art" technology available at the time to avoid or alleviate the nuisance. The trial court denied this request. Judge Howard E. Manning in Johnston County Superior Court entered the judgment in favor of the defendants on August 30, 1997. The appellate decision now focuses juries to the validity of neighbor's complaints of a nuisance when an operation is found to be so offensive as to have an intolerable impact on its neighbors. The offense can be great enough to affect neighbors ability to enjoy their homes. In a July 11, 1998 editorial of the *News & Observer*, the paper calls attention to the need for the state and local governments to set odor standards to protect the health and well being of neighbors of such facilities. In the fall of 1998, the NC Environmental Management Commission began work to regulate odors from hog operations.

In an odor study of NC residents (Schiffman, 1998) who did not live near agricultural operations, 68% of the subjects who were classified as 'More Sensitive' and 62% who were considered 'Less Sensitive' reported that exposure to animal odors would make them ill if exposed for 30 minutes or more.

Multiple factors can play a role in the changed mood of people exposed to odors from nearby swine operations. These factors include: the unpleasantness of the sensory quality of the odor; the intermittent nature of the stimulus; learned aversions to the odor; potential neural stimulation of immune responses via direct neural connections between odor censers in the brain and lymphoid tissue; direct physical effects from molecules in the plume including nasal and respiratory irritation; possible chemosensory disorders; and unpleasant thoughts associated with the odor (Schiffman et al., 1995).

The intermittent character of hog odors affecting neighboring properties is also an unpleasant factor. Although no one would want such strong odors to be constant, the fact that they come and go draws attention to the offense (Aitken et al., 1991). The odorant molecules associated with hog farms can cause nasal and respiratory irritation. The irritation of the nasal area can result in an elevation of adrenaline, which stimulates feelings of anger and tension.

1.3.2 Differences Between Psychological Findings in the North Carolina and Iowa Study

The Schiffman study results in North Carolina of the psychological effect on neighbors of swine ILOs differed from the findings in the Thu/Donham study in Iowa. The Iowa study found little difference in depressive and anxiety symptoms between their respondents and the control group. The authors concluded that the study population was neither suffering from anxiety related nor depressive psychological symptoms. In the Iowa study, researchers interviewed all subjects in person, which may have biased people from admitting to psychological problems. In the North Carolina study, subjects completed a written survey on their own. Most importantly, they were asked to complete the survey during an odor incident. This may have increased their attention to the event and their consequent response.

The Iowa and North Carolina studies took different approaches in selecting their study populations. Dr. Schiffman studied residents that lived downwind of swine intensive livestock operations and in the odor affected area. The Iowa study surveyed people who lived within a two-mile radius of a particular 4000 sow operation and did not limit themselves to subjects living in the odor-affected areas. The Iowa operation selected was one that had a history of community complaints. Schiffman chose people from multiple counties from across North Carolina. People living downwind and in the odor affected area of swine intensive operations experience the impacts of such operations differently from those that might live near-by but not necessarily affected by the odor (Schiffman et al., 1995).

1.4 Odors and the Effect on Asthmatics- A Potential Health Concern that Needs Researching

No research to date has focused specifically on the effect the exposure to air emissions from hog waste has on asthmatics. However, some reported anecdotal stories reveal that there is potentially a relationship and that further study is needed. People report that odor events trigger asthmatic episodes and cause severe reactions. Research does show that many odors, both those generally considered pleasant and offensive, can have an impact on asthmatics. Sir John Floyer first noted the relationship between exposure to certain odors and a response in asthmatics in 1698 in his classic, "A Treatise of the Asthma." Odors, both those generally associated positively, as in perfume, and negatively, as in waste or insecticides, have been found to cause asthmatic responses in some patients (Shim et al., 1986). Of 60 patients studied in Shim and Williams' research, 57 claimed a respiratory reaction to at least one odor. Of the particular odors that most frequently worsened asthma, the most common offender was insecticide (85%), then household cleaning agents (78%), cigarette smoke (75%), fresh paint smell (73%) and perfume and cologne (72%). The asthmatic response to odor is often severe. Of the 60 patients studied, 23 claimed that they had to make an emergency room visit, and 9 required hospitalization. Symptoms include shortness of breath, tightness of chest, wheezing, and cough.

Many clinicians seem to be unaware of the association between exposure to odors and an asthmatic response, however, most patients are aware of such a potential and have found ways to avoid the offending odors in their daily life (Shim et al., 1986). There has been an increase in the prevalence of asthma, especially among children under 18 years of age (Koren, 1995). Asthmatics are more sensitive to the effects from exposure to ozone, sulfur dioxide, particulate matter, and nitrogen dioxide. The increase in asthma rates is in part associated with air pollutants, including volatile organic compounds. One of the compounds found in hog odors is formaldehyde, HCHO, which can cause asthma-like symptoms. Further study needs to be conducted to determine if chronic exposures to both the gases and particulate matter can result in potential toxicity (Schiffman, 1998).

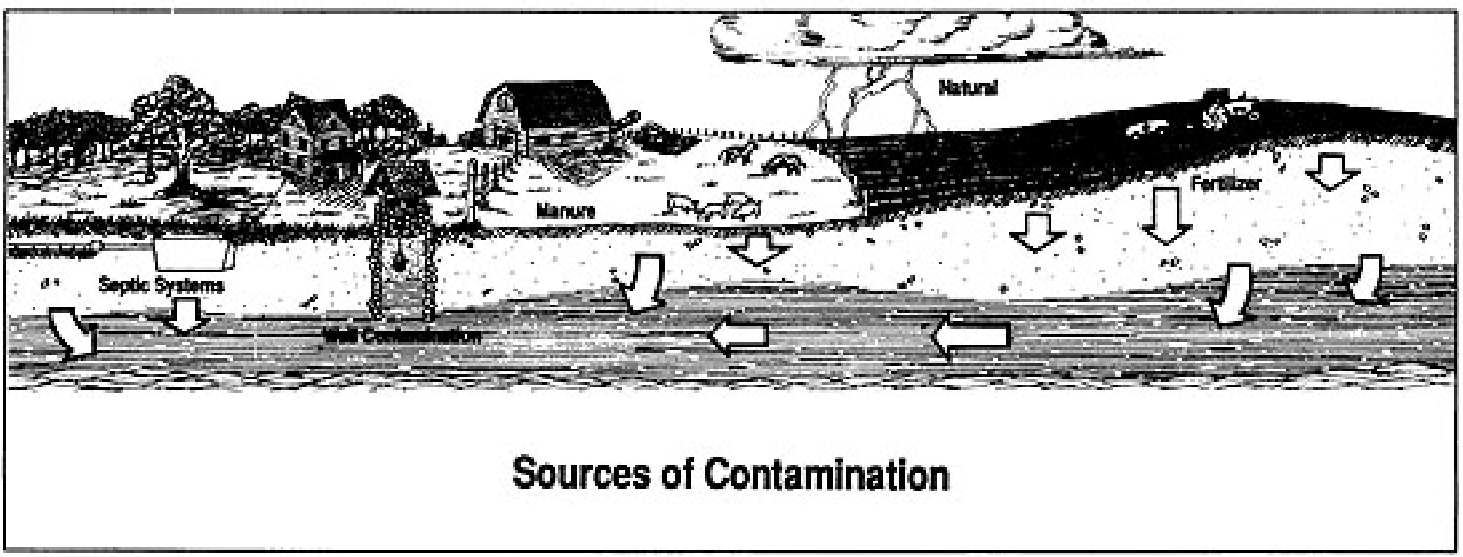
The response of asthmatics to certain odors is most commonly found in sensitive asthmatic patients and in those who are having difficulty controlling their disease. The asthmatic response to particular odors may result from reactions occurring in the bronchial mucosa. This response can be caused either by a direct irritant effect, from an immunologic reaction with secondary chemical mediator release or from a local neural reflex (Shim et al., 1986). Further research is needed to determine if the emissions from hog ILOs are having any effect on the health of nearby asthmatic populations.

Section Two: Groundwater Contamination and Related Health Issues

Hog waste when stored in lagoons and later applied to fields can result in nutrients and pathogens ending up in nearby groundwater. Although nutrients, such as nitrogen and phosphorus, are essential to life, in excess they can send aquatic ecosystems into disorder, as seen in the Neuse River in the New Bern area. Too much nitrogen is also of concern regarding human health. Once ingested through nitrogen-contaminated water, in the human gut nitrate is reduced to nitrite and is absorbed into the blood (Swinker, 1998). Nitrite compromises the oxygen-carrying capacity of the blood. Likewise, pathogens found in the gut of hogs can be of concern if they come into human contact. North Carolina has focused extensive resources on testing for nitrogen in groundwater but has not concurrently investigated for possible coliform contamination.

2.1 Nitrates and Drinking Water - A Concern for Human Health

Nitrates occur naturally in drinking water and in common vegetables, such as beets, celery, and lettuce. However, if the nitrates are present above a certain amount (10 ppm) they can be harmful to people who drink the water. Especially at risk are newborn infants. Nitrates can get into the drinking water from the overuse of chemical fertilizers and improper disposal of human and animal wastes. The nitrogen in the waste is converted to nitrates in the soil. The nitrates are highly soluble and move quickly and easily through soil, depending on soil type, and into groundwater and surface water. Once in the water, the nitrates can accumulate. In 1989 the NC Cooperative Extension Service conducted a one-year sampling program of rural drinking water supplies for nitrate-nitrogen, chloride, electrical conductivity and pH. Some of the samples were tested for pesticides. Of the over 9026 domestic wells tested, 3.2% exceeded the drinking water standard of 10 mg/L for nitrogen. Most of the contaminated wells were found to be poorly constructed and badly sited near areas of nutrient and pesticide application (Jennings et al., 1991).

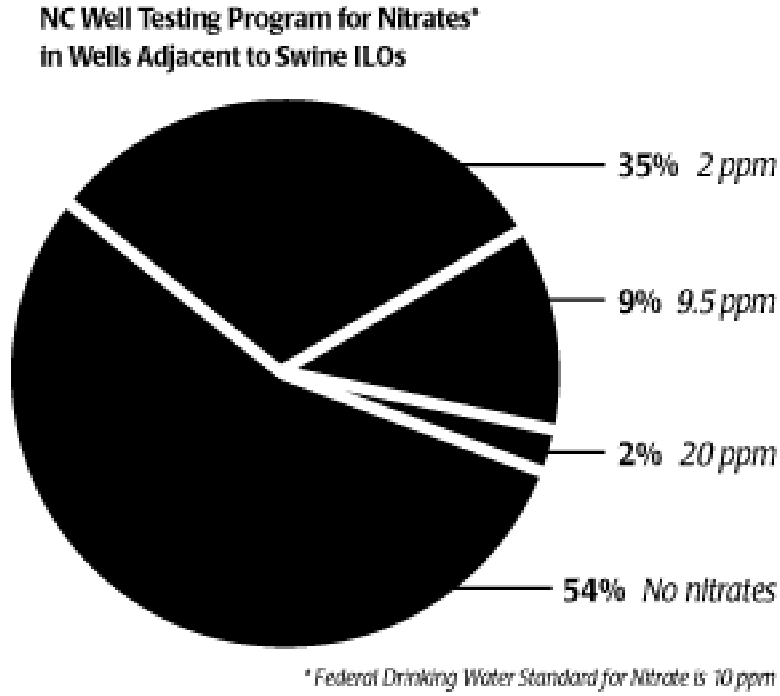


Graphic Credit: Jody Kubitz, Institute of Water Research, Michigan State University

2.2 NC Well Testing Program

In 1996 NC Governor James Hunt called for a free drinking-water well testing program for nitrates in the wells of people living adjacent to hog operations. The testing was begun after several drinking water wells in Robeson County located near swine operations were found to have nitrate levels which greatly exceeded the federal drinking water standard. Dr. Kenneth Rudo, a toxicologist with the Occupational & Environmental Epidemiology Section, headed the program. By December of 1996, Dr. Rudo had tested

948 wells in 50 counties (Rudo, 1996). Approximately 9.4% of the wells tested showed levels of nitrate that were at or exceeded the nitrate drinking water standard of 10 ppm. In a 1998 memo to the new state Health Director, Dr. Dennis McBride, Dr. Rudo stated that continued testing of wells showed that approximately 10% of the wells tested had levels of nitrate that exceeded the federal standard (Rudo, 1998). Three wells in Robeson County had nitrate levels in the 70-100 ppm range. Once tested, if the water was found to exceed the drinking water standard, residents were notified of such. Testing for other possible contaminants, such as viruses, bacteria, or pesticides was not performed, nor were health studies conducted on the families who had been drinking the nitrate contaminated well water.



The following counties in North Carolina showed the highest levels of nitrates in the drinking water wells tested by the state.

Percent of wells with nitrate* levels > 2 ppm and >9.5 ppm respectively

COUNTY	TOTAL NUMBER OF SAMPLES TAKEN
Duplin (24%, 9.9%)	121
Edgecombe (34.7%, 0%)	75
Johnston (57.1%, 4.1%)	49
Robeson (39.5%, 10%)	210
Sampson (48.3%, 22.5%)	209

Potential sources of nitrate include hog waste, human waste, and fertilizers. Further study by the state is continuing to better determine the exact source of nitrate contamination. There is concern that with the explosive growth in swine operations in the last ten years that some of the problems with well-water contamination may still lie ahead, especially in Duplin and Sampson counties where the operations are concentrated. Some Duplin County residents have expressed concern about the amount of hog waste sprayed onto crops in the region's sandy soils. Other issues include that the depth of the lagoons is only slightly higher than the water table in the area, and the status

of abandoned lagoons. These are of particular concern in rural areas where considerable poor people live and often depend on shallow wells for their drinking water.

2.3 Methemoglobin An important health effect resulting from the consumption of high levels of nitrates is methemoglobinemia, more commonly known as Blue Baby Syndrome. The nitrate as it enters the body converts to nitrite which affects the hemoglobin that carries oxygen throughout the body. The hemoglobin converts to methemoglobin that does not transport oxygen as well. This results in less oxygen getting to vital tissues, and of special concern is the brain. Methemoglobinemia can produce cyanosis, dyspnea, lethargy, and coma (Swinker, 1998). If the problem is severe and not corrected, brain damage and even death can result. The syndrome was first identified in the mid-1940s and resulted in the standard for nitrate concentrations in drinking water of 10 mg/L.

Children in the first six months of life are particularly vulnerable to high nitrates because fetal hemoglobin is more reactive than adult hemoglobin. Also the flora found in the stomach of infants facilitates conversion of nitrate to nitrite. Others who are vulnerable include pregnant women, the elderly, and adults with immune deficiencies.

When nitrates are found in groundwater and the source of the contamination is animal waste or effluent from a septic tank, the well water should be additionally tested for other contaminates of concern, such as bacteria, viruses, and protozoa. If the source of nitrates is fertilizer, then the water should be tested for the presence of pesticides. When drinking water is contaminated with nitrates at levels above 10 ppm, other sources of drinking water, such as bottled water, need to be consumed. Sometimes the drilling of a deeper well into a non-contaminated water source may provide the best solution if financing is possible. However, if the nitrate contamination is widespread in a region, a public water system should be explored. Such an option is of course expensive.

2.4 Exposure to High Nitrates and Possible Link to Reproductive Health Difficulties

Research suggests that there may be a link between the consumption of nitrate-contaminated water and human health effects including increased risk of delivery of an infant with a central nervous system (CNS) malformation (Arbuckle et al., 1988) and reproductive problems (MMWR, 1996). Three women in Indiana who lived in close proximity to each other had miscarriages a total of six times within two years. The women were all drinking well water containing high levels of nitrate, which exceeded the federal standard of 10 ppm. The women lived near a hog operation that appeared to be the source of the nitrate contamination. Other women living nearby had given birth without any problems. The women having trouble with the miscarriages all lived the closest to the hog operation. Once the women switched to bottled water they were able to have fullterm healthy pregnancies. Although it was not proven that the source of the nitrates was the hog operation, the point is that the consumption of water contaminated with nitrates appeared to have affected pregnancy outcomes. The CDC recommends that anyone drinking water from a private

well should have the water tested periodically (Meyer, 1996). Additionally, proper well construction with an emphasis on digging the wells deeper than 50 feet can alleviate many problems.

2.5 Waterborne Diseases and Well Water Consumption

Well water consumption is associated with several waterborne diseases. In the U.S. between 1991-92, 76% of the outbreaks of waterborne disease were connected to the drinking of well water (Moore et al., 1993) The total number of outbreaks has remained steady, however, new causative agents are being reported, which includes E coli 0157:H7 and Cryptosporidium (Swinker, 1998). Waterborne Cryptosporidiosis is of greatest concern when it enters municipal water treatment plants causing an outbreak of disease in the population that consumes the water. Such an outbreak occurred in Milwaukee, Wisconsin, when during the spring of 1993, over 403,000 were sickened with prolonged diarrheal illness, of which 4400 required hospitalization (CDC, 1994). Proper management of water supplies and surveillance are vital tools to the prevention of and early detection of such waterborne infectious diseases. Appropriate disposal of animal waste, including hog waste, is important in the control of Cryptosporidiosis and other infectious diseases.

2.5 Cancer and the Need for More Research

Several studies have shown an apparent increase in leukemia and brain tumors in children whose parents work in farming or in occupations where they are exposed to pesticides. A study was conducted of 323,292 children in Norway. The children were followed for a ten-year period. For children aged 0-14, there was a higher incidence of brain tumors associated with hog farming (Kristensen et al., 1996). The study, however, did not include individual exposure data and can be better used for hypothesis-generation than for drawing conclusions about risk of specific childhood cancers (Ross, 1996).

2.6 Community Concerns about Groundwater Contamination from Hog Operations- Difference Between Community Perception and Policies

Rural residents have expressed concern about the impact of hog intensive livestock operations and the impact on drinking water (Holtkamp et al., 1994). In Iowa, the number one hog producing state in the nation, 440 people completed a survey sent to residents of a two-county area in the southern part of the state. Respondents were asked to comment on their level of concern about the potential location of a new 1000 sow confinement near their home and the possible impact on groundwater.

Over 80% said they were 'somewhat' to 'seriously concerned' about the potential for nitrate contaminating their drinking water supplies. They indicated that even at a distance of five miles from a residence that they would be very concerned about the risk of contamination of their drinking water supplies. If such a facility was sited a half-mile from a residence, 77% of the

respondees said they would be 'seriously concerned' about the potential for nitrate contamination. The survey results indicate that responses were sensitive to educational level. The lower the education attainment of the respondee, the lower the concern. Overall, rural residents surveyed did indicate a strong interest in environmental issues and 58% thought it was a 'top' or 'high' national priority. Sixty percent thought it was a 'top' or 'high' priority local political issue.

Section Three: Surface Water Health Issues - Associated Issues to the Hog Industry

Increased attention and concern over the hog industry's environmental and health impacts followed the dramatic rupture of the Onslow County hog waste lagoon during the summer of 1995. This resulted in the spillage of approximately 25 million gallons of hog waste onto neighboring fields, roads, and streams, and eventually into the New River. In addition to such dramatic crisis events, waste from hog ILOs gets into surface water when it is unintentionally or illegally discharged into ditches, wetlands, and directly into streams.

Public awareness of nitrogen enrichment and the resulting eutrophication of the Lower Neuse, Cape Fear, and Tar-Pamlico river basins and the Albemarle and Pamlico sounds has focused attention on the hog industry as a major source of such nutrients (Aneja et al., 1998). The nitrogen either directly enters such surface waters from other water sources, from the runoff from ILOs, and/or through the deposition of airborne nitrogen in the form of ammonia. Ammonia emissions in eastern North Carolina are almost solely associated with intensive livestock operations, especially from hog ILOs. Testing for airborne ammonia starting in 1980 in Sampson County has shown a dramatic increase, almost a four-fold increase since 1990, in the annual mean ammonium ion concentration in rainfall (Aneja et al., 1998). Such an increase correlates with the dramatic increase in the hog population in Sampson County during that same time period.

Other sources of increased nutrients include wastewater treatment plants resulting from population growth in the Piedmont, and the residential and commercial application of fertilizers throughout the river basins. Even elementary, middle, and high schools contribute to the problem. Although the state can clearly show that there is a problem with too much nitrogen, it is less clear as to what the different sources of nitrogen are and how much they contribute. The state has called for a 30 percent reduction (Brown, 1998) in the nitrogen load of 8.7 million pounds of nitrogen that reaches New Bern, NC. The reduction plan calls for the first mandatory comprehensive environmental requirements of agricultural operations. These cumulative water contamination problems resulting in high nutrient waters, although not clearly understood, appear to be associated with the preponderance of the dinoflaggelate, Pfiesteria. The relationship between water contamination, especially nutrients, and the presence of the micro-organism continues to be studied. This micro-organism over the years has been associated with some of the major fish kills, especially in the Neuse.

There is concern about related human illness caused by exposure to Pfiesteria. Some people who

spend considerable time in the water experience skin lesions similar to those found on the fish caught in some fish kill areas. Local physicians in the area of New Bern, North Carolina have expressed concerns about the negative contribution of the hog industry on the health of their patients.

3.1 Physician Petition in Support of Hog Moratorium in Craven County

In February of 1997 seventy-five physicians of the medical staff of Craven Regional Medical Center signed a petitioning letter endorsing a one-year countywide moratorium on new and expanding hog operations for Craven County, located in eastern North Carolina. The petition was submitted at a public hearing of the Craven County Board of Commissioners. The petitioning letter read:

"We, the physicians of Craven, Jones and Pamlico counties, by our signatures on this letter, express our deepest concerns for the environment and citizens of our counties. We understand that the issues of industrial and agricultural waste disposal are complex and highly technical, while at the same time, political and intensely emotional.

Our role as physicians requires that we be vocal advocates for our patients' health and well being. We believe that the issues of agricultural and industrial waste disposal pose a health risk to our patient population. We therefore petition the Board to grant the moratorium, which will allow these areas of concern to be addressed more fully."

At the Craven County public hearing to discuss a county moratorium on new and expanded hog operations, Dr. Chris Delaney spoke in favor of such and presented slides of a man's legs with crusty sores. The sores had allegedly appeared after the man submerged his legs in the nearby Neuse River (Clabby, 1997). An increasing number of people who have had contact with the Neuse are experiencing similar skin lesions. Especially at risk are people whose livelihoods require that they spend considerable time on or in the water, such as professional underwater divers and commercial fishermen.

Dr. Peter Rowlett from Craven County in NC raised an additional issue during the public hearing. He stated that he would like to see North Carolina state health officials be more proactive (Clabby, 1997) in responding to some of the problems physicians are detecting in his area of the state. He stated that doctors routinely treat people with lesions with antibiotics with no effort made to establish the causative agent. He would like the state to establish a system where such lesions were cultured and sent to a state lab. Such a lab could assure identification of the lesion's origin and assist the state in its understanding of the cause and frequency of such.

3.2 Pfiesteria - A Potential Marker of Polluted Waters

Some medical reports indicate that humans who spend a lot of time on or in the water, such as

commercial fishermen and underwater divers, are experiencing similar problems, including but not limited to open sores on their skin. Researchers at North Carolina State University studying the dinoflagellate, Pfiesteria, experienced both short term and long-term memory problems and balance difficulties after a heavy dosage exposure to the associated toxins. Since 1991, this organism has been found in the vicinity of more than 100 fish kills in the Neuse, New and Pamlico rivers (Clabby, 1997). What stimulates the dinoflaggelates' growth and activity is still not understood (Clabby, 1997). For five years, the controversy surrounding research and scientists' and policymakers' understanding of Pfiesteria has raked the state (Barker, 1997).

Dr. JoAnn Burkholder, a North Carolina State University scientist, and a colleague in her lab, experienced neurological disorders resulting from exposure to Pfiesteria. The exposure produced short term and long-term memory problems and disorientation. Duke University researchers found that the dinoflagellate, Pfiesteria, caused serious learning impairment in rats (Levin et al., 1997). The learning deficits seen in this study may provide a partial model for the cognitive problems seen in the NCSU laboratory personnel who were accidentally exposed to Pfiesteria. However, the study does not necessarily translate to an understanding of risks involving human exposure in the field.

In August of 1997, the state of Maryland responded quickly to a major fish kill on the lower Pocomoke River, which left thousands of fish dead or covered with bleeding sores (Environmental News Service, 1997). A special medical team was brought in to respond to the reported health effects. People were reporting burning skin sensations following contact with the water, respiratory irritation, and the onset of concentration difficulties. The Maryland Department of Health and Mental Hygiene announced that a toxin, similar to Pfiesteria, might be the cause of the symptoms. Maryland Governor Parris Glendening felt there was enough information to act and felt the public should be informed of the connection to human health concerns. The response of Maryland officials brought this issue to attention of the nation. As a result, increased federal money for research is now available.

Further work investigating the connection between exposure to Pfiesteria and human health effects was reported by Maryland researchers (Grattan et al., 1998). The first complaints indicating there might be a problem for commercial fishermen began in the spring of 1997 when some reported to the county health department experiencing fatigue, headache, respiratory irritation, diarrhea, weight loss, skin irritation and rashes, and memory difficulties. The Maryland Department of Health and Hygiene called for further assessment of those exposed.

Twenty-four people who had been exposed to the Pfiesteria toxin were followed over a several month period. Exposure history and symptoms were recorded; in addition, a complete medical and laboratory assessment was conducted. A neuropsychological screening battery was performed and test results were compared to the performance of a control group. Results indicated that people with high exposure were significantly more likely to complain of symptoms such as: new or increased forgetfulness, headache and skin lesions or a burning sensation of the skin upon contact with water. Additional tests showed the exposed group was having difficulty with learning and

higher cognitive functions. The higher the exposure the greater the difficulty. Eight- seven percent of the high exposure group experienced learning and memory problems (Roper, 1998). Approximately 3-6 months after the cessation of the exposure, test scores returned to within a normal range. However, those most severely exposed continued to have some deficient performance, although overall improvement was noted. When studied again at 6 months post-exposure, they reported improvement in memory function and had normal performances on all cognitive measures.

Further research conducted by Dr. Kenneth Hudnell, an EPA neurotoxicologist, found that contact with Pfiesteria contaminated waters might adversely affect a person's ability to distinguish visual patterns. In a preliminary study, Dr. Hudnell found that one's ability to detect visual patterns was reduced by about 30 percent (Brown, 1998). Such a deficit in contrast sensitivity may cause people to perform tasks more slowly and may have the effect of increasing the risk of accidents. This was the first study to show that exposure to Pfiesteria contaminated waters may affect vision. Since none of the subjects had been exposed recently, this study is the first suggestion that persistent health effects are possible. In a North Carolina study of people potentially exposed to Pfiesteria water, medical evaluations did not find severe, chronic or widespread effects (Roper, 1998). Further studies are underway to address the limitations of the NC study.

The NC Department of the Environment and Natural Resources has recently responded to the environmental and human health concerns about Pfiesteria by:

- **creating in the Spring of 1998, a Harmful Algal Blooms program to monitor the potential health effects of Pfiesteria;**
- **setting-up a toll-free hotline for citizens to report potential problems and to receive information;**
- **posting warning signs on the Lower Neuse to advise people to avoid dead, dying and sick fish;**
- **setting a 30% nitrogen reduction plan for the Neuse;**
- **creating rapid response teams for the Neuse and Tar-Pamlico rivers to investigate fish kills and gather water quality information;**
- **using $365,000 in federal funds for further monitoring efforts;**
- **seeking $221 million to assist farmers in the control of nutrient run-off; and**
- **working with CDC and researchers from Maryland and Virginia to track 100 individuals exposed to Pfiesteria (Brown, 1998).**

3.3 Vibrio Vulnificus

Another related issue to the hog industry and the increase of nutrients in surface waters is the appearance of the marine vibrios, Vibrio Vulnificus. Researchers are studying the correlation between the deadly organism and high nutrient waters polluted in part by livestock. The CDC considers vibrios an emerging disease and NC health officials plan to track reports of the illness.

Section Four: Infectious Disease

Within the intestinal tract of swine, many bacteria, viruses, and protozoa live, some of which can be pathogenic to humans (Swinker, 1998). Those at most risk from exposure are pork production employees, especially those that have contact with the animals, carcasses, parts or by-products associated with the killing floor of the swine processing/slaughter plants (Fowler, et al., 1998). As the waste from hogs is spread over the soil and infiltrates some water supplies, there is concern about the exposure of humans to possible infectious diseases.

Coliform bacteria, when found in drinking water, indicates microbiologic activity and contamination by fecal material. The presence of E. coli is usually caused by fecal matter from warm-blooded animals. In Indiana, there was an outbreak of sickness due to exposure to E. coli bacteria (Wire Report, 1998). Twenty-one people became ill in a two-day period. E-coli causes diarrhea, nausea, fever, and dehydration. All of the Indiana cases were adults. Added to this concern of exposure to coliform bacteria is the increase in potency of some of the viruses and bacteria found in animals, including hogs. Some of these bacteria are increasingly resistant to common antibiotics.

Enteroviruses are carried by swine and predominantly spread from hogs to humans through direct contact with the animals. Enteroviruses aerosolized by agitation tanks in municipal wastewater treatment plants can pose a health risk to workers at the plants (Swinker, 1998). As North Carolina looks to phase out anerobic lagoons and possibly replace them with aerobic lagoons, exposure to enteroviruses will need to be watched. Aerobic lagoons entail the agitation of the wastewater to introduce oxygen to assist in the breakdown of organic matter. Workers will need protection from possible exposure to aerosolized Enteroviruses resulting from this process.

The intestinal tract of hogs is an important reservoir of salmonella. A study of over 2200 North Carolina swine found that 25% harbored salmonella species in their feces (Davies, et al., 1997). Several of the species found are capable of infecting humans. From 1989 to 1995 the hog population increased in North Carolina from 2.7 to 7 million. During this time, the annual incidence of reported salmonella infections was constant at a mean of 1173 cases annually. The 11 leading hog-producing counties contribute 10-12% of the reported salmonella cases, with the majority of the state's reported cases coming from the urban, more densely populated areas. At

present, modern production methods, although still fairly new in the state, do not seem to increase nor reduce the prevalence of enteric organisms in the hog population (Fowler et al., 1998). The salmonella rate should continue to be studied as the hog waste is continuously applied to soils.

Although the rate of salmonella infections has remained steady, the risk of human infection remains a concern. Future efforts to ensure the safety of pork should include attention to the control of salmonella. Due to the salmonella reservoir of infection present in hogs and the fact that modern intensive livestock production practices result in large pools of contaminated water and soil, this has raised concerns about the safety of the water supplies of nearby neighbors. The soil and water in the drainage areas should be checked for infectious agents around hog intensive livestock operations. Additionally the state should follow closely hog-associated infections. This can be facilitated by adding such infections to the list of reportable conditions (Fowler et al., 1998).

4.1 Worker Exposure to Infectious Agents

Similar to the concerns about air pollutants, workers are more at risk than the general population concerning exposure to infectious agents present in hogs. In addition to the exposure to hazardous dusts and gases, certain infectious agents that affect the respiratory tract can pass from the animals to humans. These include: swine influenza, ornithosis, and Q fever. Workers employed in the hog industry are also exposed to infectious agents through exposure to animals in the slaughterhouse and in particular on the killing floor. In 1961, a unified national program was mounted to eradicate brucellosis from the nation's hog population. Although highly successful, the bacteria is still found in some herds throughout the country. In December of 1993, thirty-four swineherds nationwide were under quarantine for brucellosis in seven states. Packing plants that handle infected swine, however, do not follow special procedures to prevent occupational exposure.

North Carolina workers who were employed in hog slaughterhouses were exposed to brucellosis (Trout et al., 1995). In a study of workers at a particular facility, 19% of the kill floor workers were found to have evidence of recent or persistent brucellosis. Of the 154 study participants, one hundred and five (68%) reported experiencing two or more symptoms consistent with brucellosis during the previous year. The most common symptoms included chills, fever, headache, and myalgia/arthralgia.

Transmission of the bacteria, brucellosis, to humans can occur during the slaughter of infected swine through worker skin lesions, inhalation of aerosols, conjunctival contact and ingestion. Person-to-person transmission is rare. The study results pointed to skin exposure, and possibly conjunctival contact, as the most likely path of exposure. Infectious tissue or body fluids appeared to be the primary exposure route in the plant. Brucellosis is considered an under-diagnosed disease. Periodic and ongoing screening is warranted, in addition to the exclusion of infected herds from entering the slaughterhouses in the first place. Although the disease is treatable with antibiotics, fatigue can persist for months afterwards and can result in disability. Relapse illness

can occur.

4.2 Increased Virulence in Salmonella

Virulent forms of salmonella are on the increase in the United States. In 1996 34% of the salmonella cultures sampled in labs were found to be of the type DT104 (Salmonella enterica serotype typhimurium), which is resistant to most major antibiotics, including ampicillin, streptomycin, and tetracycline (Glynn et al., 1998). In 1979-80 only .6% of the bacteria salmonella that was tested was found to be DT104. The routine prophylactic use of antibiotics in livestock feed has helped fuel the evolution of this drug-resistant bacterial strain. This type of Salmonella is recognized as a major cause of illness in humans and animals in Europe. Glynn et al analyzed data collected by local and state health departments and public health laboratories. They concluded that multi-drug-resistant typhimurium has become a widespread pathogen in the United States and recommended a more prudent use of antimicrobial agents in farm animals and more effective disease prevention on farms. They estimate that in the United States, that each year there are an estimated 800,000 to 4 million salmonella infections, of which 500 are fatal, and approximately 40,000 are confirmed by culture (Glynn et al., 1998).

In addition to the above mentioned Salmonella, E-coli strains that are antibiotic resistant are becoming more common. A new potent version of staph has emerged. Staphylococcus aureus bacteria has recently not responded to the most potent antibiotic, vancomycin, and appears to have taken the life of a patient in New York (Dobnik, 1998). Doctors have been warning of the emergence of several drug-resistant bacteria, which can be attributed to the misuse and overuse of antibiotics both by humans and in animal production operations.

Some of these antibiotic resistant organisms arise from animal farming practices, including swine production, which involves the prophylactic use of antibiotics in the feed. With time, bacteria develop resistance to antibiotics by various means including by creating chemicals that weaken a drug's potency. Resistance can develop quickly. Vulnerable populations, such as the elderly, infants, and people with suppressed immune systems, are at a greater risk to drug-resistant species, as are farm workers themselves (Galvin, 1998). Human exposure to antibiotic-resistant bacteria from animals is most likely caused by improper food processing, storing, or cooking.

Of great concern is the increase in infections with antibiotic-resistant staphylococci and enterococci that cannot be treated (Witte, 1998). Such infections have a prime opportunity to develop and transfer in the hospital environment given the concentrated combination of bacteria adapted to this environment, patients prone to infections, and antibiotic use. Each year it is estimated that as many as two million Americans acquire a nosocomial infection, meaning one that they did not have when they were admitted to the hospital. As estimated by Dr. Stuart B. Levy, with the Tufts University School of Medicine, over 20,000 people die from these hospital-borne infections each year (President and Fellows of Harvard College, 1997).

4.3 The Prophylactic Use of Antibiotics

The nation's hogs, poultry, and cattle are fed 19 million pounds of antibiotics a year to enhance growth and protect against disease (Mansur, 1998). In a recently released study by the National Research Council, it was acknowledged for the first time that this practice presents a threat to public health. The study noted that this practice has in some instances led to the passage of resistant bacteria from animals to humans. The prophylactic use of antibiotics at subtherapeutic amounts allows more animals to be raised at lower costs (Galvin, 1998). There are two to three times more livestock than people in the U.S. and therefore many consider the use of antibiotics in animals to be a more important contributor to the environmental pool of resistant strains than that from humans (Harvard College President and Fellows, 1997).

The dependence on the use of antibiotics is caused by the emergence of intensive livestock operations, where by definition large numbers of animals are produced in close proximity. Hogs grown in ILOs are often in constant contact with other hogs, which results in a need for antibiotics to fight against the transmission of infections. Because these animals are regularly exposed to small amounts of antibiotics, any microbes they are carrying are more likely with time to develop a resistance. Antibiotics are also used in animal husbandry for prophylaxis, chemotherapy, and growth promotion. Antibiotic resistance affects such zoonotic pathogens as Salmonella serovars and Campylobacter spp., both of which are associated with diarrheal diseases, and Escherichia coli and Enterococci. The microbial ecosystems in humans and animals are intertwined and microbial antibiotic resistance readily crosses boundaries (Witte, 1998). Since meat products are traded worldwide and bacterial populations do not limit themselves to geographic boundaries, the problems caused by the inappropriate use of antibiotics are global in nature.

An examination of current animal-management practices, and improvement of such, could result in a reduction in the need for antibiotics. In addition, the development of effective vaccines will help reduce the demand for antibiotics (Galvin, 1998).

Drug resistant bacteria can pass from animals to humans by multiple means. They include: direct contact with the animals or their waste; exposure to food contaminated by the bacteria; or by person-to-person contact (Mansur, 1998). Commenting on the National Research Council study, Jean Halloran, the director of the Consumer Policy Institute for the Consumer's Union of United States Inc., who served on the study committee stated: There's been some debate, especially in agribusiness, that says the use of antibiotics in animals is a separate pool (from the human pool) and the animal usage can't affect the resistance problem that people have....The committee concluded this was not a debate. It said, "This is not true. The pools overlap."

The Center for Science in the Public Interest has advocated for a policy response to this issue. They have asked for a government ban on the routine use of antibiotics in food animals. The findings of the National Research Council committee called for better oversight of the issue through the creation of a national database on the use of antibiotics and resistance trends; the

establishment of a national oversight panel; and more research money for the development of new antibiotics (Mansur, 1998). Vigilance is needed to prevent the spread of bacteria that are resistant to present drugs.

Of further concern is the tainting of the nation's meat supply and its impact on human health. Some polls show that consumers are more worried about bad meat than violent crime. Figures estimate that there are as many as 81 million cases of food-borne illnesses a year, which result in approximately 9,100 deaths. It is believed that the cost of foodborne disease is as high as 5 billion dollars (Altekruse et al., 1997). The exact numbers of illnesses are being challenged by the CDC, which expects to release their own numbers. The Clinton administration, however, is trying to toughen national food laws in response to these concerns regardless of the exact numbers determined (*LA Times,* 1998).

4.4 Other Swine Intestinal Tract Pathogens

The intestinal tract of swine can carry numerous bacteria, viruses, and protozoa, which can be pathogenic to humans in addition to E-coli, Salmonella and Brucellosis. They include: Trichinosis, Toxoplasmosis, Leptospirosis, Erysipelothrix, Cryptosporidium, Streptococcus suis, and Yersinia enterocolitica (Fowler, 1998; Swinker, 1998). Although these pathogens may be fairly uncommon, there may be incidents that go undetected and unreported given lack of physician training in infectious diseases and insufficient resources to properly diagnose such.

Dr. Hervey Kornegay, a Duplin County physician, has taken a leadership role in working for improved safeguards to protect public health concerning the impact of hog ILOs in his county. Dr. Kornegay's interest in the health impacts of the industry are based on his professional background as a practicing physician in the community, his community service work as Chair of the Duplin County Board of Health, and as a father. During the summer of 1996, Dr. Kornegay's son attended a camp in Johnston County. The camp's lake was situated downhill from two ILOs, including a swine ILO. While at camp, his son contracted leptospirosis, a rare and unusual disease. Dr. Kornegay's son was hospitalized for five days with high fever. Dr. Kornegay recommends that physicians in eastern North Carolina consider this pathogen as a potential causal agent when children and adults report unusual febrile illnesses. The origin of the infection was never determined, however, one possible source was from the large adjacent hog operation. Even with assistance from the Centers for Disease Control a final determination was not made.

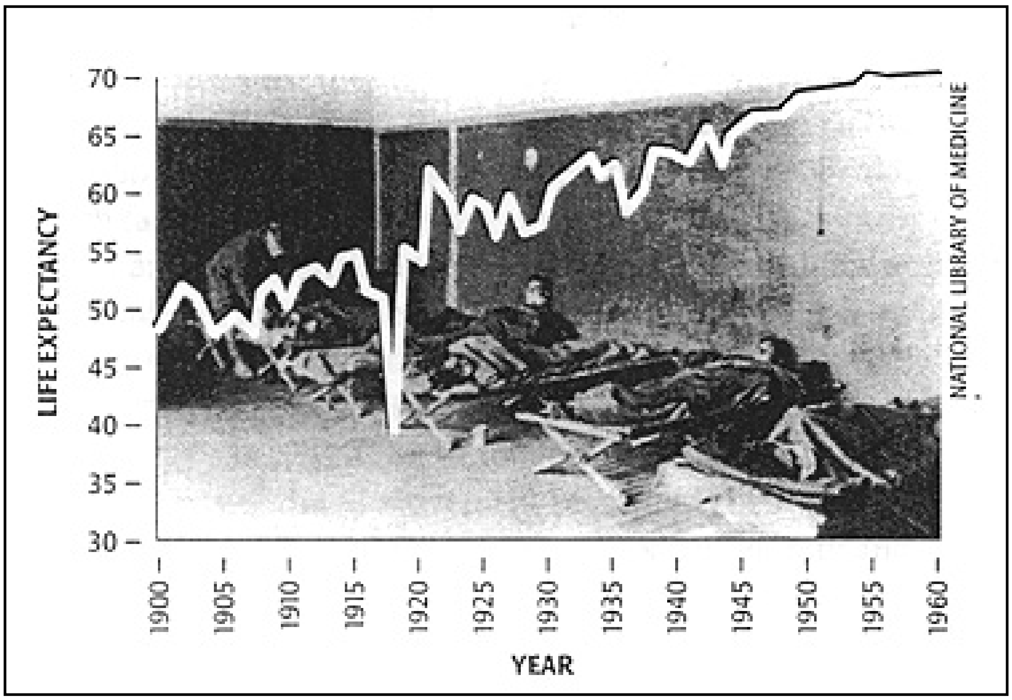
4.5 Swine or Spanish Flu of 1918

Here in North Carolina, close attention is paid to the possible transmission of disease from humans to swine. When people enter a hog house, they must wear special gear to protect the animals from exposure to viruses or bacteria carried by humans. Two family members can not seek employment in differing hog ILOs given concern over cross contamination between the herds. Equal protection and attention does not appear to be given concerning human exposure

from viruses and bacteria that are found in hogs. This omission could have catastrophic repercussions.

In 1918, the flu pandemic, which was also known as the Swine flu, Pig flu, and Spanish flu, swept the world and in little over one year's time between 20-40 million people died. In the United States alone, approximately 675,000 people died. It is estimated that 28% of the U.S. population was infected with the virus. (Taubenberger et al., 1997)

Ten times more people died from the swine flu than during World War I and almost twice as many as have died from the AIDS epidemic. Additionally, most of the deaths from the Swine Flu took place within a one-year period, whereas the AIDS' deaths have been spread over a fifteen-year period. The Swine Flu of 1918 was particularly virulent with mortality rates among the infected of over 2.5%, while other flus have less than a 0.1% death rate. The death rate was so high that it depressed the life expectancy in the U.S. by more than 10 years.



The Swine Flu of 1918 was different from any other flu before or since. Usually the youngest and oldest sectors of the population or those with a repressed immune system are most vulnerable. However, in 1918 young adult males, seemingly exceptionally healthy ones, were most likely to be struck by the disease. Once afflicted, most died within 24-48 hours. Young males were at the time of the flu outbreak mobilizing for World War I and were concentrated in military camps and on boats. This facilitated the spread of the flu among the troops and enabled the virus to move quickly through populations worldwide.

Researchers today are examining the virulent form of the Swine Flu virus. Knowledge of what this virus looks like could help health officials to identify dangerous flu strains as they emerge. Nancy Cox, a virologist with the Centers for Disease Control and Prevention (CDC), stated that study of this particular strain of the virus is needed to, "help us prepare for what we need to be prepared for." (Pennisi, 1997) The flu virus of 1918 infected the respiratory system, replicated, and then dispersed into the air via the lungs. Often all traces of the virus had disappeared by the time an autopsy was performed. Given the number of deaths from the Swine Flu, coupled with the war deaths, just disposing of the bodies became a problem, let alone having the resources to perform autopsies.

4.6 Hog as 'Mixing Vessel' for Avian and Human Viruses

Influenza A virus can escape the human immune response through antigenic drift (mutation), antigenic shift, or by the introduction of an avian virus into pigs. When introduced into the pig, the virus adapts to the new host, and is then passes on to humans (Scholtissek, 1994). Rarely does an avian influenza virus pass directly to humans. More commonly, the virus first passes through the pig. Data suggest that at the time of the Swine/Spanish Flu that a human influenza A (H1N1) virus entered the pig population and clustered with an "avian like" virus.

Pandemic strains are most likely formed when there is a reassortment of the genes. To assure the earliest possible identification of a pandemic strain, worldwide collaborating laboratories and surveillance system are required (Scholtissek, 1994). Close, continuous, and intensive monitoring of the world's swine population for the earliest possible detection of avian-like influenza viruses needs to be a basic part of global health planning.

Recent successful attempts to study the flu virus by means of polymerase chain reaction methods have allowed researchers to amplify small amounts of DNA and RNA in adequate amounts for sequence analysis. As a result, researchers now believe that the flu-virus genes closely resemble viruses isolated from pigs and that this virus had been in the pig population for a long period. The virologist, Dr. Robert Webster of St. Jude Children's Research Hospital said, "What this says is we had better watch what's happening in the pig populations of the world." (Pennisi, 1997) The pig trachea provides cell surface receptors for both human and avian influenza viruses (Ito et al., 1998). The pig trachea is conducive to viral replication and genetic reassortment. As the viruses replicate, the avian-like swine viruses develop the ability to recognize human virus receptors, and increase the possibility of their direct transmission to human populations (Ito et al., 1998). This allows the pig to act as an intermediate host for the generation of pandemic influenza viruses as seen in 1918, 1957, 1968, and 1977. Apparently, the pig acts as a 'mixing vessel' for such viruses.

Like the AIDS virus, the flu virus is ever changing. The CDC and the World Health Organization primarily spend their flu-surveillance efforts in examining flu strains in human populations, and not in the potential source itself, the hog. There is an obvious risk of studying such a virulent virus, which can be justified by the need to understand its structure and mechanism. Dr. Webster, when commenting on the 1918 Swine Flu, stated, "We want to know what killed these people. The potential is there for this kind of virus to return." (Pennisi, 1997).

Section Five: Wholistic View of Health Impacts

Our understanding of health and the determinants of health are limited and socially grounded. The language we use to describe the impacts of the intensive livestock operations and NC's newly structured vertically integrated agribusiness rarely includes the issues raised by community

members (Wing, 1998). For instance, the vocabulary used to describe the industry is determined by the industry itself, such as, the use of the word 'lagoon' vs. 'cesspool'; 'hog waste' vs. 'hog feces and urine'; 'hog farm' vs. 'hog operation'; 'growing house' vs. 'confinement facility'; 'harvest' vs. 'slaughter'; 'family farm' vs. 'factory farm'; 'agriculture' vs. 'agribusiness' and 'hog farmer' vs. 'hog operator/producer'. The top, powerful, rich controllers of the industry are called 'integrators'. The industry preferred vocabulary frames the way in which the industry is conceptualized in a more friendly, positive manner. One would not mind living near a 'lagoon' but would object to residing next to a 'feces and urine cesspool'. Don Webb, a NC community activist and founder and leader of the Alliance for a Responsible Swine Industry (ARSI), has worked arduously to recapture the vocabulary used to describe the industry, its activities and impacts.

Other issues surrounding the hog industry which are rarely raised include race and class, and the potential disproportionate effect of the industry on poor people of color. Such factors are raised by the environmental justice movement and their consideration of such are included in the Principles of Environmental Justice (EJ). In part fueled by the impacts of NC's hog industry, the state's first ever Summit for Environmental Justice was convened in Edgecombe County, in eastern NC, in October of 1998. National and state leaders of the EJ movement presented their ideas, in addition, to representatives of grassroots community and environmental organizations, state environmental organizations, state environmental agencies, and federal agencies, including EPA and NIEHS. Additional issues not often discussed are the agricultural determinants of nutrition, the loss of natural resources, and the emergence of conditions conducive to new diseases (Wing, 1998).

Policy development looks to risk factor epidemiology for guidance, which uses the paradigm of dividing the population into exposed and unexposed individuals. Such research does not ask the broader questions like:

- **where do the exposures come from;**
- **why are some groups exposed and others not;**
- **who benefits from producing the exposures;**
- **are the full range of exposures openly identified and studied;**
- **are the exposed populations fully informed;**
- **do they have access to medical care to remedy their exposure; and**
- **are the exposed populations fully compensated and made whole for their exposure (Wing, 1998)?**

These are some of the broader questions the environmental justice movement is trying to raise.

Professionals and community activists who challenge the dominant scientific account of health and disease argue that current epidemiological approaches are more likely attuned to protect the economic health of industry versus the health of the population or the environment. Such issues were discussed at the EJ Summit meeting.

By dividing scientific studies of the impact of hog intensive livestock operations into small units where specific diseases and exposures are examined one at a time, the overall effects of the industrial operations are never fully seen nor understood. This is also true of how we study the environment and set standards for emissions and is of importance when considering the effect of the hog industry. Air emissions from hog intensive operations consist of hundreds of constituent gases, none of which routinely exceed federal standards. However, the synergistic effect of all these gases has not been examined. A similar effect is seen when people take multiple medicines, where each one is taken at the prescribed amount, and yet together they can have serious effects when they interact with each other.

Victims of environmental exposures are too often blamed, questioned, isolated, and harassed (Wing, 1998; Tesh, 1988; Reich, 1991). The needs of subpopulations that are particularly vulnerable, such as pregnant women, asthmatic children, or adults with chronic respiratory disease, are rarely considered in policy deliberations. Other groups often overlooked are poor people of color.

Section Six: Recommendations

To assure the health and safety of North Carolina citizens who live near hog intensive livestock operations, further study needs to be made of the health impacts. It is particularly appropriate that such studies take place here since NC is the birthplace of this new concentrated livestock configuration. Additionally, the state's hog ILOs are more concentrated in one particular area of the state with sandy soils and a high water table. The impacts may prove to be greater here, especially due to the concentration of operations in two counties, than anywhere else in the nation. Based on the review of the research literature of hogs and human health, the following recommendations are made:

- **conduct a health study of the families who were found to be consuming nitrate contaminated well water (state and university researchers)**
- **test well water for coliform, in addition to nitrates (county staff);**
- **study the mental and physical health effects on residents of Duplin and Sampson counties, especially those living in odor affected areas (state and university researchers):**
- **conduct a clinical study of the impact of exposure to hog waste odors on asthmatics (state**

and university researchers);

- **establish a state standard for odor (NC Environmental Management Commission);**
- **check North Carolina's hog herd periodically for such infectious agents as Salmonella DT 104, and any avian viruses (NC Department of Agriculture);**
- **establish a protocol for communication and collaboration (NC Dept. of Ag., NC Occupational Safety and Health, county health department, and the State Health Director's Office) to assure a comprehensive response to the detection of swine diseases that could transfer to humans. This should include a protocol for handling potential emergencies and informing of the public; and**
- **develop a central database of demographic, psychological, and medical variables for persons exposed to hog odors and contaminated water and soil. Such a data base should include such markers as immunologic, neurologic, endocrinologic, psychologic and social factors (NC State Health Director's Office).**

Conclusions

There is a well-established literature describing the health effects of workers employed in both the confinement houses and the slaughter facilities of the hog industry. Two preliminary studies of the impact of the hog industry on neighbors were recently conducted. They concluded that in the physical health area, adjacent neighbors experience similar symptoms to employees only less severe and less frequent. In the mental health area, residents living in the odor-affected area were found to have increased incidents of mental health problems. These preliminary studies need to be redone in a more comprehensive and rigorous manner to ascertain the validity of their findings. The current status of the understanding of the potential threat to human health from hog operations is better described as what appears to be a problem or a potential problem, versus what is clearly understood. More research is needed to better understand the impacts of air and water emissions from hog intensive livestock operations and possible exposure to hog infectious agents. This is especially important given the potential for pandemic disease as the hog acts as a 'mixing vessel' between human and avian influenza viruses. Of additional concern is the increased presence of antibiotic resistant strains of bacteria, such as Salmonella (DT104), of which the pig is one of several potential sources. Increased funding for research efforts is needed to address such issues.

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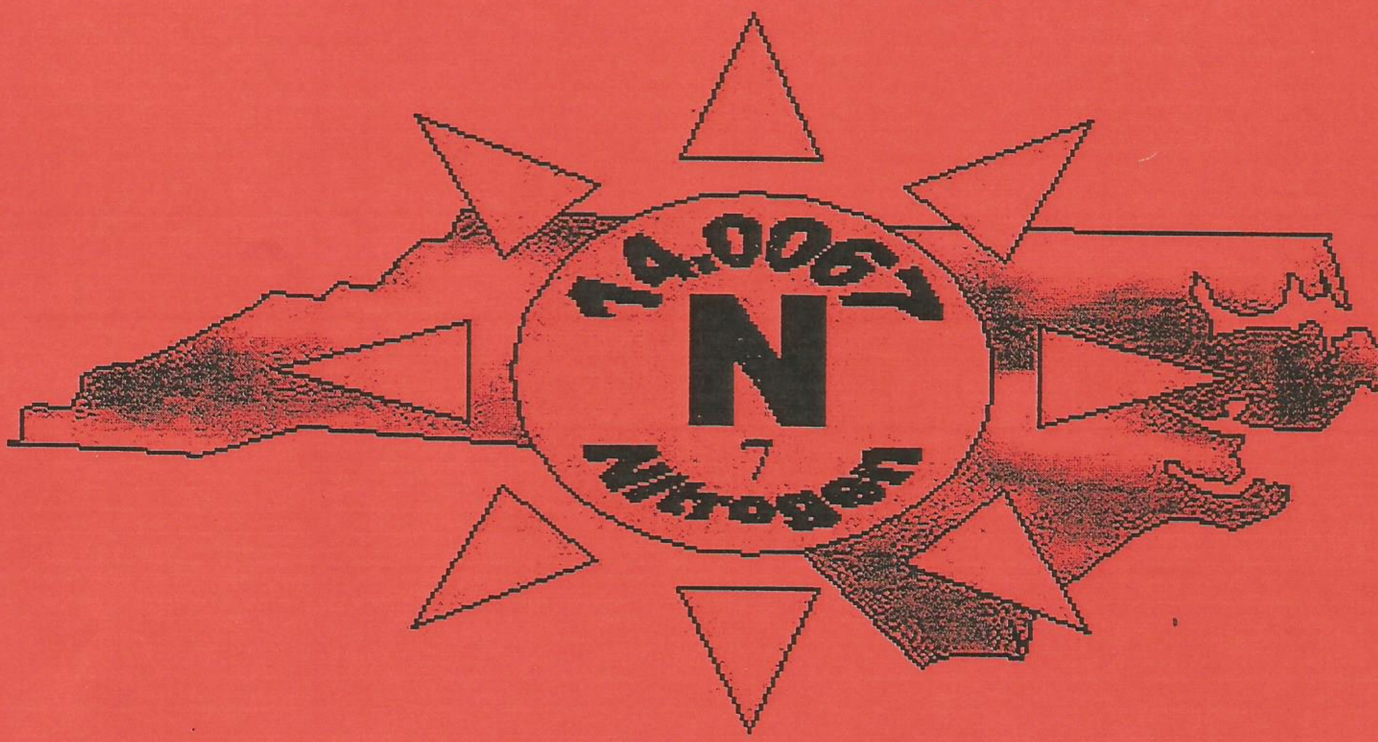
ATTACHMENT 11

Workshop On

ATMOSPHERIC NITROGEN COMPOUNDS II:

Emissions, Transport, Transformation, Deposition and Assessment

PROCEEDINGS



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Groundwater Contamination of Private Drinking Well Water by Nitrates Adjacent to Intensive Livestock Operations (ILOs)

- Nitrate standard - 10 ppm in drinking water.

- Nitrate-contaminated drinking water can cause blue baby syndrome in infants six months of age and younger resulting in serious illness and at high levels may be fatal. Nitrates may also be toxic to older children and adults.

- First private drinking water wells to be contaminated by a hog farm operation in North Carolina - October 1995, Robeson County.

- In response, Governor Hunt ordered free well water testing for all North Carolina citizens living adjacent to ILOs to protect their health.

- Well water testing program consists of testing at property owner's request and responding to request with lab analysis and health risk evaluation of testing results in each instance.

- As of August 7, 1998, 1595 wells in 57 counties have been tested for nitrates adjacent to ILOs.

- 34.2% of the wells (546 out of 1595) have exhibited nitrate contamination above 2 ppm.

- 10.2% of the wells (163 out of 1595) have exhibited nitrate contamination at or above the drinking water standard and may pose an increased health risk upon consumption.

- The percentage of contaminated wells adjacent to ILOs is approximately three times the statewide average for nitrate contamination of well water based on historical extension surveys.

- Five eastern North Carolina counties have potentially serious groundwater problems associated with ILOs - Duplin, Edgecombe, Johnston, Robeson, and Sampson Counties (see attached table).

- The majority of contaminated wells are still being investigated in regards to determining the responsible party.

- Hog farms in Robeson, Duplin, Johnston, and Lenoir Counties have been identified as responsible parties in the contamination of some offsite private wells.

- Leaking hog lagoons and hog wastewater spray fields have been responsible for well contamination in these instances.

- The poor condition of private wells, especially in eastern North Carolina counties contributes to the availability of nitrates in wells. Poorly constructed, shallow bored or drilled wells have exacerbated the nitrate contamination of many of the wells in this program.

Recommendations

- A well monitoring requirement be included in any future ILO regulations, which would be a prudent preventive public health measure that identifies groundwater contamination before it impacts private drinking water wells.

- Due to the elevated percentage of nitrate contaminated wells adjacent to ILOs, a more aggressive well sampling program will identify North Carolina citizens (especially infants under six months of age) who may be at risk from nitrate toxicity from contaminated well water.

- A well protection program that upgrades the quality of private wells, especially in eastern North Carolina, where these programs are for the most part, non-existent.

- A phasing out of the hog farm lagoon and spray field system which will provide a greater degree of proactive public health protection by eliminating the possibility of groundwater/well water nitrate contamination by hog farm operations.

- An extension of the moratorium until these public health protective measures are implemented.

Governor's Well Water Testing Project near ILOs Private Well Test Results for Nitrates

County	Total # Samples	# Samples > 2 ppm	# Samples > 9.5 ppm
Duplin	145	39	17
Edgecombe	138	74	9
Johnston	50	30	5
Robeson	317	139	34
Sampson	209	101	47
All other counties	736	163	51
Totals	1595	546	163

ATTACHMENT 12

Environmental Injustice in North Carolina's Hog Industry

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Rapid growth and the concentration of hog production in North Carolina have raised concerns of a disproportionate impact of pollution and offensive odors on poor and nonwhite communities. We analyzed the location and characteristics of 2,514 intensive hog operations in relation to racial, economic, and water source characteristics of census block groups, neighborhoods with an average of approximately 500 households each. We used Poisson regression to evaluate the extent to which relationships between environmental justice variables and the number of hog operations persisted after consideration of population density. There are 18.9 times as many hog operations in the highest quintile of poverty as compared to the lowest; however, adjustment for population density reduces the excess to 7.2. Hog operations are approximately 5 times as common in the highest three quintiles of the percentage nonwhite population as compared to the lowest, adjusted for population density. The excess of hog operations is greatest in areas with both high poverty and high percentage nonwhites. Operations run by corporate integrators are more concentrated in poor and nonwhite areas than are operations run by independent growers. Most hog operations, which use waste pits that can contaminate groundwater, are located in areas with high dependence on well water for drinking. Disproportionate impacts of intensive hog production on people of color and on the poor may impede improvements in economic and environmental conditions that are needed to address public health in areas which have high disease rates and low access to medical care as compared to other areas of the state. *Key words*: African Americans, environmental health, environmental justice, epidemiology, geographic information systems, rural health. *Environ Health Perspect* 108:225–231 (2000). [Online 8 February 2000]
http://ehpnet1.niehs.nih.gov/docs/2000/108p225-231wing/abstract.html

Environmental injustice refers to the disproportionate burden of pollution on people of color and the poor (*1–3*). In contrast to rural America's traditional image of unspoiled territory free of industrial pollution, poor rural communities have been targeted in recent years for urban, industrial, and military wastes that are unwanted by communities with larger populations and more political power (*4–6*). Other threats of environmental injustice in rural areas have come about because of the industrialization of agricultural activities (*7,8*). In this work we consider the environmental justice implications of the transformation of hog production in North Carolina from a system dominated by small independent farmers to large vertically integrated agribusiness production.

Between 1985 and 1998 North Carolina moved from fifteenth to second in hog production among U.S. states, with approximately 10 million head outnumbering the state's human population of approximately 7.5 million (*7,9*). The expansion of production has been accompanied by a declining number of operations and an increasing average size of operations (*10*). In 1998, market prices for hogs dropped to their lowest levels since the 1920s, which accelerated the demise of smaller independent producers. Most hogs are now produced by operators who work under contract to corporate integrators, which provide the management plan and own the animals, feed, and transportation; the operators own the land, buildings, and waste (*11*). In the past, hog production was dispersed throughout the state, but it has become consolidated in the coastal plain region, which concentrates waste and the potential for environmental damage in a region that is sensitive because of low-lying flood plains and high water tables (*10*).

Intensive swine production may pose environmental health dangers because of the high volume of waste, the chemical and microbial content of the waste, and the practice of using liquid waste management systems that are not isolated from the environment (*12*). In intensive hog production facilities, referred to as confined animal feeding operations (CAFOs), thousands of hogs are housed in large buildings. Waste is collected in cesspools for anaerobic decomposition and is subsequently sprayed on fields. Airborne emissions from confinement houses, cesspools, and spray fields contain ammonia, hydrogen sulfide, hundreds of volatile organic compounds, dusts, and endotoxins. These mixtures, which cause respiratory dysfunction in hog confinement-house workers (*13–28*) and possibly lower level symptoms in nearby residents (*29,30*), are highly obnoxious odorants that affect quality of life (*29–31*) and may be associated with mood disorders and lowered immune function (*32,33*).

Leaking cesspools and waste sprayed on fields can contaminate groundwater with nitrates and pathogens. The North Carolina State Health Department's (Raleigh, NC) well-testing program for the neighbors of intensive hog operations has documented elevated nitrates from hog operations (*34*). Groundwater contamination is a particular problem in eastern North Carolina because the water tables are high and many wells are shallow and unlined. No active population-based surveillance data are available to document pathogen contamination or the incidence of infections. Hog operations also contaminate surface waters, which may lead to high pathogen loads, eutrophication, and the promotion of algae and dinoflagellate growth (*35–39*).

The coastal plain region of North Carolina is also part of the southern Black Belt, a region where the agricultural economy was first built on the basis of slave labor and where a majority of rural African Americans in the United States still reside. The concentration of hog production in this poor region of the state has therefore raised the issue of environmental injustice (*40*). As in the case of other environmental justice problems, the presence of this polluting industry is a threat to public health because it may lower land values and quality of life and impede healthier economic developments that are needed in communities which suffer from low wages, lack of access to medical care, and poor nutritional options. Environmental injustice in the North Carolina hog industry has previously been investigated for counties (*7,9*) and U.S. Census Bureau (Suitland, MD) block groups (*41*). Using data for census block groups (areas of approximately 500 households), we

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examined the extent to which hog CAFOs are located disproportionately in communities with high levels of poverty, high proportions of nonwhite persons, and high percentages of households dependent on well water. In addition, because agricultural activities are located in rural areas where land is inexpensive, and because many rural areas are poor and nonwhite, we also considered whether relationships between the locations of hog CAFOs and poverty, race, and well use can be explained by the rural nature of these areas.

Materials and Methods

We obtained a list of all animal operations registered with the North Carolina Division of Water Quality (DWQ; Raleigh, NC) as of February 1998. Animal operators report information on the number of head, species and type of animals, aspects of the liquid waste management system, the latitude and longitude coordinates of the facility, and the name of the corporate integrator, if any, with whom the operator has a contract. Swine operations are required to register with the DWQ if they have > 250 head and if they use a liquid waste management system. The steady state live weight (SSLW) of the herd was calculated by the DWQ as a function of the number of head of each type (breeding sows, farrow to wean pigs, wean to feeder pigs, feeder to finish hogs, boars, and gilts) and the average weight for each type hog. Finished hogs, ready for market, weigh approximately 240 lb.

Of the 3,039 animal operations in the database, 2,585 were swine operations (Figure 1). Facilities with missing data or head counts < 250 were excluded. We located the facilities within the state using latitude and longitude data. For 257 facilities, geographic coordinates placed the facility outside of the county of operation, outside the state, or the coordinates were missing. Missing and incorrect geocoordinates were corrected using local maps, geographic information systems software, and the driving instructions provided to state inspectors. The DWQ was contacted to provide information for operations that were missing road instructions or had incomplete instructions, and on those that were out of business. Operations with coordinates inside the correct county were not examined further. Three university-owned operations, which are not subject to the same commercial location considerations as other facilities, were excluded from the analysis. The remaining 2,514 swine CAFOs were included in the analysis (Figure 1).

We used geographic coordinates for the swine operations to locate the facilities within the boundaries of block groups. The number of facilities in each block group was the dependent variable in analyses quantifying the association between number of hog CAFOs and the characteristics of block groups. Because airborne emissions from hog CAFOs may affect the environment well beyond their boundaries, we also conducted analyses considering buffer zones of 1 and 2 miles, in which the count of operations for a block group consisted of the number of hog CAFOs that were within 1 or 2 miles of the block group's boundaries.

Information on race, poverty, and water source was obtained for 1990 census block groups, the smallest geographical unit for which economic and demographic data can be obtained and the unit most closely approximating neighborhoods or communities. The 1990 census provided the most recent block group level geographic information available, and corresponded to the time during which hog production in North Carolina began to accelerate rapidly. Three environmental justice variables of interest were defined as the percentage nonwhite population, the percent of persons in poverty, and the percent of households that used well water. We also obtained the total number of persons, land area in square miles, and population density for each block group.

Some areas of the state, including metropolitan areas, have no presence of the commercial swine production industry. These areas, including mostly white Appalachia and some largely African American areas in central cities of the Piedmont, could have skewed the evaluation of the relationship between hog operations and the environmental justice variables. Therefore, we excluded from the analysis 14 of the state's 100 counties that did not border a county with a hog CAFO and the state's five cities with 1990 populations > 100,000. The remainder of the state considered in the analysis included 4,177 block groups with a population of approximately 4.9 million persons.

Relationships between the environmental justice variables (poverty, race, and water source) and the presence of hog CAFOs were first evaluated by summing the total number of hog CAFOs in quintiles of the distribution of each environmental justice variable. Because quintiles have the same number of block groups by definition, the ratio of the number of hog CAFOs in each higher quintile as compared to the lowest quintile of the variable is equal to the prevalence ratio of the number of operations per block group at higher levels as compared to the lowest level. This unadjusted measure is referred to as a crude ratio.

We prepared maps to show the spatial distribution of the major study variables. Chloropleth maps of poverty, race, and population density are keyed to bar graphs indicating the numbers of block groups in each category. Because block groups vary greatly in land area and because the visual impact of the chloropleth map is influenced by land area, categories based on quintiles of block groups are not sensitive to the spatial distribution of the variables. Therefore, we chose category boundaries for maps to reflect the distribution of each variable.

Agricultural operations of all types are located in rural areas, where population density is low and land is inexpensive. Rural areas have higher poverty rates, much of the southern Black Belt is rural, and rural areas are often not served by municipal water systems.

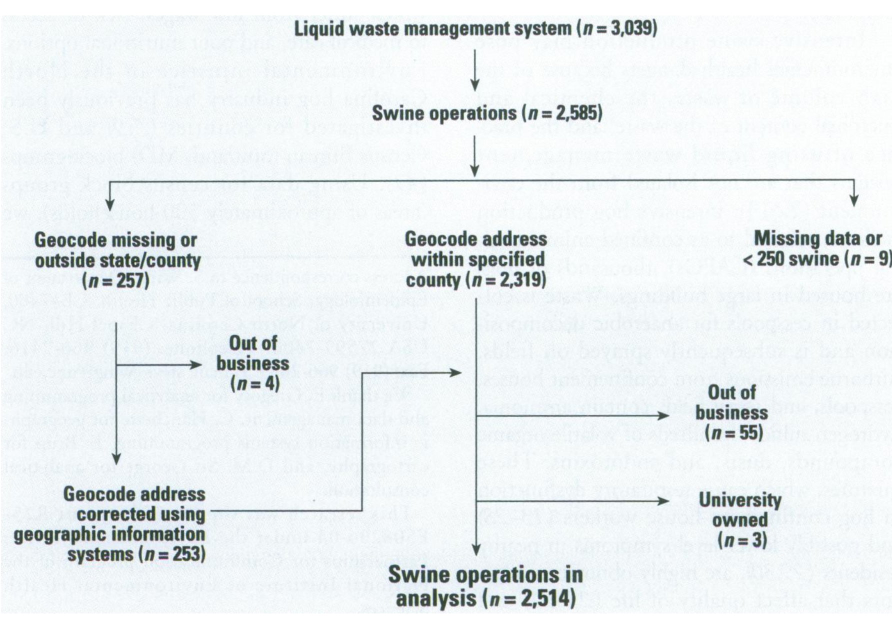

Figure 1. The identification of swine CAFOs from the DWQ data, February 1998.

It was therefore of interest to determine the extent to which excess numbers of hog operations in poor, nonwhite, and well-water-dependent communities could be considered a function of their low population density. We used Poisson regression to model the relationship between the natural log of population density and the number of hog operations per block group. We used linear, quadratic, and cubic terms for the log of population density to obtain an adequate fit of the model to the data. Higher order terms did little to improve the fit of the model.

Because Poisson models were overdispersed (model deviance/degrees of freedom > 1), we set the scale parameters for the models equal to the overdispersion values, which ranged from 1.6 to 1.8. We included indicator variables to represent each of the higher quintiles and we calculated the ratios of the number of hog CAFOs in block groups in each higher quintile of the environmental justice variables as compared to the lowest. We adjusted these ratios for population density using the cubic polynomial regression. Models were fit separately for operations under contract to corporate integrators and for those that were independent.

Results

Figure 2 shows the locations of hog CAFOs in North Carolina and the areas of the state excluded from the analysis. Each red dot represents one hog operation. The dense area of operations in the southeastern part of the state is centered on Duplin and Sampson Counties, the two largest hog-producing counties in the United States.

The size distribution of the 2,514 North Carolina hog CAFOs is shown in Table 1. The smallest 277 operations had an SSLW of < 100,000 lb each, which accounted for 11.0% of the operations and 1.4% of the state's SSLW. The SSLW of the largest 369 operations was ≥ 1 million pounds, which accounted for 14.7% of the operations and 44.4% of the SSLW in the state.

The geographic distribution of poverty is shown in Figure 3. Figure 3B shows the number of block groups in each category of poverty. For example, the categories with 0–5 and 5–10% persons in poverty each include approximately 1,000 block groups. Low-poverty areas predominate in the central Piedmont region of the state, whereas the higher poverty areas are located in the eastern coastal plain and in the northwest region (the edge of Appalachia).

Figure 4 shows the percentage nonwhite population. Most of the approximately 1,800 block groups with < 10% nonwhite population are located in the western part of the study area. These include 454 block groups that are 100% white. Areas with larger proportions of nonwhite population (mostly African Americans) are primarily in the eastern part of the state. An exception to the primarily African American makeup of the state's nonwhite population is Robeson County, located just southeast of the angle formed by the two straight lines along the central southern boundary of state. Robeson County is home to the Lumbee Indians and its population is approximately one-third Native American.

Table 2 presents the characteristics of block groups in relation to the environmental justice variables. Larger numbers of persons in the lowest categories of poverty live in a smaller land area, which results in higher population densities in areas with less poverty. Block groups in the lowest quintile of poverty contained only 43 hog CAFOs with 17.5 million lb of hogs, an average of 406.8 thousand lb/operation. In comparison, there are 225 hog operations in the second quintile

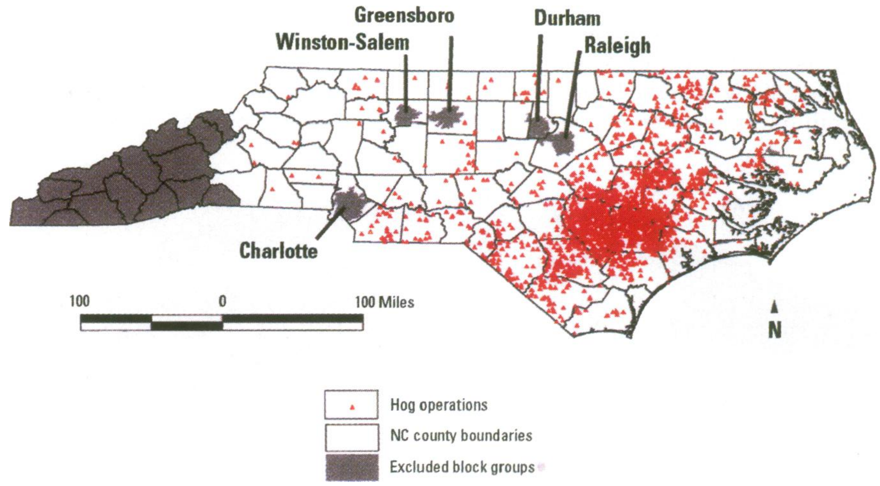

Figure 2. North Carolina study areas and locations of intensive hog operations, 1998.

Table 1. SSLW of North Carolina hog CAFOs, 1998.

SSLW (millions of pounds)	Operations (*n*)	Operations (%)	Cumulative SSLW (millions of pounds)	SSLW (%)
0.02 to < 0.10	277	11.0	20.8	1.4
0.10 to < 0.25	583	23.2	97.6	6.8
0.25 to < 0.50	708	28.2	268.2	18.6
0.50 to < 1.0	577	23.0	414.5	28.8
1.0 to < 10.1	369	14.7	639.7	44.4
Total	2,514	100	1,440.8	100

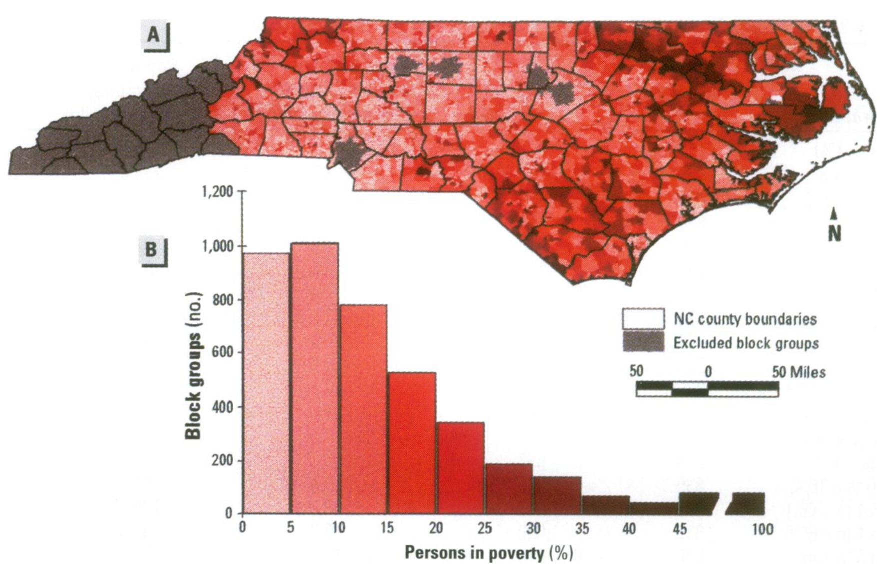

Figure 3. (*A*) The percent of persons in poverty in North Carolina, 1990. (*B*) The number of block groups in each category of poverty.

of poverty, 585 in the third, and > 800 in the fourth and fifth quintiles. Increases in total SSLW in areas with higher poverty levels are due to both larger numbers of operations and higher SSLW per operation.

Table 2 also shows the distribution of persons, land area, and hog operations for categories of the percentage nonwhite population. Population densities are lowest in the fourth and fifth quintiles of the percentage nonwhite variable. The 123 hog CAFOs in the lowest quintile have an SSLW of 48 million lb. The number of hog CAFOs in higher quintiles of the percentage nonwhite population increases to a maximum of 820 in the fourth quintile. The largest SSLW is in the highest quintile, 513 million lb, and the average size of operations increases with increases in the percentage nonwhite population.

Table 2 also presents information for block groups in quintiles of percentage of households using well water. This variable is most clearly related to population density, which declines from 1,315.4 persons/square mile in areas where < 1% of households have well water to 53.9/square mile in areas where > 85% of households have well water. Only five hog CAFOs, with a total SSLW of 1.2 million lb, are found in the lowest quintile of well-water use. Almost half of all hog CAFOs are located in block groups where > 85% of households have well water.

Although Table 2 shows clearly that there are more hog CAFOs in areas with higher percentages of persons in poverty, nonwhite persons, and households that use wells, it also shows that areas with the highest levels of these characteristics have lower population density, indicating that they are more rural areas. Population density is generally low throughout the eastern part of the state as compared to much of the Piedmont (Figure 5). Figure 6 shows that the number of hog operations per block group is strongly related to population density and that the observed number of operations per block group is predicted well by a cubic polynomial on a log–log scale. The number of operations per block group is lowest at the highest density, reaches a peak at approximately 20 persons/square mile, and declines somewhat at the lowest levels of density. The total number of operations in each category, shown in Figure 6 beside the observed values for the number of operations per block group, shows that the vast majority of operations are in block groups with fewer than 100 persons/square mile.

Table 3 summarizes the relationship between environmental justice variables and the presence of hog CAFOs in terms of the ratio of the number of operations per block group among block groups in the higher quintiles as compared to the lowest quintiles. The crude ratio of the number of operations per block group can be calculated from the data in Table 2. The ratio, adjusted for population density, is shown in the second column under each variable in Table 3. The large ratios for the higher levels of poverty, which vary from 5 to 20, are substantially reduced with adjustment for the rural nature of those areas. Adjusted ratios increase in a stepwise fashion with higher levels of poverty, from 3.0 in the second quintile to 7.2 in the highest.

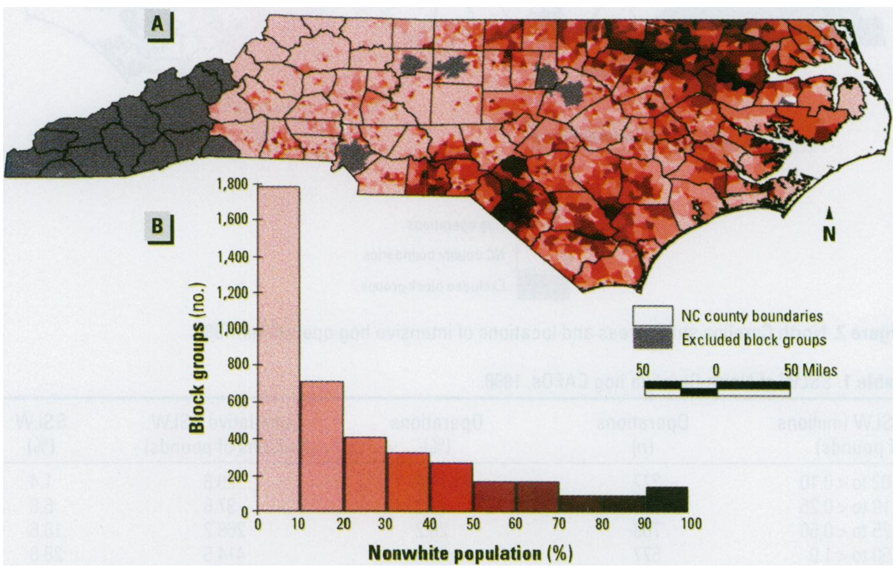

Figure 4. (*A*) The percentage nonwhite population in North Carolina, 1990. (*B*) The number of block groups in each category of the percentage nonwhite population.

Table 2. Characteristics of block groups in relation to poverty, race, and water source.

Characteristic	Block groups (*n*)	No. persons (thousands)	Land area (thousands of square miles)	Population density (people per square mile)	Total operations	Pounds of hogs (millions)	SSLW per operation (thousands)
Poverty (%)							
0 to < 4.9	835	1,118	4.7	238.0	43	17.5	406.8
4.9 to < 8.8	835	1,069	7.2	148.0	225	100.6	447.0
8.8 to < 13.6	836	966	9.4	103.0	585	284.9	486.9
13.6 to < 21.0	835	930	11.3	82.1	850	503.6	592.5
21.0 to 100	836	853	9.4	90.5	811	534.3	658.8
Nonwhite (%)							
0 to < 2.3	835	840	7.3	114.5	123	48.0	390.2
2.3 to < 9.3	835	1,048	6.3	165.2	165	78.1	473.6
9.3 to < 20.8	836	1,039	8.0	129.5	623	306.2	491.5
20.8 to < 44.2	835	1,103	10.5	105.5	820	495.5	604.3
44.2 to 100	836	907	9.9	91.7	783	513.0	655.1
Well water (%)							
0 to < 1.0	835	897	0.7	1,315.4	5	1.2	246.0
1.0 to < 16.4	835	1,068	3.4	314.4	185	91.6	495.1
16.4 to < 46.1	836	1,039	8.3	124.5	386	205.9	533.4
46.1 to < 85.5	835	1,020	12.7	80.5	734	450.5	613.7
85.5 to 100	836	914	17.0	53.9	1,204	691.6	574.4
Total[a]	4,177	4,937	42.1	117.4	2,514	1,440.8	573.1

[a]Sum for each variable.

Crude ratios for the percentage nonwhite population are smaller than the crude ratios for the percent of persons in poverty, ranging from 1.3 in the second quintile to 6.7 in the fourth quintile. Furthermore, the ratios are less affected by adjustment for population density. The ratio for the second quintile increases to 1.9, whereas ratios in the fourth and fifth quintiles are somewhat decreased. Adjusting for population density, the third, fourth, and fifth quintiles of the percentage nonwhite population have approximately 5 times as many hog CAFOs as those in the lowest quintile.

Hog CAFOs show a strong and monotonically increasing relationship to the percent of households using well water, with prevalence ratios ranging from 37.0 in the second to 240.8 in the fifth quintile. Most of this strong relationship, however, can be explained by the lower population density of areas with a high dependence on wells. Adjusted ratios in higher quintiles as compared to the lowest range between 4 and 5.

Table 4 shows the prevalence ratios for hog CAFOs in block groups cross-classified by poverty and the percentage nonwhite population, adjusted for population density. Block groups in the 0–5% poverty and 0–2% nonwhite population category are considered the referent group. Table 4 shows that increases in the percentage nonwhite population have little effect on number of hog operations among block groups in the lowest poverty group. Similarly, only modest increases in the numbers of operations are seen with increasing poverty levels among block groups in the lowest percent nonwhite category. However, prevalence ratios increase dramatically in areas with higher proportions of poor and nonwhite persons, reaching a ratio of 9 times as many operations in block groups with ≥ 12% poverty and ≥ 10% nonwhite population, adjusted for population density.

Most of the growth in NC pork production during the 1990s has been in large operations managed for corporate integrators rather than in independent operations. Therefore, we repeated the analyses for poverty and race separately for operations that listed corporate integrators on their permit applications (n = 1,603) and those that did not (n = 911). Prevalence ratios for integrator and independent CAFOs, adjusted for population density, are shown in Table 5. Although there is an excess of both types of operations in areas with greater percentages of poor and nonwhite populations, the excess is substantially larger for integrator operations at every level of poverty and race. Among the areas in the poorest quintile of block groups there are 20 times more integrator CAFOs than in the least-poor quintile, adjusted for differences in population density, whereas the excess of independent CAFOs in those areas is only 3.5 times. Similarly, block groups in the highest three quintiles of the percentage nonwhite population show an excess of integrator operations of 7 to 8 times, whereas the excess of independent operations is approximately 3 times.

Our analyses reported above consider only populations within the block groups containing hog CAFOs as potentially affected. However, airborne emissions and water pollution from CAFOs may travel some distance. Therefore, we reclassified the number of hog CAFOs in each block group considering 1- and 2-mile buffers around each operation. In these analyses, the number of hog operations in a block group is considered the number within the block group's boundaries plus the number within 1 or 2 miles of the block group, under the assumption that CAFOs located within 1 or 2 miles may impact the populations of neighboring block groups. We conducted analyses for the percent of persons in poverty and the percentage nonwhite population using the cubic polynomial model to adjust for population density. The ratios for the percent of persons in poverty were somewhat reduced, ranging between 2.2 and 5.9 under 1 and 2-mile buffers, as compared to a range of 3.0–7.2 with no buffer (Table 3). The ratios for the percentage nonwhite population were similar to ratios using a zero buffer, ranging from 1.9 to 5.3.

Discussion

We examined the locations of North Carolina's approximately 2,500 intensive hog confinement facilities in relation to poverty levels, race, and household water source of neighboring populations. These facilities are located disproportionately in communities with higher levels of poverty, higher proportions of nonwhite persons, and higher dependence on wells for household water supply. The disproportionate location of hog CAFOs in these areas raises numerous public health and social justice issues (*7,9,42,43*). Intensive swine production and its attendant pollution are concentrated in areas of North Carolina that have the highest disease rates (*44,45*), the least access to medical care, and the greatest need for positive economic development and better educational systems (*46*). The adverse effects of hog CAFOs on the quality of life and on community aesthetics (*29–31*) threaten the community economic and social developments that are fundamental to improved public health (*47*).

This study did not address siting decisions for particular hog operations. The reasons why a facility is located in a specific place are, in some ways, particular to the historical situation, business climate, local culture, and personal or family decision making. However, the pattern of location of industries reflects institutional factors and the political and economic power of local populations.

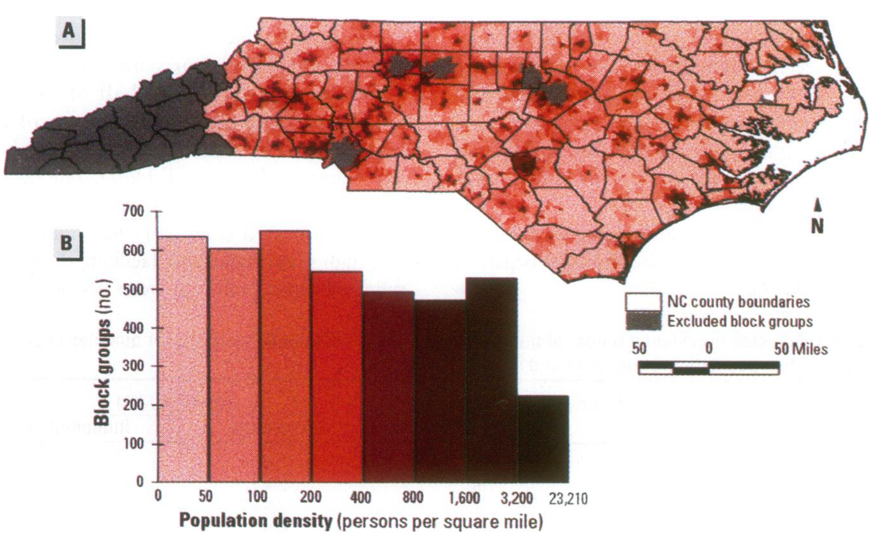

Figure 5. (*A*) North Carolina population density, 1990. (*B*) The number of block groups in each category of population density.

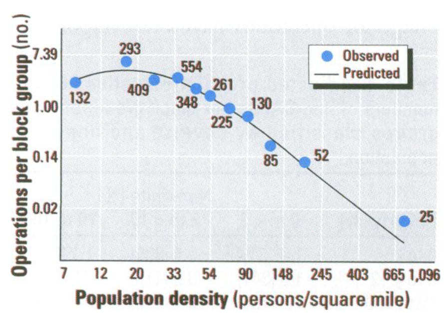

Figure 6. Number of operations per block group in relation to population density.

Table 3. Crude and adjusted prevalence ratios of numbers of hog CAFOs per block group for quintiles of poverty, nonwhite population, and well-water source.

Quintile	Poverty (%) Crude ratio[a]	Poverty (%) Adjusted ratio[b]	Nonwhite (%) Crude ratio[a]	Nonwhite (%) Adjusted ratio[b]	Well water (%) Crude ratio[a]	Well water (%) Adjusted ratio[b]
I	1.0	1.0	1.0	1.0	1.0	1.0
II	5.2	3.0	1.3	1.9	37.0	4.9
III	13.6	5.5	5.1	5.1	77.2	4.2
IV	19.8	6.4	6.7	5.1	146.8	4.2
V	18.9	7.2	6.4	4.7	240.8	4.7

[a]Unadjusted ratio of number of operations, higher quintile as compared to the first quintile. [b]Adjusted for population density, cubic polynomial.

These institutional inequalities are critically important issues to consider in addressing the public health problem of the disproportionate burden of polluting industries among poor and nonwhite populations (*1,2,5,40,48*).

Both poverty and race are strongly related to the location of hog operations, as shown in Tables 2 and 3. However, the combination of the two characteristics is of particular interest (Table 4). Increasing levels of poverty have only a modest effect in block groups with < 2% nonwhite populations. Similarly, increasing levels of nonwhite populations have little effect on the prevalence of hog operations among the block groups with < 5% poverty. It is the combination of a high percentage nonwhite populations and high poverty levels that is associated with the greatest excess of hog CAFOs, reaching a prevalence ratio of almost 10 for block groups with ≥ 12% poverty and ≥ 10% nonwhite population as compared to block groups with < 5% poverty and < 2% nonwhite population.

The industrialization of agriculture has brought about not only changes in size, but also in ownership. All of the hog operations considered in this research are large and fall under state regulations for intensive livestock operations. However, among these large operations, some are owned and operated by independent farmers who make their own management decisions. Other operations are owned by or are operated under contract with large agribusiness integrators that own and control the animals, feed, veterinary supplies, transportation, financing, and marketing of the product. Although both types of operations are large and industrialized, integrator operations have been responsible for most of the recent expansion of the industry (*7*). Because of their corporate structures, they may be in the best position to locate facilities based on economic considerations such as proximity to other operations, transportation routes, and slaughterhouses, as well as low land prices and the low local political power of host communities. Furthermore, there is a net decrease in jobs in regions where hog production has been industrialized because of the displacement of the independent producers who purchased locally (*49*). The concentration of hog CAFOs in poor and nonwhite areas is much greater for integrator than for independent operations (Table 5). Because the industry is moving rapidly toward greater economic concentration while family-owned businesses are in decline (*9,10,50*), the evidence of greater environmental injustice for integrator operations suggests that this problem may increase in the future.

This study was conducted using census block groups as the units of analysis. These areas, averaging approximately 500 households, are the smallest unit for which population data are available from the U.S. census and should provide better sensitivity and specificity to the characteristics of populations in greatest proximity to hog operations than would larger geographic units. The most recent block group data available are from 1990; more recent economic data from other sources are not available with this level of geographic detail. In any case, 1990 is an appropriate year for which to measure socioeconomic characteristics in our study of the location of hog operations because the period of rapid growth in the industry began in the late 1980s.

We depended on data from the DWQ for information on the locations and characteristics of intensive livestock operations in February 1998. Because a moratorium on the construction of new industrial operations was imposed by the North Carolina General Assembly in March 1997 (*7*) and has not yet been lifted (as of 1999), information from 1998 remains relevant. However, the validity of analyses reported here depend on the quality of information recorded by the state. We detected and corrected hundreds of errors in latitude/longitude coordinates for North Carolina hog CAFOs that were not located in the correct county according to the database (Figure 1). The extent of within-county errors in the data is unknown. Information on the size of the operation depends on the quality of data provided by the operator. The database contains information on a number of other characteristics of interest, such as the start date of the operation, the size and number of cesspools, and the acreage of spray fields. Unfortunately, these data were too incomplete to use in our analyses. Future studies of environmental justice and public health impacts of this industry would benefit from more complete and accurate data.

The public health implications of environmental injustice in the North Carolina hog industry are of special concern. Exposures in the environment of confinement houses are clearly related to impaired respiratory function, occupational asthma, and organic dust syndrome (*51*). This is an occupational health concern in areas with a large industry presence. In addition, environmental exposures to airborne emissions from hog CAFOs may be associated with respiratory effects (*29,30*) and impaired mood (*32,33*) in neighboring populations. Groundwater from hog CAFOs has been contaminated by nitrates in North Carolina (*34*). This is a special concern considering the findings presented here, which show that approximately half of the hog CAFOs are located in block groups of the state where > 85% of households depend on well water for drinking (Table 2). The eastern coastal plain of the state where most operations are located (Figure 1) has sandy soils and high water tables that facilitate the movement of water pollution from cesspools and

Table 4. Adjusted prevalence ratios[a] of the numbers of hog CAFOs per block group for block groups classified by poverty and nonwhite population.

Poverty (%)	Nonwhite (%) 0 to < 2	Nonwhite (%) 2 to < 10	Nonwhite (%) 10 to 100
0 to <5	1.0[b] (264)[c]	1.4 (335)	1.1 (254)
5 to < 12	1.8 (341)	3.6 (419)	7.0 (635)
12 to 100	1.7 (186)	3.1 (202)	9.6 (1,541)

[a]Adjusted for population density, cubic polynomial. [b]Referent group. [c]Number of block groups in parentheses.

Table 5. Adjusted prevalence ratios[a] of the numbers of hog CAFOs per block group for quintiles of poverty and nonwhite population: integrators and independents.

Quintile	Poverty (%) Integrators	Poverty (%) Independents	Nonwhite (%) Integrators	Nonwhite (%) Independents
I	1.0[b]	1.0[b]	1.0[b]	1.0[b]
II	7.2	1.9	2.4	1.5
III	16.2	2.7	7.5	3.4
IV	17.7	3.5	8.0	2.9
V	20.7	3.5	7.0	3.0

[a]Adjusted for population density, cubic polynomial. [b]Referent group.

spray fields into groundwater, and older rural homes occupied by the poor and people of color often have shallow wells with less protection from contamination. Surface water pollution is a concern because of the spread of microbial contamination and the nutrient loading of rivers and estuaries.

Community concerns about environmental injustice in the distribution of hog operations in North Carolina are real. Predominantly poor and nonwhite communities that host a disproportionate number of hog CAFOs have a great need for positive economic development, environmentally sound industry, and better schools and medical care. Such community resources are important to public health (*47*). However, future prospects for these communities are threatened by an industry that produces highly obnoxious odors and reduces the quality of life for neighbors (*29–31*), which can hamper the growth of cleaner industries, reduce land values, and contribute to loss of locally owned land (*9,40*). Our findings should be taken into consideration as growth, technological change, and environmental remediation in the industry are considered.

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ATTACHMENT 13

Air quality–related health damages of food

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Agriculture is a major contributor to air pollution, the largest environmental risk factor for mortality in the United States and worldwide. It is largely unknown, however, how individual foods or entire diets affect human health via poor air quality. We show how food production negatively impacts human health by increasing atmospheric fine particulate matter ($PM_{2.5}$), and we identify ways to reduce these negative impacts of agriculture. We quantify the air quality–related health damages attributable to 95 agricultural commodities and 67 final food products, which encompass >99% of agricultural production in the United States. Agricultural production in the United States results in 17,900 annual air quality–related deaths, 15,900 of which are from food production. Of those, 80% are attributable to animal-based foods, both directly from animal production and indirectly from growing animal feed. On-farm interventions can reduce $PM_{2.5}$-related mortality by 50%, including improved livestock waste management and fertilizer application practices that reduce emissions of ammonia, a secondary $PM_{2.5}$ precursor, and improved crop and animal production practices that reduce primary $PM_{2.5}$ emissions from tillage, field burning, livestock dust, and machinery. Dietary shifts toward more plant-based foods that maintain protein intake and other nutritional needs could reduce agricultural air quality–related mortality by 68 to 83%. In sum, improved livestock and fertilization practices, and dietary shifts could greatly decrease the health impacts of agriculture caused by its contribution to reduced air quality.

air quality | agriculture | fine particulate matter | food | pollution

The health and environmental consequences of feeding the increasingly large and affluent global population are becoming increasingly apparent. These consequences have spurred interest in identifying food production practices and diets that improve human health and reduce environmental harm. Recent work has demonstrated that many of the opportunities for food producers and consumers to improve nutritional outcomes also have environmental benefits, such as reducing greenhouse gas emissions, land and water use, and eutrophication (1–6). It is largely unknown, however, how individual foods and diets affect air quality, even though air pollution is the largest environmental mortality risk factor in the United States and globally (7, 8), and agriculture is itself known to be a major contributor to reduced air quality (8, 9). In the United States alone, atmospheric fine particulate matter ($PM_{2.5}$) from anthropogenic sources is responsible for about 100,000 premature deaths each year, one-fifth of which are linked to agriculture (10, 11).

Here, we show how different foods affect human health by reducing air quality. We consider the emission of pollutants that contribute to atmospheric $PM_{2.5}$, the chronic exposure to which increases the incidence of premature mortality from cardiovascular disease, cancer, and stroke (12, 13). These pollutants include directly emitted $PM_{2.5}$ (primary $PM_{2.5}$) and $PM_{2.5}$ formed in the atmosphere (secondary $PM_{2.5}$) from the precursors ammonia (NH_3), nitrogen oxides (NO_x), sulfur dioxide (SO_2), and nonmethane volatile organic compounds (NMVOCs). From a spatially explicit inventory of emissions of primary $PM_{2.5}$ and secondary $PM_{2.5}$ precursors from agricultural supply chain activities for commodities in the contiguous United States (*SI Appendix*, Figs. S1 and S2) (14, 15) (*Materials and Methods*), we estimate increases in atmospheric concentrations of total (primary + secondary) $PM_{2.5}$ attributable to agricultural emissions; total $PM_{2.5}$ transport, chemistry, and removal; and exposure of populations to total $PM_{2.5}$ using an ensemble of three independent air quality models (16–19). We describe damages attributable to 95 agricultural commodities and 67 final food products (full list in *SI Appendix*, Table S1), which cover >99% of US agricultural production (20).

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Results

We find that US agriculture results in 17,900 deaths (range across models: 15,600 to 20,300) per year via reduced air quality (Fig. 1 and *SI Appendix*, Figs. S3–S7). Damages are driven by NH_3 emissions (Fig. 1; "Pollutant"; 12,400 deaths; 69% of total) mainly from livestock waste and fertilizer application (Fig. 1; "Process").

Significance

Poor air quality is the largest environmental health risk in the United States and worldwide, and agriculture is a major source of air pollution. Nevertheless, air quality has been largely absent from discussions about the health and environmental impacts of food. We estimate the air quality–related health impacts of agriculture in the United States, finding that 80% of the 15,900 annual deaths that result from food-related fine particulate matter ($PM_{2.5}$) pollution are attributable to animal-based foods. By estimating these impacts and exploring how to reduce them, this work fills a critical knowledge gap. Our results are relevant to food producers, processors, and distributors, and to policymakers and members of the public interested in minimizing the negative consequences of food.

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Primary $PM_{2.5}$ is also a major contributor (4,800 deaths, 27% of total), largely from dust from tillage, livestock dust, field burning, and fuel combustion in agricultural equipment use. NO_x, SO_2, and NMVOCs are minor contributors (collective total: 700 deaths; 4% of total). Areas causing the greatest damages are spatially concentrated, with the top 10% of the most damaging counties (308 counties) together responsible for 8,400 deaths per year (47% of total deaths). These counties are mainly located in California, Pennsylvania, North Carolina, and along the Upper Midwest Corn Belt (Fig. 2 and *SI Appendix*, Fig. S4 and Table S2).

We also attribute total deaths from agricultural supply chain emissions to the production of specific commodities, which we combine into 16 groups (Fig. 1; "Commodity"). This analysis shows that 57% of deaths are from crops and 43% from livestock. However, a substantial portion of crops is used as animal feed and nonfood products (Fig. 1; "Product"). In attributing direct damages to final products, we find that 89% (15,900 deaths) of the total deaths caused by agriculture are linked to food production, with the remaining 11% (2,000 deaths) linked to biofuels and other nonfood products (e.g., plant and animal fibers) (Fig. 1; "Source"). Of food-related damages, 80% (12,700 deaths) are attributable to animal-based foods (when impacts of animal feed production are included) and 20% (3,200) to plant-based foods.

Next, we consider the per-unit damages of 11 food groups (Fig. 3 and *SI Appendix*, Table S1), taken as the production-weighted average of the foods in each group, and using four metrics suited to meet different nutritional needs (per 10^9 kg, 10^9 serving, 10^9 g protein, and 10^9 kcal, each measured as raw edible portion). We find that red meat dominates in air quality–related health damages, whether normalized by total mass, serving, protein mass, or caloric value. Per serving, production-weighted averages of red meat are 2× greater than those of eggs, 3× greater than those of dairy products, 7× greater than those of poultry, 10× greater than those of nuts and seeds, and at least 15× greater than the production-weighted average of any other plant-based food. Similar trends hold when these food groups are compared using the other three metrics. The lowest-impact production of red meat has a greater impact than the highest-impact production of any other food, absent the dietarily insignificant comparison of red meat to fruit as measured on a per-protein content basis. We observe a wide range of spatial variation in per-unit damages of major crops and livestock commodities (*SI Appendix*, Fig. S8). Damages vary spatially because of site-specific production practices, atmospheric chemistry and transport, and population density (Fig. 2 and *SI Appendix*, Fig. S4), consistent with prior studies focused on maize (21) and switchgrass (22). Limitations of supply chain information (e.g., where the crops that are fed to animals in a given location are grown) restrict our understanding of the spatial variation in per-unit damages for animal-based foods as final products.

We also estimate the air quality–related health benefits that can be achieved through the actions of food producers and consumers. We identify interventions that reduce $PM_{2.5}$-related emissions, focusing on interventions that target the most harmful agricultural processes, promote dietary shifts, reduce food loss and waste, and encourage healthy per capita consumption levels (*Materials and Methods*). We generate spatially explicit inventories for each intervention scenario and compare them to a baseline scenario for current production practices and diets (*SI Appendix*, Fig. S9 and Table S3), modeling the resulting changes in $PM_{2.5}$ concentrations and annual deaths.

We find that improvements in agricultural production, such as changing livestock feed practices to reduce the amount of excess protein ingested and therefore excreted as nitrogen, or using fertilizer amendments and inhibitors, can greatly reduce air quality–related health damages (Fig. 4). Implementing measures to reduce agricultural emissions across all producers could prevent 7,900 deaths per year (50% of total deaths from food production). The greatest benefits are from changes in livestock waste management and fertilizer application practices. Producer-side interventions in the 10% of counties with the highest mitigation potential alone could prevent 3,600 deaths per year (22% of total deaths from food production). Expanding such interventions to the top 50% of counties would prevent 41% of total $PM_{2.5}$-related deaths linked to food production.

Our findings suggest that the monetized $PM_{2.5}$-related health benefits of such interventions could greatly exceed implementation costs. For example, using a Value of Statistical Life of $10 million (23, 24), we find that the annual monetized damage cost of

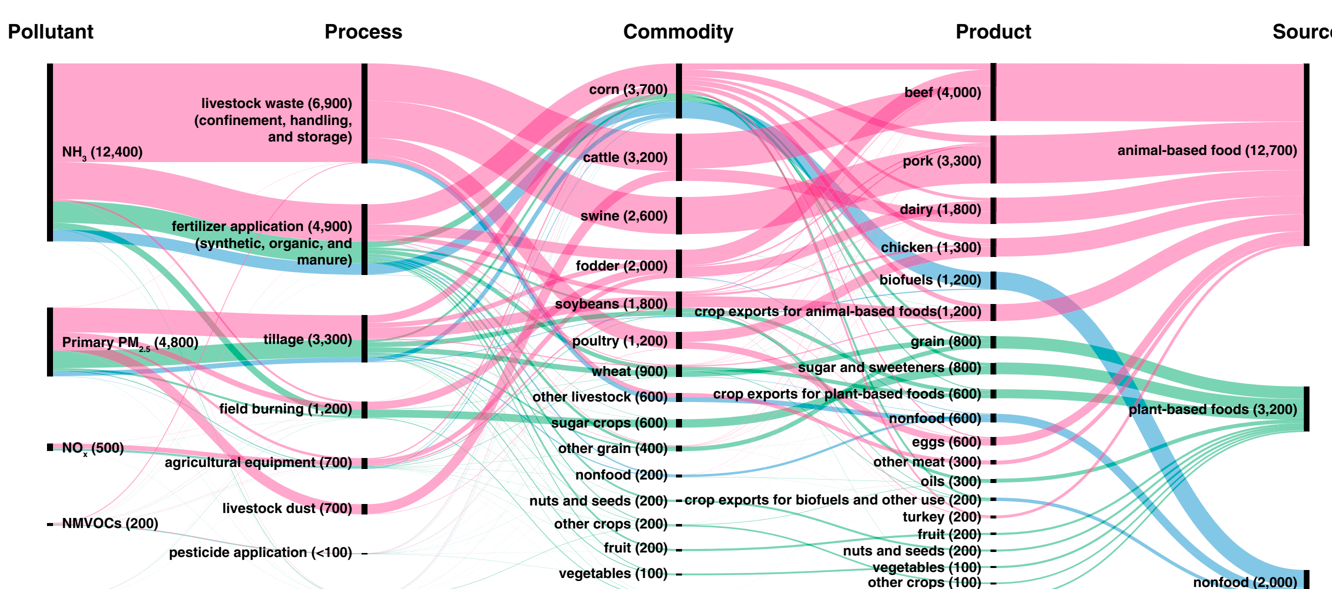

Fig. 1. Annual premature deaths attributed to increased atmospheric $PM_{2.5}$ from agriculture. Five alternate categorizations (columns) are shown: pollutant, process, commodity, product, and source. Pollutants include primary $PM_{2.5}$ and secondary $PM_{2.5}$ formed from precursor gases (NH_3, NO_x, NMVOCs, and SO_2). The height of each black bar within each column corresponds to the number of attributed deaths; deaths within each column sum to 17,900.

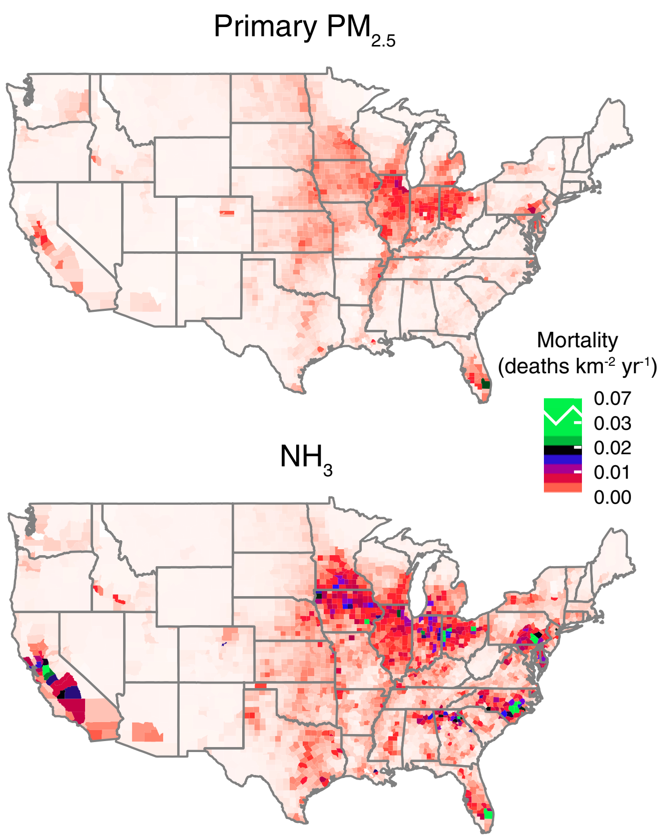

Fig. 2. Spatial distribution of $PM_{2.5}$-related mortality attributed to US agricultural production. Shown are annual premature deaths per square kilometer attributed to primary $PM_{2.5}$ (*Top*) and secondary $PM_{2.5}$ from NH_3 (*Bottom*), which together comprise 97% of agricultural $PM_{2.5}$-related deaths. Maps for the other 3% of deaths (i.e., from NO_x, NMVOCs, and SO_2) are shown in *SI Appendix*, Fig. S4. For each county, the mortality shown is that which occurs somewhere in the United States as a result of emissions from that county; that is, these maps show where the impact originates, not necessarily where it is experienced.

$PM_{2.5}$-related deaths from US food production are \$159 billion. The benefits of many of the explored interventions are 1.3 to 14.7× greater than the highest estimated implementation costs, consistent with the results of coarser resolution global analyses (25). For instance, the $PM_{2.5}$-related health benefits (range: 33.4 to 42.4 $\$ \cdot kg^{-1}$ of NH_3) of interventions for nonorganic fertilizer application, such as improvements in timing, method of application, use of amendments and inhibitors, and a shift to less emissive fertilizer types, greatly exceed the implementation costs (range: –0.8 to 3.2 $\$ \cdot kg^{-1}$ of NH_3).

We also find that nationwide dietary shifts that decrease consumption of animal-based foods can lead to large decreases in agricultural $PM_{2.5}$-related death rates, simultaneously reducing direct damages from livestock waste management and indirect damages from feed production (Fig. 4). Substituting poultry for red meat could prevent 6,300 annual deaths (40% of total deaths from food production). Even greater benefits of 10,700 to 13,100 deaths prevented per year (68 to 83%) could be achieved from more ambitious shifts to vegetarian, vegan, or flexitarian diets such as the planetary health diet of the EAT-Lancet Commission (2). Other demand-side mitigation strategies, such as decreasing caloric intake proportionally across all food groups to be in line with metabolic requirements and decreasing household food loss and waste levels, could lead to more modest reductions in agricultural $PM_{2.5}$-related death rates (range: 700 to 1,200 avoided deaths per year).

Many of the food production solutions that could reduce air quality–related health damages, such as improving nitrogen use efficiency in crop and livestock production, or decreasing food

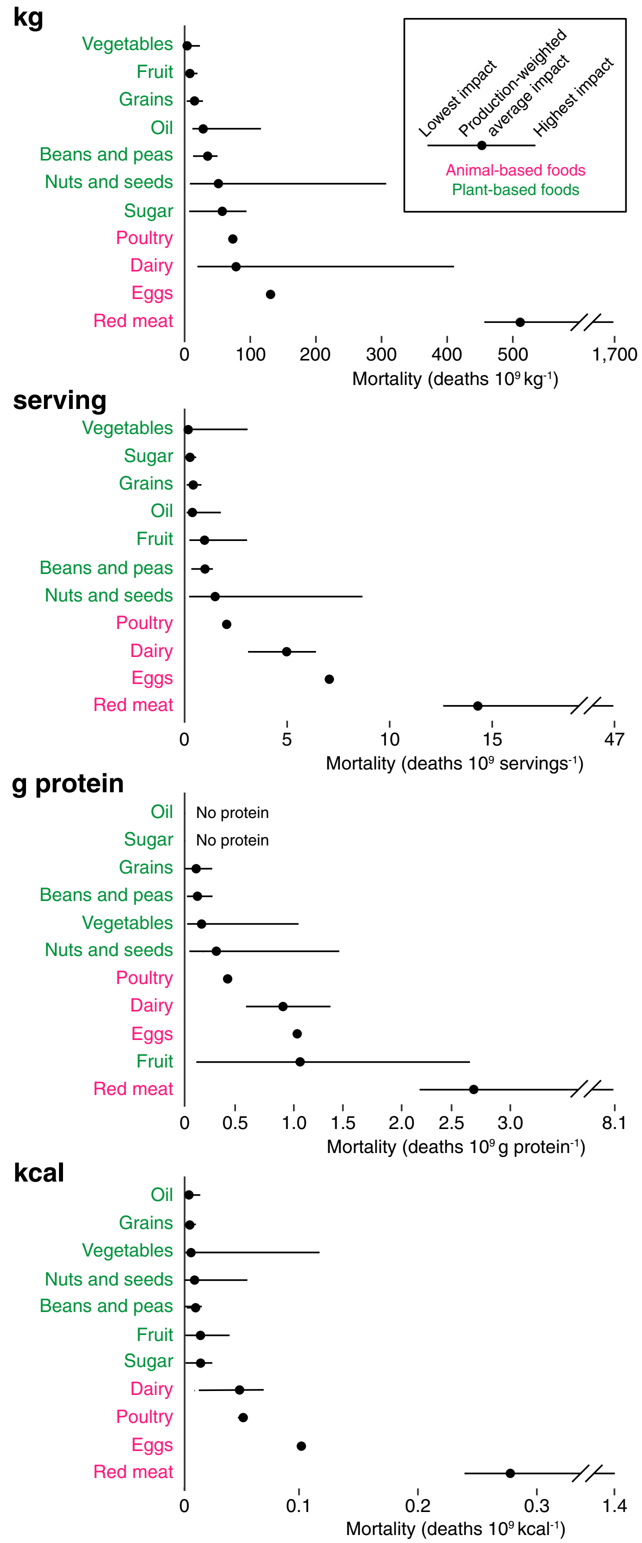

Fig. 3. Annual premature deaths attributed to total $PM_{2.5}$ per unit of food production. Annual premature mortality attributed to total $PM_{2.5}$ per 10^9 kg, 10^9 serving, 10^9 g protein, and 10^9 kcal, each measured as raw edible portion. Horizontal lines indicate the range of per-unit damages within the food group. Food groups are ordered lowest to highest within each panel.

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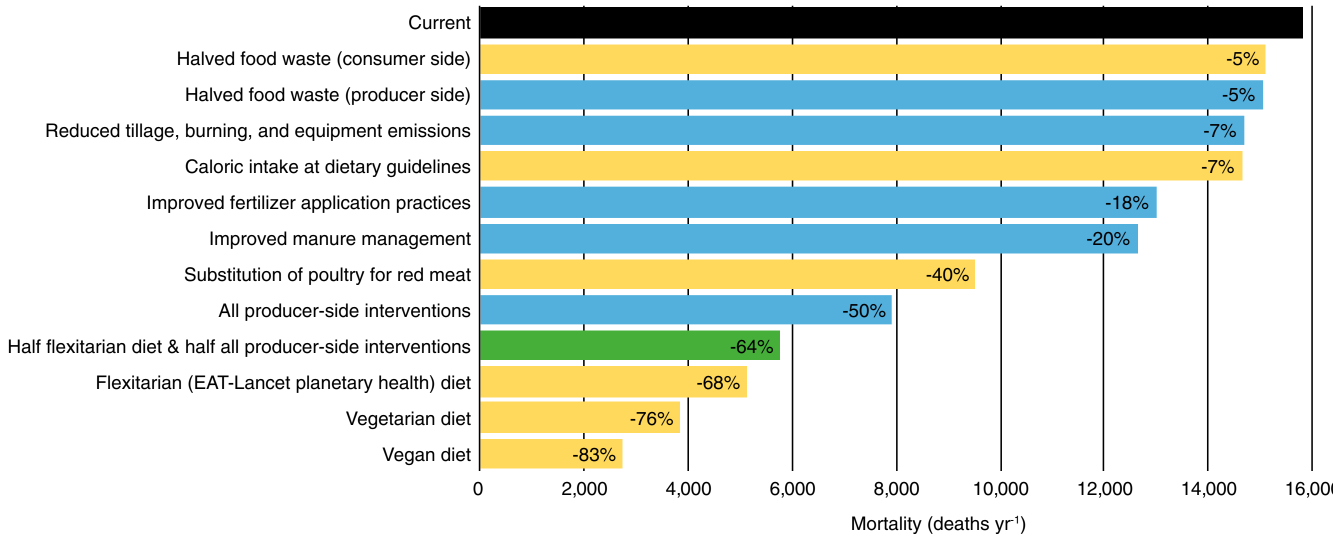

Fig. 4. Annual premature deaths attributed to total $PM_{2.5}$ from food production that could be mitigated by a given intervention or suite of interventions. Yellow bars correspond to consumer-side interventions, blue bars to producer-side interventions, and green to a combination of the two. Values for percent decrease in mortality from current mortality are shown.

loss and waste, are likely accompanied by other environmental benefits, such as decreasing greenhouse gas emissions, nutrient pollution, and undesirable land-use change (26–28). Further, dietary shifts that increase the fraction of kilocalories from plant-based foods can improve diet-related health outcomes by reducing the incidence of chronic noncommunicable diseases, such as type 2 diabetes, coronary heart disease, and cancer (29, 30).

This work contributes to a more comprehensive understanding of the air quality–related health damages of food and identifies solutions for reducing the negative impacts of food across a diverse range of diets, production practices, and other site-specific factors. Current diets and food production practices cause substantial damages to human health via reduced air quality; however, their corresponding emissions sources, particularly ammonia, are lightly regulated compared to other sources of air pollution, such as motor vehicles and electricity production. This is true despite agriculture having comparable health damages to these other sources of pollution (10, 31). Meaningful reductions in air quality–related health damages will likely require simultaneous interventions, such as dietary shifts and changes in how we manage livestock waste and apply fertilizer. Although our results are for the United States, our approach can be applied globally, with mitigation efforts anticipated to reduce premature deaths substantially. Reductions should be especially large in regions where $PM_{2.5}$ concentrations are sensitive to ammonia emissions, where agricultural burning is commonly practiced, and in densely populated regions with high $PM_{2.5}$ exposure levels (25).

Materials and Methods

We estimated the air quality–related annual deaths attributable to the US agricultural sector, which includes annual deaths attributable to 95 agricultural commodities that span the entirety of animal production, and cropland and grassland pastures captured in the 2014 US Department of Agriculture (USDA) Cropland Data Layer (CDL) (32). We then computed the per-unit annual impacts of 67 final products from 11 food groups. Finally, we estimated the air quality–related health benefits that could be achieved through producer- and consumer-side interventions.

Extraction of Agricultural Emissions. County-level air pollution impacts from the agricultural sector were estimated by identifying and extracting emissions data linked to crop and livestock production from the US Environmental Protection Agency (EPA)'s 2014 National Emissions Inventory v2 (NEI2014) (14). Emissions from the contiguous 48 states were included, covering >99% of US agricultural production. We identified agricultural processes from source classification codes (SCCs), with the seven agricultural processes listed as follows: 1) livestock waste (confinement, handling, and storage), 2) tillage, 3) fertilizer application (synthetic, organic, and manure), 4) field burning, 5) agricultural equipment fuel combustion, 6) livestock dust, and 7) pesticide application (15). As 88% of all ammonia, ammonium nitrate, urea, and other nitrogen compounds produced in the United States are used as fertilizer, the same fraction was used to allocate emissions from the production of these chemicals to a "fertilizer production" category (33). All emissions were used as published by the NEI2014, with two exceptions. First, NEI2014 estimates of primary $PM_{2.5}$ from tillage and livestock dust do not account for differences in fugitive dust emissions by land cover, as they depend on dry deposition rates and wind speeds (15). As a result, county-level transport fractions were applied to account for local effects (34). Second, we adjusted the estimates of tillage emissions in NEI2014 using data collected by the USDA in the Agricultural Resource Management Survey to better reflect the current number of tillage passes associated with individual crops (35).

Allocation of Emissions. We categorized emissions by pollutant type (i.e., primary $PM_{2.5}$, NH_3, NO_x, SO_2, and NMVOCs), agricultural process (e.g., livestock waste and fertilizer application), commodity that is the onsite emissions source (e.g., livestock types such as beef cattle, and crop types such as corn), and final product (e.g., beef).

Livestock production. Emissions from livestock production come from either livestock waste management or ancillary livestock dust. These emissions were attributed to specific animal types using SCCs within the NEI2014 (15). Livestock waste emissions of 10 major livestock types (beef cattle, dairy cattle, broilers, layers, swine, turkeys, goats, lambs, horses, and other livestock) were estimated separately by different management stages (confinement, handling and storage, and land application of manure) using the Carnegie Mellon University (CMU) Farm Emissions Model (36) but were aggregated into a livestock-specific emissions category in the NEI2014 (15). To allocate emissions by management stage, we derived the state-level distribution of emissions by management stage by first running the CMU Ammonia Model on which the Farm Emissions Model is based (36). Next, we applied that distribution to emissions published in the NEI2014. All emissions associated with confinement, handling, and storage were attributed to the livestock commodity. Emissions associated with land application of manure were attributed to crop production and further allocated to a specific crop commodity using crop production practices data from the USDA Economic Research Service (ERS) (35).

Crop production. In terms of emissions, crop production processes of interest included fertilizer production, fertilizer application, tillage, agricultural equipment use, field burning, and pesticide application. For fertilizer production and pesticide application, national-level emissions were distributed according to crop fertilizer and pesticide use data published by the USDA ERS (37, 38). For tillage and field burning, SCCs within the NEI2014 were used to allocate emissions to a specific crop type (15). Because the NEI2014 only includes tillage emissions for 21 crops, we introduced tillage emissions from an additional 71 crops available in the 2014 CDL by averaging county-level average emissions factors for annual or perennial crops in the NEI2014; we then applied the appropriate emission factor (annual or perennial) to county-level crop acreage (32).

Emissions from agricultural equipment use and fertilizer application are not preallocated to crops in the NEI2014 and therefore required additional allocation. For agricultural equipment use, we allocated county-level emissions to crop commodities using the county-level crop acreage of 92 crops (including grassland pasture) in the CDL (32). In the case of fertilizer application (for synthetic and organic fertilizers but not manure), we allocated county-level NH_3 emissions to specific crops, in proportion to crop acreages from the CDL (32) paired with crop-specific nitrogen volatilization rates and irrigation rates obtained from the Environmental Policy Integrated Climate (EPIC) model (39). However, the EPIC model only includes crop-specific data for 20 crops, with all other emissions aggregated into an "other_crops" category. As preliminary results suggested that the total emissions of the other_crops category were comparable to those of major crops such as corn and soybean, further resolution in the other_crops category was achieved by allocating emissions in proportion to the county-level distribution of crop acreage from the CDL. Emissions from the roughly 1% of counties listed in the NEI2014 but not included in the EPIC model were conserved and allocated according to the state-level distribution of emissions.

We then allocated emissions from crops to final products (e.g., crop products, animal products, exports, and biofuel) using production data from annual Yearbook Data Tables and the 2015 Agricultural Statistics Report (20, 40). This allowed us to estimate the total annual deaths associated with the production of 95 agricultural commodities as well as the per-unit annual deaths associated with the production of 67 food products (*SI Appendix*, Table S2).

Annual Deaths. We input spatially explicit emissions inventory data into three reduced-complexity chemical transport models (RCMs): Air Pollution Emission Experiments and Policy v3 (16), EASIUR (Estimating Air pollution Social Impact Using Regression) (17), and Intervention Model for Air Pollution (18). All three models include simplified representations of atmospheric chemistry and physics, which reduce computational demands relative to traditional chemical transport models, including linearization that omits meteorological coupling. This enabled us to evaluate a broad range of emissions scenarios. At the same time, each of the models has a different structure and makes different simplifying assumptions, which reduces the likelihood that all three models would make the same type of error. We chose these RCMs as they allow users to distinguish the $PM_{2.5}$-related mortality by emissions source locations and provide higher resolution than other national-scale RCMs, such as the US EPA Response Surface Model (19). The RCMs are described in the *SI Appendix*.

Because the RCMs only cover counties in the contiguous United States, we excluded NEI2014 emissions in noncontiguous states from the analysis. The RCMs are customized to estimate annual deaths according to the American Cancer Society's concentration-response function, which averages a 6% increase in annual deaths per 10 $\mu g \cdot m^{-3}$ in $PM_{2.5}$ concentration. Although NO_x and volatile organic compounds can react to form tropospheric ozone (O_3), which can also result in premature mortality, we excluded O_3 from this analysis because the resulting air quality–related health impacts are overall greatly exceeded by those of $PM_{2.5}$ (8).

Despite key differences between the formulation of the three RCMs, estimated marginal social costs per tonne of primary $PM_{2.5}$, NH_3, NO_x, and SO_2 generally fall within a factor of 2–3 for all US counties (19). The agreement of RCM model predictions is greatest for primary $PM_{2.5}$ (Pearson correlation coefficient = 0.73 to 0.81) for which the atmospheric chemistry that translates pollutant emissions to changes in $PM_{2.5}$ concentrations is relatively straightforward. It is weakest for NO_x and SO_2 (Pearson correlation coefficients of 0.35 to 0.49 and 0.07 to 0.54, respectively) for which the atmospheric chemistry is more complex. Overall, total emission-weighted annual deaths in the United States vary between 12 to 33% for ground-level sources (19).

Sensitivity Analysis. We evaluated the seasonal sensitivity of annual deaths using the seasonal social costs per tonne estimated by the EASIUR model. Specifically, we tested the seasonal sensitivity of NH_3 from livestock waste management and fertilizer application: NH_3 is the primary driver of agricultural emissions, and social costs per tonne of NH_3 are highest (roughly 2.5× greater) when seasonal emissions are relatively low (41). The NEI2014 estimates annual emissions. We obtained monthly NH_3 emissions from livestock waste management and fertilizer application by applying the monthly distribution of emissions from Pinder et al. and Goebes et al., respectively (41, 42). Damages using the seasonal option in EASIUR were comparable to those using the annual average option (*SI Appendix*, Fig. S6).

Mitigation Interventions. We estimated the air quality–related health benefits that can be achieved through interventions by producers and consumers, largely targeting NH_3 as it is a major driver of $PM_{2.5}$-related deaths attributable to food production in the agricultural sector (43). Specifically, we focused on intervention scenarios that could reduce emissions linked to livestock waste management, fertilizer application, tillage, field burning, fuel combustion, dietary shifts, food loss and waste, and per capita consumption levels.

We estimated the air quality–related health benefits of interventions by comparing health outcomes for an intervention scenario with those for a baseline scenario in which food producers and consumers behave according to business as usual. To measure the health outcomes linked to an intervention scenario, we first estimated the emissions reductions that can be achieved from a specific intervention and used that information to create a spatially explicit inventory of reduced emissions. Using the emissions inventory as input for the RCMs, we then modeled the resulting health outcomes.

We estimated emissions reductions of producer-side intervention scenarios by averaging emissions reductions linked to existing interventions as found in a survey of the literature (*SI Appendix*, Table S4). Identified interventions were grouped by agricultural process such as livestock waste management and fertilizer application. When possible, they were further grouped into subcategories such as livestock housing type or fertilizer type. For instance, emissions reductions for interventions targeting dairy cattle at the confinement stage are estimated by averaging emissions reductions for individual interventions, such as installing grooved floor systems with tooth scrapers in the confinement facilities or establishing a tree shelterbelt surrounding confinement facilities.

With regard to consumer-side interventions, we examined two caloric scenarios: 1) caloric intake is at current US levels (average of 2,590 kilocalories per capita per day), and (2) caloric intake is reduced to a level that would maintain a body mass index between 20 and 25 for an average person (average of 2,400 kilocalories per capita per day) (44). In the second scenario, we assumed that caloric intake was reduced proportionally across all food groups to achieve the target caloric level. Six isocaloric dietary scenarios from the EAT-Lancet Commission were considered: 1) business as usual in the United States, 2) the planetary health diet, 3) the planetary health diet with high milk consumption, 4) planetary health diet with high red meat consumption, 5) vegetarian, and 6) vegan (2, 3). In modeling alternative diets, the EAT-Lancet Commission considers national preferences of different food groups. Because foods can be imported or exported, the composition and volume of foods produced in the United States do not exactly match those of foods consumed in the United States, though 87% of food and beverages purchased in the United States are domestically produced (45).

We considered two options for estimating annual deaths linked to diets and food production in the United States for the business-as-usual dietary scenario: 1) we assumed annual deaths in the business-as-usual dietary scenario are equal to annual deaths from the US agricultural sector minus annual deaths from nonfood production, and 2) we assumed that annual deaths in the business-as-usual dietary scenario can be computed by matching the per-kilocalorie annual deaths associated with food groups with the caloric composition of the average United States diet from the EAT-Lancet Commission (2). The EAT-Lancet Commission estimates caloric composition of the baseline US diet using country-specific food availability data and model equations from the International Model for Policy Analysis of Agricultural Commodities and Trade. We found that the estimated annual deaths from both options differed by 8% (option one: 15,900 deaths per year and option two: 17,200 deaths per year). We used the results of the first option to analyze mitigation interventions because that option more closely represents current US food production. We estimated annual deaths linked to alternative scenarios by matching the per-kilocalorie annual deaths associated with food groups with the caloric composition of the diet being examined (assuming all food consumed in the United States is produced domestically). We assumed that the average damage of food groups remained constant with changes in production. More information on the per-kilocalorie annual deaths attributable to food groups and the caloric composition of the diets are in *SI Appendix*, Tables S2 and S3, respectively.

We considered two food loss and waste scenarios: food loss and waste at current US levels (46), and food loss and waste reduced by 50%. We distinguish food loss and waste by food type and by food supply chain stage (agricultural production, postharvest handling and storage, processing and packaging, distribution, and consumption) using estimates for the North America and Oceana region (46). We also assumed that reductions occur proportionally across both food loss and waste.

SUSTAINABILITY SCIENCE

Finally, we explored the cost effectiveness of selected NH_3 mitigation interventions, including changes in practices related to livestock feed, animal housing, manure storage, manure application, and synthetic fertilizer application.

Data Availability. All study data are included in the article and/or *SI Appendix*.

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ATTACHMENT 14

IOWA CONCENTRATED ANIMAL FEEDING OPERATIONS AIR QUALITY STUDY

Final Report

Iowa State University and The University of Iowa Study Group

February 2002

Table of Contents

Acknowledgements

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Foreword

In June, 2001, Governor Tom Vilsack asked the Presidents of Iowa State University and of The University of Iowa to assist the Iowa Department of Natural Resources and the Environmental Protection Commission with addressing public health and environmental concerns arising from air emissions from concentrated animal feeding operations (CAFOs). With the concurrence of both presidents, Iowa Department of Natural Resources Director Jeffrey Vonk charged the College of Public Health at the University of Iowa and the College of Agriculture at Iowa State University to recommend standards for air quality and address other issues regarding CAFOs.

The Colleges of Agriculture and Public Health assembled teams of faculty with appropriate expertise to complete a comprehensive review of available scientific information to address five questions asked by Director Vonk. At ISU, faculty from the College of Veterinary Medicine also made important contributions to this effort. The ISU team was led by administrators from both of these colleges. At The University of Iowa, the Environmental Health Sciences Research Center, sponsored by the National Institute for Environmental Health Sciences, assembled a team composed of faculty from the Colleges of Public Health, Engineering and Medicine. Together, these faculty delved into existing research literature, developed a ten-chapter report on the various aspects of these issues and, through a series of meetings, developed responses to Director Vonk's five questions in the form of an Executive Summary. This Executive Summary describes the consensus reached by the study group. Individual chapters are the products and views of the chapter authors. Independent national and international scientists, with appropriate expertise, reviewed and commented on both the Executive Summary and the full report.

The report is based upon the best science available to ensure that rural ambient air is as free of risk as possible in order to protect health and the quality of life at the highest possible level. These science-based recommendations were generated with the goal of providing helpful guidance to the Iowa Department of Natural Resources and the Environmental Protection Commission. It is hoped that the report will provide a sound basis for the development of appropriate administrative rules that will promote confidence in agricultural production and the quality of life in rural Iowa.

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CHAPTER 1 Executive Summary

Introduction

In mid-June of 2001, Governor Tom Vilsack requested that the faculty of the two universities address the public health and environmental impacts of concentrated animal feeding operations (CAFOs, also referred to as Concentrated Feeding Operations or CFOs). In response to this request, Richard Ross, PhD, DVM, Dean of the College of Agriculture at Iowa State University and James Merchant, MD, DrPH, Dean of the College of Public Health at The University of Iowa, were asked by the Department of Natural Resources Director Jeffrey Vonk to provide guidance **"regarding the impacts of air quality surrounding CFOs on Iowans and recommended methods for reducing and/or minimizing emissions. Specifically, I am asking your advice and recommendations on how the Department of Natural Resources should address this critically important public policy issue."**

Director Vonk asked five questions. Through a series of discussions and meetings, a combined study group of faculty and consultants (See Attachment 1) was identified, conflict of interest and confidentiality statements were signed by all faculty and consultants, definitions were discussed and agreed upon, a comprehensive report outline was developed and agreed upon and individual teams of faculty agreed to write each of the 10 chapters that constitute the full report. A technical and policy workshop was held in Des Moines on December 18 and 19, 2001, at which time chapter presentations were made and discussions were held regarding the series of five questions asked by Director Vonk. Groups were assigned to summarize the responses to these five questions in this Executive Summary. Peer review of this Executive Summary and the full report was considered to be vital to the validity and integrity of the report. This peer review, completed by national and international scientists who are experts in the areas addressed by the report (See Attachment 2), was completed in January, 2002. Their review comments, as well as comments from members of the combined study group, were discussed at meetings on January 8, 24 and 29 and were useful in completing the final report for submission to the Iowa Department of Natural Resources (IDNR). An agreed-upon glossary, which defines the many technical terms used in this report, is found in Attachment 3.

Response to Question 1

There are two questions contained in Question 1. The first is:

Based on analysis of peer-reviewed, duplicated, legitimate, published scientific research, is there direct evidence of harm to humans by emissions, byproducts, toxic waste, or infectious agents produced by CFOs?

There is now an extensive literature documenting acute and chronic respiratory diseases and dysfunction among workers, especially swine and poultry workers, from exposures to complex mixtures of particulates, gases and vapors within CAFO units. Common complaints among workers include sinusitis, chronic bronchitis, inflamed mucous membranes of the nose, irritation of the nose and throat, headaches, muscle aches and pains. Asthma and acute (cross-shift) declines in lung function are documented among CAFO workers, even though workers with pre-existing asthma usually select themselves out of such employment because of increased asthma severity. Progressive declines in lung function over years are documented among CAFO workers. Those workers with increased acute declines in lung function, which are often accompanied by chest tightness and wheezing (asthma-like syndrome), have been found to have more rapid declines in lung function over time. Very high exposures to hydrogen sulfide, which occurs during pit agitation, may result in death from asphyxia and respiratory arrest; those who survive such high dose exposures often develop reactive airways distress syndrome (RADS), bronchiolitis obliterans and severe respiratory impairment. It is therefore concluded that there is direct evidence of harm to humans from occupational exposures within CAFOs (See Chapter 6.3.2).

However, one cannot directly extrapolate occupational health risks observed among workers inside CAFOs to community health risks that may arise from CAFO emissions. While the discharge of airborne particulates and gases/vapors from CAFOs and manure handling clearly occur, the aerosols at the point source differ from ambient exposures as they move downwind, both in composition and in concentration. The populations at risk (workers) within CAFO units and within the community (community residents) also differ significantly. CAFO workers are generally a healthy population (those fit enough to work), while community residents include children, the elderly, and those with preexisting impairments. Regulatory agencies recognize the need for lower exposure limits to compensate for increased susceptibility among community residents, to allow for uncertainty factors from epidemiological study findings (and for species to species differences when animal data is used) to establish community ambient exposure limits.

The second part of the first question is:

What human research is there to confirm the existence of disease and exactly what are the specific chemical, bacterial, or aromatic causes of such diseases?

Published, controlled studies of odor experienced by community residents living in proximity to CAFOs are limited to two studies in North Carolina and one in Iowa. The first North Carolina study reported more negative mood states (tension, depression, anger, reduced vigor, fatigue and confusion) among those exposed to CAFO odor compared with control subjects. The second North Carolina study reported increased symptoms of headache, runny nose, sore throat, excessive coughing, diarrhea, burning eyes and reduced quality of life measures among community residents living in proximity to a swine CAFO compared with rural residents not living in proximity to livestock operations. The Iowa study found increases in several symptom clusters, mainly eye and upper respiratory symptoms, among those living within two miles of a swine CAFO compared with rural residents living near minimal livestock production. These studies are limited in size and scope, did not make specific environmental exposure or odor measurements, and are subject to recall bias. They are notable in that they are controlled studies that report eye and respiratory symptoms associated with concentrated livestock exposures that are similar to more prevalent and severe symptoms experienced by CAFO workers who are exposed at much higher concentrations of mixed emissions (See Chapter 6.3.3).

Also relevant in responding to this question are many experimental and epidemiological studies of non-CAFO populations exposed to low concentrations of individual chemical components of CAFO emissions, particularly hydrogen sulfide, ammonia and endotoxin. These studies document respiratory symptoms associated with low levels of these individual exposures. Because at least two of these chemicals (hydrogen sulfide and ammonia) are found in CAFO emissions that contribute to ambient community exposures, these experimental and community exposure studies are relevant to this question (See Chapter 6.3.1). Both the Environmental Protection Agency (EPA) and the Agency for Toxic Substance and Disease Registry (ATSDR)[1] have recommended ambient exposure limits for ammonia and hydrogen sulfide based on these studies.

It is concluded that no specific disease(s) *per se* among community residents can be confirmed to arise from a specific chemical, bacteria or aromatic cause. However, the findings of the limited community studies of concentrated livestock exposures are consistent with adverse health effects observed in other experimental and epidemiological studies of some specific chemicals (ammonia and hydrogen sulfide) known to be components of CAFO air emissions. It is, therefore, also concluded that CAFO air emissions may constitute a public health hazard[2] and that precautions should be taken to minimize both specific chemical exposures (hydrogen sulfide and ammonia) and mixed exposures (including odor) arising from CAFOs.

Response to Question 2

Question 2: Based on an analysis of peer-reviewed, duplicated, legitimate, and published scientific research, what specific substances, including aromatic compounds, do you believe require regulatory action to protect the public?

By consensus of the entire study group, the following substances should be considered for regulatory action: (1) hydrogen sulfide; (2) ammonia; and (3) odors. The justification for regulatory action of these substances is based on our assessment of the scientific literature, (See Chapters 2.0-8.0), recommendations by pertinent federal agencies, and review of regulations established in other states (See Chapter 9.0).

Hydrogen sulfide and ammonia are recognized degradation products of animal manure and urine (See Chapter 3.4 in the full report). Both of these gases have been measured in the general vicinity of livestock operations at concentrations of potential health concern for rural residents, under prolonged exposure (See Chapter 8.0).

The World Health Organization lists hydrogen sulfide as a toxic hazard in many environments, and recommends specific exposure limits. The ATSDR lists hydrogen sulfide and ammonia on its registry of toxic substances[1] under its federal mandate to protect the public health according to the Comprehensive Environmental Response, Compensation, and Liability Act, [42 U.S.C. 9604 et seq] as amended by the Superfund Amendments and Reauthorization Act [pub. 99-499]. Furthermore, the ATSDR has published Minimum Risk Levels (MRL's) for these substances to protect the public's health.[1] The EPA historically evaluates scientific information regarding environmental contaminants and the potential threats for human health hazards. Based on a standardized risk assessment process, the EPA identifies hydrogen sulfide and ammonia as potentially hazardous substances.[3] A detailed description of the process and justification used by the EPA and ATSDR to include ammonia and hydrogen sulfide as hazardous substances is provided in detail in Chapter 8.7.

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[1] Agency for Toxic Substances and Disease Registry, Minimal Risk Levels for Hazardous Substances (MRL's), http://www.atsdr.cdc.gov/mrls.html

[2] hazard: the potential for radiation, a chemical or other pollutant to cause human illness or injury

[3] Environmental Protection Agency, Integrated Risk Information System, www.epa.gov/iris/subst.html

Minnesota and Nebraska have established air quality standards for hydrogen sulfide based on public health concerns. California and Minnesota regulate ambient concentrations of hydrogen sulfide based upon nuisance and human health effects. Minnesota is in the process of setting standards for ammonia ambient exposures. Monitoring of ammonia ambient exposures is taking place in Missouri. The regulatory actions taken by other states in setting standards are described in Chapter 9.0.

Odors have been a major concern of residents in the vicinity of CAFOs (see Chapter 3.4, 4.0, 6.8 and 8.0). Colorado, Missouri, and North Carolina have recognized the need to promulgate odor regulations. Details of the processes of odor regulations for these states are presented in Chapter 9.0.

Response to Question 3

Question 3: Based on an analysis of peer-reviewed, duplicated, legitimate, and published scientific research, what would you recommend as Iowa or National consensus standards for any proposed substances to be regulated as emissions from CFOs?

The study group recommends that ambient air quality standards be developed to regulate the concentration of hydrogen sulfide, ammonia and odor. There has been considerable discussion on what standard levels should be established for each pollutant as well as where the measurement should take place. Some states measure concentration at the property line of the source while others measure at the residence or public use area. The U.S. EPA has determined that simultaneous exposure of two substances such as hydrogen sulfide and ammonia (both pulmonary irritants) results in an additive effect. Thus, in order to protect against the adverse effects of such binary mixtures the exposure limit for each should be reduced accordingly. While emissions from CAFOs fluctuate over time, they produce chronic rather than acute exposures. Rather than representing single doses, these exposures are recurring and may persist for days with each episode.

The study group reached consensus that measurements for hydrogen sulfide and ammonia should be taken at the CAFO property line and residence or public use area. Measurements for odor should be taken at a residence or public use area and one proposal includes measurements at the CAFO property line. The study group recommends that measurements for hydrogen sulfide and ammonia should be time weighted rather that instantaneous to allow for atmospheric variability.

With current animal production practices, stored manure must be removed and land-applied. During these times hydrogen sulfide, ammonia and odor levels at or near production facilities may be significantly higher than during normal conditions. Therefore, it is also recommended that provisions be made for allowable times to exceed the established standards to allow for proper manure application to land. Notification must be given to the Iowa DNR and nearby residents, at least 48 hours in advance when the operation expects to exceed the standards

The study group provides the following recommendations on the regulation of hydrogen sulfide, ammonia, and odor from CAFOs:

<u>Hydrogen Sulfide</u>
It is recommended that hydrogen sulfide, measured at the CAFO property line, not exceed 70 parts per billion (ppb) for a 1-hour time-weighted average (TWA) period. In addition, the concentration at a residence or public use area shall not exceed 15 ppb, measured in the same manner as the property line

measurement. It is recommended that each CAFO have up to seven days (with 48 hour notice) each calendar year when they are allowed to exceed the concentration for hydrogen sulfide.

Ammonia

It is recommended that ammonia, measured at the CAFO property line, not exceed 500 ppb for a 1-hour TWA period. In addition, the concentration at a residence or public use area shall not exceed 150 ppb, measured in the same manner as the property line measurement. It is recommended that each CAFO have up to seven days (with 48 hour notice) each calendar year when they are allowed to exceed the concentration for ammonia.

Odor

The study group was unable to reach consensus on the regulation of odors. Thus, the following two opinions for odor are presented:

Opinion 1:

It is recommended that odor, measured at the residence or public use area, shall not exceed 7:1 dilutions with an exceedence defined as two excessive measurements separated by 4 hours, in any day. It is recommended that each CAFO have up to seven days (with 48 hour notice) each calendar year when they are allowed to exceed the concentration for odor. At the CAFO property line, odor shall not exceed a 15:1 dilution, with an exceedence defined as one excessive two-hour time averaged sample, in any day. It is recommended that each CAFO have up to 14 days (with 48 hour notice) each calendar year when they are allowed to exceed the property line concentration for odor. Exceedence of a CAFO ambient air quality standard should result in regulatory action similar to that which would be required in regulatory action exceedence of a National Ambient Air Quality Standard. The IDNR should be granted the power to develop an implementation plan to reduce the emissions that led to the violation.

Opinion 2:

Odor recommendations are more difficult to establish because studies relating health impacts to odor exposure have not measured odor concentrations. However, odor concentrations related to annoyance impacts have been established. Measurements for odor should be taken at a residence or public use area. Using sampling events at the source, the frequency, duration, and concentration of exposure to odor at the residence can be modeled using tools currently available, thereby avoiding extensive monitoring.

Polls indicate that residents are willing to tolerate nuisance odors for only up to a reasonable amount of time (see Iowa Rural Life Poll, Chapter 7 in the full report). Thus, the reported odor concentration represents tolerable continuous exposure, above which, concentrations are tolerated only in relation to their frequency and duration. An odor concentration of 7:1 dilutions at a residence is a tolerable odor providing it is not exceeded for periods that extend beyond that considered reasonable.

Response to Question 4

Question 4: What do you think should be done to address any other emerging issues with respect to industrial CFOs in Iowa?

There are other important emerging issues surrounding the intensification of livestock production that extend beyond concerns over air emissions. These include concerns about water quality, the health of CAFO workers, socioeconomic impacts in rural communities, and the emergence of microorganisms resistant to antibiotics used in human and veterinary medicine. There are also concerns about the emission of greenhouse gases from CAFO sites. The effects of siting large CAFOs in or near communities should be recognized and used in making informed decisions on permitting facilities. There is a need to evaluate plans for controlling livestock epidemics and for proper disposal of carcasses in the event of an outbreak. Recent events in Europe associated with foot and mouth disease, plus renewed concerns over agricultural bioterrorism highlight this need. Lastly, the study group makes recommendations regarding the formation of a science advisory panel to advise the IDNR on agricultural and environmental health issues. Each of these issues is further described below.

Some issues discussed in this section may be outside the purview of the IDNR, but all are congruent with science-based conclusions in the body of the report. Some are appropriately addressed by other state or federal agencies, and some can only be addressed through a combination of related public policies.

Water Quality

Water quality is a major issue concerning CAFOs. Concerns include: 1) leakage or rupture of lagoons (both lined and unlined); and 2) runoff from agricultural fields where animal waste has been improperly applied. Nonpoint discharges may result in surface runoff with high concentrations of ammonia, biochemical oxygen demand (BOD), total and fecal coliform bacteria, total suspended solids, and phosphorus which can cause low dissolved oxygen in streams. Ecosystem impacts may include fish kills, changes in the natural food webs, algae growth, and losses of biological diversity in stream habitat. Both the structure and function of aquatic ecosystems can be impaired. Impacts may include increased cost for drinking water treatment of surface water supplies, reduced harvest of fish and shellfish, closed bathing beaches due to fecal coliforms, and loss of aesthetic beauty of Iowa's waterways.

Recently, Iowa has experienced an increase in the number of CAFOs as well as a greater density of animals per operation. Many larger operations are not self-sufficient in grain production and purchase feed from other sources. Therefore, applicators must follow additional application guidelines established by legislation and rules. While some study group members believe manure should never be applied to frozen ground or steep slopes, others recommend that manure application on steep slopes and frozen ground follow guidelines established by USDA Natural Resources Conservation Service "Iowa Nutrient Management Standard 590". In addition, large producers are required to file manure management plans with the IDNR.

Study group members reached consensus that as operations become more numerous and concentrated on limited land bases, there is an increased risk for deterioration of water quality. All members believe that if producers do not follow their manure management plans, the chance for runoff of nutrients and bacteria is increased. In addition, some members felt more strongly on this issue, stating that it is not possible to apply manure at high areal loading rates without runoff of nutrients and bacteria because one cannot foresee intense rainfall events. One cannot assume that manure can always be safely applied to land without a potential for runoff. These members feel the present system of CAFO production disposes of too much manure in too small an area exposed to uncontrolled meteorological conditions to realistically expect acceptable water quality.

Wastes that are stored in lagoons or earthen waste storage structures have a potential for spills and/or groundwater contamination if existing standards are not met. National Pollutant Discharge Elimination System (NPDES) permits are required for large (>1000 animal units) open feedlots which allow discharge only in the event of a 25-year, 24-hour storm. Totally roofed CAFOs are not allowed to discharge into surface waters, and therefore do not require NPDES permits. This is in contrast to small Iowa towns, all of which are required to have NPDES permits and meet effluent discharge requirements.

Occupational Health

The occupational health problems for those who work inside CAFOs have been well recognized since 1977. At least 25 percent of workers in swine CAFOs have been reported to have current respiratory health problems. Recommended maximum exposure levels designed to protect worker health have been defined (See Chapter 6.3). It is apparent that current Occupational Safety and Health Administration (OSHA) limits are not protective of CAFO worker health because a number of hazardous contaminants are not regulated. Importantly, OSHA has not promulgated any Permissible Exposure Limits specifically to protect the health of livestock production workers.

There are several important regulatory problems that have interfered with the protection of workers in CAFOs. Most of the large livestock and poultry producers have not been regulated by OSHA, even though they may have more than 10 employees and are subject to OSHA regulations. The specialization of livestock production has led to increased cumulative exposure, as workers may spend as much as 70 hours per week in these buildings. There is a need to establish exposure standards that protect workers for these extended work schedules. There is enough information to protect workers' health if recognized workplace management procedures are adopted. It is recommended that the livestock-producing industries institute comprehensive worker health protection programs.

Antibiotic Resistance

Antibiotic resistance is a health threat of great concern. Recent documents from the World Health Organization (2000), the Centers for Disease Control, and other health agencies have placed a high priority on the understanding and control of antibiotic resistance (Interagency Task Force On Antimicrobial Resistance, 2000; Tenover and Hughes, 1995). It is clear that certain antibiotic use practices in human medicine have contributed to resistance. Agricultural antibiotic use practices have also been targeted as contributing to this serious problem (Witte, 1998). In particular, the subtherapeutic use of antibiotics in food producing animals has been identified by public health officials as the key factor in the development of resistance among foodborne pathogens (Gorbach, 2001).

Antibiotic resistant organisms or the resistance genes responsible can be spread from agricultural settings into human populations through a variety of mechanisms. Ingestion of contaminated food products, especially animal-derived foods including meat and dairy products, has been linked to spread of antibiotic resistant organisms (Mead et al., 1999). Direct contact between colonized or infected animals and farm workers has also been associated with the acquisition of resistant organisms in humans (Levy et al, 1976).

Various studies have demonstrated that continued use of antibiotics in feedstuffs provides conditions favorable to the selection of resistant strains of bacteria in food animals and their environment (Chee-Sanford et al., 2001; Zahn, Anhalt, & Boyd, 2001). Yet the threats for emergence of resistant strains of bacteria through subtherapeutic use of antibiotics in livestock applies wherever these practices occur; the threat is not restricted to CAFOs. Selection pressure may be enhanced by: (1) the long-term use of antibiotics in animals having endemic subclinical infections; (2) poor environmental hygiene; and (3) management practices that allow for the introduction of naïve, susceptible animals or the movement of carrier animals into a naïve herd. This latter practice allows for the continuous passage of resistant bacteria among susceptible animals. Over the past decade, increasing numbers of organisms isolated from food animals or meat products demonstrate resistance to antibiotics including penicillins, tetracycline, sulfamethoxazole, streptomycin and other compounds (Aarestrup et al, 1998; Centers for Disease Control and Prevention, 1999; Molbak et al, 1999; Smith et al., 1999; Threlfall et al., 1996; White et al., 2001).

Antibiotics are critically important in human and veterinary medicine, and in the current context, food animal production. Organisms resistant to all classes of available antimicrobial agents have been identified in human medicine and the incidence of community acquired highly drug resistant organisms is increasing (Neu, 1992). No new classes of antimicrobial agents will be available in the foreseeable future. It is critical that the appropriate state and federal agencies and the research community in the United States take a leading role in defining the risks associated with different antibiotic use practices and develop strategies to improve our antibiotic stewardship both in human and agricultural settings (American Medical Association, 2001).

Greenhouse Gas Emissions

Regarding air pollution, air permits are not required for emissions from CAFOs, so there is not a good method to quantify their inputs. However, emissions of particulate matter, sulfur compounds, and nitrogen oxides are believed to be a very minor portion of Iowa's total emissions. CAFO emissions of these pollutants are small compared to emissions from stationary sources (power plants and industry) and mobile sources (automobiles and truck diesel). Greenhouse gas emissions from CAFOs are significant for methane. On a radiative basis (greenhouse gas impacts), methane is about 10-15% of the total greenhouse gas produced in Iowa, and methane from manure management is about 25% of the total (approximately 3% of total greenhouse gas estimated in Ney et al., 1996). The Iowa Greenhouse Gas Action Plan calls for capture of methane at large feed lots (Ney et al., 1996). Nitrous oxide emissions from manure management at CAFOs is a small contribution, and the emissions of carbon dioxide from CAFOs are a negligible portion of the state's CO_2 emissions.

Community and Socioeconomic Impacts

A number of important community and socioeconomic issues have developed with the emergence of CAFOs, as described in Chapter 7. Research has explored some of these issues, and posed and evaluated alternatives, including some alternatives for livestock production. To a significant extent, these issues are tied to overall changes in agriculture and rural life in America. Importantly, these issues are complex and generally outside the purview of the IDNR.

These issues include the concern about increased concentration of control of livestock supply chains, lack of public price discovery, and loss of family farmers' control of production. Another concern is decline in local economic activity and increases in purchases of some animal production inputs from outside the local area, as CAFOs increase in size and number. This is a complex issue since we must estimate what purchases would have been made had the structure remained the same. Of equal importance is the fact that decision-making on questions that matter at the local level are increasingly more centralized with the growth of corporate CAFOs.

Devaluation of property near hog CAFOs and related legal challenges are documented. Studies in Michigan, North Carolina, and Missouri found that the value of real estate close to CAFOs tended to fall. These and other data show that CAFOs are defined by present and potential neighbors as at least a nuisance.

Studies showing a decline in neighborliness, or community social capital, have been conducted in Iowa, North Carolina, Minnesota, and Missouri. This decline was measured by diminished opportunities to socialize, lack of trust, increased community conflict, and related variables in communities where CAFOs are concentrated.

A more diverse livestock sector that was able to remain competitive and responded to increasingly differentiated consumer preferences would likely result in greater environmental (Donham, 2000), social (Wright, et al., 2001), and economic sustainability of rural areas than one dominated by large-scale CAFOs. Policies that encourage more diverse livestock/crop farms, particularly those using sustainable production systems, could also reduce the regulatory burden of the IDNR and other agencies.

The most clearly recognizable socioeconomic issue for CAFOs that impinges on the IDNR's responsibilities is what CAFOs may do to aquatic, wildlife, and aesthetic qualities of living in Iowa, as well as tourism in Iowa. If air and water quality is compromised, the interest of persons and businesses considering relocation to Iowa will be lessened. A compromised environment could have an economic impact on tourism by keeping Iowa a low priority destination for visitors as well as driving fishing and hunting activity away from Iowa and toward less challenged environments.

Livestock Epidemic and Disposal Issues

The current state plan for Foot and Mouth Disease (FMD) in Iowa is multi-agency and is called the Foot and Mouth Disease Response and Recovery Plan. As part of its responsibilities in the state plan, the IDNR has developed the FMD Carcass Disposal Plan. Burial and composting are given high priority compared to burning, in order to reduce air pollution consequences. However, the potential impacts of a FMD epidemic like that of last year in the United Kingdom and Europe should be evaluated to assess if the current plans are sufficient for isolation of pathogens and destruction of carcasses. In addition, these plans should be evaluated for other pathogens, including bioterrorist introduction of anthrax and other potential agents of agricultural bioterrorism.

Formation of a Science Advisory Panel

To enhance the effectiveness of responses to emerging issues, the study group recommends formation of a science advisory panel to contract with the IDNR on agricultural and environmental issues. The University of Iowa and Iowa State University participants have found the current review of scientific literature on CAFOs and the ensuing discussions to be very useful. University faculty could continue in a more general role as a scientific advisory panel. This would provide the opportunity to develop closer collaboration and planning in a prospective manner. The partnership of the IDNR and other appropriate state agencies with a continuing advisory group of specialists in the sciences germane to agricultural, environmental, and public health issues would strengthen Iowa's ability to plan for prevention or remediation of emerging problems in a thoughtful and positive manner with sufficient lead-time to engage the needed resources and evaluation. A science advisory panel could suggest areas for needed research to better resolve or control the factors related to emerging issues. The panel could recommend consultants, establish standard operating procedures for resolving questions, and be prepared with the necessary background, literature resources and ongoing discussion to support science-based advice as needed by the IDNR or other agencies in Iowa.

Response to Question 5

Question 5: Finally, I am seeking your recommendations regarding available methods of reducing or minimizing the emissions from CFOs and the impact of those emissions on the ambient air surrounding sites.

Emissions from CAFOs originate from three primary sources: (1) air emissions from housing units; (2) air emissions from manure storage facilities, and (3) air emissions during and following land application events. Documented emission reduction strategies exist for all three of these sources. Some of the documented strategies are more effective than others and some are more economical than others, however, economical strategies exist for dealing with emissions from all three sources.

<u>Housing Unit Air Emissions</u>

Housing unit air emissions ultimately are carried out with the ventilation air exhausted from buildings. Emissions originate from the feeding floor itself, where deposited manure and urine decompose anaerobically resulting in airborne gases and particulates from dried fecal material. In addition, emissions originate from under-floor manure storage in slatted systems and from bedding pack in deep-bedded systems. Studies have shown that, in slatted-floor housing systems, the emission contribution from the feeding floor itself can exceed 60 percent of the total with the remaining contribution from the under-floor storage compartment. Use of smooth cleanable surfaces along with frequent and complete scraping, and/or frequent flushing of the feeding floor with minimal air exchange between the housing air and the under-floor slurry, is a good strategy for reducing housing unit emissions.

If housing unit emissions are post-processed, (i.e., exhaust ventilation air is treated), additional strategies exist. Scrubbing the ventilation air with biofilters, where the exhausted air is passed through a bed of gas-scrubbing microorganisms, has been shown to reduce ammonia and odor emissions by more than 90 percent. However, effective use of biofilter technology requires simultaneous use of power ventilation. Biofilters are difficult to implement under high ventilation rate situations typical of Iowa summers and, of course, are not useful in naturally ventilated housing systems.

Gases and odors adhere to dust particles. Natural biomass filters such as corn stalks and chopped-straw have been used to capture a portion of the larger dust particles emitted with ventilation air. The evidence on this strategy is still being documented but research to date indicates that about 60 percent of the odor can be reduced using this technique.

Tree barriers are being evaluated for effectiveness in reducing odor and particulates and enhancing mixing and dilution. However, the impact on a large scale relative to livestock or poultry production sites is unknown. Tree barriers surrounding production sites have high aesthetic value.

Storage Unit Air Emissions

Outside manure storage systems can be a source of additional gas emissions. Regardless of whether the storage system is formed concrete, steel-lined, or earthen basin, these open exposures to the atmosphere can result in high emission rates. Emission rates are highly influenced by weather conditions. The most effective and economically feasible strategy for reducing emissions from outside storage units (not including anaerobic lagoons) is accomplished by covering the entire surface area of the storage unit. Research has been conducted on many covering materials, ranging from expensive impermeable covers, to relatively inexpensive chopped-straw covers with a maintained minimum depth of coverage. Inexpensive, chopped-straw cover, with a maintained minimum depth is as effective in reducing emissions as the more expensive covers. However, the key to success with this strategy is maintenance of a minimum depth of straw.

The best method for minimizing odors from anaerobic lagoons is to simply practice good management. It is most important to use adequate dilution water and load at or below design capacity. There has been much discussion recently about the use of anaerobic digesters which can significantly reduce storage odors and generate energy in the form of methane gas.

Air Emissions from Land Applied Manure

Emissions during land application of livestock and poultry manure can be intense if the manure is surface-applied. The majority of total emissions, roughly 80 percent, occur during the first six hours after land application. To significantly reduce emissions of gases and odors during land application, injection or immediate coverage (within 1 hour) is required. Odor reduction is, in turn, dependent upon the degree of soil coverage. Poorly injected manure slurry with little soil coverage is only marginal in effectiveness in reducing gas and odor emissions. To take full benefit of the natural odor absorption capacity of soils, the slurry must be completely covered. The evidence is clear that 85-90 percent emission reduction is possible with complete soil coverage compared to surface application when coverage is delayed for more than 3-6 hours.

Policy Strategies for Long-Term Viability of the Livestock Industry in Iowa

Emission of gases and particulates from livestock and poultry systems is an inevitable outcome requiring special attention. Strategies for emission reduction for all stages of production have been outlined, with most being economically feasible. The strategies outlined previously are documented techniques that have gained fairly widespread acceptance with scientists and engineers working in this area.

A few strategies have been discussed for years. They lack the scientific evidence to document their specific benefits, but nevertheless deserve discussion. The study group is unanimous in the belief that a long-term strategy of better facility siting, setbacks, and landscape considerations, in addition to the implementation of available odor and gas reducing technologies, will benefit both the producer and residents in the community. The study group strongly urges that the following topics receive careful consideration.

Statewide Spatial Planning

Facilities built today, under current siting and setback practices, have a lifetime of roughly 15 years. In the long-term, guidelines should be established based on siting and spatial planning considerations that require siting of new and replaced facilities in accordance with a statewide spatial plan. Some areas of the state are currently over-populated with facilities. A statewide spatial plan, based for example on animal units per acre, would help guide and distribute animals in a manner that takes full advantage of Iowa's soil/nutrient capabilities and minimizes the impacts of air emissions on the community.

Local Siting Guidelines

The study group feels strongly that current siting guidelines are outdated and not reflective of the changing demographics in rural Iowa. Current siting guidelines use a simple distance and size regulation for new facilities. The study group feels that this method of siting is not conducive to the long-term viability of the livestock and poultry industries in Iowa. A strategy that takes into account proposed facility size and type, distance and orientation to surrounding neighbors, local weather patterns, odor control measures, existing recreational and public-use facilities, and other existing production facilities in a community would provide better placement guidance of facilities and contribute positively to spatial planning considerations. Siting models that utilize the above mentioned inputs have been developed, are currently being calibrated, and should be used in community-wide applications.

Aesthetic Considerations for Livestock and Poultry Production Sites

Evidence exists in the literature that foliage (primarily trees) will enhance mixing and capture some of the odor-producing gases and particulates emitted from livestock and poultry production facilities. Currently, research projects are being planned, and some have already been conducted, to test the use of strategically placed tree barriers around production sites. Although evidence documenting odor, gas, and particulate-capture-percentages on a production-size scale is limited, the study group feels strongly that landscape changes such as strategically placed tree lines will positively impact producer/community relationships. This is a researchable area and one that holds promise as a natural, aesthetically pleasing strategy for producers to implement.

Conclusion to Executive Summary

The consensus responses summarized in this Executive Summary provide a science-based summary of this inquiry from the Iowa Department of Natural Resources. The study group recognizes the importance of livestock production and the vital role it plays in the livelihoods of Iowa producers and suppliers and the state's economy. It is, therefore, critically important that science-based policies be developed to sustain livestock production. It is equally vital that such policies protect the public's health, sustain and enhance the communities in which livestock production takes place, and protect and enhance the environment and Iowa's natural resources through sound production practices, environmental controls and the development of a long-range, sustainable, community health and environmentally conscious spatial plan for CAFOS.

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Chapter 2. Industry Structure and Trends in Iowa

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Abstract

Animal production trends in the United States and Iowa are reviewed to illustrate the changes in the animal industry over the past 50 years. Total production from the major industries are presented along with the changes in numbers of producers and average size of production units. Rapid consolidation of the industry is evident in both poultry and swine production systems in Iowa. Cattle numbers continue to decrease in the state.

2.0 Introduction

The structural changes of the animal industry in Iowa and the related concentration trends are very similar to those seen in most industries in the United States. Overall consumption of animal products has either increased or remained stable over the past 20 years while the number of farms producing these products has greatly diminished. These trends are very similar to those seen in other industries such as construction, food processing, banking, general manufacturing, real estate, services and pharmacy. This results in a large increase in the average size of the active farms in Iowa. The number of active farms in Iowa has been reduced from over 200,000 in 1950 to fewer than 100,000 in the late 1990s as seen in Figure 1(6).

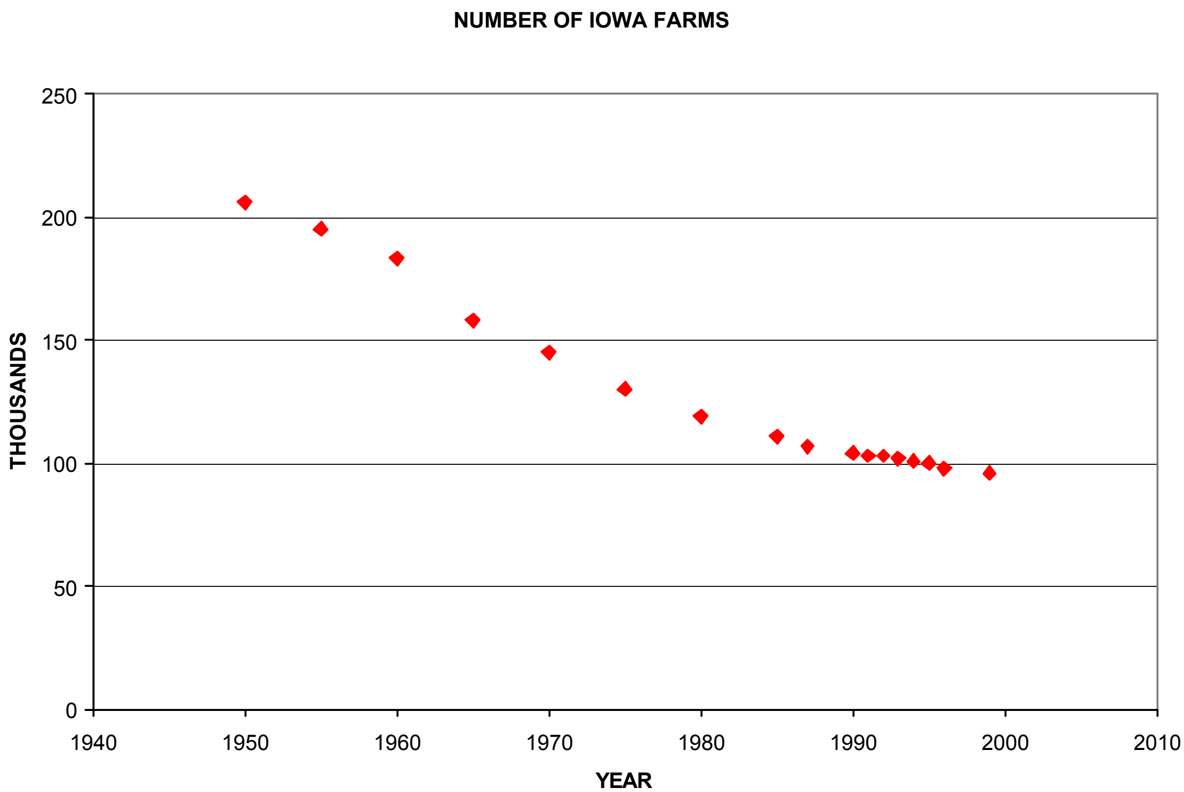

Figure 1. Number of Iowa farms.

A farm in Figure 1 is defined as any operation that sold more than $1,000 in agricultural products. The number of farms owning and operating confined animal feeding operations (CAFOs) will be much less than the above table. However, the trend in declining numbers of farms is obvious. The trend toward fewer farms in Iowa is accompanied by a reduction in the percentage of Iowa farms that have hogs or cattle as a component of their agricultural business. Figure 2 shows that in the early 1960s over 80% of Iowa farms had cattle as part of their operation and 70% had hogs as part of their farming operation. The percent of Iowa farms that included cattle in their farming operation has declined to less than 40% as of 2000, while the percent of Iowa farms that included hogs in their farming operation has declined to approximately 12%.

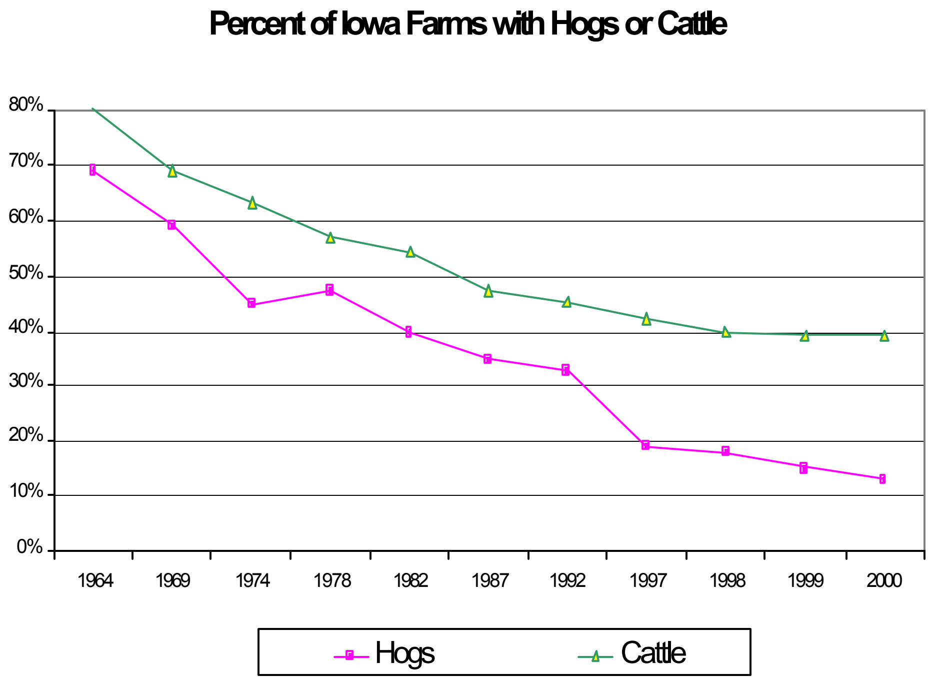

Figure 2. Percent of Iowa farms with hogs or cattle.

2.1 Swine Industry Changes

There are several very distinct trends that can be seen in the U.S. pork industry in the areas of production, processing, environment, vertical integration/coordination and the adoption of technology. The trends being seen in production of pork are shown in the following Table 1. (Lawrence and Grimes, 2001).

Table 1. Changes in USA Pork Production in Number of Farms and Percentage of U.S. Marketings

	Number of Farms			% Marketings	
Herd Size	1997	2000	% Change	1997	2000
1-50	69,460	54,513	-27%	3%	2%
50-250	20,142	17,464	-15%	28%	17%
250-500	1,978	2,627	+33%	10%	10%
500-2500	1,318	2,501	+90%	16%	19%
2500-25,000	127	136	+7%	16%	17%
25,000+	18	20	+11%	27%	35%

The production structure of the U.S. swine industry has changed dramatically in terms of size and location over the past few years. The above table shows the change in numbers of pig-producing farms and marketing percentages over just the past three years. We have recently seen a great reduction in the number of small hog farms (<250 sows) as producers have either gone out of pork production or have increased their herd size to function under the new terms of commodity pork production. The percentage of pigs marketed by this small producer type has decreased from 31% of all pigs marketed in 1997 to only 19% of pigs marketed in 2000. This dropout from production of the smaller farms has been largely picked up by expansion within the corporate pig production segment (greater than 25,000 sows) as the percent of marketing accounted for by this segment has grown by 8%. The mid-level swine production segment has picked up the rest of the fallout from the small producer. Interestingly, the increase in farm numbers (at 250+ sows) coincides with the minimum farm size to implement a weekly farrowing schedule, one of the most basic management technologies. And the herd size that has seen the greatest increase in size (500+ sows) coincides with the minimum herd size needed to market pigs in lot sizes that fit semi-trailer delivery, the most preferred method of delivery by packers (7).

One reason for the increased herd sizes is the greater potential for profit. The following table shows the profitability by herd size recently reported in the United States by Lawrence and Grimes (2001).

Table 2. Profitability by herd size (number of sows) in the U.S. (2000).

Herd Size	Net Profit	Breakeven	Net Loss
1-50	50%	30%	20%
50-250	70%	20%	10%
250-500	78%	13%	9%
500-2500	77%	12%	11%
2500-25,000	90%	5%	5%
25,000+	95%	5%	0%

This table shows the percentage of farms that reported a profit for the year 2000. It is clear that a higher percentage of smaller farms were in the breakeven or net loss return categories when compared to larger farms. The reasons for this are many, but do include those mentioned earlier. Very simply, larger farms are more consistently making a profit when compared to smaller farms.

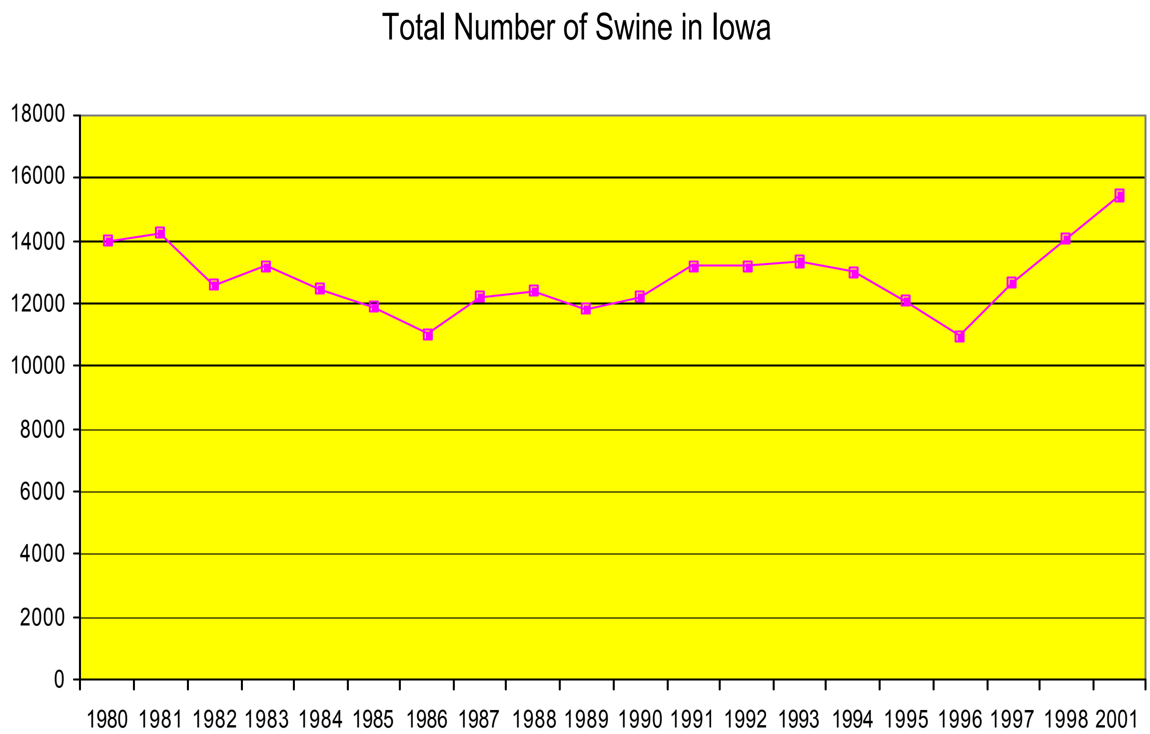

Figure 3 shows the number of swine in Iowa has stayed fairly constant in the past 20 years. However, while the numbers of pigs in Iowa have been somewhat stable, the proportion of hogs that are breeding sows versus market swine has changed markedly, as shown in Figures 4 and 5.

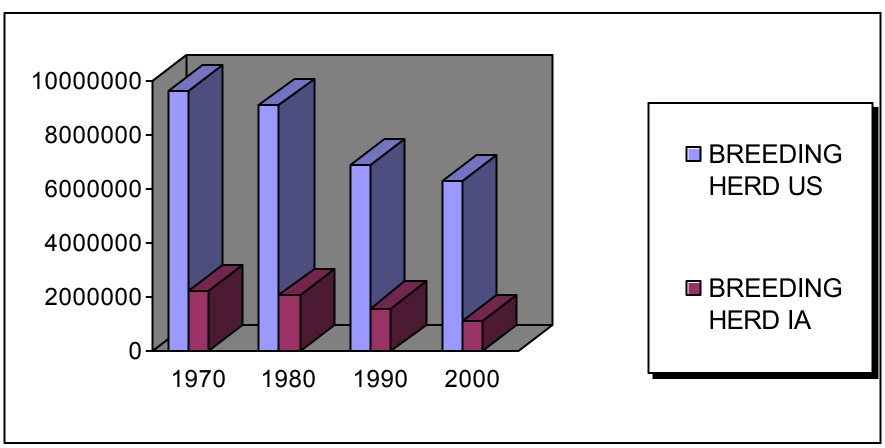

Figure 4. Swine breeding herd in USA and Iowa.

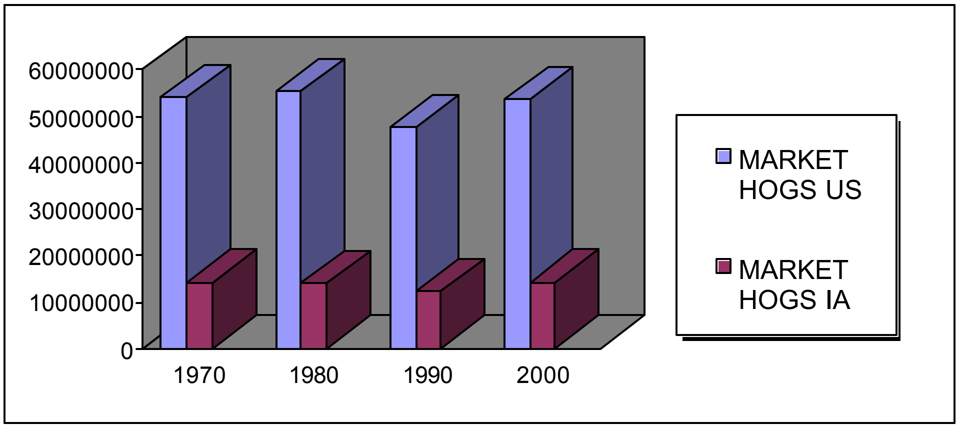

Figure 5. Market hog inventory for USA and Iowa.

Over the past 30 years the breeding herd size in the United States has decreased from just under 10 million sows to just over 6 million sows (Figure 4). However, increases in productivity have allowed total swine production to remain fairly steady. The size of the breeding herd in Iowa has declined from over 2 million sows in 1970 to 1.1 million in 2000. However, the number of market hogs in Iowa has not seen the same decline (Figure 5). One of the primary trends that has potential environmental implications is the trend towards farms having more concentration of hogs. As the number of farms with hogs has declined and the number of total hogs has been more stable, the inevitable result is that the average number of hogs per farm has increased, as shown in Figure 6. As production units increase, there is the associated concentration of waste produced in fewer, larger units. More workers are concentrated to work in the facilities, as well as larger volumes of feed and manure transport. In addition to the increased potential for emissions from these operations compared to smaller units, there is increased traffic volume servicing the unit. Increased traffic volume has the unintended affect of more dust and noise in and around the production unit.

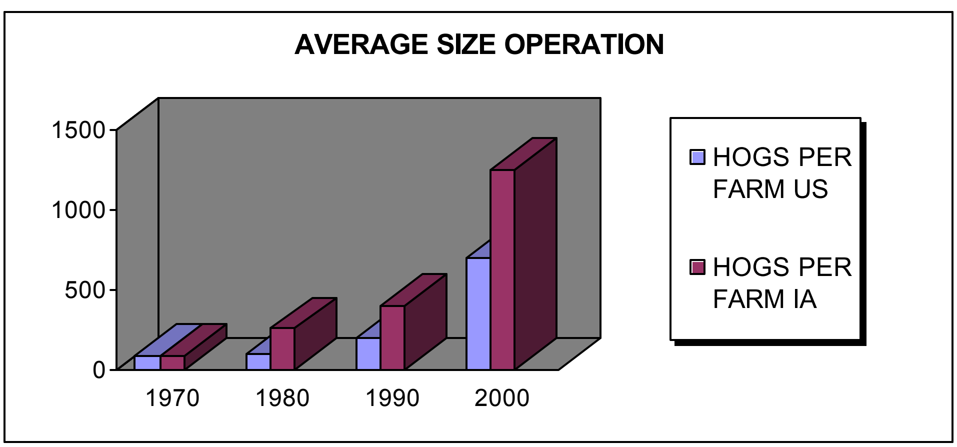

Figure 6. Average swine farm size in USA and Iowa.

The trend in unit size in Iowa mirrors that of the rest of the United States in that the number of hogs per farm has increased greatly over the past 30 years.

2.2 Beef and Dairy Cattle Industry Changes

The total number of cattle on farms in the United States has been somewhat stable over the past 20 years, but has declined over the past 40 years, as shown in Figure 7. However, Iowa has seen a steady decrease in the number of cattle on farms since a peak in the late 1970s to early 1980s. These cattle numbers can be broken down into three primary groups: dairy cattle, beef cows and cattle on feed. The number of dairy cows compared to beef cows on Iowa farms is shown in Figure 8.

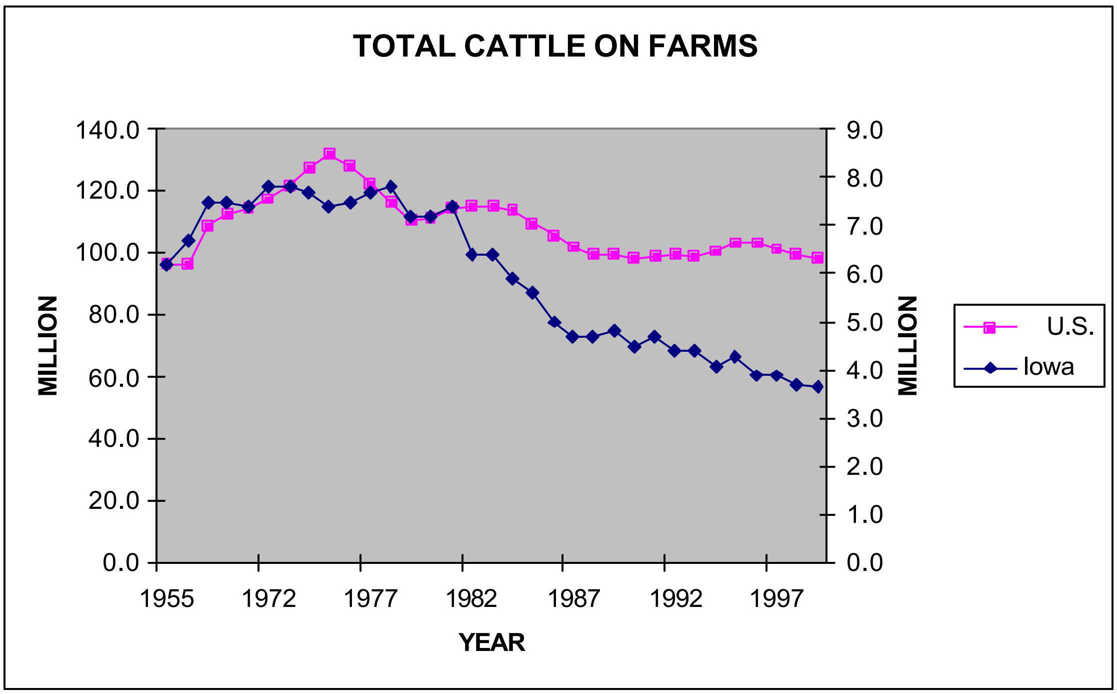

Figure 7. Total cattle on farms in USA and Iowa.

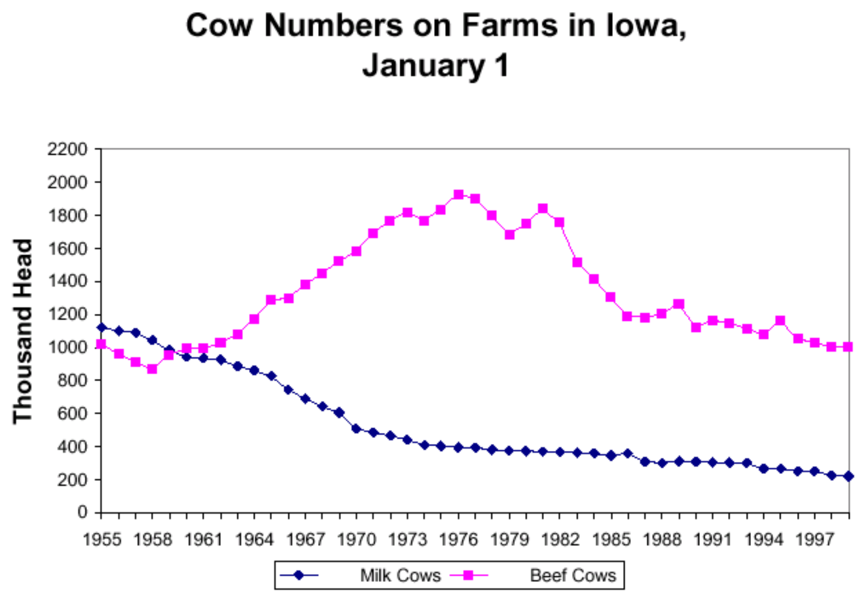

Figure 8. Beef and dairy cow numbers in Iowa.

A slow and steady decline in the number of dairy cows in Iowa has occurred over the past 40 years. Beef cow numbers increased from 1955 until the late 1970s and then began to decline in numbers.

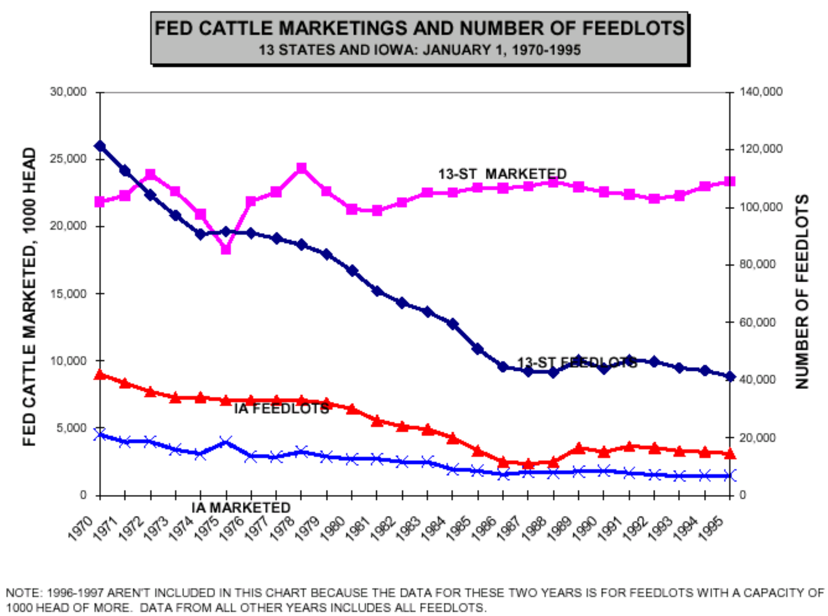

Figure 9. Fed Cattle marketed and number of feedlots for 13 states and Iowa.

The numbers of feedlots and fed cattle in Iowa and 13 states are shown in Figure 9. Iowa has dropped from the number one state in fed cattle production in 1970 to number six in fed cattle marketings. Fed cattle marketings have decreased from 4.7 million annually in 1968 to 1.7 million in 1999. This loss was experienced as fed cattle marketings increased in the Southwest. The number of Iowa feedlots has declined by almost one-half since 1970 while the number of cattle marketed has declined at a somewhat slower rate. This suggests the average size of a beef feedlot in Iowa has increased over the past 30 years.

Poultry Industry Changes

The poultry industry in Iowa consists primarily of egg production and turkey production. The greatest changes in Iowa have been seen in the layer industry as shown in Figure 10. While

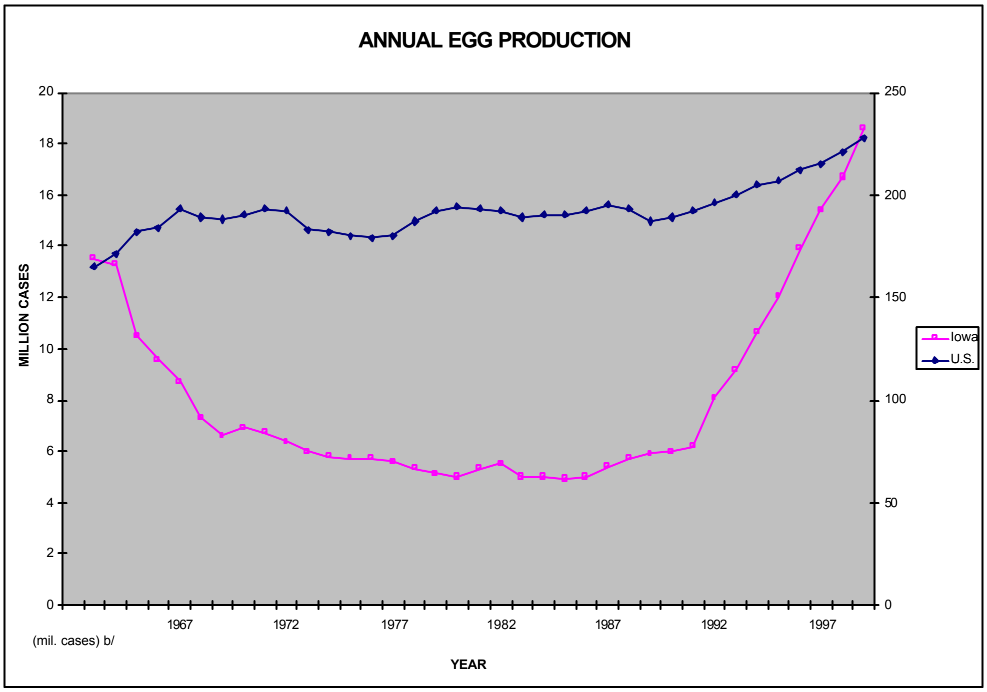

Figure 10. Egg production in USA and Iowa.

Egg production in the US has slowly increased over the past 40 years, the egg production industry in Iowa dropped off dramatically in the 1950s and stayed very small until the 1990s. Since 1990, the egg production industry in Iowa has rapidly grown to the point that Iowa is now number one in the United States in layer numbers.

The trend in turkey production has also been stable, as shown in the Figure 11.

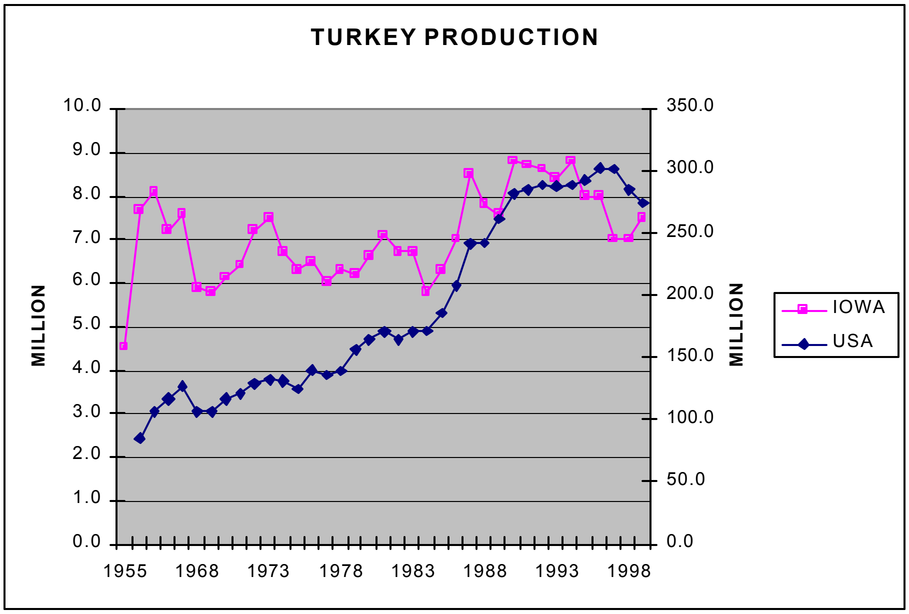

Figure 11. Turkey production in the US and Iowa

2.3 Census of Agriculture Information

Many of the trends shown before in this chapter are documented by Census of Agriculture data comparing changes from 1987 to 1997 presented in tabular form in Table 3. This information is somewhat dated since significant changes have occurred since the last Census of Agriculture in 1997. However, this information shows that major livestock sectors are being restructured in Iowa. As Buttel and Jackson-Smith (1997) point out, this process involves a sharp decline in number of farms, increasing scale, concentration of market power, and increased vertical integration, generally involving greater subordination of the producer to stronger actors in the supply chain. This process can be seen in certain—but not all—livestock sectors in Iowa. During the decade from 1987 to 1997 (date of the most recent U.S. Agricultural Census), hog and poultry production became much more concentrated on fewer farms.

The impact of these changes is greatest in hog production: the number of farms raising hogs halved while the number of hogs and pigs sold per farm more than doubled—and total production grew steadily (by 17% over 10 years). (Recent reports indicate that number of swine farms in Iowa in 2000 is now less than 11,000). The greatest *percentage* shifts occurred in broilers and laying hens. Numbers of layers grew 2.6 times over the period, but the number of layers/farm increased nearly seven-fold. In 1997, Iowa ranked third in the nation in egg production, and has since moved to number one, surpassing Pennsylvania and Ohio. The number of farms engaged in dairying fell by 45 % and the number of dairy cows declined by nearly 25% in the decade, although milk production declined less. The dairies that remain are only modestly larger than before, indicating that the scale revolution in dairying has not greatly affected Iowa, apart from the shift of production to large dairy farms in the West. Stock cattle production declined modestly and beef herd size grew only modestly. The decline in number of farms raising beef cattle (-15%) paralleled the decline in total numbers of farms of all kinds in Iowa (-13.7%). Ruminants are efficient in converting roughage and thus resist complete industrialization. It appears that cattle feedlots did not grow in size, partly because beef CAFOs in Iowa were limited by capital and environmental concerns, while fed cattle production continued the shift to the Great Plains.

Table 3. Changes in Livestock and Poultry Production, Iowa, 1987-1997

Cattle:	**Livestock/farm numbers,1997**	**1987**	**1992**	**1997**
				% Chg 87-97
Farms with cattle/calf sales	38,548			*- 23.7%*
Cattle & calf numbers sold	2,881,122			*- 18.6%*
Cattle and calves sold per farm		**70.0**	**73.6**	**74.7**
Farms with beef cows	27,452			*- 15.0%*
Beef cow numbers (inventory)	1,029,172			*- 8.4%*
Beef cows per farm		**34.8**	**35.5**	**37.5**
Farms with dairy cows	4,208			*- 45.7%*
Dairy cow numbers (inventory)	222,142			*- 24.7%*
Dairy cows per farm		**38.1**	**44.1**	**52.8**
Hogs:				
Farms with hog/pig sales	18,370			*- 52.5%*
Number of hogs/pigs sold	27,495,818			*+ 17.1%*
Hogs and pigs sold per farm		**608**	**787**	**1497**
Poultry:				
Farms with laying hens*	1,892			*- 61.4%*
Inventory of laying hens*	24,876,834			*+ 160.0%*
Layers and pullets* per farm		**1956**	**4770**	**13,148**
Farms selling broilers	519			*- 51.0%*
Broilers sold	6,852,810			*+ 928.9%*
Broilers sold per farm		**628**	**14,110**	**13,203**

* Includes pullets 13 weeks old and over.
Source: U.S. Census of Agriculture: Iowa, 1987, 1992, 1997.
http://ia.profiles.iastate.edu/data/census/county/agcensus.asp?sCounty=19000 Midwest PROfiles, Public Resources Online, Department of Economics, Iowa State University (accessed 12/17/01)

2.4 Iowa DNR Permitted CAFOs

The Iowa Department of Natural Resources has recently estimated current livestock numbers in the state and the numbers of operations large enough to have manure management plans or operation permits. These values are given in table 4.

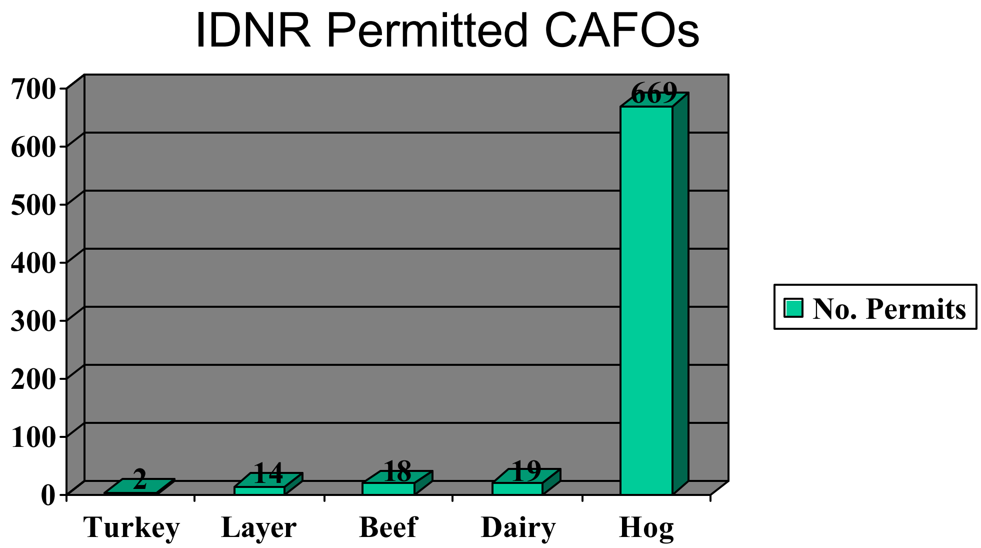

Figure 12. IDNR permitted CAFOs.

The Iowa Department of Natural Resources issues construction permits to confined animal feeding operations that are above a certain threshold of capacity based on live animal weight. The distribution of these permitted CAFOs as of 2001 is shown in Figure 12.

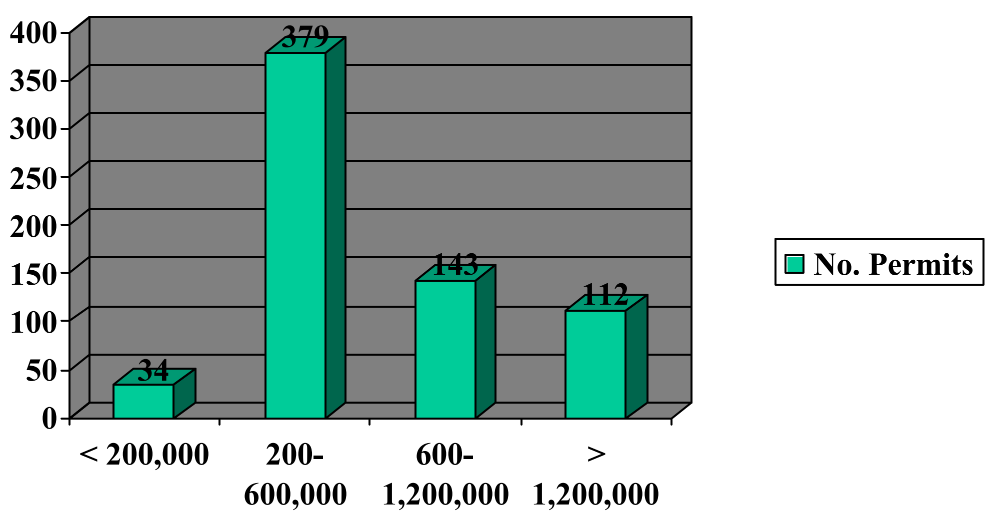

Figure 13. IDNR permitted swine operations by size (weight).

The most prevalent permitted CAFOs in Iowa at the present time are those occupied by hogs. These permitted hog CAFOs are somewhat variable in their size, as shown Figure 13.

Table 4. IDNR animal number estimates.

Livestock Production Numbers

PORK		Animal Unit conversion	
12,900,000 head	Pork produced in facilities large enough to require manure management plans (including permitted operations) = 3,500 operations (85% of IA hogs raised)	5,160,000 au	
2,277,000 head	Pork produced in facilities not required to submit manure management plans, nor required to be permitted.	910,000 au	
15,177,000 head	Total Production		

BEEF		Animal Unit conversion	
365,000 head	Beef produced in facilities containing over 1,000 head	365,000 au	
635,000 head	Beef produced in facilities containing less than 1,000 head	635,000 au	
1,000,000 head	Total Production		
COW/CALF		Animal Unit conversion	
1,200,000 head	Iowa Cattlemen's Association estimation	1,200,000	

DAIRY		Animal Unit conversion	
32,400 head	Dairy animals produced in facilities requiring a manure management plan	45,360 au	
183,600 head	Dairy animals produced in facilities that are not required to have a manure management plan	257,040 au	
216,000 head	Total Production		

TURKEY		Animal Unit conversion	
7,5000,000 head	Estimate production from Iowa Turkey Federation	135,000 au	
7,5000,000 head	Total Production		

POULTRY		Animal Unit conversion	
35,000,000 head	Layers – estimated by Iowa Poultry Association	350,000 au	
5,000,000 head	Broilers – estimated by Iowa Poultry Association	50,000 au	
40,000,000 head	Total Production		

Source: Iowa Department of Natural Resources

Locational Trends in Iowa

Figure 14 illustrates the location of CAFOs in Iowa where there are registered open feedlots or manure management plans have been required under current Iowa regulations. There is a definite concentration of these units in north central, west central, and the extreme northwest corner of the state. Manure management plans are required for all operations with animal weight capacity of over 400,000 pounds of cattle or more than 200,000 pounds for all other species and the operation was constructed or expanded after May 31, 1985.

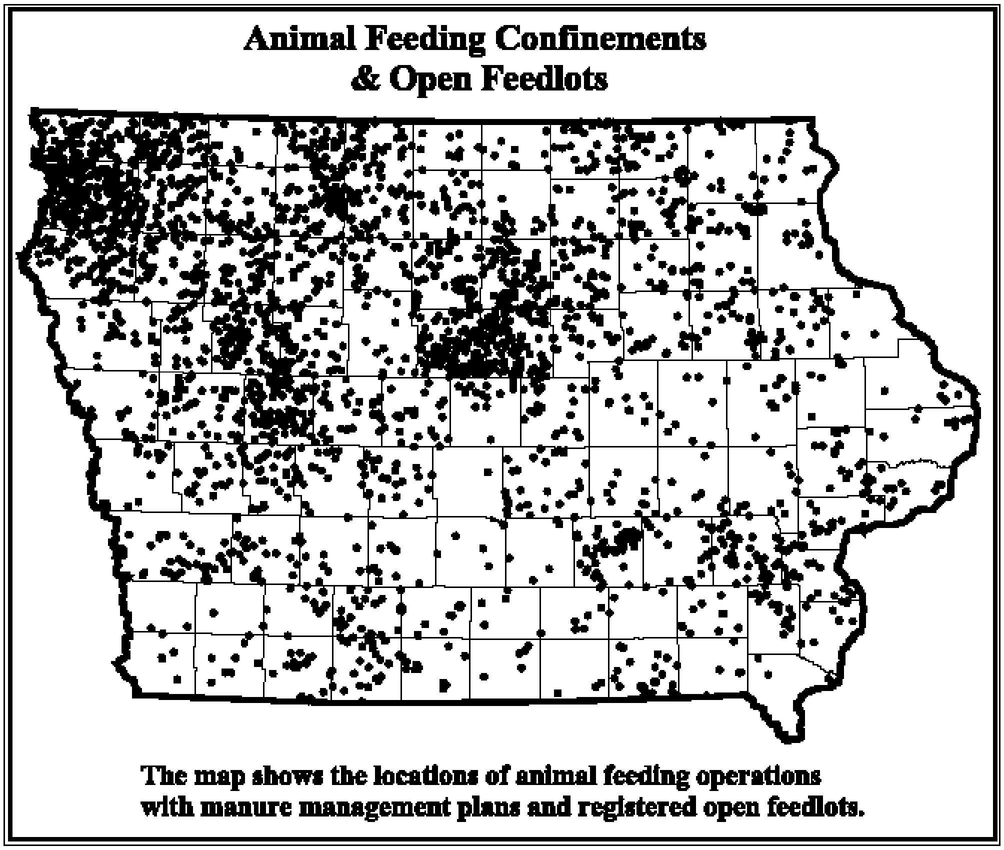

Figure 14. Location of larger animal feeding operations in Iowa(Source IDNR)

Figures 15 and 16 illustrate the changes in concentration of the swine industry in Iowa over the ten-year period from 1987 to 1997. In 1987, there is a relatively uniform distribution of animals across the state whereas in 1997, there are significant concentrations of swine in various parts of the state, especially in northwest and north central Iowa where significant new operations were developed during that time period. The total number of animals has not changed significantly. Therefore some areas have lost swine populations while others have gained significant numbers during the ten-year period. This trend for concentration has continued since 1997.

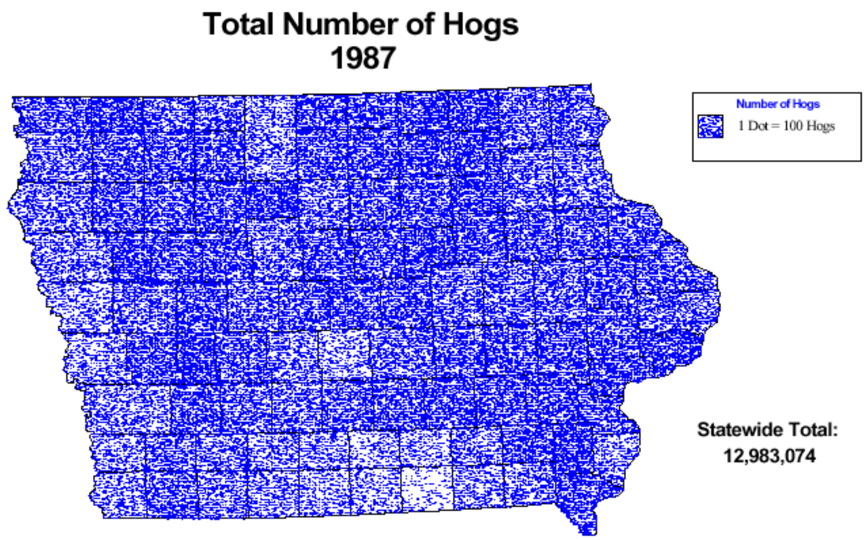

Figure 15. Map of swine numbers in Iowa per county, 1987(Miller, 2002)

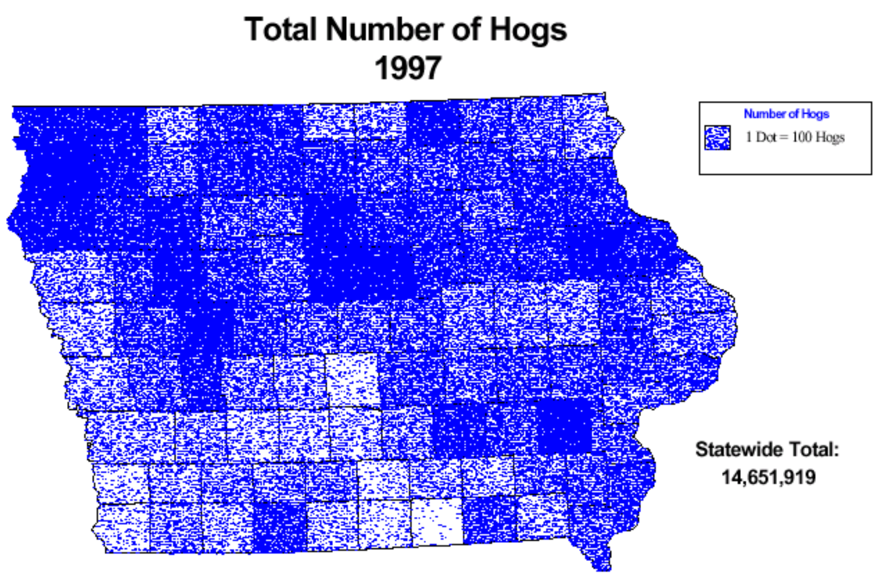

Figure 16. Map of swine numbers in Iowa per county, 1997(Miller, 2002)

Conclusions

It is obvious that animal agriculture in the United States and in Iowa has changed over the past years and will continue to change. There is an increased awareness of environmental and other problems associated with current production systems. This awareness is leading to a rethinking of our current approach to animal production. Changing consumer preferences and lifestyles offer new options and alternatives for animal production. Policies are needed to protect both producer and consumer from being adversely affected.

Many forces impact the livestock industry. The bottom line is that profitability and sustainability are needed. Over time the industry had fewer and larger farms with a higher level of specialization. Access to information is becoming more vital for effective management decisions such as technology adoption. These decisions can be odor management or a host of other production/management issues. Collaborative efforts are increasing. These efforts involve all industry stakeholders, input suppliers, producers, processors, retailers, and policy makers. Information access is increasingly important and cuts across all stakeholders. Among other issues it aids in establishing workable and effective policy decisions.

Animal production is an important part of the Iowa economy but this production needs to be conducted in environmentally sound and sustainable systems to provide the best quality product to consumers while protecting the environment. Iowa can and should remain a leader in production of high quality, environmentally sound animal products.

Odors and emissions from CAFOs have been of concern in Iowa for many years. However, the concentration of animals into larger, more concentrated units has increased the visibility of the potential problems resulting from these major structural changes. The remainder of this report addresses the potential community health impacts of CAFOs.

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Chapter 3.0 Air Quality Issues

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This chapter will describe the agents that emanate from livestock facilities, waste storages and manure application sites associated with livestock production. This will include those agents of concern within barns and air contaminants beyond the barn. These may be on the farm in the vicinity of CAFOs or off the farm at locations or in communities adjacent to CAFOs. This chapter will also briefly describe the measurement approaches and the sources of data for these compounds. The toxic properties of these agents, their emission rates and the concentrations at which they appear are presented in subsequent chapters.

3.1 Sources of data

Air quality data for CAFOs are quite limited. There are relatively few monitoring programs for large-scale livestock production compared to other industries that are regulated. This is further complicated by the fact that the air emissions from CAFOs include a wide array of toxicants including gases, vapors, odoriferous compounds, particulates, and bioaerosols. There are no federally mandated monitoring programs in the United States and only a small number of states have instituted their own monitoring (see Chapter 9). Efforts to institute local controls have generally focused on siting, set backs and zoning rather than compliance with standards for hazardous air pollutants. In Europe, the situation is different. For instance, the Netherlands has established programs based on manure handling practices and for control of emissions from CAFOs. Initially these covered only intensive livestock producers, but now these regulations will extend to all farms. The European Union has issued a number of directives designed to limit emissions of ammonia, methane and odors.

The majority of the monitoring and exposure data available has come from academic researchers interested in characterizing the emissions either for studies of occupational and community health or for studies to address emission rates and efficacy of control approaches. Recently, citizens and citizen groups have begun setting up their own hydrogen sulfide monitoring as a means to provide exposure data to the debate over CAFOs. The swine industry has not engaged in monitoring of air emissions in the United States except when required by court settlements or regulatory action.

3.2 Particulate Matter

Particulate matter associated with CAFOs is composed of fecal matter, feed materials, skin cells, and the products of microbial action on feces and feed (Table 3-1). Components of feed include plant proteins, starches and carbohydrates; feed additives such as vitamins, minerals, amino acids and other supplements; and antibiotics. The most common approaches to measurement of particulate matter emissions are gravimetric sampling, nephelometry, or particle counting. Gravimetric sampling is performed by pre-weighing specialized air sampling filters using a precision microbalance, sampling in the test environment by pulling a measured amount of air through the filter, and then post-weighing the filters and correcting the weight gain for the change in the blanks. This corrected weight change is then divided by the volume of air that was pulled through the filter to determine the airborne dust concentration in mg per cubic meter of air. Different fractions of dust can be selectively sampled by changing the design of the air sampling device and the airflow rate through the device.

When dust is inhaled by humans or animals, a higher proportion of small particles than large particles will travel deep into the lung and be deposited. Thus, environmental health professionals often choose to collect fractions of the total suspended particulates (TSP) to gain more insight into the potential for toxic effects on the lung. Two such categories of smaller fractions are the inhalable dust fraction (50% of particulate mass less than 100 micron [μm]) and the respirable dust fraction (50% of particulate mass less than 3.5 μm). These terms are widely applied in the occupational health literature. Environmental health specialists who study community ambient air pollution more commonly measure two other fractions of particulate matter. These are called PM_{10} and $PM_{2.5}$. PM_{10} refers to particulate matter less than 10 μm in diameter and $PM_{2.5}$ is less than 2.5 μm in diameter. In general, finer particulate fractions contain a higher proportion of anthropogenic dust and lower levels of wind blown soil and plant pollens. Since lung problems associated with CAFOs include airway disease, it is important to consider inhalable particulate fraction and PM_{10}. While gravimetric sampling methods contribute the lion's share of the data on particulate matter concentrations, light scattering and particle counting devices are important as well. These latter methods provide real-time monitoring data and size-specific particle counting necessary for understanding pulmonary deposition and lung burdens.

Bioaerosols are a major component of the particulate matter from CAFOs. Bioaerosols are simply particles of biological origin that are suspended in air. These include bacteria, fungi, fungal and bacterial spores, viruses, mammalian cell debris, products of microorganisms, pollens, and aeroallergens (Table 3-1). Bacterial and fungal bioaerosols may be of infectious or non-infectious species. Bacterial products or components exist as bioaerosols and include endotoxins, exotoxins, peptidoglycans, lipoteichoic acids, and bacterial DNA bearing CpG motifs. Fungal products or components of note include conidia and microconidia, hyphal fragments, mycotoxins and glucans. Settings with very high bioaerosol concentrations include swine, poultry, and dairy confinement buildings; grain and feed mills, grain loading terminals, mushroom production facilities, composting facilities, and sawmills. Typical aerosol sizes for these bioaerosols in indicated in Table 3-2.

Table 3-1 Components of CAFO Particulate Matter

Feed dust	<u>Bioaerosols</u>
plant materials	Microorganisms
proteins	bacteria
starches	bacterial spores
carbohydrates	fungi
feed additives	fungal spores
vitamins	viruses
minerals	
amino acids	Products of bacteria
antibiotics	endotoxins
	exotoxins
Mammalian cell debris	peptidoglycans
	lipoteichoic acids
Aeroallergens	bacterial DNA bearing CpG motifs
plant pollens	Products of fungi
mite fecal allergens	conidia and microconidia
arthropod debris	hyphal fragments
	mycotoxins
	glucans

Sources: Heedeerik, Thorne, and Douwes, 2002; Douwes et al, 2002

Table 3-2 Typical Sizes of Bioaerosols

Bioaerosols	Typical Sizes, µm
Tree/grass pollens	30 - 50
Fungi	20 - 100
Bacteria	2 - 20
Fungal conidia	5 - 15
Bacterial spores	0.5 - 3.0
Viruses	0.01 - 0.05
Droplet nuclei	5 - 10

Source: Thorne and Heederik, 1999b

Genera of bacteria found in air samples from swine barns include the Gram-negative organisms *Enterobacter, Acinetobacter, Enterococcus, Moraxella, Pseudomonas,* and *Escherichia coli,* and the Gram-positive organisms *Enterococcus, Staphylococcus, Streptococcus, Bacillus, Aerococcus,* and *Micrococcus* (Kiekhaefer et al 1995, Cormier et al. 1990). Gram-positive microorganisms (especially Enterococci) represent the majority of bacteria and gram-negative organisms are generally less than 25% of the viable bacteria (Clark et al 1983, Heederik et al 1991). The most commonly found fungi are the mold genera *Aspergillus, Scopulariopsis, Penicillium, Geotrichum, Mucor*, and *Fusarium*. Yeasts found in swine environments include *Candida, Cryptococcus, Toruopsis, Trichosporon, Rhodotorula,* and *Hansenula*. However, variations in housing conditions and feed ingredients can impact the gastric flora of the animals. The concentrations of non-culturable aerobic and anaerobic organisms in the particulate matter in swine barns is known to be 10 to 100-fold higher than the culturable organisms (Lange et al 1997b, Heederik et al 2002). However, the bacterial genera represented in these bioaerosols have not been adequately studied.

Much research has been conducted on methodology for assessment of bioaerosol concentrations in the agricultural environment. This body of work has been recently reviewed (Heederik et al 2002). Methods for assessment of culturable organisms rely on collecting bioaerosols using jet-to-agar samplers or using liquid impingers with dilution plating onto agar (Thorne and Heederik 1999a). Cultures are then allowed to grow in incubators and are enumerated to determine airborne concentrations. Individual colonies may be sub-cultured and identified. Impinger collection fluids may be cultured on a variety of media to quantify mesophilic bacteria, thermophilic bacteria, fungi and selective microbial groups (Thorne et al 1992, Kiekhaefer et al 1995, Cormier et al 1990, Lange et al 1997a, Kullman et al 1998). Since many of the airborne organisms are not culturable, it is necessary to employ non-culture based methods. These include use of direct count methods with DNA staining and epifluorescence microscopy, fluorescent in situ hybridization, and PCR techniques (Thorne et al 1992, Lange et al 1997b, Kullman et al 1998). Significant advances have arisen in the past few years in PCR-based techniques and these will advance the science of bioaerosol sampling in and around swine barns (Heederik et al 2002).

Endotoxin is a lipopolysaccharide (LPS) component of the outer cell wall of Gram-negative bacteria. Since Gm- organisms are ubiquitous in the environment, so is endotoxin. Endotoxin is a potent inflammatory agent that produces systemic effects and lung obstruction, even at low levels of exposure. Livestock confinement units present some of the highest concentrations seen anywhere. The concentration of endotoxin is best determined from liquid impingers or air sampling filters (Duchaine et al 2001) and analyzed using the *Limulus* amebocyte lysate (LAL) assay (Thorne et al 1997, Douwes et al 1995). The LAL bioassay is based on the exquisite sensitivity of an enzymatic clotting cascade in amebocytes taken from the hemolymph of horseshoe crabs (*Limulus polyphemus*) and related species (Thorne 2000). Samples are typically extracted in sterile, pyrogen-free water with 0.05% Tween-20 with continuous shaking. Extracts are centrifuged and supernatants are analyzed using the kinetic chromogenic LAL assay. To provide the highest quality analysis, a twelve-point calibration curve of standard endotoxin from *E. coli* 0111:B4 and four-point endotoxin determination for samples is performed (Thorne 2000). Assay reagent blank wells serve as reference and control. Quality assurance spiking assays are performed to assess matrix interference or enhancement. A number of studies have demonstrated refinements for use of this assay for agricultural environments (Thorne et al 1997, Douwes et al 1995, Gordon et al 1992, Hollander et al 1993, Duchaine et al 2001). Four studies have reported comparisons of endotoxin assay between laboratories (Thorne et al 1997, Reynolds et al 2001, Chun et al 2000, Chun et al 2001).

?(1-3)-glucans are cell wall components of fungi that have been associated with lung inflammation, although at exposure levels well above the levels of endotoxin required for comparable effects (Roy et al 1999). Studies of the past five years have provided evidence that glucans may also be important immunomodulators (Rylander et al 1999, Fogelmark et al 1997). ß(1? 3)-glucans are glucose polymers with variable molecular weight and degree of branching that may appear in triple helix, single helix or random coil structures (Williams 1994). ß(1? 3)-glucans originate from a variety of sources, including fungi, bacteria, and plants (Stone and Clarke 1992). They are water insoluble structural cell wall components of these organisms, but may also be found in extracellular secretions of microbial origin. Glucans may account for up to 60% of the dry weight of the cell wall of fungi, of which the major part is ß(1? 3)-glucan (Klis 1994).

There are currently three principal methods in use for the assay of ? (1-3)-glucans (Heederik et al 2001). Two are based upon the bioactivity of this molecule in the factor G-mediated *Limulus* coagulation pathway. These methods are extremely expensive and not feasible for large field studies. A polyclonal antibody-based immunoassay for ? (1-3)-glucans that is totally independent of the horseshoe crab hemolymph has also been developed (Douwes et al 1996). One laboratory in the Unites States has recently produced several monoclonal antibodies for glucans directed specifically against branched ? (1-3, 1-6)-glucans and ? (1-3)-glucans. This should facilitate future toxicology and exposure assessment studies for glucans.

3.3 Gases and Vapors

Hazardous gases and vapors are emitted from swine barns, lagoons, manure storage piles and from sites of manure land application. These compounds arise from the urine and feces, but especially from microbial degradation of liquid manure in storage or as manure compost. Table 3-3 lists volatile organic compounds; vapors and gases; and odoriferous volatile fatty acids, phenolic compounds and nitrogen-containing compounds. Many of these agents are sensory and respiratory irritants. In combination, they are associated with nasal, sinus, and eye irritation; coughing; wheezing; dyspnea and feelings of malaise (Schenker et al 1998).

While there are real time monitors available for some (e.g. Jerome meters for hydrogen sulfide) most compounds are determined using GC-MS or HPLC-MS methods on air samples collected in impermeable bags or by extraction or purging from collection media. Some vapors, such as ammonia, exist at significant concentrations in both the vapor phase as well as adsorbed to particulate matter. For quantification of these compounds, it is necessary to assay for both the solid and vapor phase. This can be accomplished with annular or honeycomb denuders that collect the vapor phase by reaction with citric acid and the particulate phase by analysis of material deposited on air sampling filters. Of the multitude of compounds in this mixture, those most commonly measured are ammonia, hydrogen sulfide and methane.

Table 3-3. Gases and Vapors Emanated from CAFOs

Volatile Organic Compounds
- acetaldehyde
- acetone
- acetophenon
- acrolein
- benzaldehyde
- benzene
- bis (2-ethylhexyl) phthalate
- 2-butanone
- carbon disulfide
- carbonyl sulfide
- chloroform
- crotonaldehyde
- ethyl acetate
- formaldehyde
- formic acid
- hexane
- isobutyl alcohol
- methanol
- 2-methoxyethanol
- naphthalene
- phenol
- pyridine
- tetrachloroethylene
- toluene
- triethylamine
- xylene

Vapors and gases
- ammonia
- hydrogen sulfide
- dimethyl sulfide
- hydrazine
- sulfur dioxide
- carbon dioxide
- carbon monoxide

Odoriferous microbial compounds
- volatile fatty acids including:
  - butyric and isobutyric acid
  - caproic and isocaproic acid
  - valeric and isovaleric acid
  - propionic and phenylpropionic acid
  - lauric acid
  - acetic and phenylacetic acid

Phenolic compounds
- phenol
- ethyl phenol
- cresols

Nitrogen-containing compounds
- ammonia
- amines
- pyridines
- indole
- skatole
- trimethylamine
- trimethyl pyrazine
- tetramethyl pyrazine

Sources: Banwart and Bremmer 1975, Cole et al 2000, Donham and Popendorf 1985, Hammond and Smith 1981, Hammond et al 1979, Hammond et al 1981, Hammond et al 1989, Hartung 1985, Hartung 1988, Heederik et al 1990, Merkel et al 1979, Minnesota Environmental Quality Board 2001, O'Neill and Phillips 1992, Ritter 1989, Schaefer 1977, Schenker et al 1998, Spoelstra 1980.

3.4 Odors

Odors are one of the most significant community concerns associated with CAFOs. The chemicals that evoke these odors can be an extreme nuisance and can induce adverse health effects with sufficient exposure. The breakdown of feed in the gut of the animals and of the manure after excretion produces odoriferous organic compounds. Bacteria attack organic matter in order to gain energy for life and growth. Bacteria will act on molecules in manure by dehydrogenating these compounds producing reduced oxygen species (Cheremisinoff and Young 1975). Sulfur in proteins is broken down to SO_4 ions. These and organic matter react under the influence of sulfate-reducing bacteria (e.g. *Vibrio desulfuricans*) to produce hydrogen sulfide:

$$SO_4^{-2} + \text{organic matter} \rightarrow S^{-2} + H_2O + CO_2$$

$$S^{-2} + 2H^{+} \rightarrow H_2S$$

In a similar fashion, when oxidized organic compounds are reduced to organic acids, mercaptans, skatoles or indoles they become orders of magnitude more odoriferous.

Some of the most objectionable compounds produced are the organic acids including acetic acid, butyric acids, valeric acids, caproic acids, and propanoic acid; sulfur containing compounds such as hydrogen sulfide and dimethyl sulfide; and nitrogen-containing compounds including ammonia, methyl amines, methyl pyrazines, skatole and indoles. Table 3-4 lists some smells associated with example compounds.

Table 3-4 Examples of Odor Qualities

Chemical Name	Smell
Hydrogen sulfide	rotten eggs
Dimethyl sulfide	rotting vegetables
Butyric, isobutyric acid	rancid butter
Valeric acid	putrid, fecal smell
Isovaleric acid	stinky feet
Skatole	fecal, nauseating
Indole	intense fecal

Source: Cheremisinoff and Young 1975

Methods are well established for characterization of the odor threshold of an air sample. (ASTM Standard Practice E679-91 Determination of Odor and Taste Thresholds by a Forced-Choice Ascending Concentration Series of Limits). Odor thresholds are quantified using an olfactometer and a panel of smellers. These panelists are non-smoking adults that are carefully selected and trained according to ASTM Special Technical Publication 758 Guidelines for Selection and Training of Sensory Panel Members. Eight panelists sniff a two-fold serially-diluted odor sample as it is discharged from one of three ports. The other two ports deliver clean air. The panelist must select which of the randomly assigned ports is the sample and declares whether the selection is based upon recognition, detection, or a guess. The panel then samples the odor at a two-fold higher concentration. Analysis of results from the panel utilizes the triangular forced-choice method in an ascending concentration series.

3.5 Environmental Pollution: Acidifying Emissions and Greenhouse Gases

It is recognized that ammonia emissions from the livestock sector contribute significantly to eutrophication and acidification of the environment. Acidification can put stress on species diversity in the natural environment. Reduction of ammonia emissions requires injection of liquid manure into soil and elimination of surface application. Covering of manure storages and livestock housing that controls emissions are also beneficial. CAFOs are known sources of greenhouse gases such as methane and nitrous oxide. These gases may contribute to global climate change and are the subject of national and international air pollution control strategies. Methane is produced during the digestive process by ruminants while nitrous oxide arises primarily from the microbial degradation of manure.

3.6 Summary

Potentially hazardous air pollutants arise from CAFOs and their associated manure storages and land application sites. These air emissions include coarse and fine particulates, bioaerosols, endotoxins, hydrogen sulfide, ammonia, volatile organic compounds, odoriferous microbial organic compounds, and greenhouse gases. While methods are established for monitoring concentrations of all these compounds, little monitoring has been done in the vicinity of CAFOs. However, occupational health studies have characterized exposures within animal houses. Quantifying odors has relied on olfactometry which uses panels of human subjects to determine odor thresholds. In addition to direct effects on humans, greenhouse gas emissions and volatilization and environmental deposition of ammonia are air quality concerns from CAFOs.

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Chapter 4. Emissions and Community Exposures from CAFOs

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Abstract

This chapter is a review of research and peer-reviewed literature on the emission rates and emission models for dispersion of gases from CAFOs. Emissions originate from the housing ventilation air, manure storage units, and during land application of manure. Refereed publications were sought that identified ammonia, methane, hydrogen sulfide, particulate, bioaerosol, and volatile organic compound (VOCs, including "odor") emission from swine, dairy/cattle, and poultry production systems. The vast majority of published data is related to ammonia emission, and where available, the remaining components were cited and reported. A lack of data exists that reports downwind concentration of gases and particulates from CAFOs as a function of facility type and emission rate.

Dispersion models predict the relationship between concentrations at a receptor and emissions from a source. Appropriate dispersion models for use in predicting concentrations of compounds are reviewed, with reference to current and peer-reviewed literature reports. Dispersion models are generally found to poorly predict absolute concentrations, but are adequate for predicting trends and the expected relationship between reduced emissions and reduced downwind concentrations.

A series of tables is provided at the end of this chapter summarizing the reported emission rates of the above-mentioned components for swine, dairy/cattle, and poultry production systems. An overall summary of reported emission ranges for ammonia, from refereed publications, for ventilation air and manure storages is given below:

Species	Source	Ammonia	Units
Swine	**VA**	**5-311**	**g/AU-day**
	ST	**0.3-144**	**g/m2-day**
Dairy/Cattle	**VA**	**6-43**	**g/AU-day**
	ST	**0.3-18**	**g/m2-day**
Poultry	**VA**	**14-300**	**g/AU-day**
	ST (litter)	**3-5**	**g/m2-day**

VA=ventilation air, ST=storage, AU=animal unit (500 kg)

4.0. Introduction

This chapter summarizes published and refereed research data on gas and particulate emissions from swine, beef cattle, dairy cow, and poultry production systems. Emission refers to the rate at which gases or particulates are being emitted from either the housing unit, manure storage unit, or during land application events. This is in contrast to concentration-only measurements. Emission rates are determined by multiplying the concentration of a component by the volumetric rate at which a component at a given concentration is being emitted. This chapter reports published data, using units reported from each publication. At the end of this chapter, published emission rates are summarized into a table using common units. Research findings are organized by species, individual gases/particulates, and by the particular source, whether building, storage, or land application unit. Each species section is concluded with a summary of research results on published source emission levels versus downwind concentration. At the conclusion of all species-specific discussion, a section describing gas and particulate dispersion models is present. Concluding this chapter is a discussion on how different states address odor emissions and complaints. The terminology used in this chapter is defined below:

<u>Animal Unit:</u> Many emission quantities published are based on a per animal unit (AU) basis. Unless otherwise noted, one AU is equivalent to 500 kg body weight (1,100 lbs).

<u>Bioaerosol:</u> Includes the sub-class of viable particulates that has an associated biological component.

<u>Housing Unit:</u> Any facility used to house livestock or poultry incorporating either a mechanical or natural ventilation system for providing fresh-air exchange.

<u>Inhalable:</u> The class of particulates or bioaerosols having a mean aerodynamic diameter at or below 100 um (micrometers).

<u>Land Application Unit:</u> The process of applying animal manure to the soil.

<u>Manure Storage Unit:</u> Any structure used to store manure, including long-term storage inside the housing unit. Includes above- and below-ground structures.

<u>mg, ug, ng:</u> Respectively, milligrams (10^{-3}), micrograms (10^{-6}), and nanograms (10^{-9}).

<u>Particulate:</u> Includes the class of both inert and viable aerosols. Includes total, inhalable, and respirable fractions.

<u>ppm, ppb:</u> Respectively, parts per million and parts per billion.

<u>Respirable:</u> The class of particulates or bioaerosols having a mean aerodynamic diameter at or below 5 um.

Published emission data is presented in this chapter using original units reported from each citation. Where possible, emission rates from housing unit ventilation air were converted to grams of component per animal unit per day (g/AU-day) and presented in parenthesis after the cited levels. For reported manure storage emission rates, the published levels, where possible, were converted to grams of component per square meter of storage area per day (g/m2-day)

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4.1. Swine System Emissions

4.1.1. Housing Unit Emissions

Ammonia

Aarnink *et al.* (1995) studied the ammonia emission patterns of nursery and finishing pigs raised on partially slatted flooring. They found that for nursery pigs, an average increase of 16 mg NH3/pig-day was measured and this increased to 85 mg NH3/pig-day for finishing pigs. The overall average ammonia emission measured was between 0.70 and 1.20 g NH3/pig-day for nursery pigs (19-33 g NH3/AU-day) and between 5.7 and 5.9 g NH3/pig-day for finishing pigs (42-43 g NH3/AU-day). They found an increase in ammonia emission during the summer months for nursery pigs due to higher ventilation rates but this same trend was not found for finishing pigs. They also found that removing the under-floor stored slurry reduced the ammonia emission by about 20 percent for a period of 10 hours, after which time the ammonia emission regained the pre-removal emission level.

Demmers *et al.* (1999) investigated the exhausted concentrations and emission rates of ammonia from mechanically ventilated swine buildings. They reported ammonia concentrations in a swine finishing house between 12 and 30 mg NH3/m3 with an average ammonia emission rate of 46.9 kg NH3/AU-yr (160 g NH3/AU-day).

Burton and Beauchamp (1986) studied the relationship between outside temperature, ventilation system response, in-house ammonia concentration, and the resulting emission of ammonia from the housing unit. They showed very clearly the inverse relationship of in-house ammonia concentration with outside temperature and the direct relationship of ammonia emission from the housing unit with outside temperature. This trend was attributed to the increased ventilation rates required during the summer to control inside climate temperatures for the housed animals. They summarized results over a one-year period and reported the monthly averages. February had the highest in-house concentration at 15 mg NH3-N/liter corresponding to the lowest emission rate at 0.9 kg NH3-N/day. August had the lowest in-house concentration of 4 mg NH3-N/liter and, correspondingly, the highest emission rate of 3.2 kg NH3-N/day, on average.

Ni *et al.* (2000) investigated the exhausted concentrations and emission rates of ammonia in and from a deep-pit swine finishing building with and without the presence of animals and with pits that were roughly half full (130 cm depth, 240 cm depth capacity). They investigated the gas release rates with and without the effect of heating the building through unit space heaters. Without the presence of animals, they measured ammonia concentrations between 6 and 15 ppm with emission rates between 40 and 58 mg NH3/m2-h (5-8 g NH3/AU-day). When the buildings were re-stocked with pigs, exhaust air concentrations of ammonia were on average 15.2 ppm with corresponding emission rates of 233 mg NH3/m2-h (40-50 g NH3/AU-day).

Groot Koerkamp *et al.* (1998) conducted an extensive study of ammonia emissions from swine housing facilities. They investigated both indoor ammonia levels and with simultaneous measurements of building ventilation rates, reported the resulting emission rate. In general, ammonia concentrations varied between 5 and 18 ppm, with average emission rates between 649 and 3751 mg NH3/AU-h (16-90 g NH3/AU-day). A more complete listing of the ammonia emission rates recorded as a function of maturity level and flooring is given below:

Table 4.1. Swine house ammonia emissions (Groot Koerkamp *et al,* 1998)

Species	Flooring Type	Low Average mg/AU-h (g NH3/AU-day)	High Average mg/AU-h (g NH3/AU-day)
Sows	Litter	744 (18)	3248 (78)
Sows	Slats	1049 (25)	1701 (41)
Nursery Pigs	Slats	649 (16)	1526 (37)
Finishing Pigs	Litter	1429 (34)	3751 (90)
Finishing Pigs	Slats	2076 (50)	2592 (62)

Hinz and Linke (1998) investigated the indoor concentrations and emissions of ammonia from a mechanically ventilated swine finishing facility during a grow-out period where pigs ranged between 25 and 100 kg. Interior ammonia concentrations during the grow-out varied from 10 to 35 ppm and these were inversely proportional to outside temperature. Emission rate of ammonia varied from 70 g NH3/hr (38 kg average pig weight) to 210 g NH3/hr (83 kg average pig weight) resulting in an average ammonia emission rate of 66 g NH3/AU-day.

Zahn *et al.* (2001b) studied the ammonia emission rate from both deep-pit and pull-plug swine finishing facilities during summer periods. He found that the ammonia emission rates were very similar for these two facility types and grouped the emission data into an overall average of 66 ng NH3/cm2-s (311 g NH3/AU-day).

Zhu *et al.* (2000) studied the daily variations in ammonia emissions from various mechanically and naturally ventilated swine housing systems. For a mechanically ventilated swine gestation facility, they measured internal ammonia concentrations between 9 and 15 ppm, with emission rates consistent at about 5 ug NH3/m2-s (2.2 g NH3/AU-day). For a mechanically ventilated farrowing facility, they measured internal ammonia concentrations between 3 and 5 ppm, with emission rates ranging between 20 and 55 ug NH3/m2-s (15-42 g NH3/AU-day). For a mechanically ventilated nursery facility, they measured internal ammonia concentrations between 2 and 5 ppm, with emission rates ranging between 20 and 140 ug NH3/m2-s (23-160 g NH3/AU-day). For a mechanically ventilated finishing facility, they measured internal ammonia concentrations between 4 and 8 ppm, with emission rates ranging between 20 and 55 ug NH3/m2-s (10-26 g NH3/AU-day). For a naturally ventilated finishing facility with pit exhaust fans, they measured internal ammonia concentrations between 7 and 15 ppm, with emission rates ranging between 60 and 170 ug NH3/m2-s (28-80 g NH3/AU-day).

Osada *et al.* (1998) investigated ammonia emission from a swine finisher over an eight week period comparing under-floor stored manure (reference) and under-floor manure removed weekly (treatment). They reported only slight differences in ammonia emission rates with the reference at 11.8 kg NH3/AU-yr (32 g NH3/AU-day) and the treatment at 11.0 kg NH3/AU-yr (30 g NH3/AU-day).

Hartung *et al.* (2001) investigated the effect of a mature and new biofilter on the ammonia emission rate from a swine finisher's ventilation air. They found that with an ammonia load from the ventilation air averaging 4475 mg NH3/m3-h, a reduction in ammonia emission of 15 to 36 percent was measured. This level of ammonia emission reduction was found to be highly dependent on airflow rate and therefore the retention time within the biofilter medium. For the biofilter tested, an airflow rate of 4000 m3/h through the filter bed resulted in a 60 percent ammonia emission reduction and this dropped to zero percent at an airflow rate of about 9000 m3/h.

Methane

Zahn *et al.* (2001b) studied the methane emission rate from both deep-pit and pull-plug swine finishing facilities during summer periods. He found that the methane emission rates were very similar for these two facility types and grouped the emission data into an overall average of 34 ng CH_4/cm^2-s (160 g CH_4/AU-day).

Osada *et al.* (1998) investigated methane emission from a swine finisher over an eight week period comparing under-floor stored manure (reference) and under-floor manure removed weekly (treatment). They reported only slight differences in methane emission rates with the reference at 19.7 kg CH_4/AU-yr (54 g CH_4/AU-day) and the treatment at 17.5 kg CH_4/AU-yr (48 g CH_4/AU-day).

Hydrogen Sulfide

Ni *et al.* (2000) investigated the exhausted concentrations and emission rates of hydrogen sulfide in a deep-pit swine finishing building with and without the presence of animals and with pits that were roughly half full (130 cm depth, 240 cm depth capacity). They investigated the gas release rates with and without the effect of heating the building through unit space heaters. They measured hydrogen sulfide concentrations ranging from 221 to 1492 ppb (parts per billion) with corresponding emission rates between 1.6 and 3.8 mg H_2S/m^2-h (0.22-0.49 g H_2S/AU-day). When the buildings were re-stocked with pigs, exhaust air concentration of hydrogen sulfide averaged 423 ppb with a corresponding emission rate of 9.4 mg H_2S /m^2-h (1.25 g H_2S/AU-day).

Zahn *et al.* (2001b) studied the hydrogen sulfide emission rate from both deep-pit and pull-plug swine finishing facilities during summer periods. He found that the hydrogen sulfide emission rates were very similar for these two facility types and grouped the emission data into an overall average of 0.37 ng H_2S/cm^2-s (1.7 g H_2S/AU-day).

Zhu *et al.* (2000) studied the daily variations in hydrogen sulfide emissions from various mechanically and naturally ventilated swine housing systems. For a mechanically ventilated swine gestation facility, they measured internal hydrogen sulfide concentrations between 500 and 1200 ppb, with emission rates consistent at about 2 ug H_2S/m^2-s (1 g H_2S/AU-day). For a mechanically ventilated farrowing facility, they measured internal hydrogen sulfide concentrations between 200 and 500 ppb, with emission rates consistent at about 5 ug H_2S/m^2-s (4 g H_2S/AU-day). For a mechanically ventilated nursery facility, they measured internal hydrogen sulfide concentrations between 700 and 3400 ppb, with emission rates ranging between 20 and 140 ug H_2S/m^2-s (23-160 g H_2S/AU-day). For a mechanically ventilated finishing facility, they measured internal hydrogen sulfide concentrations between 300 and 600 ppb, with emission rates consistent at about 10 ug H_2S/m^2-s (5 g H_2S/AU-day). For a naturally ventilated finishing facility with pit exhaust fans, they measured internal hydrogen sulfide concentrations between 200 and 400 ppb, with emission rates ranging between 5 and 15 ug H_2S/m^2-s (2-7 g H_2S/AU-day).

Trace Gases

Hartung and Phillips (1994) summarized average measured and reported concentrations of trace gases found in swine housing ventilation air. They did not provide corresponding ventilation rates from which to determine the emission rates of these trace gases. Zahn *et al.* (2001c) measured VOC concentrations in swine finishing facilities incorporating either deep-pit or pull-plug manure handling systems. The trace gases reported from Hartung and Phillips (1994) and those reported from Zahn *et al.* (2001c) are included in the table below. These VOCs are included as an identification of trace gases that might be expected in swine house air (Zahn *et al.*, 2001c list not complete).

Table 4.2. VOC components from swine house ventilation air

Measured In-House Trace Gas	Average Measured Concentration in Air mg/m3 (Hartung and Phillips, 1994)	Average Measured Concentration in Air mg/m3 (Zahn *et al.*, 2001c)**
Fatty Acids		
Acetic acid	0.189	0.281
Propionic acid	0.156	0.126
n-butyric acid	0.318	0.142
I-butyric acid	0.040	0.023
n-valeric acid	0.035	0.043
I-valeric acid	0.049	0.073
n-hexanoic acid	0.010	
I-hexanoic acid	0.004	
Heptanoic acid	0.003	Nd
Octoanoic acid	0.005	
Pelargonic acid	0.004	
Phenols and Indoles		
Phenol	0.023	0.009
p-cresol	0.039	0.085
Indole	0.0011	Nd
Skatole	0.0011	0.0005
Methylamines		
Dimethylamine	2	
Trimethylamine	2.2	
Other Gases		
Acetone	0.33	
Ammonia	8.5	9.6
Hydrogen sulphide	2	0.054
Methane	0.004	5.0
Total (nonmethane VOCs)	1.22	0.81

** list not complete from Zahn *et al.*, 2001c. Total reported nonmethane VOCs from both studies are based on complete listing.

Zahn *et al.* (2001c) did provide simultaneous ventilation rate measurements and therefore was able to assess VOC emissions. For the complete listing of identified VOCs, they reported a VOC emission rate of 89.9 g VOC/system-h.

Zhu *et al.* (2000) studied the daily variations in odor emissions from various mechanically and naturally ventilated swine housing systems. For each housing system studied, a deep-pit manure storage system was used, and each was pit-ventilated. They investigated odor strength from both the pit-fan exhaust-air and the air emitted from inside the building itself. From the pit-fan exhaust, odor strength was highest from the nursery facility ranging between 500 and 2400 OU (dilutions to threshold). The odor strength was lowest from the naturally ventilated finishing facility averaging between 200 and 400 OU. Odor strengths measured from inside the house were significantly lower than those measured from the pit-exhaust fans, with the highest measured levels from the nursery averaging between 250 and 900 OU. The lowest internal odor strength measurements were reported from the gestation facility averaging between 200 and 300 OU. For all five swine facilities monitored, with the exception of the nursery facility, the emission rate of odors ranged from about 5 to 20 OU m3/m2-s. For the nursery facility studied, the odor emission rate was significantly higher than the gestation, farrowing, or finishing facilities, averaging between 8 and 50 OU m3/m2-s (emitted OU strength multiplied by ventilation rate and divided by the floor area of the facility).

Hartung *et al.* (2001) investigated the effect of a mature and new biofilter on the odor emission rate from a swine finisher's ventilation air. They found that with an odor load from the ventilation air averaging 326 OU/m3-h, a reduction in odor emission of 78 to 81 percent was measured. This level of odor emission reduction was not found to be highly dependent on airflow rate. The biofilter experiments conducted resulted in an average air retention time through the filter medium of six seconds.

Particulates
Takai *et al.* (1998) conducted an extensive study of dust emissions from swine housing units. They investigated both indoor concentration levels of dust and the corresponding emission rates. They found significant differences in concentrations and emissions by housing type. The overall average indoor concentrations measured were 2.19 and 0.23 mg/m3 for inhalable and respirable dust concentrations, respectively. The average emission rate from the housing systems monitored were 762 and 85 mg/AU-h for the inhalable and respirable fractions, respectively (18 and 2 g particulates/AU-day). Seasonal effects were found to be significant for the inhalable dust emission rates from the pig houses monitored where, emissions were higher in summer periods, with indoor concentrations higher in winter than summer. There was no similar correlation found for the respirable fraction. A more complete table of results is presented below:

Table 4.3. Swine house particulate matter emissions (Takai et al, 1998)

		Inhalable Dust*		Respirable Dust	
Species	Flooring Type	Low Average Mg/AU-h	High Average mg/AU-h	Low Average mg/AU-h	High Average mg/AU-h
Sows	Litter	144 (3.5)	753 (18)	46 (1.1)	49 (1.2)
Sows	Slats	121 (2.9)	949 (22.8)	13 (0.3)	141 (3.4)
Nursery Pigs	Slats	687 (16.5)	1364 (32.7)	51 (1.2)	122 (2.9)
Finishing Pigs	Litter	561 (13.5)	890 (21.4)	69 (1.7)	73 (1.8)
Finishing Pigs	Slats	418 (10)	895 (21.5)	34 (0.8)	133 (3.2)

* levels in () are g particulates/AU-day.

Bioaerosols
Seedorf *et al.* (1998) conducted a comprehensive study of the emissions of endotoxin and mircroorganisms in the air fraction from pig housing facilities. They found average emission rates of inhalable and respirable endotoxin averaged 51 and 6 ug/AU-h, respectively (1.2 and 0.14 mg/AU-day). The table below gives a more complete listing of the average measured endotoxin emissions from various facilities:

Table 4.4. Swine house endotoxin emissions (Seedorf et al, 1998)

Species	N	Average Inhalable EE	Average Respirable EE	Maximum Inhalable EE	Maximum Respirable EE
Sows	43	37.4 (0.9)	3.7 (0.1)	961.6 (23)	68.7 (1.6)
Nursery Pigs	25	66.6 (1.6)	8.9 (0.2)	347.8 (8.3)	39.8 (1.0)
Finishing Pigs	39	49.8 (1.2)	5.2 (0.1)	299.7 (7.2)	56.1 (1.3)

N=number of buildings sampled, EE=endotoxin emission in ug/AU-h, values in () are mg/AU-day.

Microorganism emissions from these same facilities were categorized into total bacteria, enterobacteriaceae, and fungi. The results from this analysis are given in the table below, with results presented as the Log of the number of colony forming units (cfu) emitted per hour and per AU.

Table 4.5. Swine house microorganism emissions (Seedorf et al, 1998)

Species	N	Total	Enterobacteriacae	Fungi
			Log cfu /AU-h	
Sows	43	7.7	6.0	6.5
Nursery Pigs	25	7.1	6.9	5.8
Finishing Pigs	39	7.6	6.9	6.1

Endotoxin is a hazardous component of airborne particulates in CAFOs. It arises from the degradation of the cell wall of bacteria and is ubiquitous in the agricultural environment. Endotoxin is a potent inflammatory agent that produces systemic effects and lung obstruction, even at very low levels of exposure. It is consistently measured in high concentrations in CAFOs. Nine studies have reported endotoxin exposures in livestock confinement barns using rigorous quantitative methodology as summarized in the table below.

Table 4.6. Swine house interior endotoxin concentrations

Environment	Sites	Range, EU/m³	Mean, EU/m³	Reference
Swine Units	31	--	2400	Donham *et al*,1989
	21	2030 – 11300	4380	Duchaine *et al,* 2001
	350	56 – 15030	920	Preller *et al*,1995
	6	2190 – 24100	8080	Thorne *et al*,1997
	18	210 – 4200	900	Clark *et al*, 1983
Poultry Houses	6	200 – 4500	1360	Thorne *et al*, 1997
	7	1200 – 5000	3600	Clark *et al*, 1983
	25	1300 – 10900	--	Thelin *et al*, 1984
Dairy Barns	85	42 – 34800	742	Kullman *et al*, 1997

There have been few studies that have evaluated offsite transmission of endotoxin from CAFOs. One recent Iowa study (published only as an abstract) investigated in-barn and downwind endotoxin concentrations on 9 occasions over the course of 15 months (Thorne *et al.*, 2001). The study was conducted at one site with three hoop barns housing a mean total of 570 pigs and a conventional confinement site 15 miles away housing 1500 pigs. Grand mean in-barn endotoxin concentrations were 7230 EU/m^3 for the hoop barns and 9950 EU/m^3 for the conventional confinement facilities compared to upwind mean values of 17 EU/m^3 at both sites. Despite these high in-barn levels, there was a sharp diminution of airborne levels downwind of the barns. Endotoxin values 500 feet downwind had reached the 50 EU/m^3 level that is considered a no effect threshold (Dutch Expert Committee on Occupational Standards, 1998). However, it should be recognized that these facilities were small and a larger operation would be expected to produce higher levels of endotoxin. The endotoxin data from this study are summarized in table 4.7.

Table 4.7. Downwind concentrations of endotoxin (Thorne et al, 2001)

	Hoops		Conventional confinement	
	100 ft downwind	500 ft downwind	100 ft downwind	500 ft downwind
Mean, EU/m^3	837	51	155	44
Range, EU/m^3	22 – 3904	20 – 142	18 - 408	-143

4.1.2. Swine Manure Storage Unit Emissions

Ammonia

Aneja *et al.* (2001) studied the ammonia-nitrogen flux from lagoons in North Carolina and found that the emission rates were correlated with lagoon water temperature and aqueous ammonia concentration. They developed a correlation for ammonia nitrogen flux (NH3-N) as ln(NH3-N)=1.0788+0.0406*T+0.0015(NHx) where NH3-N is in ug N/m2-min, T is the lagoon surface temperature in Celsius, and NHx is the total ammonia-nitrogen concentration in mg N/liter.

Aneja *et al.* (2000) studied the seasonal variations in ammonia-nitrogen flux from an anaerobic lagoon in North Carolina and found maximum ammonia emissions during the summer (4017 ug N/m2-min) with minimum levels in the winter (305 ug N/m2-min). Mild weather emissions ranged from 844 (fall) to 1706 (spring) ug N/m2-min. These emission rates were correlated with lagoon surface temperature (measured 15 cm below the lagoon surface) as Log10(NH3-N)=2.1+0.048*T where NH3-N is in ug N/m2-min and T is the lagoon surface temperature in Celsius.

Zahn *et al.* (2001a) studied the efficiency of a polymer-based biocover on the reduction of gas emissions from a single-stage lagoon using micrometeorological techniques. Ammonia flux averaged 18 ng NH3/cm2-s (16 g NH3/m2-day) between summer and fall conditions.

Zahn *et al.* (2001b) studied the ammonia emission rates from 29 swine manure storage systems in Iowa (n=24), Oklahoma (n=2), and North Carolina (n=3). They found that the 29 manure storage systems could be grouped into four main "types", categorized by the total phosphorous and sulfur in the slurry and were able to show distinctions between these 29 storage systems into these four general manure storage "types". These four general types all exhibited similar gas and VOC emission characteristics, allowing grouping of emission results to be made. The four general types were, (1) housing units with long and short term under-floor manure storage configured as deep-pit or pull-plug systems, (2) earthen basin, concrete lined, or above-ground steel tanks, (3) lagoons without photosynthetic blooms, and (4) lagoons with photosynthetic blooms. A summary of the ammonia emission rates from these four types, based on averages within type, are given below:

Type	Description	Ammonia Flux Rate, ng NH3/cm2-s*
I	deep-pit, pull-plug	66 (57)
II	earthen, concrete-lined, steel tanks	167 (144)
III	lagoons without photosynthetic blooms	109 (94)
IV	lagoons with photosynthetic blooms	89 (77)

* values in () are g NH3/m2-day.

Hobbs *et al.* (1999) investigated the emission of odors and gases from stored swine manure with storage times between 0 and 112 days. They reported average daily emissions of ammonia at 4.35 g NH3/m2-day.

Sommer *et al.* (1993) conducted a series of controlled experiments to determine the ammonia emission from stored swine slurry. If the slurry was left uncovered, without allowing a crust to form, the ammonia emission rate was on average 4.3 g NH3-N/m2-day (5.2 g NH3/m2-day). If a crust was allowed to form (between 16-30 cm thick), the ammonia emission reduced to between 0.5 and 1.5 g NH3-N/m2-day (0.6-1.8 g NH3/m2-day). If this slurry was covered with chopped wheat straw at a thickness ranging from 15-23 cm, the ammonia emission was reduced to between 0.2 and 1 g NH3-N/m2-day (0.3-1.2 g NH3/m2-day). If this same slurry was capped with a lid, the ammonia emission reduced to between 0 and 0.3 g NH3-N/m2-day (0-0.4 g NH3/m2-day).

Methane

Zahn *et al.* (2001a) studied the efficiency of a polymer-based biocover on the reduction of gas emissions from a single-stage lagoon using micrometeorological techniques. Methane flux ranged from 134 ng CH4/cm2-s (116 g CH4/m2-day) in summer to 80 ng CH4/cm2-s (69 g CH4/m2-day) in fall.

Hobbs *et al.* (1999) investigated the emission of methane from stored swine manure with time between 0 and 112 days of storage. They reported average daily emissions of methane at 21.4 g CH4/m2-day respectively.

Zahn *et al.* (2001b) studied the methane emission rates from 29 swine manure storage systems, as described previously. A summary of the methane emission rates from the four type classifications, based on averages within type, are given below:

Type	Description	Methane Flux Rate, ng CH4/cm2-s*
I	deep-pit, pull-plug	34 (29)
II	earthen, concrete-lined, steel tanks	178 (154)
III	lagoons without photosynthetic blooms	218 (188)
IV	lagoons with photosynthetic blooms	200 (173)

* values in () are g CH4/m2-day

Hydrogen Sulfide

Zahn *et al.* (2001a) studied the efficiency of a polymer-based biocover on the reduction of gas emissions from a single-stage lagoon using micrometeorological techniques. Hydrogen sulfide flux ranged between 0.73 ng H2S/cm2-s (0.63 g H2S/m2-day) for the summer and 2.11 ng H2S/cm2-s (1.8 g H2S/m2-day) in fall.

Hobbs *et al.* (1999) investigated the emission of odors and gases from stored swine manure with time between 0 and 112 days of storage. They reported average daily hydrogen sulfide emissions of 66.6 g H2S/m2-day.

Zahn *et al.* (2001b) studied the hydrogen sulfide emission rates from 29 swine manure storage systems, as described previously. A summary of the emission rates from the four type classifications, based on averages within type, are given below:

Type	Description	Hydrogen Sulfide Flux Rate, ng H2S/cm2-s*
I	deep-pit, pull-plug	0.37 (0.32)
II	earthen, concrete-lined, steel tanks	1.1 (0.95)
III	lagoons without photosynthetic blooms	0.32 (0.28)
IV	lagoons with photosynthetic blooms	0.24 (0.21)

* values in () are g H2S/m2-day

Arogo *et al.* (2000) investigated the influence of water supply sulfate concentration on the emission of hydrogen sulfide from under-floor stored swine manure. They found a positive correlation between these two parameters in a controlled laboratory condition.

<u>Trace Gases</u>

Hobbs *et al.* (1999) investigated the emission of odors and gases from stored swine manure with storage times between 0 and 112 days. They measured and recorded several volatile organic compounds. Of these measured VOC's, acetic acid had the highest average emission at 1.49 g/m2-day. Phenols on average were emitted at 0.018 g/m2-day with indoles emitted at less than 0.001 g/m2-day. The cumulative odor emission rate was also reported at 802,483 OU/m2-min (odor threshold, OU, multiplied by release rate, m3/min, divided by surface area, m2).

Zahn *et al.* (2001b) studied the total VOC emission rates from 29 swine manure storage systems, as described previously, and summarized the VOC emission rate from the four type classifications, based on averages within type, as:

Type	Description	Total VOC Emission Rate (g VOC/system-h)
I	deep-pit, pull-plug	89.9
II	earthen, concrete-lined, steel tanks	394
III	lagoons without photosynthetic blooms	113.1
IV	lagoons with photosynthetic blooms	14.5

4.1.3. Swine System Emission Rates versus Downwind Concentrations

Seedorf *et al.* (1998) summarized downwind concentrations of endotoxin from swine facilities from work conducted by others:

Downwind Distance (m)	Endotoxin Concentration (ng/m3)
50	60
115	15

Seedorf *et al.* (1998) also summarized research of others on the simultaneous source emission and downwind concentration of microorganisms from swine facilities for both cold and mild weather ventilation conditions:

Downwind Distance (m)	Bacteria Concentration (log cfu/m3)	
	Winter	Spring
0 (source emission)	6.04	5.76
100	3.23	2.97

Zhu *et al.* (2000) studied the downwind concentrations of odor from five dairy/cattle facilities, 18 swine facilities, and five poultry facilities. These facilities ranged widely between manure handling and ventilation methods. Although individual building versus downwind odor strength data was not presented, insight into the downwind odor strength can be gained from this study. At 100 m from any of the sources investigated, the maximum odor strength measured was 270 OU (dilutions to threshold). At 200 m from any of the sources, the maximum odor strength measured was 70 OU, and this reduced to 50 OU at 300 m, and further reduced to 13 OU at 400-500 m downwind. All recordings were taken during daytime hours and the odor strength, reported as OU, was evaluated by personnel trained using a scale developed with n-butanol.

Cattle and Dairy System Emissions

4.1.4. Cattle and Dairy Housing Unit Emissions

Ammonia

Braam *et al.* (1997) investigated the influence of manure handling on the emission of ammonia from dairy cow housing. They investigated two new under-floor manure handling systems incorporating urine gutters with traditional slatted floor systems and found that ammonia emissions from dairy cow housing using slatted floor arrangements could be reduced by as much as 65 percent with special under-floor manure handling. If the under-floor slurry was designed as a sloping floor with a special gutter used to quickly remove urine from the slurry, ammonia emissions were reduced by as much as 50 percent. If in addition to this manure handling system, water was added 12 times per day at a rate of 6 liters/day-cow, the ammonia emission reduction was 65 percent, again relative to a under-floor pit with traditional slats.

Groot Koerkamp *et al.* (1998) conducted an extensive study of ammonia emissions from cattle housing facilities. They investigated both indoor ammonia levels and with simultaneous measurements of building ventilation rates, reported the emission rate. In general, ammonia levels inside the cattle buildings monitored were low, averaging 8 ppm, with average ammonia emission rates ranging between 315 and 1797 mg NH3/AU-h (7.6 and 43 g NH3/AU-day). A more complete listing of the ammonia emissions measured for various species and flooring type are given in the table below:

Table 4.8. Cattle house ventilation air ammonia emission (Groot Koerkamp et al, 1998)

Species	Flooring Type	Low Average mg/AU-h (g NH3/AU-day)	High Average mg/AU-h (g NH3/AU-day)
Dairy Cows	Litter	260 (6.2)	890 (21.4)
Dairy Cows	Cubicles	843 (20)	1769 (42.5)
Beef Cattle	Litter	431 (10.3)	478 (11.5)
Beef Cattle	Slats	371 (9)	900 (21.6)
Calves	Litter	315 (7.6)	1037 (25)
Calves	Slats	1148 (28)	1797 (43)

Jeppsson (1999) studied the influence of bedding material on the ammonia emission rate from cattle housing. Bedding consisting of chopped straw, long straw (ie unchopped), and chopped straw with a peat mixture (2:3 ratio) were tested. Bedding was added to each pen at a rate of 2.7 kg/animal-day over a six month period. They found that pens with chopped straw added to a peat mixture in a 2:3

ratio reduced the ammonia emission by nearly 60 percent relative to pens bedded with long straw. In total, the chopped straw/peat bedding resulted in an average ammonia emission rate of 319 mg/m2-h (8 g NH3/m2-day), while the pens with long straw resulted in an average ammonia emission rate of 747 mg/m2-h (18 g NH3/m2-day). They attributed this reduction to the ability of peat to absorb ammonia, lower the pH level, it's high carbon-to-nitrogen ratio, and it's ability to absorb water. Chopped straw alone, without the addition of peat, reduced the average ammonia emission rate to 547 mg/m2-h (13 g NH3/m2-day). For this study, cattle had access to an unbedded walkway with the reported ammonia emission from this area of the barn averaging 297 mg/m2-h (7.1 g NH3/m2-day).

Kroodsma *et al.* (1993) investigated the contributions of the slurry pit, feeding floor, and the influence of flushing on ammonia emission rates from free-stall dairy facilities. Overall, they reported that from all in-house contributions of ammonia emission, on average results were 1.0 to 1.5 kg NH3/cow-month produced, which equates to between 1344 and 2016 mg NH3/cow-h. They also studied the contributions of ammonia emission from different aspects of the dairy house, as summarized in the table below:

Table 4.9. Cattle house floor and slurry ammonia emission (Kroodsma et al, 1993)

Emission Source	Measured Ammonia Emission Rate mg NH3/m2-h (g NH3/m2-day)
Dirty Slatted Floor	400 (9.6)
Scraped Slatted Floor	380 (9.1)
Unstirred Slurry Below Slats	320 (7.7)
Stirred Slurry Below Slats	290 (7.0)
Dirty Solid Floor	670 (16)
Scraped Solid Floor	620 (15)
Flushed Solid Floor	210 (5)

These results point out the relative equal contributions from the flooring system itself and the stored slurry below the floor. Also, flushing manured floor surfaces can drastically reduce ammonia emissions, as shown. They tested many flushing regimes and found that flushing the floors at 60kPa nozzle pressure, for two seconds every two hours (50 liters water/cow-day), resulted in the best ammonia reduction levels, as reported above for the flushed solid floor.

Swierstra *et al.* (2001) studied the effectiveness of a specially grooved slatted flooring system for free-stall dairy housing with under-floor slurry storage. The flooring system tested had grooved channels with periodic perforations to quickly channel urine from the feeding floor and this was combined with frequent scraping (every two hours) of the slatted flooring to an opening that delivered manure to the under-floor pit area. This opening (to the under-floor pit) was closed during non-scraping events. This method of manure handling was compared with a conventional slatted flooring system based on ammonia emission rates. They consistently found that the ammonia emission rate was reduced by 46 percent compared with the conventional slatted floor system (11.7 g NH3/h vs 21.6 g NH3/h). On a per cow-day basis, these levels correspond to 28.1 g NH3/cow-day and 51.8 g NH3/cow-day. A follow-up field study confirmed this level of ammonia reduction.

Zhu *et al.* (2000) studied the daily variations in ammonia emission from a naturally ventilated dairy housing unit. During one day of monitoring, they measured a consistent 1 ppm of ammonia concentration inside the housing unit with a resulting emission rate averaging 4 ug NH3/m2-s (0.35 g NH3/m2-day).

Elzing and Monteny (1997) studied, in a controlled laboratory setting, the ammonia emission rate from manure and urine fouled slats and from the under-floor storage tank from dairy-cow manure. They found that peak ammonia emissions, from soiled slats covered with fresh manure and urine deposits, had a peak ammonia emission level at about two hours after deposition. The peak ammonia emission rate from the slats was positively correlated with both slat surface temperature and airspeed levels above the slats. They found that the combined slat and under-floor storage unit resulted in 10 g NH3 being emitted after 10 hours of fresh manure deposition on the slats and 12 g NH3 after 20 hours of fresh manure deposition on the slats. During these same periods of 10 and 20 hours, they reported that the contribution of this total ammonia emission from the under-floor storage unit was constant at 3.3 g NH3.

Methane

Kaharabata and Schuepp (2000) investigated emission of methane from dairy cows using a tracer-ratio method. They studied emissions from dairy cow housing and reported measured levels of 542 L CH4/day-cow.

Hydrogen Sulfide

Zhu *et al.* (2000) studied the daily variations in hydrogen sulfide emission from a naturally ventilated dairy housing unit. During one day of monitoring, they measured variations in internal concentrations between 4 and 26 ppb with resulting emission rates averaging roughly 3 ug H2S/m2-s (0.26 g H2S/m2-day).

Trace Gases

Zhu *et al.* (2000) studied the daily variations in odor emissions from a naturally ventilated dairy housing unit. During one day of monitoring, they measured a consistent internal odor strength of 50 OU (dilutions to threshold). The resulting odor emission rate was on average 2 OU m3/m2-s (odor strength, OU, multiplied by the estimated ventilation rate, m3/s, divided by the floor area of the housing unit, m2).

Particulates

Takai *et al.* (1998) conducted an extensive study of dust emissions from cattle housing. They investigated both indoor concentration levels of dust and the emission rates. They found significant differences in concentrations and emissions by housing type. The overall average indoor concentrations measured were 0.38 and 0.07 mg/m3 for inhalable and respirable dust concentrations, respectively. The average emission rate from the cattle housing systems monitored was 145 and 24 mg/AU-h (3.5 and 0.6 g/AU-day) for the inhalable and respirable fractions, respectively. Seasonal differences in concentration and emission of dust for cattle buildings for both inhalable and respirable fractions were not significant. A more complete table of results is presented below:

Table 4.10. Cattle house ventilation air particulate emission (Takai et al, 1998)

		Inhalable Dust*		Respirable Dust	
Species	Flooring Type	Low Average mg/AU-h	High Average mg/AU-h	Low Average mg/AU-h	High Average mg/AU-h
Dairy Cows	Litter	60 (1.4)	142 (3.4)	6 (0.1)	84 ((2.0)
Dairy Cows	Cubicles	21 (0.5)	338 (8.1)	13 (0.3)	54 (1.3)
Beef Cattle	Litter	36 (0.9)	135 (3.2)	6 (0.1)	26 (0.6)
Beef Cattle	Slats	78 (1.9)	144 (3.5)	5 (0.1)	29 (0.7)
Calves	Litter	64 (1.5)	190 (4.6)	14 (0.3)	40 (1.0)
Calves	Slats	63 (1.5)	192 (4.6)	14 (0.3)	22 (0.5)

* values in () are g/AU-day.

Bioaerosols

Seedorf et (1998) conducted a comprehensive study of the emissions of endotoxin and mircroorganisms from cattle housing facilities. They found average emission rates of inhalable and respirable endotoxin in cattle buildings of 9 and 1 ug/AU-h, respectively. The table below gives a more complete listing of the average measured endotoxin emissions from various facilities:

Table 4.11. Cattle house ventilation air endotoxin emission (Seedorf et al, 1998)

		Average	Average	Maximum	Maximum
Species	N	Inhalable EE*	Respirable EE	Inhalable EE	Respirable EE
Cows	31	2.9 (0.07)	0.3 (0.007)	11.4 (0.28)	1.9 (0.05)
Beef	18	3.7 (0.09)	0.6 (0.01)	22.8 (0.55)	9.3 (0.22)
Calves	17	21.4 (0.50)	2.7 (0.06)	90.1 (2.18)	44.8 (1.08)

* EE=endotoxin emission in ug/AU-h. Values in () are mg/AU-day.

Microorganism emissions from the facilities studied in Seedorf *et al.* (1998) were summarized in terms of total bacteria, enterobacteriaceae, and fungi. The results from this analysis are given in the table below, with results presented as the Log of the number of colony forming units (cfu) emitted per hour and per AU.

Table 4.12. Cattle house ventilation air microorganism emission (Seedorf et al, 1998)

		Total	Enterobacteriacae	Fungi
Species	N		Log cfu /AU-h	
Cows	31	6.8	6.2	6.0
Beef	18	6.7	6.2	5.9
Calves	17	7.3	6.1	6.5

4.1.5. Cattle and Dairy Manure Storage Unit Emissions

Ammonia

Kellems *et al.* (1979) conducted experiments to investigate ammonia emission from cattle slurry as the proportions of feces, urine, and water changed. They found clear trends in ammonia emission rates with various proportions. From the urine fraction only, the ammonia emission rate was 426 ug NH3/h, representing the worst-case scenario. From the feces fraction only, 3.2 ug NH3/h was emitted. With a 1:1 ratio of feces and urine, the emission rate of ammonia was 120 ug NH3/h.

Dewes (1999) studied ammonia emission characteristics of liquid and solid cattle manure over the initial 16 days of storage time. Over this short initial time period, solid manure with 15 kg of straw added per animal per day had the highest emission rate of ammonia at 6300 ug NH3-N/h-kg of manure with liquid manure having the lowest emission rate of ammonia at 663 ug NH3-N/h-kg of manure. Projections were made for longer storage periods and conclusions were made that after a storage period of 28 days, the ammonia emission rate would be greatest with manure stored as a liquid. However, it was also concluded that solid manure systems that use heaped piles can result in higher ammonia emission rates versus liquid manure systems since the emitting area for a stored pile is large.

Sommer *et al.* (1993) conducted a series of controlled experiments to determine the ammonia emission from stored cattle slurry. If the slurry was left uncovered, without allowing a crust to form, the ammonia emission rate was on average 4.5 g NH3-N/m2-day (5.5 g NH3/m2-day). If a crust was allowed to form, at 7 cm thickness, the ammonia emission reduced to 1.3 g NH3-N/m2-day (1.6 g NH3/m2-day). If this same slurry was capped with a lid, the ammonia emission reduced to between 0.2 and 0.4 g NH3-N/m2-day (0.25-0.5 g NH3/m2-day).

Methane

Kaharabata and Schuepp (2000) investigated emission of methane from dairy cows using a tracer-ratio method. They studied emissions from the feedlot and reported average emissions of 631 L CH4/day-cow.

Poultry System Emissions

4.1.6. Poultry Housing Unit Emissions

Ammonia

Demmers *et al.* (1999) investigated the exhausted concentrations and emission rates of ammonia from a mechanically ventilated poultry building. They reported ammonia concentrations between 1 and 37 mg/m3. Emission rates of ammonia averaged 18.6 kg NH3/AU-yr (51 g NH3/AU-day).

Wathes *et al.* (1997) studied extensively the emission of ammonia from broiler and layer facilities. They reported average emissions of ammonia at 9.2 g NH3/AU-h (221 g NH3/AU-day). This ammonia emission rate was consistent across both layer and broiler facilities. A complete table of their findings is presented below:

Table 4.13. Poultry house ventilation air ammonia emission (Wathes et al, 1997)

Poultry Type	Season	Average Ammonia Emission g NH3 /AU-h (g NH3/AU-day)
Caged Layers	Winter	8 (192)
Broilers	Winter	9 (216)
Caged Layers	Summer	12.5 (300)
Broilers	Summer	9 (216)

Groot Koerkamp *et al.* (1998) conducted an extensive study of ammonia emissions from poultry housing facilities. They investigated both indoor ammonia levels and with simultaneous measurements of building ventilation rates, reported the emissions. In general, ammonia levels inside the buildings ranged between 5 and 30 ppm with the average emission rate of ammonia between 602 and 10892 mg NH3/AU-h (14 and 261 g NH3/AU-day). A more complete listing of the ammonia emissions measured for various species and flooring types is given in the table below:

Table 4.14. Poultry house ventilation air ammonia emission (Groot Koerkamp et al, 1998)

Species	Flooring Type	Low Average mg/AU-h (g/AU-day)	High Average Mg/AU-h (g/AU-day)
Laying Hens	Litter	7392 (177)	10892 (261)
Laying Hens	Cages	602 (14)	9316 (224)
Broilers	Litter	2208 (53)	8294 (199)

Zhu *et al*. (2000) studied the daily variations in ammonia emission from a mechanically ventilated broiler house using litter bedding. During one day of monitoring, they measured internal concentrations of ammonia between 9 and 13 ppm with a resulting ammonia emission rate averaging between 4 and 20 ug NH3/m2-s (7-33 g NH3/AU-day).

Methane

Wathes *et al*. (1997) studied the emission rate of methane from broiler and layer facilities. They reported average methane emissions of 0.85 g CH4/AU-h (19 g CH4/AU-day) for caged layers and 0.25 g CH4/AU-h (0.6 g CH4/AU-day) for broilers. A summary table of reported methane emissions is given below:

Table 4.15. Poultry house ventilation air methane emission (Wathes et al, 1997)

Poultry Type	Season	Average Methane Emission g CH4 /AU-h (g CH4/ AU-day)
Caged Layers	Winter	0.80 (19)
Broilers	Winter	0.25 (6)
Caged Layers	Summer	0.90 (22)
Broilers	Summer	0.25 (6)

Hydrogen Sulfide

Zhu *et al*. (2000) studied the daily variations in hydrogen sulfide emission from a mechanically ventilated broiler house using litter bedding. During one day of monitoring, they measured internal concentrations of hydrogen sulfide between 40 and 150 ppb with a resulting hydrogen sulfide emission rate averaging less than 2 ug H2S/m2-s (3.3 g H2S/AU-day).

Trace Gases

Misselbrook *et al*. (1993) studied the relationship between odor emission and intensity for broiler house air. They determined a relationship using a 0-6 point intensity scale versus the concentration of odors emitted from broiler houses. The intensity scale used is given below:

Intensity (I)	Description
0	No odor
1	Very faint odor
2	Faint odor
3	Distinct odor
4	Strong odor
5	Very strong odor
6	Extremely strong odor

They found a relationship that described 84 percent of the variability in their data where I=2.35 (Log_{10} C) + 0.30 where C is the dilution to threshold concentration of odor. They further summarized their data to give indications of the odor intensity with the dilution threshold concentration as given below:

Intensity	Broiler House Air Odor Concentration (OU/m3)
0	0-1.2
1	1.2-3.3
2	3.3-8.8
3	8.8-23.4
4	23.4-62.6
5	62.6-167
6	> 167

From this study, and from cited work of others, they concluded that an odor intensity at or below an intensity of 2 (faint odor) may be considered acceptable, which further implies that for broiler house ventilation air the odor concentration should be below about 3.3 OU/m3.

Zhu *et al.* (2000) studied the daily variations in odor emissions from a mechanically ventilated broiler house with litter bedding. During one day of monitoring, they measured a consistent internal odor strength of about 100 OU (dilutions to threshold). The resulting odor emission rate was less than 2 OU m3/m2-s (determined by multiplying the odor threshold, OU, by the ventilation rate, m3/s, and dividing through by the floor surface area of the housing unit, m2).

Particulates

Wathes *et al.* (1997) studied the emission of dust from broiler and layer facilities. They reported average inhalable and respirable dust emissions of 1.0 g/AU-h and 0.17 g/AU-h (24 and 4 g/AU-day) for caged layers and 6.7 g/AU-h and 0.79 g/AU-h (161 and 19 g/AU-day) for broilers, respectively. A summary table of emissions is given below:

Table 4.16. Poultry house ventilation air particulate emission (Wathes et al, 1997)

Poultry Type	Season	Inhalable Dust g/AU-h (g/AU-day)	Respirable Dust g/AU-h (g/AU-day)
Caged Layers	Winter	0.9 (22)	0.24 (5.8)
Broilers	Winter	5.2 (125)	0.60 (14.4)
Caged Layers	Summer	1.1 (26)	0.09 (2.2)
Broilers	Summer	8.2 (197)	0.88 (21.1)

Takai *et al.* (1998) conducted an extensive study of dust emissions from poultry housing facilities. They investigated both indoor concentration levels of dust and the emission rates to the atmosphere. They found significant differences in concentrations and emissions by housing type. The overall average indoor concentration was 3.60 and 0.45 mg/m3 for inhalable and respirable dust concentrations, respectively. The average emission rate from the various poultry housing systems was 3165 and 504 mg/AU-h (76 and 12 g/AU-day) for the inhalable and respirable fractions, respectively. Seasonal effects were found to be significant for the inhalable dust emission rates with emissions higher in summer periods, and indoor concentrations higher in winter than summer. There was no similar correlation found for the respirable fraction. A more complete table of results is presented below:

Table 4.17. Poultry house ventilation air particulate emission (Takai et al, 1998)

		Inhalable Dust*		Respirable Dust	
Species	Flooring Type	Low Average mg/AU-h	High Average mg/AU-h	Low Average mg/AU-h	High Average mg/AU-h
Laying Hens	Cages	398 (9.6)	872 (21)	24 (0.6)	161 (3.9)
Broilers	Litter	1856 (45)	6218 (149)	245 (5.9)	725 (17.4)

* values in () are g/AU-day.

<u>Bioaerosols</u>

Wathes *et al.* (1997) studied the emission of endotoxin from broiler and layer facilities. They reported average emissions of endotoxin between 1 and 45 ug/AU-h (0.024 and 1.1 mg/AU-day) with very strong seasonal effects, with summer emissions 3 to 45 times higher for caged layers and broilers, respectively. A summary table of endotoxin emissions is given below:

Table 4.18. Poultry house ventilation air endotoxin emission (Wathes et al, 1997)

Poultry Type	Season	Average Endotoxin Emission g/AU-h (g/AU-day)
Caged Layers	Winter	10 (240)
Broilers	Winter	< 1 (<24)
Caged Layers	Summer	30 (720)
Broilers	Summer	45 (1080)

Hinz and Linke (1998) investigated the indoor concentration and emission of endotoxin from a naturally ventilated broiler house. Endotoxin was measured in the broiler with reported levels ranging between 0.05 and 0.45 ug/m3 with no apparent seasonal trends, unlike the trends observed for inhalable dust.

Seedorf *et al.* (1998) conducted a comprehensive study of the endotoxin emissions from poultry housing facilities. They found average emission rates of inhalable and respirable endotoxin in poultry facilities, averaging 678 and 43 ug/AU-h (16 and 1 mg/AU-day), respectively. The table below gives a more complete listing of the average measured endotoxin emissions from various facilities:

Table 4.19. Poultry house ventilation air endotoxin emission (Seedorf et al, 1998)

Species	N	Average Inhalable EE	Average Respirable EE	Maximum Inhalable EE	Maximum Respirable EE
Layers	43	538.3 (13)	38.7 (0.9)	5247.1 (127)	342.5 (8.3)
Broilers	19	817.4 (20)	46.7 (1.1)	6836.3 (165)	294.6 (7.1)

EE=endotoxin emission in ug/AU-h. Values in () are mg/AU-day.

Microorganism emissions from the facilities studied were summarized in terms of total bacteria, enterobacteriaceae, and fungi. The results from this analysis are given in the table below, with results presented as the Log of the number of colony forming units (cfu) emitted per hour and per AU.

Table 4.20. Poultry house ventilation air microorganism emission (Seedorf et al, 1998)

Species	N	Total	Enterobacteriacae Log cfu /AU-h	Fungi
Layers	43	7.1	7.1	6.0
Broilers	19	9.5	6.1	7.8

4.1.7. Poultry Manure Storage Unit Emissions

Ammonia

Brewer and Costello (1999) investigated the emission of ammonia from broiler house litter, comparing new bedding consisting of either rice hulls or rice hulls mixed with pine shavings and re-used bedding of the same. On average, new bedding resulted in an average ammonia emission of 149 mg NH3-N/m2-h (4.3 g NH3/m2-day) with a maximum emission of 314 mg NH3-N/m2-h (9.1 g NH3/m2-day). When the bedding was re-used for subsequent grow-out periods, the average ammonia emission increased to 208 mg NH3-N/m2-h (6.0 g NH3/m2-day) with a maximum emission of 271 mg NH3-N/m2-h (7.9 g NH3/m2-day).

4.2. Emissions During Land Application of Livestock Manure

Ammonia

Svensson (1994) investigated the factors that affect ammonia volatilization and thus emission from land applying swine and cattle manure. He pointed out that the major factors influencing ammonia emission were (1) meterorological, (2) soil/manure characteristics, and (3) the application technique. For meteorological factors, wind speed, air temperature, and thermal stratification near the soil surface were most important. Regarding soil/manure characteristics, soil temperature, soil pH, soil porosity, and soil water content were most important. Finally, the application technique was noted as having a large impact on ammonia emission rates. Svensson (1994) conducted a series of controlled experiments to quantify the influence of these factors, mainly by recording the equilibrium ammonia concentration above the soil after a land application event. This equilibrium ammonia concentration was then used to determine the relative potential of ammonia emission rates from land application of both cattle and pig slurry. Soil temperature was found to be a critical factor. At soil temperatures of 24 C, the equilibrium ammonia concentration was over three times that for soil temperatures at 14 C (18 versus 5 ppm ammonia). Manure solids content was also found to be an important contributor to ammonia emission. A pig slurry of 5.4 percent solids had an equilibrium ammonia concentration of about 4 ppm, and this increased to 23 ppm ammonia for pig slurry at 14.4 percent solids. Application technique had the largest effect on the equilibrium ammonia concentration above the soil surface after spreading. If the slurry was injected, the average equilibrium ammonia concentration one hour after land applying was less than 1 ppm. If this same slurry was surface applied with no follow-up coverage, the equilibrium ammonia concentration one hour after land applying rose to 39 ppm. Svensson (1994) further investigated the influence of land application technique using pig urine only. If this "slurry" was broadcast spread with no follow-up cover, ammonia was emitted at about 700 g NH3/hectare-h during the first four hours. If this same slurry was broadcast spread with immediate covering *via* harrowing, the ammonia emission reduced to about 120 g NH3/hectare-h over the same time period, representing an 83 percent reduction. Clearly, injecting or *immediate* covering of slurry has a substantial reducing effect on ammonia emission.

Trace Gases

Misselbrook *et al.* (1993) studied the relationship between odor emission and intensity for land applied swine manure. They determined a relationship between a 0-6 point intensity scale and the concentration of odors emitted from land applied slurry. Their intensity scale used is given below:

Intensity (I)	Description
0	No odor
1	Very faint odor
2	Faint odor
3	Distinct odor
4	Strong odor
5	Very strong odor
6	Extremely strong odor

They found a relationship that described 68 percent of the variability in their data with $I=1.61(\mathrm{Log}_{10} C) + 0.45$ where C is the dilution to threshold concentration of odor. They further summarized their data to give indications of the odor intensity with the dilution threshold concentration as given below:

Intensity	Pig Slurry Odor Concentration (OU/m3)
0	0-1.1
1	1.1-4.5
2	4.5-18.8
3	18.8-78.9
4	78.9-331
5	331-1390
6	> 1390

From this study, and from cited work of others, they concluded that an odor intensity at or below an intensity of 2 (faint odor) may be considered acceptable, which further implies that for pig slurry the odor concentration should be on average below about 4.5 OU/m3. For a barely perceptible odor, indicated by an Intensity level of 1, the odor concentration should be on average below about 1.1 OU/m3.

Pain *et al.* (1991) investigated the concentrations and emissions of odors from land applied pig and cattle slurry. They investigated the emission rates of odors as a function of the land application method, and found odor emission rates (OU/s-m3 of slurry applied) of 8600 if the slurry was immediately plowed under versus 53700 for surface applied slurry, representing an 84 percent reduction in odor emission rates. For all experiments conducted, peak emissions occurred within one hour after spreading, and exponentially decayed rapidly to a level of about 10 percent the initial emission rate six hours after spreading. They stated that waiting 3-6 hours after surface applying before incorporating the slurry gave no benefit to the odor load experienced.

4.3. Dispersion Models

Predicting downwind concentrations of air pollutants released from concentrated animal feeding operations (CAFOs) is difficult because the emissions vary over time and they tend to be emitted from a variety of source types within a small area. This section provides a brief overview of the state of the science of the issue with focus on 1) classic methods for predicting pollutant concentrations downwind of a source and 2) recent reports in the peer-reviewed literature.

The Gaussian Plume model is the classic method of predicting downwind concentrations of air pollutants released from a single source. The model is based on a statistical model of diffusion from an origin. Its most important assumption is that of steady state conditions from a single source. The model assumes a constant state of meteorological conditions and emission rates. Given this assumption, however, the model can be used to examine how factors such as turbulent dispersion in the vertical and horizontal directions, wind speed, atmospheric stability, and emission rates will affect concentrations of the pollutant downwind. See figure below.

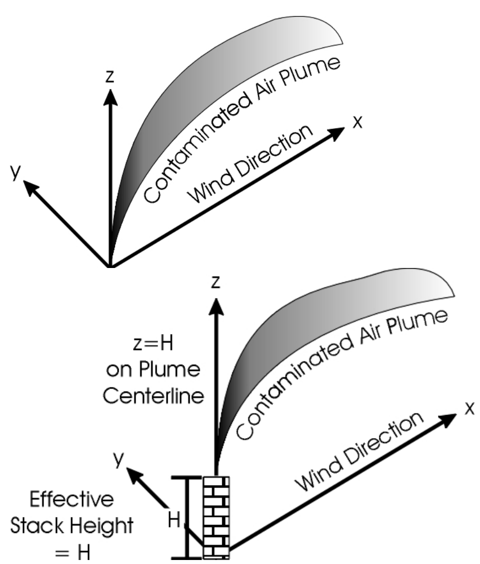

Figure 4.1. Coordinates of the Gaussian Plume model. The top figure describes a ground-level emission and the bottom figure describes a stack point-source emission.

The general equation for the concentration of pollutants can be derived as:

$$C(x,y,z) = \frac{Q}{2\pi\sigma_y\sigma_z}\exp\left(-\frac{y^2}{2\sigma_y^2}\right)\exp\left(-\frac{(z-H)^2}{2\sigma_z^2}\right)$$

where C is the concentration (mass/volume); x is the distance downwind of the source; y is the horizontal distance perpendicular to the x direction; z is altitude; Q is the emission rate (mass/time); σ_y and σ_z are the dispersion coefficients in the horizontal and vertical directions, respectively; and H is the effective stack height of the plume.

The Gaussian plume model is a reasonable screening level approach for estimating the concentration of pollutants released from a source. It can be modified to incorporate reflection or absorption of the pollutants by the ground and reactions in the atmosphere. This model is useful in examining the effect of atmospheric stability and in estimating the point of maximum downwind concentrations. Use of the Gaussian plume model can be useful in addressing questions about downwind concentrations of air pollutants from CAFOs such as:

- What level of improvement in air concentrations is predicted with a reduction in the emission rate?
- What wind directions and atmospheric stabilities will result in higher concentrations?
- What affect would installation of a stack exhaust have on downwind concentrations?
- How do meteorological conditions affect the diurnal and seasonal variability of air concentrations?
- What is the relationship between decreasing concentrations and distance from the source?

4.3.1 EPA Dispersion Models

There are a number of U.S. EPA approved computer models that are based on the Gaussian plume approach, with specific adaptations for local terrain, non-constant emissions, ground level and small area sources, and atmospheric deposition. As noted above, the Gaussian Plume models are especially useful as screening tools, designed to quickly address basic hypotheses about the relationship between sources and downwind concentrations.

Several dispersion models approved by the EPA have been evaluated for their use in association with confinement operations. The Industrial Source Complex Short Term (ISCST3) model is commonly used to model the dispersion from industrial point-sources. This model and two others: the AMS/EPA Regulatory Model (AERMOD), and the non-steady state CALPUFF model were evaluated for their effectiveness in modeling emissions from feedlot facilities (Earth Tech, 2001). The sophistication of the AERMOD and CALPUFF models give them certain advantages such as: flexibility for defining the area source geometry (AERMOD); and a realistic simulation of multi-facility impact assessment (CALPUFF). However, the ISCST3 model was chosen as the best model for evaluation of a single facility primarily because of its ease of use and familiarity. This model has been used in the past for modeling emissions from agricultural sources because, in addition to modeling plumes from tall stacks, it can also account for ground-level sources as would be the case for gases emitted from a manure pit (Gassman, 1992). Modifications to this model have also been made to increase the accuracy of its use for hydrogen sulfide and ammonia by the application of specific dispersion coefficients for these gases (Rege and Tock, 1996).

4.3.2. Livestock System Based Dispersion Models

Dispersion models have been used to predict downwind concentrations of ammonia from CAFO facilities. Quinn *et al.* (2001) tested several atmospheric dispersion models. They predicted air concentrations close (<100 m) to the CAFO source with some success. They tested a computational fluid dynamics (CFD) model linked to a modified diffusion model. The latter approach best reasonably fit the data although it underestimated concentrations for the majority of the points. The modeling was most successful in predicting the decrease in concentration with distance for near (<100m) sites. It should be noted that the success of this study is due, in part, to the design of the study. In this study, ammonia gas was released in known quantity, so the modeling effort benefited from use of a quantitative emission source.

Dispersion of odorous compounds has been considered using a modified Gaussian plume model. In one of the earlier papers on this subject, Carney and Dodd (1989) compared a modified Gaussian plume model used for odor dispersion with actual data from a number of sources, including a 450-sow swine facility, and determined that modeling adequately predicted actual plume dispersion. However, Li et al. (1994) found that the Gaussian model was inadequate for odor prediction from a 200-sow facility. The model's predicted plumes were too wide compared to those in the field and the model's emission rates were unreasonably high. Furthermore, Heinemann and Wahanik (1998) studied the application of this model to the dispersion of odors from a composting facility and found that instantaneous measurements taken during field samples may differ considerably from model predictions because of the large averaging time used by the model. Gassman (1992) reviewed odor modeling using the Gaussian-plume method and stated that the method was adequate when used on a relative basis for comparing differences between different scenarios, but did not recommend this method for finding absolute odor concentrations.

One of the ultimate utilities of odor-dispersion modeling is its use for estimating odor concentrations for the purpose of establishing setback distances and dilution ratios (Jacobson *et al.*, 2001; Zhu *et al.*, 2000b). However, researchers recognize that the use of dispersion models for this purpose will involve considerable field validation, which includes an understanding of the effects of various weather conditions on model accuracy and odor intensity. Previous field validation studies have demonstrated that the INPUFF-2 dispersion model simulated odor intensity in agreement with field odor measurements and may be the best model for the purpose of establishing odor setback distances (Zhu *et al.*, 2000b; Jacobson *et al.*, 2000).

4.3.3 Uncertainties and Recommended Uses of Models for CAFO Emissions

The Gaussian plume model and its modifications assume an emission source that is singular or made up of specific single sources: a point source, line source, or homogeneous area sources. Emissions from CAFOs are none of these. Animal operations in Iowa are increasingly compacted, and some facilities include an integration of the animal life-cycle from farrow to finish as well as outside manure storages (See Figure 4.2). There are a variety of potential gas and particulate emission sources. Possible sources may include farrowing/nursery barns, finishing barns, outside storages and fields where manure is applied. When barns are ventilated with single fans, the sources may be modeled as point sources, but outside storage units are clearly more like area sources. Barns with a series of ventilation fans behave as something in between area and point sources. Modeling emission sources from such a variety of source types makes achievement of an accurate prediction difficult. In addition, the emission rates often vary throughout the day, with local climate, and as the need for ventilation changes. For these reasons, Gaussian plume models will not excel at predicting actual pollutant concentrations downwind from a source or sources.

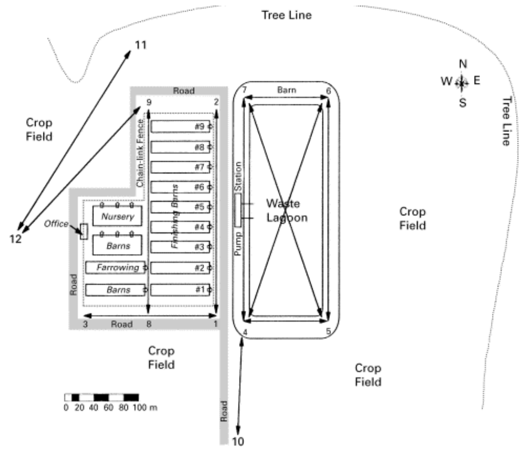

Figure 4.2. Integrated industrial swine production facility. From Childers *et al.*, 2001, Atmos. Environ. (35) 1923-1936.

Atmospheric dispersion models have limited utility for predicting absolute concentrations of atmospheric particles and gaseous compounds released from concentrated animal facilities. This limitation is primarily because the magnitude and variability in the emission sources that are difficult to quantify. Therefore, models cannot be used in lieu of direct measurements.

Noting the difficulties above, models still have essential uses in controlling and interpreting downwind concentrations of air pollutants released from CAFOs. First of all they successfully address how changes in the emission source, meteorology, and time of day or year affect concentrations (see bulleted questions above). Use of models for these purposes is a valuable and well-accepted mechanism for abatement of air pollutants. Models can be successfully used to predict the effect of emission reductions on ambient air concentrations. For example, models are a routine component of state implementation plans (SIPs) for reduction of criteria pollutants (U.S.EPA, 2002). State environmental agencies use models to estimate the relationship between local emission sources and measured concentrations of air pollutants. Using these findings, state agencies issue permits to emitters limiting emissions or requiring specific air pollution control devices or procedures. It is reasonable that states would also use this strategy to issue permits for CAFO emissions.

Dispersion models are useful as screening methods for predicting trends and percent changes in concentrations of atmospheric compounds released from CAFOs. Factors that may aggravate downwind conditions and that can be addressed with dispersion models include: trends caused by meteorological conditions, relative changes in source strengths, and dilution factors with distance away from the source. If direct measurements of emissions or concentrations very close to the facility are made, then dispersion models can be used to estimate dilution of the atmospheric compounds as a function of distance from the facility. Use of dispersion models to predict relative decreases in air concentrations as a result of decreases in emission rates is one of the most powerful uses of the models.

4.4. Evaluating Community Exposures to Odor

One method for evaluating the influence of CAFOs on surrounding residences is to review odor complaint records and the methods used for evaluating these complaints. Several states have procedures in place for documenting and evaluating odor complaints, as discussed in this section.

There is not a consistent method used in addressing odor complaints from animal feeding operations in the United States. Not all states have odor standards to address odor complaints. Some states use an arbitrary odor scale. Some states use dilution-to-threshold for odor evaluation. Others states use hydrogen sulfide as a surrogate method to measure odors. In Iowa, one of the most frequent complaints from livestock operations is odor. Since Iowa has no regulations pertaining to odors, some field offices do not record odor complaints.

Between July 1994 to October 2001, Iowa had 306 odor complaints. North Carolina Department of Air Quality (DAQ) started keeping a database for odors in February 1999. North Carolina has reported 415 separate complaints with a follow-up inspection of all complaints. As of December 2001, DAQ in North Carolina confirmed the presence of "objectionable odors" at 6 complainant locations involving 11 farms. Most of the complaints reported in Iowa and North Carolina were from swine facilities. Missouri has a state odor standard for industrial emission which will also be applied to the Class 1A CAFO's beginning January 2002.

4.4.1. Methods of evaluation of complaints

Some states do have odor standards or regulations governing odor emissions from industry, livestock or both. The methods on how complaints are addressed ranges from lay observers, to trained observers going to the site from a regulatory agency to simply registering the complaints from an individual. Several states and municipalities have odor standards.

The methods used to evaluate odor complaints range from a person to a group of people going to the location of the complaint and observing the source, strength of odor at the location, or measuring a surrogate odorous gas concentrations. There is limited data available on odor complaints from livestock.

Methods of evaluation complaint evaluation by states

Methods of odor evaluation are not consistent among states. The methods range from no protocol to arbitrary methods, to odor threshold measurement, to using hydrogen sulfide as a surrogate method of measurement. The length of time for evaluation and protocol for evaluation also differs.

No protocol

Some states do not regulate odors and therefore do not have an approved procedure for evaluating odor complaints. Iowa currently does not have an odor standard; therefore, does not have an adopted protocol for measuring odors. The livestock odor complaints in Iowa basically go unverified, since there are no standards. Within Iowa some municipalities have standards which use the scentometer to evaluate odors.

Arbitrary protocol

Some states use an arbitrary odor scale to evaluate odor strength with 0 being no odor and a higher number being a very strong odor. North Carolina uses an arbitrary scale of 0 to 5 with 0 being no odor and 5 being very odorous. This method often uses a team approach of more than one person taken to the site for investigation. Average values from the panel are used to evaluate odor strength.

Odor Threshold

The scentometer made by Barnebey and Sutcliffe is the primary method used when the protocol uses dilution-to threshold techniques for evaluating odors. Table 1 shows (Sweeten, 1990) a list of states that uses this method as a standard. The accepted standard level of odor threshold varies from state to state as shown in the table. Also the location of measurement differs between standards used, i.e., on site, property line, or neighbor's residence.

Hydrogen sulfide

Hydrogen sulfide concentrations are sometimes used as a substitute for odor evaluation. This standard is used in both Minnesota and Nebraska. Minnesota has a state hydrogen sulfide standard at 30 ppb not to be exceeded more than twice per 5 days in a 30-minute time period at the property line. Minnesota allows for a time period of 21 days during the year when this standard of 30 ppb is exempt. Nebraska has a similar standard of 100 ppb that cannot be exceeded more than 30 minutes. Both of these states give the counties jurisdiction for siting livestock facilities and the allowable odor level is left up to the county.

4.4.2. Odor Complaint Evaluation Discussion by State

Iowa

The Iowa Department of Natural Resources (DNR) Compliance and Enforcement Bureau consists of six field offices that are located throughout the state. Each field office is responsible for conducting routine inspections of agricultural facilities and handling complaints from the public in their designated counties. Animal feeding operations, or AFOs, are the source of many types of complaints, including well contamination, waste runoff, improper disposal of dead animals, and many others. Although the field offices receive a variety of complaints, one of the most frequent causes of complaints is odor. Animal feeding operations generate odors from several sources, such as the buildings where animals are housed, waste treatment systems such as lagoons or earthen basins, and the spreading of manure. More specifically, odors can occur from:

- stockpiling manure,
- untimely disposal of dead animals,
- improper compost pile management,
- spilling manure on roads or highways,
- spreading manure on snow, and
- spreading manure without injection.

Citizens that have complaints are encouraged to call the field office in their area. Odor complaints taken at field offices are not referred to the Air Quality Bureau or central offices. Although similar in nature, the complaint forms vary for each field office. Each contains the following information:

- the date the complaint was received,

- basic information on the complainant,
- basic information on who the complaint is against,
- program area (such as wastewater, air, solid waste, etc.),
- a statement of the complaint, and
- action/resolution.

The program area section lists different areas in which to classify the compliant, but varies in content and detail for each form. Some forms list the DNR employee to whom the complaint was referred, and others assign each complaint a complaint number. After a complaint form is filled out, each complaint gets logged in a spreadsheet. Again, although similar in nature, the categories listed in the spreadsheet vary for each field office. Several issues arose while completing this study:

- Most odor complaints go unrecorded at the field offices. There is no written protocol established for receiving and recording incoming odor complaints because the DNR does not regulate odor.
- Many odor complaints are never called in. Once citizens learn that there are no odor regulations, they realize the DNR may not be able help them, so they don't place the call.
- Citizens may call about odor and an additional problem, and the complaint gets logged under the additional problem.

Odor complaint records involving confined animal feeding operations in the state of Iowa were evaluated from 7/1/94 to the present (10/15/01). There were 306 total complaints, which fell into the following livestock categories: 86.9% swine, 5.6% cattle, 3.9% poultry, 3.6% horse, and <1% ostrich.

Several field office staff made the statement that most complaints occur during spring and fall due to manure application. This may also be attributed to the amount of time people spend outdoors.

There is a lack of consistency in recording, processing, and responding to odor complaints in Iowa. Since Iowa has no regulations pertaining to odor, some field offices do not record complaints when odor is the primary concern. There is no written protocol established for receiving and recording incoming odor complaints. The complaint form should be standardized for each field office as well as the central offices, and the database system where complaint records finally end up should also be standardized. A well organized complaint system for the state of Iowa would allow simple queries that could quickly determine how many times a facility has been referred, or how many times a certain individual has called in a complaint.

<u>Missouri</u>

The state of Missouri has an odor standard for industrial emissions using the scentometer. Missouri's odor standard states that no person may cause, permit, or allow the emission of odorous matter in concentrations and frequencies or for durations that odor can be perceived when one (1) volume of odorous air is diluted with seven (7) volumes of odor-free air for two (2) separate trials not less than fifteen (15) minutes apart within the period of one (1) hour. One exception of this standard was livestock production units. Class 1A CAFOs; however, was added to the list of regulated odors. A Class 1A livestock operation has a population of greater than 4,900 head of dairy cows; 17,500 head of finishing hogs; or 210,000 layer hens. Missouri has 20 Class 1A CAFOs. All 1A CAFOs operating on or after January 1, 1999, shall prepare and implement an odor control plan.

These plans must be submitted no later than July 1, 2000. After January 1, 2002, no Class 1A Concentrated Animal Feeding Operation (CAFOs) may cause, permit or allow the emission of odorous matter in concentrations and frequencies or for durations that the odor can be perceived when one (1) volume of odorous air is diluted with five and four-tenths (5.4) volumes of odor-free air for two separate trials not less than fifteen minutes apart within the period of one hour. This odor evaluation shall be taken at a site not at the installation and will be used as a screening evaluation. A positive screening evaluation for odor shall require an odor sample to be taken and evaluated by olfactometry. There were no odor complaint charts found for the state of Missouri.

North Carolina

North Carolina uses an arbitrary scale of 0 to 5 for panel members to evaluate odor complaints on site. Zero is no odor detected. A 5 is considered a very strong odor. Normally, two or more observers go to the complainant site to determine if an odor problem exists. This sometimes requires evaluation during night-time conditions.

The Department of Air Quality (DAQ) in North Carolina is maintaining a database of complainants and complaint locations. The DAQ database was begun February 23, 1999. The following information was gathered from the database:

- There have been 255 individual complainants/complaint locations listed in their database. There was a DAQ staff follow-up site visit in each case.
- There were 415 separate complaints listed in the database from the above complainants.
- As of (December, 2001), DAQ regional inspectors have confirmed the presence of "objectionable odors" at 6 complainant locations involving 11 farms.
- The Director has required the submission of 6 BMPs per regulations for 5 of the 6 complainant locations.
- For those sites where the presence of objectionable odors was confirmed, it took between 7 and 14 visits by DAQ staff to confirm the presence of the objectionable odor in response to a complaint. Each odor determination investigation typically requires 2 or more DAQ staff and most objectionable odor conditions occur outside of normal business hours.
- The odor complaints were greatest in 1999, lesser in 2000 and 2001 (Saunders, 2001).
- Most complaints are from smaller units that fall below the required size for odor management plans (Saunders, 2001).

Odor Management Plans are required under DAQ, 2D.1802(d), for swine operations based upon steady state live weight (SSLW). The regulations have the following schedule for submittal.

- January 15, 2001, number of farms with SSLW of more than 4 million pounds required to submit odor management plan response to DAQ;7 farms.
- July 15, 2001, number of farms with SSLW of more than 2 million pounds but less than 4 million pounds required to submit odor management plan response to DAQ;78 farms.

Table 4.21. Summary of Odor Standards in the United States (Sweeten,1990)

State or Political Division	Regulatory Limit			
	Residential	Commercial	Industrial	Other
Scentometer (D/T):				
Colorado	7	7	15	127
Illinois	8	8	15	16
Kentucky	7	7	24	16
Missouri	7	7	7	
North Dakota	2	2	2	2
Nevada	8	8	8	
Oregon				2
Wyoming	7	7	7	
District of Columbia	1	1	1	
Dallas,Texas	2	1	1	
Southwest WA State, AGMA	1-2	1-2	8-32	8-32
Polk County, Iowa	7	7	7	7
Cedar Rapids, IA	4	8	20	8
Omaha, Nebraska	4	8	20	8
Chattanooga, Tennessee	0	4	4	4

Table 4.22. Ammonia Emission Summary

Species	Species Phase	Notes	Unit*	Ammonia Emission	Reported Units	Reference Source	Reference Year	VA g NH3/AU-day	ST g NH3/m2-day	LA mg NH3/m2-hr
Pigs	Nursery		VA	700 - 1,200	mg NH3/pig-day	Aarnink et al	1995	19 - 33		
Pigs	Finishing		VA	5,700 – 5,900	mg NH3/pig-day	Aarnink et al	1995	42 - 43		
Pigs	Finishing		VA	46.9	kg NH3-N/AU-yr	Demmers et al	1999	160		
Pigs	Finishing		VA	0.9 - 3.2	kg NH3-N/day	Burton and Beauchamp	1986			
Pigs	Finishing	Deep-pit only	VA	40 - 58	mg NH3/m2-h	Ni et al	2000	5 - 8	1 - 1.4	
Pigs	Finishing	Deep-pit + pigs	VA	233	mg NH3/m2-h	Ni et al	2000	40 - 50		
Pigs	Sows	on bedding	VA	744 - 3,248	mg NH3/AU-h	Groot Kooerkamp et al	1998	18 - 78		
Pigs	Sows	on slats	VA	1,049 - 1,701	mg NH3/AU-h	Groot Kooerkamp et al	1998	25 - 41		
Pigs	Nursery		VA	649 - 1,526	mg NH3/AU-h	Groot Kooerkamp et al	1998	16 - 37		
Pigs	Finishing	on bedding	VA	1,429 - 3,751	mg NH3/AU-h	Groot Kooerkamp et al	1998	34 - 90		
Pigs	Finishing	on slats	VA	2,076 - 2,592	mg NH3/AU-h	Groot Kooerkamp et al	1998	50 - 62		
Pigs	Finishing	Nursery-to-finishing	VA	70-210	g NH3/h	Hinz and Linke	1998	66		
Pigs	Finishing	Lagoon	ST			Aneja et al	2001			
Pigs	Finishing	Lagoon	ST			Aneja et al	2000			
Pigs	Finishing	Lagoon	ST	18	ng NH3/cm2-s	Zahn et al	2001		16	
Pigs	Finishing		ST	4.35	g NH3/m2-day	Hobbs et al	1999		4.4	
Pigs	Finishing	Uncovered, no crust	ST	4.3	g NH3-N/m2-day	Sommer et al	1993		5.2	
Pigs	Finishing	Uncovered, with crust	ST	0.5 - 1.5	g NH3-N/m2-day	Sommer et al	1993		0.6 - 1.8	
Pigs	Finishing	Uncovered, with straw	ST	0.2 - 1.0	g NH3-N/m2-day	Sommer et al	1993		0.25 – 1.2	
Pigs	Finishing	capped with lid	ST	0 - 0.3	g NH3-N/m2-day	Sommer et al	1993		0 - 0.36	
Pigs	Finishing	Deep-pit or pull-plug	VA	66	ng NH3/cm2-s	Zahn et al	2001	311	57	
Pigs	Finishing	Earthen, concrete, or steel-lined	ST	167	ng NH3/cm2-s	Zahn et al	2001		144	
Pigs	Finishing	Non-phototrophic lagoons	ST	109	ng NH3/cm2-s	Zahn et al	2001		94	
Pigs	Finishing	Phototrophic lagoons	ST	89	ng NH3/cm2-s	Zahn et al	2001		77	
Pigs	Gestation	Mechanically ventilated	VA	5	ug NH3/m2-s	Zhu et al	2000	2.2		
Pigs	Farrowing	Mechanically ventilated	VA	20-55	ug NH3/m2-s	Zhu et al	2000	15-42		
Pigs	Nursery	Mechanically ventilated	VA	20-140	ug NH3/m2-s	Zhu et al	2000	23-160		
Pigs	Finishing	Mechanically ventilated	VA	20-55	ug NH3/m2-s	Zhu et al	2000	10-26		
Pigs	Finishing	Naturally ventilated, pit fans	VA	60-170	ug NH3/m2-s	Zhu et al	2000	28-80		
Pigs	Finishing	Slurry removed weekly	VA	11	kg NH3/AU-yr	Osada et al	1998	30		
Pigs	Finishing	Deep-pit manure storage	VA	11.8	kg NH3/AU-yr	Osada et al	1998	32		
Dairy		on bedding	VA	260 - 890	mg NH3/AU-h	Groot Kooerkamp et al	1998	6.2 - 21.4		
Dairy	free-stall		VA	843 - 1,769	mg NH3/AU-h	Groot Kooerkamp et al	1998	20 - 42.5		
Beef		on bedding	VA	431 - 478	mg NH3/AU-h	Groot Kooerkamp et al	1998	10.3 - 11.5		
Beef		on slats	VA	371 - 900	mg NH3/AU-h	Groot Kooerkamp et al	1998	9 - 21.6		
Calves		on bedding	VA	315 - 1,037	mg NH3/AU-h	Groot Kooerkamp et al	1998	7.6 - 25		
Calves		on slats	VA	1,148 - 1,797	mg NH3/AU-h	Groot Kooerkamp et al	1998	28 - 43		
Beef		On chopped straw	ST	547	mg NH3/m2-h	Jeppsson	1999		13	
Beef		on unchopped straw	ST	747	mg NH3/m2-h	Jeppsson	1999		18	
Beef		on chooped straw + peat	ST	319	mg NH3/m2-h	Jeppsson	1999		8	
Dairy	free-stall	Manured slatted floor	ST	400	mg NH3/m2-h	Kroodsma et al	1993		9.6	
Dairy	free-stall	Scraped slatted floor	ST	380	mg NH3/m2-h	Kroodsma et al	1993		9.1	
Dairy	free-stall	Unstirred slurry below slats	ST	320	mg NH3/m2-h	Kroodsma et al	1993		7.7	
Dairy	free-stall	Stirred slurry below slats	ST	290	mg NH3/m2-h	Kroodsma et al	1993		7	

Dairy	free-stall	Manured solid floor	ST	670	mg NH3/m2h	Kroodsma et al	1993		16	
Dairy	free-stall	Scraped solid floor	ST	620	mg NH3/m2h	Kroodsma et al	1993		15	
Dairy	free-stall	Flushed solid floor	ST	210	mg NH3/m2h	Kroodsma et al	1993		5	
Dairy	Free-stall		ST	4	ug NH3/m2-s	Zhu et al	2000		0.35	
Beef		Uncovered, no crust	ST	4.5	g NH3-N/m2-day	Sommer et al	1993		5.5	
Beef		Uncovered, with crust	ST	1.3	g NH3-N/m2-day	Sommer et al	1993		1.6	
Beef		capped with lid	ST	0.2 - 0.4	g NH3-N/m2-day	Sommer et al	1993		0.25 – 0.5	
Poultry	Caged layers	Winter	VA	8	g NH3/AU-h	Wathes et al	1997	192		
Poultry	Caged layers	Summer	VA	12.5	g NH3/AU-h	Wathes et al	1997	300		
Poultry	Broilers	Winter	VA	9	g NH3/AU-h	Wathes et al	1997	216		
Poultry	Broilers	Summer	VA	9	g NH3/AU-h	Wathes et al	1997	216		
Poultry	Broilers	on litter	VA	4-20	ug NH3/m2-s	Zhu et al	2000	7-33		
Poultry	laying hens	on litter	VA	7,392 - 10,892	mg NH3/AU-h	Groot Kooerkamp et al	1998	177 - 261		
Poultry	laying hens	Cages	VA	602 - 9,316	mg NH3/AU-h	Groot Kooerkamp et al	1998	14 - 224		
Poultry	Broilers	on litter	VA	2,208 - 8,294	mg NH3/AU-h	Groot Kooerkamp et al	1998	53 - 199		
Poultry	Broilers	on litter	VA	18.6	kg NH3/AU-yr	Demmers et al	1999	51		
Poultry	Broilers	First flock on new bedding	ST	149-314	mg NH3-N/m2-h	Brewer and Costello	1999		4.3-9.1	
Poultry	Broilers	After four flocks on bedding	ST	208-271	mg NH3-N/m2-h	Brewer and Costello	1999		6.0-7.9	
Pigs		Surface applied, urine only	LA	700	g NH3/hectare-h	Svensson	1994			70
Pigs		Surface applied + immediate cover, urine only	LA	120	g NH3/hectare-h	Svensson	1994			12

* VA=ventilation air, ST-storage, LA=land application

Table 4.23. Methane Emission Summary

Species	Species Phase	Notes	Unit	Methane Emission (Reported Level)	Reported Units	Reference Source	Reference Year	**VA** g CH4/AU -day	**ST** g CH4/m2-day
Pigs	Finishing	Fall	ST	80	ng CH4/cm2-s	Zahn et al	2001		69
Pigs	Finishing	Summer	ST	134	ng CH4/cm2-s	Zahn et al	2001		116
Pigs	Finishing		ST	21.4	g CH4/m2-day	Hobbs et al	1999		21.4
Pigs	Finishing	deep-pit or pull-plug	VA	34	ng CH4/cm2-s	Zahn et al	2001	160	29
Pigs	Finishing	earthen, concrete, or steel-lined	ST	178	ng CH4/cm2-s	Zahn et al	2001		154
Pigs	Finishing	non-phototrophic lagoons	ST	218	ng CH4/cm2-s	Zahn et al	2001		188
Pigs	Finishing	Phototrophic lagoons	ST	200	ng CH4/cm2-s	Zahn et al	2001		173
Pigs	Finishing	slurry removed weekly	VA	17.5	kg CH4/AU -yr	Osada et al	1998	48	
Pigs	Finishing	Deep-pit manure storage	VA	19.7	kg CH4/AU -yr	Osada et al	1998	54	
Dairy			VA	542	L CH4/cow-day	Kaharabata and Schuepp	2000		
Dairy		Feedlot	ST	631	L CH4/cow-day	Kaharabata and Schuepp	2000		
Poultry	Caged layers	Winter	VA	0.8	g CH4/AU -h	Wathes et al	1997	19	
Poultry	Caged layers	Summer	VA	0.9	g CH4/AU -h	Wathes et al	1997	22	
Poultry	Broilers	Winter	VA	0.25	g CH4/AU -h	Wathes et al	1997	6	
Poultry	Broilers	Summer	VA	0.25	g CH4/AU -h	Wathes et al	1997	6	

Table 4.24. Hydrogen Sulfide Emission Summary

Species	Species Phase	Notes	Unit	H2S Emission (Reported Level)	Reported Units	Reference Source	Reference Year	**VA** g H2S/AU-day	**ST** g H2S/m2-day
Pigs	Finishing	deep-pit only	VA	1.6 - 3.8	mg H2S/m2-h	Ni et al	2000	0.22 - 0.49	0.04 - 0.09
Pigs	Finishing	deep-pit + pigs	VA	9.4	mg H2S/m2-h	Ni et al	2000	1.25	0.23
Pigs	Gestation	Mechanical ventilation	VA	2	ug H2S/m2-s	Zhu et al	2000		
Pigs	Farrowing	Mechanical ventilation	VA	5	ug H2S/m2-s	Zhu et al	2000	1	
Pigs	Nursery	Mechanical ventilation	VA	20-140	ug H2S/m2-s	Zhu et al	2000	4	
Pigs	Finishing	Mechanical ventilation	VA	10	ug H2S/m2-s	Zhu et al	2000	23-160	
Pigs	Finishing	Natural ventilation, pit fans	VA	5-15	ug H2S/m2-s	Zhu et al	2000	5	
Pigs	Finishing	Fall	ST	2.11	ng H2S/cm2-s	Zahn et al	2001	2-7	1.8
Pigs	Finishing	Summer	ST	0.73	ng H2S/cm2-s	Zahn et al	2001		0.63
Pigs	Finishing		ST	66.6	g H2S/m2-day	Hobbs et al	1999		66.6
Pigs	Finishing	deep-pit or pull-plug	VA	0.37	ng H2S/cm2-s	Zahn et al	2001	1.7	0.32
Pigs	Finishing	earthen, concrete, or steel-lined	ST	1.1	ng H2S/cm2-s	Zahn et al	2001		0.95
Pigs	Finishing	non-phototrophic lagoons	ST	0.32	ng H2S/cm2-s	Zahn et al	2001		0.28
Pigs	Finishing	Phototrophic lagoons	ST	0.24	ng H2S/cm2-s	Zahn et al	2001		0.21
Poultry	Broiler	on litter	VA	2	ug H2S/m2-s	Zhu et al	2000	3.3	
Dairy	Free-stall		ST	3	ug H2S/m2-s	Zhu et al	2000		0.26

Table 4.25. VOC Emission Summary

Species	Species Phase	Notes	Unit	VOC Emission (Reported Level)	Reported Units	Reference Source	Reference Year
Pigs	Average	acetic acid	VA	NA		Hartung and Phillips	1994
Pigs		acetic acid	ST	1.49	g/m2-day	Hobbs et al	1999
Pigs	Average	Propionic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	n-butyric acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	I-butyric acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	n-valeric acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	I-valeric acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	n-hexanoic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	I-hexanoic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	Heptanoic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	Octoanoic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	Pelargonic acid	VA	NA		Hartung and Phillips	1994
Pigs	Average	Phenol	VA	NA		Hartung and Phillips	1994
Pigs		Phenol	ST	0.018	g/m2-day	Hobbs et al	1999
Pigs	Average	p-cresol	VA	NA		Hartung and Phillips	1994
Pigs	Average	Indole	VA	NA		Hartung and Phillips	1994
Pigs		Indole	ST	< 0.001	g/m2-day	Hobbs et al	1999
Pigs	Average	Skatole	VA	NA		Hartung and Phillips	1994
Pigs	Average	Dimethylamine	VA	NA		Hartung and Phillips	1994
Pigs	Average	Trimethylamine	VA	NA		Hartung and Phillips	1994
Pigs	Average	Acetone	VA	NA		Hartung and Phillips	1994
Pigs		Odors	ST	802,483	OU/m2-min	Hobbs et al	1999
Broilers		Odors	VA			Misselbrook et al	1993
Swine		Odors	LA			Misselbrook et al	1993
Pigs	Surface applied, no cover	Odors	LA	53,700	OU/s-m3 of slurry	Pain et al	1991
Pigs	Surface applied, immediate cove rage	Odors	LA	8,600	OU/s-m3 of slurry	Pain et al	1991
Pigs	Finishing	Type 1 (deep-pit or pull-plug)	VA	89.9	g VOC/system-h	Zahn et al	2001
Pigs	Finishing	Type 2 (earthen, concrete, or steel-lined)	ST	39.4	g VOC/system-h	Zahn et al	2001
Pigs	Finishing	Type 3 (non-phototrophic lagoons)	ST	113.1	g VOC/system-h	Zahn et al	2001
Pigs	Finishing	Type 4 (phototrophic lagoons)	ST	14.5	g VOC/system-h	Zahn et al	2001

Table 4.26. Particulate Emission Summary

Species	Species Phase	Notes	Emission Source	Partic.ulate Emission	Reported Units	Reference Source	Reference Year	**VA** g particulates/AU - day
Pigs	Sows	on bedding, inhalable	VA	144 – 753	mg/AU-h	Takai et al	1998	3.5 - 18
Pigs	Sows	on slats, inhalable	VA	121 – 949	mg/AU-h	Takai et al	1998	2.9 - 22.8
Pigs	Nursery	Inhalable	VA	687 - 1,364	mg/AU-h	Takai et al	1998	16.5 - 32.7
Pigs	Finishing	on bedding, inhalable	VA	561 – 890	mg/AU-h	Takai et al	1998	13.5 - 21.4
Pigs	Finishing	on slats, inhalable	VA	418 – 895	mg/AU-h	Takai et al	1998	10 - 21.5
Pigs	Sows	on bedding, respirable	VA	46 – 49	mg/AU-h	Takai et al	1998	1.1 - 1.2
Pigs	Sows	on slats, respirable	VA	13 – 141	mg/AU-h	Takai et al	1998	0.3 - 3.4
Pigs	Nursery	Respirable	VA	51 – 122	mg/AU-h	Takai et al	1998	1.2 - 2.9
Pigs	Finishing	on bedding, respirable	VA	69 – 73	mg/AU-h	Takai et al	1998	1.7 - 1.8
Pigs	Finishing	on slats, respirable	VA	34 – 133	mg/AU-h	Takai et al	1998	0.8 - 3.2
Dairy		on bedding, inhalable	VA	60 – 142	mg/AU-h	Takai et al	1998	1.4 - 3.4
Dairy		free-stall, inhalable	VA	21 – 338	mg/AU-h	Takai et al	1998	0.5 - 8.1
Beef		on bedding, inhalable	VA	36 – 135	mg/AU-h	Takai et al	1998	0.9 - 3.2
Beef		on slats, inhalable	VA	78 – 144	mg/AU-h	Takai et al	1998	1.9 - 3.5
Calves		on bedding, inhalable	VA	64 – 190	mg/AU-h	Takai et al	1998	1.5 - 4.6
Calves		on slats, inhalable	VA	63 – 192	mg/AU-h	Takai et al	1998	1.5 - 4.6
Dairy		on bedding, respirable	VA	6 – 84	mg/AU-h	Takai et al	1998	0.1 - 2
Dairy		free-stall, respirable	VA	13 – 54	mg/AU-h	Takai et al	1998	0.3 - 1.3
Beef		on bedding, respirable	VA	6 – 26	mg/AU-h	Takai et al	1998	0.1 - 0.6
Beef		on slats, respirable	VA	5 – 29	mg/AU-h	Takai et al	1998	0.1 - 0.7
Calves		on bedding, respirable	VA	14 – 40	mg/AU-h	Takai et al	1998	0.3 - 1
Calves		on slats, respirable	VA	14 – 22	mg/AU-h	Takai et al	1998	0.3 - 0.5
Poultry	Caged layers	winter, inhalable	VA	0.9	g/AU-h	Wathes et al	1997	22
Poultry	Caged layers	summer, inhalable	VA	1.1	g/AU-h	Wathes et al	1997	26
Poultry	Broilers	winter, inhalable	VA	5.2	g/AU-h	Wathes et al	1997	125
Poultry	Broilers	summer, inhalable	VA	8.2	g/AU-h	Wathes et al	1997	197
Poultry	Caged layers	winter, respirable	VA	0.24	g/AU-h	Wathes et al	1997	5.8
Poultry	Caged layers	summer, respirable	VA	0.09	g/AU-h	Wathes et al	1997	2
Poultry	Broilers	winter, respirable	VA	0.6	g/AU-h	Wathes et al	1997	14
Poultry	Broilers	summer, respirable	VA	0.88	g/AU-h	Wathes et al	1997	21
Poultry	Caged layers	Inhalable	VA	398 – 872	mg/AU-h	Takai et al	1998	9.6 - 21
Poultry	Broilers	on litter, inhalable	VA	1,856 - 6,218	mg/AU-h	Takai et al	1998	45 - 149
Poultry	Caged layers	Inhalable	VA	24 – 161	mg/AU-h	Takai et al	1998	0.6 - 3.9
Poultry	Broilers	on litter, inhalable	VA	245 – 725	mg/AU-h	Takai et al	1998	5.9 - 17.4

Table 4.27. Bioaerosol Emission Summary

Species	Species Phase	Notes	Unit	Bioaerosol Emission	Reported Units	Reference Source	Reference Year	VA mg /AU-day	VA Log CFU/AU -day
Pigs	Sows	average, inhalable endotoxin	VA	37.4	ug/AU-h	Seedorf et al	1998	0.9	
Pigs	Nursery	average, inhalable endotoxin	VA	66.6	ug/AU-h	Seedorf et al	1998	1.6	
Pigs	Finishing	average, inhalable endotoxin	VA	49.8	ug/AU-h	Seedorf et al	1998	1.2	
Pigs	Sows	average, respirable endotoxin	VA	3.7	ug/AU-h	Seedorf et al	1998	0.09	
Pigs	Nursery	average, respirable endotoxin	VA	8.9	ug/AU-h	Seedorf et al	1998	0.2	
Pigs	Finishing	average, respirable endotoxin	VA	5.2	ug/AU-h	Seedorf et al	1998	0.1	
Dairy		average, inhalable endotoxin	VA	2.9	ug/AU-h	Seedorf et al	1998	0.07	
Beef		average, inhalable endotoxin	VA	3.7	ug/AU-h	Seedorf et al	1998	0.09	
Calves		average, inhalable endotoxin	VA	21.4	ug/AU-h	Seedorf et al	1998	0.5	
Dairy		average, respirable endotoxin	VA	0.3	ug/AU-h	Seedorf et al	1998	0.007	
Beef		average, respirable endotoxin	VA	0.6	ug/AU-h	Seedorf et al	1998	0.01	
Calves		average, respirable endotoxin	VA	2.7	ug/AU-h	Seedorf et al	1998	0.06	
Poultry	Caged layers	winter, endotoxin	VA	10	g/AU-h	Wathes et al	1997	240000	
Poultry	Caged layers	summer, endotoxin	VA	30	g/AU-h	Wathes et al	1997	720000	
Poultry	Broilers	winter, endotoxin	VA	< 1	g/AU-h	Wathes et al	1997	< 24000	
Poultry	Broilers	summer, endotoxin	VA	45	g/AU-h	Wathes et al	1997	1080000	
Poultry	Caged layers	average, inhalable endotoxin	VA	538.3	ug/AU-h	Seedorf et al	1998	13	
Poultry	Broilers	average, inhalable endotoxin	VA	817.4	ug/AU-h	Seedorf et al	1998	20	
Poultry	Caged layers	average, respirable endotoxin	VA	38.7	ug/AU-h	Seedorf et al	1998	0.9	
Poultry	Broilers	average, respirable endotoxin	VA	46.7	ug/AU-h	Seedorf et al	1998	1.1	
Pigs	Sows	average, total bacteria	VA	7.7	Log CFU/AU -h	Seedorf et al	1998		185
Pigs	Nursery	average, total bacteria	VA	7.1	Log CFU/AU -h	Seedorf et al	1998		170
Pigs	Finishing	average, total bacteria	VA	7.6	Log CFU/AU -h	Seedorf et al	1998		182
Dairy		average, total bacteria	VA	6.8	Log CFU/AU -h	Seedorf et al	1998		163
Beef		average, total bacteria	VA	6.7	Log CFU/AU -h	Seedorf et al	1998		161
Calves		average, total bacteria	VA	7.3	Log CFU/AU -h	Seedorf et al	1998		175
Poultry	Caged layers	average, total bacteria	VA	7.1	Log CFU/AU -h	Seedorf et al	1998		170
Poultry	Broilers	average, total bacteria	VA	9.5	Log CFU/AU -h	Seedorf et al	1998		228

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Chapter 5. Fate and Transport of Air Pollutants from CAFOs

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5.1 Introduction

A schematic of the fate and transport of air emissions from Confined Animal Feeding Operations (CAFOs) is given by Figure 1. Many sources contribute to the overall fugitive emissions from such operations: the animals themselves, their manure, manure applied to farm fields nearby, and waste lagoons. Emissions can be as particles or gases, and they may serve as reactants for aerosol formations (micron and submicron size solid and liquid suspensions).

Particles emitted from CAFOs that may cause problems include odorants, dusts, animal dander and other allergens. Generally, these are dispersed rapidly in the atmosphere by mixing processes and are deposited to the land surface.

Gases are also of concern. These may include odorants, hydrogen sulfide (H_2S), ammonia (NH_3), methane (CH_4) and other trace gas constituents. Some of these persist in the atmosphere for hours or days, and they may be transported hundreds of kilometers (Table 1). Ammonia and sulfur compounds from CAFOs participate in reactions that can form secondary particles and aerosols in the atmosphere. These may limit visibility, cause health effects to sensitive individuals, and be precursors of acid rain at a regional scale. Secondary particles include ammonium sulfate $(NH_4)_2SO_4$, ammonium bisulfate $(NH_4)HSO_4$, and ammonium nitrate NH_4NO_3.

Large amounts of manure at feedlots can undergo partial anaerobic degradation by bacteria to form gases such as methane (CH_4) and nitrous oxide (N_2O). These are potent greenhouse gases at a global scale, and they contribute a significant fraction of Iowa's greenhouse gas emissions to the global atmosphere.

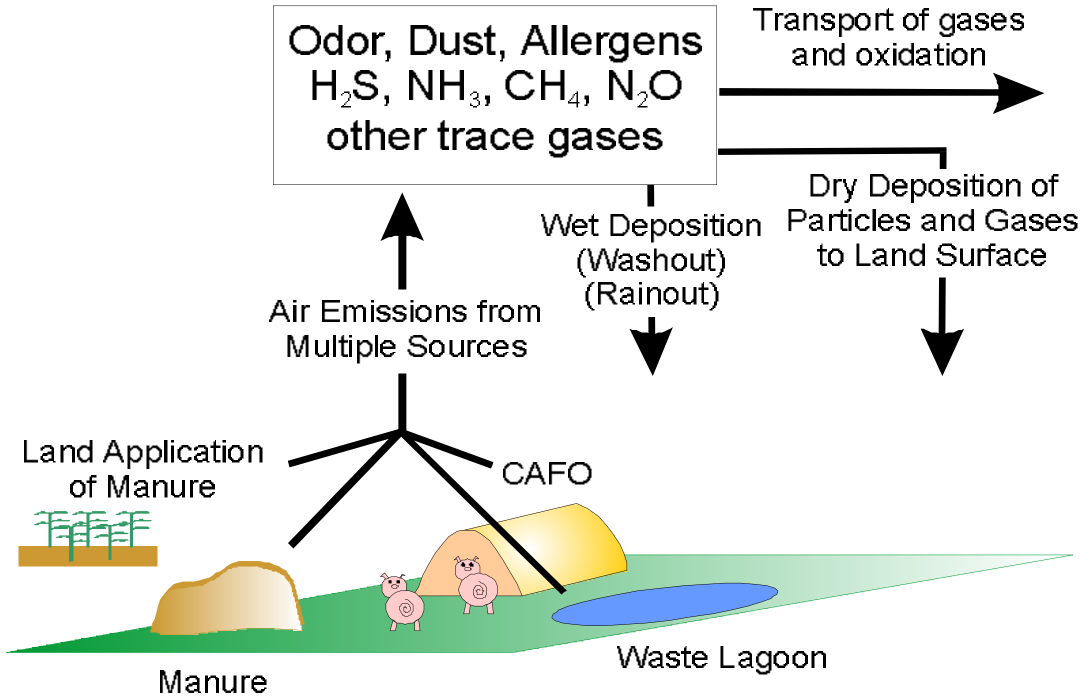

Figure 1. Fate and Transport of Air Emissions Associated with Confined Animal Feeding Operations

Table 1. Transport of Air Emissions Associated with Confined Animal Feeding Operations

	Local Scale		Regional to Global Scale	
	Short Range Transport (<10 km)	Fate Processes	Medium-to-Long Range Transport (10-1000 km)	Fate Processes
Particles	Odor (particles) Dust (animal Dander) Allergens	Dispersion Dry Deposition	Secondary Particle Formation (e.g., $(HN_4)_2SO_4$, NH_4NO_3, $(NH_4)HSO_4$, Aerosols)	Dispersion Dry Deposition Washout Rainout
Gases	Odor (gases) Dimethyl sulfide (DMS) Mercaptans	Dispersion Rapid Rxn.	Hydrogen Sulfide (H_2S) Carbon Disulfide (CS_2) Ammonia (NH_4) Sulfur oxides (SO_x) Methane (CH_4) Nitrous Oxide (N_2O)	Dispersion Rxn. with hydroxyl radicals Washout Dry Deposition

A simplified stoichiometric equation for the anaerobic digestion of manure is:

$$\underset{\text{“manure”}}{C_5H_7O_2NS_{0.5}} + 3.025\ H_2O \xrightarrow{\text{anaerobes}} 2.5125\ CO_{2(g)} + 2.4875\ CH_{4(g)} + NH_{3(g)} + 0.05\ H_2S_{(g)}$$

Four primary gases are formed including carbon dioxide, methane, ammonia, and hydrogen sulfide. These are the major constituents emanating from CAFOs, but they are not necessarily the most problematic. Hydrogen sulfide and ammonia are toxic to humans. However, odorants, dust, and allergens may cause greater nuisance problems for nearby citizens and communities.

The above chemical reaction may occur wherever manure is handled in large quantities. Piles of manure below CAFO gratings and in penned areas, manure that is improperly knifed (mixed) into soil on agricultural fields, and anaerobic waste lagoons can all produce these gases.

5.2 Sulfur Compounds

Confined animal feeding operations (CAFOs) produce significant amounts of reduced sulfur compounds (H_2S, mercaptans, carbon disulfide, dimethyl sulfide DMS, and others) from anaerobic microbial action on fecal matter on a local scale. Reduced forms of sulfur in the atmosphere are oxidized by hydroxyl radicals. For H_2S and DMS the reaction is reasonably fast with lifetimes in the troposphere (lower atmosphere) of 4.4 days and 0.6 days, respectively (Warnek, 1988). Thus, they can be transported tens or even hundreds of kilometers from the animal feeding operation prior to oxidation. A series of free radical reactions occurs for H_2S in the presence of hydroxyl radicals, formaldehyde, or ozone in the atmosphere.

$$\cdot OH + H_2S \rightarrow H_2O + HS\cdot$$

$$HS\cdot + HCHO \rightarrow H_2S + HCO\cdot$$

or

$$HS\cdot + O_3 \rightarrow HSO + O_2$$

The $HS\cdot$ radical does not build-up in the atmosphere, and it is thought that formaldehyde and ozone react with $HS\cdot$. Likewise, dimethyl sulfide (DMS) undergoes a series of free radical reactions beginning with the ubiquitous hydroxyl radical $\cdot OH$.

$$\cdot OH + CH_3SCH_3 \rightarrow CH_3S(OH)CH_3$$

$$CH_3S(OH)CH_3 + O_2 \rightarrow (O_2\text{ - DMS - }OH)\text{ complex}$$

$$\text{complex} \rightarrow \underbrace{CH_3(SO_3)H}_{MSA} + CH_3\cdot$$

Methane sulfonic acid (MSA) is a major product of the reaction between hydroxyl radicals and DMS. It is a condensable product that helps to form aerosol particles in the atmosphere (cloud condensation nuclei). But these reactions would occur far from the confined animal feeding operation at a regional or global scale.

Carbon disulfide (CS_2) also reacts rapidly with hydroxyl radicals in the atmosphere and has a lifetime of ~12 days which transports it hundreds of kilometers from the source (Warnek, 1988). Of course, the concentration dissipates quickly due to mixing (dispersion), dry deposition, and washout by precipitation.

$$\cdot OH + CS_2 \rightarrow SCSOH$$
$$SCSOH + O_2 \rightarrow COS + SO_2 + H$$

Carbonyl sulfide (COS) is slow to be oxidized. Its lifetime in the atmosphere is on the order of 44 years (Warnek, 1988). Carbonyl sulfide is transported and mixed at trace concentrations on a global scale.

$$\cdot OH + COS \rightarrow CO_2 + H_2S$$

Sulfur oxides can also be emitted from CAFOs , and/or they may form as an oxidation product of reduced sulfur emissions from CAFOs. Both gas and aqueous phase reactions are important in the oxidation of sulfur dioxide in the atmosphere. Oxygen and hydroxyl radicals can oxidize SO_2 to SO_4^{2-} in the gas phase, and humidity in the air (H_2O) can convert SO_3 to acidic aerosol particles.

$$SO_2 + \cdot OH \rightarrow HOSO_2 \xrightarrow{O_2} SO_3 + HO_2(\text{superoxide})$$
$$SO_3 + H_2O \rightarrow H_2SO_4 \text{ (aerosol)}$$

Aqueous phase reactions for SO_2 include reaction with ozone O_3 and hydrogen peroxide H_2O_2; both can be important depending on the concentrations of ozone and hydrogen peroxide in clouds.

$$SO_2 + H_2O \rightarrow SO_3^{2-} + 2H^+$$
$$SO_3^{2-} + O_{3(aq)} \rightarrow SO_4^{2-} + O_2$$
$$SO_3^{2-} + H_2O_{2(aq)} \rightarrow SO_4^{2-} + H_2O$$

Lifetimes for the above reactions in clouds are on the order of 1-50 days. Clouds process a tremendous amount of air and water vapor. They serve as a concentrating vortex for particles and gases that react with SO_2. These are all long-range transport processes that take place far from the original CAFO operation.

Most CAFO sulfur emissions are in the form of reduced sulfur species and SO_2. Sulfur falls back to earth (continents and oceans) in the form of $SO_{2(g)}$ (dry deposition), sulfate aerosols (H_2SO_4, $(NH_4)_2SO_4$ NH_4NO_3, $MgSO_4$, $CaSO_4$ in dry deposition), and sulfate ions (H_2SO_4 and $CaSO_4$ in wet deposition). Sulfate aerosols and cloud condensation nuclei play an important role as a negative feedback effect to global warming by increasing the earth's albedo on a global scale. $SO_{2(g)}$ results in H_2SO_4 (sulfuric acid) and acid deposition. However, emissions from CAFOs are very small compared to coal-fired power plants, smelters, industrial emissions, and even volcanoes. In Figure 2, CAFOs contribute negligible amounts of hydrogen sulfide H_2S, DMS, COS, and CS_2 to the global atmosphere; these gases are in turn oxidized to $SO_{2(g)}$ and eventually to sulfate, both of which are deposited to land and oceans.

Ammonia $NH_{3(g)}$ is a weak base that reacts with water to form ammonium and hydroxide ions in CAFO air. This increases the pH of water vapor in CAFO settings and helps to neutralize sulfuric acid from SO_2 emissions (Figure 3). When water is evaporated from the atmosphere, one of the principal salts that form as aerosols and causes decreased visibility is ammonium sulfate $(NH_4)_2SO_4$.

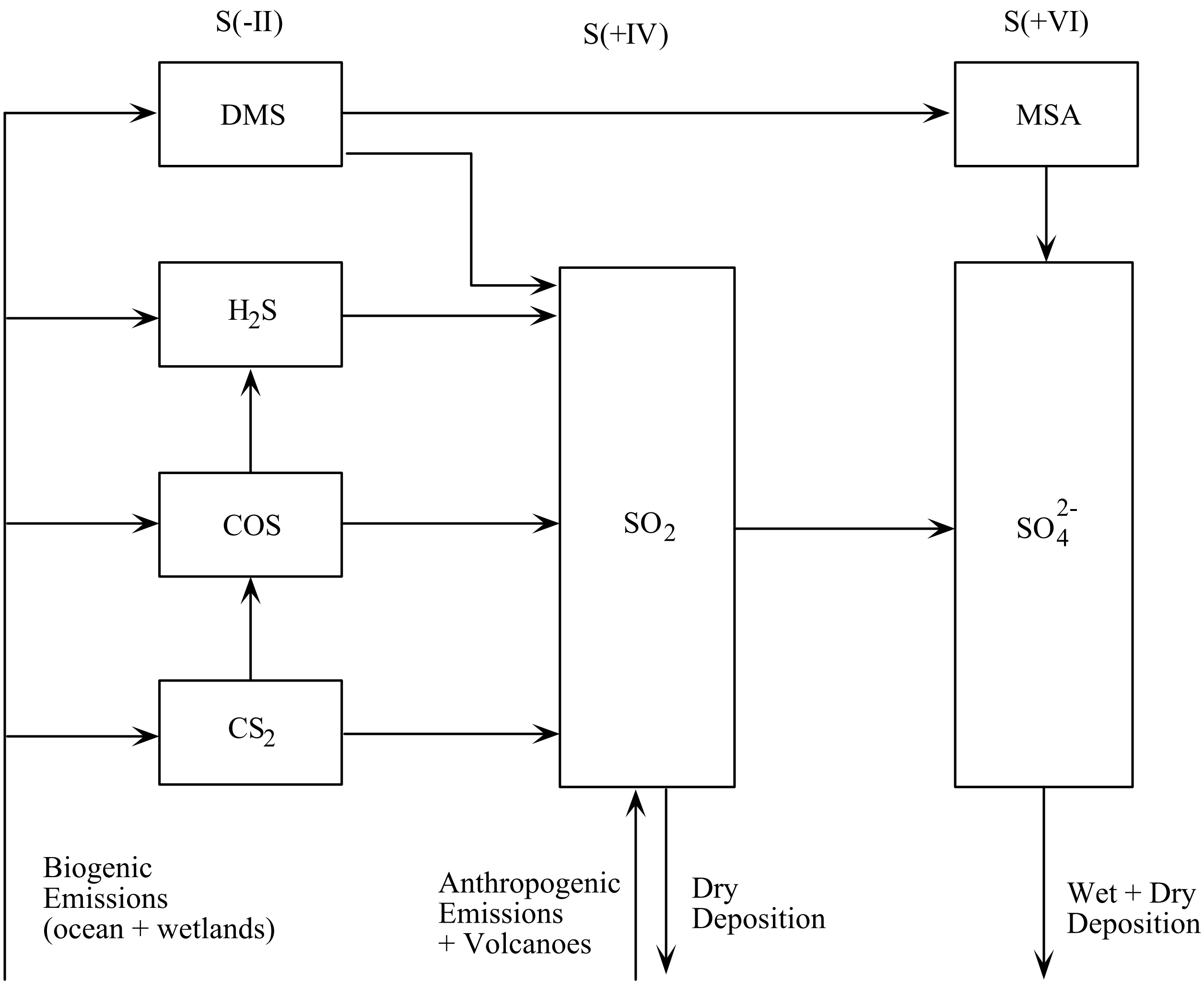

Figure 2. Global Reactions and Transport of Sulfur Species in the Atmosphere

The atmosphere is a small reservoir for sulfur species, only 4.6 x 10^{12} g-S resides in the atmosphere resulting in a mean residence time of only 4.9 days. SO_2 travels 500-2000 km by long range transport, but it does not accumulate in the atmosphere.

Table 2. Global Sulfur Budget[a] to and from the Atmosphere (from Schnoor, 1996)

Sources and Sinks	Tg-S yr^{-1}
Sulfur Sources to Atmosphere[b]	
Volcanoes (SO_2 + H_2S)	20
Dust ($CaSO_4$)	20
Emissions (SO_2)	93
Soil and wetlands (H_2S + COS)	22
Sea salt (Na_2SO_4)	144
Ocean flux (DMS)	43
TOTAL	342
Sulfur Sinks from the Atmosphere	
Wet and dry deposition (terrestrial)	84
Deposition (oceanic)	258
TOTAL	342

[a]10^{12} g S yr^{-1} = 1 million metric tonnes = Tg-S yr^{-1}
[b]CAFO sources are negligible on a global scale

5.3 Methane and Nitrous Oxide, Greenhouse Gas Emissions

Trace gases in the atmosphere include methane (CH_4), and nitrous oxide (N_2O), a small amount which emanates from CAFO sources. Methane and nitrous oxide are potent greenhouse gases with radiative effects 25 and 200 times greater than carbon dioxide, respectively. The global budget for N_2O is the least well known, especially regarding its sinks. Sources include industrial emissions and incomplete combustion of fossil fuels, 1 Tg-N_2O/yr, and biomass burning ~1 Tg-N_2O/yr. Natural ecosystems emit 3-9 Tg-N_2O/yr as an intermediate oxidation state (leakage) from the nitrogen cycle. Fertilized fields are thought to emit up to ten times more N_2O/m^2 than nature. Thus, emissions of N_2O from farm fields receiving large amounts of manure application could be a significant source of N_2O emissions on a global basis. Deforestation and the opening of the soil nitrogen cycle after clear-cutting may account for another large source and, in addition, surface ocean waters could be emitting N_2O because they are ~4% supersaturated with N_2O. Since N_2O is increasing in the atmosphere at only 0.2%/yr, it is thought that there must be a large sink in the soil (oxidation-reduction reactions to $N_{2(g)}$ or $NO_{x(g)}$). Atmospheric sinks for N_2O include a slow oxidation with singlet oxygen to form NO.

Methane is another trace greenhouse gas that occupies 1.7 ppm by volume of the atmosphere. Anthropogenic sources of methane rival natural sources with flooded rice agriculture and ruminant animals as the largest sources (Table 2). Wetlands, including CAFOs and waste lagoons, emit large amounts of methane due to methanogenic conditions in anaerobic sediments and soils (Paterson, 1993). Methane reacts with hydroxyl radicals in the atmosphere as the principal sink. Eventually methane oxidizes to form CO_2, but the reactions are slow. Table 3 is a compilation of some trace gas reactions for carbon species in the atmosphere including methane assuming pseudo-steady state approximations. Methane reacts with ·OH to form formaldehyde, HCHO; and formaldehyde undergoes photolytic oxidation to form carbon monoxide, which eventually yields carbon dioxide. Methane has a long residence time in the atmosphere (5-10 years). Natural emissions of non-methane hydrocarbons (NMHC) are also important sources of formaldehyde and carbon monoxide to the atmosphere. They enter the photolytic cycle and participate in the formation of ozone and smog. NMHCs are primarily C_{10} and higher alkenes that are emitted by vegetation, such as terpene and isoprene. They are responsible for the haze found in the Smokey Mountains of Appalachia.

Table 3. Methane Balance for the Global Atmosphere (Schnoor, 1996)

Sources and Sinks	$Tg\text{-}CH_4/yr$
Sources	
Anthropogenic	
Biomass Burning	44
Coal Extraction	37
Waste Systems	52
Natural Gas Losses	51
Rice Production	99
Ruminant Animals[a]	82
Subtotal	365
Natural	
Biomass Burning	10
Freshwater	5
Hydrates-Clathrates	5
Oceans	10
Termites	21
Wetlands	109
Subtotal	160
TOTAL	525
Sinks	
Oxidation with Hydroxyl Radical	436
Oxidation with Chlorine in Stratosphere	26
Accumulation in Atmosphere	26
Oxidation by Soil Microorganisms	37
TOTAL	525

[a]CAFO sources are a moderate and increasing portion of methane emissions on a global scale.

Greenhouse gas emissions from agriculture compose about 21% of emissions from all sources in Iowa (Table 4). Capturing methane by anaerobic digestion of manure at large feedlots (CAFOs with more than 5000 animals) could reduce methane emissions by 700,000 tons CO_2 equivalents per year in Iowa, about 1% of total greenhouse gas emissions (Ney et al., 1996). Most of this reduction would be possible at large hog lots where 102,000 tons CH_4 per year (~250,000 tons CO_2 equivalents per year) are emitted due to the management of pig manure (Table 5).

Table 4. Greenhouse Gas Emissions from Iowa Agriculture and Total Emissions, Base Year 1990.

Iowa Source of Greenhouse Gas	Gas	Emissions (tons CO_2 equivalent/yr) 1990
Fossil Fuel Combustion On-farm	CO_2	2,540,000
Fertilizer use	N_2O	4,480,000
Manure Management	CH_4	2,590,000
Livestock (domesticated animals)	CH_4	8,360,000
Subtotal Agriculture		17,970,000
TOTAL ALL EMISSIONS		86,700,000

Source: Ney *et al.*, (1996)

Table 5. Iowa Greenhouse Gas Emissions from Animal Agriculture, Base Year 1990

	Iowa Emissions of Methane Tons Methane per Year	
	Manure Management	Direct Emissions from Livestock
Cattle	14,900	352,000
Pigs	102,000	22,400
Poultry	1,770	NA
Sheep	208	4,312
Horses/Mules	170	983
Sub-total	119,000	380,000

Source: Ney *et al.*, (1996)

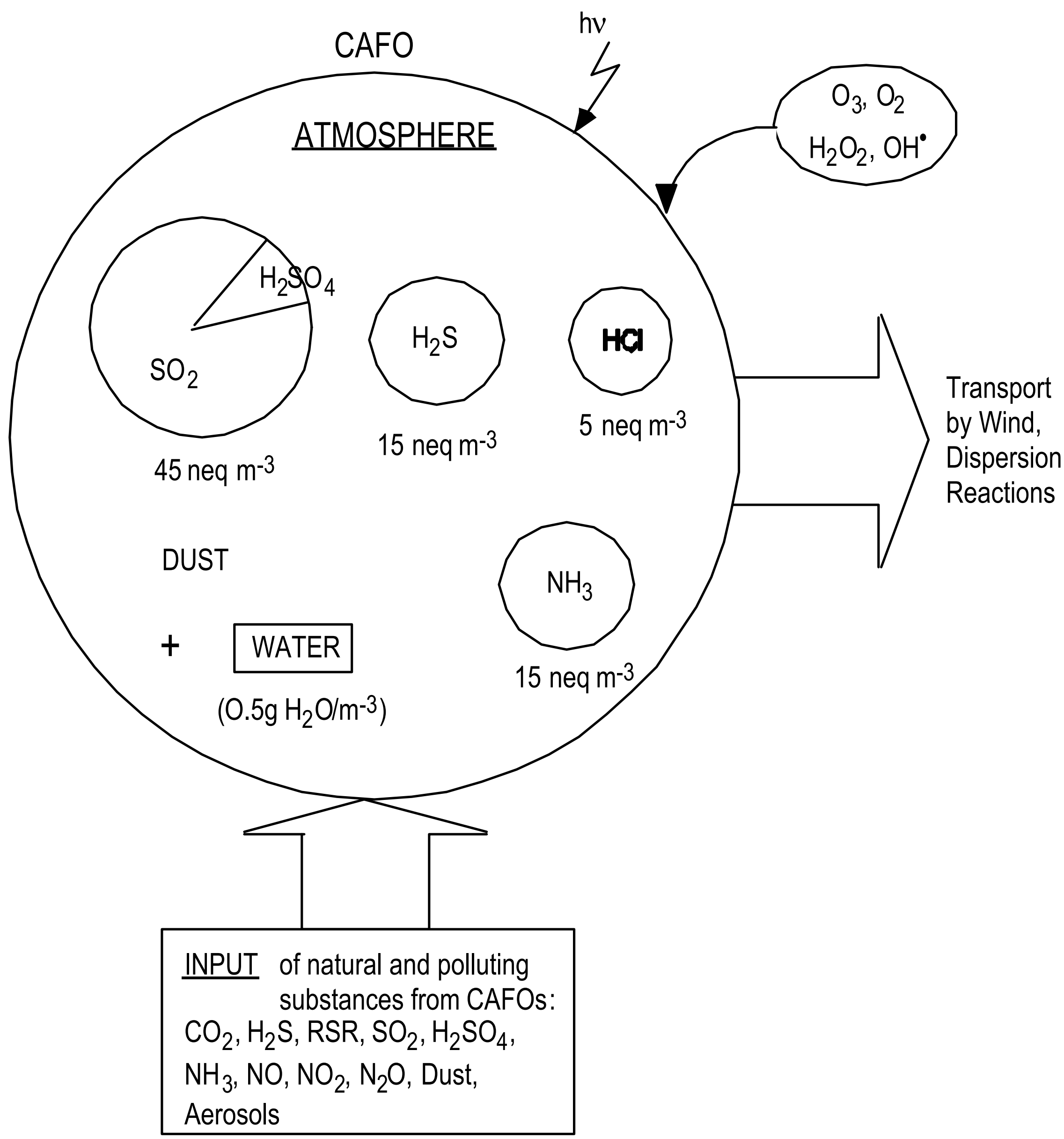

Figure 3. Air Emissions (Inputs) and Formation of Gas Composition in Vicinity of CAFOs.

Rain clouds process a considerable volume of air over relatively long distances and thus are able to absorb gases and aerosols from a large region. Because fog is formed in lower air masses, fog droplets are efficient collectors of pollutants close to the earth's surface. The influence of local emissions (such as NH_3 and H_2S from CAFOs) is reflected in the local fog composition. Fog waters typically contain total ionic concentrations of 0.5-15 meq/ℓ. Remarkably different pH values can be observed in fog. In addition to neutral fogs (pH 5-7) – some of which have very high anion concentrations – other fogs contain acidity (Sigg and Stumm, 1989). Ammonia emissions from feedlots may cause alkaline fog waters. As the fog evaporates, it can decrease visibility (haze) by formation of aerosols, especially ammonium nitrate and ammonium sulfate aerosols.

5.4 Wet and Dry Deposition

Wet deposition occurs when pollutants fall to the ground or water surface by rainfall, snowfall, or hail/sleet. *Dry deposition* is when gases, particles, and aerosols are intercepted by the earth's surface in the absence of precipitation. Wet deposition to the surface of the earth is directly proportional to the concentration of pollutant in the rain, snow, or ice phase.

Wet deposition flux is defined by equation (1) below

$$F_{wet} = IC_w \tag{1}$$

where F_{wet} is the areal wet deposition flux in $\mu g/cm^2$-s, I is the precipitation rate in cm/s (as liquid H_2O), and C_w is the concentration of the pollutant associated with the precipitation in $\mu g/cm^3$. Wet deposition is measured with a bucket collector and a rain gauge. The rain gauge is placed at the receptor site and provides an accurate measure of precipitation rate, I. The wet bucket collector is open only during the precipitation event, and its contents are analyzed for pollutant concentration, C_w.

The concentration of pollutants in wet deposition is due to two important effects with quite different physical mechanisms:

- aerosol particle scavenging
- gas scavenging

Aerosols begin their life cycle after nucleation and formation of a submicron hygroscopic particle, e.g., $(NH_4)_2SO_4$, which hydrates and grows very quickly due to condensation of water around the particle. At this stage, it is neither solid nor liquid, but merely a stable aerosol with a density between 1.0-1.1 g/cm^3. Mass quantities of air are processed by clouds, creating updrafts which cleanse the air of pollutants. Cloud droplets are very small, on the order of 10 μm in diameter. Typically one million cloud droplets are needed to comprise a 1 mm diameter raindrop. Assuming an average spacing of 1 mm between cloud droplets, condensation of 10^6 cloud droplets into a 1 mm raindrop would scavenge enough air for a washout ratio of 10^6.

$$W = \frac{C_w}{C_{ae}} \tag{2}$$

where C_w is the concentration of the pollutant in precipitation water in $\mu g/cm^3$, C_{ae} is the concentration of the pollutant associated with aerosol droplets in air in $\mu g/cm^3$, and W is the washout ratio for aerosols, dimensionless (cm^3 air/cm^3 precipitation).

Details of the physics of the scavenging process are beyond the scope of this report, but reference texts include Schwartz and Slinn (1992), and Pruppacher *et al.* (1983), and Eisenreich (1981). Because clouds process such large quantities of air and pull-up polluted air from the surface, washout is caused predominantly by in-cloud processes. Washout ratios for particles are typically on the order of 10^5-10^6. In other words, they are removed rapidly by cloud processes and/or rained-out efficiently. *Rainout* sometimes refers to below-cloud processes, whereby pollutants are scavenged as raindrops fall through polluted air.

Washout for gas scavenging operates by a different mechanism than aerosol particle scavenging. Here, Henry's law is applicable because chemical equilibrium for absorption processes in the atmosphere is on the time scale of one second. Gas scavenging, therefore, is reversible, while aerosol scavenging is an irreversible process. Ammonia is quite efficiently scavenged by washout processes and absorbed into the aqueous phase of water vapor or precipitation where it forms ammonium ions, NH_4^+. Hydrogen sulfide is less efficiently scavenged.

If we express Henry's constant K_H in units of M atm^{-1}, the following equations apply for Henry's law and the washout ratio.

$$C_w = K_H \, p_{atm} \tag{3}$$

$$W = \frac{C_w}{C_g} = K_H RT \tag{4}$$

where C_w is the concentration in the water phase (M), p_{atm} is the atmospheric partial pressure (atm), W is the washout ratio (dimensionless, i.e., ℓ H_2O/ℓ gas), C_g is the concentration in the gas (mol/ℓ gas) and RT is the universal gas law constant times temperature (24.46 atm/M at 25°C).

Some estimates for washout ratios of ammonia gas and selected pesticides are presented in Table 6. Henry's constants are provided in Schwarzenbach *et al.* (1993). In general, washout ratios are large for soluble and polar compounds, intermediate for semi-volatiles (such as DDT, dieldrin, dioxin, and PCBs), and low for volatile organic chemicals. Semi-volatile pollutants are an interesting case because these gases can be transported long distances and recycled many times before being deposited in polar regions by a "cold-trap" effect. Although washout ratios of gases by snow are smaller than by rain, there can be appreciable liquid water contained in snow that absorbs gases. Adsorption of gases to snowflake surfaces can also be significant.

The washout ratio does not give enough information to calculate the mass of pollutant in a column of air that is actually "washed-out" by rain, but such information is provided in reference texts such as Schnoor (1996).

Dry deposition takes place (in the absence of rain) by two pathways.

- aerosol and particle deposition
- gas deposition

There are three resistances to aerosol and gas deposition: 1) aerodynamic resistance, 2) boundary layer resistance, and 3) surface resistance. Aerodynamic resistance involves turbulent mixing and transport from the atmosphere (~ 1 km elevation) to the laminar boundary layer in the quiescent zone above the earth's surface. Boundary resistance refers to the difficulty of pollutant transport through the laminar boundary layer, and surface resistance involves the physical and chemical reactions that may occur at the surface of the receptor (sea surface, vegetation, snow surface, etc.). Dry deposition velocity encompasses the electrical analog of these three resistances in series

$$V_d = \frac{1}{r_a + r_b + r_s} \quad (5)$$

where V_d is defined as the dry deposition velocity (cm/s), r_a is the aerodynamic resistance, r_b is the boundary layer resistance, and r_s is the resistance at the surface.

Table 6. Estimates of Washout Ratios for Selected Gases, 25˚C (Schnoor, 1996)

Chemical	Henry's Constant K_H, M-atm^{-1}	Washout Ratio $W = K_H RT^*$
ammonia	63	1,500
aldrin	100	2,450
benzene	0.18	4.4
benzo(a)pyrene	830	20,300
CCl_4	0.042	1.0
dioxin	20	490
DDT	105	2,570
dieldrin	89	2,200
di-n-butyl phthalate	780	19,000
methane	0.0015	0.037
naphthalene	2.3	56
parathion	2,630	64,000
trichloroethene	0.093	2.3
toluene	0.15	3.7
2,2',5,5'-PCB	3.5	86

* RT = 24.46 atm/M at 25˚C

The deposition velocity is affected by a number of factors including relative humidity, type of aerosol or gas, aerosol particle size, wind velocity profile, type of surface receptor, roughness factor, atmospheric stability, and temperature. V_d increases with wind speed because sheer stress at the surface causes increased vertical turbulence and eddies. A summary of dry deposition measurements and a comparison of collector surfaces are given by Davidson and Wu (1990).

For aerosol particles, the deposition velocity is dependent on particle diameter. A minimum deposition velocity (~10^{-2} cm/s) exists for fine aerosol particles in the size range from 0.1-1.0 μm. Larger particles are deposited much more rapidly.

In reality, aerosols change constantly due to changes in relative humidity; they evaporate or condense into water continually. The mass median diameter (MMD) is a measure of the particle size distribution. Milford and Davidson (1985) showed a general power-law correlation for the dependence of V_d on particle size

$$V_d = 0.388\ MMD^{0.76} \quad (6)$$

where V_d is the deposition velocity in cm/s and MMD is the mass median diameter of the particle in μm. Table 5 is a compilation of dry deposition velocities for chemicals of interest from Davidson and Wu (1990).

In general, gases that react at the surface (e.g., H_2S, SO_2, HNO_3, HCl, and O_3) tend to have slightly higher deposition velocities, on the order of 1.0 cm/s. HNO_3 vapor has a very large deposition velocity because there is no surface resistance -- it is immediately absorbed and neutralized by vegetation and/or water. Some gases such as NO_x display higher V_d values in daylight because vegetation transpires at that time, and gas exchange through the stomata serves to increase the concentration gradient and the flux at the leaf surface.

The receptor surface is critical. Deposition velocities in Table 7 are mostly to natural earth surfaces. Surrogate surfaces tend to underestimate the actual dry deposition because of differences in reactivity at the surface, differences in surface area, and aerodynamic differences around the collector. Natural vegetation and trees are relatively efficient interceptors of gases and particles based on specific surface areas. SO_2 dry deposition velocity for a coniferous forest may be several times higher than for an open field or a snow field. Buffer strips of trees around CAFOs could intercept and remove some of the gases and particles by dry deposition.

Table 7. Dry Deposition Velocities for a Number of Aerosol Particles and Gases

Pollutant	V_d, cm/s^{-1} Typical Range	V_d, cm/s^{-1} Typical Median	Conditions
$SO_2(g)$	0.3-1.6	0.95	to natural vegetation
	0.04-0.22	0.13	to snow field
SO_4^{2-}	0.01-1.2	0.55	submicron aerosols in field (micrometeorological)
	0.01-0.5	0.26	to surrogate surfaces
NO_3^-	0.1-2.0	0.7	aerosol particle deposition
NH_4^+	0.05-2.0	0.8	aerosol particle deposition
$HNO_{3(g)}$	1-3	1.4	gas, no surface resistance
$NO_{x(g)}$	0.01-0.5	0.05	night, closed stomata
	0.1-1.7	0.6	day, open stomata
Cl^-	1-5	2	particles, MMD = 1-4 μm
$HCl_{(g)}$	0.6-0.8	0.7	sorption by dew
$O_{3(g)}$	0.01-1.5	0.4	by measured gradients
Pb	0.1-1.0	0.26	aerosol particle deposition from autos MMD < 1 μm
Crustal metals (Ca, Mg, K, Fe, Mn)	0.3-3.0	1.5	associated w/coarse particles MMD = 1-4 μm
Enriched (anthropogenic metals-Ag, As, Cd, Cu, Zn, Pb, Ni)	0.1-1.0	0.3	assoc. w/fine particles, enriched MMD < 1 μm
Fine Particles	0.1-1.2	0.4	submicron particles

Source: from Davidson and Wu (1990)

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6.0 Adverse Health Effects
6.1 Toxicology

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6.1 Abstract - Toxicology

Valid evaluation of the health effects of airborne substances released from animal production units should be based on the important and well-established toxicological principles of dosage and response. Dosage is the most important factor that determines response to poisons. Toxicity is the quantitative amount of toxicant required to produce a defined effect, but the hazard or risk of toxicosis depends not only on the inherent toxicity of the agent, but on the probability of exposure to the toxicant under conditions of use. Acute, subacute, and chronic toxicity are different chronological quantitations of chemical toxicity and are determined by relative dosage and time of exposure. Many factors can alter animal or human response to toxicants, including those inherent in the toxicant, the organism, the environment and the combinations of these major factors. Toxicological evaluation depends heavily on determination of exposure and evidence for the contribution of interacting factors that can alter toxicity. Quantitative expressions of toxicity and exposure are essential for thorough toxicological evaluation and prognosis.

Response to exposure by airborne toxicants is likely to involve the respiratory system because it is a portal of entry. Study of CAFO issues suggests consideration of the mechanisms of injury by volatile agents and particulates, as well as understanding the potential effects of both acute and chronic exposure. Respiratory system effects are manifest in relatively limited ways (bronchoconstriction, pulmonary edema, asthma, carcinogenesis), and careful attention must be given to evidence for cause and effect from among a wide range of insults and levels of exposure. Similar considerations are important for systemic effects that are manifested in other parts of the body.

Laboratory animals are often as experimental models of human disease to help establish the mechanism of action and the correlation between exposure levels of airborne toxicants and clinical response. Clinical response to these pollutants depends not only on the concentration of the specific compound, but also the frequency and duration of exposure.

Studies of aerial ammonia in laboratory animals have demonstrated dose-effect and duration-effect patterns for damage to the respiratory tract similar to that observed in humans. Acute exposures to moderate concentrations of ammonia irritate the upper respiratory tract. Prolonged or repeated exposures to lower levels of ammonia produce inflammation and lesions of the respiratory tract. Exposures to high concentrations of ammonia result in severe damage to the upper and lower respiratory tract and alveolar capillaries.

Controlled studies with hydrogen sulfide in laboratory animals have shown that levels of 500 ppm or greater are likely to be lethal, similar to the response observed in humans. Exposure to sub-lethal levels of hydrogen sulfide have produced progressive effects ranging from increased respiratory rate, to pulmonary edema, to histopathological changes in the nasal cavity and lung tissue.

Endotoxins, glucans, and microorganisms maybe important components of bioaerosols associated with animal production units. Inhalation of these compounds have been shown to produce respiratory system effects including airway constriction and obstructive breathing pattern, inflammatory tissue responses, and overt infection of lung tissue.

6.1 Toxicology

6.10 Overview of Toxicology

Toxicology is the study of poisons, and their effects on living organisms. This includes an understanding of sources of poisons, circumstances of exposure, their effects, diagnosis and treatment and the application of management or educational strategies to prevent poisoning. More than many of the specialties in veterinary medicine, toxicology is based on the important principle of dose and response. Response is dependent not only on presence of a potential toxicant but on the amount of exposure as well.(Osweiler, 1996) With emphasis in this report on accountability of Concentrated Animal Feeding Operations (CAFOs) for substances released from animal production units, there is increasing need to be aware of and apply the dosage and response principle to best estimate the need for regulation or remediation.

Determinants of exposure that affect dosage may be more than simply the gross amount of material with access to animals or man. Rather, the effective dosage at a susceptible receptor site determines the ultimate response. Thus, environmental factors that influence exposure, species differences in organisms within an exposure area, vehicle differences that affect absorption, specific drug or chemical interactions that potentiate response, and organ dysfunction that limits elimination may all be factors which influence the ultimate dosage and the outcome of exposure. (Osweiler, 1996)

Toxicological Principles Of Evaluation For Cafo Issues

A **poison** or **toxicant** is any natural or synthetic solid, liquid or gas that when introduced into or applied to the body can interfere with homeostasis of the organism or life processes of cells of the organism by its own inherent qualities, without acting mechanically and irrespective of temperature. For CAFOs, toxicants considered are natural products that would normally be handled by ecological assimilation, but may be locally in unnatural or excessive concentration. Knowledge of the chemical nature and specific effects of toxicants and their combinations is the only certain way to assess hazard from such exposure. Suggestions about potential adverse effects of natural products from livestock waste may be gained from comparative experimental studies, from know effects of substances at high concentrations within CAFOs, and from well-controlled and properly interpreted epidemiological studies. This chapter will review the known biological effects of compounds identified in CAFOs, and will also present evidence gained from epidemiological studies.

Toxicological conventions should be followed in assessment of risk to different populations. **Toxicity** is the quantitative amount or dosage of a poison that will produce a defined effect. For example, the acute lethal dosage of hydrogen sulfide to swine could be described as a **concentration** in air, e. g. 1,000 parts-per-million or as the equivalent amount on a body weight basis. Toxicity values do not describe the biological effects, but only the quantitative amount (dosage) required to produce a defined effect (e.g. death, respiratory distress, immune suppression, etc). **Dosage** is the correct terminology for toxicity expressed as amount of toxicant per unit of body weight. Commonly accepted dosage units are mg/kg body weight or moles or micromoles of agent/per kg body weight. In comparative toxicology, relative effects in large and small animals relate dosage to the body surface area, which is approximately equal to (body weight) $^{2/3}$. This relationship, and others relevant to interspecies comparisons, should always be considered when comparing laboratory or farm animal toxicity data against risk for humans. Generally, as animals increase in weight, the body surface area increases proportionally less, and this may affect the rate of metabolism, excretion and receptor interaction with toxicants. For many toxicants, larger animals will be poisoned by relatively lower body weight dosages than are smaller mammals. (Eaton and Klaassen, 2001; Osweiler, 1996)

From a public health and diagnostic toxicology perspective it is essential to know what exposure level will not cause any adverse health effect. This level is usually referred to as the "no observed adverse effect level" (NOAEL). (Eaton and Klaassen, 2001) Usually a NOAEL in laboratory animals is based on chronic exposures ranging from ninety days to two or more years depending on the species. The inhalation toxicity for gases or aerosols, including particulates, is often expressed as the concentration of material (i.e. the weight of compound per volume or weight of air). The no-effect level is the largest dosage or concentration that does not result in detrimental effects. In industrial hygiene, the concept of protecting human health from exposure is quantified to an assumed normal work day exposure and given a value called the Threshold Limit Value (TLV), which includes a safety factor between exposure allowed and concentrations where adverse effects may be expected.

Response to Toxicants

Toxicant evaluation is usually classified according to chronological scale that accounts for both dosage and response. **Acute toxicity** refers to effects of a single dose or multiple doses measured during a twenty-four-hour period. Toxic effects apparent over a period of several days or weeks are classified as **subacute**. **Subchronic** toxicity refers to toxic effects that occur between 30 days and ninety days exposure. **Chronic** effects are those produced by prolonged exposures of three months to a lifetime. Chronic effects are affected by the cumulative tendencies of the toxicant. The ratio of the acute to chronic LD_{50} dosage is called the **chronicity index**. (Eaton and Klaassen, 2001) Compounds with strong cumulative properties have larger chronicity index. The potential for individual products from CAFOs to cause cumulative effects should include evaluation of their cumulative potential or chronicity index. Conversely, organisms may develop **tolerance** for a compound such that repeated exposure increases the size of the dose required to produce lethality. For example, the single dose LD_{50} of potassium cyanide in rats is 10 mg/kg, while rats given potassium cyanide for ninety days are able to tolerate a dosage of 250 mg/kg without lethality.

Toxicity and Risk

The concept of risk or hazard is important to toxicology. While toxicity defines the amount of a toxicant that produces specific effects at a known dosage, hazard or risk is the probability of poisoning under the conditions of expected exposure or usage. Compounds of high toxicity may still present low hazard or risk if exposure to the toxicant is limited. CAFO risk evaluation should include estimation of dosage at remote or off site locations, and measurement or estimation of exposure at such locations is essential. Factors discussed in previous chapters relating to dispersion and dilution in the environment are essential in estimating the risk for a compound, even if it is of high inherent toxicity. Moreover, binding of toxicant gases to particulates may either reduce or increase their toxic properties so that risk is a function of all factors and interactions.

Factors That Affect Response To Toxicants

Many factors inherent in the toxicant, the animal or the environment can alter a toxicity value determined under defined experimental conditions. The toxicity of a compound may vary with the route of exposure. Usual routes of exposure to environmental agents are oral, dermal and inhalation. Gases are absorbed directly through pulmonary membranes, but aerosols including dusts may be deposited in lower airways or lungs if they are in a range between 0.1 and 5.0 um. Systemic retention occurs when macrophages laden with particles gain access to the pulmonary lymphatic drainage. Retention of inhaled particles in the gastrointestinal tract can occur when large particles trapped by cilia and mucus in the nasopharynx and trachea are swallowed. (Eaton and Klaassen, 2001)

Many environmental and physiological factors can influence the toxicity of compounds, and such factors, or others possibly unknown, can substantially influence response to toxicants. Accurate evaluation of CAFO risk to both on-site and off-site persons must consider multiple factors and their interactions to properly support regulatory and remedial activity. Some examples of factors that alter response to toxicants are presented in Table 1

TABLE 1. SELECTED FACTORS THAT MAY ALTER RESPONSE TO TOXICANTS	
Alteration or Change	**Mechanism or Example**
Changes in chemical composition or salts of inorganic agents	Toxicity of metals may be altered by valence state. Sodium salts are more water soluble than parent compounds, promoting absorption.
Instability or decomposition of chemical	Volatile compounds can decompose or change to more toxic form upon exposure to sunlight, as with nitrogen and nitrogen oxides.
Ionization	Generally, compounds that are highly ionized are poorly absorbed and thus less toxic. The pH of the source of pit gases may influence ionization of some products.
Vehicle effects	Non-polar and lipid soluble vehicles usually increase toxicity of toxicants by promoting absorption and membrane penetration.
Protein binding	Binding to serum albumin is common for many drugs and toxicants, limiting the bioavailability of the agent and reducing toxicity.
Chemical or drug interactions	Chemicals may directly bind, inactivate or potentiate another. One chemical may also induce microsomal enzymes to influence the metabolism of another.
Biotransformation	Prior exposure to the same or similar chemical may induce increased metabolic activity of microsomal mixed function oxidases (MFOs). Foreign compounds activated by MFOs can then be conjugated by Phase II metabolism and excreted. If toxicants are activated by MFO activity, then toxicity may be increased. Liver disease, very young or very old animals, and specific breeds or strains of animal can alter ability of MFO to begin metabolism followed by Phase II

	detoxification of foreign compounds.
Liver disease	Reduced synthesis of conjugating or binding agents (glutathione, metallothionein), essential proteins and coagulation factors may alter response to absorbed chemicals.
Nutrition and diet	Vitamin C and vitamin E can aid in scavenging of free radicals and repair of cellular protective mechanisms.

Respiratory System Response to Injury

Response of airways and lung to injury is dose dependent and expressed in chronological terms as acute, subacute or chronic. Response of the respiratory tract to toxicants is manifest in relatively few ways in response to many different chemicals, and a few specific mechanisms of injury are known. (Haschek and Rousseaux, 1998; Witschi, 2001)

Mechanisms of Respiratory System Injury

Respiratory damage depends on relatively few recognized molecular and cellular mechanisms that account for a wide variety of toxicant exposures. Many recognized effects are related to the oxidative burden imposed on the respiratory tract. (Witschi, 1997) This includes generation of unstable and reactive free radicals that lead to oxidative chain reactions and subsequent cellular damage or destruction. Cellular injury then results in release of microsomes and flavoproteins, neutrophils, moncytes and macrophages that can sustain the conversion of molecular oxygen to reactive oxygen metabolites. Many of these effects are an excessive response to what is a normal respiratory defense mechanism against microorganisms and low- or high-molecular-weight antigenic materials. Immunologic consequences are triggered when foreign materials in the respiratory tract sensitize the lung or airways to further exposure of the same material. (Witshci, 2001). Further consequences of oxidative damage or covalent binding in the pulmonary systems can result from damage and cross linking of DNA with potential subsequent development of carcinogenesis. The consequences of these mechanisms can be acute or chronic respiratory damage and the physiological dysfunction that accompanies each.

Acute Respiratory Injury

Acute airway damage in the transport passages (nasopharynx, trachea, bronchi, bronchioles) is reflected as bronchoconstriction and/or excess or reduced mucus and ciliary function. (Haschek and Rousseaux, 1998; Witschi, 2001). Response to irritants in nasal passages can cause acute or chronic rhinitis or, at higher concentrations, pause in respiration which develops as a reflex protective mechanism. Autonomic nervous system response to irritants is associated with acute reflex contraction of trachea and bronchi, resulting in decreased airway diameter and increased resistance to air flow. This results in wheezing, coughing, dyspnea and reduced exercise tolerance. This response is most likely triggered by irritant gases with moderate water solubility. Effects of short-term exposure resolve quickly when the irritant gas is no longer present and if no permanent cellular damage has occurred; long-term exposure may lead to chronic effects.

Acute lung damage can result in two major effects on lung tissue. Toxic pulmonary edema, which is characterized by alveolar or interstitial fluid accumulation and a thickened alveolar-capillary interface results in reduced oxygen and carbon dioxide exchange. Highly water-soluble irritant gases, including ammonia and hydrogen sulfide, which reach the lung parenchyma can damage cellular membranes and allow fluid leakage leading to pulmonary edema. Inflammatory response and cellular accumulation may accompany the edema and, if severe, result in prolonged changes including fibrogenesis. Acute alveolar endothelial damage and necrosis stimulates Alveolar Type II cell proliferation. These cells are physically thicker than Type I cells, and as immature replacements of Type I cells (alveolar endothelium) markedly reduce oxygen and carbon dioxide exchange (Witschi, 2001).

Chronic Respiratory Injury

Chronic response to injury may come from excessive and prolonged acute injury or from low-level or subclinical damage. In either event, manifestation is commonly as fibrosis or other chronic inflammatory change, emphysema, asthma or carcinogenesis.

Fibrosis is the result of excessive production of collagen in lung parenchyma and can occur at the alveolar, alveolar duct and bronchiolar levels. Type I and III collagen constitute approximately 90 percent of lung collagen. Increases in collagen, especially Type I, increase stiffness of the lung and reduce compliance, with severe fibrosis resulting in reduced vital capacity and reduced exercise tolerance.

Emphysema is characterized by "abnormal enlargement of the airspaces distal to the terminal bronchiole, accompanied by destruction of the walls, without obvious fibrosis". (Snider et al, 1985) Emphysema arises from interference with or lack of alpha$_1$-antiprotease, leading to loss of pulmonary elastin and subsequent alveolar wall breakdown. This leads to reduced alveolar surface and hyperinflation of alveoli and lungs with excessive compliance.

Asthma is characterized by increased airway activity with excessive contraction of large airways in response to irritants. Effects may be initiated by exposure to antigens or by chemicals that serve as haptens, with contributing influences by inflammatory cells and cytokines (Barnes et al 1998). Effects are mild to severe dyspnea, which can be acute, recurring and influenced by inhalation of a variety of pollutants (Witschi, 2001).

Respiratory carcinogenesis, especially lung cancer in humans is common and associated with environmental, industrial and personal exposures to a variety of chemicals. For most lung cancers, there is likely a dose-response relationship but clinical disease is often manifested later in life after long-term exposure. Animal studies are helpful in definition of mechanisms and in selected dose-response considerations. However, animal studies are important to intepret carefully in the context of significant differences in laboratory animal susceptibility and for the dosages used in experimental studies compared to ambient exposures of human populations (Hahn, 1997; Malkinson, 1998).

Systemic Effects of Airborne Toxicants

Airborne toxicants can affect systems other than or in addition to the respiratory tract. Lung is an efficient absorption organ and readily transports volatile compounds to the systemic circulation. Neurological and immune system consequences may occur secondary to inhalation exposure. A limited amount of xenobiotic metabolism is possible in lung, so that some bioactivation of toxicants can occur upon first pass pulmonary absorption. Effects of absorbed volatile agents will depend on the eventual target organs and susceptible receptors. These specific effects in target tissues and organs will be discussed in detail in subsequent sections of this chapter.

6.1.1 Toxicology of ammonia

Experimental studies indicate that the concentration of aerial ammonia which is acutely lethal to laboratory animals is dependent on the duration of the exposure. The lethal concentration of ammonia in rats and mice increases 5-10 times as the duration of exposure decreases from 16 hours to several minutes (Hilado et al. 1977; Kapeghian et al. 1982; Weedon et al. 1940). Exposure frequency also appears to be an important factor in determining lethality. Continuous exposure to 653 ppm of ammonia for 25 days resulted in nearly 64% lethality in rats, whereas intermittent exposure to nearly twice this concentration was tolerated for 42 days (Coon et al. 1970). It also appears that male rats are more sensitive than female rats to the lethal effects of aerial ammonia (Appelman et al. 1982).

Studies in laboratory animals have demonstrated dose-effect and duration-effect patterns for damage to the respiratory tract similar to that observed in humans. Acute exposures to moderate concentrations of ammonia ($\leq$1000 ppm) irritate the upper respiratory tract, whereas exposures to high concentrations ($\geq$4000 ppm) result in severe damage to the upper and lower respiratory tract and alveolar capillaries (Coon et al. 1970; Kapeghian et al. 1982; Mayan and Merilan 1972; Richard et al. 1978a,b; Schaerdel et al. 1983). Prolonged or repeated exposures to lower levels of ammonia ($\geq$150 ppm) produce inflammation and lesions of the respiratory tract (Broderson et al. 1976; Coon et al. 1970).

No overt symptoms of neurological disorders were reported in guinea pigs or monkeys that were exposed to up to 1105 ppm ammonia for 6 weeks (Coon et al. 1970). However, acute exposure to low levels of ammonia (100 ppm) has been shown to depress free-access wheel running behavior in rodents (Tepper et al. 1985). This may represent avoidance of sensory or upper airway irritation, but these same effects can be seen after injection of ammonium salts.

6.1.2 Toxicology of hydrogen sulfide

Controlled studies using dogs, rats, mice, and rabbits exposed acutely to high concentrations of hydrogen sulfide gas for various periods of time have shown that levels of 500 ppm or greater are likely to be lethal, similar to the response observed in humans exposed to high levels (Beck, 1979; Elovaara, 1978; Higuchi and Fukamachi, 1977; Haggard, 1922; Lopez,1987,1988a,1988b,1989; Kage, 1992; Khan, 1990; Prior, 1988, 1990; Savolainen, 1980; Smith and Gosselin, 1964; Tansy,1981).

In addition to an increase in respiration rate that was noted in rats exposed to 100-200 ppm hydrogen sulfide for 1 hour (Higachi and Fukamachi, 1977), a number of histological and biochemical changes were noted in the respiratory tissues and fluids of rats acutely exposed to 200, 300 or 400 ppm hydrogen sulfide for 4 hours (Lopez, 1987; Green,1991). Histopathological changes were reported in the nasal cavity of rats exposed to greater than 200 ppm hydrogen sulfide for 4 hours (Lopez, 1988b). Moderate-to-massive pulmonary edema was evident in rats exposed to 375 ppm hydrogen sulfide for 4 hours (Prior, 1990), and slight pulmonary congestion was found in rats exposed to 75 ppm hydrogen sulfide for 1 hour (Kohno, 1991). Significant decreases in numbers of viable pulmonary alveolar macrophages were noted in the lung lavage fluid of rats exposed for 4 hours to 400 ppm hydrogen sulfide (Khan, 1991).

The effects of intermediate-duration exposures to hydrogen sulfide have been examined in rats, mice, and pigs. Respiratory effects were not observed in two strains of rats exposed to hydrogen sulfide at concentrations up to 80 ppm 6 hours/day, 5 days/week, for 90 days (CIIT 1983b, CIIT

1983c). In contrast to rats, inflammation of the nasal mucosa described as minimal to mild was observed in mice exposed to hydrogen sulfide at 80 ppm (CIIT 1983a). Respiratory effects were not observed at 30.5 ppm. No mortality was noted during 90-day studies in which rats and mice were exposed for 6 hours/day, 5 days/week, to up to 80 ppm hydrogen sulfide (CIIT 1983b, 1983c). (CIIT 1983a).

Guinea pigs exposed daily to 20 ppm of hydrogen sulfide for 11 days developed fatigue, somnolence, and dizziness (Haider, 1980). Neurochemical analyses revealed decreased cerebral hemisphere and brain stem total lipids and phospholipids. Lethargy was observed in rats following exposure to 400 ppm of hydrogen sulfide for 4 hours (Lopez, 1988b).

Rats were exposed to average concentrations of 100-200, 200-300, 300-400, or 400-500 ppm hydrogen sulfide; at 200-300 ppm, a decreased response rate in a discriminated avoidance task was observed (Higuchi and Fukamachi, 1977). Except at the highest concentrations tested, the response rates and percent avoidances recovered rapidly when ventilation with clean air was provided, although even at 400-500 ppm, they were almost normal the following day. When these same animals were tested for Sidman-type conditioned avoidance response at response-shock intervals of 10 or 30 seconds, an inverse relationship between hydrogen sulfide concentration and response rate was noted; this effect dissipated when exposure stopped (Higuchi and Fukamachi 1977). Excitement was observed when mice were exposed to 100 ppm of hydrogen sulfide for 2 hours at 4-day intervals (Savolainen, 1980). Exposure also resulted in decreased cerebral ribonucleic acid (RNA), decreased orotic acid incorporation into the RNA fraction, and inhibition of cytochrome oxidase. An increase in the glial enzyme marker, 2',3'-cyclic nucleotide-3'-phosphohydrolase, was seen. Neurochemical effects have been reported in other studies. Decreased leucine uptake and acid proteinase activity in the brain were observed in mice exposed to 100 ppm hydrogen sulfide for 2 hours (Elovaara, 1978). Inhibition of brain cytochrome oxidase and a decrease in orotic acid uptake were observed in mice exposed to 100 ppm hydrogen sulfide for up to 4 days (Savolainen, 1980).

The intermediate-duration effects of hydrogen sulfide on neurological function were examined by the measurement of motor and sensory nerve conduction velocities of the tail nerve or morphology of the sciatic nerve but, no neurotoxic effects were observed in rats exposed to 50 ppm hydrogen sulfide for 5 days a week, for 25 weeks (Gagnaire, 1986).

Neurologic function and neuropathology were evaluated in rats exposed to 0,10.1, 30.5, or 80.0 ppm hydrogen sulfide for 6 hours/day, 5 days/week, for 90 days (CIIT, 1983c). Although absolute brain weights were decreased (5%) in rats exposed to 80 ppm hydrogen sulfide in this study, there were no treatment-related effects on neurological function or neuropathology. In addition, no signs of neurotoxicity were noted in a similar study in which mice and rats were exposed to 0, 10.1, 30.5, or 80.0 ppm hydrogen sulfide for 90 days (CIIT, 1983a, CIIT, 1983b).

6.1.3 Toxicology of bioaerosols

Endotoxin

The bioaerosol constituent present in swine barns that has been most studied is endotoxin. Endotoxin is a lipopolysaccharide (LPS) component of the outer cell wall of Gram negative (Gm-) bacteria. Endotoxin has been shown in both humans (Schwartz et al 1995, Jagielo et al 1996, Deetz et al 1997) and animals (Schwartz et al 1994, Jagielo et al 1996, Thorne et al 1998, Thorne 2000) to be a potent pro-inflammatory agent through its ability to activate the innate immune system. Endotoxin is an amphipathic molecule consisting of a phospholipid fraction, called lipid A, bound to a polysaccharide. The polysaccharide has two components: the O-antigen and the core polysaccharide (Rietschel et al 1996). In swine CAFOs, endotoxin most likely includes pieces of other membrane materials in association with LPS. The biological activity of endotoxin rests largely with the lipid A fraction. Once inhaled, endotoxin will interact with macrophages or soluble CD14 inducing signal transduction via the TLR-4 receptor (Medzhitov et al 1997, Faure et al 2000, Gao et al 1998). Through multiple transcription factors (Gao et al 1998), the initiation of transcription of several genes coding for inflammatory mediators can trigger the production of pro-inflammatory cytokines. The cytokines most associated with inhalation of endotoxin are Interleukin (IL)-1, tumor necrosis factor (TNF)a, IL-6, IL-8 (humans), and MIP-2 (mice) (Thorne et al 1998, Deetz et al 1997). Recent evidence suggests a regulatory role for IL-10, IL-12 (Shnyra et al 1998), and interferon ? (IFN?) (Kline et al 1998). An aggressive response to endotoxin exposure results in a cascade of events producing airway narrowing and an obstructive breathing pattern (Pauwels et al 1990). Chronic inhalation exposure in mice has been shown to induce airway remodeling and collagen formation (George et al 2001).

Glucans

Studies of the past five years have provided evidence that glucans may also be important immunomodulators (Rylander 1999, Fogelmark et al 1997). ß(1? 3)-glucans are glucose polymers with variable molecular weight and degree of branching that may appear in triple helix, single helix or random coil structures (Williams 1994). ß(1? 3)-glucans originate from a variety of sources, including fungi, bacteria, and plants (Stone and Clarke 1992). They are water insoluble structural cell wall components of these organisms, but may also be found in extracellular secretions of microbial origin. Glucans may account for up to 60% of the dry weight of the cell wall of fungi, of which the major part is ß(1? 3)-glucan (Klis 1994). Recently it has been suggested that ß(1? 3)-glucans play a role in bioaerosol induced inflammatory responses and resulting respiratory symptoms (Williams 1994, Rylander et al 1992, Fogelmark et al 1994).

Microorganisms

Infectious microorganisms may present an occupational hazard when inhaled (Thorne 2001, Douwes et al 2002). Fortunately, airborne transmission of zoonotic pathogens at sufficient doses to cause disease appears to be uncommon in CAFOs. The most notable infectious bioaerosol in agricultural occupational environments is *Mycobacterium tuberculosis* (Schenker et al. 1998). However, this arises from transmission from person-to-person. Tuberculosis occurs with high prevalence among immigrant farm laborers. More germane to CAFOs in Iowa is concern over the emergence of antibiotic resistant pathogenic organisms that may arise under the influence of antibiotics added to feed.

Non-infectious microorganisms are a more significant problem in CAFOs by virtue of the enormously high concentrations at which they occur. There has been limited study of the effects of inhaled bacteria and fungi in laboratory animal models of human disease. Most of the studies in the literature have used a lung infection model to study host defense against lung pathogens or to assess the efficacy of antimicrobial therapies. However, a few studies are informative. McCray et al (1999) demonstrated severe inflammation with neutrophilic infiltration to the lungs of mice following 4 hr inhalation exposure to *Pseudomonas aeruginosa* at a concentration of 3.3×10^8 CFU/m^3. This study used bacterial lung burdens that resemble those attainable in CAFOs. The bacteria were cleared from the lungs within 24 hours and the inflammation resolved by 72 hours after exposure. Thorne and Gassman studied the relative potency of inhaled Gram-negative organisms and Gram-positive organisms for lung inflammation in mice (Gassman et al 2000). This study demonstrated that the Gram-negative bacteria: *Enterobacter agglomerans* and *Pseudomonas aeruginosa* were orders of magnitude more potent that the Gram-positive organisms: *Bacillus magaterium* and *Micrococcus luteus* at initiating inflammation. In this study, markers of inflammation included influx of neutrophils to the lung and increased concentration of interleukin-6 (IL-6) and tumor necrosis factor (TNFa). It was concluded that the endotoxin derived from the Gram-negative organisms was the cell component primarily responsible for the inflammation.

Fungi and fungal conidia are also found airborne in CAFOs. Fungi have been studied primarily as allergens and as sources of mycotoxins. There is no reported evidence of animal or human health problems due to mycotoxin delivery arising from inhalation of fungal spores for the common fungi found in CAFOs. Studies of allergen potency for fungi found in CAFOs have focused on human studies rather than on animal models.

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Chapter 6.2 Animal Health Effects

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Chapter 6.2. Animal Health Effects

The preponderance of scientific studies on the effects of air contaminants and emissions on animal health has been conducted in and around swine facilities. Air contaminants can be divided into gases, particulates, bioaerosols, and toxic microbial by-products. Excess ammonia has been associated with lowered average number of pigs weaned, arthritis, porcine stress syndrome, muscle lesions, abscesses, and liver ascarid scars. Particulates (dust) have been related to reduced growth in growing pigs and turbinate pathology. Bioaerosols have been associated with lowered feed efficiency, decreased growth, and increased morbidity and mortality due to respiratory disease and abscesses. There are few scientific studies regarding the health effects and productivity problems of air contaminants on cattle and other livestock. Ammonia and hydrogen sulfide are the two most important inorganic gases affecting the respiratory system of cattle raised in confinement facilities. These gases affect the mucociliary transport and alveolar macrophage functions of the respiratory system lessening its protective responses.

6.2.1 Ammonia - Livestock Health Effects

At concentrations usually found in livestock facilities (<100 ppm), the primary impact of aerial ammonia is as an irritant of the eye and respiratory membranes; and as a chronic stressor that can affect the course of infectious disease as well as directly influence the growth of healthy young animals (Lillie, 1972; Curtis, 1983).

A series of experiments at the University of Illinois measured the effects of various levels of aerial ammonia on young pigs. The rate of gain of young pigs was reduced by 12% during exposure to aerial ammonia at 50 ppm, but no lesions were observed in the respiratory system. At both 100 and 150 ppm aerial ammonia, rate of gain was reduced by 30% and tracheal epithelium and nasal turbinates showed lesions consitent with a tissue irritant (Drummond et al., 1980). Aerial ammonia at 50 and 75 ppm reduced the ability of healthy young pigs to clear bacteria from their lungs (Drummond et al., 1978). At 50 and 100 ppm, aerial ammonia exacerbated nasal turbinate lesions in young pigs infected with *Bordetella bronchiseptica,* but did not add to the infection-induced reduction in the pig's growth rate (Drummond et al., 1981a). In another study, 100 ppm aerial ammonia reduced the rate of gain by 32%; while effects of 100 ppm ammonia and concurrent ascarid infection were additive to where the rate of gain was reduced by 61% (Drummond, et al., 1981b). In a study of 28 swine farms in Sweden, a higher incidence of arthritis, porcine stress syndrome lesions, and abscesses had a positive correlation with levels of aerial ammonia in the facilities (Donham, 1991)

It has recently been recommended that the maximum long-term ammonia exposure limit for swine should be less than 20 ppm as both pathological data (Hamilton, 1996) and immunological data (Urbain, 1994) suggest that exposure to ammonia concentrations of 10 to 15 ppm reduce resistance to infection (Jones, 1997). British workers utilized operant conditioning techniques giving pigs the choice between ambient ammonia levels of 0, 10, 20, and 40 ppm to demonstrate that pigs have an aversion to atmospheres containing even relatively low levels of ammonia (Jones, 1997).

Ammonia has been considered as the most significant air pollutant in cattle barns as its irritating effect on the respiratory epithelium appears to directly reduce the number of ciliated cells and thus decrease the efficiency of mucociliary transport (Marschang, 1973). Ammonia concentrations within cattle facilities varied greatly from 80 to 2001 mg/h per animal depending on the type of housing (concrete floors vs slatted flooring, ventilated vs closed), bedding, age of animals, environmental conditions, waste storage system employed, frequency of cleaning, and ration (Koerkamp et al, 1998; Wathes et al, 1998; Pitcairn et al, 1998; Gurk et al, 1997). At concentrations less than 100 ppm and in a poorly ventilated facility, ammonia appears to affect pulmonary function in cattle. Five mechanisms protect the lungs from invasion of foreign materials: cellular and humoral immunity, mucociliary transport, macrophage function, cough reflex, and nasopharyngeal filtration. Of these defensive mechanisms, mucociliary transport and alveolar macrophage functions are most severely affected by ammonia and possibly hydrogen sulfide (Lillie and Thompson, 1972).

In poultry, ammonia is considered the most harmful gas in broiler chicken housing (Carlile, 1984). Ambient ammonia levels of 50 ppm for prolonged periods irritate respiratory airways and predispose chickens to respiratory infections with the added risk of secondary infections; and development of lesions of keratoconjunctivitis of the eye is associated with ambient ammonia levels of 60 ppm (Hauser, 1988). A reduced rate of bacterial clearance from the lungs was measured in turkeys exposed to 40 ppm aerial ammonia (Nagaraja, 1984). Excessive mucous production, matted cilia, and deterioration of normal mucociliary apparatus was found in turkeys exposed to ammonia concentrations as low as 10 ppm for 7 weeks (Nagaraja, 1983).

6.2.2 Hydrogen Sulfide - Livestock Health Effects

Hydrogen sulfide is a potentially lethal gas produced by anaerobic bacterial decomposition of protein and other sulfur containing organic matter. This colorless gas with the distinctive odor of rotten eggs is heavier than air and may accumulate in manure pits, holding tanks, and other low areas in a facility. The sources of hydrogen sulfide presenting the greatest hazard in an agricultural setting are liquid manure holding pits which are commonly under slatted floors of livestock facilities. Although most of the continuously produced hydrogen sulfide is retained within the liquid of the pit, the gas is rapidly released into the ambient air when the waste slurry is agitated to suspend solids prior to being pumped out. While the concentration of hydrogen sulfide usually found in closed animal facilities (<10 ppm) is not harmful, the release of this gas from manure slurry agitation may produce concentrations up to 1000 ppm or higher (Lillie, 1972; Carson, 1998; Donham, 2000).

Hydrogen sulfide is an irritant gas producing local inflammation of the moist membranes of the eye and respiratory tract. The irritant action of hydrogen sulfide is fairly uniform throughout the respiratory tract, although the deeper pulmonary structures suffer the greatest damage often producing pulmonary edema (Curtis, 1983).

Differences between mammalian species susceptibility to toxic concentrations of hydrogen sulfide are small, as demonstrated by the following reported acutely toxic levels of hydrogen sulfide: goat – 900 ppm; guinea pig – 750 ppm; dog – 600 ppm; rat – 500 ppm (Sayer, 1923). However, chickens were found to be less sensitive to hydrogen sulfide than mammals, with exposures of 4,000 ppm not resulting in immediate death (Klentz, 1978).

Early experiments examining various levels of acute hydrogen sulfide gas exposure in pigs reported the following associated clinical effects; 50 to 100 ppm - nothing significant; 250 ppm – distress; 500 to 700 ppm – semicomatose; 1000 ppm – intermittent spasms, cyanosis, unconsciousness, convulsions, death (O'Donoghue, 1961). At low levels of hydrogen sulfide exposure, no effect was measured on rate of body weight gain or respiratory tract structure in young pigs breathing air containing 8.5 ppm hydrogen sulfide for 17 days (Curtis, 1975)

6.3.3 Particulates

Particulates are derived from two primary sources: pigs and feed. The primary particulate component from the pigs is dried fecal material. After drying fecal material becomes aerosolized by movement of the pigs and air currents. This dust is very fine, and up to 40% is inhalable (Donham, 2000). Dried fecal material is heavily contaminated with microbes and microbial by-products. Animals and workers in nursery and farrowing facilities would be exposed to greater concentrations of fecal dust than would those in finishing facilities where feed dust would predominate (Donham, 2000).

6.3.4 Bioaerosols and Endotoxins

Air quality, as defined in ventilation parameters, influences the aerosol spread of potential viral and bacterial pathogens that colonize the respiratory epithelium. However, rarely does one find pathogens in the air. They generally are less viable and found in fewer numbers relative to the nonpathogens and saprophytes (Donham, 2000). Bacteria, fungi, and yeast heavily contaminated the atmosphere of swine confinement facilities. Total microbial concentration (cfu per cubic meter) range from 100,000 – 10,000,000 (Donham, 2000). Maximum concentration for swine health is approximately 430,000 (Donham, 2000).

Of recent importance is the concentration of endotoxin detected in the atmosphere of confinement facilities. Endotoxin is a phospholipid-polysaccharide macromolecule that comprises the cell wall of Gram-negative bacteria. It is released when the integrity of the cell wall is disturbed. A typical range for endotoxin in the atmosphere of a confined building is 150 -1000 units (Donham, 2000). Maximum concentration for swine health has been approximated at 150 units. Endotoxin is a highly inflammatory substance and is believed to play a major role in respiratory disease of workers (Donham, 2000).

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6.3 Human Health Effects

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6.3 Human Health Effects

While other public health impacts, which include human exposures to polluted water or antibiotic resistant microorganisms that may arise from CAFOs, are not being addressed in this chapter, occupational exposures to CAFO environments will be reviewed and discussed because of their relevance to human response to CAFO air emissions.

The lung contains the largest epithelial surface in the body, consisting of more than 100 square meters of surface area in the average adult male (compared with approximately 2 square meters of skin). The average adult male inhales up to 15 kg of air daily, and children inhale proportionately more for their size. Because of this high surface area and high volume of air exchanged, the lung is capable of absorbing vast quantities of inhaled substances. Defense mechanisms of the lung, including the cough reflex, mucociliary transport, and the innate immune system are efficient at combating inhaled particulate matter and microorganisms. Gases, vapors, and aerosols (of "respirable" size, approximately 1-10 microns in diameter) are readily inhaled and absorbed.

Health effects associated with inhalation of toxins and bioaerosols are manifold. Medical problems commonly associated with inhaled agents include respiratory diseases (asthma, hypersensitivity pneumonitis, industrial bronchitis), cardiovascular events (sudden death associated with particulate air pollution), and neuropsychiatric conditions (due to odor as well as delayed effects of toxic inhalations).

Most studies of human exposures to airborne agricultural hazards have focused on occupational exposures in agricultural settings. With the rise of large, industrial CAFOs as the preeminent form of livestock production and their associated higher production of gases, vapors, and fumes, these exposures now have the potential to affect larger numbers of individuals, including members of the neighboring community not involved in agriculture or related industrial livestock production. Few studies have directly examined the health effects of proximity or exposure to CAFOs in the community, thus extrapolations must be considered from well-documented effects of these toxins in laboratory settings and occupational exposure studies. Donham and colleagues (1977) first reported that workers in swine confinement facilities described significantly more respiratory symptoms than non-exposed workers; subsequent studies have confirmed this symptomatology and have also documented increased risks of respiratory infections, progressive declines in pulmonary function, and poisoning from hydrogen sulfide in this occupational group.

For many reasons, standards for community exposures to the toxic agents released from CAFOs must be stricter than that for occupational exposures. First of all, community members may include subgroups of especially susceptible individuals, for example the elderly, children, and those with pre-existing impairments. Secondly, community members may be exposed continuously to released substances rather than for a workshift or less; this is especially true for those who do not work outside the home, and for pre-school children. Moreover, exposed community members may not have chosen to live in proximity to a CAFO, whereas occupationally exposed individuals have some choice in their employment. Thus, ambient exposure levels arising from CAFOs, including ammonia and hydrogen sulfide, must be significantly lower than occupational levels; notwithstanding, many components of the CAFO environment, e.g. bioaerosols including endotoxins and glucans, have no current recommended or mandated occupational exposure limits.

6.3.1 Studies of Adverse Health Effects from Specific Exposures found in CAFO Emissions

It is important to examine the literature regarding adverse health effects arising from individual chemicals and mixtures of chemical compounds, often referred to as odorants, exposures known to be components of emissions from CAFOs. The following is a summary of available, published findings from clinical, experimental and epidemiological observations for several categories of these exposures. The concentration of exposure is not always known or measured, but there have been several studies of individual chemical exposures that have documented both concentrations and durations of exposure, some at very low levels. The vast majority of these observations come from occupational, experimental, and non-CAFO community exposures, many of which were made among selected populations of workers or healthy volunteer subjects. Regulatory agencies have used many of these findings, taking into account uncertainty and susceptibility factors, in making their recommendations regarding exposure limits for exposed communities (See Chapter 8.0).

6.3.1.1. Ammonia

Ammonia is both a component of animal waste and released in waste treatment processes. Well recognized as a human toxin, the current OSHA PEL for ammonia is a TWA of 50 ppm (also its odor threshold), although ACGIH and NIOSH recommend a lower TWA of 25 ppm. Concentrations of greater than 100 ppm have been regularly reported in poultry confinement operations (Mulhausen et al, 1987). The EPA has found that animal agricultural operations are responsible for almost three fourths of ammonia air pollution in the United States (Harris et al, 2001), although numerous other industries are associated with inhalation exposure to ammonia. EPA has recommended as reference concentration for chronic inhalation of ammonia of 1.4 ppm. ATSDR has recommended a long-term MRL of 300 ppb for community exposures (See Chapter 8.0 for detailed discussion).

Water-soluble, ammonia is rapidly absorbed in the upper airways, with the result of damaging upper airway epithelia. Moderate concentrations (50-150 ppm) can lead to severe cough and mucous production; higher concentrations (>150 ppm) may cause scarring of the upper and lower airways (Close et al, 1980; Leduc et al, 1992). A consequence of these inflammatory responses, in some cases, is reactive airways dysfunction syndrome (RADS) and associated persistent airway hyperresponsiveness (Bernstein and Bernstein, 1989; Flury et al, 1983). At higher concentrations, sufficient ammonia may bypass the upper airways to cause lower lung inflammation and pulmonary edema (Close et al, 1980; Sobonya, 1977). Massive exposure to ammonia can be fatal, including in the agricultural sector, a consequence to disruption of tanks of anhydrous ammonia (Sobonya, 1977). These fatalities, as well as the chronic lung disease seen following as little as two minutes of exposure to high concentrations of ammonia gas may result in the development of bronchiolitis obliterans (de la Hoz et al, 1996; Kass et al, 1972; Sobonya, 1977; Walton, 1973), restrictive lung disease (de la Hoz et al, 1996), and bronchiectasis (Leduc et al, 1992).

In addition to pulmonary disease, exposure to ammonia also leads to irritation of the eyes, sinuses, and skin. Exposure to 100 ppm ammonia for short (30 second) duration leads to nasal irritation and increases in nasal airway resistance (McLean et al, 1979). When increasing concentrations of ammonia are delivered by spontaneous respiration, severe nasal irritation develops at 134 ppm after 5 minutes; some individuals report symptoms as low as 32 ppm (Keplinger et al, 1973). Clinical sinusitis has been reported following accidental exposure to ammonia as well (Brautbar, 1998). Chemical burns to the skin and eyes are also commonly seen following high-concentration ammonia exposures (Latenser and Loucktong, 2000).

Although the most serious adverse effects of ammonia inhalation are usually seen with concentrations of ammonia that have been associated with fatal exposures (in the range of 500 ppm), evidence exists that lower concentrations of ammonia can reach the alveoli and may be adsorbed to respirable particulates, as may be seen in complex bioaerosols such as those found in the agricultural setting resulting in a research-recommended occupational exposure limit of 7 ppm (See Section 6.3.2.2). Similar occupational exposures to ammonia (9 ppm) have been studied among soda ash workers (Holness et al, 1989) who reported increased symptoms of coughing, wheezing, nasal complaints, eye irritation, throat irritation, and skin complaints; however, no changes in lung function were observed when measured over a working shift. It was noted that this was a cross-sectional study of a small population and that selection bias may have therefore occurred.

6.3.1.2. Hydrogen Sulfide

Hydrogen sulfide is one of the most important of the gases arising from the storage, handling, and decomposition of animal wastes. Smelling like rotten eggs, this gas that is recognized as both an irritant and an asphyxiant, is a prominent component of odorants released from CAFOs. Current OSHA PEL for H_2S are 10 ppm (with STEL of 15 ppm), while NIOSH has recommended a time weighted average occupational exposure limit of 10 ppm. For community exposures, EPA has recommended a reference concentration for long-term exposure of 7 ppb (See Chapter 8.0 for full discussion).

Levels as high as 1,000 ppm have been reported (Donham and Gustafson, 1982) following the perturbation of manure lagoons, and levels greater than 100 ppm are considered immediately hazardous to life and health. Exposure to these elevated levels of H_2S can cause rapid loss of consciousness, and H_2S has been implicated in a number of deaths when encountered in confined environments in agricultural settings. The primary mode of absorption of H_2S is through inhalation (Bhambhani et al, 1996a).

One particular hazard is that, although the odor threshold is quite low (less than 1 ppm), at levels over 6 ppm the intensity of the smell only modestly increases; above 150 ppm, exposure to hydrogen sulfide may actually reduce the sense of smell, hindering the olfactory detection of high concentrations of the gas and making H_2S monitoring equipment mandatory in occupational settings (van Aalst et al, 2000). The toxic effects of hydrogen sulfide are based on its property as a chemical asphyxiant; it binds to the mitochondrial enzyme cytochrome oxidase, blocking oxidative phosphorylation and ATP production. This leads to anerobic metabolism and the development of lactic acidosis (Nichols and Kim, 1982).

Experimental exposure studies have been carried out examining the effects of inhalation of low levels of H2S on healthy volunteers (Bhambhani et al 1996a, 1996b, 1997). Inhalation of 5 ppm of H_2S by exercising men leads to a significant decrease in the concentration of citrate synthase, a marker of aerobic metabolism, in muscle biopsy tissue, although no increases in lactic acidosis were noted (Bhambani et al, 1996b). Levels of 10 ppm cause no change in physiologic measures of pulmonary function (Bhambani et al, 1996a), but do cause a significant decline in maximal oxygen update (VO2max) and an associated increase in blood lactate in exercising men and women (Bhambani et al, 1997). Jappinen and colleagues (1990) exposed a group of asthmatics (severe asthmatics were eliminated from the study) to 2 ppm of hydrogen sulfide for 30 minutes. Three complained of headache and two were found to have increased airway resistance, but there was no change in other lung function values or associated symptoms. Members of a Mobile Monoriting Team of the Texas Natural Resource Conservation Commission (TNRCC) evaluated hydrogen sulfide concentrations downwind from an oil refinery and reported 0.09 ppm 30-minute averages over a period of five hours (Texas Natural Resources Conservation Commission, 1998). Six staff members reported eye and throat irritation, headache, and nausea. These experimental studies indicate consistent patterns of adverse health effects after short, low concentrations of exposure to hydrogen sulfide.

Epidemiological studies of workers exposed to hydrogen sulfide exposure include pulp mill workers who reported increased respiratory symptoms (irritation and cough), as well as increased headache and migraine; it was noted that these workers were also exposed to other sulfur compounds including sulfur dioxide and mercaptans (Partti-Pellinen et al, 1996). Jappinen and colleagues (1990) studied pulp mill workers thought to be exposed to hydrogen sulfide levels usually below a maximum permitted concentration of 10 ppm and reported no significant changes in lung function and airway hyperresponsiveness at the end of the workday, compared with control values. Hessel and colleagues (1997) studied oil and gas workers at undefined, but probably moderately high, exposures to hydrogen sulfide (as some of the workers lost consciousness); nearly third of the workers reported symptoms.

Several additional epidemiological studies of community residents exposed to low levels of hydrogen sulfide have been reported. A U.S. Public Health Service study of a general population exposed to levels in excess of 0.3 ppm reported adverse health effects including shortness of breath, eye irritation, nausea, and loss of sleep (United States Public Health Service, 1964). Jaakkola and colleagues (1991) studied chronic community exposure to hydrogen sulfide and TRS (total reduced sulfur) compounds (hydrogen sulfide annual means of 0.006 ppm and daily means of 0.07 ppm) and found that both asthma and chronic bronchitis were slightly more prevalent, that eye and nasal symptoms were found significantly more often, and that these symptoms were dose-related. They concluded that the WHO standard of 0.1 ppm (24 hour average) did not protect against these adverse health effects. Jaakkola and colleagues (1991) also studied the respiratory infection rate among infants exposed to ambient hydrogen sulfide levels of 0.001 ppm, and at half-hour maximal exposures of 0.125 ppm, and reported that exposed infants had higher rates of respiratory infection, but that combined effects of other air pollutants may have been contributing factors. Haahtela and colleagues (1992) studied community residents exposed to peak exposures of hydrogen sulfide of 0.095 ppm (four hour average) and 0.025 and 0.030 ppm over two days of exposure, compared to control days, with four hour exposures ranging between 0.00007 and 0.002 ppm. Cough, throat irritation, and eye symptoms were observed significantly more often during the peak exposure period. The author concluded that the WHO guideline of 0.10 ppm for a 24 hour average did not provide adequate protection from adverse health effects. Rossi and colleagues (1993) studied the occurrence of asthma attacks in relation to air pollution events (hydrogen sulfide levels ranged from the highest 1 hour mean of 0.011 ppm and daily 24 hour means of 0.002 ppm), and reported significant associations between the frequency of asthma attacks at an emergency room and nitrogen sulfides, sulfur dixoxide, total suspended particulates, and hydrogen sulfide. Partti-Pellinin and colleagues (1996) studied a general population exposed to TRS levels of up to 0.1 ppm over a 24-hour period. Based on a self-administered questionnaire, the authors concluded that the exposed community reported more cough, respiratory infections, and headaches than the reference community, and also that headaches, depression, tiredness, and nausea were more often reported on days when the 1 hour or daily mean TRS levels exceeded 0.028 ppm (both communities were exposed to similar levels of sulfur dioxide). These community studies of hydrogen sulfide and TRS exposures are especially useful because they report measured low levels of exposure and associated adverse health effects. However, as is the case with community exposures to CAFOs, these are invariably mixed exposures to hydrogen sulfide and other chemicals, some of which may contribute to the adverse health effects described in these studies. Campagna and colleagues (2001) studied the effects of ambient hydrogen sulfide and TRS levels on hospital visits for respiratory diseases among children and adults in Dakota City and South Sioux City, Nebraska. While peak levels of hydrogen sulfide were as high as 1,375 mean levels over an entire day were much lower. An increase in asthma hospital visits was seen a day following peak TRS exposures among children and an increase in hospital visits for all respiratory disease was seen following peak exposures for both TRS and hydrogen sulfide.

Finally, Xu and colleagues (1998) has reported a retrospective epidemiological study of spontaneous abortion among a large cohort of female workers in a petrochemical plant in Beijing, China. Among women exposed only to hydrogen sulfide (concentrations were not reported because of the retrospective nature of the study), a rate of spontaneous abortion of 12.3% was observed and a significant association with hydrogen sulfide exposure was reported (OR, 2.3, CI 1.2-4.4).

Chronic low-level exposure is associated with anosmia, the loss of ability to detect odors. At higher levels, hydrogen sulfide exposure causes loss of consciousness, shock, pulmonary edema, coma and death. Survivors of hydrogen sulfide poisoning are reported to commonly have neuropsychiatric defects which may be permanent; a recent study by Kilburn of University of Southern California has demonstrated that even exposure to low concentrations of hydrogen sulfide leads to significant neuropsychologic abnormalities, including impaired balance, visual field performance, color discrimination, hearing, memory, mood, and intellectual function (Kilburn, 1997). These effects may be due to anoxic encephalopathy.

6.3.1.3. Particulates

The air in CAFOs is contaminated with high concentrations of particulates, approximately one quarter of which is protein; about one third of suspended dust is considered respirable (< 10 microns in diameter, PM10). Occupational and environmental studies have demonstrated an average of 2-6 mg/m^3 dust concentrations, and levels up to 20 mg/m^3 may be encountered. National ambient air standards for PM10 are an annual average of 50 mcg/m^3 with a 24-hour average of 150 mcg/m^3. Of these, particles between 4 and 10 microns are deposited in the airways and smaller particles (< 2.5 microns) progress into and may be absorbed by the terminal bronchiols and alveoli. Particles which settle in the upper airways are associated with asthma and bronchitis; smaller particles may be absorbed and have systemic effects including, in studies of urban air pollution, increased rates of cardiac death. In addition to direct inflammatory response to inhaled allergens, dust can also convey inflammatory and/or irritating gases or chemicals (such as ammonia, hydrogen sulfide, or endotoxin) deeper into the lung, thereby enhancing their toxic effects.

Although certain mineral particulates, such as silica dioxide, lead to characteristic pulmonary inflammatory and scarring conditions known as pneumoconioses, even inhalation of seemingly inert dust particles appear to have adverse long-term consequences. In a number of occupational settings, cumulative exposure to dust particles in the respiratory range is one of the most important causes of persistent respiratory symptoms and progressive declines in lung function (Healy et al, 2001; Ulvestad et al, 2001); and this has also been reported in non-occupational settings (Dockery and Pope, 1994; Dockery et al, 1993; Pope et al, 1995, Lippmann et al, 2000).

6.3.1.4. Bioaerosols

An important component of the environment released from CAFOs is microbiologic in origin. Swine manure contains up to 10^8 coliform bacteria/gram, and CAFOs contain these organisms in airborne and respirable particles; total organism load may exceed 10^{10} cfu/m^3 at times. Some of the microorganisms that are present in the CAFO environment are human pathogens, creating a potential risk of infection for those exposed to these agents. Dust in CAFOs and other agricultural settings, contains far more than merely viable organisms. Microbial products of medical importance include antigens, glucans, and endotoxins.

Exposure to protein antigens derived from plants, animals, and microbes are known to cause a variety of medical problems. Inhalation of thermophilic bacteria, commonly found in moldy hay and other damp locations, leads to a condition known as hypersensitivity pneumonitis, a respiratory condition characterized by granulomatous inflammation of the lung, restrictive physiology, and progressive dyspnea. Associated with detection of antibodies to these organisms in the blood, hypersensitivity pneumonitis (also known as "farmer's lung" in agricultural settings), is found among agricultural workers and others occupationally exposed to these agents (Skorska et al, 2000).

Asthma may also be caused or exacerbated by exposure to conditions common in CAFOs. Atopic asthma is caused, in susceptible individuals, by sensitization to and subsequent inhalation of allergens, agents that can lead to asthma in previously non-sensitized individuals. Those with a previous diagnosis of asthma may have their asthma triggered in a non-specific way by exposure to the dust and irritant-inducing agents arising from the CAFO environment. CAFOs contain, among other compounds, high concentrations of grain dust, dust mites, animal dander, pollen grains, molds and fungal spores, and dried fecal particles, each of which may induce or exacerbate asthma. Proximity to CAFOs, and periodic/seasonal agricultural activities (e.g., agriculture chemical and manure applications), are frequently cited by rural asthma patients as exposures resulting in asthma exacerbation making asthma control more difficult.

Endotoxins are lipopolysaccharide complexes that are products of gram-negative bacterial cell walls. Ubiquitous in the environment, they are present in high concentrations in agricultural settings such as grain elevators, feed barns, and CAFOs. Endotoxins are important components of exposures responsible for the adverse health effects following inhalation of organic agricultural dust. Acute effects of endotoxin inhalation include symptoms of cough, chest tightness, and dyspnea and alterations in pulmonary function characterized most typically by a decline in FEV1; over a working shift and overtime; systemic effects include fever, rigors, myalgia, arthralgia, and other "flu-like" symptoms. Although no occupational standards currently exist for endotoxin in the United States, Dutch Expert Committee on Occupational Standards of the National Health Council has proposed a limit of 50 EU/m^3 (4.5 ng/m^3) over an 8-hour exposure period (Heederik and Douwes, 1997).

Kline and colleagues (1999) evaluated the responses of 72 normal, non-smoking, non-atopic, non-asthmatic volunteers who were exposed to graded doses of endotoxin by inhalation in a clinical exposure facility. Each subject first inhaled 0.5 mcg of endotoxin then underwent spirometry prior to inhaling a greater concentration of endotoxin. Cumulative levels of endotoxin inhalation consisted of 0.5, 1.5, 3.5, 6.5, 11.5, 21.5, 41.5 mcg. The protocol was terminated for decline in FEV1 to < 90% of baseline or a total of 41.5 mcg. Among study participants, a wide range of sensitivity to the bronchospastic effects of inhaled endotoxin was found; some individuals demonstrated a 20%

decline in FEV1 following inhalation of as little as 1.5 mcg whereas others were resistant to these effects and did not even decline by 10% following inhalation of over 41.5 mcg. In a separate study, asthmatic individuals were found to have an enhanced degree of symptoms and bronchospasm following inhalation challenges compared with normal control subjects (Kline et al, 2000). Other studies have also found that inhalation exposure to endotoxin and endotoxin-containing grain dust leads to the development of bronchospasm and airway inflammatory responses (Blaski et al, 1996; Jagielo et al, 1996; Michel et al, 1989; Michel et al, 1996; Michel et al, 1997; Schwartz et al, 1995a).

Most of the reports of community, occupational, and ambient effects due to endotoxin exposure are related to inhaled endotoxin; this is clearly different than the case of patients suffering from gram-negative infections, who are typically exposed to endotoxin via the blood stream. The greatest effect of inhaled endotoxin is on airway inflammation and the induction of bronchial hyperresponsiveness, both characteristic of asthma. Interestingly, some recent studies have demonstrated a protective effect of endotoxin exposure relative to the development of allergic disease. Von Mutius and colleagues (2000) recently reported that environmental endotoxin exposure of farmers' children protects them from the development of atopy; Gereda and colleagues (2000), in a study of urban homes, found that home levels of endotoxin inversely correlated with likelihood of allergen sensitization in infants. In a similar vein, Gehring and colleagues (2001) found that environmental exposure to endotoxin protected infants from the development of atopic eczema. These effects of endotoxin on early-life development of allergic responsiveness may be due to the deviation away from a Th2-type response to allergens and towards a Th1-type response, however alternate explanations are possible.

Exposure of adults, however, (and infants and children in some studies) appears to be clearly detrimental with regards to airway function and asthma. In contrast to the studies showing protective effects of endotoxin on the development of disease among infants, Park and colleagues (2001) reported that infants with at least one asthmatic/allergic parent were placed at increased risk of developing wheezing when their home environment contained higher levels of ambient endotoxin. Douwes and colleagues (2000), in a community study of household dust, found that endotoxin content of dust was associated with increased peak flow variability among asthmatic children. Michel and colleagues (1991) reported that asthmatic patients with higher levels of home endotoxin exposure develop more symptoms and require more intensive treatment than those from homes with lower levels of endotoxin. In a separate study, the same group confirmed that asthma severity correlates with endotoxin exposure (Michel et al, 1992). In a study conducted in Brazil, Rizzo and colleagues (Rizzo et al, 1997) found that endotoxin (but not dust mite) content of dust significantly correlated with symptom scores in asthmatic children.

Controlled laboratory studies of endotoxin exposure confirm that inhalation induces airway inflammation and bronchial hyperreactivity. Blaski and colleagues (1996) reported that both normal control subjects and atopic individuals developed airway neutrophilia and reduced airflow following inhalation of 0.4 mcg/kg of endotoxin. Jagielo and colleagues (1996) found that the endotoxin content of grain dust was responsible for its ability to induce inflammation and obstructive airway physiology in normal volunteers. Michel and colleagues (1989) found that endotoxin inhalation by asthmatics resulted in significantly more airflow reduction than in normals. Among asthmatics, the reduction in airflow (Michel et al, 1992) and development of symptoms of chest tightness and dyspnea (Kline et al, 2000) are greater than the difference in development of airway inflammation. Even among non-asthmatics, a significant variability in responsiveness to the effects of inhaled endotoxin can be seen (Kline et al, 1999); this appears to be explained, at least in part, by genetic factors (Arbour et al, 2000).

Mycotoxins, beta-glucans, and other components of fungal pathogens appear to have a similar range of toxicity to endotoxins, including both inflammatory and immunostimulatory effects. These compounds, however, have been less well studied in human exposures, and their concentration in CAFOS is unknown (American Thoracic Society, 1998).

6.3.1.5. Volatile Organic Compounds

Of the thousands of gases, vapors, particles, and aerosols present in CAFOs, over 24 odorous chemicals, often referred to as odorants, have been identified (Cole et al, 2000). Volatile acids, mercaptans, and amines are particularly odorous even in miniscule concentrations. Ammonia and hydrogen sulfide, as noted above, are also pungently aromatic.

Although long recognized as a neighborhood nuisance, recent studies have suggested that odiferous exposures emitted from CAFOs may well have adverse health effects (Schiffman et al, 2000). Odor appears to play a significant role in the recognition of and concern over symptoms in neighbors of hazardous waste sites (Shusterman, 1992; Shusterman et al, 1999). Schiffman and colleagues (1995) from Duke University have reported that indicators of altered mood, assessed using validated scales, are significantly worse in subjects who live in the vicinity of intensive swine operations compared with control subjects.

Chen and colleagues (1999) have demonstrated, using odor threshold dilution analysis, that odor intensity in swine buildings is reproducible and measurable. Zahn and colleagues (2001) have analyzed malodorous volatile organic compound components of swine production facility air samples, and have demonstrated, using an artificial swine odor solution, that alterations in the concentrations of these components can be detected by study subjects. No odor studies were found that related the quantitative measurement of odor intensity in the downwind air stream from livestock facilities with adverse health effects among community residents. However, there is an extensive literature relating non-CAFO odors and adverse health effects that are relevant to community exposures to CAFO exposures.

Of the hundreds of gases, vapors, particles, and aerosols present in CAFOs, 331 volatile organic compounds (VOCs) and fixed gases were recently characterized by Schiffman and colleagues (2001). These compounds, assessed at the point of emission, included many acids, alcohols, aldehydes, amides, amines, aromatics, esters, ethers, fixed gases, halogenated hydrocarbons, hydrocarbons, ketones, nitriles, other nitrogen-containing compounds, phenols, sulfur-containing compounds, steroids, and other compounds. The authors (Schiffman et al, 2001) further observed that the vast majority of these compounds were found at concentrations below their published irritant or odor thresholds, yet human assessments of the combined odors and their irritant effects were described as "strong" at a distance of 1000 feet.

While CAFO odors have long been recognized as a neighborhood nuisance, recent studies have suggested that odiferous exposures emitted from CAFOs may well have adverse health effects (Schiffman, 1997; Schiffman et al, 1995; Thu et al, 1997; Wing and Wolf, 2000). Direct measurement of odorous or other noxious substances were not made in these studies, therefore, a direct linkage to level of exposure could not be reported. A Duke University workshop summarized by experts in assessing the potential health effects of odor from animal operations (Schiffman, Walker, Dalton, Lorig, Raymer, Shusterman and Williams) addressed this issue (Schiffman et al, 2001). They observed that health symptoms have been reported with increasing frequency from low level exposures from manures and biosolids; "the most frequently reported health complaints include eye, nose, and throat irritation, headache, nausea, diarrhea, hoarseness, sore throat, cough, chest tightness, nasal congestion, palpitations, shortness of breath, stress, drowsiness, and alteration in mood". They further observed that these symptoms usually occurred briefly at the time of exposure, but that hypersensitive individuals, such as asthmatics, could have their condition exacerbated with persisting symptoms.

Exactly how odors from CAFOs may result in these symptoms is not well understood. The Duke workshop discussed freeways, or paradigms, by which ambient odors may produce health symptoms (Schiffman et al, 2000). In the first paradigm, the symptoms may occur at levels of exposure that would also be expected to cause irritant effects from combinations of irritants that may be additive or synergistic in their effect. In this paradigm, the adverse health effect typically occurs at a higher level than the concentration at which the odor would first be detected.

In the second paradigm, symptoms may occur at odor concentrations below that expected from irritants. The mechanism by which these odorants may cause their adverse effects is not known (Schiffman et al, 2000). Schiffman and colleagues (1995) reported that CAFO odors perceived as unpleasant can impair mood. Shusterman and colleagues (1991) observed increased symptom prevalence and an "odor worry" interaction associated with odor from hazardous waste sites. Schiffman and colleagues (2000) summarized evidence that negative mood, stress, and environmental worry may lead to biochemical and physiological effects with subsequent health outcomes. Other studies suggest that bias concerning odors can alter the response relating to health effects (Dalton et al, 1997). These results provide evidence that both the perceived odor and cognitive expectations about a chemical can significantly affect individual response. Other studies have also demonstrated that ones current cognitive state can bias ancillary characteristics of an odor such as preference or acceptability (Knasko, 1993). Some studies have shown that persons can report experiencing strong odors as an outcome by showing that cognitive factors can lead to reports of odors when none are present (Knasko, 1992; O'Mahoney, 1978). Knasko and colleagues (1990) have also observed that an odorant stimulus is greatly influenced by the environment surrounding the exposure, which can include the social context or the perceiver's mental state. It is also recognized that those working in an odorous environment may adapt to the odor following long term exposure. Dalton and Wysocki (1996) have advocated for the development of laboratory procedures that combine long-term odor exposure in a naturalistic setting with psychological tests.

A third way for paradigm, is when the odorant is a part of a mixture that contains bioactive pollutants such as bioaerosols containing organic dust, endotoxin, glucans, allergens, microorganisms, or other toxins (Schiffman et al, 2000). In this paradigm, the individual is exposed to odors, but the adverse health effect is likely to arise from a non-odorant toxin. Relevant to this paradigm is the study of Reynolds and colleagues (1997) who sampled at 60 meters for hydrogen sulfide, ammonia, endotoxin, and total dust. A reason to sample for dust and ammonia together is that it is now recognized that some ammonia absorbs to respirable dust particles thereby providing a vehicle to transport ammonia and dust-latent toxins, like endotoxin, deep into the lung.

To date there has been relatively little research quantifying odorants. Zahn and colleagues (2001) completed a multi-component analysis of malodorous DOCs found in air samples from 29 swine production facilities using a 19-component artificial swine odor solution. The results of this study concluded that this approach can be applied toward estimating perceived odor intensity. Schiffman and colleagues (2001) studied six swine operations in North Carolina. In addition to quantifying the DOCs and fixed gases from these facilities, they used six methods for trained human panel members to assess the intensity of odor at varying distances from swine facilities. Scentometer measurements were made at 12 feet, 750 feet, and 1250 feet from the swine facilities and range from a high of 170 D/T (dilutions to threshold) to a low of 2 D/T.

It is recognized that there is great variability between odors arising from CAFOs, and that odorous gases may be transformed through interactions with other gases and particulates between the source and the receptor (Peters and Blackwood, 1977). It is also recognized that there is variability in odor persistence, "persistence factor" defined as the relative time that odorous gases will remain perceptible (Summer, 1971). There is a need to combine quantitative assessments of odors with environmental measurements in well-designed, controlled studies of symptoms and other health outcomes at the community level.

6.3.1.6. Experimental Occupational Exposures among Naïve Subjects

Workers in CAFOS are exposed, on a daily basis, to a wide array of gases, vapors, dusts, and other compounds. Thus, it is challenging to identify, in this occupational setting, which specific components of their exposure is responsible for health outcomes. Experimental occupational exposures among normal volunteers have addressed this issue.

Two clinical epidemiological studies of normal volunteers in swine CAFOs have been reported, both from Canada. Cormier and colleagues (1997) exposed 7 previously non-exposed, normal subjects to a swine building and found significant respiratory symptoms, declines in lung function, and clear evidence of a marked inflammatory response via analysis of bronchoalveolar lavage (BAL) fluid post exposure. Total dust, endotoxins, and ammonia were measured but no individual exposures, rather a mixed exposure, appeared to be responsible for these adverse health effects. Senthilselvan and colleagues (1997) made similar observations among 20 naive subjects, while also showing that treatment of the swine facility with canola oil significantly reduced symptoms, declines in lung function, airway hyperresponsiveness, and mean dust and endotoxin concentrations.

6.3.2. Occupational Health Effects

The first description of health hazards to people working in these CAFO's was in 1977 (Donham et al, 1977). This early study revealed that over 60 percent of veterinarians working in these facilities experienced one or more respiratory health symptoms. This report led to many subsequent studies in the US, Canada, and Europe (Donham, 1993). In addition to respiratory illnesses, other occupational health problems associated with CAFOs have been documented, including traumatic injuries, noise-induced hearing loss, needle sticks, hydrogen sulfide and carbon monoxide poisonings, and infectious diseases (Donham et al, 1982a; Donham et al, 1982b; Donham, 1985).

Workers in confined poultry and dairy operations are also at risk, but most beef operations are in open lots, thus reducing worker respiratory exposures. The increasing industrialization of livestock production will continue to result in more independent producers leaving the industry, or becoming quasi-employees of large-scale producers as contract growers. Furthermore, many minority workers are becoming employees of larger producers, raising potential legal issues of undocumented workers and further need of OSHA regulation of these large operations. In the past, OSHA has been restricted in agriculture because of a federal law that restricts enforcement on farms with ten or fewer employees. Many of the large industrial CAFOs now employ hundreds of workers and these workers will work full shifts in animal confinement buildings in contrast to smaller, independently owned CAFOs where periods of exposure are typically much shorter. This increase in large, industrial CAFOs will, therefore, likely lead to increased cumulative exposure and thus greater risk to adverse health effects. To date, OSHA has not addressed the CAFO issue, in spite of strong evidence of worker health risks.

The worker health component of this review is assembled to characterize the range of occupational health hazards associated with large-scale livestock production, but concentrates on health effects from air toxics and a brief discussion on measures needed to decrease health risks among workers.

Table 2 lists major categories of hazards and then further classifies diseases or health outcomes within those categories. The order does not necessarily relate to incidence, prevalence, or severity—these are common health risks among all intensive livestock production operations. The vast majority of the research in this area has been with swine production. Therefore, this report will deal largely with swine operations. However, similar exposures and adverse health effect observations have been made among those working in concentrated poultry production (Bar-Sela et al, 1984; Lenhart et al, 1990; Morris et al, 1991).

The principal health risks for CAFO workers result from respiratory exposures to a wide range of toxic, irritant, and inflammatory substances emitted into the air. Ammonia, hydrogen sulfide, carbon monoxide, particulate matter, endotoxin, and other bioaerosols have received the majority of research attention. However, infectious diseases, noise, trauma, fires, explosions, electrocutions, thermal stress, poisonings, and drowning are all also important causes of morbidity and mortality (Randolph and Rhodes, 1993). Often overlooked are emotional stress and chronic musculoskeletal pain that can lead to significant impairment and to disability in this workforce. This report will be limited to air toxics and resultant respiratory diseases.

6.3.2.1. Respiratory Diseases

Respiratory exposures lead to the most common health hazard among swine farmers and CAFO workers. There are both acute illnesses and chronic respiratory diseases among CAFO workers. The most serious acute hazard is hydrogen sulfide poisoning, which results from sudden exposure to high levels (> 500 ppm) of this gas. This is a confined space entry hazard (areas that are not vented and may trap toxic gases) in CAFOs, with hydrogen sulfide the principle hazard (Donham et al, 1982a; Osbern and Crapo, 1981). Acute respiratory distress syndrome (ARDS), or pulmonary edema, can result in CAFO workers from acute or chronic exposure to hydrogen sulfide (H_2S). There have been at least 19 acute deaths in workers resulting from sudden H_2S exposure of above 500 ppm secondary to liquid manure agitation. These people may collapse and stop breathing following only a few breaths at this high exposure (hydrogen sulfide is an asphyxiant). Severe pulmonary edema from the irritant properties of hydrogen sulfide and death may result. Longer-term lower exposure may also lead to ARDS during or following an accumulative or multiple period exposure (Donham et al, 1982a).

Other respiratory illnesses result from less acutely toxic exposures and lead to non-fatal acute lung insults as well as chronic declines in lung function (Bongers et al, 1987; Choudat et al, 1994; Cormier et al, 1997; Crook et al, 1991; Donham et al, 1984). Respiratory problems associated with this environment are listed in Table 3 by upper respiratory tract, airway, interstitial, and mixed airway and interstitial lung diseases. The pathogenesis of these respiratory diseases is primarily acute and chronic airway inflammation. Classical immunologically mediated asthma and hypersensitivity pneumonitis appear to be uncommon among CAFOs workers (Matson et al, 1983).

Acute bronchitis is the most common complaint among CAFOs workers, affecting as many as 70 percent of those exposed. This is an irritant-induced inflammatory condition of the airways. The symptoms of bronchitis are cough and sputum production. Chronic bronchitis is noted by chronic phlegm for two or more years. This condition affects about 25 percent of CAFO workers. Acute and chronic bronchitis may be accompanied by an asthma-like condition, with symptoms of chest tightness, wheezing, difficulty breathing, and shortness of breath (the symptoms most typically reported).

Frequent upper respiratory tract conditions include sinusitis and rhinitis. Some studies have referred to these collectively as mucus membrane irritation (MMI) (Rylander, 1994; Rylander et al, 1989). MMI may be attributable to the combination of bioaerosol, endotoxin and ammonia and other irritant exposures (Donham, 1986; Donham et al, 1986a).

Sinusitis is often chronic among CAFO workers who may complain of a continual or frequent cold "they just cannot shake," of a stuffy head, difficulty in breathing through the nose, headache, and/or "popping ears." These symptoms are a result of a noninfectious, toxic inflammation and swelling of the mucus membranes of the sinus cavities and the Eustachian tubes leading to the middle ear. This is often accompanied by a chronic irritant rhinitis and pharyngitis.

Allergic rhinitis (also called hay fever) has rarely been attributed to confinement exposures. Such persons may have a specific allergy to some component of the swine environment. These symptoms are similar to irritant rhinitis, except it usually develops after only brief exposure to the environment and may be accompanied by itchy, watery eyes and possibly acute chest tightness (allergic asthma). Workers with pre-existing allergic rhinitis often self-select themselves out of CAFO work which contributes to a selected, or survivor, population of CAFO workers.

An asthma-like syndrome, similar to byssinosis (a condition of workers exposed to cotton and other vegetable textile dusts), has been described among CAFO workers. This condition is characterized by chest tightness, wheezing, and/or cough on return to work after two or more days of work absence, and mild acquired airway hyperresponsiveness. It may occur early in exposure to the CAFO environment and is not an immunologically mediated condition. It was documented in 11 percent of a population-based study of Iowa swine confinement workers (Donham et al, 1990).

Occupational asthma includes periodic airway obstruction, chest tightness, wheezing, and dyspnea, does not occur on first exposure, but may develop after weeks to months of CAFO exposure. CAFO workers with pre-existent asthma typically experience severe asthma upon first exposure to animal confinement facilities and select themselves out of these jobs. Occupational asthma may result from repeated exposure to the work environment. It has two basic mechanisms: 1) immunologically mediated or allergic, or 2) chronic irritation. Rarely have there been documented allergic (IgE) mediated causes for CAFO workers' illnesses. These "susceptible" workers almost always leave the work force early because of severe asthma, and the condition is very difficult to manage among workers who continue to work in the CAFO environment. Non-allergic occupational asthma, asthma-like syndrome, and/or reactive airways disease, has been found to be common (up to 20 percent) of current CAFO workers. This condition may lead to progressive declines in lung function and chronic obstructive pulmonary disease, which is a chronic irreversible condition (Schwartz et al, 1992; Schwartz et al, 1995b). CAFO exposures, dust concentration, endotoxin concentration, and cross-shift decline in lung function (FEV1) have been found to be significant determinants of progressive decline in lung function over time (Reynolds et al, 1996; Schwartz et al, 1995a; Schwartz et al, 1995b; Vogelzang et al, 1998; Vogelzang et al, 2000).

Occupational asthma is distinct from organic dust toxic syndrome (ODTS). ODTS results in a flu-like spectrum of symptoms including headache, joint and muscle pain, fever, fatigue and weakness, cough, shortness of breath, and irritation of the airways and the cells lining the small sacs of the lung. ODTS may be clinically mistaken for farmer's lung, as they have similar acute symptoms, e.g., the delayed onset of severe influenza-like symptoms, following exposure. However, farmer's lung (hypersensitivity pneumonitis) is seen (now rarely) in mainly dairy farming operations, but has not been documented in swine workers (Rylander, 1994). However, 33 percent (Donham et al, 1990) of swine producers have reported episodes of ODTS, which is an influenza-like illness followed by exposure to a higher than usual dust load, e.g., moving and sorting hogs. A chronic or sub acute condition (a possible variant of ODTS) has been described among swine workers and is characterized by chronic fatigue and possibly persistent mild pulmonary infiltrates (Auger, 1992). However, there are only anecdotal cases observed and no human studies that have been conducted (Donham, 1993); there is some evidence for a persistent pulmonary infiltrate condition from one animal study (Donham and Leininger, 1984).

It is recognized that several of these respiratory conditions may occur in an individual CAFO worker, and they may occur at the same time. It is possible, for instance, for an individual CAFO worker to have signs and symptoms of an asthma-like condition, bronchitis, and episodes of ODTS. This produces an interrelated group of conditions (a syndrome) of illness caused by exposure to the swine building environment (Table 1).

6.3.2.2. Control of the Occupational Environment

CAFO worker health risks can be significantly reduced through a comprehensive program of environmental monitoring and control through the use of management practices, engineering controls, judicial use of personal protective equipment, and health surveillance. However, such programs are exceedingly rare in today's CAFO industry. There is little to no exposure monitoring except for research purposes, and routine health surveillance in this worker population is rare. Engineering controls are generally implemented if they will benefit hog production, but rarely with worker health as the principal motivation. There is some evidence to suggest that healthy swine confinement workers can usually tolerate exposures to total dust (2.5 mg/m^3), respirable dust (0.23 mg/m^3), ammonia (7 ppm), endotoxin (100 EU/m^3), and micro-organisms/m^3 (10^5) without experiencing significant acute respiratory symptoms (Donham et al, 1986a; Donham et al, 1986b; Donham et al, 1990; Reynolds et al, 1996). However, further studies are needed to confirm these findings and to assess the combined effects of common CAFO exposures, including ammonia, endotoxin, and the use of disinfectants, which together appear to influence respiratory disease outcomes (Preller et al, 1995).

It is important to recognize that CAFO workers are a survivor population, meaning that the most severely affected workers have already left the workplace. In addition, there is evidence that workers exposed to inhaled endotoxin develop a tolerance (at least to acute symptoms) to this toxicant.

However, long-term exposure may lead to chronic airway obstruction, even in the absence of acute symptoms. Some previously unexposed individuals in the general community population would be expected to react acutely to lower concentrations of CAFO exposures.

Management practices and engineering controls can significantly reduce exposures to inhaled toxicants (Senthilselvan et al, 1997). These include frequent facility cleaning (frequent power washing from floor to ceiling, at least every three weeks); addition of extra fat and a urease inhibitor, e.g., microaid, to the feed; self-cleaning flooring; and improved lagoon operation (Mutel et al, 1992). The ventilation system, by itself, cannot necessarily assure a healthful environment. Health surveillance and the management procedures, mentioned above, must also be implemented. Also, the ventilation system must be properly engineered and maintained; very often, higher cool weather exchange ventilation rates are needed; and lower animal density (swine mass per unit of barn volume) may be required.

Personal protective equipment should not be considered an effective alternative to good management practices and engineering controls. Without a properly supervised respirator program, it is very difficult to assure that exposed personnel will wear the right respirator and that it fits properly, functions properly, and is worn at the appropriate time. Respirators are not well tolerated, especially for strenuous work in a hot, humid environment. The Occupational Safety and Health Administration (OSHA) requires that if respirators are worn to protect workers, they must be worn at all times, and be fit, maintained, and stored properly through an appropriately supervised respirator program. Respirators are an adjunct to management practices, engineering controls, and health surveillance, especially for specific tasks that result in higher-than-normal exposures or for workers in need of increased protection.

Special attention should be given to pregnant women who work in swine confinement facilities. The unborn fetus is susceptible to carbon monoxide and hormonal drugs used in swine production (e.g., oxytocin and prostaglandins). Pregnant women may be at increased risk for spontaneous abortion if they work in swine barns (Donham and Gustafson, 1982).

6.3.2.3. Relationships Between Indoor and External Air Environments

One cannot directly extrapolate occupational health risks observed inside CAFOs to community health risks outside swine production. Although there is discharge of airborne particulates and gases/vapors from the swine barns to the exterior environment, the aerosols differ considerably in composition and in the concentration of specific agents. As aerosols and gases/vapors emanate from a point source travel downwind, the aerosols disperse, become less concentrated and adsorbed gases/vapors may be stripped from particles. There may also be photochemical reactions and ground deposition. Volatile organics present in the outdoor air in the vicinity of a swine production facility may arise from outdoor manure storage facilities and manure application, in addition to particulate and gases in air discharged from the confinement facilities.

Although there is theoretically a definable dose-response relationship for respiratory diseases by individual compounds, the exposures inside CAFOs are always a complex, mixed exposures and differ in many ways from those outside. Perhaps equally important is the fact that the CAFO and community populations are quite different in terms of susceptibility factors. Some members of the general population, including susceptible children, the elderly, asthmatics, and other susceptible individuals, would be expected to develop responses at much lower doses than healthy workers.

Furthermore, individuals living in the vicinity of CAFOs and who may have their quality of life and social and economic conditions affected and feel stress because they have no control over their living conditions.

6.3.2.4. Conclusion

The scientific literature is quite clear that workers in swine or poultry CAFOs are at risk to acute and chronic respiratory diseases from concentrated emissions inside CAFOs. There is, however, adequate information to protect workers, if the industry and regulators take steps to do so--including monitoring engineering, administrative, and personal protective equipment. The swine and poultry industry needs to develop and manage exposures to their workers, and OSHA should take action to protect the health and safety of workers under their jurisdiction.

6.3.3. Community-Based Studies

Community exposures to environmental contamination, most of which has arisen from industrial and agricultural technology over the last 100 years, are now well-recognized public health problems. Exposures include a vast array of chemicals, noise, and ionizing radiation. Other sources of environmental contamination have arisen from the products of armed conflicts, including the some 250,000 American veterans and their families who were exposed to ionizing radiation during the above ground atomic bomb testing program from 1945 to 1962 (Ellis et al, 1992), those exposed to a variety of environmental agents, in addition to a hostile environment, in the Persian Gulf War (Schwartz et al, 1997) and community residents living in proximity to industrial sites in California (Shusterman et al, 1991). These examples, like community exposures to CAFOs, involve environmental exposures under circumstances in which there is little or no environmental control by the affected community.

Ellis has defined community environmental contamination "as a stress that is unique in terms of: 1) its physical characteristics and resultant adaptational dilemmas, 2) the agent or cause of the injury, and 3) the institutional responses to the contamination" (Ellis et al, 1992). Asked by the Centers of Disease Control to assess any adverse health effects of Iowans who served in the Persian Gulf War, The University of Iowa Persian Gulf War Study Group assessed a number of specific and non-specific health outcomes and a number of environmental exposures as well as global exposure to the Persian Gulf War theater among a sample (n=3695) of active and reserve military personnel who served in the war theater and elsewhere during the study period (Schwartz et al, 1997). Significantly higher prevalence of symptoms of depression, posttraumatic stress disorder, chronic fatigue, cognitive dysfunction, bronchitis, asthma, fibromyalgia, alcohol abuse, anxiety, and sexual discomfort were observed. Assessment of health-related quality of life demonstrated diminished mental and physical differences among the PGW as compared with non-PGW military personnel. While significant associations were observed with a number of self-reported environmental exposures during this time period, the exposures and constellation of symptoms did not fit well into an established category of disease or syndrome, but were similar to previous reports of veterans from previous wars thought to arise from the stresses of war. The specific environmental causes of the increased adverse health effects could not ascertained (nor could they be ruled out) from this study and recall bias, to which any such survey is subject, could also not be ruled out as a contributing factor to these associations. Shusterman and colleagues (1991) studied both "environmental worry" and self-perceived environmental odors (especially petrochemical) among 2000 Californians living in proximity to three industrial sites, as well as control sites. Observations found that both "environmental worry" and perceptions regarding odor were associated both independently and interactively with symptom reporting. Recall bias was recognized as a potential confounder for some of these findings. These methodological approaches are relevant to studies of other community environmental exposures, such as those that arise from CAFOs that include both specific environmental agent exposures and more global (odors/mixed exposures) community exposures, arising from a given source(s) of environmental exposures.

6.3.3.1. Community Studies of Concentrated Livestock Exposures

Schiffman and colleagues (1995) studied North Carolina residents who lived in the vicinity of intensive swine operations (n=44), and compared with matched control subjects who did not live near such operations (n=44). Using a validated Profile of Mood States (Schiffman et al, 1995) they found more negative mood states among those living in proximity to swine operations. The factors affected included tension, depression, anger, reduced vigor, fatigue, and confusion. Greater total mood disturbance was also reported by those living near swine operations. These authors suggested that a variety of factors may have affected the mood of those exposed to odors and living in proximity to swine facilities.

Thu and colleagues (1997) found no difference in the clinical levels of depression or anxiety between Iowans (n=18) living within two miles of a 4,000 sow CAFO and a random sample of demographically similar rural residents (n=18) living near minimal livestock production. However, higher rates of four clusters of symptoms common among CAFO workers and associated with toxic air exposures were observed: (Cluster 1: sputum, cough, shortness of breath, chest tightness, wheezing, p=.02; Cluster 2: nausea, dizziness, weakness, fainting, p=.04; Cluster 3: headaches, plugged ears, p=.06; Cluster 4: runny nose, scratchy throat, burning eyes, p=.12), whereas other symptoms including muscle aches, hearing problems, skin rash, and fever did not differ between the two groups. The authors drew attention to the similarities between the pattern of symptoms among these community residents and CAFO workers and suggested that a larger population-based study was needed.

Wing and Wolf (2000) conducted a population-based study of three rural North Carolina communities, one of which was in the vicinity of a 6,000-head hog operation, one in the vicinity of two intensive cattle operations, and a third area without "liquid waste" livestock operations. A standardized questionnaire was administered by trained interviewers to ascertain health symptoms and indicators of quality of life during the previous 6 months. 155 interviews were completed with a participation rate of 86%. Those living in proximity to the swine operation reported increased rates of headaches, runny nose, sore throat, excessive coughing, diarrhea, and burning eyes compared to rural residents with no livestock operation. Quality of life measures among those living in the vicinity of the swine operation were greatly reduced. The authors were aware of potential recall bias and, therefore, presented the study as a "rural health" study which did not include any questions about hogs, livestock, or odors. They also pointed out that eight symptoms in the miscellaneous category did not differ between the hog and control communities, thereby minimizing the likelihood of significant recall bias.

Hodne and her University of Iowa colleagues are currently testing the relative power of aspects of medical models and bio-psychosocial models to assess the mental health consequences of CAFO community exposures. For example, they report greater traumatic cognitions associated with post-traumatic stress disorder among residents of rural areas with many CAFOs and areas with traditional livestock production than among rural residents in areas with very little livestock (Hodne, 2001).

They are also exploring the types of stress responses in CAFO neighbors that may mediate the relationship between air emissions and odors and physical and mental health outcomes.

The three published, peer reviewed studies of community residents exposed to CAFO emissions are limited and should be interpreted with caution because of the relatively small numbers of participants, because they did not report environmental exposure data and likely contain some recall bias. However, they are notable because they were all well designed, controlled studies and because the two of the three that examined respiratory and other symptoms common among CAFO workers found similar symptom patterns (while not as prevalent or severe) as those observed among CAFO workers. Two of the three studies also reported indicators associated with diminished a quality of life among those living in proximity to livestock facilities as compare to community controls.

6.3.4. Conclusion

Numerous occupational studies have documented significant increases in respiratory disease and other respiratory adverse health effects, including CAFO-related deaths, acute and chronic respiratory diseases and associated symptoms and acute losses in exposure-related lung function and progressive respiratory impairment, among those who work in CAFOs. However, it is recognized that the CAFO workforce is generally healthy, while those in the general community, including children, the elderly, those with chronic impairments such as pre-existing asthma or chronic obstructive pulmonary disease, are expected to be much more susceptible to CAFO exposures. There is experimental and epidemiological evidence that very low levels of exposures to ammonia, hydrogen sulfide, known to be ambient air toxic gases arising from CAFOs, may result in adverse health effects among healthy volunteers and community residents. While limited in number and scope, the currently published, peer reviewed, community-based studies of adverse health affects associated with CAFO exposures find an increased prevalence of similar symptom patterns, especially respiratory symptoms, and similar indicators of reduced quality of life. Taken together with other experimental and epidemiological observations of adverse health effects observed with low levels of exposures to chemical components (ammonia, hydrogen sulfide) of CAFO emissions, these findings support a conclusion that CAFO air emissions constitute a public health hazard, deserving of public health precautions as well as larger, well controlled, population-based studies to more fully ascertain adverse health outcomes and their impact on community health services.

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TABLE 1
Volatile Compounds Associated with Pig Wastes

Methanol	Methanal	
Ethanol	Ethanal	Ammonia
1-Propanol	Propanal	Methylamine
2-Propanol	Butanal	Ethylamine
1-Butanol	Pentanal	Trimethylamine
2-Butanol	Hexanal	Triethylamine
2-Methyl-1-propanol	Heptanal	Carbonsulphide
3-Methyl-1-butanol	Octanal	Hydrogen sulphide
2-Ethoxy-1-propanol	Decanal	Methanethiol
2-Methyl-2-pentanol	2-Methyl-1-propanal	Dimethylsulphide
2,3-Butanediol	Ethylacetate	Dimethyldisulphide
	Methanaoic acid	Dimethyltrisulphide
	Ethanaoic acid	Diethyldisulphide
3-Hydroxy-2-butanone	Propanoic acid	Propanethiol
Propanone	Butanoic acid	Butanethiol
2-Butanone	2-Methylpropanoic acid	Dipropldisulphide
3-Pentanone	Pentanoic acid	2-Methylthiophene
Cyclopentane	3-Methylbutanoic acid	Propylprop-1-enyldisulphide
1-Octanone	Hexanoic acid	2,4-Dimethylthiophene
2,3-Butanidione	4-Methylpentanoic acid	2-Methylfuran
	Heptanoic acid	
	Octanoic acid	
Phenol	Nonanoic acid	
4-Methylphenol	Phenylacetic acid	
4-Ethylphenol	2-Phenylpropanoic acid	
Toluene		
Xylene		
Indone		
Benzaldehyde		
Benzanoic acid		
Methylphthalene		
Indole		
Skatole		
Acetphenone		
o-Aminoacetophenone		
Aneline		

Source: 1995. Proceedings, "Understanding the Impacts of Large-Scale Swine Production," June 29-30, Des Moines, IA. The University of Iowa Printing Service, Iowa City, IA, pg 51.

TABLE 2
Major Hazard Categories in Swine Production.

Hazards	Subcategories	Examples
Chemical Hazards	Asphyxiation	Carbon monoxide
	lung injury	Nitrogen oxides, ammonia
	contact dermatitis	Allergic, irritant
	Poisonings	Pesticides, fuels, cleaning agents
	Intoxication	Solvents, silo gas, substance abuse
	Immunomodulation	Adjuvants: biocides, phytotoxins
		Immunosuppressants: pesticides
Biological Hazards	Microorganisms	Pathogenic
		Non-pathogenic
	organic dust	Bacterial toxins: endotoxins, exotoxins, enterotoxins
		Fungal toxins: mycotoxins, glucans
		Phytotoxins
		Inflammatory agents
	Aeroallergens	Arachnid detritus
		Animal proteins
		Allergenic fungi
Infectious Hazards	Zoonotic	Systemic
	non-zoonotic	Lung
	antibiotic resistance	Skin
	emerging pathogens	Ocular conjunctivitis
Biomechanical Stress	Trauma	Animal bites
		Falls
		Needle sticks
		Punctures, lacerations, abrasions, burns
		Crushing injuries
		Repetitive trauma
	Noise	Noise-induced hearing loss
		Reduced safety from impaired hearing
Thermal Stress	heat stress	
	cold stress	
Emotional Stress	Occupational	Suicide
	Marital	Depression
	Financial	Anxiety
Drowning		Lagoons
		Pits
		Farm ponds
Fires/explosions	Chemical	Methane in pits
	Electrical	Ignited building materials or feed
	Welding	Ignited building materials or feed
	organic material	Grain, grain dust, compost, hay
Electrocution		Faulty wiring
		Water associated
Chronic pain	Biomechanical stress	Arthralgia
	Arthritis	Myalgia
Fatigue	sleep deprivation	Planting, harvesting
	chronic fatigue syndrome	Chronic endotoxin exposure

Source: 1995. Proceedings, "Understanding the Impacts of Large-Scale Swine Production," June 29-30, Des Moines, IA. The University of Iowa Printing Service, Iowa City, IA, pg 156.

TABLE 3: Respiratory Diseases Associated with Swine Production

Upper Airway Disease
Sinusitis
Irritant Rhinitis
Allergic Rhinitis
Pharyngitis
Lower Airway Disease
Organic Dust Toxic Syndrome (ODTS)
Occupational Asthma
Nonallergic asthma, hyperresponsive airways disease, or reactive airways disease syndrome (RADS)
Allergic asthma (IgE mediated)
Acute or Subacute Bronchitis
Chronic Bronchitis
Chronic Obstructive Pulmonary Disease (COPD)
Interstitial Disease
Alveolitis
Chronic Interstitial Infiltrate
Pulmonary Edema

Source: 1995. Proceedings, "Understanding the Impacts of Large-Scale Swine Production," June 29-30, Des Moines, IA. The University of Iowa Printing Service, Iowa City, IA, pg 158

Chapter 7. Social and Community Impacts

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Introduction
The impacts of Concentrated Animal Feeding Operations (CAFOs) should be judged in terms of their socioeconomic impacts on rural Iowa and its communities as well as their impacts on human and animal health. Regulations and management practices should support socially and economically desirable community outcomes, as well as protect human and animal health. It is the role of government to select from among the regulatory options that contribute to economically viable, socially equitable, and environmentally sound communities (President's Council on Sustainable Development, 1996).

7.1 Quality of Life and Community Social Capital
Quality of life factors are emphasized in recent literature addressing the community impacts of CAFOs. The state of Minnesota recently brought together the scientific and public policy communities to advise state government on how to address several CAFO issues, resulting in a Generic Environmental Impact Statement (GEIS) for animal agriculture. It suggests, "Quality of life is related to perceptions of 1) having alternatives in what one does on a daily or life cycle basis, and 2) being respected by family and communities of interest and place." (Flora et al., 1999:A24).

An important aspect of community quality of life is social capital, which includes mutual trust, reciprocity, and shared norms and identity. In general, communities with greater social capital provide greater quality of life (Flora, 1998; Flora, Sharp, Flora, Newlon, 1997; Sharp, Agnitsch, Ryan, Flora, 2001). Also, social capital emerges as an internal resource in instances of controversies.

7.1.1 Agricultural Structure, Quality of Life, and Economic Vitality
Quality of life issues related to the structure and scale of agriculture were examined as early as the 1940's. More than half a century ago, Goldschmidt (1978; originally published in 1946) compared two rural California communities where the structure and size of farms were different, but where total value of farm production was almost identical. In the town where farms were larger and industrialized (with a higher proportion of absentee ownership and employing a higher proportion of farm workers per unit of output) there was greater separation of social classes, i.e. greater social inequality. More decisions about local affairs were made outside the community. This contrasted with the other community where farms were smaller, more likely to be owner operated, and utilized the labor of the operating family with some hired labor. This community had a richer civic and social fabric: residents of all social classes were more involved in community affairs, more community organizations served people of both middle and working class background, and there were more local businesses and more retail activity because more agricultural and consumer purchases were made locally and more income was in the hands of the classes with a greater propensity to spend. MacCannell, in a macro study that included family-farm and industrial-agricultural communities in 98 industrial-farm counties in California, Arizona, Texas, and Florida, found that mean farm size (in acres), gross farm sales, as well as high levels of mechanization "significantly predict declining community conditions not merely at the local agricultural community level, but in the entire county." (1988, p. 63.)

Recent studies, including those in the Midwest, reveal tendencies of economic decline in communities with greater concentration of CAFOs, similar to Goldschmidt's thesis of greater rural community decline with greater industrialization of agriculture. The econometric analysis conducted by Gomez and Zhang (2000) over a decade revealed the negative impact of swine CAFOs on economic growth in rural Illinois counties, as indicated by sales tax receipts. They found that purchases from small businesses declined as concentration of CAFOs intensified. In a Michigan study, Abeles-Allison and Connor (1990) found that local purchases of supplies for swine production decrease as CAFO concentration increases. Local expenditures per hog were calculated at $67 for the small farms and $46 for the large farms. The difference is largely due to bulk feed purchases from outside the community by the larger farms, but is also related to somewhat greater total expenditures per hog on the smaller farms. Durrenberger and Thu's (1996) finding that increased food stamp utilization is associated with industrialized hog production in Iowa suggests either that industrial agriculture generates inequalities or that industrial agriculture thrives in counties with greater inequalities

Foltz, Jackson-Smith and Chen (2000) examined local purchasing patterns of large and small dairy farms in Wisconsin. They found that the percent of dairy feed purchased locally declined as herd size increased. Stronger indicators of local feed purchasing were the physical nearness to and social attachment to the community. In Minnesota, Chism and Levins (1994) found that local spending was not related to gross sales volume on *crop* farms. However, local farm-related expenditures fell sharply when the scale of livestock operations increased.

Otto, Swenson, and Lawrence (cited in Kliebenstein, 1998) found that local property tax revenues and state revenues in Iowa, calculated on a per sow basis, were as follows:

Table 7.1. Net Benefits And Net Revenues To Local And State Governments From Farrow To Finish Operations, Iowa

Size of operation	150 sows	300 sows	1,200 sows	3,400 sows
Net Local Government Benefit per sow	$8.84	$9.35	$10.43	$8.23
Net revenues to State Government per Sow	$16.01	$17.19	$14.59	$12.86
Sum of local and state revenues	$24.85	26.54	$25.02	$21.09

Overall, more moderate-sized farrow-to-finish operations generated more local and state revenues per sow than did small or very large ones.

Quality of life issues that relate to agricultural structures are evident in Eastern North Carolina. This region experienced a tremendous growth in the hog industry beginning in the 1980's that includes both contract and corporate production facilities and meatpacking plants. Many citizens there perceive that this ha s left them with a power structure in which the interests of large pork producers dominate those of local residents at all levels of government (McMillan and Schulman, 2001; Thu and Durrenberger, 1994).

In North Carolina, Wing, Cole, and Grant (2000) have found patterns of disproportionate siting of corporate CAFOs in rural lower-income and African-American communities. This places residents of these communities at disproportionate risk for health and socioeconomic problems.

7.1.2 Quality of Life, Community Social Capital and Community Conflict

Wing and Wolf's (2000) study of 50-55 individuals from each of three North Carolina rural communities showed that quality of life was greatly diminished among who residents near a

6,000-head swine confinement operation, compared to residents near two intensive cattle operations or near an agricultural area without livestock operations that required liquid waste management. Quality of life was indicated by the number of times that neighbors could not open their windows or go outside even during nice weather due to CAFO odors.[1] Thirty percent of respondents from around the hog CAFO as compared to a maximum of three percent from the other two communities indicated that each of these problems had occurred 12 or more times during the past six months. Many rural residents comment that it is difficult to plan social activities in their homes because of the uncertainty of whether the air will be tolerable for guests (see Donham & Thu, 1996; Wright et al., 2001, pp. 28-30, for similar health and social responses near Minnesota CAFOs). Such limitations on social relations with one's neighbors indicate a decline in community social capital (Ryan, Terry, & Besser, 1995).

Lasley's Iowa Farm and Rural Life Poll (1998) shows substantial concern among Iowa farmers about hog odors. In the 1992 and 1998 polls, respondents were asked "how many days per year they would be willing to tolerate odors from a neighbor's livestock operation before they would consider it a major nuisance." Fourteen percent were unwilling to tolerate more than two days; 34% were willing to tolerate only a week or less, and fully 50% would view odors as a major nuisance if they affected them as many as ten days out of the year. The latter figure rose from 44% in 1992 (Lasley, 1995). Three-fourths of Iowa farmers live within half a mile of a neighbor. In addition the proportion of respondents agreeing with the statement, "Increasingly, manure management is a major issue in the livestock industry," rose from 61% to 85% of Iowa farm respondents between 1992 and 1998.

Characteristics of the nearest CAFO and of the affected neighbor influence the latter's level of annoyance with CAFO odors. Van Kleek and Bulley (1985), in a study conducted in the early 1980s in British Columbia, chose 14 swine farms, 14 beef feed lots, 11 laying hen firms, and 10 broiler farms located at least 800 meters (somewhat less than ½ mile) from any other livestock farm. A least 12 residences (non-producers of livestock) were within 800 meters of each livestock farm. Those residents rated their perception of the livestock farm "as it relates to your living here" on a five-point scale from "no nuisance/very compatible" to "severe nuisance/incompatible."

The authors found that nuisance potential decreased with distance, but it decreased the least for hog farms. Larger farms were a greater nuisance than smaller ones, but the difference disappeared for residences that were at very close ranges from the livestock farm. Hog farms were considered the greatest nuisance, followed by cattle feedlots and then by poultry CAFOs. Odor represented 75% of the total nuisance, but the proportion differed according to the type of farm; for hog farms, 95% of the nuisance responses related to odor; for broilers, ¾; for layers, 2/3; and for feedlots, only about half. People with rural backgrounds were less tolerant of livestock farms than were those who had come from urban areas; those with farm backgrounds did not differ from those without farm backgrounds. Lohr (1996) found that among neighbors of a swine farm, tenure of residence, previous contact with the

[1] Miedema and Ham (1988) used an independent dispersion olfactometric testing method in a study designed to determine if specific complaints and symptoms from odors were indeed correlated with independent measurement of the presence of agricultural and industrial odors. Individuals living near a pig sty, a rapeseed oil extraction plant, and an electric wire insulation factory, were surveyed. Level of annoyance with the odor and reported frequency of having to shut windows because of the odor were linearly related to the frequency of detection of odor using the olfactometric test. Interestingly, the pattern of relation was not specific to the type of odor being measured.

farmer, and economic dependence on farming all negatively correlated with the degree of odor annoyance.

Debate continues, in popular and academic circles, on whether CAFO odors are best characterized as primarily nuisances of varying degrees or whether these odors are also linked to negative health outcomes (Thu, 1998). Donham (2000) describes possible non-toxic mechanisms for CAFO odors to generate physical symptoms through complex interactions of the brain and somatic systems. Shusterman (1992) describes some of these mechanisms in his review of the health impacts of environmental odor pollution.
The well-researched linkage of physical symptoms to the uncontrollability of various stressors including environmental stressors (e.g., noise) may be applicable to CAFO odors as noted in Chapter 6.3. In addition, the variety of family, neighborhood, and community stressors sometimes associated with CAFOs may also generate stress-induced symptoms and illness. However, these possible linkages have not yet been reported.

All sides of CAFO controversies tend to frame their issues and identities in terms of rights and entitlements, as described in McMillan and Schulman's (2001) research on the hog industry in North Carolina. For example, producers defend their property rights and a right to earn a living from their land, while neighbors defend their right to enjoy their own property. De Lind (1995) documents that in response to local opposition to a corporate CAFO or "hog hotel" in Parma township in Michigan, the Farm Bureau, the Pork Producers Council, and other agricultural interests defended the right of "hog hotels" to exist without regulation by appealing to the right to farm.

Constance and Bonanno (1999) document actions of anti-CAFO groups in the Texas Panhandle. They focus on episodes of resistance carried out by local residents and environmental groups who were mainly motivated by human health and property value concerns. Corporate responses to community resistance primarily involved reconstruction of their corporate image as environmentally friendly.

A decline in social capital is associated with swine CAFOs, according to rural residents of Iowa, North Carolina, Minnesota, Michigan, and Missouri who describe violations of core rural values of honesty, respect, and reciprocity, as reported in an interdisciplinary workshop held in Iowa on swine CAFOs (Thu et al., 1995, p. 76). For example, CAFO neighbors often consider it a violation of respect when their concerns are labeled as emotional, perceptual, and subjective or are dismissed as invalid or unscientific.

Recent findings are presented by Kleiner, Rikoon and Seipel (2000), who found that in two northern Missouri counties where large-scale corporately owned swine CAFOs are dominant, citizens expressed more negative attitudes regarding trust, neighborliness, community division, networks of acquaintanceship, democratic values, and community involvement. The county that was dominated by independently owned swine operations had the most positive attitudes regarding trust, neighborliness, community division and networks of acquaintanceship.

The siting of a swine confinement facility in Parma, Michigan in the mid-80s (DeLind 1995, 1998) generated conflict when the firm established a five-unit CAFO with manure lagoons. Neighbors believed the three open-air 42 million gallon lagoons compromised their health and quality of life. Local resistance culminated in the emergence of two grassroots organizations and a four-year litigation process. Consequences of this conflict were anger on the part of residents who believed that their environment and their integrity had been violated, resentment towards public officials, polarization within the community, vandalism, alienation, and verbal threats and physical aggression by both sides. Although the opponents of the CAFO won the battle on the local level (the CAFO went bankrupt), when they were interviewed a few years later, they felt the personal acrimony and divisions in the community resulting from conflict over the smell from the lagoons were too high a price to pay.

Wright et al. (2001) reported results from a six-county study in southern Minnesota regarding changes in animal agriculture. Over one hundred producers, community leaders, and others were interviewed, either in roundtable discussions or individually. Three patterns reflect the decline of social capital that resulted from the siting of CAFOs in all six rural communities: 1) widening gaps between farmers who produce livestock within CAFOs and their neighbors, including non-CAFO livestock producers; 2) harassment of vocal opponents of CAFOs; and 3) perceptions by both CAFO supporters and opponents of hostility, neglect or inattention by public institutions that resulted in perpetuation of an adversarial and inequitable community climate.

The North Central Regional Center for Rural Development (1999) examined recent, dramatic increases in corporate hog production and meatpacking in a rural Oklahoma county. Social capital indicators measured mutual trust, reciprocity, and shared norms and identity. Individual security was measured in terms of crime, and community conflict was measured in terms of civil court cases. The overall crime rate increased dramatically between 1990 and 1997. Violent crimes increased 378 percent compared to the average 29 percent decrease in violent crimes over the same period in comparison farming-dependent counties with no dramatic changes in animal agriculture. Theft-related crimes also increased in the case county by 64 percent, compared to a decrease of 11 percent in comparison counties. Civil court cases, indicating community conflict, increased in the county by 7 percent, while they decreased 11 percent in comparison counties. This study dramatically reveals the costs to social capital in counties experiencing rapid and dramatic change in the structure of animal agriculture.

7.2 Agricultural Restructuring and Population Trends

The primary purpose of this section is to provide background for partially answering Director Vonk's question 4: "What do you think should be done to address any other emerging issues with respect to industrial CAFOs in Iowa?" It is useful to begin with a discussion of rural population patterns in Iowa since the beginning of WWII. That is followed by an examination of recent changes in the structure of animal agriculture (and crop agriculture insofar as it interacts with animal agriculture) and how public policy relates to those changes. The general trends in livestock and poultry production are presented in Chapter 2.

7.2.1 Rural Population Dynamics since WWII

Agricultural restructuring since the initiation of WWII transformed the landscape of rural Iowa. As a result, Iowa's rural population generally has decreased across the decades. Using a definition of rural as an incorporated place with fewer than 2,500 residents plus those who live on farms or in the open country, Iowa had about 1,454,000 rural residents in 1940 and 1,094,000 in 1990. Although final figures are not available from the 2000 census, it appears that a slight increase occurred in Iowa's rural population in the 1990s. Major differences have emerged among three sectors—residents of farms, small towns, and the country. The first of these dropped substantially[2], the second remained much the same[3], and the third grew substantially[4] across the decades.

Small towns tend to have the oldest age structure of the three types; that is, that have proportionately greater numbers of older and fewer numbers of younger residents than do the farm or country categories. This is because many older residents do not move in later life (or if they do change residences, they move from the countryside to nearby towns) and many high school graduates seek urban-based educational and occupational opportunities. This loss of youth is later magnified as they form families elsewhere. The farm population approaches a pyramidal shape, in part because many older residents move from the farm in later life; some others stay on the farmstead but no longer operate the farm (which may be absorbed into a neighbor's farm operation). Of the three groups of rural residents, the country population most closely approaches the classical pyramidal age structure. It includes younger residents with children. Country residents often are newcomers to the area; they may have perspectives that differ from those held by long-term residents. It is not easy to categorize country residents because of their more diverse origins and backgrounds.

7.2.2 Restructuring of Livestock Production in the Past Decade

Until the past decade or so, the industrialization of farm production had largely bypassed Iowa, with the exception of the fat cattle industry, which had its heydey in Iowa in the 1950s and 1960s (see Table 2.9 in this report). In the 1990s, Iowa hog and poultry (particularly egg) production were transformed (see Chapter 2, Table 3 of this volume). Furthermore, different types of animal production systems may generate different socioeconomic impacts at the level of the farm and community. Farmers, rural residents, and others express concern that increasing CAFO production is having negative impacts on the traditional family farm structure (e.g., Halverson, 2000). Buttel and Jackson-Smith (1997) surveyed 1,100 randomly selected Wisconsin farmers in 1995 and repeated the survey with 1400 farmers in 1999 (Jackson-Smith, et al., 2000) regarding their views toward large-scale livestock production. Only 17 percent of the respondents perceived expansion in the livestock industry as a good initiative, while 45 percent perceived it to be negative. Only 15 percent indicated that non-farm investors should invest in dairying in the local community (Buttel and Jackson-Smith, 1997). Results were similar in 1999.

Wisconsin farmers' views towards livestock expansion were not shaped primarily by concerns about the environment but instead by concerns about farm structure in their state. Farmers' responses indicated strong support for family-scale operations as opposed to large-scale farms using hired labor-type and to investor-owned dairy operations.[5] The authors

[2] Number of persons resident on farms has declined across many censuses. Since 1940, when 917,000 lived on farms, Iowa lost at least 120,000 farm people each decade to 1990, when 257,000 were counted, the most recent data available (the 2000 farm population will be released later in 2002). The number of farms in Iowa decreased from about 213,000 farms in 1940 to 91,000 in 1997.

[3] Small towns (fewer than 2,500 inhabitants) contained about the same number of residents in 1990 (460,000) as they held in 1940 (471,000). From 1990 to 2000, 464 of the 829 towns with fewer than 2,500 residents in 1990 increased in size. Only among the smallest-sized category—places with fewer than 100 residents in 1990—did a majority of towns decline in population.

[4] About 66,000 country residents were counted in 1940 and 377,000 in 1990. In 1990, for the first time, country residents outnumbered farm residents in Iowa. Strong increases among country residents have occurred in each decade for which data are available. Gains among country residents tended to occur across counties regardless of the trends among farm or small-town residents.

[5] The Iowa Farm and Rural Life Poll (Lasley 1999) has not asked questions that get as directly to views of the structure of agriculture, but they appear to hold similar views. In the 1999 poll, over half of farmers

concluded that the bulk of the farmers who oppose livestock expansion do so because of a strong concern that it would erode the status of family farming in the state.

The increasing production of hogs through contract relationships, following that of poultry (Morrison, 1998), is becoming central to socioeconomic, health, and environmental concerns regarding CAFOs. One reason that agribusiness firms contract with producers, or contract with intermediary firms who subsequently contract with producers is to gain greater control over the production process (Welsh 1997), moving decision-making from the farm level to higher levels in the vertical system. Rarely do poultry growers own the birds they raise, and the pork industry appears to be moving in that direction (Morrison, 1998). Among major livestock production systems, cow-calf operations remain the most staunchly controlled at the farm-level.

In Kentucky the fulcrum of recent agricultural policy debate has been a proposed joint liability provision within state regulations. This provision would make corporations that retain ownership of animals (integrators) and the growers who raise animals jointly liable for resultant environmental damages or production facility closings. Burmeister (2000) suggests this joint liability provision reflects a societal attempt to control the social risk of changes in animal agriculture.

Research on the social/community impacts of different forms of contracting versus spot markets is scarce. For example, there has been no systematic research on animal producers who lose production contracts. Certain contract livestock producers are organizing to gain more regulatory and contractual protection (Hamilton, 1995; Roth, 1995). Whether such protection will generate substantial socioeconomic and environmental benefits to these producers and their communities may be measurable in the future.

Contract farming, while seen by some livestock growers as their best available option for remaining in farming, is problematic for others. In 1999, 70 percent of Iowa farmers favored greater regulation of contracts in farming (Lasley, 1999). Other alternatives should be encouraged—particularly ones that are compatible with changes in consumer demands and with environmental quality. A growing proportion of consumers are concerned about sub-therapeutic use of hormones (as discussed in the Executive Summary), humane treatment of animals[6], and the health and well being of producers. The socioeconomic, health, and ecological benefits of sustainable methods of agricultural production, including pork production as described by Ikerd (1998), are gaining recognition. For example, Lyson and Barham's (1998) found evidence of greater sustainability of middle-size, family farm operations over large-scale, corporate farms. They used measures of profitability, decreased

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responding strongly agreed with the statement, "There is too much economic power concentrated in a flew large agribusiness firms, and when the "agreed" category, the proportion agreeing rises to nine in ten farmers. The percentage agreeing with the statement, "If things continue as they are now, in a few years farmers will be treated like employees on their own farms," was only modestly lower (46% and 85%, respectively).

[6] In an unpublished survey conducted by the Animal Industry Foundation in 1989 nearly 80 percent of those polled supported current practices of farm animal treatment (cited in Ohlendorf, Jenkins and Tomazic, forthcoming). But in the same survey two-thirds of those polled were in favor of increased regulation of production practices. Following up on this data, Ohlendorf et al. asked more than 2,700 consumers whether they agreed or disagreed with the statement "I would be willing to pay more for meat if it meant more humane treatment of farm animals." While 23 percent of those surveyed were undecided, one-half of all respondents agreed with the statement. There is no significant variation in agreement with this more pro-animal attitude across economic classes. This is at odds with the prevalent notion that consumer concern is much more different socioeconomic groups would be willing to pay.

resource use, and stable or increasing farm numbers in a community (See also Lasley, Hoiberg, & Bultena, 1993).

Thus, it is not necessary that CAFOs be the only, or necessarily even the dominant, way in which livestock will be fattened or milk or eggs will be produced in the future. Perhaps, it would be more correct to say that public policy—the collective will—could lead animal production either toward a continued growth of CAFOs at the expense of all others, or toward more pluralistic production regimes—which would undoubtedly include CAFOs without their necessarily being the dominant form of production.

7.2.3 Market Restructuring

While the structure of livestock production is changing rapidly, so is the marketing structure. The most important shift in livestock marketing is the expansion of vertical integration and the potential of an alternative form, vertical coordination (see Tweeten & Flora, 2001, for a thorough treatment of this topic). Vertical *integration* occurs through a *supply chain,* while vertical *coordination* operates through a *value chain.* Table 7.2 indicates important differences between the two.

Supply chains are oriented by myriad decisions of many producers—usually in an atomized market or perhaps nudged by government supply-limitation (until 1996) or supply-encouraging (after 1996) incentives. Value chains respond to the demands of consumers. Increasingly supply chains have come to be vertically integrated, reducing the freedom of the farmer to make on-farm and marketing decisions. The poultry grower neither owns the birds, nor makes decisions about how they will be produced. S/he is required to market

Table 7.2 Comparison of Features of Supply Chains and Value Chains

SUPPLY CHAIN	VALUE CHAIN
Producer oriented	Consumer oriented
Supply driven	Demand driven
Emphasis on reducing costs	Emphasis on creating value
Focus on volume	Focus on quality
Undifferentiated commodity	Differentiated products
Source (of commodity) is anonymous	Product may be traced to specific producer (identity preservation)
Many independent decisions (particularly at producer level)	Few cascading decisions
Open entry of new producers	Entry of new producers is limited
Susceptible to vertical integration	Requires at least some vertical coordination

Table adapted from C. Flora, et al. (1999), who adapted it from Cook (1997) and Hughes (1998).

to the integrator, and cannot be certain of the price s/he will receive for growing the birds. This lack of market discovery is also becoming more common in hog and cattle marketing, as processors, who increasingly buy directly from the farmer, are not required to publicly disclose the prices they pay. In the poultry business, contracts are from year to year. If they are terminated, there may be little likelihood of finding another integrator to sell to, since generally only one or two poultry integrators is active in a particular locale (Bjerklie, 1995; Griffith 1993; Heffernan & Jenkins, 1983).

The processor has typically controlled vertical integration, but increasingly retailers[7] are gaining the balance of power in the food supply chain. Vertical coordination has the potential to be more collaborative and decentralized. Value chains are more amenable to a team approach, since flexibility in production is essential if production is to respond to changing consumer preferences. Farmers have little power under vertical integration, while they may band together to control or share control through vertical coordination. Vertical coordination does not ensure farmer power, but it is certainly amenable to farmers collectively exercising that power—if they are willing to key their production on diverse consumer desires and to devise ways to shorten the supply chain (Tweeten & Flora, 2001). Of course, state and local governments and institutions of higher learning can be helpful with information and linkages, particularly if they address previous constraints to promoting sustainable agricultural practices (Lacy, 1993).

At present, hog production—though much more concentrated than it was a decade ago--is much less concentrated than is pork processing.[8] Heffernan, Hendrickson, and Gronski, (1999) estimated that in 1998, the fifty largest producers controlled about one half of all marketed hogs, and only one of the top five producers had substantial presence in Iowa. Most states where corporate hog production predominates are states where large numbers of hogs were not produced previously or where farms are smaller and less prosperous. One author argues convincingly that broiler integrators chose to focus on the South precisely because small farmers often were underemployed and desperately needed additional income (Bjerklie, 1995). The degree to which integrated hog contracts in Iowa and other parts of the Midwest are favorable or unfavorable to farmers will depend on the overall vitality of the rural parts of those states. When growers have or perceive they have few other options, they are more likely to sign unfavorable contracts.

7.2.4 Impetus for Alternatives in Production, Processing, and Marketing

One means of preserving identity is through shortening the value chain—bringing producer and consumer closer together. Shortening the value chain is important for the development of alternative production systems. Reducing the steps between producer and consumer contributes to quality control. Trust can be substituted for costly inspection systems, and immediate and direct feedback will occur when quality is inadequate. In addition, quality may be redefined in unconventional ways. For instance, the consumer may be willing to forego cuts of meat in uniform and predictable sizes if s/he has assurance that sub-therapeutic hormones are not used, or that animals are treated humanely.

If this sounds like each farm family would do its own direct marketing (which often falls to the female partner in a producer family), it does not have to be. A critical piece is socializing the transaction costs involved in identity preservation and quality assurance. This can be accomplished through devising novel collaborative means of marketing and identity preservation that are satisfying to the consumer, but which do not require each producer family to make its own marketing links or to individually organize its own system of quality assurance. Different kinds of producer-controlled or -influenced value chains, such as marketing cooperatives, joint ventures between corporate entities and producer associations, producer-consumer coalitions such as Community Supported Agriculture (CSA) groups

[7] Between 1997 and 2000, the market share of the top five food retailers operating in the U.S. rose from 24% to 42%. Hendrickson, et al. (2001) argue that increasingly, market power is shifting from processors to food retail chains.

[8] In 1998, the top four pork processors marketed 57% of all hogs in the country. The following year, according to the New York Times, the top six firms processed 75% of all market hogs. In 2001, the largest processor, Smithfield, bought IBP, which had ranked second in 1998 (Heffernan et al., 1999; 16)

(Cone & Myhre, 2000), or local marketing cooperatives (Ziegenhorn, 1998) can lift the marketing burden from the shoulders of individual producers.

Only with involvement of market (private for-profit firms, including family firms and farms), state (governments at different levels), and civil society (not-for-profit organizations, such as producer organizations, certification entities, etc.) can vertically coordinated value chains compete with vertical integration and supply chains. We often forget just how large a role various levels of government play in subsidizing commodity supply chains and vertically integrated firms within our food system (see North Central Regional Center for Rural Development, 1999: 6-20, for a detailed discussion of the "incentives" used to encourage Seaboard Corporation to build a pork packing plant in Guyman, OK).

Which of these factors may influence the future of the livestock industry in Iowa and how might they relate to odor regulation? Clearly, Iowa's competitive advantage in grain and livestock production is an important element. Some argue that Iowa may regain market share in cattle and hog feeding that has recently been lost to the Great Plains (cattle) and to North Carolina (hogs), given Iowa's competitive advantage in cheap grains. The 1996 Freedom to Farm Act, by dismantling price supports and the supply management system, encouraged production of corn and soybeans (Harl, 2001). Currently, low grain prices do not encourage farmers to shift to higher value crops, since loan deficiency payments increase as market prices decline. This has encouraged CAFO production in the Midwest where grain is cheap. It has also favored CAFO production over diversified family farming. CAFOs can purchase feed grains at market prices lower than costs to family farmers of feeding their own grain, since market prices have recently been below cost of production for family farmers.

Another important factor is the differential contribution of environmental protection to the cost of production by region. All other things equal, the more dense the human population, the greater the cost of environmental protection to the producer—if there are mechanisms for internalizing those costs, rather than their being paid by the society at large. The initial moratorium on building new hog CAFOs in North Carolina and its recent extension suggest that hog odors and water contamination can provide the political impetus for internalizing these costs in heavily populated areas. Should the Environmental Protection Agency increase the amount of land that is required for disposal of manure because of concern about excess phosphorus application, production in Iowa would be favored over North Carolina, although Iowa might be disadvantaged vis-a-vis the Great Plains.

Policy makers' consideration of alternative means of regulating odors must take into account which farmers are disadvantaged by the regulations and what those regulations may mean in terms of encouraging certain desired futures for rural Iowa—and Iowa in general.

In this section we have provided evidence that industrialized commodity production and corporate controlled supply chains are not the only alternative. Regulation of odors and other airborne products should take into account various options, and encourage those that are more socially desirable.

7.3 Changes in Property Values

In the next section we consider changes in animal agriculture as they relate to the final form of community capital - financial capital. Several studies examine effects of nearness to a CAFO on real estate values. Abeles-Allison and Conner (1990) chose eight Michigan hog CAFOs and then examined residential sales within a five-mile square block centered on each CAFO. They analyzed data on 288 sales between 1986 and 1989. For every thousand hogs added in the five-mile area, they found an average drop in sale price of $430 per property. The depression of sale price was much greater when the residential property was less than 1.6 miles away from the respective hog farm. Using state-wide data, they found, for the first half of 1989, that odor complaints were 50 times more likely to be lodged against any particular hog CAFO of over 500 head than against smaller hog operations.

Palmquist, Roka, and Vukina (1997) studied residential property values close to hog CAFOs in North Carolina. Controlling for other characteristics of the property, they examined patterns of non-farm home sales prices (n=237) over an 18-month period in 1992 and 1993. They found that nearness to large hog CAFOs and the amount of nearby manure jointly acted as a significant depressor of sales prices of up to nine percent, depending on the number of hogs and their distance from the house. Phillips et al. (1999), suggest that odors cannot be separated from other local effects from CAFOs that could also depress sales prices. These could be noise, dust from trucks, or a general decline in the natural beauty of the area.

Hamed, Johnson, and Miller (1999) found that an average vacant parcel within three miles of a CAFO in Missouri lost about 6.6% in value, but if a parcel with a house on it was within 1/10 mile of the CAFO, it lost 88% of its value!

Finally, Taff, et al., (1996) examined housing sale prices in two counties of southwestern Minnesota. The measures used to indicate feedlot proximity included distance, total animal units within a defined distance, and whether the home was downwind from any feedlots. Feedlot proximity was associated with *higher* sales prices. The authors suggest that perhaps workers desired to live close to their work.

7.4 Impact on Social and Health Services

While not examined here, studies of broader changes taking place in agriculture link housing, public services, natural resources and land use, and historical and cultural resources to the changing structure of animal agriculture. These changes are also reflected in the examples related specifically to animal agriculture.

NCRCRD research in Oklahoma (1999) found that housing rental rates increased nearly 85 percent over seven years in the county where production and meatpacking expansion occurred, compared to a 61 percent increase in comparison counties. At the same time, the influx of new workers resulted in a 47 percent decrease in housing availability. The combined result is overcrowding and shared housing situations, or a commute to neighboring counties with available and more affordable housing. These commuting costs add to the household costs of workers. Of course, the housing industry, among others, benefits from such growth.

The same research notes important implications for local educational systems. While total school enrollment increased 12 percent, resulting in construction of a new elementary school, there was a 125 percent increase in the number of bilingual or limited English speaking students. Despite an 81 percent increase in the county school budget between 1990 and 1997, both dropout rates and student/teacher ratios increased. Community costs due to increased demand on services, such as court costs from increased criminal and civil cases; law enforcement costs, and applications for public assistance and food stamps were also noted.

Other research points to additional costs of large-scale animal production to community resources: impacts on tourism and recreation due to livestock odors (McMillan & Schulman 2001); deterioration of bridges and hard surface roads (Constance 2000); and significant changes in rural landscapes and the number and condition of farm sites (Bowen 2000).

In 1990, the minority population accounted for about 4 of every 100 Iowans (4.1%); by 2000, that figure had increased to more than 7 of every 100 (7.4%). The minority population grew by 103,000 while the (white non-Hispanic) majority increased by 47,000 during the 1990s. For the first time, a significant portion of that growth in minority population occurred outside Iowa's metropolitan areas. These new Iowans were mainly attracted by jobs in meatpacking, and secondarily, in plant nurseries, construction, and certain low-wage service jobs.

While we were unable to find data on the extent of employment of immigrants and other minority groups in CAFOs in Iowa, it is clearer that industrial agriculture (packing plants in particular) employs a growing number of new residents who are culturally different from the long-term residents of rural Iowa (see Grey, 1997, 1998). Turnover in packing plant employment and hence in population (rather than presence of minority groups, per se) contributes to a number of social problems and a need for more local services, but it also brings in young, hard working, entrepreneurial (especially immigrant) families, shoring up the base of population pyramids and offering a larger working age population for years to come in certain communities that before the 1990s were aging steadily. Whether long-term residents and leadership in these communities will see these new residents as a gift or as a threat is still to be seen.

7.5 Concluding Remarks

Generally, Iowa's rural areas have had more difficulty holding their populations than have urban sections of the state. With more deaths than births[9] and greater out- than in-migration, some of these counties have had problems sustaining their populations. The encouraging news is that the only decade in the 20th century during which Iowa had more people enter than leave was the 1990s; net in-migration totaled about 50,000. Even in that case, however, 43 of Iowa's 99 counties had more residents leaving than entering in the 1990s. Although there were some major exceptions, rural counties were more frequently listed among those 43 with net out-migration than were urban counties.

If this migration turnaround is to be sustained, additional attention needs to be given to issues of quality of life. That means that the physical environment, the quality and diversity of services (particularly health and educational services), and employment opportunities will need attention. If jobs are not available, it is unlikely that others will move to the area unless

[9] In 2000, 48 counties had more deaths than births (called net natural decrease) and most were rural; only a few had an incorporated place with at least 10,000 residents. Due to the out-migration of younger people from many rural counties and the tendency of older residents to age in place, a declining proportion of the population is in the reproductive age groups. Hence, in recent decades, the number of counties experiencing net natural decrease has gradually grown. In Iowa as a whole, however, about 100,000 more births than deaths occurred throughout the 1990s.

natural and social amenities provide the premium that would attract them. Some Iowa counties have physical environments (e.g., rivers, lakes, open space) that attract residents. At present, many of the people moving to such locales already live in the state. And natural amenities are likely to be magnets only for the somewhat more affluent. On the other hand, urban areas are much more likely to benefit from employment-related moves. But then the characteristics of jobs also are related to the residents that they attract; that is, the types of employment that become available dictate at least in part the characteristics of those who will move to an area. To attract residents to a rural area, then, requires the perception that such a move may raise the quality of life through improved employment opportunities, and increasingly, access to amenities—both natural and social.

Demographic changes have a number of implications for CAFOs and vice versa. While in the 50 years between 1940 and 1990, the farm population dropped at about twice the rate that the (non-farm) country population increased, many residences remain close to livestock operations (Lasley, 1998). Since it appears that for the past decade the gap between farm population decline and the country population may be closing, hog, and perhaps poultry, CAFO odors will be a growing issue among rural dwellers.

A related issue that is suggested by the demographic patterns is the potential conflict that CAFOs and industrial agriculture generate between employment and amenities. Those communities where odors and health problems from CAFOs remain or become an issue may have a more difficult time attracting or holding population that would otherwise come because of rural communities being "a good place to live and raise a family." The amenity scale may go down not simply because of these problems themselves, but because the odor and health issues generates conflict, reducing social capital and the ability of the community to act collectively to enhance local social and natural amenities. Resolving these questions through alternative livestock production methods may make it easier for communities to encourage employment *and* to increase amenities. For example, a 2001 informal survey of 13 Iowa State University Extension livestock specialists (Honeyman et al., 2001) documented the existence of at least 2100 hoop structures in Iowa, which, with appropriate management practices, can be more environmentally friendly than CAFOs. In conjunction with appropriate marketing structures, other ecological production regimes, such as use of A-frames and rotational pasturing may be feasible.

A final set of demographic issues surrounds the health risks and desires for justice expressed by elderly rural residents residing near CAFOs. They often express concern about being at risk for respiratory problems, as well as concern that antibiotic treatments may fail them when needed. The siting of CAFOs near the rural elderly, who are less likely to move in the later years, seems inequitable to some, as does the decline in quality of life for those who have worked productively for many years, including in support of others in their communities.

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Chapter 8. Exposure Limits Related to Air Quality and Risk Assessment

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Abstract
This chapter reviews the literature with regards to health hazard substances emitted from CAFOs. Furthermore, we reviewed the risk assessment process of pertinent federal agencies in regards to hazardous emissions from CAFOs. Occupational health hazards, for those working in CAFOs, have been long recognized. Research documents that current recommended or legal occupational exposure levels are not sufficient to protect workers. Although the research on occupational exposures of CAFO workers documents the hazardous nature of CAFO emissions at concentrations found inside buildings, the concentration of these hazardous substances are much lower in the ambient air of the community surrounding CAFOs. As occupational exposure limit recommendations are not highly relevant to the community, specific exposure standards are needed to help protect community residents as well as workers.

Regarding community standards, the risk assessment processes of the Environmental Protection Agency (EPA), and the Agency for Toxic Substances and Disease Registry (ATSDR) are the most relevant agencies in making recommendations for limits to community exposures. The EPA estimates levels safe for a lifetime exposure and ATSDR list levels for acute, intermediate or chronic levels. For ammonia, the EPA list 144 ppb for lifetime exposures and the ATSDR list 500 ppb for acute and 300 ppb for chronic exposure. For hydrogen sulfide, the EPA lists 0.7 ppb for lifetime exposure, and ATSDR lists 70 ppb for acute and 30 ppb for intermediate exposures. Considering these recommendations made by EPA and ATSDR, concentration recommendations, recommendations made in surrounding states, and consideration of the possible additive or synergistic effect of mixed exposures, hydrogen sulfide, ammonia, and odors should be regulated. The levels that should be considered are as follows: hydrogen sulfide, one hour time-weighted average of no more than 15 ppb at the residence or 70 ppb at the property line; ammonia, one hour time-weighted average of no more than 150 ppb at the residence and no more than 70 ppb at the property line; odors should be no more than a 1:7 dilution at the residence and no more than 1:15 at the property line.

8.1 Introduction

This chapter will review the scientific literature on exposure limits for occupational and ambient conditions, relative to CAFOs. Also, the relevance of existing standards for the health protection of workers and community residents will be discussed. Furthermore, the circumstances of mixed exposures will be reviewed. Finally, a risk assessment and recommendations for appropriate standards will be discussed.

8.2 Existing Occupational Health Exposure Limits or Recommendations

In the US, there are four sources of recommendations in regards to occupational exposure limits. These include the American Conference of Governmental Industrial Hygienists (ACGIH, 2001 TLV's for Chemical Substances and Physical Agents & Biological Exposures Indices), the American Industrial Hygiene Association (AIHA, AIHA Press, Fairfax VA, 2001), The National Institute for Occupational Safety and Health (NIOSH, Pocket Guide to Chemical Hazards, 1997) and the Occupational Safety and Health Administration (OSHA, Code of Federal Regulations, Chapter 29). The first two organizations (AIHC and ACGIH) are private professional organizations. The third, (NIOSH) is a governmental educational and research organization. OSHA is the only regulatory and enforcement agency of these four. AIHC, ACGIH, and NIOSH, only recommend worker-exposure standards, but develop science-based recommendations, and not subject to the stakeholder pressures from the administration, industry, and labor, and other constituents groups, as is OSHA. The terminology for exposure limits is different for each of these organizations. AIHC, ACGIH, and NIOSH issue, respectively, Emergency Response Planning Guidelines/Workplace Environmental Exposure Level Guides (ERPPGs/WEELs), Threshold Limit Values (TLV) and Time Weighted Average Exposure Limits (TWA). OSHA issues Permissible Exposure Limits (PEL's).

The primary exposures of occupational concern in CAFOs include ammonia (NH3), hydrogen sulfide (H2S), carbon monoxide (CO), carbon dioxide (CO2), particulate matter (PM), bioaerosols, and endotoxin. However, none of the bodies mentioned above have specified limits for bioaerosols or endotoxin. Table 1 lists the indoor concentration levels for each of these bodies for the agents specified.

Table 1. Maximum Concentration Levels Listed for Occupational Health

	NH3	H2S	CO	CO2	Total Particulate Matter	Respirable Dust	Bioaerosols	Endotoxin
AIHA	25 ppm	0.1 ppm	200 ppm	Not listed	Not listed	Not Listed	Not listed	Not listed
ACGIH	25 ppm	10 ppm	25 ppm	5000 ppm	4 mg/m^3 (Grain dust) 10 mg/m^3 (Nuisance dust)	3 mg/m^3 (Grain dust)	Not listed	Not listed
NIOSH	25 ppm	10 ppm	35 ppm	5000 ppm	4 mg/m^3 (Grain dust)	Not Listed	Not Listed	Not Listed
OSHA	50 ppm	20 ppm	50 ppm	5000 ppm	10 mg/m^3 (Grain dust) 15 mg/m^3 (Nuisance dust)	5 mg/m^3	Not Listed	Not Listed

8.2.1 Occupational Dose Response Data For Humans

Exposure-response studies in workers have included an assessment of the response to the amount of time exposed, for particulate matter (PM), endotoxin, NH3, and H2S. Endotoxin and PM concentrations have had the strongest and most consistent relationships to respiratory symptoms and decrements in pulmonary function tests (PFT) (Donham et al., 1989; Donham et al., 1995;

Reynolds et al., 1996). A significant relationship was seen between microbial concentration and bronchitic symptoms (cough and phlegm) (Donham et al., 1989). A weaker relationship of bioaerosol concentrations to tightness of chest and febrile syndromes (flu-like illness with fever) was found (Donham et al., 1989). There was no relationship of bioaerosol to pulmonary function changes. Ammonia did show some relationship to decreased baseline pulmonary function in four different studies (Donham et al., 1989; Donham et al., 1995; Reynolds et al., 1996; Cumro et al., 2001, in press). In one of the studies, the levels of microbes showed a significant dose response relationship to symptoms of hyper-reactive airways (Donham et al., 1989).

A study in The Netherlands (Heederik et al., 1991) suggested that both endotoxin and Gram-negative bacteria were related to reductions in pulmonary function, as measured by forced expiratory volume in one second, (FEV_1) and forced vital capacity (FVC). Also, significant relationships were shown between symptoms of bronchitis, or Organic Dust Toxic Syndrome (ODTS) to endotoxin or Gram-negative bacteria exposure.

8.2.2 Occupational Exposure Limit Studies

There is little scientific doubt that disease symptoms and work-shift declines in pulmonary function are related to several components of the mixture of particulate matter, bioaerosols and gases found inside CAFOs. These components include dust, endotoxin, hydrogen sulfide, and ammonia. However, the most important question in this regard is how much exposure creates a health hazard? Knowledge of the appropriate exposure limits is extremely important for controlling the work environment.

Data, which suggest the exposure limits in relation to adverse pulmonary function and symptoms, are found in four dose-response studies (Donham et al., 1989; Donham et al., 1995; Reynolds et al., 1996; and Cumro et al., 2001, in press). The first is a study of workers on 54 pig farms in Sweden (Donham et al., 1989). Several significant correlations were found between respiratory symptoms and PFT and PM, endotoxin, ammonia, and carbon dioxide. Significant relationships were seen between health measures and environmental measures taken at stationary locations in the buildings. More recent data analyses from US studies have corroborated the previous exposure limit study (Donham et al., 1995; Reynolds et al., 1996). A longitudinal study of 208 swine farmers (randomly selected from a stratified sample of all pig producers in Iowa) resulted in consistent evidence of a dose-response relationship of exposure to the dust and gases found in pig buildings and respiratory symptoms, and decreased pulmonary function. Furthermore, multiple regression analyses of the data, provided results consistent with the exposure limits previously mentioned in the Swedish study.

The fourth dose-response study mentioned previously was conducted in the poultry industry with 149 poultry production workers (Donham, Leistikow et al., 1989). This study analyzed respiratory symptoms and PFT associated with exposures to PM, endotoxin, and ammonia. Regression analysis was used to determine maximum exposure levels that predicted more than 5% pulmonary function decline with adverse health responses (Donham et al., 2000).

These four studies reviewed above are in close agreement in regard to concentration levels of contaminants that represent hazardous exposures to workers in either swine or poultry CAFOs. Table 2, lists the recommended maximum levels from the scientific literature of environmental exposures based on the four studies reviewed above. Recommended maximum exposures for swine health are also listed for comparisons sake. The worker health and swine health levels are reasonably close, indicating that protecting the health of workers also can provide benefits for health and production of swine.

Table 2. Human and pig exposure thresholds for various bioaerosol components found in swine buildings. Exposure to concentrations of contaminants in excess of values given are associated with a higher proportion of ill-health in workers, and with disease, or lower production parameters in pigs. Taken from' Donham et al., (1989);[1] Donham et al., (1995);[1] Donham (1991);[2] Reynolds et al., (1996);[2] Donham et al., (2000).[1]

Bioaerosol component	Human health[1]	Swine health[2]
Total dust mg/m^3	2.4	3.7
Respirable dust mg/m^3	0.23	0.23
Endotoxin EU/m^3	100	150
Carbon dioxide (ppm)	1,540	1,540
Ammonia (ppm)	7.0	11.0
Total microbes cfu/m^3	$4.3x10^5$	$4.3x10^5$

8.3 Ambient Exposure Limits

The EPA currently has national ambient standards for particulate matter (PM), sulfur dioxide, oxides of nitrogen, ozone, lead, and carbon monoxide. Generally, speaking, these emissions are not relevant to CAFOs, except PM. However, tracing the source of PM is difficult at this time, (although there are at least two possible methods for use, LIDAR and chemical analysis of signature molecules attached to particulates.) The U.S. EPA has promulgated standards in response to the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA: 40 C.F.R. Part 302). Under this act, regulated hazardous substances (CERCLA 40 CFR Parts 355 and 370) emitted from a point source may not exceed 100 lb/day for ammonia, hydrogen sulfide and a number of other pollutants. Ammonia emissions from four CAFOs studied swine production systems in Iowa (Zahn et al., 2001a; Zahn et al., 2001b) were recently reported to violate release, reporting requirements for NH_3 under the U.S. EPA Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, U.S. EPA, 2002). The range for ammonia emissions from these swine production sites ranged from 224 lbs NH3/day to 813.9 lbs. NH_3 day^{-1} (nder warm weather conditions). The observed aggregate emission rates for swine production facilities evaluated in this latter study were reported to exceed CERCLA reporting requirements for NH_3 by 55% to 88%. There is an additional federal act that may be relevant to CAFOs. The Emergency Planning and Community Right-to-Know Act (EPCRA) section 329(4), defines a facility to include stationary structures on a single site, or on contiguous or adjacent sites owned or operated by the same person. Under this definition, the aggregated emission rate of registered hazardous substances from all swine production facility point sources is subject to release reporting requirements. As part of the release reporting requirements, the polluting facility must develop an EPA-approved emission abatement plan to curb emissions from the emitting point sources.

Generally, there has been little published information available indicating that CAFO emissions exceed present federal Clean Air Act regulations. The EPA's 1998-draft strategy for addressing CAFO issues has not included health or air quality provisions. However, the pending revision of the

Clean Air Act will likely address these issues. There has been a USDA Air Quality Task Force working on the issues. This Task Force issued a report dated July 19, 2000, titled, "Air Quality Research and Technology Transfer White Paper and Recommendations for Concentrated Animal Feeding Operations" (http://www.nhq.nrcs.usda.gov/faca/Archives/2000/Policy/CAFO.htm) Currently, EPA has commissioned the National Academy of Science to conduct a study evaluating the human health impacts of emissions from CAFOs. This 14-month study has just begun. Generally, this issue has been left up to the individual states. The states of Colorado, and Missouri have odor regulations, based on the sentometry at 7:1, and 5.4:1 dilutions respectively at the property boundary (Colorado Department of Public Health and Environment Air Pollution Control Divisions Odor Concentration Measurement, Scentometry Test Policy for Housed Commercial Swine Feeding Operations, Colorado Department of Public Health and Environment, Denver, Colorado, January 25, 2001, www.Cdphe.state.co.us/ap/hog_policies:html, and Missouri. Pollution Control Agency, Feedlot Air Quality Summary: Data Collection, Enforcement, and Program Development, March 1999). Minnesota and California have state H2S regulations, which are 50 ppb, for not more than one-half hour, and not more than two occurrences per year, and 30 ppb for not more than one-half hour for not more than two occurrences in a 5-day period (property line of the emitter). There is also a provisional 60 ppb human risk value (HRV) limit for not more than one hour (at the receptor) (MN Pollution Control Agency). Current regulations and recommendations in regards to federal and state agencies are reviewed in more detail in chapter 9.0.

8.3.1 EPA Risk Assessments

Risk assessment has been defined as "the characterization of the potential adverse health effects of human exposures to environmental hazards" (NRC, 1983). In a risk assessment, the extent to which a group of people has been or may be exposed to a certain chemical is determined, and the extent of exposure is then considered in relation to the kind and degree of hazard posed by the chemical, thereby permitting an estimate to be made of the present or potential health risk to the population exposed. Regarding the primary inhalation exposures in CAFOs, the U.S. EPA has completed risk assessment evaluations for ammonia and hydrogen sulfide. Both are limited to chronic (24 hour/day lifetime exposure) health hazard assessments for noncarcinogenic effects. The completed risk assessments represent a consensus opinion of EPA health scientists representing various Program Offices and the Office of Research and Development.

The consensus process includes interpreting the available scientific literature applicable to health effects of a risk agent, and using established methodologies to develop values for inhalation reference concentration. With regard to multiple exposure routes, the U.S. EPA's position is that the potential for health effects manifested via one route of exposure (i.e. dermal or respiratory) is relevant to considerations of any other route of exposure, unless convincing evidence exists to the contrary. In other words, if there is convincing data of a health hazard to a specific substance from respiratory exposure, then the EPA assumes dermal exposures are also hazardous, unless there is convincing evidence to the contrary. As more epidemiological, animal studies, and new scientific information becomes available for CAFO-related exposures, EPA intends to review it, as appropriate, and develop more complete risk assessments.

Chronic Health Hazard Assessments for Noncarcinogenic Effects
The inhalation reference concentrations (RfC) and chronic health hazard summaries for NH_3 and $H2_S$ are listed in Tables 3 and 4, respectively. The No-Observed-Adverse-Effect Level (NOAEL) is

the highest exposure level at which there are no statistically or biologically significant increases in the frequency or severity of adverse effect between the exposed population and its appropriate control. Although some effects may be produced at this level, they are not considered adverse, nor precursors to adverse effects. The Lowest-Observed-Adverse-Effect Level (LOAEL) is the lowest exposure level at which there are statistically or biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group. The Reference Concentration (RfC) is an estimate (with uncertainty spanning perhaps an order of magnitude) of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime. The RfC is derived from a NOAEL, LOAEL, or benchmark concentration, with uncertainty factors (UF) generally applied to reflect limitations of the data used. The RfC is generally used in EPA's noncancer health risk assessments.

For ammonia, an uncertainty factor of 10 is used to allow for the protection of sensitive individuals. Additionally, a factor of 3 is used to account for several database deficiencies including the lack of chronic data and the lack of reproductive and developmental toxicology studies. Based on these factors, EPA sets the limit for lifetime exposures to ammonia at 144 ppb. For hydrogen sulfide, the uncertainty factor of 1000 reflects a factor of 10 to protect sensitive individuals, a factor of 10 to adjust from sub-chronic studies to a chronic study, and a factor of 10 for both interspecies conversion and data base deficiencies. Based on these factors, EPA sets the limit for lifetime exposures to ammonia hydrogen sulfide at 0.7 ppb.

Table 3. Environmental Protection Agency Reference Concentrations for Chronic Inhalation Exposure to Ammonia

Critical Effect	Exposures*	UF	RfC
Lack of evidence of deceased pulmonary function or changes in subjective syptomatology {Occupational Study}	NOAEL (HEC): 2.3 mg/cu.m	30	0.1 mg/cu.m (144 ppb)

*The NOAEL is based on an 8-hour TWA occupational exposure.
(HEC) is the adjusted human equivalent dose.
[1]USEPA, last revised 1991.

Table 4. Environmental Protection Agency Reference Concentrations for Chronic Inhalation Exposure to Hydrogen Sulfide[1]

Critical Effect	Exposures	UF	RfC
Inflammation of the nasal mucosa {Mouse Sub-chronic Inhalation Study}	NOAEL (HEC)[2]: 1.01 mg/cu.m (0.73 ppm)	1000	0.001 mg/cu.m (0.7 ppb)

[1]USEPA, last revised 1995.
[2] NOEL (HEC) = No Effect Exposure Level, Human equivalent dose.
*See appendix A for references for these hazard assessment recommendations.

8.3.2 ATSDR Recommended Limits

Ambient exposure guidelines are also provided in the reviews produced by the Agency for Toxic Substances and Disease Registry, the federal agency charged with evaluating possible health risks from chemicals released at waste sites where the general public may be exposed. In their Toxicological Profiles, this Agency has reviewed the extensive literature concerning health effects of ammonia (ATSDR, 1990, reviewing more than 350 articles to assess possible human health effects of this compound) and hydrogen sulfide (ATSDR, 1999, reviewing about 470 articles), probably the two major contaminants of concern from animal operations as far as is currently known. While the ATSDR guidelines are not generally applicable and enforceable ambient standards, their focus is on protection of the public, including sensitive individuals, and thus they are relevant to the situation under consideration here.

The product of ATSDR reviews are generally information and guidelines related to public exposures near waste sites. They state:

> During the development of toxicological profiles, Minimal Risk Levels (MRLs) are derived when reliable and sufficient data exist to identify the target organ(s) of effect or the most sensitive health effect(s) for a specific duration for a given route of exposure. An MRL is an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse noncancer health effects over a specified duration of exposure. MRLs are based on noncancer health effects only and are not based on a consideration of cancer effects. These substance-specific estimates, which are intended to serve as screening levels, are used by ATSDR health assessors to identify contaminants and potential health effects that may be of concern at hazardous waste sites. It is important to note that MRLs are not intended to define clean up or action levels. MRLs are derived for hazardous substances using the no-observed-adverse-effect level/uncertainty factor approach. They are below levels that might cause adverse health effects in the people most sensitive to such chemical-induced effects. MRLs are derived for acute (1-14 days), intermediate (15-364 days), and chronic (365 days and longer) duration and for the oral and inhalation routes of exposure (ATSDR, 1999, page A-1).

Thus the MRLs are designed to protect sensitive populations. However, as the MRLs are derived for individual contaminants; mixtures of chemicals such as CAFO emissions are potentially more hazardous, but difficult to assess from a health effect standpoint. The situation of mixed exposures is discussed in section 8.3.2.

The ATSDR report on ammonia (ATSDR, 1990) establishes a short-term (less than or equal to 14 days) MRL of 500 ppb for inhalation. A long-term (defined as greater than 365 days in this earlier Toxicological Profile) MRL of 300 ppb at the receptor is established. It appears that the 300ppb MRL would be the appropriate comparison value for public exposures beyond the property limits of a CAFO (table 6). Using the occupational 8-hour time-weighted-average recommendation for workplace exposure (nearly 100 times this value), while appropriate for the healthy adult working population, would be inappropriate for continuous exposure of the general public which includes sensitive populations, including infants, the elderly, and those with pre-existing conditions. Observed or estimated CAFO concentrations of 250 ppb are at times uncomfortably close to the long-term ammonia MRL (Subramanian, et al., 1996; Reynolds et al., 1997).

Hydrogen sulfide is another major contaminant of concern near confinements. The July 1999 ATSDR "Toxicological Profile for Hydrogen Sulfide" (ATSDR, 1999) derives "an acute inhalation MRL of 70 ppb" and "an intermediate MRL of 30 ppb" (p. 139); these would correspond to the 1-14 day and 15-364 day durations of exposure, respectively, and would be appropriate for those living adjacent to CAFOs (table 6). These MRLs are public health exposure guidelines, much lower than the occupational limit of 10,000 ppb.

Generally, there is limited peer-reviewed published literature on community assessments of hydrogen sulfide in the vicinity of CAFOs. There is a non-peer reviewed article by Jacobsen, (1997), for both ammonia and hydrogen sulfide. There have been several studies by the USEPA of continuous monitoring around CAFOs. One of these is a 1999 study in Northern Missouri is available from the EPA (Secrest, C.D., "Field Measurements of Air Pollutants Near Swine Confinement Animal Feeding Operations Using UV DOAS and FTIR," Office of Regulatory Enforcement, Air Enforcement Division USEPA, MS 2242A, 1200 Pennsylvania Avenue, NW, Washington, DC, 20460). Furthermore, the Minnesota Pollution Control Agency has conducted monitoring of numerous CAFOs. Their report on "Feedlot Air Quality Summary Data Collection, Enforcement, and Program Development (March 1999)," can be seen at http://www.pca.state.mn.us/hot/feedlots.html. These reports indicate that observed off-site concentrations near CAFOs at times may approach or exceed these ATSDR recommended limits.

There is a very important point to note, that there is variation in concentrations that can be measured, depending on atmospheric conditions. Stable atmospheres, particularly in the evening are condusive to build up of contaminants in the vicinity of CAFOs. Therefore, it is important that measurement periods take these predicable variations into account. In other words, just measuring during the evening as well as the day is important to obtaining an accurate assessment of actual exposure at the receptor.

Table 6. Agency for Toxic Substances and Disease Registry Minimum Risk Levels (MRL) for Ammonia and Hydrogen Sulfide[1]

Substance	Acute Exposure (1-14 days)	Intermediate Exposure (15-364 days)	Chronic Exposure (365 days and longer)
Ammonia	500 ppb	(None listed)	300 ppb
Hydrogen Sulfide	70 ppb	30 ppb	(None listed)

8.4 Relevance of legal or other recommended limits to occupational and ambient air quality associated with CAFOs.

Regarding OSHA occupational health exposure regulations, the PEL's listed for the hazardous substances found in CAFOs is not highly relevant. The reasons are as follows:

1. The scientific literature document s that endotoxin is one of the most hazardous substances to CAFO workers (Rylander, Jacobs, Organic Dusts, Exposure, Effects, and Prevention. CRC Press, 1994). However OSHA has no PEL standard for endotoxin
2. The OSHA PEL for PM is based on a non-biologically active (nuisance) dust. However, the PM inside CAFOs is highly biologically active, (high concentrations of microbes, endotoxins, and glucan) and is hazardous at much lower levels than in the 10 mg/m^3 published PEL (Donham and Scallon, 1986, and Donham and Reynolds 1996).
3. The PEL's are written assuming exposures to one toxic substance. CAFOs result in complex mixed exposures, which lowers the allowable exposure to each individual component of the mixture (Donham and Scallon, 1986, and Donham and Reynolds, 1996). Therefore, the OSHA or other recommended limits are not highly relevant. Although NIOSH, ACGIH, and AIHA are more stringent than OSHA, they are still much higher than research findings indicate they should be to offer, adequate worker protection in mixed exposure situations like CAFOs.

8.4.1 Mixed Exposures – Occupational

OSHA has established a Permissible Exposure Limit (PEL) for nuisance dust of 15 mg/m^3. The OSHA TWA's for respirable particles and ammonia are, respectively, 5 mg/m^3 and 50 ppm. Threshold limit values (TLV's) established by the American Conference of Governmental Industrial Hygienists (ACGIH) include 10 mg/m^3 for nuisance dusts, 4 mg/m^3 for grain dusts, 3 mg/m^3 for respirable dusts, and 25 ppm for ammonia (Table 1, NIOSH, 1994; ACGIH, 1994). However, several published research manuscripts (Donham KJ, et al., 1995, Reynolds S, 1996, Donham KJ et al., 2000) document that these limits are too high for CAFOs where a mixture of biologically active agents can combine to produce respiratory and systemic effects at much lower levels (Cumro et al., in press).

Multiple agents, multiple etiologies, and the potential for multiple interactions make thorough evaluation of health effects from CAFO emitants a very difficult task. The assignment of unquestionable causality to a single agent for a single adverse health effect or dysfunction in confinement workers is unlikely at best. The 2001ACGIH publication for threshold limit values for chemical substances and physical agents states that when mixed exposures are present, and unless other data indicate differently, the effects should be considered additive. For example where C1, C2, and Cn are measured concentrations of hazardous substances, and T1, T2, and Tn, are their respective TLV's, then the relationship to determine if the level is under legal TLV's, the relationship is defined mathematically as follows:C1/T1 + C2 /T2 + Cn/Tn = < 1.

There may be instances when the effects of two substances are greater than additive, defined as a synergistic interaction. If synergy is present then mixed exposures are even more hazardous than if the effects were merely additive. Such a relationship between NH3 and PM in CAFOs, has been defined by Cumro and Donham, (in press). Data were analyzed from an exposure- response study of 149 poultry CAFO worker. Analysis of this data-set revealed prominent dose-response relationships between increasing PM, NH3, and endotoxin concentrations with corresponding cross-shift declines in worker lung function. Specific threshold concentrations were defined including total dust, 2.4 mg/m^3; respirable dust, 0.16 mg/m^3; total endotoxin, 100 EU/m^3; respirable endotoxin, 0.35 EU/m^3; and NH3, 12 ppm (Donham and Cumro, et al., 2000). As health effects to poultry workers from exposure to both dust and ammonia were less than half the published ACGIH TLV's, investigations were undertaken to study possible interactions between these substances. The results demonstrated that when workers are exposed to both PM and NH3, the adverse effect on pulmonary function is up to 156% greater than the individual effects of these gases (Cumro, et al., in press). Assuming a typical swine CAFO winter concentration of 10 ppm of NH3 and PM of 3.5 mg/m^3, and the TLV for grain dust of 4 mg/m^3, the correct relationship to determine if exposure limits are exceeded in this situation would be as follows: ([NH3]/TLV of NH3 + [PM]/TLV of PM) x 1.56. An example for a typical swine building would be as follows: (10 ppm / 25 ppm + 3.5 mg/m^3 / 4 mg/m^3) x 1.56 = 2.0. In other words, a typical building might exceed our recommended limit by two times. Synergy of simultaneous dust and ammonia exposures in a working environment raises the question of redefining exposure limits for organic dust and ammonia when workers are exposed simultaneously to these substances.

8.4.2 Mixed Exposures – The Community Setting

The EPA, in fact, treats mixed exposures in the community as additive (as ACGIH treats occupational exposures) unless there is information to indicate otherwise (USEPA 600890066F Methods for Derivation Inhalation Reference Concentrations and Application of Inhalation Dosimetry http://www.epa.gov/cgi-bin/claritgw). Existing data are clear that the community exposure concentrations are much less than in the occupational setting. The logical public health question is do mixed exposures in the community setting also have additive or synergistic health effects? Fundamental toxicologic principles would predict there would be additive or synergistic health effects of mixed exposures in the community, (as there would be in the occupational setting) if the hazardous substances effect the same body tissues or organ(s).

In the case of CAFOs, ammonia and hydrogen sulfide both have direct effects on the respiratory system, although ATSDR also warns that hydrogen sulfide is also a broad-spectrum poison. Whether exposure indices for these two respiratory irritants with similar short or intermediate term MRLs can or should be added is not immediately clear but certainly possible. A potential method to establish limits in mixed exposures would be to ratio the concentrations to the appropriate MRLs, with a sum below 1 suggesting no respiratory threat (similar to ACGIH for occupational exposures

ACGIH TLV's for Chemical Substances and physical agents and biological exposures indices). Note that a sum above 1 would not necessarily imply overexposure unless known toxic limits were reached, but would be an "indeterminate human health hazard" under the ATSDR classification scheme.

ATSDR notes hydrogen sulfide is considered a broad-spectrum poison. This means that it can poison several different systems in the body. Thus, in addition to possibly additive or synergistic effects on the respiratory system in the presence of ammonia, there may also be additive effects with other components of CAFO emissions. These materials occur together, not only with each other, but also potentially with a variety of other contaminants in hog manure. For example, there are endotoxins and other bioaerosols along with various other substances that contribute to the observed effect. Unfortunately, available research does not allow quantitive assessment of the health effects of all the mixtures of all substances in CAFO emissions.

8.5 Summary of Occupational Exposure Limits as Recommended from the Scientific Literature
There can be no questions that exposure to emissions while working in CAFOs can be a health hazard. There are over 50 publications documenting the risks. There are now 4 dose – response studies that agree quite closely, regarding the lowest observed health effect levels are. As the concentration of the livestock industry continues, and becomes more specialized, we have greater worker exposure because more are working full-time inside the buildings, rather than spending time in other farming activities as in previous diversified farms. OSHA has left the industry alone for the most part, but with many more large operations (with more than 10 employees), this segment of the industry clearly falls under OSHA's mandate. However, as previously discussed, the current OSHA limits are not highly relevant to protection of CAFO workers. The following concentration, listed in table 7, are scientifically supportable guidelines for occupational exposures, and are listed adjacent to current OSHA standards. (Donham et al., 1989; Donham et al., 1995; Reynolds et al., 1996; and Donham et al., 2000.

Table 7. Summary of Scientific Recommendations of Maximum Exposure Concentrations for Occupational Health Considerations of Swine and Poultry CAFO Workers.

	Human Health[1]	Current OSHA
Total dust mg/m^3	2.5	15
Respirable dust mg/m^3	0.23	5
Endotoxin EU/m^3	100	NA
Carbon dioxide (ppm)	1,540	5000
Ammonia (ppm)	7.0	50

8.6 Summary of Ambiant Exposure Limits as Recommended from Federal Agencies and Regional State Regulations

There has been no published literature on dose – response relationships of CAFO emissions and life quality or chronic health effects among community residents. However, several states have adopted emission standards based on the weight of evidence regarding individual chemical exposures (see chapter 9.0) Furthermore, ATSDR and the EPA have made recommendations based on hazard assessment evaluations. Also, consideration for mixed exposures should lower levels set for individual exposures. The following concentrations could be supported for CAFOs, based on the relevant information reviewed above.

H2S:
15 ppb at the residence for a one-hour average measure and 70 ppb at the property line. No more that seven exceedences would be allowed, per calendar year (with notice to the residents and DNR).
NH3:
150 ppb at the residence and 500 ppb at the property line for a one-hour average measure. There should be no more than 7 exceedences (with notice to residence and DNR), per calendar year.
Odor:
Odor would not exceed 1:7 dilutions at the receptor, or public use area, No more than 14 exceedences (with notice), per calendar year. An additional consideration could be given to a 1:15 dilution at the property line. Monitoring would be conducted via scentometry.

8.7 Justification for Recommendations of Exposure Limits

The concentrations listed in section 8.6 above, are based on a combination of data gained from relevant regulations in other states, and recommendations from made by several public health related agencies, including the World Health Organization, the US Environmental Protection Agency (EPA), and the US Agency for Toxic Substances and Disease Registry (ATSDR). The basis for the regulations promulgated in other states are reported in Chapter 9. The justification for levels recommenced by the EPA and ATSDR are described below.

The ATSDR minimal risk levels (MRL's) were developed in response to the mandate for the agency to list hazardous substances commonly found at listed facilities, the toxicologic profiles of these substances and to ascertain significant human exposures. That mandate is specified in The Comprehensive Environmental Response, Compensations and Liability Act (CERCLA) [42 U.S.C. 9604 et seq.], as amended by the Superfund Amendments and Reauthorization Act (SARA) [Pub. L.99-499]. The ATSDR has adopted a method similar to the EPA to determine the MRL's for respiratory exposures, or Reference concentrations RfC's. These levels are estimates of the daily human exposure to a hazardous substance that is not likely to cause adverse (non-cancerous) health effects, over a specified exposure period (acute – 1-14 days, intermediate – 15-364 days, and chronic, greater than 365 days). MRL's are derived when the ATSDR determines there is sufficient data to determine specific and sensitive health effects for a specific duration. Consistent with principles of public health, the MRL's are set to protect sensitive individuals, and that there is a safety factor built in as they are set below levels that might cause adverse health effects. The public health protection principle is also used by utilizing uncertainty factors (UF) when less than complete data are available. The MRL's undergo a rigorous review process, both internal, and external to the agency, are peer reviewed, and are submitted for public comment. As of June 1, 2001, 286 MRLs had been determined, including hydrogen sulfide, and ammonia. The MRL's can be found on the ATSDR website at www.atsdr.cdc.gov/mrls.html. The ATSDR also publishes, "Toxicologic Profiles," which reviews the literature on the toxicology and public health significance, and justifications for MRL's determined for each of the substances for which an MRL is determined (www.atsdr.cdc.gov/toxpro2.html).

As mentioned previously, the more detailed methods ATSDR uses for determination of MRL's are very similar to the EPA methods for setting their risk levels, which are called reference dose concentration guidelines, (RfD's, for oral exposures, or RfC's for respiratory exposures). The EPA method is described in detail here to help explain how EPA and the ATSDR develop their exposure guidelines. The EPA Risk Assessment Method, are described in detail in the 416 page document 600890066F, entitled "Methods for Derivation Inhalation Reference Concentrations and Application Dosimetry" (www.epa.gov/cgi-bin/claritgw). The EPA has a long history of evaluating scientific information and in developing benchmark values for regulatory action to protect the public from adverse health effects. The National Academy of Sciences (NAS) has been charged with the evaluation of risk assessment processes performed by federal agencies to assure that regulations are based on best judgment and analysis of available scientific knowledge (Risk Assessment in the Federal Government: Managing the Process, NAS, 1983, and NAS Report on Sciences and Judgment in Risk Assessment, National Research Council, 1994). The NAS recommends that risk assessment should be separate from policy aspects of risk management to help assure recommendations for protection on the publics' health are not compromised by the political process. Furthermore, NAS defines risk assessment as "characterization of the potential adverse human health effects of exposures to environmental hazards and consists of the following steps:

1. Hazard identification: to determine the cause-health effect linkages of suspected hazardous substances;
2. Dose-response assessment: the estimation of the relation between the magnitude of exposures and the occurrence of the health effects in question;
3. Exposure assessment: determination of the extent of human exposure;
4. Risk characterization: determination of the nature and magnitude of human exposure, along with attendant uncertainty.

The EPA adopted its reference dose concentration guidelines (RfD's) and analogous guidelines for respiratory exposures (RfC's) based on the NAS guidelines, but the method is more rigorously defined and includes guidance for uncertainty factors (UF's) to help guide extrapolation in instances such as applying animal data to human exposures, and incomplete data (Barnes and Dourson, 1988). The process is a quantitative approach to interpretation of toxicology and epidemiologic data to determine a dose-response estimate, followed by a comparison to exposure estimate to analyze risk characterization. The RfC is defined as: An estimate (with uncertainty spanning perhaps an order of magnitude) of a continuous inhalation exposure to the human population (including sensitive subgroups) that is likely to be without appreciable health risks during a lifetime (24 hours per day for 70 years).

The steps to calculating an RfC are as follows:

1. Determination of a no-observed-adverse-effect-level (NOAEL), which is the highest dose where no health effects are seen, or threshold level (Klaassen, 1986).
2. Determination of a human equivalent concentration (HEC) of the NOAEL, if the latter is based on animal data.
3. Determination of uncertainty factors (UF) that may include necessary extrapolations from:
    a. average healthy to sensitive humans
    b. animal to human data
    c. sub-chronic to chronic data
    d. lowest effect level to NOAEL
    e. incomplete to complete data base
4. Determination of any necessary modifying factors (MF) not addressed by the UF's, such as adjustments for low sample sizes, or poor exposure characterization.

The RfC determination could be defined by the following notation:

$$RfC = NOAEL[HEC] / (UF \times MF)$$

Usually a subjective confidence level is assigned to the RfC, based on the quality and completeness of the data and the extent of UF's used. These are issued not to disregard those with medium or low confidence levels, but to indicate that the values may change as more information becomes available. RfC's with a high confidence level may not expect to change in the future, relative to those with a low confidence level. The EPA's Integrated Risk Information System (IRIS), lists all the RfC's established, and discusses the UF's used in their determination.

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APPENDIX A
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APPENDIX B

References for ATSDR Exposure Limit Recommendations

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Chapter 9. Relevant Laws, Regulations and Decisions

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9.1 Introduction

Air emissions and odor from Concentrated Animal Feeding Operations (CAFOs) has become the subject of discussion and government action in a number of states. Sometimes the news media, environmental disasters or the large number of constituent contacts, push legislative changes or administrative action on how CAFOs are sited and managed. Occasionally a new scientific study on the topic changes the tone of the debate.

This chapter documents action taken to regulate and manage CAFO air emissions and odor in jurisdictions outside Iowa. It gives examples of the kind of control various levels of government have imposed on the management of these facilities in an attempt to balance economic advantage with public health and welfare.

9.2 Distinguishing the Consequences of Air Emissions: Nuisance or Health Effects

Air emissions from animal waste storage systems, buildings and land application of animal waste contain a number of gasses and particles that include hydrogen sulfide, ammonia, methane, particulate matter and bioaerosols. Hydrogen sulfide is an example of a substance that is both a direct toxic health risk and odorous. Furthermore, this odorous substance, as well as other odorous substances, may also cause adverse health symptoms, from an indirect toxic mechanism via interactions with the central and peripheral nervous system. [See Chapter 6]

Odorous compounds also decrease the quality of life of neighbors. Hydrogen sulfide, for instance, causes corrosion of metals and damage to plants. When a state or local government acts to reduce air emissions, it might describe this as intervening to protect public health or to enhance the quality of life or the government may say nothing about what motivates its action.

The Minnesota Pollution Control Agency (MPCA) places limits on hydrogen sulfide emissions at CAFOs. Minnesota is similar to other states that have recently intervened to reduce air emissions from a type of activity that was formerly thought to be only a local issue.

> In the past, the Minnesota Pollution Control Agency (MPCA) has viewed odors as a natural result of animal production that could best be addressed through good land use planning, with the primary responsibility for land use planning at the local level of government (see Minn.R. 7020.0100).[1]

Just as states have struggled to determine whether state or local regulation is appropriate, they have not found it simple to distinguish the rationale for regulation. It is not simple to say which air emission is a health issue and which a nuisance. Rather there is a continuum from life threatening acute health effects, through acute effects that are not life threatening, to chronic effects that appear after longer exposure, to annoyance smell. Odor or a constituent part of the air emission can have a range of effects. Consider this example. Two Minnesota agencies, the MPCA and the Minnesota Department of Health (MDH) have established values to limit the various adverse effects from hydrogen sulfide emissions from CAFOs. The MDH explains why both agencies have acted.

> The MPCA standard for hydrogen sulfide will protect against symptoms of headache, nausea, and maintain a quality of life for Minnesotans. The MDH acute Health Risk Value (HRV) will protect against respiratory effects by ensuring that hydrogen sulfide is included in

> hazard indexes calculations where hydrogen sulfide is one of many chemicals emitted to air potentially having respiratory impacts.[2]

9.3 Minnesota GEIS on Animal Agriculture

The state of Minnesota has recently brought the scientific and public policy community together to advise state government on how to proceed on several CAFO issues. This extensive process resulted in a Generic Environmental Impact Statement (GEIS) for animal agriculture that was presented at a public hearing on December 10, 2001. Because the technical work papers for the GEIS date from the first half of 2001, this chapter relied heavily on three GEIS Technical Work Papers -- Air Quality and Odor, Role of Government, and Human Health. One finding of the GEIS Technical Work Papers is that air quality has not been the driving force behind government action.

> Existing laws and programs have mostly emerged out of a long-standing concern over surface water impacts, which, while valid, have meant that air, groundwater and other emerging issues are not adequately factored into government decision-making.[3]

Government's slow involvement with air emissions and odor is substantiated by data presented in 1998 survey on Animal Confinement Policy designed by a National Task Force of 15 Extension Specialists representing all regions of the nation. For only 13 of 48 states did those surveyed answer yes to the following question: Are odor standards imposed as a matter of state government policy or court decisions in your state?[4] More than twice as many states regulate discharge to surface water. However, some states like Missouri, which recorded an answer of 'no' to the preceding question, now have odor standards.

9.3.1 Iowa

We will discuss Iowa CAFOs at length below. However it is informative to see the view of our state from our neighbor to the north. The Minnesota GEIS Technical Work Paper on Air Quality and Odor Impacts investigated regulatory programs in a number of states including Iowa. According to the researchers,

> {d}espite having an estimated 3,000 large animal feeding operations that have the capacity for more than 1,000 animal units and receiving many odor complaints from neighbors, the State of Iowa has essentially no program in place for addressing odors or air emissions from animal agriculture facilities.[5]

This strong language points to the inadequacy of present Iowa rules and regulations. However, two members of this research team, one from the University of Iowa and one from Iowa State University were part of a study group which put the following language into Iowa Administrative Rules of the former Air Quality Commission in 1977 when an odorous substance standard was defined as:

> The emission of an odorous substance from an odorous substance source shall constitute a violation of these rules if the emission is of such frequency, duration, quality, and intensity to be harmful or injurious to human health and welfare, or as to unreasonably interfere with the comfortable use and enjoyment of life and property or so as to constitute a nuisance as defined in sections 657.1 and 657.2 of the Code.

The former paragraph was part of the state Administrative Code for only a short time. The rule was rescinded in 1984 when the term "odorous substances" was also deleted from the Administrative Rules.*[6]

While the Minnesota GEIS was our starting point to seek out governmental action on air and odor, we also talked to government officials and staff of non-governmental organizations in pork producing states. We have omitted setback requirements, one type of policy that nearly every state has adopted to reduce the effects of air emissions and odor from CAFOs. We will look at regulations for three constituents of air emissions, hydrogen sulfide, ammonia and odor. We will look at local government activities and we will look at the results of action in the courts. In each category, rather than an exhaustive look we will choose a few jurisdictions to serve as exemplars for the kind of government intervention to control CAFO air emissions.

9.4 Hydrogen Sulfide Standards

The Agency for Toxic Substances and Disease Registry (ATSDR) of the US Department of Health and Human Services, lists 24 states with regulations or guidelines for hydrogen sulfide. In this section we will look at three states with regulations especially relevant to CAFOs.[7]

9.4.1 Minnesota

Minnesota is interesting because it addresses air quality issues from animal agriculture in several ways. First, the MPCA maintains a two-component Ambient Air Quality Standard for hydrogen sulfide. 50 parts per billion (ppb) is not to be exceed for ½ hour twice per year and 30 ppb is the ½ hour average not to be exceed more than 2 times in any 5 consecutive days.[8] These are emitter property line standards for animal feeding operations over 1000 animal units in size.

Second, subsequent to establishing these standards, another Minnesota agency, the MDH, proposed a draft acute Inhalation Health Risk Value (HRV) for hydrogen sulfide of 60 ppb as a 1-hour average and a draft subchronic (3-month average) HRV of 7 ppb.[9] These standards are evaluated at the receptor rather than the emitter's property line. The MDH plans to adopt both HRVs without public hearing in the very near future. In addition, the state addresses air quality issues from CAFOs by requiring each facility with a capacity of 1000 animal units to include an Air Emission Plan in its water quality permit.

The many different air emission limits in a single state for only one constituent makes clear the difficulty in describing laws, regulations and decisions relating to CAFO air emissions and odor. Many authors do not make a distinction between air emissions and odor or between health effects and nuisance. In this chapter we will deal with both odor and air emissions and will try to record accurately which a particular law, regulation or decision is being referred to.

9.4.2 Nebraska

The Nebraska Department of Environmental Quality (NDEQ) implemented an ambient air quality health-based standard of 100 ppb for total reduced sulfur (TRS) in September 1997 (Revised January 1999). The impetus for this action was industrial emissions in Dakota City, Nebraska and not

* Rule 400--4.5 was published as an adopted and filed rule on 6/16/77. On 6/22/83 the rule was transferred to WAWM as rule 900--23.5. An amendment published 9/12/84 changed the catchwords from "Odorous substances" to "Anaerobic lagoons"; rescinded subrules 23.5(1) and 23.5(2) and renumbered subrules 23.5(3) and 23.5(4) as 23.5(1) and 23.5(2), respectively. Rule 23.5 is still in the Iowa Administrative Code.

CAFOs. However, since the revision in 1999, the standard applies to CAFOs. NDEQ prepared an extensive background research paper that focused on low-level exposure to hydrogen sulfide and TRS through inhalation.[10]

> Total Reduced Sulfur (TRS) consists of the total sulfur from the following compounds: hydrogen sulfide (H2S), methyl mercaptan (CH3SH), dimethyl sulfide ((CH3)2S), and dimethyl disulfide (CH3SSCH3) (87). These TRS compounds occur naturally in the environment. H2S makes up the greatest proportion of TRS.[11]

As part of their research paper, NDEQ surveyed the 49 other states and found 27 states that had standards for H2S or TRS. These states based standards on a variety of issues including, odor or nuisance, welfare effects, and health effects. Standards varied considerably to as low as 0.7 ppb for a yearly average (New York) and 5 ppb averaged over 24 hours (Pennsylvania). Many of the standards were based on nuisance including Minnesota's 30 ppb and 50 ppb standards. The lowest standard that was reported to be health based was a 10 ppb 8-hour 10 ppb standard (Illinois).[12]

The authors of the background research paper recommended Nebraska's present health standard of 100 ppb, averaged over 30 minutes. The authors also recommended a much lower 30-day standard of 10 ppb or 5 ppb (depending on average humidity level in the air) to protect against other effects of sulfur compounds. While the state adopted the 100 ppb health-based standard, it has not yet adopted the lower, welfare standard for TRS.

9.4.3 California

The California Ambient Air Quality Standard (CAAQS) for hydrogen sulfide of 30 ppb for one hour was adopted in 1969 and reviewed but not changed in 1980 and 1984. A year 2000 review states the purpose of the standard was to decrease odor annoyance from industry rather than CAFOs. However, the review notes that significant adverse health effects might occur at levels of exposure below the CAAQS.[13] More recently, the California Office of Environmental Health Hazard has adopted a chronic reference inhalation standard of H2S at 8 ppb.

The three states give three motivations for a sulfurous emission standard. Nebraska has a health-based standard of 100 ppb, averaged over 30 minutes at the receptor. Minnesota's health standards vary from an acute 60 ppb 30-minute standard to a sub-chronic 7 ppb, 3-month standard measured at the receptor or nearer the source of the emission. The state also has two nuisance-based property line standards of 50 ppb and 30 ppb averaged over 30-minutes. California's much older 30 ppb for one hour standard is based on nuisance but the state is looking into whether significant adverse health effects might occur at similar levels. The California Office of Environmental Health Hazard has adopted a chronic reference inhalation standard of hydrogen sulfide of 8 ppb.[14] Standards for the three states as well as for federal agencies and the World Health Organization are contained in Table 1.

9.5 Ammonia Measurements

The ATSDR has published a toxicological profile for ammonia, which contains a list of regulations and advisories from various states. The list of acceptable ambient air concentration levels for Ammonia based on 1988 information, contains standards in place in 11 states.[15] (See Appendix and Table 2.) The ammonia standards for the three following jurisdictions are contained in Table 2 at the end of the chapter.

9.5.1 Minnesota

Besides establishing a Health Risk Value (HRV) for hydrogen sulfide, the Minnesota Department of Health has filed a draft HRV for ammonia as well. This HRV, like the other, will receive final approval in the next few months. Both HRVs are scientifically measured standards that protect the public from adverse health effects. The point of measurement would often be at the receptor but it is possible that the agency will take measurements at the property line as well. A brochure prepared for the public explains that being below the HRV does not necessarily take care of odor problems.

> However, keeping emissions at or below the HRV does not necessarily eliminate odors from the agricultural animal operation and may not eliminate health effects from odors.[16]

9.5.2 Netherlands

While The Netherlands is very different from any US state, it is similar in terms of livestock production.[17] [18] The numbers of livestock animals in The Netherlands are 14 million hogs, 108 million chickens, 4.2 million cattle and 1.4 million sheep. On four times as much land mass Iowa has 15 million hogs, 37.8 million chickens, 3.7 million cattle and 270,000 sheep. In The Netherlands, ammonia emissions from agriculture are responsible for 42% of the acidification attributable to domestic sources. Policies set forth in the 1990s aim to reduce by 70% the ammonia emissions compared to the 1980 benchmark. The European Union is working on a directive to reduce ammonia emissions that is being modeled after the Dutch regulation.

Features of the Dutch policy that regulates phosphate, nitrogen and ammonia include a mandated reduction of the pig population by 10%, compulsory minerals accounting and reporting by all intensive livestock farms, a total ban on application of manure in autumn and winter, a ban on application on frozen ground, compulsory use of injection manure application, compulsory covering of manure storage tanks and reservoirs, strict requirements for ammonia emissions from intensive livestock facilities including a requirement that all new livestock housing meet the strict ALARA (as low as reasonably achievable) standards. All facilities will be required to meet ALARA by 2008. Farmers must have adequate manure storage facilities to store their manure from September through February.

Methane and nitrous oxide arising from CAFOs are greenhouse gases that may contribute to global climate change. In The Netherlands, agricultural activities account for an estimated 45% of the total methane emissions and 35 to 40% of the N2O emissions. Measures being taken to reduce ammonia emissions should allow The Netherlands to reduce greenhouse gas emissions to levels mandated by the Kyoto Protocol. Thus far, the U.S. has not taken any steps to control emissions of greenhouse gases from livestock production.

9.5.3 Missouri

The Missouri Department of Natural Resources has been monitoring a Premium Standard Farm (PSF) concentrated animal feeding operation in northern Missouri since the beginning of 2000.[19] Hydrogen sulfide and ammonia concentrations are monitored on a 24-hour basis. The monitor has recorded high concentrations of both ammonia and hydrogen sulfide. Missouri has an ambient air standard for hydrogen sulfide and an ambient acceptable level (AAL) for ammonia of 144 ppb. The Department added a second monitor at another PSF facility late in Fall 2001.[20]

The Missouri Department of Health, the Missouri Department of Natural Resources, the US EPA and the US Agency for Toxic Substances & Disease Registry cooperated in a health evaluation near some of PSF's facilities in the fall of 2001. The health evaluation concentrated on the two pollutants hydrogen sulfide and ammonia. A health evaluation is to determine if a more full-scale health study is needed. The results of the evaluation have not been released.[20] In a recent Consent Decree with the US EPA and a citizen's group, PSF has agreed to continue to monitor for a number of air emissions including ammonia.[21]

9.6 Odor legislation

States can regulate air emissions without referring to a specific chemical constituent. The 1998 national survey of animal confinement policies, referred to above, found thirteen states where odor standards were imposed as a matter of state policy or court decisions.[4] We look at three states that require that odor from CAFOs be held below a threshold. In Missouri and Colorado, the threshold is based on a dilution standard. The Colorado dilution standard of 7:1 means that an air sample collected at the emitter property line is diluted with seven volumes of fresh air. If odor can still be detected by using an olfactometer and panel of smellers, there is a violation. (See description in section 3.4) North Carolina has an idiosyncratic method of deciding on an odor violation, which is covered below. Table 3 summarizes the information in this section.

9.6.1 Missouri

On January 1, 2002, all very large CAFOs in Missouri must have an odor control plan in place describing measures to be used to control odor emissions (10 CSR 10-3.090, Code of State Regulations). All Class 1A CAFOs, those having more than 7000 animal units must comply (twenty-one facilities in total). This air quality specific program approach dates from 1999. A number of farm organizations went to court alleging the state lacked authority to regulate emissions. The state has prevailed at the County Circuit Court and the Missouri Court of Appeals.[22] At this time only 1 of Missouri's 21 Class 1A CAFOs has an approved odor control plan. All the others have not been approved. In general, the disapproved odor control plans lacked specific odor control and reduction strategies. Nearly all of the CAFO owners have appealed the state's disapproval of their plans.[20]

Missouri uses a dilution threshold as a standard. An instrument called a scentometer is used in the field at a dilution threshold of 5.4:1 to determine if a significant odor is present. If odor is detected, an air sample is taken and sent for further evaluation by an olfactometry panel. If the panel detects the odor at a dilution threshold of 7:1 or greater, or at an intensity greater than a reference standard of 225 ppm of n-butanol, then a violation has occurred.

9.6.2 North Carolina

North Carolina CAFOs with liquid waste systems are required to first meet a number of best management practices for things such as dead animal disposal. Besides these management practices requirements, certain swine operations fall under the regulation's complaint response and odor management program. Compliance with the rules depend on facility size and distance from an occupied residence, business, school, hospital, church, outdoor recreation facility, park, historic property, or childcare center. According to materials gathered in the Minnesota GEIS process, the North Carolina complaint response system is quite involved and seems to be a time consuming process. It consists of the following steps.

Complaint response system

When a citizen complains to the state, they are asked to log complaints and weather conditions for 30 days on a form provided by the North Carolina Air Quality Division (NCDAQ). Once the logbook is returned to the state, the following formal investigation takes place.

a. An inspection is scheduled during weather conditions and time of day similar to when typical objectionable odor was reported
b. Evaluation is made at the location of the residence of the complainants
c. An "odor snapshot" is made by regional office investigator (one of 5 rankings)
d. The snapshot evaluation is reported to a regional supervisor
e. The regional office submits a recommendation to Division Director
f. The Division of Air Quality Director makes a final decision whether an objectionable odor exists.

If a determination of Objectionable Odor is made, the NCDAQ will require a Best Management Plan (BMP) of a facility—this is a revision of the original submitted plan. The BMP must be submitted within 90 days. Then ensues a process of plan approval and revision. If the BMP is found to be inadequate, the NCDAQ notifies the operation that it must submit a revised BMP under the same time schedules. Only then can the state order a facility to initiate any specific action.

> If the revised plan fails to adequately control, odors, the facility is required to install add-on control equipment and must submit a permit application for this installation within 90 days of receiving notification that their revised BMP was not adequate.[24]

Thus far only 25 facilities have had an Objectionable Odor determination. Each is currently in the process of providing a BMP to NCDAQ. As of early 2001, none had moved to the final step in the process, the installation of add-on control technology.

9.6.3 Colorado

Missouri and North Carolina demonstrate the significant length of time required to decide what action to take to abate odor. A different approach was followed by the state of Colorado where a referendum on the state ballot led to regulations addressing odorous gases and odor emissions from new and existing housed commercial swine feeding operations. The list of rules is extensive. They include, a cover requirement of anaerobic process wastewater lagoons, aerobic lagoon requirements, land waste application setback requirements, and mortality waste handling requirements. Housed commercial swine feeding operations must use technologies to minimize off-site odor emissions from all aspects of the operation (confinement structures, waste treatment facilities, manure management and land application), develop a comprehensive odor management plan and obtain an operating permit.

The regulation applies to all CAFOs that contain more than 800,000 pounds of live animal weight. Colorado CAFOs of this size must meet two ambient odor concentration standards, a dilution standard of 7:1 at the facility boundary and a dilution standard of 2:1 at any receptor (building, school or a municipal boundary). "The plans must also identify the odor monitoring that the facility intends to conduct in order to ensure compliance with the odor standards identified above."[25] While the requirements apply to more than 110 individual facilities, there are only eight owners of these facilities. The new regulations have reduced odor complaints substantially according to Phyllis Woodford of the Department of Public Health and the Environment.[26]

9.7 County and Local Action

Citizens who are not satisfied with state level governmental action to mitigate the effects of CAFOs have two other venues to protect their rights. They may pursue restrictions at the local government level or they can go to court in a private cause of action.

Rural counties have not generally adopted the zoning protection of more urban areas. However this seems to be changing.[27] In Missouri, reticence to zoning restrictions in a rural county was overcome by the arrival of large confinement operations. A resident of a township next to Premium Standard Farm's facility in Missouri, describes the area's change of view that caused it to adopt zoning.

> You've got to make plans and provide for the control of the situation before it occurs. Otherwise, by the time you realize you need zoning, its too late, and they've set the hook.[28]

While the eminent arrival of a CAFO might cause citizens to demand more protection from local government, local government is often prevented from playing a part in how CAFOs are regulated. State legislation to regulate animal agriculture has often been passed with the provision that local governments are prevented from intervening. Preemption of local action has been widely discussed in the literature.[27] Abdalla and Becker give several examples of the preemption of local government's abilities to deal with CAFOs. The authors explain resort to preemption laws by agricultural interests as simple economics.

> The economics of political influence clearly leads to a general preference for state level regulatory authority by organized interest groups. Monitoring and lobbying at the state level is much less expensive than providing these services at hundreds of local governmental units.[29]

9.7.1 Iowa

In Kuehl v. Cass County (1996) the Iowa Supreme Court held that all agriculture, including an animal feeding operation, is exempt from any county zoning. Before this decision, Humboldt County adopted four ordinances governing "large livestock confinement feeding facilities." While a district court upheld three of the ordinances, as a proper application of "home rule" authority, the Iowa Supreme Court struck down all the ordinances in their decision in Goodell v. Humboldt County, Iowa in 1998.[30] Presently an ordinance from Worth County is proceeding through the courts. This will test whether counties can regulate CAFOs based on public health. Whatever the outcome of the latest case, the Iowa preemption law has made county government reticent to try to regulate the location of CAFOs.

9.7.2 North Carolina

A website at the School of Public Health at the University of North Carolina contains reports on six county in the state that have passed ordinances regulating CAFOs. Ordinances required such things as operating permits, closure plans, graduated setback requirements and well testing. The Moore County ordinance for instances, required that confinement buildings and lagoons be set 2 miles from any golf course.[31] In 2001, two North Carolina court decisions struck down two county ordinances and put the remainder in jeopardy. The courts found that the General Assembly did not want to impose an unnecessary economic burden on hog production caused by each county passing its own set of rules. Chatham County has appealed the Court of Appeal's decision to the NC Supreme Court. A decision is expected in early 2002.[32]

9.7.3 South Carolina

The preemption strategy is not always successful as demonstrated by an attempt in South Carolina to push local government out of the regulatory picture.

> What started out as an attempt to adopt state laws that preempt counties from enacting measures to deal with confined animal feeding operations resulted in a measure that provides for considerable regulation of the activity and significant local involvement in the process.[33]

9.8 State Moratorium on Expansion

We have omitted one form of government action to this point. Moratoria have sometimes been adopted to give state officials time to review and update environmental regulations. In April 2001, Governor Jim Hodges of South Carolina imposed a 15-day moratorium on CAFO expansion to give environmental regulators more time to consider permit regulations.[34] Short-term limitations on any expansion of CAFOs also have taken place in Kentucky, Missouri and Arkansas.[30]

In one state the controversy over CAFO expansion has been so contentious that a continuous moratorium on large CAFO expansion has been put in place. North Carolina placed a two-year moratorium on the expansion of CAFOs with lagoon systems, when House Bill 515 passed in 1997. The moratorium has been extended twice and is now due to expire in September 2003.[35] Such reaction leads one to speculate whether stricter regulation earlier in the process may have better for the industry in North Carolina. The moratorium originally resulted from a number of lagoon breaks but also from the perception that the original regulation was too lax.

It is not even clear that weak legislation, preferred by nearly every industry, is in the best interest of producers in other parts of the nation.

> The Mo and Abdalla study found that overall, the stringency of environmental regulation did not appear to impact hog inventory growth." and "…the amount of staff devoted to animal waste management had an unexpected, but strongly positive relationship to hog inventory growth.[36]

9.9 Individual Legal Action

Local citizens have access to the courts where one can bring a private cause of action under nuisance. However, in many states, state government has attempted to blunt individual legal action through "right to farm" legislation. Hamilton explains the motivation for such statutes.

> Most lawyers and farmers have more than a passing familiarity with the legal concept upon which the laws were originally based—existing farm operations should not become nuisances due to the later development of non-agricultural uses in the surrounding area.[37]

Right to farm legislation has prevented neighbors and environmental groups from using individual nuisance action to require management changes or new locations for CAFOs. DeLind found that successful court action by neighbors against a Michigan swine confinement operation was the impetus for changing the law to give neighbors fewer rights by providing right to farm protection for what she calls hog hotels.[38]

> The official outcome, in other words, undermined both the original set of grassroots concerns and weakened the basis for further local-level action and representation.[39]

However, courts in several states including Iowa have ruled that right to farm laws give only limited protection from nuisance action. Richardson and Feitshans point to the Bormann case decided in

1998 by the Iowa Supreme Court as reducing the effectiveness of this protection for animal agriculture.[40] In that case, removing a citizen's right to nuisance action within a declared agricultural area was found to be a categorical taking of private property for public purposes without just compensation. Thus, the Iowa Legislature had exceeded its authority by authorizing the use of property in such a way as to infringe on the rights of others.[41]

In addition to limiting nuisance suits, another method of reducing the risk of animal agriculture operations from individual legal action has been the "fee shifting" provision. Hamilton gives the example of a 1995 Iowa law that assesses all costs and expenses of the defense side to the losing plaintiff in a nuisance action against a CAFO. Hamilton finds that "From a legal standpoint there are several reasons why this type of "soft" fee shifting is not a significant threat to most people who would file a nuisance challenge."[42]

In the last several months, two Iowa cases have set the stage for an expansion of the Bormann decision to land not in a designated agricultural area. In August 2001, an Iowa district court judge ruled for the first time that an Iowa law that protects CAFOs against nuisance suits is unconstitutional.[43] The decision allowed the Gacke case against Pork Xtra, L.L.C. to proceed in a Sioux County, Iowa court.[44] In January 2002, Pork Xtra was assessed $100,000 in damages.[45]

In December 2001, a court in Calhoun County Iowa made a similar determination that the defendants in Kleemeier v. Beazly Group, Inc and Pork Innovations could go forward. The judge found that the defendant's affirmative defense against nuisance action by neighbors had relied on an unconstitutional statute.[46]

Actual examples of substantial plaintiff victories from CAFOs under nuisance exist in other states. On September 9, 2001 Buckeye Egg Farm in Ohio, was hit with a judgment of $19.7 million for nuisance violations including fly infestations and odor. According to Feedstuffs, Buckeye, which has barn capacity of 11 million hens and 4 million chicks, is considering bankruptcy protection. Buckeye is the fifth largest commercial egg producer in the U.S.[47]

Nuisance need not be only a private court action. The Illinois Attorney General is presently prosecuting at least two swine CAFO operations under two counts of state law--air pollution and public nuisance--as well as under a third count of common law nuisance. There are two noteworthy dimensions of these cases. First, the Illinois AG cites considerable case law indicating that technical measures and depictions of odor and emissions are not required to prosecute air pollution and nuisance violations. Indeed, the nuisance statute itself was created, in part, to allow general citizens equitable access to courses of legal action without recourse to expensive technical measurements or scientific assessments.

In fact, a ruling by a Ninth Circuit Court Judge in one of these cases, affirmed the evidentiary possibility of neighbors proving their case by experiential testimony. Moreover, the state Illinois Environmental Protection Agency (IEPA) that follows-up on odor complaints, does not conduct technical assessments of odor or emissions. Instead, representatives from the IEPA will respond to neighbor odor complaints by making site visits and carefully documenting the presence or absence of odor with their own sensory judgments. Interestingly, the IEPA has issued warnings to prospective CAFO builders and operators (prior to construction of a CAFO) that just because they receive an approved state operating permit does not preclude action against them for violating the state's odor and nuisance standard. In fact, there are cases in Illinois where the Illinois Department of Agriculture will issue an approval for construction and operation while the IEPA will issue a simultaneous warning of potential air quality violations against neighbors.

It should be noted that the air pollution and nuisance statutes of Illinois are similar, if not identical to those found in other states. Hence the issue becomes not one of statutory authority of the Attorney General to prosecute such cases on behalf of neighbors, but more likely a political decision based upon a weighing of competing interests. More importantly, these statutes and associated case law recognize the evidentiary value of experiential assessments of odor by neighbors.

9.10 Non Regulatory Approaches

9.10.1 North Carolina

In July 2000, then North Carolina Attorney General Mike Easley signed an agreement that required Smithfield Foods, by far the state's largest pork producer, to pay $15 million to fund research and testing on better technologies to treat hog waste. Premium Standard Farms has committed $2.5 million for the same research questions under a similar agreement with Easley who is now Governor. In July 2002, a report is due.

> Smithfield-affiliated farmers then have three years to convert their facilities to the recommended technologies. In addition, the agreement requires Smithfield to pay $50 million for environmental improvements such as mapping and closing abandoned waste lagoons in the eastern half of the state.[48]

The technologies being examined must make substantial reduction in a number of emissions including ammonia, odor, disease vectors and airborne pathogens. Since this is early in the agreement, it is well to withhold judgment on whether or not Smithfield facilities will solve their odor and water emission problems.

9.10.2 Oklahoma

Oklahoma is another state in which a livestock producer has signed an agreement with the State Attorney General to change waste treatment systems at facilities.

> Seaboard Farms, the state's largest corporate hog producer, signed an agreement Tuesday (12/04/01) to spend about $3 million to better treat sewage and -–for the first time – control odors scientifically.[49]

Seaboard agreed to several measures including installing a manure treatment system similar to human waste treatment systems and agreed to share monitoring results with the state. The agreement allowed the company to open a second 25,000-sow facility similar to the one where the new treatment devices will be installed.

9.10.3 Missouri

A third case of a settlement of a court case resulted in changes by Premium Standard Farms (PFS) in Missouri. In 1997 Citizens Legal Environmental Action Network (CLEAN) filed suit against PFS for its waste handling procedures. In 1999, the US EPA joined the citizen suit. Settlement of the court case in Missouri has resulted in a civil penalty of $1,000,000. However PSF was allowed to receive credit for payment of $650,000 to the State of Missouri for a previous State Consent Decree. The PSF website describes the settlement as $350,000.[50] The payment of civil penalty was small in comparison to what PSF pledged to invest in upgrades to its facilities in Missouri, which has been reported as high as $50M.[51] [52] Although the PFS agreement is a legal settlement, we treat it in this section because of the requirement that new technologies be introduced and the joining of government entities in citizen suits.

9.11 Role of Research in Public Policy

Government does not always wait for research recommendations before taking action. We have referred often to Minnesota's Animal Agriculture GEIS process for which the state legislature committed $1.4 million beginning in 1998. While the process was underway, in the 2000 legislative session, the ability of the Minnesota Pollution Control Agency (MPCA) to enforce feedlot rules was compromised as follows.

> Lacking evidence of an immediate public health threat, the MPCA may not require operators of feedlots under 300 AU to spend more than $3000 without 75% cost-share, and feedlots under 500 AU cannot be required to spend more than $10,000 without cost-share of 75% of the upgrade, or $50,000, whichever is less.[53]

One reason to expect legislators not to wait for a research process to be complete is the same reason courts side with neighbors who have only their own experience and not exhaustive studies to impart. When legislators heard from their constituents who produced small numbers of livestock, that they did not want to be caught up in regulations for "the big guys" the legislature acted on what they felt was adequate evidence.

Our two colleges have been asked to bring science to regulatory decisions. Similarly in Minnesota and Nebraska, regulatory action was based on a survey of the scientific literature on health and welfare effects of pollutant emissions or on a survey of action in other states. However, both researchers and legislators assert that a scientific recommendation need not necessarily have all the answers before regulations can be promulgated. Regulating air quality from CAFOs can be made on the basis of precaution.

The Precautionary principle provides a guide to environmental policy that places the burden on the proponents of a potentially harmful activity to prove that their actions do not harm human health or the environment. The principle has been stated in many different places and contexts. The1998 Wingspread Statement, which is a consensus document produced by those attending a conference on the issue at the Wingspread Conference Center in Wisconsin, states in part:

> When an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically. In this context the proponent of an activity, rather than the public, should bear the burden of proof.[54]

The precautionary principle already forms the basis of at least a dozen treaties and international laws including the 1987 Montreal Protocol on Substances that Deplete the Ozone Layer, the 1992 UN Framework Convention on Climate Change and the 2000 Cartagena Protocol on Biosafety.[55] [56] [57]

The precautionary principle has often been applied to the introduction of new chemicals into the environment. Kriebel and Tickner assert that science informs policy in many ways. They find that, "A shift from reaction to precaution is entirely consistent with the core values of public health practice."[58]

A public health approach to marshalling evidence may be different from a strictly hard scientific approach according to Krimsky. He describes this difference by referring to Type I and Type II scientific errors. He maintains that minimizing false positives is the priority of the hard sciences while a public health perspective prefers to err on the side of overstating risks (prefer type II errors).[59]

Krimsky demonstrates the wide acceptance of some amount of precaution when he quotes an issue of the journal Chemistry & Industry, which states, "For one thing, it is crucial to avoid even inadvertently suggesting that the 'no evidence of harm' somehow equals 'evidence of no harm.'"[60] However, when considering implementation of precautionary measures, it is imperative that all consequences of such measures be thoroughly evaluated.

9.12 Conclusion

Governments have intervened to mediate between CAFOs and their neighbors in a number of ways. This chapter has sought to demonstrate the range of such intervention. States regulate hydrogen sulfide and TRS. Other states have limited the emissions of odor and ammonia. Local governments have sometimes been allowed to intervene to protect citizens against air emissions, but in most cases the state legislature has reserved this role for itself. Both, where states have acted and where they have not, the courts have intervened to give neighbors of CAFOs protection from air emissions and odor. This chapter is designed to demonstrate to the Iowa DNR and to Iowa government in general that there are examples of Laws, Regulations and Decisions designed to regulate air emissions and odor from CAFOs.

Appendix: Federal and International Air Quality Standards

Federal Standards

CAFO effects on water quality have been addressed by a Unified National Strategy developed by the USDA and USEPA in 1999. Air emission effects of CAFOs have not yet found the same level of federal attention. USDA formed an Agricultural Air Quality Task Force, which has been meeting and in July of 2000 drew up a white paper on research and technology transfer.[61]

Both the USEPA and the Agency for Toxic Substances and Disease Registry (ATSDR) have standards for both H2S and ammonia. These have been cited in Chapter 8.

Specifically for Hydrogen Sulfide, acute exposure guideline levels have been printed in the Federal Register for March 15, 2000.[62] There are several Proposed Acute Exposure Guideline Levels (AEGLs) applicable to the general population. AEGL-1 is set at 30 ppb for both a 10-minute and 30-minute exposure. AEGL-1 is designed to limit exposure to prevent "discomfort, irritation, or certain asymtomatic, non-sensory effects" which are not disabling. According to the web page of the American Petroleum Council. http://www.api.org/ehs/h2s/FalkeAbstract.htm AEGL-1 has not passed all the various reviews and is still considered a draft while two other AEGLs have been adopted.*

* AEGL values were developed for hydrogen sulfide by the National Advisory Committee on Acute Exposure Guideline Levels (NAC/AEGL Committee). These values were published in the Federal Register on March 15, 2000 (U.S. EPA, 2000) for public comment. After reviewing comments, the AEGL values were sent unchanged from the Federal Register Notice to the National Academies for review at their meeting on July 24-25, 2000. Following verbal

International Examples

Jurisdictions often base their standards on peer-reviewed literature and upon choices made by other jurisdictions. The State of Nebraska in adopting their standard of 100 ppb for TRS [see 9.4.2 for the relation between HS and TRS], cited data from the World Health Organization.

> The World Health Organization (WHO) reviewed information on health effects and recommended a daily (24-hour) value of 0.1 ppm H_2S. This value was based on the eye irritation effects at 10 ppm and a safety factor of 100. WHO noted that changes in heme synthesis were found at 1 ppm in pulp mill workers. Since the WHO made its recommendation in 1983, Bhambhani and Jappinen have conducted studies that indicate that eye irritation is not the most sensitive critical health effect.[63]

Table 10-1
Hydrogen Sulfide Standards for Various Jurisdictions

Jurisdiction	Type	Standard
Minnesota MPCA	Nuisance	30 ppb and 50 ppb
Minnesota Dept. of Health	Acute	60 ppb
Minnesota Dept. of Health	Sub-chronic	7 ppb
Nebraska Dept. of Health	Acute	100 ppb
California OEHH	Nuisance	30 ppb
California OEHH	Chronic	8 ppb
EPA – IRIS Chronic	Chronic	.7 ppb
EPA -- AEGL-1 (proposed)	Acute, non-disabling	30 ppb
ATSDR	Acute	70 ppb
ATSDR	Acute	30 ppb
WHO		100 ppb

comments at the National Academies' review, the AEGL-1 values are currently being re-evaluated by the NAC/AEGL Committee for endpoint and key study selection.

Table 10-2
Ammonia Standards for Various Jurisdictions

Jurisdiction	Type	Standard
Minnesota Dept. of Health-draft	Acute	3200 ug/m3
Minnesota Dept. of Health-draft	Chronic	115 ppb
Netherlands Dept. of Agriculture	Not a number std.	
Missouri Dept. of Natural Resources	One producer	141 ppb
EPA	Chronic	141 ppb
ATSDR	Acute	500 ppb
ATSDR	Intermediate	300 ppb

Table 10-3
Odor Standards for Various Jurisdictions

Jurisdiction	Standard
Colorado Dept. of Public Health and Environment	7 to 1 dilutions at the property line
Colorado Dept. of Public Health and Environment	2 to 1 dilutions at the property line
Missouri Dept. of Natural Resources	5.4 to 1 dilutions at the property line
North Carolina Division of Air Quality	Objectionable odor at the source

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Chapter 10. Emission Control Systems

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Chapter 10 Emission Control Systems

Abstract

Emissions odors, gases, and dust from livestock production facilities arise primarily from three sources; buildings, manure storages, and land application (1). Emissions from buildings and storages form a baseline emission level. Eliminating emissions from one of the sources will likely not eliminate emissions entirely. Control technologies often address only one of the three sources. *Many of the technologies reduce emissions; none eliminate them.*

Emissions from buildings can be reduced by inhibiting contaminant generation, or by treating the air as it leaves the building. Frequent manure removal is one of the best ways of reducing contaminant generation within the building. Frequent removal requires outdoor storage. Other methods include the use of bedding, oil sprinkling, chemical additives, and diet manipulation. Treating the air leaving a building can be done with washing walls or biofilters. Natural or manmade windbreak walls may be beneficial.

There are four types of storages; deep pits, outdoor slurry storage, anaerobic lagoons, and solid stacks. Outdoor storages are the most apparent source of odors. Controls include permeable and impermeable, natural and synthetic covers. They have been shown to be effective when managed properly. Solids separation has not been proven effective to reduce odors. Proper aeration will eliminate odors from outdoor storages, but is expensive in a liquid system. Composting works well for solid manure. Anaerobic digesters reduce odors, but are also not economically feasible. Manure additives are generally not reliable.

Emission control during land application is best done by direct injection.

General Introduction

Buildings, whether they are naturally ventilated (depend on natural breezes for ventilation), or mechanically ventilated (depend on fans for ventilation), buildings must have continuous air movement through them for the health of the animals and workers. Building emissions, along with emissions from the manure storage, form a baseline emission level for a production site (2).

Different types of storages are used for confinement systems. Outdoor pits and lagoons have the advantage of removing manure from the buildings more frequently than the "deep pit", which stores the manure under the building. Because they're exposed to the wind, outdoor storages may have a greater potential for odor and gas release.

Odor and gas releases are reduced during cold weather. Outdoor storages freeze, and building ventilation is reduced.

A high percentage of complaints each year occur due to land application of manure (3). Unlike buildings and storages, land application only occurs once or twice a year, and the impact is for short period of time. Air quality impacts involve a combination of intensity and duration. Buildings and storages represent the "baseline" emission levels. Land application can cause short term, more intense emissions.

10.1 Emission Control Strategies from Building Sources

Introduction

There are two basic approaches to minimizing odor emissions from buildings. The ideal odor control method is to minimize the odor generated in the building. The second option is to capture and treat odor as it is emitted from the building. The first method benefits the people and animals in the building as well as the neighbors. Either method helps minimize effects on neighbors.

Minimizing Odors within a Building

There are a number of recognized methods of minimizing odors generated within buildings. One of the more popular and effective is simply frequent manure removal. By using anaerobic lagoons so that "dilute" liquid is available to flush the areas where the manure collects frequently. The concept is very similar to human waste management where we flush the stool after each use. Animal facilities that flush once a week have better air quality than those that flush less frequently.

Bedded solid manure is thought to release fewer odors than liquid systems (4). Although firm scientific data has not proved it, most people feel that odors from bedded systems are less bothersome than from liquid systems, although dust may be worse from bedded systems than liquid.

Sprinkling vegetable oil in very small amounts inside swine buildings has been shown to control dust as well as odor and some gases (5,6,7). Once a day sprinkling at 0.5 ml/ft^2 has reduced dust 40-50%, odor up to 60%, and H_2S up to 60% (8). No reduction in NH_3 was found. A disadvantage is that building surfaces become oily and requires the use of degreasers in cleanup.

Chemical additives to reduce manure odors and gases have been popular with producers and vendors for many years. Unfortunately researchers have found it very difficult to prove the effectiveness of the many additives that are available. Of the products tested, relatively few have been shown to significantly reduce odor or gases. The most recent study was done by the National Pork Board (9). It investigated 35 products. Of the 35, none reduced odors at the 95% confidence level. Hydrogen sulfide was reduced at that level by 7 of the products, and 8 products reduced ammonia. Only one product was effective for both gases. These results are typical of studies over the years where given products may work for one gas, but not for anything else. Another reason additives are not recommended is their cost, which can be significant.

One application where an additive has been shown to be effective is in the poultry industry. Alum has been shown to reduce ammonia volatilization very significantly during a 42-day incubation of poultry litter (10), and also found to be cost effective due to increased production when it is used.

Ozone is a powerful oxidizing agent and germicide that has been investigated for its odor control characteristics. It's a natural component of air, and has been used to disinfect water supplies for years (11,12). It is being tested in a swine barn where it's distributed with ventilation air (13,14). Ozone's disadvantages are that it is very unstable, so it doesn't last long, and it can be very toxic and corrosive at high levels. OSHA's exposure limit is 0.1 part per million for an 8-hour exposure (15).

Diet manipulation has significant potential for ammonia reduction by reducing nitrogen (protein) in the feed (16,17). This concept is becoming more popular, but must be used with care since production can be significantly effected if protein levels are reduced too far.

Capturing and/or Treating Odor Emitted from Buildings

There are several ways of treating air before it's released from a building to lessen its odor and gas emission potential. The following are some methods that have been researched to some degree.

Washing walls is a concept that has been tested to reduce dust and odors (18). Water is used to "scrub" air as it leaves buildings similar to systems used in industry. Water recirculates through evaporative pad scrubber as exhaust fans blow air from the building. Such a system requires power ventilation systems (not natural ventilation). Washing Walls used in a swine finisher reduced total dust 20-60%, NH_3 33-50%, and reduced odors only slightly. As might be expected, better cleanup was achieved with low airflow rates compared to high rates.

Biofilters similar to those used in Europe have been adapted in the US. They use biomass and microorganisms to treat ventilation air as it leaves the building. Design parameters have been tested on a full-scale 750-head sow facility in Minnesota (19,20,21,22). At that facility the biofilter achieved odor and H_2S reduction of 80-90%, and NH_3 reduction of 50-60%. Weed control and rodent control were the primary problems experienced. A critical element in the use of biofilters is their dependence on power ventilated buildings where fans push the air through the filter. They don't work on naturally ventilated buildings.

A similar system is the biomass filter (23). Although not quite as effective as a biofilter, biomass filters do not depend on microbes to the extent of a biofilter, and they don't restrict airflow as much. Like a biofilter, power ventilation is required to use a biomass filter.

Windbreak walls are a type of wall that has been tested in the Southeast US to deflect exhaust air upward from tunnel ventilated building so it mixes with clean air, which dilutes odors and gases (24).

Windbreak walls can be constructed of various materials such as metal, straw, or wood. Without a wall, exhaust air moves along the ground and is not diluted. A Windbreak wall helps to direct barn exhaust air upward for better dispersion/dilution.

Natural windbreaks accomplish some of the same things (25). They, however, take some time to establish. Odor reduction not well researched, but thought to be beneficial through mixing, and dispersion. Natural windbreaks are naturally esthetically pleasing.

10.2 Emission Control Strategies from Manure Storages

Introduction

There are four basic types of storages that require different treatment for air quality preservation. They are deep pits outdoor slurry storage basins or tanks, anaerobic lagoons, and solid manure storage systems. The following briefly defines each type:

Deep Pits and Slurry Storage

A deep pit is a manure storage area underneath, or in the "basement," of a livestock production building. The manure storage is not visible from outside the building, and wind does not blow across the storage unit and pick up odors and gases. Manure is typically removed from deep pits only once or twice a year. No extra dilution water is added to the manure. Outdoor slurry storages may be used in place of deep pits. They may be made of earth, concrete, or steel. The earthen storages were popular due to their low cost (less than ½ the cost of the others) until regulations made them unfeasible to construct. Outdoor storages have the advantage of more frequent removal of manure from the building to provide better air quality within the building.

Anaerobic Lagoons

Anaerobic lagoons are considerably larger than earthen storage since they are designed as treatment method (26). Originally designed as an odor control method, they use microbes to digest manure solids and stabilize the manure (27). To avoid accumulating concentrations of some constituents (particularly ammonia) that are toxic t the microbes, dilution water is added. Earthen construction is used due to the large storage volume needed to accommodate the manure and dilution water.

Solid Stacks

Solid stacks can result either from using bedding to create solid manure, or from solids separated from liquid streams. Either type should be solid enough to pile up in a stack. Stacks may or may not be composted. They typically compost naturally somewhat, but may become anaerobic if piled too deep, or if the particles are too fine to admit enough oxygen for composting (28). If a stack becomes anaerobic it can be a source of odors and gases like a liquid system. Properly composted solids emit few odors (29).

Air Quality Control Technologies

Storages are the most "apparent" odor source on many farms. Since many people know that the odors coming from animal farms originate with the manure, it is natural for them to focus on the manure storage facility. The visibility of manure sources can make a difference in both the odor frequency and magnitude of what people smell. Landscaping improves the appearance of production and manure storage facilities and helps hide storages.

Covers

Synthetic impermeable covers hold gases and odors inside tank. Covers may be either rigid (wooden, concrete, fiberglass), or flexible (plastic). Synthetic plastic covers may either float on the liquid surface, be inflated, or be held above the liquid level by cables. Inflated covers are difficult to protect from high winds, so floating covers are the most common. Gas and odor reductions have been reported from 40 – 90% (2, 30).

Biocovers such as straw or cornstalks protect liquid manure from air passing over storage. Even though they are permeable, they still reduce diffusion from liquid surface to gas above (31). Some researchers feel that aerobic action occurs within the cover. In some situations a natural crust will develop which accomplishes the same result as an artificial biocover.

Other synthetic permeable covers have been developed such as a geotextile cover for earthen storages, and clay ball covers (Leka rock) on concrete slurry pit

The benefits of some of the above covers have been shown to be significant, while others are less successful (31). All covers require additional management, whether it's extra chopping of straw to avoid plugging lines, or peeling back synthetic covers to provide access for pumpout. The capital cost of covers also reduces their acceptance by producers. Approximate costs of the various types of covers follows:

Biocovers (straw)	$0.10/sq ft each year	$0.40 per head
Clay balls (LEKA)	$2-5/sq ft every	$0.33 - .80* " "
Geotextile	$0.20-$0.40/sq ft every	$0.03 - .07 " "
Plastic cover	$1-$2/sq ft every	$0.16 – 0.33 " "

* Assumes 10 year life, 10% annual interest

Solids Separation

Separating solids from liquid manure reduces the load on anaerobic lagoons, which should help reduce odors from the lagoons. Solids separation is very difficult to accomplish with liquid swine manure. Removal rates may range from 5% 5o 50% (32,33). Although the theory is sound, odor/gas reductions have not been documented due to solids reductions resulting from separating solids (34). Solids separation also creates a second waste stream to manage which may be detrimental to overall air quality if the system is not managed correctly. The cost of solids separation ranges from $10-$20 per1000 lb bodyweight per year making it very expensive. Overall solids separation is not a good alternative for air quality protection in most instances.

Aeration

Complete aerobic treatment nearly eliminates odors and undesirable gases. Many municipalities and industries use aeration for waste treatment. Continuous aeration can be achieved via floating aerators, fixed aerators, or submerged air lines. Air can be bubbled up through the liquid, whipped into the liquid, or the liquid sprayed up into the air. The disadvantage of aeration (and the reason producers don't use it) is that it requires very large amounts of energy (electricity) to accomplish the air entrainment necessary (35). The energy cost for aerating liquid manure is estimated to be $20-$40/1,000-lb bodywgt. -year

Partial Aeration can reduce odors and gases, although if under designed may actually increase odors. Floating aerators may be used for partial aeration, with the number of units determining the completeness of the aeration.

Composting is a method of aerating solid manure. Like liquid aeration, it significantly reduces odors (28). In addition, it is less energy intensive, since periodic mixing can be done relatively cheaply. Bedding in solid manure tends to make the manure "fluffy" so air naturally mixes with it to help maintain aerobic conditions. The mix of gases released is different than anaerobic treatment. Composting costs can vary significantly, but some estimates are \$0.20-\$0.40/1,000-lb bodywgt per year.

Anaerobic Treatment

Anaerobic treatment takes place in the absence of oxygen. The most common type is the anaerobic lagoon. Although the general public has a poor opinion of anaerobic lagoons, a properly operating one emits low odors. Lagoon design is based on volatile solids or COD loading, with the objective of keeping the bacterial populations in the lagoon in balance. When they are in balance, odors are minimal. In Iowa, cold weather interferes with balanced, steady state operation, and makes odor control more difficult. Oversized lagoons are sometimes used to reduce concentrations within the lagoon, thus reducing odors. The cost of oversizing lagoons can be expected to be about \$200 per 1000 lb bodyweight capital cost for extra earthwork.

Anaerobic digesters are very different from anaerobic lagoons. Digesters provide more "intense" treatment. Digesters are heated and the manure is thicker than lagoons. An overloaded condition can cause intense odors and gases. Digesters reduce odors by containing the gas that is produced so that it can be burned, and by stabilizing the liquid before it goes to the open storage tank or lagoon. Anaerobic digesters are misunderstood by the general public. Digesters are complex living organisms that are expensive to install, and require significant additional management. They do reduce odors, BOD/COD, and provide energy as heat, electricity, or both. But they do <u>not</u> reduce the volume or nutrient concentration of the manure significantly. The cost of constructing an anaerobic digestion system is approximately \$100/pig, or \$500-1,000/dairy cow capital cost. Some of the costs can be offset by the captured energy, but without higher energy prices or large government grants anaerobic digesters are not economically viable (36).

Manure Additives

Many additives are available to add to pits, lagoons, or animal feed. They work in a variety of different ways. Microbiological additives include digestive deodorants. They may be designed to enhance solids degradation, and may be pH or temperature dependant. One of the main factors that is discouraging about microbiological additives is that to work effectively, they must become the predominate bacteria. Since most bacteria are ubiquitous, if the environment favored the selected bacteria, it would already predominate in the manure (37).

Chemical (non microbiological) additives may include several mechanisms for control:
pH control
chemical oxidation
precipitation
odor masks or perfumes
adsorbents

A recently completed study of 35 additives conducted by the National Pork Board found that none of the additives decreased odors at the 95% confidence level, 6 decreased hydrogen sulfide, and 8 reduced ammonia (8).
Cost of biological/chemical additives
$0.20-$1.00/pig mktd

Emission Control from Land Application of Manure

Applying manure to cropland returns nutrients to the soil. The manure provides nutrients to the crops that would otherwise have to be purchased as commercial fertilizer. The other reason manure is land applied is *because federal law forbids discharging agricultural wastes to waters of the state or nation.* Unfortunately, land application can result in very significant odor occurrences. Even though they are not long lasting, odors from land application can be very obnoxious.

The best way to reduce odors from land application of liquid manure is direct injection of the manure below the soil surface (38,39). Research has shown that injection that accomplishes good soil cover of the manure results in odor reductions up to 90% compared to broadcast manure. Lack of complete coverage reduces odor control. Broadcasting followed by rapid incorporation also significantly reduces odors compared to broadcasting only (40), but it is not as effective as direct injection. The additional cost to inject manure is typically 1/10th of a cent per gallon more than broadcast. Some of the additional cost is offset by better nitrogen retention.

Other methods of reducing odors from land application include dilution with clean water, placement below the crop canopy (the canopy reduces air movement across the manured soil), and other potential treatments. Pretreatments have been shown to reduce odors 80%, and certain specific gases such as hydrogen sulfide up to 90% (41), but pretreatment with additives is unreliable and expensive. Research is being conducted with ozone to remove odors and reduce ammonia and hydrogen sulfide, but results aren't yet known. It's known that the technology works, but cost and management requirements haven't been proved.

Solid manure is generally less odorous than liquid, but still deserves some attention. Because it cannot be injected, rapid incorporation of solid manure is the best method to minimize odors.

Some of the best odor control results for observing common sense rules that account for wind direction and speed. Watching weather forecast, and not spreading when the wind is blowing towards neighbors can minimize severe odor "events". Several models have been developed by universities and government agencies, such as EPA's INPUFF and Minnesota's OFFSET model (42, 43), to predict odor movement and estimate their effects on neighbors.

Summary

Table 1 summarizes methods to reduce gas, odor, and dust emissions from animal facilities. Odor and gas emission sources associated with animal production facilities can be broken down into three categories: buildings, storages, and land application. Eliminating emissions from any one of the three will not eliminate emissions entirely. A number of technologies exist that are capable of reducing emissions from all three. Cost, increased management requirements, and lack of economic or regulatory incentives to encourage their use are the primary reasons more producers have not adopted the technologies. Technologies that work well, are easily managed, and are affordable have seen increased use throughout the state. These include biocovers on outside storages, utilization of deep pits (eliminating outside storages), greater use of bedded systems and composting, and manure injection during land application.

Table 1. Summary Table of Emission Reducing Strategies

Emission Source*	Emission Reducing Strategy	Targeted Components**	Documented Reduction
Housing Unit Emissions (25)			
Feeding floor (60)	Frequent, short-term pressure washing	dust, odors	65 - 70 %
Feeding floor (60)	Urine separation, complete scraping to sealed under-floor storage	dust, odors	50 - 65 %
Under-floor storage (40)	Frequent, complete scraping, water follow-up	odors	
Under-floor storage (40)	Air exchange avoidance with room air	odors	80 %
Ventilation air exhausted (100)	Dust suppression using oil sprayed on internal building surfaces	dust, odors	50 - 60 %
Ventilation air exhausted (100)	Dust suppression using biomass filters	dust, odors	50 - 60 %
Ventilation air exhausted (100)	Dust and gas suppression using biofilters	dust, odors	85 - 90 %
Storage Unit Emissions (25)			
	Floating permeable man-made covers	odors	60 - 75 %
	Floating impermeable man-made covers	odors	80 %
	Impermeable man-made covers	odors	95 %
	Chopped-straw covers	odors	75 %
	Natural crusting of manure surface	odors	75 %
	Anaerobic digestion of manure	odors	80 - 85 %
Land Applying Unit Emissions (50)			
	Surface applied, incorporation delayed 24 hours	odors	0 - 5 %
	Surface applied, incorporation delayed 12 hours	odors	0 - 5 %
	Surface applied, incorporation delayed 6 hours	odors	0 - 5 %
	Surface applied, incorporation delayed 3 hours	odors	0 - 10 %
	Surface applied, incorporated immediately by plowing	odors	50%
	Injection with full soil coverage	odors	85 - 90 %

* ( ) implies roughly the percent of total system emissions (Kroodsma et al, 1993)
** odors implies all gases emitted from livestock production systems

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Attachment 1
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Attachment 3. Glossary of Terms

ACGIH - American Conference of Governmental Industrial Hygienists
Acidic equivalent[1] - pollutants differ in their acidic effect per gram. A pollutant's effect on acidification is expressed in acidic equivalents
Acidification[1] - the process by which a soil becomes increasingly acidic. This can be caused by emissions of sulphur dioxide, nitrogen dioxide and ammonia.
Acid precipitation[1] - the mechanisms by which acidity reaches the earth's surface. These include gaseous and particle pollutants in dry, occult or wet deposition.
Acute toxicity - effects of a single dose or multiple doses measured during a twenty-four-hour period
Adverse effect[2a] - change in morphology, physiology, growth, development or life span of an organism exposed to air pollution, which results in impairment of functional capacity or impairment of capacity to compensate for additional stress or increase in susceptibility to the harmful effects of other environmental influences
Aeration[3] - a process forcing intimate contact between air and a liquid by one or more of the following methods: spraying the liquid in the air; bubbling air through the liquid; agitating the liquid to promote absorption of oxygen through the air liquid interface
Aerobic bacteria[3] - bacteria that require free elemental oxygen for their growth. Oxygen in chemical combination will not support aerobic organisms
Aerobic decomposition[3] - reduction of the net energy level of organic matter by aerobic microorganisms
Aerosols[4] - an assembly of liquid or solid particles suspended in a gaseous medium long enough to enable observation or measurement.
Agitation[3] - the turbulent mixing of liquids and slurries
ALARA principle[1] - the "As Low as Reasonably Achievable Principle" according to which rules and regulations are based on a balanced assessment of available technology, economic costs and environmental interests
Ambient[5] - surrounding, as in the surrounding environment. The medium surrounding or contacting an organism (e.g., a person), such as outdoor air, indoor air, water, or soil, through which chemicals or pollutants can be carried and can reach the organism
Anerobic bacteria[3] - bacteria not requiring the presence of free or dissolved oxygen. Facultative anaerobes can be active in the presence of dissolved oxygen, but do not require it.
Animal health[6] - a state of physical and psychological well-being and of productivity including reproduction
Animal unit - many emission quantities published are based on a per animal unit (AU) basis. Unless otherwise noted, one AU is equivalent to 500 kg body weight (1,100 lbs.)
Application regulations[1] - regulations governing when and how livestock manure, sewage sludge, compost, black soil and combinations of the above may be applied on land
Appraisal[7] - cognitive process of assessing the extent to which a threat, challenge, or loss exists and the availability of needed coping resources
Asphyxia[8] - impaired or absent exchange of oxygen and carbon dioxide on a ventilatory basis.
Asthma[9] - a lung disease with the following characteristics: 1) airway obstruction (or airway narrowing) that is reversible (but not completely so in some patients) either spontaneously or with treatment; 2) airway inflammation; and 3) airway hyper-responsiveness to a variety of stimuli.
Bacteria[1] - A group of universally distributed, rigid, essentially unicellular procaryotic microorganisms. Bacteria usually appear as spheroid, rod-like or curved entities, but occasionally appear as sheets, chains, or branched filaments.
Bioaerosol - includes the sub-class of viable particulates that has an associated biological component
Biogas[3] - gaseous product of anaerobic digestion that consists primarily of methane and carbon dioxide
Bioterrorism[10] - the overt or covert dispensing of disease pathogens by individuals, groups, or governments for the explicit purpose of causing death or disease in humans, animals, or plants. Biological terrorism agents include both living microorganisms (bacteria, protozoa, viruses, and fungi), and toxins (chemicals) produced by microorganisms, plants, or animals.
Blue baby syndrome[11] - see Methemoglobinemia
Bronchiolitis obliterans - a disease of the airways of the lung that is characterized by fibrosis (scarring) of the small airways (bronchioles). Known causes include some viral infections, rejection of a transplanted lung, and inhalation of some mineral dusts and irritant fumes.

CAFO - Concentrated Animal Feeding Operation; also known as Confined Animal Feeding Operation; generally, a facility where large numbers of farm animals are confined, fed, and raised, such as dairy and beef cattle feedlots, hog production facilities, and closed poultry houses. EPA has developed a specific regulatory definition of CAFO for the purpose of enforcing the Clean Water Act.[2]

Chronicity index[12] - ratio of the acute to chronic LD50 dosage

Chronic effects - effects produced by prolonged exposures of three months to a lifetime

Clean Water Act[11] - federal legislation administered by the U.S. EPA that serves as the primary means of protecting and regulating the surface water quality of the United State. The goal of this legislation is to eliminate the discharge of contaminants into Untied States waters and to achieve a level of water quality capable of supporting propagation of fish and wildlife and water-based recreation

Cognitive[7] - relating to thinking processes and related brain functioning

Coliform-group bacteria[1] - a group of long-living bacteria predominantly inhabiting the intestines of warm blooded animals, but also found in soil. It includes all aerobic and facultative anaerobic, gram-negative, nonspore-forming bacilli that ferment lactose with production of gas. This group of "total" coliforms include escherichia coli which is considered the typical form of fecal origin. The fecal coliforms are often used as an indicator of the potential presence of pathogenic organisms.

Concentrate feed[1] - animal feed containing mineral supplements

Concentration[7] - the strong trend of monopolization and vertical integration in agricultural production, processing, and marketing, as well as in the manufacturing of farm inputs

Contract feeding[7] - a method of livestock production in which companies provide farmers with young animals, feed, medications, etc. and the farmers provide the building, equipment, and labor, while receiving a set amount per pound or head and absorbing many of the risks of production

Control condition[7] - condition in which no treatment occurs, thus allowing comparison of the effects of the experimental treatment

Coping[7] - efforts to decrease, tolerate, or master the demands created by stressors; may be adaptive or maladaptive

Depression[7] - disorder related to brain chemistry and biologic factors that is characterized by sadness, despair, low self-esteem, low positive affect, sleep disorders, or change in appetite

Designated areas[1] - areas protected by law, in this case areas vulnerable to leaching

Disease[13] - any deviation from or interruption of the normal structure or function of the body that has a characteristic set of symptoms and signs for which there are objective findings (e.g., medical tests, x-rays) and which fits the definition of a specific disease as seen in the International Code of Diseases (ICD-9).

Disposal[11] - the discharge, deposit, injection, dumping, spilling, leaking, or placing of any solid waste or hazardous waste into the environment (land, surface water, ground water, and air)

Diversified operations[7] - farms that produce a variety of grains and livestock in ways (e.g., crop rotation) that promote environmental sustainability

Dosage - toxicity expressed as amount of toxicant per unity of body weight

Emissions - the rate at which gases or particulates leave a surface or ventilated structure. An emission rate is calculated by multiplying the concentration of a gas (mass or volume basis) by the airflow rate (volume of air per unit time) associated with this concentration

Empowerment[7] - enhancement of sense of capability, on individual and social levels, as distinct from power over others

Epidemiology - study of the distribution and determinants of health-related states or events in particular populations; application of this study to the prevention and control of health problems

Escherichia coli, E. coli[3] - one of the species of coliform bacteria in the intestinal tract of warm-blooded animals. Its presence is considered indicative of fresh fecal contamination.

Eutrophication[1] - excessive concentrations of phosphate and nitrogen enter the environment and upset the balance of water and soil ecosystems and diminish the quality of drinking water

Externalization of costs[7] - political and economic processes by which publicly unacceptable (e.g., polluting) aspects of manufacturing or production are directly or indirectly paid by the public, rather than by the manufacturer, such as through hiding or ignoring costs, passing costs along to consumers, or receiving public subsidies

Facultative bacteria[11] - bacteria that can grow in the presence, as well as the absence, of oxygen

Farm commodities[7] - the grain, livestock, fiber, and other materials produced by farmers

FEV1 - forced expiratory volume in one second

FVC - forced vital capacity

Groundwater[14] - that portion of the water below the surface of the ground at a pressure equal to or greater than atmospheric

Hazard[15] - potential for radiation, a chemical or other pollutant to cause human illness or injury
Health[2b] - health is a state of complete physical, social and mental, and social well-being and not merely the absence of disease or infirmity
Housing unit - any facility used to house livestock or poultry incorporating either a mechanical or natural ventilation system for providing fresh-air exchange
H_2S - hydrogen sulfide
Impermeable - not permitting fluids to pass through
Inhalable - the class of particulates or bioaerosols having a mean aerodynamic diameter at or below 100 um
Input standard[1] - the maximum amount of minerals per acre that may be deposited on land. The standard encompasses both the manure produced on the farm and manure or fertilizer inputted at the farmgate.
Inputs[7] - materials needed for farm production, e.g., seed, fertilizer, pesticides
Industrialized agriculture[9] - large-scale, highly capitalized farm production that favors corporate production over family farm production
Irritant[16] - toxicant that exerts its deleterious effects by causing inflammation of mucous membranes with which they came into contact. Irritants principally act on the respiratory system and can cause death from asphyxiation due to lung edema. Other mucous membranes that may be affected by irritants are those of the eyes.
Lagoon[3] - an earthen facility for the biological treatment of wastewater. It can be aerobic, artificially aerated, anaerobic or facultative depending on the loading rate, design, and type of organisms present.
Land application[3] - application of manure, sewage sludge, municipal wastewater, and industrial wastes to land either for disposal or for utilization of the fertilizer nutrients, organic matter, and improvement of soil tilth.
Land application unit - the process of applying animal manure to the soil
Laughing gas[1] - NO_2, forms naturally during nitrification. It is a greenhouse gas.
Loss standard[1] - the amounts of phosphate and nitrogen that may be released into the environment. When losses exceed the loss standard, a levy is raised on the difference.
Low emission manure application techniques[1] - techniques where manure is not spread on the surface but is injected into the sod or ploughed in to prevent ammonia emission.
Low-emission housing[1] - livestock housing with a lower ammonia emission than conventional housing
Manure[3] - the fecal and urinary excretion of livestock and poultry. Often referred to as livestock waste. This material may also contain bedding, spilled feed, water or soil. It may also include wastes not associated with livestock excreta, such as milking center wastewater, contaminated milk, hair, feathers, or other debris. Manure may be described in different categories as related to solids and moisture content. These categories are related to handling equipment and storage types.
Manure disposal contract[1] - contract between a livestock farmer with a manure surplus on his farm and an arable farmer or other user of agricultural land with a manure shortage, or a manure processing establishment
Manure storage unit - any structure used to store manure, including long-term storage inside the housing unit. Includes above- and below-ground structures.
Meteorological[17] - pertaining to the atmosphere and its phenomena, especially of its variations of heat and moisture, of its winds, etc.
Methemoglobinemia[11] - illness caused by high levels of nitrate in drinking water, above about 45 ppm, which infants are particularly susceptible to.
Methane[1] - a gas that is released during the digestive processes of ruminants. Methane is a greenhouse gas
Microorganism - a microscopic organism as a bacteria or fungi
Minerals accounting system[1] - registration of nitrogen and phosphate inputs and outputs on a farm. Input and output should be balanced although some loss is considered acceptable (loss standard).
Minimum risk level (MRL)[18] - an estimate of the daily human exposure to a hazardous substance that is likely to be without appreciable risk of adverse non-cancer health effects over a specified duration of exposure.
Multiplier effect[7] - the multiplying of economic activities, including at the community level, including that achieved through raw material production
NH_3 - ammonia
Nitrification[3] - the biological oxidation of ammoniacal nitrogen to nitrite and then to nitrate
NO_2 - nitrogen dioxide

Nonpoint source pollution[19] - Nonpoint source pollution, unlike pollution from industrial and sewage treatment plants, comes from many diffuse sources. Nonpoint source pollution is caused by rainfall or snowmelt moving over and through the ground. As the runoff moves, it picks up and carries away natural and human-made pollutants, finally depositing them into lakes, rivers, wetlands, coastal waters, and even our underground sources of drinking water. In rural areas these pollutants include bacteria and nutrients from livestock, soil sediments, fertilizers, herbicides, and insecticides.

Nutrient pollution[11] - contamination by excessive inputs of nutrient: a primary cause of eutrophication of surface waters, in which excess nutrients, usually nitrogen or phosphorus, stimulate algal growth. Sources of nutrient pollution include runoff from fields and pastures, discharges from septic tanks and feedlots, and emissions from combustion.

Odor threshold[3] - the lowest concentration of an odor in air that can be detected by the human olfactory sense

Operating costs[7] - the costs of farm inputs, labor, credit, energy, etc.

OSHA - Occupational Safety and Health Administration

Particulate - includes the class of both inert and viable aerosols. Includes total, inhalable, and respirable fractions

Parity prices[7] - equality in prices for farm commodities in which farmers get a fair return in relation to their costs of production; historically maintained by government support of farm commodity prices at a level fixed by law and indexed for inflation

PEL - Permissible Exposure Limit

Point source pollution - pollution from a particular source

Poison - see Toxicant

Pollutant[11] - a contaminant that adversely alters the physical, chemical, or biological properties of the environment. The term includes toxic metals, carcinogens, pathogens, oxygen-demanding materials, heat, and all other harmful substances, contaminants, or impurities

Pollution[11] - presence of a contaminant to such a degree that the environment (land, water, or air) is not suitable for a particular use

Price support[7] - a policy mechanism such as the non-recourse loan that sets a floor under farm commodities and thus requires exporters or processors to pay a minimum price. This is in contrast to an "income support" that involves direct payments from the U. S. Treasury to support farm income but does not directly influence market prices.

Pulmonary[8] - relating to the lungs, to the pulmonary artery, or to the aperture leading from the right ventricle into the pulmonary artery

Regulation[11] - a requirement or rule passed by an agency or department of federal, state, or local government that is authorized to create and enforce a requirement or rule through an authorizing statute or constitutional authority

Resistance - the extent to which a disease or disease-causing organism is unaffected by antibiotics or other medications

Respirable - the class of particulates or bioaerosols having a mean aerodynamic diameter at or below 5 um

Restructuring (agricultural restructuring)[20] - changes in the relationships among ownership, management, and labor in the agriculture-food system, with particular emphasis on the production component. Restructuring generally involves technological changes (induding shifts in levels of specialization/diversification) as cause or effect, and may include changes in vertical and horizontal integration or coordination, in ownership of resources (including tenancy and leasing), in farm/firm size, in geographic location of specific agri-food activities, in composition of the work force, and in levels of concentration at various levels in the supply chain.

Risk assessment - the characterization of the potential adverse health effects of human exposures to environmental hazards

Runoff[21] - occurs when input of water exceeds infiltration. Pesticide runoff includes losses from the dissolved and sediment-absorbed pesticide. Though runoff generally results directly in the contamination of surface water, it can also contribute to ground water contamination through recharging ground water by the surface water.

Setback[18] - specific distance that a structure or area must be located away, from other defined areas or structures

Sinusitis[8] - inflammation of the lining membrane of any sinus, especially of one of the sinuses alongside the nose.

Siting[11] - choosing a location for a facility

Social capital - mutual trust, reciprocity, and shared norms and identity that are inherent in relationships between and among groups

Spot market - a market in which buyer and seller come together with no pre-arranged commitment or price with the expectation of exchanging a good or service. The terms of the transaction are public, and, jointly with other similar transactions of the day, define a market price for that day.

Statistically significant difference - a research finding that is unlikely (usually less likely than 5 percent) to be due to chance

STEL - short-term exposure limit
Stress[7] - emotional, physical, behavioral, and social reactions to stressors
Stressor[7] - short-term or ongoing conditions, situations, or relationships that cause stress, often involving change, conflict, or pressure
Subacute toxic effects - toxic effects apparent over a period of several days or weeks
Subchronic toxicity - toxic effects that occur between 30 days and 90 days exposure
Supply chain[22] - the chain of transactions and product transformations that take place between the producer and consumer of a particular commodity. Historically, in agriculture, supply chains have implied openness of entry for new producers, and hence involve mass production of an undifferentiated commodity.
Tolerance - condition in which repeated exposure increases the size of the dose required to produce lethality
Toxicity - the quantitative amount or dosage of a poison that will produce a define effect
Toxicant - any natural or synthetic solid, liquid or gas that when introduced into or applied to the body can interfere with homeostasis of the organism or life processes of cells of he organism by its own inherent qualities, without acting mechanically and irrespective of temperature
Trace element[1] - chemical elements (such as copper, zinc) present in minute quantities in plant or animal tissues and considered essential to these organisms' physiological processes. An overdose, however, is harmful for the organism. Non-essential trace elements such as cadmium are harmful even in very low concentrations.
TWA - Time Weighted Average
USDA - U. S. Department of Agriculture; federal agency that is responsible for select state and local programs regarding agricultural production, conservation, and food
Value-added agriculture[7] - production of farm commodities that are fully or partially processed before being marketed by farmers (as individuals or in groups, e.g., ethanol cooperatives), thus enhancing the income of farmers and rural communities
Value chain[22] - a supply chain characterized at least in part of its links by vertical coordination. Value chains generally involve limited entry at the various levels, or links in the chain, and are focused on providing particular consumer groups with a product that fits their preferences. The emphasis is on quality (or specific qualities), rather than on producing an inexpensive product.
Vertical coordination[23] - synchronization of the vertical stages of a production/marketing system
Vertical integration[24] - coordination of two or more stages in the food chain under ownership via management directive
VOC - volatile organic compound

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ATTACHMENT 15

Mortality and Health Outcomes in North Carolina Communities Located in Close Proximity to Hog Concentrated Animal Feeding Operations

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BACKGROUND Life expectancy in southeastern North Carolina communities located in an area with multiple concentrated animal feeding operations (CAFOs) after adjusting for socioeconomic factors remains low. We hypothesized that poor health outcomes in this region may be due to converging demographic, socioeconomic, behavioral, and access-to-care factors and are influenced by the presence of hog CAFOs.

METHODS We studied mortality, hospital admissions, and emergency department (ED) usage for health conditions potentially associated with hog CAFOs—anemia, kidney disease, infectious diseases, and low birth weight (LBW)—in North Carolina communities located in zip codes with hog CAFOs (Study group 1), in zip codes with > 215hogs/km^2 (Study group 2), and without hog CAFOs (Control group). We compared cause-specific age-adjusted rates, the odds ratios (ORs) of events in multivariable analyses (adjusted for 6 co-factors), and the changes of ORs relative to the distance to hog CAFOs.

RESULTS Residents from Study groups 1 and 2 had higher rates of all-cause mortality, infant mortality, mortality of patients with multimorbidity, mortality from anemia, kidney disease, tuberculosis, and septicemia, and higher rates of ED visits and hospital admissions for LBW infants than the residents in the Control group. In zip codes with > 215hogs/km^2, mortality ORs were 1.50 for anemia ($P < 0.0001$), 1.31 for kidney disease ($P < 0.0001$), 2.30 for septicemia ($P < 0.0001$), and 2.22 for tuberculosis ($P = 0.0061$).

LIMITATIONS This study included a lack of individual measurements on environmental contaminants, biomarkers of exposures and co-factors, and differences in residential and occupational locations.

CONCLUSION North Carolina communities located near hog CAFOs had higher all-cause and infant mortality, mortality due to anemia, kidney disease, tuberculosis, septicemia, and higher hospital admissions/ED visits of LBW infants. Although not establishing causality with exposures from hog CAFOs, our findings support the need for future studies to determine factors that influence these outcomes, as well as the need to improve screening and diagnostic strategies for these diseases in North Carolina communities adjacent to hog CAFOs.

Among North Carolina communities, including both high-income and low-income communities, the lowest life expectancy was observed in southeastern North Carolina [1]. Higher risks of chronic kidney disease and low birth weight (LBW) infants have also been reported for this region [2, 3]. These geographic variations in life expectancy and health outcomes have been suggested to correlate with region-specific health behaviors, access to care, and environmental characteristics [1]. One unique environmental characteristic of southeastern North Carolina is the presence of multiple hog concentrated animal feeding operations (CAFOs) [4]. The average number of hogs per farm in North Carolina is much higher than in the areas with hog CAFOs in 2 other US leaders in hog industry—the states of Iowa and Minnesota. Because the population density in southeastern North Carolina is substantially higher than in the areas with hog CAFOs in Iowa and Minnesota, the population of the communities adjacent to hog CAFOs is much greater. Consequently, the proximity of multiple high-density hog CAFOs to a large population makes this region uniquely suited to studying the potential impact of CAFOs on environment and human health.

Previous studies of the potential relationship between health and hog CAFOs were mostly focused on the occupational health risks among CAFO workers [3, 5, 6]. The residents living in close proximity to hog CAFOs may also be at risk as they are chronically exposed to contaminants from land-applied wastes and their overland flows, leaking lagoons, and pit-buried carcasses, as well as airborne emissions, resulting in higher risks of certain diseases [3, 6, 7-20]. In fact, previous survey based studies of residential communities reported significant health risks for residents, including higher risks of bacterial infections, higher frequencies of symptoms of respiratory and neurological disorders, and depression [3, 6, 7, 12, 14, 19, 20-22].

We identified the established health conditions and indicators that were previously used to evaluate community health, including the known medical conditions associated

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with occupational or residential exposure to CAFOs. These included an increased risk of anemia and kidney disease (which may serve as an indicator of chronic exposure to toxins) [23-26], miscarriage [27], and LBW infants (which may serve as an indicators of maternal and fetal health) [2]. In addition, a higher prevalence and broader spectrum of antibiotic-resistant microorganisms in areas adjacent to hog CAFOs [28-30] has raised concerns about infections in both occupational and residential settings [31]. Therefore, the outcomes of anemia and kidney disease, acute infection (septicemia) and chronic communicable infection (tuberculosis), and LBW infants were analyzed as indicators of health in communities adjacent to hog CAFOs.

We focused our study on assessing the outcomes of these specific disorders in residential communities in southeastern North Carolina. Our objective was to determine whether, or to what extent, poor health outcomes are associated with the additional impact of hog CAFOs beyond disparities associated with demographics, socioeconomic characteristics, behavioral risks factors, or access to medical care. Furthermore, these health conditions served as potential opportunities for interventions if the determined health outcomes were poor.

Materials and Methods

Data. Data on disease-specific mortality were obtained from a publicly available data source at the State Center for Health Statistics for 2007-2013 [32]. Data on emergency department (ED) visits and hospital admissions were obtained from the Healthcare Cost and Utilization Project's (HCUP) State Emergency Department Database (SEDD) [33] and State Inpatient Database (SID) [34] for 2007-2013. The North Carolina analysis represents part of the larger study on health outcomes in the communities adjacent to hog CAFOs that includes other US states with commercial hog production (eg, Iowa and Minnesota). Therefore, we used the HCUP's state-specific database containing the data in a uniform format facilitating multi-state comparisons and analyses of geographic patterns and time trends in health care utilization, access, and outcomes across multiple US states. The SEDD captures discharge information on all ED visits that do not result in an admission and contains more than 100 clinical and non-clinical variables. Information on patients that are initially seen in the ED and then admitted to the hospital is included in SID, which encompasses almost 97% of all US hospital discharges. The SID and SEDD data for North Carolina for the period analyzed in this study had several issues that were addressed in performed analysis. For example, the 2011–2012 North Carolina SEDD included 2 types of erroneous records, such as duplicated records for ED visits that did not result in an admission to the same hospital and records for ED visits that did result in an admission to the same hospital. The SID dataset for North Carolina for 2007-2008 had problems with the coding of discharge disposition. These issues were identified and resolved according to the guidelines provided by the HCUP Data Center.

The list of swine animal operations registered in North Carolina contained information on geographic locations and the number of swine in each CAFO facility. Information was obtained from the North Carolina Division of Water Resources (NC DWR) for the year 2009. The animal operations are defined by General Statute 143-215.10B as feedlots involving 250 or more swine with a liquid waste management system.

Zip-code-level data on median household income (scaled by $10,000) and education level (defined as a percentage of people aged 25+ who attained an educational level higher than a bachelor's degree) were obtained from the 2010-2014 American Community Survey. County level data on the numbers of primary care providers (per 100,000 residents) and the percent of uninsured individuals was obtained from the Area Health Resources Files (AHRF) for 2008 and 2010-2013. County level data on prevalence of current smokers in age-specific groups were obtained from the Behavioral Risk Factor Surveillance System (BRFSS, CDC) for 2008-2013.

Methods. We studied the health outcomes in two study groups. Study group 1 included the residents of North Carolina communities located in zip codes with hog CAFO(s): 221 zip codes with approximately 2,260,000 residents. Study group 2 represented a subset of Study group 1. This group included North Carolina communities located in zip codes with the highest upper quartile of hog density (with > 215hogs/km^2): 56 zip codes with approximately 400,000 residents. North Carolina communities located in zip codes without hog CAFOs represented the Control group: 601 zip codes with approximately 7,200,000 residents. Geographic locations of zip codes for two Study groups and the Control group are shown in Figure 1.

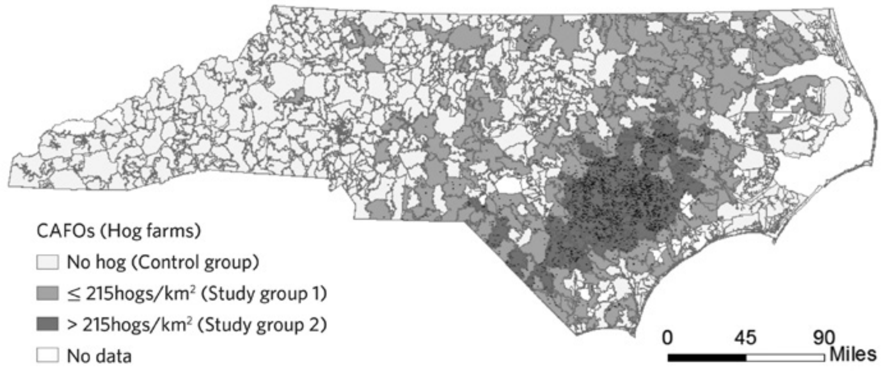
We compared disease-specific mortality, hospital admissions, and ED visits in these groups for the 2007-2013 period. All-cause, infant mortality, and outcomes of anemia, kidney disease, tuberculosis, septicemia, and LBW infants (see Appendix for respective ICD codes) were studied as the health indicators, with disease-specific mortality as primary outcome. The main predictor was the presence of a hog CAFO in a given zip code. Analyses were performed for underlying cause of death/primary diagnosis and for underlying-plus-secondary causes of death/primary-plus-secondary diagnoses. The illustration of the relations in assessment of potential impact factors/outcome associations used in multivariable analysis is shown in Supplemental Figure S1 in the Appendix.

Age-adjusted rates. We empirically estimated disease-

APPENDIX 1.
The International Classification of Diseases (ICD) codes used in the analysis

This appenix is available in its entirety in the online edition of the NCMJ.

FIGURE 1.
Density of Hogs in Zip Codes in Study Group 1 and Study Group 2, Locations of Registered at NC DWR Hog CAFOs in NC, and Locations of Zip Codes without Registered at DWR Hog CAFOs (Control Group), 2009



specific, age-adjusted rates of mortality, hospital admission, and ED visits (per 100,000). 95% confidence intervals (CIs) were estimated based on the approximation suggested by Keyfitz [35]. We compared these rates between Study groups 1 and 2 and the Control group, and additionally to North Carolina and the US average (for mortality rates).

Logistic regression analysis. We used logistic regression analysis (adjusted by age, median household income, education, health insurance coverage, numbers of primary care providers, and smoking prevalence) to evaluate whether a proportion of disease-specific deaths (as well as a proportion of disease-specific hospital admissions and ED visits) among all-cause deaths/all hospital admissions/all ED visits statistically differed between the studied groups. The Control group was a referent group for calculating ORs. This analysis allowed for minimization of potential bias due to uncertainties in population counts in North Carolina zip codes over the study period. SAS Proc Logistic (the SAS 9.4 statistical package; SAS Institute, Cary, NC) was used to evaluate ORs, 95% CIs, and p-values.

The DiSC analysis. We developed and applied an approach we termed the Distance from the Source of potential Contamination (DiSC) analysis to investigate the changes in ORs for all studied health outcomes with closer proximity to the CAFO. The core of this analysis is the new zip-code-specific continuous measure of potential exposures from hog CAFOs constructed using the exact address of each CAFO and the population counts in all census blocks in each zip code. We hypothesized that the risk of mortality (or hospital admission or ED visit) is proportional to the number of hogs in a CAFO, maximal at the location of a CAFO, and decreases with remoteness from a CAFO according to two-dimensional normal distribution (ie, "bell-shaped" distribution) of potential contaminants. Its standard deviation σ is the measure of the distance from the CAFO at which the level of potential contaminants drops 2-fold. The functional form is justified by the theory of diffusion from a point source [36]. The zip-code-specific measures of potential contaminants from CAFOs were modeled by summing the contributions of all census block groups in a given zip code:

$$E_z(\sigma) = \frac{\sum_i p_{iz} \sum_n N_n f(d_{niz}, \sigma)}{\sum_i p_{iz}}$$

where n enumerates all CAFOs; N_n is the number of hogs in the CAFO n; i enumerates all census block groups in a zip code z; p_{iz} is population of census block group i in zip-code z; and

$N_n f(d_{niz}, \sigma)$; where $f(d_{niz}, \sigma) = \dfrac{1}{\sqrt{2\pi}\sigma} \exp\left(-\dfrac{d_{niz}^2}{2\sigma^2}\right)$

is the modeled contaminant level from a specific CAFO in a census block group (where d_{niz} is the distance between them). Since there are no direct measurements that allow for estimating σ, we performed radius-specific analyses corresponding to 4 values of σ: ie, at 2, 5, 10, and 20 kilometers (km). A zip-code-specific value of $E_z(\sigma)$ was then used in the logistic regression analysis to evaluate the associations with disease-specific outcomes in multivariable analysis. The evaluated ORs are per a unit of $E_z(\sigma)$. The OR estimates for different σ are comparable because the measures are normalized equally: sums of contaminant levels over all zip-codes equal the total number of hogs in all CAFOs for any σ.

Sensitivity analyses. Because hog CAFOs are predominantly located in rural North Carolina, and access to medical care likely differs in urban and rural areas, we i) excluded zip codes of the cities of Charlotte and Raleigh, and also ii) excluded 18 urbanized areas defined in the US Census Bureau criteria for urban-rural areas as having ≥ 50,000 residents.

FIGURE S1.
Illustration of the Relations in the Assessment of Potential Impact Factors-Outcome Associations

This figure is available in its entirety in the online edition of the NCMJ.

We also used the generalized estimating equation (GEE) method to account for possible correlations between records in specific zip codes.

We used the greedy matching algorithm [37] to perform propensity score-based matching of zip codes from Control group to zip codes in Study group 2 by demographic and socioeconomic characteristics (see Appendix for detailed description of the matched groups and their characteristics presented in Table S1).

Ethics statement. All data analyses were designed and performed in accordance with the ethical standards of a responsible committee on human studies and with the Helsinki Declaration (of 1975, revised in 1983) and have been approved by the Duke University Health System Institutional Review Board.

Results

Demographic and socioeconomic characteristics. The residents of communities adjacent to hog farms were more diverse than the average North Carolina community. There were more African-American (28.8% vs. 19.3%, $P < 0.001$) and American-Indian (2.4% vs. 0.8%, $P < 0.05$) residents in zip codes with hog CAFOs (Study group 1) compared to the Control group (see Supplemental Tables S2 and S3 in Appendix). Study group 1 also had a lower median household income ($39,005 vs. $46,414, $P < 0.001$), fewer college-educated people with bachelor's or higher degrees (16.5% vs. 24.2%, $P < 0.001$), and a lower number of primary care health providers (54 vs. 76 per 100,000 residents, $P < 0.001$). The differences were even more pronounced for the residents of communities located in zip codes with > 215hogs/km^2 (Study group 2): 31.3% ($P < 0.001$) of the residents were African Americans and 4.1% were American Indians ($P < 0.001$). People from Study group 2 had the lowest (among the studied groups) median household income ($36,520, $P < 0.001$), percent of residents with bachelor's or higher degrees (13.7%, $P < 0.001$), and number of primary care providers (51/100,000, $P < 0.001$) (see Supplemental Tables S2 and S3).

TABLE S1.
Characteristics of Matched Group A, Matched Group B, and Study Group 2, NC, 2007-2013

This table is available in its entirety in the online edition of the NCMJ.

[1]Means are evaluated without weights representing zip-code populations.
n/a, non-applicable.

TABLE S2.
Descriptive Table of the 3 Studied Groups of NC Communities with and without Hog Concentrated Feeding Animal Operations (CAFOs): Race-Specific Population Groups, Socioeconomic Characteristics, Smoking Prevalence, and Access-To-Care Characteristics, NC, 2007-2013

This table is available in its entirety in the online edition of the NCMJ.

*$P < 0.05$.
**$P < 0.001$.

TABLE S3.
Person-Years of Observations in Race-Specific Groups of the Residents of NC Communities from the 3 Studied Groups, NC, 2007-2013

This table is available in its entirety in the online edition of the NCMJ.

Mortality rates. Cause-specific mortality rates of all studied diseases were higher in North Carolina communities located in zip codes with > 215hogs/km^2 (Study group 2) compared to the North Carolina and US averages (see Table 1). The all-cause mortality rate in Study group 2 was as high as 934/100,000.

The residents from Study group 2 aged ≤ 24 years old had much higher all-cause mortality rates (92.7/100,000) than mortality rates in North Carolina (69.8/100,000) and the US (62.2/100,000) for this age group (see Table 1). Conditions originating in the perinatal period may have substantially contributed to the differences in mortality at younger ages; the mortality rate among infants under 1 year old in Study group 2 was as high as 495/100,000. This is much higher than both the US average (317/100,000) and the North Carolina average (398/100,000). The groups that contributed the most to increased mortality rates due to perinatal conditions were newborns affected by maternal trauma and by disorders related to length of gestation and fetal growth (see Table 1). The rates of infant death related to maternal trauma were much higher in North Carolina communities located in zip codes with > 215hogs/km^2 (149/100,000) than the United States and North Carolina averages. The rates of death related to the length of gestation and fetal growth were higher in both North Carolina (North Carolina average) and Study group 2 compared to the US average.

Patients from Study group 2 with multimorbid conditions such as co-existing septicemia and kidney disease, septicemia and anemia, or septicemia and kidney disease and anemia had mortality rates 1.5-2.2 times greater than North Carolina and 1.8-1.9 times greater than the US average mortality rates for patients with the same respective co-existing diseases (see Supplemental Figure S2 in Appendix). For all studied diseases, the age-adjusted mortality rates were higher in Study group 1 than in the Control group, but lower than in Study group 2 (see Table 2), except for tuberculosis: its mortality did not significantly differ between Study groups 1 and 2.

To highlight the magnitude of higher mortality in the region, we modeled Study group 2 as an independent geo-

TABLE 1.
Age-Adjusted Mortality Rates (Per 100,000) in NC Communities with > 215hogs/km^2 (Study Group 2) Compared to the NC and US Average, 2007-2013. (95% Confidence Intervals are Shown in the Parentheses)

Disease	Age and race group	The US average[1]	The NC average[a]	NC communities with > 215hogs/km^2
All-cause mortality	All ages, all races	750 (749.5-750.2)	**803**[a] (801.3-805.6)	**934**[ab] (922.7-944.8)
	White, all ages	745 (744.5-745.2)	**780**[a] (777.9-782.6)	**858**[ab] (844.7-871.2)
	AA,[2] all ages	903 (901.6-904.1)	**923**[a] (917.4-928.4)	**969**[ab] (947.9-989.4)
	Age ≤ 24 years old, all races	62.2 (62.0-62.4)	**69.8**[a] (68.7-70.9)	**92.7**[ab] (86.3-99.1)
Conditions of perinatal period	All races, age < 1 year old	317 (314.4-318.6)	**398**[a] **(381.1-408.5)**	**495**[ab] **(420.7-569.5)**
Newborns affected by maternal trauma	All races, age < 1 year old	74.6 (73.6-75.6)	**102**[a] **(95.7-109.1)**	**149**[ab] **(110.6-195.3)**
Disorders related to length of gestation and fetal growth	All races, age < 1 year old	112 (110.6-113.1)	**163**[a] **(154.8-171.8)**	**169**[a] **(128.3-218.4)**
Anemia (underlying cause)	All races, all ages	1.5 (1.5-1.5)	**1.9**[a] (1.8-2.0)	**2.6**[ab] (2.1-3.2)
	AA, all ages	3.0 (2.9-3.0)	**3.6**[a] (3.3-4.0)	**5.3**[ab] (3.9-7.1)
Kidney disease (underlying cause)	All races, all ages	14.6 (14.5-14.6)	**18.3**[a] (18.0-18.6)	**24.8**[ab] (23.0-26.6)
	White, all ages	13.3 (13.3-13.4)	**14.8**[a] (14.5-15.2)	**18.3**[ab] (16.3-20.2)
	AA, all ages	28.1 (27.9-28.3)	**34.9**[a] (33.8-36.0)	**37.7**[a] (33.6-41.8)
Tuberculosis (underlying + secondary cause)	All races, all ages	0.31 (0.30-0.32)	0.30 (0.26-0.35)	**0.63**[ab] (0.32-0.81)
Septicemia (underlying cause)	All ages, all races	10.8 (10.7-10.8)	**13.5**[a] (13.2-13.67)	**16.6**[ab] (15.1-18.1)

[1]Mortality rates are obtained from the Centers for Disease Control and Prevention Multiple Cause of Death data (https://wonder.cdc.gov/mcd.html).
[2]African-American.
[a]Statistically significant difference compared to the US average.
[b]Statistically significant difference compared to NC average.

graphic unit and compared its overall and disease-specific mortality rates to the US states with the highest mortality rates (see Supplemental Table S4 in Appendix). In this model, the geographic area encompassing Study group 2 would be ranked number 4 in the United States for the highest all-cause mortality, number 1 in the United States for mortality from anemia as underlying cause, number 1 for kidney disease, number 2 for septicemia, and number 3 for tuberculosis as underlying-plus-secondary cause.

The rates of hospital admissions and ED visits. For most of the studied diseases, the rates of hospital admissions and ED visits (see Table 2) were higher in Study group 1 than in the Control group, but lower than in Study group 2. Rates did not differ between Study groups 1 and 2 for anemia hospital admissions and ED visits (as primary diagnosis), ED visits for tuberculosis, and LBW hospital admissions (as primary-plus-secondary diagnosis); however, these rates were still higher than in the Control group.

Logistic regression analysis. After adjustment for 6 co-

FIGURE S2.
Mortality Rates among Patients with Co-Existing Anemia, Kidney Disease, and Septicemia: The US Average, NC Average, and NC Communities with > 215hogs/km^2 (Study Group 2), 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

This figure is available in its entirety in the online edition of the NCMJ.

TABLE S4.
Age-Adjusted Mortality Rates (Per 100,000) in NC Communities with > 215hogs/km^2 (Study Group 2): Ranks of This Area among the US States and District of Columbia with the Highest Mortality, 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

This table is available in its entirety in the online edition of the NCMJ.

[a]Mortality rates were calculated using the Multiple Cause of Death data from the Centers for Disease Control and Prevention (https://wonder.cdc.gov/mcd.html).

factors, the ORs for death, hospital admissions, and ED visits for most of the studied diseases in Study group 1 were > 1.0 (see Table 3). The ORs in Study group 2 were significantly higher than in Study group 1 for kidney disease (all 3 outcomes), tuberculosis (hospital admissions), anemia (all 3 outcomes), tuberculosis (ED visits), septicemia (mortality), and LBW (ED visits) (see Table 3).

DiSC analysis. After adjustment for 6 co-factors, the studied outcomes had similar distance-related patterns: the ORs were higher in close proximity to a hog CAFO than in more distant communities (see Table 4). For example, mortality ORs for kidney disease were the highest in communities located within 2 km of a CAFO (OR = 1.14, $P < 0.0001$), then decreased to 1.02 ($P < 0.0001$) at 20 km. For hospital admissions, the OR for kidney disease was 1.22 ($P < 0.0001$) at 2 km, 1.08 at 5 km ($P < 0.0001$), 1.04 at 10 km ($P < 0.0001$), and 1.03 at 20 km ($P < 0.0001$). The most pronounced changes in ORs were observed between 2 km and 5 km from the CAFO.

Sensitivity analysis. After exclusion of urban areas, no significant changes were observed for mortality risks. Slightly lower ORs than in the main analysis were observed for hospital admissions, and slightly higher ORs were observed for ED visits. The results of GEE analysis also confirmed the main study results; one exclusion was some minor changes in hospital admissions.

Locations of matched zip codes are shown in Supplemental Figure S3 (Appendix): compared to "clustered" locations of zip codes with > 215hogs/km^2, non-CAFOs zip codes are sparsely located in different regions of North Carolina. The mortality rates of all studied diseases and hospital admission/ED visit rates of kidney disease, tuberculosis, and LBW were higher in Study group 2 than in matched zip codes

TABLE 2.
Age-Adjusted Rates (per 100,000) of Mortality, Hospital Admissions, and ED Visits in NC Communities with Hog CAFOs (Study Group 1), NC Communities with > 215hogs/km^2 (Study Group 2), and NC Communities without Hog CAFOs (Control Group), 2007-2013. Underlying Cause/Primary Diagnosis and Underlying-Plus-Secondary Cause/Primary-Plus-Secondary Diagnosis. (95% Confidence Intervals Are Shown in the Parentheses)

		Underlying cause/Primary diagnosis			Underlying+secondary cause/ Primary+secondary diagnosis		
Outcome	**Disease**	**Study group 1**	**Study group 2**	**Control group**	**Study group 1**	**Study group 2**	**Control group**
Mortality	All-cause mortality	**866**[a] (861.1-870.0)	**934**[b] (922.7-944.8)	773 (770.4-775.2)	**866**[a] (861.1-870.0)	**934**[b] (922.7-944.8)	773 (770.4-775.2)
	Anemia	**2.3**[a] (2.1-2.6)	**2.6**[a] (2.1-3.2)	1.7 (1.6-1.8)	**28.4**[a] (27.6-29.2)	**35.5**[ab] (33.4-37.7)	17.0 (16.7-17.4)
	Kidney disease	**21.1**[a] (20.4-21.8)	**24.8**[ab] (23.0-26.6)	17.1 (16.7-17.5)	**101**[a] (99.1-102.1)	**119**[ab] (114.6-122.5)	75.4 (74.7-76.2)
	Tuberculosis	**0.32**[a] (0.21-0.42)	**0.24**[a] (0.04-0.43)	0.13 (0.12-0.14)	**0.52**[a] (0.42-0.61)	**0.63**[a] (0.32-0.81)	0.23 (0.22-0.34)
	Septicemia	**15.5**[a] (14.9-16.1)	**16.6**[a] (15.1-18.1)	12.7 (12.4-13.0)	**67.9**[a] (66.7-69.1)	**75.1**[ab] (71.9-78.2)	50.9 (50.3-51.5)
Hospital admissions	Anemia	**112**[a] (110.7-114.0)	**113**[a] (108.6-116.4)	87.4 (86.6-88.2)	**1,989**[a] (1,982-1,996)	**2,179**[ab] (2,162-2,196)	1,642 (1,638-1,645)
	Kidney disease	**164**[a] (162.3-166.2)	**187**[ab] (181.6-191.4)	128 (126.6-128.6)	**1,809**[a] (1,802-1,815)	**2,031**[ab] (2,015-2,048)	1,369 (1,366-1,372)
	Tuberculosis	**1.8**[a] (1.6-2.0)	**3.1**[ab] (2.4-3.7)	1.0 (0.9-1.1)	**4.0**[a] (3.7-4.3)	**6.2**[ab] (5.3-7.1)	2.4 (2.3-2.6)
	Septicemia	**296**[a] (293.6-298.8)	**313.1**[ab] (306.7-319.5)	239 (237.8-240.4)	**437**[a] (433.9-440.2)	**468**[ab] (460.3-475.9)	344 (342.1-345.2)
	Low birth weight	n/a	n/a	n/a	**2.2**[a] (1.9-2.4)	**2.5**[a] (1.9-3.1)	1.5 (1.4-1.6)
ED visits	Anemia	**84.8**[a] (83.3-86.2)	85.4[a] (81.9-88.9)	71.4 (70.6-72.1)	**605**[a] (600.8-608.4)	**682**[ab] (672.2-691.7)	480 (478.1-481.9)
	Kidney disease	**26.4**[a] (25.6-27.2)	**33.2**[ab] (31.1-35.3)	19.6 (19.2-20.0)	**547**[a] (543.4-550.5)	**643**[ab] (634.0-652.3)	376 (373.9-377.2)
	Tuberculosis	0.22 (0.13-0.32)	0.33 (0.12-0.53)	0.14 (0.11-0.14)	**1.04**[a] (0.8-1.13)	**1.42**[a] (1.03-1.93)	0.72 (0.62-0.74)
	Septicemia	**15.4**[a] (14.8-16.0)	**20.1**[ab] (18.4-21.7)	13.7 (13.4-14.0)	**26.2**[a] (25.4-26.9)	**35.4**[ab] (33.3-37.6)	21.1 (20.7-21.5)
	Low birth weight	n/a	n/a	n/a	**3.0**[a] (2.7-3.3)	**4.7**[ab] (3.9-5.5)	1.6 (1.5-1.7)

[a]Statistically significant difference compared to the Control group.
[b]Statistically significant difference compared to Study group 1.
n/a, non-applicable.

TABLE 3.
Age-Adjusted Rates (per 100,000) of Mortality, Hospital Admissions, and ED Visits in NC Communities with Hog CAFOs (Study Group 1), NC Communities with > 215hogs/km^2 (Study Group 2), and NC Communities without Hog CAFOs (Control Group), 2007-2013. Underlying Cause/Primary Diagnosis and Underlying-Plus-Secondary Cause/Primary-Plus-Secondary Diagnosis. (95% Confidence Intervals Are Shown in the Parentheses)

		Underlying cause/Primary diagnosis		Underlying+secondary cause/ Primary+secondary diagnosis	
Outcome	Disease	Study group 1	Study group 2	Study group 1	Study group 2
Death	Anemia	**1.24** (1.11-1.36), *P* = 0.0012	**1.39** (1.15-1.64), *P* = 0.0077	**1.34** (1.30-1.38), *P* < 0.0001[#]	**1.50**[a] (1.43-1.57), *P* < 0.0001[#]
	Kidney disease	**1.13** (1.09-1.17), *P* < 0.0001[#]	**1.27**[a] (1.19-1.35), *P* < 0.0001[#]	**1.18** (1.16-1.20), *P* < 0.0001[#]	**1.31**[a] (1.27-1.35), *P* < 0.0001[#]
	Tuberculosis	**2.77**[a] (2.33-3.21), *P* < 0.0001[#]	2.12 (1.19-3.04), *P* = 0.1125	**2.23** (1.93-2.54), *P* < 0.0001[#]	**2.22** (1.65-2.79), *P* = 0.0061
	Septicemia	**1.07** (1.02-1.12), *P* = 0.0120	1.08 (0.97-1.17), *P* = 0.1633	**1.18** (1.15-1.20), *P* < 0.0001[#]	**2.30**[a] (2.11-2.48), *P* < 0.0001[#]
Hospital admissions	Anemia	**1.07** (1.05-1.09), *P* < 0.0001[#]	**1.07** (1.03-1.11), *P* = 0.0022	**1.03** (1.03-1.04), *P* < 0.0001[#]	**1.12**[a] (1.11-1.14), *P* < 0.0001[#]
	Kidney disease	**1.09** (1.07-1.11), *P* < 0.0001[#]	**1.21**[a] (1.18-1.24), *P* < 0.0001[#]	**1.15** (1.15-1.16), *P* < 0.0001[#]	**1.33**[a] (1.32-1.34), *P* < 0.0001[#]
	Tuberculosis	**1.48** (1.31-1.64), *P* < 0.0001[#]	**2.81**[a] (2.54-3.08), *P* < 0.0001[#]	**1.39** (1.28-1.50), *P* < 0.0001[#]	**2.30**[a] (2.11-2.48), *P* < 0.0001[#]
	Septicemia	**1.03** (1.02-1.04), *P* < 0.0001[#]	**1.03** (1.00-1.05), *P* = 0.0324	**1.06** (1.05-1.07), *P* < 0.0001[#]	**1.08** (1.06-1.10), *P* < 0.0001[#]
	LBW	n/a	n/a	**1.44** (1.25-1.62), *P* < 0.0001[#]	1.40 (1.04-1.76), *P* = 0.0661
ED visits	Anemia	1.02 (1.00-1.05), *P* = 0.0721	**1.08**[a] (1.03-1.13), *P* = 0.0028	**1.08** (1.07-1.09), *P* < 0.0001[#]	**1.21**[a] (1.19-1.23), *P* < 0.0001[#]
	Kidney disease	**1.05** (1.00-1.09), *P* = 0.0431	**1.26**[a] (1.18-1.34), *P* < 0.0001[#]	**1.23** (1.22-1.24), *P* < 0.0001[#]	**1.43**[a] (1.41-1.45), *P* < 0.0001[#]
	Tuberculosis	1.38 (0.84-1.93), *P* = 0.2451	2.26 (1.33-3.19), *P* = 0.0868	1.24 (1.01-1.47), *P* = 0.0721	**2.22**[a] (1.84-2.61), *P* < 0.0001[#]
	Septicemia	**0.89** (0.82-0.96), *P* = 0.0013	**0.82** (0.69-0.96), *P* = 0.0057	0.98 (0.92-1.03), *P* = 0.3671	0.99 (0.89-1.09), *P* = 0.8742
	LBW	n/a	n/a	**1.53** (1.34-1.73), *P* < 0.0001[#]	**2.45**[a] (2.13-2.76), *P* < 0.0001[#]

[a]Statistically significant difference between the Study groups 1 and 2.
[#]Remains significant under Bonferroni correction.
n/a, non-applicable.

without CAFOs (the results are presented in the Appendix, Table S5).

Discussion

We found that people living in southeastern North Carolina communities located near hog CAFOs had poorer outcomes for a variety of health conditions in different age groups than the residents of North Carolina communities located in zip codes without hog CAFOs; they had higher mortality due to infections, anemia, kidney disease, and perinatal conditions, and higher rates of hospital admissions and ED visits for LBW infants. The observed higher rate of all-cause mortality is consistent with the lower life expectancy in this area [1].

While the precise causes of higher anemia rates observed in our study are unclear, other studies have suggested that exposure to ammonia, hydrogen sulfide, methane, and particulate matters (PMs) near the CAFOs [23, 24], contamination of water and soil with zinc [25], exposure to the antibiotic chloramphenicol previously widely used to treat infections in hogs [26], and inappropriate human use of veterinary medications (certain NSAIDs or antibiotics) [38] cause anemia. Moreover, anemia is an independent risk factor of death in patients with chronic diseases [39, 40], a complication of renal failure [41] and tuberculosis [42], and a risk factor for preterm birth and LBW infants [43].

Earlier studies reported that workers in the swine

FIGURE S3.
Locations of Matched NC Zip Codes without Hog CAFOs (Matched Group A and Matched Group B) and Locations of Zip Codes with > 215hogs/km^2 (Study Group 2)

This figure is available in its entirety in the online edition of the NCMJ.

TABLE 4.
The Distance from the Source of Potential Contamination ("DiSC") Analysis: ORs of Mortality, Hospital Admissions, and ED Visits in NC Communities Located within Different Distances from Hog CAFOs: Underlying-Plus-Secondary Causes of Death/Primary-Plus-Secondary Diagnoses, Logistic Regression, Multivariable Analysis (Adjusted by Age, Income, Education, Health Insurance, Smoking, and Availability of Primary Care Providers), 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

		The distance from hog CAFO			
Outcome	Disease	2 km	5 km	10 km	20 km
Death	Anemia	1.11 (1.05-1.18), $P < 0.0001$	1.05[a] (1.03-1.07), $P < 0.0001$	1.04 (1.03-1.05), $P < 0.0001$	1.03 (1.03-1.04), $P < 0.0001$
	Kidney disease	1.14 (1.11-1.18), $P < 0.0001$	1.06 a (1.05-1.07), $P < 0.0001$	1.03 (1.03-1.04), $P < 0.0001$	1.02 (1.02-1.03), $P < 0.0001$
	Tuberculosis	1.37 (0.95-1.79), $P = 0.1442$	1.12 (0.96-1.27), $P = 0.1621$	1.09 (1.02-1.16), $P = 0.0231$	1.07 (1.03-1.11), $P < 0.0001$
	Septicemia	1.11 (1.06-1.15), $P < 0.0001$	1.04[a] (1.03-1.06), $P < 0.0001$	1.03 (1.02-1.03), $P < 0.0001$	1.02 (1.02-1.09), $P < 0.0001$
Hospital admissions	Anemia	1.06 (91.05-1.07), $P < 0.0001$	1.02[a] (1.02-1.03), $P < 0.0001$	1.01 (1.01-1.02), $P < 0.0001$	1.01 (1.01-1.01), $P < 0.0001$
	Kidney disease	1.22 (1.21-1.23), $P < 0.0001$	1.08[a] (1.08-1.09), $P < 0.0001$	1.04[a] (1.04-1.04), $P < 0.0001$	1.03[a] (1.03-1.03), $P < 0.0001$
	Tuberculosis	1.59 (1.44-1.75), $P < 0.0001$	1.18[a] (1.13-1.24), $P < 0.0001$	1.09[a] (1.06-1.12), $P < 0.0001$	1.06 (1.04-1.07), $P < 0.0001$
	Septicemia	1.10 (1.08-1.11), $P < 0.0001$	1.04[a] (1.03-1.04), $P < 0.0001$	1.02[a] (1.02-1.02), $P < 0.0001$	1.02 (1.01-1.02), $P < 0.0001$
	LBW	1.21 (0.97-1.46), $P = 0.1272$	1.06 (0.97-1.15), $P = 0.1913$	1.04 (0.99-1.08), $P = 0.1112$	1.03 (1.01-1,06), $P = 0.0082$
ED visits	Anemia	1.15 (1.14-1.17), $P < 0.0001$	1.05[a] (1.05-1.06), $P < 0.0001$	1.03[a] (1.02-1.03), $P < 0.0001$	1.02 (1.02-1.02), $P < 0.0001$
	Kidney disease	1.23 (1.21-1.24), $P < 0.0001$	1.08[a] (1.08-1.09), $P < 0.0001$	1.04[a] (1.04-1.05), $P < 0.0001$	1.03[a] (1.03-1.03), $P < 0.0001$
	Tuberculosis	1.99 (1.69-2.29), $P < 0.0001$	1.30[a] (1.19-1.40), $P < 0.0001$	1.13[a] (1.08-1.18), $P < 0.0001$	1.07[a] (1.04-1.10), $P < 0.0001$
	Septicemia	1.14 (1.06-1.22), $P < 0.0001$	1.06[a] (1.03-1.09), $P < 0.0001$	1.03 (1.02-1.04), $P < 0.0001$	1.02 (1.01-1.03), $P < 0.0001$
	LBW	2.28 (2.12-2.44), $P < 0.0001$	1.39[a] (1.34-1.45), $P < 0.0001$	1.20[a] (1.17-1.22), $P < 0.0001$	1.13[a] (1.11-1.14), $P < 0.0001$

[a]Statistically significant difference from the value of the result at shorter vs. longer distances (eg, 5 km vs. 2 km, or 10 km vs. 5 km) within the same row in the table.

industry have a higher risk for tuberculosis; however, this disease has been recently eradicated from US livestock [44]. Our findings on higher rates of tuberculosis likely result from the impact of a combination of factors in this North Carolina region where co-existing medical and social determinants may exacerbate each other [6, 10]. While no information is currently available on potential risk of occurrence of antibiotic-resistant strains of *Mycobacterium tuberculosis* in the communities adjacent to hog CAFOs, this aspect may require detailed analysis. The increased risk of undiagnosed latent tuberculosis that may be present in these communities, which may have a higher number of foreign-born residents [45], also requires attention. Co-existence of factors that may promote tuberculosis from its latent to active form (eg, diabetes, immunosuppression, and other conditions) needs to be accounted for when developing a strategy for improving identification of latent and active cases (ie, through screening) and treatment adherence in patients who require therapy.

Higher mortality rates for infants living in North Carolina zip codes with > 215hogs/km^2 represent an important health issue for this population that requires the immediate attention of public health and health care specialists. Maternal trauma and the length of gestation and fetal growth contribute the most to infant mortality in these North Carolina communities and can be targeted by special programs on maternal and child health. Higher rates of LBW infants in North Carolina communities adjacent to hog CAFOs are an important parameter of maternal and child health, not only because of the immediate medical care needed for such infants, but also because of their increased lifetime risk of chronic diseases (eg, higher risk of development of diabetes mellitus, arterial hypertension, ischemic heart disease, depression, respiratory diseases, and chronic kidney disease) [46]. Targeted programs in North Carolina communities adjacent to hog CAFOs could provide information about health issues related to women's and children's health to women of childbearing potential, as well as supporting mothers and children from pregnancy through birth and beyond.

The DiSC analysis in our study highlighted a potential opportunity for associating residential and occupational exposures in communities located in close proximity to hog CAFOs; poorer health outcomes among the residents of communities located within 2-5 km from CAFOs could be due to additional exposures because of potential employment at CAFOs. That may provide some guidance as to the most efficient use of resources to screen and diagnose diseases/conditions found to be highly prevalent in these communities.

In this study we do not establish causality between exposures from hog CAFOs and higher risk of mortality, hospital admissions, or ED visits for studied diseases in communities adjacent to CAFOs. One interpretation of our findings could be that people who reside in such communities may simultaneously be affected by multiple risk factors including low income and education, higher smoking prevalence, and lower access to medical care. Nonetheless, after adjusting for such co-factors or comparing zip codes with similar co-factors, persistently poorer health outcomes were observed in the communities located in zip codes with hog CAFOs. Furthermore, the DiSC analysis demonstrated a higher risk of poorer health outcomes in closer proximity to the CAFO. Our sensitivity analysis showed that patterns of use of medical care among the residents of these North Carolina communities may also contribute to the differences in health outcomes. For example, residents of rural North Carolina areas (where most of the hog CAFOs are located) are more likely to use EDs when searching for medical assistance and less likely use hospitals (due to problems with access such as transportation issues, problems with medical insurance coverage, or behavioral patterns of preferring EDs to a staying in a hospital).

The limitations of this study include: i) a lack of individual measurements of exposure, co-factors, and potential biomarkers of exposure; ii) potential misclassification of exposure from spray fields, accounting for weather, season and wind direction, exposure to poultry facilities, and coal power plants; iii) limited list of population characteristics in currently available dataset to match the compared population groups; and iv) potentially different residential and occupational locations for the same person. Further studies must address these limitations. The problems of identifying potential causative agents and evaluation of dose-response relationships in hog CAFOs studies are discussed in the literature; it is difficult to account for all required factors in occupational health studies, but the detection of specific exposures and diseases in residential communities is even more challenging due to additional complexities caused by dispersion of environmental agents, different exposure pathways, and variability of individual susceptibility to contaminants [6].

Community based research has been gaining prominence as a source of information for medical decision-making. It has been recognized that detailed individual-level data on co-factors are rarely available in the US; therefore, opportunities for individual-level analyses that account for multiple risk factors are very limited. To obtain information on health outcomes in certain populations, public health specialists and policymakers have begun to shift their attention from an exclusive focus on individual-level studies toward community level analyses. When contributions of specific risk factors to health outcomes in communities can be evaluated, this information can be used for optimization of resource allocation for medical interventions designed to improve health outcomes [47].

TABLE S5.
Age-Adjusted Cause-Specific Rates (per 100,000) of Mortality, Hospital Admissions, and ED Visits in Communities Located in Zip Codes with > 215hogs/km^2 (Study Group 2) and in Communities Located in Zip Codes Matched by Percent of African Americans, Percent of Children and Adults Aged 65+ in Population, and Median Household Income (Matched Group A) and Additionally Matched by Percent of the Residents Aged 25+ with Bachelor or Higher Degree (Matched Group B), NC, 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

This table is available in its entirety in the online edition of the NCMJ.

*Statistically significant difference when compared to Study group 2.
n/a, non-applicable.

Conclusion

Southeastern North Carolina communities located in close proximity to hog CAFOs are characterized by poor indicators of health that are not solely due to the impact of converging demographic, socioeconomic, behavioral, and access-to-care factors, but are also due to the additional impact of multiple hog CAFOs located in this area. Although causality with specific exposures from hog CAFOs was not established, our findings suggest research is needed in environmental factors that may influence these outcomes. In addition, these findings suggest an immediate need for improved screening, diagnosis, and intervention for conditions including infant mortality and LBW infants that were found to be overrepresented in these communities. Poor health outcomes in North Carolina communities adjacent to hog CAFOs may also need to be addressed by improving access to medical resources, and future studies to determine the contribution of factors that influence these outcomes are needed. NCMJ

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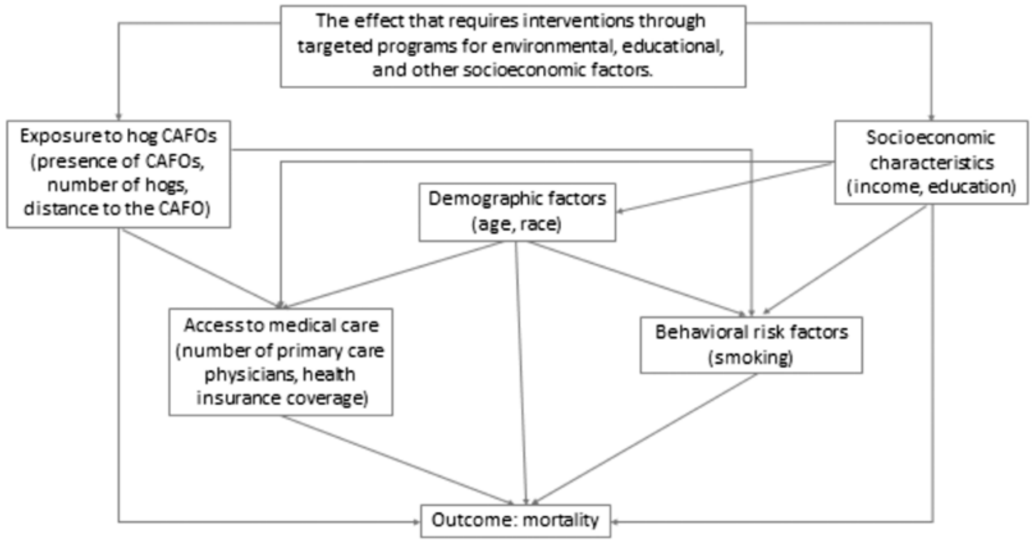
APPENDIX 1.
The International Classification of Diseases (ICD) Codes Used in the Analysis

ICD-9 codes (used for analysis of HCUP data)	
280-285	Anemia (includes Iron deficiency anemias, Other deficiency anemias, Hereditary hemolytic anemias, Acquired hemolytic anemias, Aplastic anemia and other bone marrow failure syndromes, Other and unspecified anemias)
580-589	Kidney disease (Nephritis, Nephrotic Syndrome, and Nephrosis)
010-018	Tuberculosis
038	Septicemia, 995.91 – Sepsis
V21.3	Low birth weight
ICD-10 codes (used for analysis of Multiple Cause of Death data)	
D50-D53, D55-D59, D60-D64	Anemia (includes Nutritional anemias, Hemolytic anemias, Aplastic and other anemias and other bone marrow failure syndromes)
N00-N19	Kidney disease (includes Glomerular diseases, Renal tubule-interstitial diseases, Acute kidney failure and chronic kidney disease)
A15-A19	Tuberculosis
A40, A41	Septicemia (includes Streptococcal sepsis, Other sepsis)
P07.1	Low birth weight newborn
P00-P96	Conditions originating in perinatal period
P00-P04	Newborns affected by maternal trauma
P10-P15	Disorders related to length of gestation and fetal growth

79502 APPENDIX 2.

FIGURE S1.
Illustration of the Relations in the Assessment of Potential Impact Factors-Outcome Associations



APPENDIX 3.
Sensitivity Analysis

2a) Proc Genmod was used for GEE analysis

2b). The propensity score for matching zip codes without CAFO to zip codes with > 215hogs/km^2 (Study group 2) was evaluated using the percent of African Americans, percent of children and people aged 65+ among the residents, as well as median household income, and percent of people with a bachelor's or higher degree. The greedy matching algorithm [37] was used to match zip codes with close propensity scores.

The Matched group A included 56 zip codes that were matched by using the percent of African Americans, percent of children (aged 0-19) and people aged 65+ among the residents, and median household income. The Matched group B included 55 zip codes matched by above listed characteristics of Matched group A and additionally by the percent of people with a bachelor's or higher degree. Characteristics of matched zip codes (i.e., the results on balancing the variables in the matched groups) for the Matched group A and Matched group B are presented in Table S1.

79502 APPENDIX 4.

TABLE S1.
Characteristics of Matched Group A, Matched Group B, and Study Group 2, NC, 2007-2013

Variable	Matching design 1 Mean[1]±SE (95%CI)		Matching design 2 Mean[1]±SE (95%CI)	
	Matched group A	**Study group 2**	**Matched group B**	**Study group 2**
% of African-Americans	28.4±2.9% (22.8%-34.1%)	28.92±1.8% (25.5%-32.4%)	27.1%±3.0% (21.2%-33.0%)	28.9%±1.8% (25.4%-32.4%)
% of children (0-19 years old)	26.8±0.6% (25.7%-27.9%)	27.1±0.4% (26.2%-27.9%)	25.5%±0.6% (24.4%-26.6%)	27.3%±0.4% (26.3%-28.0%)
% of adults (65+ years)	14.0±0.6% (12.9%-15.2%)	14.3±0.4% (13.5%-15.2%)	15.0%±0.6% (13.8%-16.2%)	14.3%±0.4% (13.5%-15.2%)
Median household income (US dollars)	\$35,640±\$1,118 (\$33,450-\$37,831)	\$36,521±\$919 (\$34,719-%38,322)	\$34,933±\$1,161 (\$32,658-\$37,208)	\$36,527±\$936 (\$34,693-\$38,362)
% of people with bachelor or higher degree education among those aged 25+ years	n/a	n/a	9.16%±0.8% (7.7%-10.7%)	11.1%±0.5% (10.2%-12.0%)

[1]Means are evaluated without weights representing zip-code populations.
n/a, non-applicable.

Then, age-adjusted total mortality rate and cause-specific rates of mortality, hospital admissions, and ED visits were compared between Matched group A and B and Study group 2 for underlying cause of death or primary diagnosis and for underlying-plus-secondary cause of death or primary-plus-secondary diagnosis. As shown in Table S5, mortality rates for total mortality and anemia and kidney as underlying causes were higher in Study group 2 than in Matched group A and B. Also, mortality rates of anemia, kidney disease, tuberculosis, and septicemia were higher in Study group 2 than in both matched groups for these diseases as underlying-plus-secondary causes of death. Hospital admission and ED visit rates were higher in Study group 2 than in Matched group A and B for kidney disease and tuberculosis (for primary diagnoses and for primary-plus-secondary diagnoses). ED visits rate for children with LBW also was higher in Study group 2 than in both matched groups (for primary-plus-secondary diagnosis).

TABLE S2.
Descriptive Table of the 3 Studied Groups of NC Communities with and without the Hog Concentrated Feeding Animal Operations (CAFOs): Race-Specific Population Groups, Socioeconomic Characteristics, Smoking Prevalence, and Access-To-Care Characteristics, NC, 2007-2013

Characteristics	NC communities with hog CAFOs (Study group 1)	NC communities with > 215hogs/km^2 (Study group 2)	NC communities without hog CAFOs (Control group)
Race (%):			
White	63.9%**	58.3%**	73.7%
African-American (AA)	28.8%*	31.3%**	19.3%
American Indian	2.4%*	4.1%**	0.8%
Asian	0.8%**	0.3%**	2.5%
Other	4.1%	6.0%**	3.7%
Median household income	$39,005**	$36,520**	$46,414
Bachelor or higher degree education	16.5%**	13.7%**	24.2%
Availability of primary care providers (per 100,000 population)	54**	51**	76
Percent of uninsured individuals	18.2%	18.5%	17.8%
Smokers prevalence among those aged 24+ years old	24.4%	25.9%**	24.0%

*P < 0.05.
**P < 0.001.

79502 APPENDIX 6.

TABLE S3.
Person-Years of Observations in Race-Specific Groups of the Residents of NC Communities from the 3 Studied Groups, NC, 2007-2013

Race	NC communities with hog CAFOs (Study group 1)	NC communities with > 215hogs/km^2 (Study group 2)	NC communities without hog CAFOs (Control group)
White	10,054,073	1,588,477	36,675,276
African-American (AA)	4,528,375	851,839	9,593,021
American Indian	370,901	111,226	411,900
Asian	129,901	8,574	1,242,243
Other	642,425	162,896	1,870,849

FIGURE S2.
Mortality Rates among Patients with Co-Existing Anemia, Kidney Disease, and Septicemia: The US Average, NC Average, and NC Communities with > 215hogs/km² (Study Group 2), 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

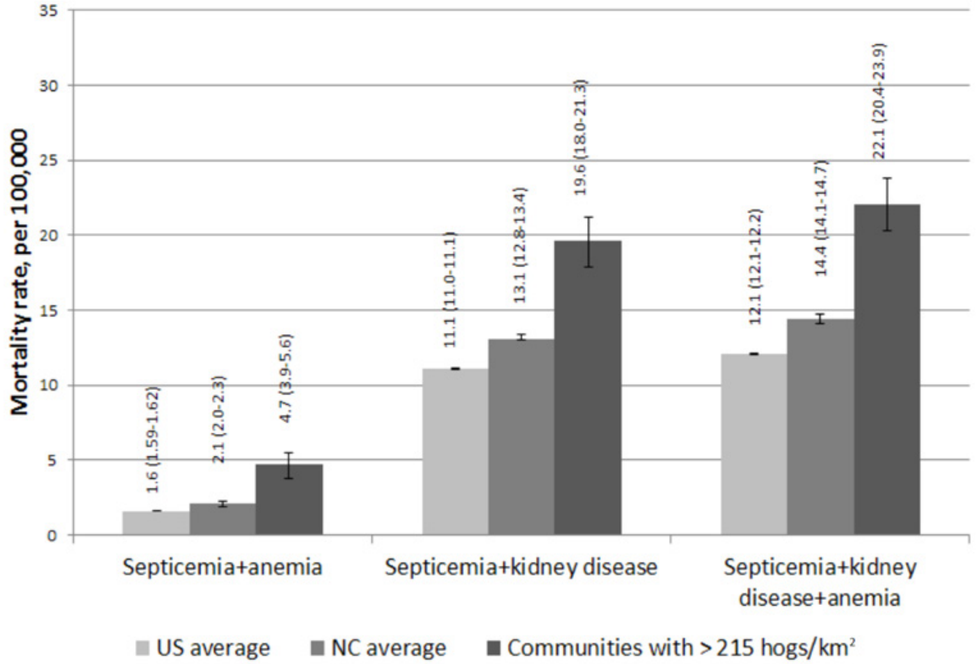


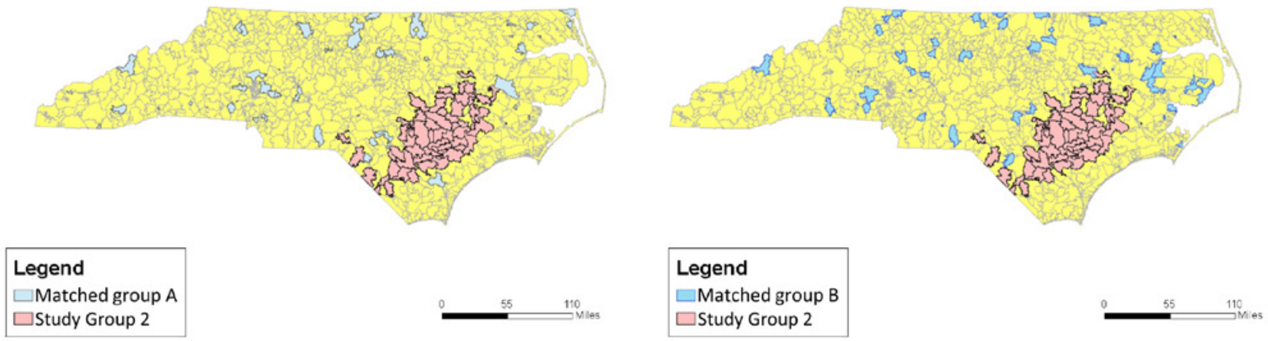
TABLE S4.
Age-Adjusted Mortality Rates (per 100,000) in NC Communities with > 215hogs/km^2 (Study Group 2): Ranks of This Area among the US States and District of Columbia with the Highest Mortality, 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

Disease, cause of mortality	NC communities with > 215hogs/km^2	Rank of the area with > 215hogs/km^2 among the US states with the highest mortality	The US states (with their current respective ranks)[a] with mortality rates closest to the rates of the area with > 215hogs/km^2	
All-cause mortality	934 (922.7-944.8)	#4	#3 Alabama	940 (936.7-943.1)
Anemia:				
• as underlying cause,	2.6 (2.1-3.2)	#1	#1 Mississippi	2.3 (2.1-2.5)
• as underlying+secondary cause	**35.5** (33.4-37.7)	#1	#1 West Virginia	24.4 (23.7-25.2)
Kidney disease:				
• as underlying cause,	24.8 (23.0-26.6)	#2	#1 Louisiana	26.2 (25.7-26.8)
• as underlying+secondary cause	**119** (114.6-122.5)	#1	#1 West Virginia	96.2 (94.7-97.7)
Tuberculosis:				
• as underlying+secondary cause	0.63 (0.32-0.81)	#3	#2 District of Columbia	0.73 (0.49-1.04)
Septicemia:				
• as underlying cause,	16.6 (15.1-18.1)	#7	#6 Alabama	17.0 (16.6-17.4)
• as underlying+secondary cause	75.1 (71.9-78.2)	#2	#1 District of Columbia	83.6 (80.7-86.4)

[a]Mortality rates were calculated using the Multiple Cause of Death data from the Centers for Disease Control and Prevention (https://wonder.cdc.gov/mcd.html).

79502 APPENDIX 9.

FIGURE S3.
Locations of Matched NC Zip Codes without Hog CAFOs (Matched Group A and Matched Group B) and Locations of Zip Codes with > 215hogs/km² (Study Group 2)



79502 APPENDIX 10.

TABLE S5.

Age-Adjusted Cause-Specific Rates (per 100,000) of Mortality, Hospital Admissions, and ED Visits in Communities Located in Zip Codes with > 215hogs/km^2 (Study Group 2) and in Communities Located in Zip Codes Matched by Percent of African Americans, Percent of Children and Adults Aged 65+ in Population, and Median Household Income (Matched Group A) and Additionally Matched by Percent of the Residents Aged 25+ with Bachelor or Higher Degree (Matched Group B), NC, 2007-2013. (95% Confidence Intervals Are Shown in the Parentheses)

		Underlying cause/Primary diagnosis			Underlying+secondary cause/ Primary+secondary diagnosis		
Outcome	**Disease**	**Study group 2**	**Matched group A**	**Matched group B**	**Study group 2**	**Matched group A**	**Matched group B**
Mortality	Total mortality	**934** (922.7-944.8)	**867*** (857.9-875.3)	**920*** (908.6-930.8)	**934** (922.7-944.8)	**867*** (857.9-875.3)	**920*** (908.6-930.8)
	Anemia	**2.65** (2.2-3.2)	**2.1*** (1.6-2.5)	**1.8*** (1.3-2.2)	**35.5** (33.4-37.7)	**20.6*** (19.2-21.9)	**24.1*** (22.3-25.9)
	Kidney disease	**24.8** (23.0-26.6)	**20.9*** (19.6-22.3)	**22.5*** (20.7-24.2)	**119** (114.6-122.5)	**90.1*** (87.2-92.9)	**107*** (103.3-110.9)
	Tuberculosis	0.21 (0.04-0.38)	0.11 (0.01-0.20)	**0.04*** (0.04-0.13)	**0.55** (0.28-0.82)	**0.25*** (0.10-0.40)	**0.24*** (0.06-0.42)
	Septicemia	16.6 (15.1-18.1)	15.9 (14.7-17.1)	16.7 (15.2-18.2)	**75.1** (72.0-78.2)	**62.7*** (60.3-65.0)	**67.6*** (64.6-70.6)
Hospital	Anemia	113 (108.6-116.4)	116 (112.3-118.6)	**141*** (136.3-145.3)	2,179 (2,162-2,196)	**1,867*** (1,854-1,880)	2,165 (2,148-2,183)
	Kidney disease	**187** (181.6-191.4)	**152*** (148.5-155.8)	**175*** (170.4-180.1)	**2,031** (2,015-1,2048)	**1,713*** (1,701-1,725)	**1,864*** (1,848-1,880)
	Tuberculosis	**3.1** (2.4-3.7)	**1.7*** (1.4-2.1)	**0.86*** (0.51-1.21)	**6.2** (5.3-7.2)	**3.7*** (3.2-4.3)	**2.4*** (1.9-3.0)
	Sepsis	313.1 (306.7-319.5)	**272*** (267.4-277.2)	**324*** (317.2-330.4)	468 (460.3-475.9)	**396*** (390.4-402.2)	466 (458.4-474.3)
	Low birth weight	n/a	n/a	n/a	2.5 (1.9-3.1)	**1.5*** (1.2-1.9)	2.3 (1.7-2.9)
ED visits	Anemia	85.4 (81.9-88.9)	88.5 (85.8-91.3)	**115*** (111.0-119.3)	682 (672.2-691.7)	**570*** (563.0-577.0)	**729*** (718.6-739.0)
	Kidney disease	**33.2** (31.1-35.3)	**25.1*** (23.6-26.6)	31.7 (29.6-33.8)	643 (634.0-652.3)	**517*** (510.7-524.2)	633 (623.7-642.3)
	Tuberculosis	**0.32** (0.11-0.53)	**0.15*** (0.04-0.25)	**0.08*** (0.03-0.18)	**1.4** (1.0-1.9)	**0.89*** (0.62-1.17)	**0.61*** (0.32-0.90)
	Sepsis	20.1 (18.5-21.7)	**12.1*** (11.1-13.2)	21.3 (19.6-23.0)	35.5 (33.3-37.6)	**20.1*** (18.7-21.4)	33.1 (31.0-35.2)
	Low birth weight	n/a	n/a	n/a	**4.7** (3.9-5.5)	**1.04*** (0.74-1.34)	**1.9*** (1.4-2.5)

*Statistically significant difference when compared to Study group 2.
n/a, non-applicable.

ATTACHMENT 16

FIPS	state	county	corn	fodder	soybeans	wheat	sugar_crops	other_grain	non_food	other_crops	nuts_and_seeds	fruit	vegetables	beans_and_peas	cattle	swine	poultry	other_livestock	total
1001	alabama	autauga	0.03	0.39	0.01	0.01	0.00	0.00	0.05	0.01	0.01	0.00	0.00	0.00	0.32	0.00	0.00	0.09	0.92
1003	alabama	baldwin	0.15	1.06	0.21	0.11	0.00	0.07	0.18	0.26	0.90	0.00	0.07	0.03	0.73	0.01	0.26	0.30	4.35
1005	alabama	barbour	0.79	0.62	0.08	0.04	0.00	0.04	0.07	0.02	0.05	0.00	0.00	0.01	0.32	0.06	1.53	0.04	3.66
1007	alabama	bibb	0.04	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.07	0.04	0.50
1009	alabama	blount	3.62	0.63	0.44	0.18	0.00	0.15	0.07	0.01	0.01	0.00	0.01	0.00	1.29	0.01	6.55	0.30	13.28
1011	alabama	bullock	0.20	0.36	0.02	0.01	0.00	0.01	0.01	0.01	0.01	0.00	0.00	0.02	0.17	0.00	0.41	0.04	1.26
1013	alabama	butler	0.77	0.16	0.07	0.04	0.00	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.14	0.06	1.35	0.05	2.69
1015	alabama	calhoun	1.12	0.20	0.15	0.06	0.00	0.07	0.05	0.03	0.00	0.00	0.00	0.00	0.48	0.12	1.87	0.21	4.37
1017	alabama	chambers	0.04	0.32	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.02	0.07	0.07	1.01
1019	alabama	cherokee	1.51	0.19	0.36	0.14	0.00	0.07	0.34	0.09	0.00	0.00	0.01	0.02	0.51	0.12	2.73	0.14	6.23
1021	alabama	chilton	0.07	0.45	0.01	0.01	0.00	0.00	0.04	0.02	0.01	0.01	0.00	0.00	0.46	0.13	0.09	0.20	1.50
1023	alabama	choctaw	0.02	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.03	0.03	0.29
1025	alabama	clarke	0.01	0.22	0.00	0.00	0.00	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.05	0.00	0.00	0.04	0.37
1027	alabama	clay	0.95	0.23	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.09	1.79	0.06	3.71
1029	alabama	cleburne	1.40	0.14	0.13	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.28	0.01	2.54	0.07	4.71
1031	alabama	coffee	2.54	0.47	0.25	0.15	0.00	0.15	0.18	0.01	0.18	0.00	0.00	0.00	0.57	0.08	4.50	0.11	9.20
1033	alabama	colbert	1.04	0.36	0.25	0.23	0.00	0.03	0.08	0.25	0.00	0.02	0.02	0.02	0.73	0.00	1.16	0.18	4.37
1035	alabama	conecuh	0.04	0.22	0.01	0.00	0.00	0.00	0.08	0.00	0.02	0.00	0.00	0.00	0.15	0.00	0.18	0.03	0.73
1037	alabama	coosa	0.04	0.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.06	0.03	0.42
1039	alabama	covington	0.55	0.51	0.05	0.03	0.00	0.03	0.14	0.02	0.06	0.00	0.00	0.00	0.40	0.00	1.33	0.07	3.18
1041	alabama	crenshaw	1.45	0.26	0.13	0.08	0.00	0.06	0.02	0.00	0.02	0.00	0.00	0.03	0.44	0.00	2.59	0.04	5.12
1043	alabama	cullman	5.18	0.44	0.67	0.27	0.00	0.21	0.03	0.00	0.00	0.00	0.04	0.00	2.20	0.01	11.12	0.49	20.68
1045	alabama	dale	1.36	0.46	0.13	0.07	0.00	0.06	0.14	0.01	0.22	0.00	0.00	0.00	0.24	0.00	2.51	0.07	5.28
1047	alabama	dallas	0.05	0.47	0.07	0.03	0.00	0.01	0.10	0.01	0.02	0.00	0.00	0.01	0.40	0.00	0.04	0.07	1.30
1049	alabama	de kalb	7.12	0.59	0.87	0.37	0.00	0.30	0.03	0.03	0.01	0.02	0.02	0.05	2.06	5.59	13.21	0.42	30.68
1051	alabama	elmore	0.06	0.46	0.02	0.01	0.00	0.00	0.15	0.01	0.01	0.00	0.00	0.03	0.30	0.02	0.07	0.16	1.29
1053	alabama	escambia	0.02	0.43	0.01	0.01	0.00	0.01	0.09	0.01	0.14	0.00	0.00	0.01	0.20	0.00	0.00	0.06	1.00
1055	alabama	etowah	1.75	0.25	0.23	0.09	0.00	0.07	0.10	0.01	0.00	0.00	0.01	0.00	0.73	0.01	3.09	0.29	6.63
1057	alabama	fayette	0.33	0.18	0.07	0.02	0.00	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.28	0.01	0.47	0.05	1.46
1059	alabama	franklin	1.49	0.30	0.18	0.09	0.00	0.06	0.02	0.01	0.00	0.00	0.00	0.00	0.68	0.01	2.74	0.15	5.73
1061	alabama	geneva	1.55	0.36	0.16	0.09	0.00	0.09	0.22	0.02	0.28	0.00	0.00	0.00	0.57	0.00	2.59	0.09	6.04
1063	alabama	greene	0.00	0.21	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.02	0.42
1065	alabama	hale	0.05	0.23	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.06	0.07	0.04	0.91
1067	alabama	henry	0.34	0.31	0.04	0.02	0.00	0.03	0.15	0.02	0.18	0.00	0.00	0.01	0.25	0.01	0.67	0.05	2.09
1069	alabama	houston	0.13	0.85	0.03	0.02	0.00	0.05	0.44	0.03	0.96	0.01	0.01	0.00	0.76	0.12	0.09	0.18	3.69
1071	alabama	jackson	1.48	0.37	0.45	0.09	0.00	0.05	0.05	0.02	0.00	0.04	0.02	0.02	1.17	0.11	2.22	0.38	6.46
1073	alabama	jefferson	0.11	0.63	0.01	0.01	0.00	0.01	0.01	0.01	0.00	0.00	0.00	0.00	0.31	0.03	0.19	0.37	1.69
1075	alabama	lamar	0.07	0.16	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.11	0.06	0.55
1077	alabama	lauderdale	0.73	0.46	0.35	0.25	0.00	0.02	0.31	0.15	0.00	0.03	0.00	0.01	1.01	0.01	0.71	0.43	4.48
1079	alabama	lawrence	2.95	0.38	0.54	0.32	0.00	0.11	0.21	0.21	0.01	0.00	0.01	0.01	0.88	0.01	4.45	0.29	10.38
1081	alabama	lee	0.05	0.24	0.01	0.00	0.00	0.00	0.01	0.01	0.01	0.00	0.00	0.00	0.16	0.00	0.08	0.15	0.72
1083	alabama	limestone	1.40	0.39	1.44	0.85	0.00	0.06	0.44	0.41	0.00	0.00	0.01	0.00	1.18	0.13	1.46	0.40	8.17
1085	alabama	lowndes	0.63	0.50	0.08	0.03	0.00	0.03	0.02	0.01	0.01	0.00	0.00	0.01	1.07	0.00	1.04	0.07	3.51
1087	alabama	macon	0.01	0.30	0.00	0.00	0.00	0.00	0.04	0.02	0.01	0.00	0.00	0.02	0.15	0.08	0.00	0.07	0.71
1089	alabama	madison	0.86	0.38	0.74	0.39	0.00	0.05	0.65	0.13	0.02	0.06	0.15	0.02	0.95	0.03	0.23	0.47	5.14
1091	alabama	marengo	0.01	0.38	0.01	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.04	0.79
1093	alabama	marion	1.30	0.21	0.14	0.07	0.00	0.05	0.01	0.01	0.00	0.00	0.00	0.00	0.27	0.01	2.38	0.12	4.57
1095	alabama	marshall	3.90	0.42	0.46	0.20	0.00	0.16	0.02	0.03	0.00	0.00	0.00	0.00	1.36	0.36	7.57	0.40	14.88
1097	alabama	mobile	0.07	0.71	0.04	0.03	0.00	0.00	0.12	0.01	0.22	0.00	0.00	0.00	0.75	0.01	0.01	0.29	2.27
1099	alabama	monroe	0.04	0.19	0.02	0.02	0.00	0.01	0.20	0.00	0.16	0.00	0.00	0.00	0.20	0.06	0.04	0.04	0.99
1101	alabama	montgomery	0.54	0.87	0.07	0.04	0.00	0.03	0.08	0.01	0.02	0.00	0.00	0.04	2.77	0.00	0.85	0.27	5.58
1103	alabama	morgan	2.34	0.61	0.41	0.16	0.00	0.12	0.08	0.07	0.00	0.03	0.05	0.01	1.75	0.05	3.56	0.42	9.68
1105	alabama	perry	0.02	0.23	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.02	0.51	0.00	0.00	0.04	0.90
1107	alabama	pickens	0.89	0.29	0.10	0.05	0.00	0.04	0.02	0.00	0.01	0.00	0.00	0.00	0.21	0.06	1.69	0.07	3.42
1109	alabama	pike	0.96	0.27	0.09	0.05	0.00	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.43	0.06	1.79	0.06	3.81
1111	alabama	randolph	1.35	0.17	0.13	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.55	0.00	2.78	0.06	5.17
1113	alabama	russell	0.07	0.41	0.01	0.02	0.00	0.00	0.04	0.02	0.04	0.00	0.00	0.00	0.20	0.12	0.08	0.14	1.14
1117	alabama	shelby	0.04	0.45	0.06	0.01	0.00	0.00	0.17	0.08	0.00	0.00	0.02	0.00	0.34	0.01	0.01	0.33	1.50
1115	alabama	st clair	1.19	0.22	0.13	0.06	0.00	0.05	0.01	0.02	0.00	0.00	0.01	0.00	0.35	0.13	2.02	0.20	4.38
1119	alabama	sumter	0.08	0.25	0.01	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.38	1.03	0.10	0.04	1.93
1121	alabama	talladega	0.39	0.31	0.11	0.02	0.00	0.01	0.09	0.05	0.00	0.00	0.01	0.03	0.42	0.00	0.47	0.20	2.12
1123	alabama	tallapoosa	0.11	0.34	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.17	0.08	0.88
1125	alabama	tuscaloosa	0.48	0.28	0.19	0.03	0.00	0.01	0.08	0.05	0.00	0.00	0.00	0.00	0.28	0.00	0.73	0.25	2.38
1127	alabama	walker	0.57	0.47	0.06	0.03	0.00	0.02	0.00	0.04	0.00	0.00	0.01	0.00	0.35	0.01	1.00	0.27	2.83
1129	alabama	washington	0.07	0.18	0.01	0.00	0.00	0.01	0.00	0.01	0.03	0.00	0.00	0.00	0.11	0.01	0.19	0.05	0.67
1131	alabama	wilcox	0.00	0.28	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.18	0.00	0.00	0.04	0.53
1133	alabama	winston	0.57	0.19	0.05	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.28	0.00	1.15	0.13	2.44
4001	arizona	apache	0.13	0.02	0.01	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.05	0.00	0.44	1.05
4003	arizona	cochise	0.60	0.59	0.04	0.03	0.00	0.06	0.02	0.03	0.20	0.01	0.03	0.02	1.01	0.36	0.01	0.08	3.10
4005	arizona	coconino	0.26	0.02	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.68	0.07	0.00	0.28	1.36
4007	arizona	gila	0.11	0.00	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.06	0.45
4009	arizona	graham	0.08	0.16	0.01	0.10	0.00	0.03	0.20	0.00	0.04	0.02	0.01	0.01	0.14	0.01	0.00	0.02	0.82
4011	arizona	greenlee	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.00	0.00	0.01	0.27
4012	arizona	la paz	0.07	0.90	0.01	0.08	0.00	0.00	0.22	0.02	0.00	0.00	0.02	0.01	0.30	0.00	0.00	0.00	1.64
4013	arizona	maricopa	8.90	8.56	0.95	2.45	0.02	1.58	2.32	0.16	0.04	0.09	0.65	0.06	45.10	6.03	0.01	1.89	78.81
4015	arizona	mohave	0.51	0.17	0.05	0.03	0.00	0.02	0.07	0.00	0.00	0.00	0.00	0.00	1.66	0.03	0.00	0.04	2.58
4017	arizona	navajo	0.10	0.03	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.27	0.00	0.00	0.29	0.72
4019	arizona	pima	0.58	0.42	0.06	0.14	0.00	0.14	0.32	0.00	0.01	0.02	0.00	0.03	1.97	0.95	0.01	0.38	5.00
4021	arizona	pinal	8.10	6.39	0.68	1.56	0.00	1.70	4.98	0.09	0.14	0.05	0.39	0.02	30.10	0.08	0.01	0.55	54.83
4023	arizona	santa cruz	0.16	0.00	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.35	0.00	0.00	0.03	0.59
4025	arizona	yavapai	0.29	0.02	0.03	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.74	0.01	0.01	0.10	1.27
4027	arizona	yuma	0.39	0.96	0.06	0.20	0.00	0.04	0.21	0.47	0.01	0.12	0.52	0.02	1.27	0.01	0.00	0.03	4.31
5001	arkansas	arkansas	0.69	0.05	2.56	0.36	0.00	2.88	0.00	0.10	0.00	0.00	0.01	0.82	0.08	0.05	0.00	0.01	7.60
5003	arkansas	ashley	0.33	0.19	0.39	0.03	0.00	0.26	0.36	0.08	0.00	0.00	0.01	0.27	0.05	0.00	0.00	0.03	2.02
5005	arkansas	baxter	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.00	0.00	0.07	0.37
5007	arkansas	benton	3.70	0.63	0.35	0.20	0.00	0.15	0.02	0.00	0.00	0.04	0.00	0.00	2.49	0.06	7.95	0.23	15.81
5009	arkansas	boone	0.36	0.21	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.76	0.00	0.73	0.13	2.26
5011	arkansas	bradley	0.14	0.04	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.02	0.00	0.44	0.02	0.73
5013	arkansas	calhoun	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.05	0.08	0.01	0.22
5015	arkansas	carroll	1.49	0.21	0.14	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.17	0.01	3.51	0.11	6.80
5017	arkansas	chicot	0.81	0.07	2.61	0.29	0.00	1.42	0.54	0.02	0.00	0.00	0.07	0.35	0.07	0.00	0.00	0.01	6.27
5019	arkansas	clark	0.02	0.09	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.14	0.05	0.14	0.04	0.61
5021	arkansas	clay	0.77	0.77	2.28	0.14	0.00	3.05	1.09	0.09	0.01	0.13	0.01	0.65	0.11	0.00	0.16	0.05	9.31
5023	arkansas	cleburne	0.30	0.14	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.01	0.62	0.10	1.68
5025	arkansas	cleveland	0.87	0.03	0.08	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.01	0.05	0.05	1.62	0.01	2.83
5027	arkansas	columbia	0.27	0.02	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.53	0.03	1.01
5029	arkansas	conway	1.43	0.31	0.37	0.18	0.00	0.14	0.01	0.01	0.10	0.00	0.00	0.02	0.69	0.68	2.38	0.09	6.41
5031	arkansas	craighead	0.99	0.79	2.82	0.20	0.00	3.38	2.60	0.09	0.02	0.07	0.01	0.81	0.09	0.01	0.00	0.08	11.97
5033	arkansas	crawford	0.12	0.10	0.09	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.26	0.09	1.03
5035	arkansas	crittenden	0.55	0.22	8.26	1.49	0.00	4.83	1.27	0.37	1.41	0.01	0.03	1.83	0.05	0.14	0.00	0.04	20.48
5037	arkansas	cross	0.39	0.23	3.11	0.39	0.00	3.53	0.21	0.13	0.01	0.00	0.02	0.83	0.04	0.00	0.02	0.02	8.92
5039	arkansas	dallas	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.00	0.01	0.08
5041	arkansas	desha	0.83	0.02	2.35	0.17	0.00	0.97	0.47	0.02	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.01	5.44
5043	arkansas	drew	0.30	0.43	0.35	0.02	0.00	0.26	0.09	0.02	0.00	0.00	0.00	0.08	0.08	0.05	0.28	0.03	2.05
5045	arkansas	faulkner	0.04	0.42	0.13	0.08	0.00	0.16	0.00	0.02	0.06	0.00	0.01	0.15	0.77	0.01	0.00	0.26	1.98
5047	arkansas	franklin	0.73	0.39	0.10	0.04	0.00	0.04	0.00	0.00	0.00	0.03	0.00	0.00	0.62	0.00	1.36	0.05	3.35
5049	arkansas	fulton	0.02	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.02	0.00	0.07	0.73
5051	arkansas	garland	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.07	0.14	0.09	0.47
5053	arkansas	grant	0.10	0.12	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.17	0.05	0.69
5055	arkansas	greene	0.61	0.18	1.53	0.19	0.00	3.49	0.34	0.06	0.01	0.07	0.02	0.36	0.19	0.01	0.00	0.08	7.12
5057	arkansas	hempstead	1.20	0.37	0.11	0.06	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.61	0.17	2.48	0.06	5.13
5059	arkansas	hot spring	0.02	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.01	0.45	0.10	0.99
5061	arkansas	howard	1.00	0.13	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.86	1.89	0.04	4.57
5063	arkansas	independence	0.56	0.39	0.34	0.06	0.00	0.37	0.00	0.03	0.07	0.00	0.00	0.15	0.54	0.00	0.85	0.12	3.48
5065	arkansas	izard	0.27	0.17	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.05	0.44	0.07	1.39
5067	arkansas	jackson	0.36	0.04	2.37	0.35	0.00	3.60	0.02	0.17	0.05	0.00	0.04	0.78	0.09	0.00	0.02	0.03	7.94
5069	arkansas	jefferson	0.92	0.08	1.72	0.38	0.00	1.89	0.13	0.09	0.03	0.00	0.01	0.63	0.04	0.00	0.47	0.04	6.43
5071	arkansas	johnson	0.58	0.10	0.07	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.05	1.06	0.06	2.36
5073	arkansas	lafayette	0.78	0.40	0.13	0.08	0.00	0.09	0.01	0.01	0.03	0.00	0.00	0.09	0.36	0.05	1.09	0.02	3.15
5075	arkansas	lawrence	0.33	0.19	0.90	0.12	0.00	1.94	0.00	0.04	0.07	0.00	0.00	0.44	0.32	0.06	0.43	0.06	4.85
5077	arkansas	lee	0.74	0.07	3.05	0.60	0.00	2.15	1.46	0.08	0.01	0.00	0.00	0.60	0.03	0.05	0.02	0.00	8.88
5079	arkansas	lincoln	1.35	0.07	1.42	0.22	0.00	0.96	0.39	0.04	0.00	0.00	0.05	0.39	0.15	0.05	1.58	0.02	6.70
5081	arkansas	little river	0.41	0.33	0.11	0.07	0.00	0.02	0.00	0.01	0.01	0.00	0.00	0.02	0.47	0.04	0.65	0.06	2.21
5083	arkansas	logan	1.17	0.18	0.15	0.06	0.00	0.05	0.01	0.01	0.00	0.05	0.00	0.00	0.76	0.05	2.06	0.06	4.62
5085	arkansas	lonoke	1.11	0.30	2.66	0.45	0.00	3.43	0.22	0.07	0.09	0.00	0.01	0.64	0.30	0.00	0.14	0.09	9.52
5087	arkansas	madison	0.93	0.23	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.02	0.00	0.00	0.63	0.01	1.90	0.11	4.00
5089	arkansas	marion	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00	0.03	0.09	0.57
5091	arkansas	miller	0.26	1.01	0.22	0.16	0.00	0.04	0.00	0.09	0.18	0.00	0.00	0.05	0.43	0.05	0.22	0.06	2.79
5093	arkansas	mississippi	0.70	0.07	4.77	0.70	0.00	2.82	4.24	0.41	0.10	0.55	0.02	1.53	0.02	0.00	0.00	0.02	15.94
5095	arkansas	monroe	0.69	0.02	2.13	0.42	0.00	2.35	0.10	0.10	0.00	0.00	0.00	0.25	0.01	0.05	0.01	0.01	6.14
5097	arkansas	montgomery	0.20	0.05	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.25	0.42	0.02	1.14
5099	arkansas	nevada	0.30	0.09	0.11	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.16	0.05	0.64	0.02	1.41
5101	arkansas	newton	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.17	0.08	0.12	0.09	0.53
5103	arkansas	ouachita	0.16	0.08	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.01	0.28	0.03	0.61
5105	arkansas	perry	0.26	0.08	0.17	0.03	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.08	0.17	0.06	0.49	0.05	1.42
5107	arkansas	phillips	0.92	0.14	4.07	0.59	0.00	1.78	0.68	0.18	0.01	0.00	0.01	0.88	0.02	0.00	0.01	0.01	9.30
5109	arkansas	pike	0.27	0.07	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.22	0.76	0.02	1.61
5111	arkansas	poinsett	0.49	0.19	3.36	0.33	0.00	4.18	0.81	0.09	0.04	0.00	0.01	1.55	0.04	0.00	0.02	0.02	11.13
5113	arkansas	polk	0.77	0.12	0.08	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.11	1.47	0.09	3.00
5115	arkansas	pope	0.91	0.23	0.14	0.07	0.00	0.10	0.00	0.00	0.06	0.00	0.00	0.01	0.41	0.47	1.62	0.12	4.14
5117	arkansas	prairie	0.37	0.22	2.17	0.15	0.00	2.55	0.04	0.04	0.04	0.00	0.00	0.31	0.07	0.05	0.00	0.04	6.05
5119	arkansas	pulaski	0.34	1.11	0.71	0.27	0.00	0.39	0.00	0.02	0.10	0.00	0.00	0.08	0.24	0.03	0.20	0.22	3.70
5121	arkansas	randolph	0.21	0.26	0.36	0.03	0.00	0.84	0.00	0.03	0.02	0.00	0.00	0.14	0.47	0.01	0.18	0.10	2.67
5125	arkansas	saline	0.01	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.01	0.00	0.15	0.56
5127	arkansas	scott	0.90	0.11	0.09	0.05	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.26	0.05	1.82	0.04	3.39
5129	arkansas	searcy	0.01	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.03	0.00	0.07	0.52
5131	arkansas	sebastian	0.94	0.42	0.20	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.01	0.71	0.00	1.59	0.10	4.07
5133	arkansas	sevier	1.01	0.12	0.09	0.05	0.00	0.04	0.00	0.00	0.04	0.00	0.00	0.00	0.34	0.18	1.78	0.05	3.73
5135	arkansas	sharp	0.52	0.15	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.00	1.10	0.07	2.28
5123	arkansas	st francis	0.40	0.16	3.27	0.51	0.00	2.52	0.96	0.07	0.00	0.00	0.01	0.51	0.04	0.00	0.00	0.03	8.46
5137	arkansas	stone	0.26	0.07	0.03	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.00	0.56	0.05	1.30
5139	arkansas	union	0.38	0.02	0.03	0.04	0.00	0.02	0.01	0.09	0.00	0.00	0.00	0.00	0.07	0.00	0.46	0.04	1.15
5141	arkansas	van buren	0.09	0.12	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.05	0.14	0.06	0.74
5143	arkansas	washington	3.04	0.53	0.28	0.15	0.00	0.13	0.01	0.00	0.00	0.05	0.00	0.09	1.72	0.09	6.36	0.33	12.70
5145	arkansas	white	0.33	0.65	0.80	0.14	0.00	0.66	0.00	0.02	0.03	0.00	0.00	0.12	0.89	0.02	0.40	0.29	4.37
5147	arkansas	woodruff	0.47	0.02	2.35	0.27	0.00	2.31	0.13	0.07	0.16	0.00	0.01	0.39	0.02	0.00	0.12	0.00	6.31
5149	arkansas	yell	1.47	0.24	0.21	0.09	0.00	0.06	0.01	0.01	0.04	0.00	0.00	0.01	0.39	0.43	2.86	0.08	5.88
6001	california	alameda	2.25	2.12	0.21	0.25	0.00	0.17	0.01	0.00	0.01	0.02	1.08	0.00	8.44	0.01	0.01	0.83	15.42
6003	california	alpine	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.13	0.00	0.19
6005	california	amador	0.28	0.24	0.03	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.97	0.00	0.00	0.19	1.75
6007	california	butte	0.37	5.36	0.05	0.17	0.00	3.58	0.01	0.10	3.55	0.33	0.17	0.02	1.07	0.22	0.00	0.22	15.22
6009	california	calaveras	0.24	0.13	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.04	0.00	0.82	0.01	0.01	0.17	1.46
6011	california	colusa	0.50	2.00	0.05	0.55	0.00	5.43	0.01	0.46	2.98	0.10	0.75	0.15	0.91	0.03	0.26	0.09	14.28
6013	california	contra costa	4.47	3.59	0.25	0.54	0.00	0.34	0.01	0.34	0.26	0.40	0.40	0.00	9.59	0.03	0.00	1.77	22.01
6015	california	del norte	0.10	1.36	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.00	1.86
6017	california	el dorado	0.13	0.17	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.02	0.00	0.18	0.96
6019	california	fresno	15.02	14.12	1.57	7.13	0.11	2.15	4.05	0.23	16.71	17.50	9.14	0.43	51.48	1.15	2.31	8.79	151.91
6021	california	glenn	1.47	3.51	0.13	0.19	0.00	1.91	0.04	0.11	1.31	0.05	0.14	0.02	5.01	0.03	0.00	0.11	14.04
6023	california	humboldt	0.73	0.40	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	2.63	0.01	0.00	0.07	3.99
6025	california	imperial	4.49	8.51	0.44	1.86	0.55	0.26	0.11	0.15	0.01	0.23	1.70	0.00	17.45	0.25	0.00	0.39	36.41
6027	california	inyo	0.10	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.38	0.00	0.00	0.01	0.61
6029	california	kern	19.12	68.84	1.88	3.60	0.01	1.64	2.84	0.10	17.07	7.18	3.21	0.10	73.83	0.21	0.68	3.19	203.50
6031	california	kings	11.62	5.07	0.99	4.76	0.01	1.90	5.58	2.20	5.92	1.19	2.23	0.15	39.33	0.05	0.30	2.83	84.16
6033	california	lake	0.06	0.90	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.20	0.03	0.00	0.09	1.31
6035	california	lassen	0.48	0.72	0.05	0.13	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	1.65	0.01	0.00	0.10	3.21
6037	california	los angeles	9.38	42.91	0.95	0.48	0.00	1.10	0.07	0.08	1.01	1.93	1.18	0.00	21.04	7.42	11.70	27.70	126.95
6039	california	madera	6.03	3.36	0.59	0.73	0.00	0.50	0.11	0.03	5.96	7.42	0.53	0.00	23.19	0.04	0.06	1.95	50.49
6041	california	marin	4.25	0.45	0.40	0.21	0.00	0.19	0.02	0.00	0.00	0.01	0.00	0.00	15.91	0.05	0.01	0.92	22.41
6043	california	mariposa	0.34	0.06	0.03	0.02	0.00	0.01	0.00	0.00	0.04	0.05	0.00	0.00	1.17	0.00	0.00	0.09	1.82
6045	california	mendocino	0.29	0.62	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.99	0.05	0.00	0.14	2.16
6047	california	merced	15.22	7.00	1.34	1.81	0.00	1.98	1.69	0.19	4.74	1.90	2.19	0.01	55.53	0.03	2.37	3.22	99.22
6049	california	modoc	0.22	0.42	0.02	0.11	0.00	0.05	0.00	0.01	0.00	0.00	0.03	0.00	0.76	0.00	0.00	0.03	1.65
6051	california	mono	0.09	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.21	0.10	0.02	0.65
6053	california	monterey	2.34	11.50	0.53	0.18	0.01	0.35	0.13	0.05	0.09	1.92	0.69	0.03	6.07	0.01	0.01	0.60	24.53
6055	california	napa	0.41	1.63	0.04	0.02	0.00	0.02	0.00	0.00	0.01	0.86	0.00	0.00	1.35	0.01	0.01	0.11	4.49
6057	california	nevada	0.11	0.31	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.02	0.00	0.21	1.03
6059	california	orange	0.25	3.19	0.08	0.09	0.00	0.09	0.03	0.00	0.01	0.07	0.05	0.01	0.32	0.02	0.03	0.93	5.18
6061	california	placer	0.52	2.96	0.05	0.07	0.00	0.39	0.00	0.04	0.04	0.01	0.00	0.00	1.81	0.08	0.02	0.55	6.55
6063	california	plumas	0.24	0.09	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.81	0.00	0.00	0.02	1.21
6065	california	riverside	6.64	12.09	0.65	4.12	0.40	0.41	1.26	0.21	0.03	2.96	3.36	0.00	26.79	0.35	5.81	4.15	69.22
6067	california	sacramento	8.41	14.60	0.56	1.07	0.00	1.73	0.03	0.53	0.23	2.85	0.81	0.04	22.26	0.14	0.35	4.14	57.74
6069	california	san benito	0.70	0.68	0.07	0.06	0.00	0.04	0.00	0.00	0.00	0.01	0.03	0.00	2.59	0.62	0.01	0.20	5.03
6071	california	san bernardino	13.47	2.32	1.25	0.68	0.00	0.59	0.06	0.00	0.01	0.01	0.06	0.00	52.72	0.72	5.14	2.53	79.57
6073	california	san diego	3.04	25.13	0.32	0.16	0.00	0.14	0.03	0.01	0.01	0.02	0.04	0.00	9.67	1.27	6.97	4.66	51.45
6075	california	san francisco	0.01	0.14	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.24
6077	california	san joaquin	38.24	56.69	1.56	8.95	0.00	8.22	0.10	2.66	32.79	37.85	16.75	0.94	63.60	2.28	2.16	4.32	277.12
6079	california	san luis obispo	1.42	2.22	0.17	0.10	0.00	0.27	0.02	0.01	0.01	0.39	0.18	0.00	4.94	0.04	0.00	0.60	10.37
6081	california	san mateo	0.22	0.83	0.03	0.01	0.00	0.02	0.00	0.00	0.00	0.01	0.05	0.01	0.71	0.04	0.06	0.66	2.66
6083	california	santa barbara	1.47	3.46	0.35	0.12	0.01	0.11	0.09	0.02	0.05	1.25	0.26	0.02	3.73	0.06	0.00	0.45	11.44
6085	california	santa clara	1.77	3.51	0.21	0.22	0.00	0.10	0.03	0.01	0.01	0.09	0.23	0.01	6.36	0.04	0.03	0.88	13.51
6087	california	santa cruz	1.57	1.82	0.28	0.15	0.01	0.10	0.06	0.04	0.04	0.21	0.25	0.01	1.34	3.20	1.69	0.29	11.08
6089	california	shasta	0.53	1.94	0.05	0.03	0.00	0.02	0.00	0.05	0.00	0.00	0.01	0.00	1.74	0.03	0.00	0.18	4.59
6091	california	sierra	0.20	0.02	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.40	0.18	0.01	1.11
6093	california	siskiyou	0.43	1.41	0.06	0.25	0.00	0.13	0.01	0.01	0.00	0.01	0.06	0.00	1.35	0.02	0.00	0.06	3.80
6095	california	solano	5.18	7.64	0.37	2.77	0.00	1.93	0.02	4.70	1.72	0.69	1.83	0.10	14.43	0.07	0.06	2.62	44.12
6097	california	sonoma	6.98	4.38	0.66	0.35	0.00	0.46	0.03	0.00	0.01	1.35	0.01	0.00	24.13	0.20	2.09	2.48	43.14
6099	california	stanislaus	14.15	11.50	1.35	1.01	0.00	1.38	0.09	0.64	5.53	2.03	0.65	0.07	56.25	2.15	2.37	3.98	103.17
6101	california	sutter	0.60	3.38	0.02	0.74	0.00	7.64	0.02	0.97	2.65	0.55	1.02	0.39	0.70	0.05	0.01	0.19	18.92
6103	california	tehama	1.11	1.70	0.11	0.06	0.00	0.06	0.01	0.00	0.12	0.02	0.03	0.00	3.90	0.07	0.00	0.32	7.52
6105	california	trinity	0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.03	0.00	0.01	0.25
6107	california	tulare	37.64	7.21	3.46	4.18	0.01	2.13	0.83	0.57	7.89	9.07	0.71	0.08	142.82	7.23	0.29	3.67	227.78
6109	california	tuolumne	0.08	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.87	1.29
6111	california	ventura	0.48	14.14	0.17	0.04	0.01	0.05	0.05	0.00	0.03	2.52	0.09	0.00	0.69	0.03	0.02	0.38	18.70

6113	california	yolo	1.51	8.46	0.10	4.51	0.00	5.14	0.01	4.17	4.83	2.34	5.70	0.20	2.63	0.15	0.01	0.50	40.26
6115	california	yuba	0.38	1.44	0.04	0.06	0.00	1.14	0.00	0.08	0.42	0.14	0.14	0.00	1.31	0.02	0.00	0.15	5.34
8001	colorado	adams	1.23	1.25	0.06	9.41	0.02	0.73	0.01	0.35	0.00	0.01	0.53	0.01	1.43	0.03	0.01	0.45	15.54
8003	colorado	alamosa	0.06	0.12	0.01	0.03	0.00	0.04	0.00	0.01	0.00	0.00	0.06	0.00	0.11	0.00	0.00	0.04	0.48
8005	colorado	arapahoe	0.26	0.29	0.03	2.06	0.00	0.19	0.00	0.01	0.00	0.00	0.46	0.00	0.77	0.02	0.01	0.29	4.37
8007	colorado	archuleta	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.02	0.17
8009	colorado	baca	0.29	0.62	0.02	1.11	0.00	0.35	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.29	0.00	0.01	3.10
8011	colorado	bent	0.15	0.23	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.33	0.00	0.03	1.20
8013	colorado	boulder	0.23	0.39	0.02	0.05	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.01	0.20	0.37	1.79
8014	colorado	broomfield	0.04	0.08	0.00	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.36
8015	colorado	chaffee	0.07	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.08	0.43	0.00	0.02	0.68
8017	colorado	cheyenne	0.15	0.37	0.01	0.37	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.31	0.00	0.01	1.48
8019	colorado	clear creek	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02
8021	colorado	conejos	0.08	0.07	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.06	0.41
8023	colorado	costilla	0.03	0.09	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.06	0.00	0.00	0.01	0.22
8025	colorado	crowley	0.33	0.06	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.00	0.00	0.02	1.36
8027	colorado	custer	0.07	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.36	0.00	0.02	0.61
8029	colorado	delta	0.20	0.17	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.01	0.01	0.43	0.01	0.00	0.17	1.07
8031	colorado	denver	0.52	0.00	0.05	0.29	0.00	0.07	0.00	0.00	0.00	0.00	0.42	0.00	0.02	5.76	0.00	0.11	7.25
8033	colorado	dolores	0.01	0.05	0.01	0.07	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.02	0.04	0.00	0.00	0.01	0.23
8035	colorado	douglas	0.20	0.07	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.01	0.01	0.66	1.48
8037	colorado	eagle	0.03	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.09	0.00	0.00	0.03	0.28
8041	colorado	el paso	0.40	0.15	0.04	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.17	0.00	1.23	0.02	0.01	0.57	2.62
8039	colorado	elbert	0.37	0.12	0.04	0.14	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.00	0.81	0.01	0.00	0.22	1.76
8043	colorado	fremont	0.08	0.03	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.26	0.00	0.00	0.14	0.61
8045	colorado	garfield	0.07	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.11	0.47
8047	colorado	gilpin	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09
8049	colorado	grand	0.19	0.06	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.89	0.00	0.08	1.51
8051	colorado	gunnison	0.07	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.04	0.35
8053	colorado	hinsdale	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.03	0.00	0.04
8055	colorado	huerfano	0.09	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.42	0.00	0.02	0.74
8057	colorado	jackson	0.12	0.13	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00	0.00	0.02	0.64
8059	colorado	jefferson	0.08	0.07	0.01	0.12	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.01	0.45	0.93
8061	colorado	kiowa	0.11	0.04	0.01	0.66	0.00	0.09	0.00	0.02	0.00	0.00	0.00	0.00	0.13	0.31	0.00	0.01	1.39
8063	colorado	kit carson	1.08	0.69	0.06	1.50	0.00	0.17	0.00	0.03	0.00	0.00	0.00	0.00	1.60	0.30	0.00	0.01	5.45
8067	colorado	la plata	0.06	0.06	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.11	0.40
8065	colorado	lake	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.52	0.00	0.00	0.59
8069	colorado	larimer	0.55	0.40	0.04	0.08	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.01	1.12	0.01	0.00	0.48	2.75
8071	colorado	las animas	0.13	0.05	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.03	0.63
8073	colorado	lincoln	0.19	0.07	0.02	0.68	0.00	0.13	0.00	0.01	0.00	0.00	0.00	0.00	0.42	0.00	0.00	0.02	1.55
8075	colorado	logan	1.04	0.86	0.07	0.80	0.02	0.23	0.00	0.00	0.00	0.00	0.00	0.01	2.17	0.14	0.00	0.03	5.38
8077	colorado	mesa	0.45	0.90	0.04	0.08	0.00	0.02	0.00	0.00	0.00	0.02	0.00	0.00	1.21	0.01	0.10	0.27	3.11
8079	colorado	mineral	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.08
8081	colorado	moffat	0.09	0.07	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.27	0.68
8083	colorado	montezuma	0.08	0.20	0.01	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.02	0.21	0.00	0.00	0.08	0.66
8085	colorado	montrose	0.30	0.27	0.02	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.06	0.03	0.62	0.01	0.05	0.16	1.54
8087	colorado	morgan	1.52	0.38	0.11	0.52	0.01	0.16	0.01	0.00	0.00	0.00	0.00	0.00	3.37	0.03	0.00	0.06	6.18
8089	colorado	otero	0.30	0.21	0.02	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.04	0.00	0.81	0.00	0.00	0.06	1.52
8091	colorado	ouray	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.15
8093	colorado	park	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.07	0.36
8095	colorado	phillips	0.67	0.19	0.02	0.42	0.00	0.09	0.00	0.00	0.00	0.00	0.01	0.00	0.28	0.31	0.00	0.01	2.02
8097	colorado	pitkin	0.06	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.06	0.45	0.00	0.01	0.69
8099	colorado	prowers	0.34	0.45	0.03	0.20	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.29	0.00	0.03	2.06
8101	colorado	pueblo	0.41	0.33	0.04	0.03	0.00	0.02	0.00	0.00	0.00	0.01	0.02	0.01	1.27	0.01	0.00	0.20	2.34
8103	colorado	rio blanco	0.08	0.14	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.27	0.00	0.00	0.12	0.73
8105	colorado	rio grande	0.06	0.12	0.01	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.09	0.00	0.00	0.05	0.42
8107	colorado	routt	0.16	0.18	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.11	0.96
8109	colorado	saguache	0.09	0.13	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.17	0.00	0.00	0.02	0.47
8111	colorado	san juan	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.18	0.00	0.00	0.28
8113	colorado	san miguel	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.01	0.09
8115	colorado	sedgwick	0.39	0.07	0.01	0.37	0.00	0.05	0.00	0.01	0.00	0.00	0.02	0.00	0.16	0.32	0.00	0.01	1.41
8117	colorado	summit	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.05
8119	colorado	teller	0.13	0.00	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.06	1.22	0.00	0.02	1.48
8121	colorado	washington	0.43	0.25	0.03	0.96	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.56	0.33	0.00	0.03	2.78
8123	colorado	weld	5.59	4.13	0.45	1.93	0.14	0.50	0.02	0.07	0.01	0.00	0.04	0.09	16.12	0.06	2.02	2.50	33.66
8125	colorado	yuma	1.72	0.85	0.09	0.26	0.02	0.11	0.00	0.01	0.00	0.00	0.05	0.04	2.62	0.30	0.00	0.05	6.15
9001	connecticut	fairfield	0.05	0.15	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.01	0.01	0.00	0.34	0.01	0.03	0.43	1.07
9003	connecticut	hartford	0.46	0.38	0.03	0.03	0.00	0.02	0.00	0.11	0.00	0.02	0.22	0.01	1.23	0.07	0.02	0.74	3.35
9005	connecticut	litchfield	1.03	0.61	0.09	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.02	0.00	4.14	0.05	0.03	0.78	6.87
9007	connecticut	middlesex	0.06	0.08	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.35	0.01	0.01	0.26	0.80
9009	connecticut	new haven	0.26	0.35	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.03	0.03	0.00	1.01	0.07	0.02	0.52	2.37
9011	connecticut	new london	0.61	0.28	0.05	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.02	0.00	2.15	0.07	0.03	0.34	3.58
9013	connecticut	tolland	0.63	0.14	0.05	0.04	0.00	0.02	0.00	0.01	0.00	0.00	0.01	0.00	2.22	0.02	0.01	0.27	3.42
9015	connecticut	windham	0.71	0.18	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	2.34	0.05	0.10	0.19	3.70
10001	delaware	kent	4.59	0.59	2.35	1.10	0.00	0.60	0.02	0.30	0.01	0.15	0.41	0.01	2.06	0.20	4.30	0.42	17.11
10003	delaware	new castle	1.55	0.39	1.44	0.61	0.00	0.24	0.01	0.12	0.00	0.02	0.49	0.00	0.83	0.01	1.45	0.45	7.61
10005	delaware	sussex	12.36	0.71	2.88	1.25	0.00	0.70	0.06	0.48	0.01	0.45	0.35	0.02	1.33	0.22	16.23	0.17	37.23
11001	district of columbia	washington	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
12001	florida	alachua	0.17	1.87	0.03	0.01	0.00	0.02	0.01	0.02	0.07	0.01	0.01	0.00	2.15	0.07	0.01	0.51	4.97
12003	florida	baker	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.04	0.28
12005	florida	bay	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.03	0.00	0.00	0.04	0.42
12007	florida	bradford	0.16	0.15	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.01	1.70	0.09	2.68
12009	florida	brevard	0.07	2.82	0.02	0.01	0.00	0.01	0.01	0.02	0.00	0.20	0.06	0.00	1.76	0.03	0.02	0.21	5.23
12011	florida	broward	0.00	8.37	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.03	0.00	0.00	0.47	0.02	0.01	1.01	10.00
12013	florida	calhoun	0.01	0.52	0.01	0.00	0.00	0.00	0.08	0.00	0.10	0.00	0.01	0.00	0.19	0.03	0.00	0.07	1.03
12015	florida	charlotte	0.17	14.85	0.09	0.02	0.00	0.02	0.03	0.00	0.01	0.88	0.08	0.00	2.20	0.01	0.00	0.09	18.46
12017	florida	citrus	0.11	1.49	0.01	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.87	0.04	0.01	0.15	2.71
12019	florida	clay	0.11	0.29	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.73	0.02	0.00	0.23	1.43
12021	florida	collier	0.32	1.77	0.18	0.04	0.02	0.06	0.06	0.00	0.02	0.88	0.15	0.00	0.52	0.00	0.00	0.06	4.08
12023	florida	columbia	0.08	0.82	0.01	0.00	0.00	0.01	0.00	0.01	0.03	0.00	0.00	0.00	0.84	0.03	1.40	0.16	3.40
12027	florida	de soto	0.52	11.28	0.18	0.04	0.01	0.04	0.05	0.00	0.02	0.81	0.10	0.00	4.09	0.01	0.01	0.11	17.26
12029	florida	dixie	0.09	0.17	0.01	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.01	0.00	0.22	0.01	0.89	0.03	1.47
12031	florida	duval	0.19	0.14	0.02	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.04	0.00	1.27	0.02	0.00	0.27	1.98
12033	florida	escambia	0.13	0.85	0.07	0.06	0.00	0.04	0.47	0.05	0.43	0.00	0.01	0.00	0.41	0.02	0.01	0.23	2.78
12035	florida	flagler	0.01	0.29	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.13	0.02	0.00	0.02	0.52
12037	florida	franklin	0.03	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.59	0.00	0.69
12039	florida	gadsden	0.04	1.16	0.01	0.00	0.00	0.00	0.02	0.01	0.06	0.00	0.00	0.00	0.19	0.02	0.02	0.09	1.63
12041	florida	gilchrist	0.60	1.25	0.06	0.03	0.00	0.03	0.00	0.00	0.05	0.00	0.01	0.00	3.15	0.01	0.01	0.16	5.37
12043	florida	glades	0.09	4.65	0.03	0.01	10.05	0.01	0.01	0.00	0.00	0.11	0.01	0.00	2.19	0.01	0.00	0.03	17.20
12045	florida	gulf	0.04	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.67	0.00	0.77
12047	florida	hamilton	0.14	1.02	0.01	0.01	0.00	0.01	0.00	0.02	0.03	0.00	0.01	0.00	0.15	0.06	0.16	0.02	1.66
12049	florida	hardee	1.08	9.84	0.21	0.07	0.00	0.06	0.05	0.00	0.01	0.46	0.10	0.00	6.18	0.02	0.00	0.07	18.14
12051	florida	hendry	0.42	7.37	0.23	0.05	31.37	0.05	0.08	0.00	0.02	1.22	0.17	0.00	2.31	0.07	0.00	0.04	43.42
12053	florida	hernando	0.44	1.41	0.05	0.02	0.00	0.02	0.00	0.00	0.00	0.05	0.05	0.00	2.32	0.11	4.40	0.66	9.53
12055	florida	highlands	0.61	8.45	0.17	0.05	0.35	0.04	0.05	0.01	0.01	0.74	0.08	0.00	5.95	0.06	0.00	0.09	16.66
12057	florida	hillsborough	1.99	6.08	0.71	0.24	0.02	0.18	0.24	0.23	0.09	1.43	0.40	0.00	6.53	0.68	7.39	2.19	28.41
12059	florida	holmes	0.11	0.56	0.03	0.01	0.00	0.01	0.02	0.00	0.04	0.00	0.00	0.00	0.49	0.01	0.13	0.08	1.50
12061	florida	indian river	0.36	1.00	0.20	0.04	0.01	0.04	0.07	0.00	0.02	1.88	0.12	0.00	1.00	0.01	0.00	0.06	4.82
12063	florida	jackson	0.13	1.58	0.04	0.02	0.00	0.02	0.42	0.01	0.56	0.01	0.01	0.00	1.49	0.02	0.01	0.17	4.49
12065	florida	jefferson	0.05	0.30	0.01	0.00	0.00	0.01	0.00	0.00	0.03	0.00	0.00	0.00	0.41	0.02	0.00	0.07	0.89
12067	florida	lafayette	0.71	0.47	0.07	0.03	0.00	0.03	0.00	0.00	0.02	0.00	0.00	0.00	2.13	0.03	0.51	0.02	4.04
12069	florida	lake	0.21	9.75	0.11	0.02	0.00	0.02	0.04	0.00	0.01	0.84	0.07	0.00	2.69	0.13	0.02	0.86	14.78
12071	florida	lee	0.12	0.90	0.07	0.01	0.03	0.01	0.02	0.00	0.01	0.64	0.07	0.00	0.78	0.13	0.01	0.43	3.25
12073	florida	leon	0.01	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.07	0.02	0.00	0.12	0.57
12075	florida	levy	0.07	2.88	0.01	0.00	0.00	0.01	0.00	0.01	0.20	0.00	0.01	0.00	0.90	0.02	0.00	0.26	4.37
12077	florida	liberty	0.00	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.02	0.00	0.01	0.27
12079	florida	madison	0.40	0.99	0.04	0.02	0.00	0.02	0.01	0.00	0.07	0.00	0.00	0.00	0.59	0.02	0.54	0.04	2.74
12081	florida	manatee	2.13	8.01	0.86	0.20	0.03	0.20	0.27	0.00	0.08	1.39	0.55	0.00	6.08	0.04	0.06	0.34	20.24
12083	florida	marion	0.04	4.29	0.02	0.01	0.00	0.01	0.01	0.00	0.13	0.04	0.01	0.00	3.06	0.20	0.04	4.50	12.37
12085	florida	martin	0.39	3.22	0.15	0.04	10.89	0.04	0.05	0.05	0.01	0.54	0.12	0.00	2.60	0.05	2.80	0.23	21.18
12086	florida	miami-dade	0.42	16.91	0.21	0.04	0.42	0.04	0.07	5.88	0.02	0.10	1.14	0.02	0.57	0.16	0.15	1.24	27.39
12087	florida	monroe	0.00	1.19	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.21
12089	florida	nassau	0.13	0.35	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.73	0.04	0.03	0.13	1.43
12091	florida	okaloosa	0.02	0.88	0.01	0.00	0.00	0.01	0.02	0.03	0.15	0.00	0.00	0.00	0.21	0.02	0.02	0.17	1.53
12093	florida	okeechobee	1.42	6.79	0.15	0.07	0.00	0.06	0.01	0.00	0.00	0.13	0.04	0.00	11.59	0.03	0.00	0.15	20.45
12095	florida	orange	0.11	1.78	0.06	0.01	0.00	0.01	0.02	0.00	0.01	0.22	0.03	0.00	2.19	0.15	0.03	0.48	5.11
12097	florida	osceola	0.09	14.03	0.05	0.01	0.00	0.01	0.02	0.01	0.00	0.38	0.09	0.00	6.85	0.01	0.00	0.20	21.72
12099	florida	palm beach	1.07	2.66	0.58	0.12	484.83	0.52	0.20	0.28	0.06	0.50	0.46	0.00	1.03	0.04	0.02	1.31	493.68
12101	florida	pasco	0.53	3.77	0.10	0.03	0.00	0.03	0.02	0.00	0.01	0.30	0.08	0.00	4.00	0.06	5.01	0.62	14.58
12103	florida	pinellas	0.04	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.00	0.07	0.19	0.39
12105	florida	polk	1.65	12.58	0.58	0.27	0.02	0.16	0.22	0.46	0.06	1.96	0.34	0.00	6.79	0.08	0.03	0.53	25.73
12107	florida	putnam	0.12	0.94	0.02	0.01	0.00	0.01	0.01	0.01	0.00	0.01	0.05	0.00	0.68	0.02	0.13	0.11	2.12
12113	florida	santa rosa	0.03	0.89	0.03	0.04	0.00	0.00	0.37	0.03	0.55	0.00	0.01	0.00	0.17	0.00	0.01	0.16	2.29
12115	florida	sarasota	0.19	3.34	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.08	0.00	0.00	2.17	0.02	0.00	0.25	6.11
12117	florida	seminole	0.06	0.65	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.07	0.00	0.58	0.09	0.07	0.61	2.32
12109	florida	st johns	0.13	0.52	0.07	0.01	0.00	0.01	0.02	0.01	0.01	0.02	0.33	0.00	0.07	0.01	0.02	0.11	1.34
12111	florida	st lucie	0.57	5.39	0.26	0.06	0.01	0.06	0.09	0.01	0.03	1.85	0.15	0.00	2.84	0.03	0.00	0.08	11.42
12119	florida	sumter	0.15	2.66	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	3.73	0.05	0.01	0.38	7.02
12121	florida	suwannee	2.21	2.63	0.22	0.11	0.00	0.10	0.02	0.01	0.08	0.01	0.02	0.00	2.66	0.04	3.29	0.14	11.54
12123	florida	taylor	0.00	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.01	0.01	0.25
12125	florida	union	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.22	0.02	0.00	0.04	0.63
12127	florida	volusia	0.27	2.29	0.04	0.02	0.00	0.01	0.01	0.02	0.00	0.07	0.01	0.00	2.03	0.16	0.04	0.93	5.91
12129	florida	wakulla	0.00	0.26	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.01	0.02	0.32
12131	florida	walton	0.10	0.61	0.03	0.01	0.00	0.01	0.03	0.00	0.06	0.00	0.00	0.00	0.47	0.02	0.20	0.08	1.62
12133	florida	washington	0.02	0.34	0.01	0.00	0.00	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.23	0.01	0.00	0.03	0.68
13001	georgia	appling	0.73	0.48	0.08	0.04	0.00	0.03	0.15	0.01	0.09	0.02	0.00	0.00	0.41	0.00	1.24	0.04	3.31
13003	georgia	atkinson	0.49	0.26	0.05	0.03	0.00	0.02	0.07	0.00	0.03	0.00	0.00	0.00	0.11	0.00	0.83	0.02	1.92
13005	georgia	bacon	0.29	0.37	0.03	0.01	0.00	0.01	0.10	0.00	0.03	0.05	0.00	0.00	0.20	0.00	0.99	0.02	2.10
13007	georgia	baker	0.16	0.73	0.02	0.01	0.00	0.01	0.10	0.00	0.24	0.00	0.00	0.00	0.11	0.00	0.28	0.01	1.67
13009	georgia	baldwin	0.05	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.17	0.05	0.03	0.57
13011	georgia	banks	2.83	0.07	0.26	0.14	0.00	0.12	0.01	0.00	0.00	0.00	0.00	0.00	0.51	0.21	5.83	0.10	10.10
13013	georgia	barrow	0.99	0.14	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.33	1.96	0.16	4.26
13015	georgia	bartow	2.05	0.56	0.22	0.12	0.00	0.08	0.05	0.01	0.00	0.00	0.00	0.00	1.05	0.01	3.62	0.20	7.99
13017	georgia	ben hill	0.04	0.37	0.01	0.00	0.00	0.01	0.06	0.00	0.06	0.00	0.00	0.00	0.03	0.00	0.03	0.06	0.67
13019	georgia	berrien	0.09	0.58	0.02	0.01	0.00	0.01	0.31	0.01	0.18	0.01	0.01	0.00	0.19	0.00	0.26	0.06	1.73
13021	georgia	bibb	0.28	0.35	0.03	0.02	0.00	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.17	0.29	0.44	0.03	1.63
13023	georgia	bleckley	0.07	0.82	0.04	0.02	0.00	0.01	0.06	0.00	0.09	0.00	0.00	0.00	0.13	0.00	0.04	0.03	1.32
13025	georgia	brantley	0.01	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.12	0.02	0.49
13027	georgia	brooks	0.60	2.00	0.16	0.06	0.00	0.04	0.53	0.02	0.31	0.03	0.05	0.00	2.04	0.08	0.35	0.15	6.41
13029	georgia	bryan	0.00	0.29	0.01	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.01	0.00	0.02	0.37
13031	georgia	bulloch	0.15	1.62	0.07	0.02	0.00	0.02	0.53	0.01	0.32	0.01	0.01	0.00	0.21	0.16	0.18	0.07	3.36
13033	georgia	burke	0.44	2.23	0.08	0.08	0.00	0.05	0.47	0.01	0.49	0.00	0.01	0.00	1.41	0.03	0.00	0.06	5.37
13035	georgia	butts	0.04	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.07	0.06	0.41
13037	georgia	calhoun	0.07	0.74	0.01	0.02	0.00	0.01	0.08	0.00	0.19	0.00	0.00	0.00	0.06	0.10	0.03	0.01	1.32
13039	georgia	camden	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.01	0.08
13043	georgia	candler	0.03	0.48	0.01	0.01	0.00	0.00	0.12	0.00	0.04	0.00	0.00	0.00	0.10	0.11	0.12	0.02	1.05
13045	georgia	carroll	4.79	0.31	0.44	0.24	0.00	0.20	0.02	0.00	0.00	0.00	0.00	0.00	1.00	0.02	8.53	0.35	15.91
13047	georgia	catoosa	1.17	0.26	0.11	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.04	2.47	0.12	4.68
13049	georgia	charlton	0.03	0.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.17	0.02	0.52
13051	georgia	chatham	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.23	0.01	0.34
13053	georgia	chattahoochee	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11
13055	georgia	chattooga	0.29	0.30	0.04	0.02	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.39	0.26	0.67	0.08	2.06
13057	georgia	cherokee	0.70	0.19	0.07	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.45	0.02	1.26	0.42	3.18
13059	georgia	clarke	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.26	0.00	0.05	0.48
13061	georgia	clay	0.02	0.22	0.00	0.01	0.00	0.00	0.05	0.01	0.08	0.00	0.00	0.00	0.03	0.10	0.10	0.01	0.63
13063	georgia	clayton	0.09	0.16	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.05	0.47
13065	georgia	clinch	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.11
13067	georgia	cobb	0.07	0.17	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.01	0.01	0.24	0.84
13069	georgia	coffee	1.08	0.71	0.12	0.06	0.00	0.05	0.31	0.01	0.20	0.02	0.01	0.00	0.20	0.53	1.80	0.07	5.18
13071	georgia	colquitt	1.08	0.83	0.15	0.08	0.00	0.06	0.71	0.02	0.41	0.03	0.06	0.00	0.27	0.00	1.80	0.05	5.56
13073	georgia	columbia	0.08	0.20	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.29	0.08	0.10	0.90
13075	georgia	cook	0.18	0.53	0.03	0.01	0.00	0.01	0.21	0.02	0.13	0.00	0.01	0.00	0.09	0.13	0.25	0.03	1.65
13077	georgia	coweta	0.08	0.40	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.00	0.09	0.25	1.25
13079	georgia	crawford	0.60	0.42	0.06	0.03	0.00	0.03	0.01	0.01	0.03	0.01	0.00	0.00	0.07	0.21	1.00	0.04	2.54
13081	georgia	crisp	0.12	0.71	0.03	0.02	0.00	0.04	0.28	0.00	0.17	0.01	0.01	0.00	0.11	0.00	0.12	0.02	1.63
13083	georgia	dade	0.70	0.08	0.07	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	1.31	0.08	2.50
13085	georgia	dawson	1.70	0.06	0.16	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.10	0.00	3.05	0.06	5.31
13089	georgia	de kalb	0.00	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.29
13087	georgia	decatur	0.29	1.44	0.05	0.02	0.00	0.02	0.26	0.02	0.48	0.00	0.01	0.00	0.48	0.01	0.18	0.02	3.28
13091	georgia	dodge	0.23	1.17	0.04	0.02	0.00	0.02	0.10	0.01	0.09	0.01	0.00	0.00	0.12	0.00	0.35	0.07	2.23
13093	georgia	dooly	0.18	0.50	0.04	0.04	0.00	0.02	0.55	0.02	0.28	0.01	0.01	0.00	0.08	0.12	0.21	0.02	2.07
13095	georgia	dougherty	0.06	0.55	0.01	0.00	0.00	0.00	0.04	0.01	0.22	0.00	0.00	0.00	0.09	0.15	0.04	0.02	1.20
13097	georgia	douglas	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.09	0.21
13099	georgia	early	0.08	0.87	0.02	0.02	0.00	0.01	0.21	0.02	0.44	0.00	0.00	0.00	0.27	0.10	0.10	0.03	2.17
13101	georgia	echols	0.02	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.03	0.01	0.22
13103	georgia	effingham	0.03	0.38	0.02	0.00	0.00	0.00	0.04	0.01	0.03	0.00	0.00	0.00	0.07	0.00	0.04	0.05	0.69
13105	georgia	elbert	1.23	0.19	0.12	0.07	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.35	0.01	2.46	0.09	4.58
13107	georgia	emanuel	0.06	1.45	0.03	0.01	0.00	0.01	0.23	0.01	0.19	0.00	0.01	0.00	0.15	0.00	0.03	0.05	2.23
13109	georgia	evans	0.36	0.34	0.04	0.02	0.00	0.02	0.07	0.00	0.03	0.00	0.00	0.00	0.07	0.00	0.71	0.02	1.68
13111	georgia	fannin	0.24	0.06	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.06	0.01	0.62	0.06	1.11
13113	georgia	fayette	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.42	0.00	0.10	0.66
13115	georgia	floyd	1.89	0.53	0.22	0.10	0.00	0.08	0.03	0.00	0.00	0.00	0.00	0.00	0.47	0.01	3.19	0.20	6.71
13117	georgia	forsyth	1.11	0.09	0.11	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.01	2.36	0.23	4.31
13119	georgia	franklin	7.11	0.20	0.66	0.36	0.00	0.30	0.03	0.00	0.01	0.00	0.00	0.00	0.87	0.01	14.74	0.10	24.39
13121	georgia	fulton	0.01	0.39	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.01	0.04	0.32	0.98
13123	georgia	gilmer	5.35	0.07	0.50	0.27	0.00	0.22	0.02	0.00	0.00	0.00	0.00	0.00	0.23	0.25	9.77	0.05	16.74
13125	georgia	glascock	0.02	0.33	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.10	0.00	0.14	0.01	0.63
13127	georgia	glynn	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.14
13129	georgia	gordon	5.77	0.40	0.62	0.29	0.00	0.24	0.02	0.02	0.00	0.00	0.00	0.00	0.85	0.01	10.53	0.23	19.00
13131	georgia	grady	0.41	0.61	0.05	0.02	0.00	0.02	0.25	0.00	0.27	0.00	0.01	0.00	0.31	0.04	0.55	0.09	2.63
13133	georgia	greene	0.36	0.23	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.64	0.05	1.85
13135	georgia	gwinnett	0.14	0.21	0.01	0.01	0.00	0.01	0.00	0.01	0.01	0.00	0.00	0.00	0.12	0.74	0.22	0.15	1.63
13137	georgia	habersham	2.38	0.05	0.22	0.12	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.49	0.23	4.74	0.08	8.41
13139	georgia	hall	2.87	0.15	0.28	0.15	0.00	0.12	0.01	0.00	0.00	0.00	0.00	0.00	1.31	0.04	9.14	0.33	14.41
13141	georgia	hancock	0.00	0.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.05	0.37
13143	georgia	haralson	1.28	0.07	0.12	0.06	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.19	0.01	2.22	0.07	4.08
13145	georgia	harris	0.04	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.02	0.06	0.10	0.59
13147	georgia	hart	3.25	0.15	0.32	0.18	0.00	0.14	0.01	0.00	0.00	0.00	0.00	0.00	0.91	0.19	7.11	0.15	12.44

13149	georgia	heard	0.70	0.12	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	1.40	0.05	2.55
13151	georgia	henry	0.08	0.20	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.01	0.14	0.15	0.82
13153	georgia	houston	0.09	0.42	0.03	0.03	0.00	0.01	0.09	0.01	0.19	0.00	0.00	0.00	0.24	0.00	0.06	0.08	1.27
13155	georgia	irwin	0.20	0.72	0.03	0.01	0.00	0.01	0.35	0.01	0.24	0.01	0.01	0.00	0.16	0.00	0.23	0.04	2.01
13157	georgia	jackson	4.20	0.24	0.39	0.21	0.00	0.18	0.02	0.00	0.00	0.00	0.00	0.00	0.96	0.02	10.98	0.27	17.47
13159	georgia	jasper	0.07	0.15	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.30	0.09	0.94
13161	georgia	jeff davis	0.16	0.31	0.02	0.01	0.00	0.01	0.16	0.00	0.07	0.01	0.00	0.00	0.07	0.10	0.34	0.02	1.28
13163	georgia	jefferson	0.15	2.34	0.03	0.04	0.00	0.03	0.10	0.01	0.13	0.00	0.00	0.00	0.61	0.00	0.00	0.02	3.47
13165	georgia	jenkins	0.07	0.79	0.01	0.00	0.00	0.01	0.16	0.00	0.12	0.00	0.00	0.00	0.11	0.12	0.04	0.02	1.45
13167	georgia	johnson	0.03	1.04	0.01	0.01	0.00	0.01	0.00	0.00	0.03	0.00	0.00	0.00	0.07	0.01	0.04	0.02	1.29
13169	georgia	jones	0.06	0.24	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.23	0.07	0.08	0.96
13171	georgia	lamar	0.96	0.21	0.09	0.05	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.16	0.22	1.86	0.07	3.66
13173	georgia	lanier	0.02	0.35	0.01	0.00	0.00	0.00	0.07	0.01	0.03	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.57
13175	georgia	laurens	0.07	2.31	0.08	0.05	0.00	0.03	0.08	0.02	0.11	0.00	0.00	0.00	0.24	0.02	0.00	0.05	3.06
13177	georgia	lee	0.15	1.72	0.04	0.05	0.00	0.02	0.14	0.01	0.37	0.01	0.00	0.00	0.29	0.00	0.00	0.03	2.83
13179	georgia	liberty	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.10
13181	georgia	lincoln	0.03	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.05	0.02	0.24
13183	georgia	long	0.13	0.26	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.23	0.01	0.68
13185	georgia	lowndes	0.04	0.90	0.02	0.00	0.00	0.00	0.11	0.01	0.09	0.00	0.01	0.00	0.13	0.01	0.00	0.07	1.42
13187	georgia	lumpkin	0.77	0.04	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.24	1.37	0.07	2.75
13193	georgia	macon	1.27	0.66	0.14	0.09	0.00	0.06	0.13	0.05	0.22	0.03	0.00	0.00	1.61	0.00	1.70	0.04	5.99
13195	georgia	madison	3.80	0.19	0.36	0.20	0.00	0.16	0.02	0.00	0.00	0.00	0.00	0.00	0.69	0.03	7.28	0.17	12.90
13197	georgia	marion	0.33	0.85	0.03	0.02	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.66	0.03	2.06
13189	georgia	mcduffie	0.05	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.06	0.07	0.69
13191	georgia	mcintosh	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.08
13199	georgia	meriwether	0.02	0.55	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.04	0.00	0.09	0.97
13201	georgia	miller	0.10	0.94	0.01	0.02	0.00	0.01	0.21	0.01	0.36	0.00	0.00	0.00	0.16	0.10	0.03	0.01	1.97
13205	georgia	mitchell	1.03	1.50	0.11	0.05	0.00	0.06	0.32	0.00	0.43	0.01	0.02	0.00	0.43	0.01	1.67	0.05	5.69
13207	georgia	monroe	0.53	0.28	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.00	0.89	0.07	2.07
13209	georgia	montgomery	0.04	0.22	0.01	0.01	0.00	0.00	0.03	0.00	0.05	0.00	0.00	0.00	0.03	0.01	0.16	0.03	0.59
13211	georgia	morgan	1.53	0.63	0.15	0.09	0.00	0.07	0.01	0.01	0.00	0.00	0.00	0.00	1.89	0.03	2.55	0.21	7.16
13213	georgia	murray	1.61	0.55	0.19	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.30	0.01	2.87	0.10	5.79
13215	georgia	muscogee	0.02	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.36	0.03	0.51
13217	georgia	newton	0.10	0.42	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.32	0.42	0.12	1.82
13219	georgia	oconee	1.22	0.14	0.12	0.06	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.38	0.00	2.05	0.20	4.25
13221	georgia	oglethorpe	2.95	0.60	0.28	0.16	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.41	3.06	5.11	0.14	12.85
13223	georgia	paulding	0.60	0.27	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	1.04	0.08	2.18
13225	georgia	peach	0.05	0.28	0.05	0.01	0.00	0.00	0.02	0.01	0.17	0.01	0.00	0.00	0.04	0.00	0.07	0.09	0.85
13227	georgia	pickens	1.49	0.07	0.14	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.09	0.00	2.70	0.11	4.73
13229	georgia	pierce	0.06	0.63	0.01	0.01	0.00	0.00	0.08	0.00	0.06	0.01	0.00	0.00	0.08	0.00	0.15	0.04	1.13
13231	georgia	pike	0.21	0.34	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.33	0.09	1.26
13233	georgia	polk	0.86	0.29	0.10	0.05	0.00	0.04	0.01	0.01	0.00	0.00	0.00	0.00	0.42	0.03	1.45	0.16	3.41
13235	georgia	pulaski	0.42	0.64	0.05	0.04	0.00	0.04	0.18	0.01	0.15	0.00	0.00	0.00	0.01	0.00	0.67	0.02	2.26
13237	georgia	putnam	0.15	0.31	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.85	0.00	0.18	0.06	1.59
13239	georgia	quitman	0.01	0.07	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.02	0.11	0.12	0.00	0.34
13241	georgia	rabun	0.44	0.01	0.04	0.02	0.00	0.02	0.00	0.00	0.02	0.01	0.00	0.00	0.04	0.01	0.76	0.01	1.38
13243	georgia	randolph	0.08	0.55	0.01	0.02	0.00	0.01	0.07	0.00	0.19	0.00	0.00	0.00	0.09	0.00	0.00	0.00	1.03
13245	georgia	richmond	6.54	0.49	0.25	1.64	0.00	0.45	0.39	5.08	0.02	0.00	0.10	0.01	0.07	0.29	0.08	0.08	15.49
13247	georgia	rockdale	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.42	0.00	0.09	0.61
13249	georgia	schley	0.21	0.23	0.02	0.01	0.00	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.02	0.00	0.48	0.01	1.02
13251	georgia	screven	0.08	1.25	0.02	0.01	0.00	0.01	0.27	0.00	0.19	0.00	0.00	0.00	0.20	0.06	0.00	0.04	2.15
13253	georgia	seminole	0.09	0.67	0.02	0.01	0.00	0.00	0.17	0.01	0.26	0.00	0.00	0.00	0.17	0.02	0.00	0.01	1.44
13255	georgia	spalding	0.12	0.32	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.17	0.18	1.08
13257	georgia	stephens	1.06	0.06	0.10	0.05	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	1.93	0.02	3.60
13259	georgia	stewart	0.03	0.34	0.01	0.00	0.00	0.00	0.03	0.00	0.06	0.00	0.00	0.00	0.02	0.00	0.03	0.01	0.53
13261	georgia	sumter	0.35	1.69	0.06	0.07	0.00	0.02	0.30	0.02	0.26	0.01	0.00	0.00	0.34	0.12	0.38	0.03	3.65
13263	georgia	talbot	0.03	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.05	0.03	0.39
13265	georgia	taliaferro	0.05	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.15	0.04	0.01	0.50
13267	georgia	tattnall	1.56	1.01	0.18	0.08	0.00	0.07	0.16	0.00	0.10	0.01	0.00	0.00	0.24	0.64	2.85	0.07	7.00
13269	georgia	taylor	0.18	0.90	0.04	0.02	0.00	0.01	0.01	0.01	0.02	0.02	0.00	0.00	0.09	0.00	0.28	0.03	1.61
13271	georgia	telfair	0.01	0.89	0.01	0.00	0.00	0.00	0.03	0.00	0.03	0.00	0.00	0.00	0.07	0.02	0.00	0.03	1.11
13273	georgia	terrell	0.07	0.85	0.02	0.03	0.00	0.01	0.13	0.00	0.17	0.00	0.00	0.00	0.04	0.00	0.00	0.02	1.35
13275	georgia	thomas	0.23	0.89	0.05	0.03	0.00	0.02	0.52	0.01	0.38	0.01	0.01	0.00	0.39	0.17	0.42	0.06	3.19
13277	georgia	tift	0.09	0.51	0.03	0.01	0.00	0.02	0.21	0.01	0.13	0.01	0.02	0.00	0.19	0.12	0.04	0.03	1.41
13279	georgia	toombs	0.07	0.77	0.03	0.01	0.00	0.01	0.06	0.00	0.06	0.00	0.02	0.00	0.15	0.11	0.12	0.03	1.45
13281	georgia	towns	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.01	0.12
13283	georgia	treutlen	0.01	0.30	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.02	0.11	0.00	0.01	0.49
13285	georgia	troup	0.06	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.06	0.08	0.61
13287	georgia	turner	0.11	0.99	0.02	0.01	0.00	0.01	0.34	0.01	0.23	0.01	0.00	0.00	0.09	0.00	0.14	0.04	1.99
13289	georgia	twiggs	0.05	1.01	0.01	0.00	0.00	0.00	0.05	0.00	0.03	0.00	0.00	0.00	0.06	0.02	0.06	0.04	1.33
13291	georgia	union	0.05	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.01	0.07	0.06	0.48
13293	georgia	upson	0.40	0.16	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.67	0.12	1.60
13295	georgia	walker	2.55	0.38	0.27	0.15	0.00	0.11	0.01	0.01	0.00	0.00	0.00	0.00	0.73	0.01	4.60	0.22	9.04
13297	georgia	walton	0.44	0.43	0.04	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.68	0.01	1.36	0.26	3.30
13299	georgia	ware	0.05	0.35	0.01	0.00	0.00	0.00	0.03	0.00	0.02	0.01	0.00	0.00	0.09	0.11	0.15	0.02	0.83
13301	georgia	warren	0.04	0.43	0.00	0.01	0.00	0.00	0.01	0.00	0.02	0.01	0.00	0.00	0.11	0.00	0.04	0.05	0.71
13303	georgia	washington	0.05	1.26	0.02	0.02	0.00	0.01	0.01	0.00	0.07	0.00	0.00	0.00	0.19	0.03	0.04	0.05	1.76
13305	georgia	wayne	0.14	0.77	0.03	0.01	0.00	0.01	0.07	0.00	0.04	0.00	0.00	0.00	0.11	0.00	0.72	0.03	1.93
13307	georgia	webster	0.02	0.14	0.01	0.00	0.00	0.00	0.09	0.00	0.06	0.00	0.00	0.00	0.04	0.00	0.00	0.02	0.37
13309	georgia	wheeler	0.01	0.46	0.01	0.01	0.00	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.02	0.01	0.00	0.01	0.57
13311	georgia	white	1.30	0.02	0.12	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.36	0.01	2.58	0.06	4.58
13313	georgia	whitfield	2.01	0.35	0.19	0.10	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.41	0.00	3.95	0.12	7.24
13315	georgia	wilcox	0.44	0.82	0.05	0.03	0.00	0.02	0.22	0.01	0.15	0.00	0.01	0.00	0.13	0.00	0.81	0.02	2.71
13317	georgia	wilkes	0.90	0.21	0.08	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.16	1.45	0.04	3.63
13319	georgia	wilkinson	0.03	0.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.05	0.04	0.54
13321	georgia	worth	0.28	1.83	0.04	0.03	0.00	0.02	0.57	0.01	0.46	0.01	0.01	0.00	0.25	0.00	0.28	0.05	3.85
16001	idaho	ada	2.11	1.11	0.16	0.56	0.11	0.20	0.01	0.07	0.00	0.01	0.07	0.09	7.58	0.07	0.01	0.29	12.44
16003	idaho	adams	0.02	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.01	0.18
16005	idaho	bannock	0.08	0.38	0.01	0.45	0.00	0.10	0.00	0.03	0.00	0.00	0.06	0.00	0.31	0.00	0.00	0.05	1.47
16007	idaho	bear lake	0.05	0.27	0.01	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.36	0.00	0.25	0.00	0.00	0.04	1.06
16009	idaho	benewah	0.00	0.16	0.00	0.75	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.03	0.01	0.00	0.00	0.01	0.99
16011	idaho	bingham	0.45	0.80	0.09	1.92	0.24	0.22	0.02	0.01	0.01	0.02	0.77	0.00	1.24	0.00	0.00	0.14	5.94
16013	idaho	blaine	0.02	0.12	0.00	0.02	0.00	0.07	0.00	0.00	0.00	0.00	0.18	0.00	0.07	0.00	0.00	0.05	0.52
16015	idaho	boise	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.02	0.00	0.00	0.00	0.21
16017	idaho	bonner	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.03	0.11
16019	idaho	bonneville	0.37	0.78	0.06	0.89	0.00	1.30	0.01	0.02	0.00	0.01	0.36	0.00	1.33	0.00	0.00	0.06	5.20
16021	idaho	boundary	0.02	0.06	0.00	0.20	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.01	0.40
16023	idaho	butte	0.02	0.21	0.00	0.03	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.01	0.44
16025	idaho	camas	0.02	0.23	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.00	0.22	0.00	0.07	0.00	0.00	0.00	0.59
16027	idaho	canyon	2.27	1.54	0.17	1.03	0.24	0.18	0.02	0.25	0.00	0.25	0.57	0.38	6.67	0.01	0.04	0.41	14.01
16029	idaho	caribou	0.10	0.35	0.01	0.47	0.00	0.64	0.00	0.00	0.00	0.00	0.07	0.00	0.38	0.00	0.00	0.04	2.07
16031	idaho	cassia	0.95	0.56	0.09	0.61	0.20	0.27	0.01	0.01	0.00	0.01	0.37	0.06	3.43	0.01	0.01	0.06	6.63
16033	idaho	clark	0.02	0.17	0.00	0.12	0.00	0.02	0.00	0.00	0.00	0.00	0.30	0.00	0.07	0.00	0.00	0.02	0.71
16035	idaho	clearwater	0.01	0.05	0.00	0.17	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.04	0.00	0.00	0.00	0.29
16037	idaho	custer	0.04	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.14	0.00	0.00	0.02	0.35
16039	idaho	elmore	0.72	0.37	0.06	0.20	0.03	0.04	0.00	0.02	0.00	0.00	0.39	0.01	2.65	0.00	0.00	0.03	4.55
16041	idaho	franklin	0.24	0.39	0.02	0.18	0.00	0.08	0.00	0.02	0.00	0.00	0.42	0.01	0.86	0.00	0.23	0.10	2.55
16043	idaho	fremont	0.06	0.26	0.02	0.34	0.00	0.61	0.01	0.01	0.00	0.01	0.19	0.00	0.15	0.00	0.00	0.04	1.69
16045	idaho	gem	0.18	0.36	0.01	0.03	0.00	0.02	0.00	0.00	0.00	0.02	0.59	0.00	0.66	0.00	0.00	0.07	1.95
16047	idaho	gooding	2.16	0.34	0.18	0.19	0.02	0.12	0.01	0.00	0.00	0.00	0.06	0.02	7.75	0.00	0.00	0.07	10.93
16049	idaho	idaho	0.04	0.22	0.00	0.73	0.00	0.05	0.00	0.03	0.00	0.00	0.00	0.01	0.13	0.00	0.00	0.03	1.24
16051	idaho	jefferson	0.32	0.51	0.04	0.40	0.00	0.39	0.01	0.00	0.00	0.01	0.13	0.00	1.19	0.00	0.00	0.11	3.11
16053	idaho	jerome	1.71	0.75	0.15	0.41	0.18	0.43	0.01	0.00	0.00	0.00	0.17	0.15	6.44	0.00	0.00	0.03	10.44
16055	idaho	kootenai	0.03	0.62	0.01	0.66	0.00	0.02	0.00	0.04	0.00	0.00	0.40	0.04	0.10	0.03	0.00	0.09	2.05
16057	idaho	latah	0.02	0.25	0.00	0.83	0.00	0.07	0.00	0.02	0.00	0.00	0.00	0.24	0.05	0.01	0.00	0.03	1.52
16059	idaho	lemhi	0.05	0.10	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.02	0.38
16061	idaho	lewis	0.02	0.38	0.00	1.10	0.00	0.07	0.00	0.07	0.00	0.00	0.00	0.09	0.06	0.00	0.00	0.00	1.78
16063	idaho	lincoln	0.37	0.27	0.03	0.10	0.04	0.05	0.00	0.00	0.00	0.00	0.01	0.00	1.36	0.01	0.00	0.02	2.27
16065	idaho	madison	0.08	0.20	0.03	0.49	0.00	0.48	0.01	0.00	0.00	0.01	0.27	0.00	0.13	0.00	0.00	0.03	1.72
16067	idaho	minidoka	0.25	0.30	0.04	0.36	0.22	0.31	0.01	0.00	0.00	0.01	0.20	0.03	0.59	0.00	0.03	0.18	2.54
16069	idaho	nez perce	0.03	0.61	0.00	0.71	0.00	0.07	0.00	0.04	0.00	0.00	0.00	0.21	0.12	0.00	0.00	0.02	1.83
16071	idaho	oneida	0.05	0.39	0.00	0.38	0.00	0.03	0.00	0.03	0.00	0.00	0.41	0.00	0.23	0.00	0.00	0.02	1.56
16073	idaho	owyhee	0.69	0.58	0.05	0.22	0.04	0.06	0.00	0.01	0.00	0.00	0.02	0.02	2.44	0.00	0.00	0.05	4.19
16075	idaho	payette	0.48	0.37	0.03	0.13	0.03	0.02	0.00	0.02	0.00	0.04	0.07	0.03	1.54	0.00	0.00	0.05	2.82
16077	idaho	power	0.22	0.17	0.05	1.16	0.13	0.04	0.02	0.10	0.00	0.01	0.52	0.00	0.36	0.00	0.00	0.01	2.80
16079	idaho	shoshone	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
16081	idaho	teton	0.02	0.19	0.00	0.11	0.00	0.35	0.00	0.00	0.00	0.00	0.04	0.00	0.07	0.00	0.00	0.01	0.80
16083	idaho	twin falls	1.47	1.20	0.12	0.58	0.11	0.58	0.01	0.01	0.00	0.01	0.16	0.39	4.78	0.02	0.00	0.10	9.54
16085	idaho	valley	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.07
16087	idaho	washington	0.11	0.34	0.01	0.09	0.01	0.01	0.00	0.00	0.00	0.01	0.01	0.01	0.40	0.00	0.00	0.05	1.05
17001	illinois	adams	5.99	0.53	3.28	0.33	0.00	0.04	0.01	0.02	0.00	0.01	0.02	0.01	1.23	1.59	0.00	0.11	13.20
17003	illinois	alexander	0.33	0.04	0.84	0.09	0.00	0.05	0.00	0.01	0.00	0.00	0.08	0.01	0.03	0.00	0.00	0.02	1.49
17005	illinois	bond	3.28	0.15	2.18	0.39	0.00	0.02	0.00	0.03	0.00	0.00	0.01	0.00	0.43	0.05	0.01	0.04	6.60
17007	illinois	boone	5.74	0.24	2.23	0.19	0.00	0.03	0.01	0.01	0.00	0.00	0.05	0.01	0.84	0.44	0.09	0.30	10.19
17009	illinois	brown	1.42	0.11	0.57	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.40	0.01	0.03	2.76
17011	illinois	bureau	11.82	0.82	4.09	0.11	0.00	0.05	0.02	0.00	0.01	0.01	0.05	0.01	0.54	1.95	0.20	0.51	20.20
17013	illinois	calhoun	1.03	0.10	0.38	0.05	0.00	0.02	0.00	0.01	0.01	0.00	0.01	0.00	0.16	1.69	0.01	0.03	3.50
17015	illinois	carroll	7.56	0.41	1.56	0.11	0.00	0.07	0.01	0.00	0.00	0.01	0.04	0.01	2.28	1.44	0.02	0.07	13.59
17017	illinois	cass	3.46	0.10	1.21	0.07	0.00	0.04	0.01	0.01	0.00	0.00	0.01	0.00	0.13	2.91	0.00	0.04	7.99
17019	illinois	champaign	15.17	0.84	10.99	0.19	0.00	0.04	0.03	0.01	0.01	0.02	0.05	0.03	0.76	0.38	0.01	0.14	28.67
17021	illinois	christian	9.60	0.12	5.51	0.10	0.00	0.06	0.01	0.00	0.00	0.01	0.03	0.01	0.27	1.45	0.00	0.05	17.24
17023	illinois	clark	3.58	0.90	3.06	0.13	0.00	0.10	0.01	0.03	0.00	0.02	0.03	0.00	0.14	2.23	0.02	0.08	10.32
17025	illinois	clay	3.11	0.72	2.17	0.27	0.00	0.27	0.01	0.04	0.00	0.00	0.02	0.01	0.20	2.23	0.02	0.06	9.13
17027	illinois	clinton	6.10	0.33	3.27	0.99	0.00	0.08	0.01	0.06	0.00	0.00	0.01	0.00	3.00	2.19	0.40	1.37	17.83
17029	illinois	coles	6.09	0.76	4.34	0.07	0.00	0.02	0.01	0.01	0.00	0.01	0.02	0.01	0.16	0.84	0.03	0.17	12.54
17031	illinois	cook	0.57	0.58	0.32	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.14	1.20	0.08	1.21	4.14
17033	illinois	crawford	2.19	0.77	2.07	0.12	0.00	0.04	0.01	0.06	0.00	0.05	0.03	0.00	0.13	0.31	0.03	0.05	5.87
17035	illinois	cumberland	2.83	0.11	1.94	0.07	0.00	0.03	0.01	0.01	0.00	0.00	0.01	0.00	0.70	0.61	0.00	0.04	6.37
17037	illinois	de kalb	23.83	0.31	7.85	0.64	0.00	0.44	0.03	0.00	0.01	0.02	0.20	0.04	3.90	14.82	0.21	0.25	52.55
17039	illinois	de witt	3.40	0.55	3.25	0.02	0.00	0.02	0.01	0.00	0.00	0.01	0.01	0.01	0.10	1.14	0.00	0.04	8.55
17041	illinois	douglas	5.17	0.18	3.43	0.05	0.00	0.06	0.01	0.01	0.00	0.01	0.01	0.01	0.59	0.12	0.05	0.40	10.09
17043	illinois	du page	0.32	1.94	0.31	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.56	3.16
17045	illinois	edgar	6.62	0.49	4.58	0.07	0.00	0.02	0.01	0.01	0.00	0.02	0.03	0.02	0.23	0.77	0.00	0.07	12.95
17047	illinois	edwards	1.10	0.08	1.00	0.13	0.00	0.01	0.00	0.04	0.00	0.07	0.00	0.00	0.08	0.36	0.00	0.04	2.93
17049	illinois	effingham	4.37	0.21	2.49	0.26	0.00	0.13	0.01	0.05	0.00	0.01	0.01	0.01	1.43	3.85	0.00	0.11	12.94
17051	illinois	fayette	4.05	0.23	2.99	0.23	0.00	0.20	0.01	0.05	0.00	0.01	0.01	0.01	0.41	0.31	0.00	0.11	8.63
17053	illinois	ford	7.31	0.28	4.31	0.19	0.00	0.05	0.01	0.01	0.00	0.01	0.02	0.02	0.19	4.06	0.04	0.09	16.59
17055	illinois	franklin	2.03	0.16	1.87	0.26	0.00	0.06	0.00	0.33	0.00	0.00	0.04	0.02	0.18	0.38	0.23	0.07	5.64
17057	illinois	fulton	5.48	0.32	2.70	0.11	0.00	0.04	0.01	0.00	0.00	0.01	0.04	0.01	0.82	1.16	0.00	0.09	10.78
17059	illinois	gallatin	2.18	0.39	1.69	0.13	0.00	0.03	0.00	0.10	0.00	0.00	0.01	0.02	0.05	0.00	0.34	0.00	4.95
17061	illinois	greene	4.20	0.18	2.28	0.10	0.00	0.02	0.01	0.02	0.00	0.00	0.01	0.01	0.36	0.85	0.00	0.04	8.08
17063	illinois	grundy	5.70	0.16	5.27	0.07	0.00	0.01	0.01	0.00	0.00	0.01	0.03	0.00	0.27	0.21	0.00	0.05	11.81
17065	illinois	hamilton	1.48	1.03	1.78	0.25	0.00	0.02	0.00	0.16	0.00	0.00	0.05	0.01	0.09	0.00	0.01	0.03	4.92
17067	illinois	hancock	6.19	0.28	3.27	0.15	0.00	0.05	0.01	0.00	0.01	0.01	0.03	0.00	0.70	4.02	0.00	0.05	14.77
17069	illinois	hardin	0.11	0.04	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.06	0.00	0.01	0.34
17071	illinois	henderson	2.90	0.12	1.01	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.41	0.46	0.00	0.04	5.02
17073	illinois	henry	11.08	1.16	4.95	0.10	0.00	0.08	0.02	0.00	0.05	0.01	0.04	0.00	1.66	3.91	0.46	0.22	23.75
17075	illinois	iroquois	15.16	0.84	8.32	0.27	0.00	0.06	0.02	0.01	0.01	0.02	0.04	0.02	1.47	1.96	0.49	3.37	32.06
17077	illinois	jackson	1.57	0.21	1.95	0.39	0.00	0.20	0.00	0.21	0.05	0.00	0.02	0.00	0.37	0.09	0.00	0.07	5.15
17079	illinois	jasper	4.23	0.72	2.67	0.19	0.00	0.07	0.01	0.09	0.00	0.01	0.01	0.01	0.49	4.57	0.00	0.03	13.10
17081	illinois	jefferson	1.46	1.29	1.68	0.38	0.00	0.09	0.00	0.17	0.00	0.00	0.11	0.02	0.37	0.27	0.04	0.13	6.02
17083	illinois	jersey	3.28	0.15	1.56	0.08	0.00	0.01	0.00	0.03	0.02	0.00	0.03	0.00	0.30	0.04	0.00	0.07	5.57
17085	illinois	jo daviess	5.18	0.91	1.16	0.33	0.00	0.16	0.07	0.81	0.00	0.00	0.02	0.00	2.65	0.39	0.02	0.11	11.83
17087	illinois	johnson	0.26	0.14	0.26	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.03	0.00	0.17	0.22	0.01	0.05	1.16
17089	illinois	kane	12.04	3.71	5.24	0.37	0.00	0.10	0.02	0.02	0.00	0.01	0.04	0.03	1.52	4.17	0.01	0.74	28.03
17091	illinois	kankakee	12.78	1.17	5.36	0.35	0.00	0.05	0.02	0.06	0.01	0.02	0.07	0.02	0.40	3.62	0.88	3.11	27.90
17093	illinois	kendall	8.00	0.29	5.87	0.12	0.00	0.04	0.01	0.05	0.00	0.01	0.07	0.02	1.08	1.03	0.06	0.21	16.87
17095	illinois	knox	6.96	0.33	3.26	0.08	0.00	0.06	0.01	0.00	0.00	0.01	0.02	0.00	0.68	3.50	0.02	0.08	15.01
17099	illinois	la salle	18.31	2.00	10.91	0.22	0.00	0.05	0.02	0.00	0.01	0.02	0.17	0.05	0.99	0.10	0.00	0.22	33.07
17097	illinois	lake	0.82	0.29	0.76	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.22	0.00	0.01	0.76	3.02
17101	illinois	lawrence	2.81	0.59	2.05	0.26	0.00	0.22	0.01	0.03	0.00	0.04	0.03	0.01	0.07	1.13	0.68	0.03	7.96
17103	illinois	lee	12.94	0.19	3.47	0.10	0.00	0.04	0.02	0.00	0.00	0.01	0.13	0.03	0.78	1.69	0.00	0.06	19.47
17105	illinois	livingston	18.24	0.22	11.25	0.50	0.00	0.12	0.03	0.01	0.01	0.02	0.04	0.02	0.70	8.90	0.09	0.10	40.26
17107	illinois	logan	6.80	0.76	4.49	0.07	0.00	0.05	0.01	0.00	0.00	0.01	0.03	0.02	0.29	1.57	0.02	0.07	14.20
17115	illinois	macon	6.83	0.36	4.19	0.04	0.00	0.02	0.01	0.00	0.00	0.01	0.04	0.01	0.48	0.33	0.00	0.07	12.41
17117	illinois	macoupin	8.55	0.26	3.81	0.22	0.00	0.03	0.01	0.08	0.00	0.01	0.06	0.01	0.88	0.99	0.03	0.06	15.01
17119	illinois	madison	9.06	0.47	5.73	1.06	0.00	0.11	0.02	0.78	0.01	0.03	0.06	0.02	1.08	0.46	0.04	0.21	19.12
17121	illinois	marion	2.16	0.98	1.86	0.29	0.00	0.17	0.01	0.13	0.00	0.00	0.03	0.01	0.25	0.47	0.35	1.26	7.98
17123	illinois	marshall	4.56	0.63	2.41	0.06	0.00	0.03	0.01	0.00	0.00	0.00	0.04	0.00	0.27	0.58	0.00	0.04	8.64
17125	illinois	mason	5.51	0.08	2.62	0.16	0.00	1.17	0.01	0.01	0.00	0.01	0.13	0.01	0.15	0.94	0.00	0.02	10.83
17127	illinois	massac	0.97	0.15	0.95	0.08	0.00	0.01	0.00	0.08	0.00	0.00	0.04	0.01	0.14	0.43	0.00	0.05	2.90
17109	illinois	mcdonough	5.01	0.21	2.58	0.05	0.00	0.02	0.01	0.00	0.00	0.01	0.01	0.00	0.34	0.96	0.02	0.05	9.29
17111	illinois	mchenry	13.26	1.41	4.90	0.45	0.00	0.08	0.02	0.00	0.00	0.01	0.17	0.02	3.23	1.41	0.03	0.89	25.90
17113	illinois	mclean	16.98	0.95	11.93	0.17	0.00	0.12	0.04	0.01	0.01	0.03	0.06	0.03	1.19	6.83	0.00	0.13	38.49
17129	illinois	menard	2.83	0.10	2.23	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.01	0.14	0.09	0.00	0.05	5.59
17131	illinois	mercer	5.32	0.26	2.37	0.05	0.00	0.05	0.01	0.00	0.02	0.01	0.03	0.03	0.34	2.47	0.00	0.05	11.00
17133	illinois	monroe	4.60	0.25	3.28	1.29	0.00	0.08	0.01	0.78	0.03	0.05	0.07	0.00	0.60	1.35	0.02	0.10	12.49
17135	illinois	montgomery	7.04	0.17	3.84	0.30	0.00	0.05	0.01	0.05	0.00	0.01	0.08	0.01	0.31	3.11	0.00	0.07	15.05
17137	illinois	morgan	5.94	0.15	2.98	0.09	0.00	0.02	0.01	0.00	0.00	0.01	0.03	0.00	0.53	0.74	0.02	0.05	10.58
17139	illinois	moultrie	3.79	0.10	2.38	0.02	0.00	0.02	0.01	0.00	0.00	0.00	0.01	0.00	0.18	0.04	0.05	0.18	6.79
17141	illinois	ogle	14.75	0.51	4.05	0.28	0.00	0.11	0.02	0.01	0.01	0.01	0.15	0.02	2.60	4.50	0.68	0.56	28.25
17143	illinois	peoria	6.33	0.34	3.24	0.11	0.00	0.04	0.01	0.00	0.00	0.01	0.10	0.02	0.64	1.01	0.00	0.12	11.99
17145	illinois	perry	1.89	0.19	1.79	0.45	0.00	0.05	0.00	0.60	0.00	0.00	0.01	0.00	0.34	0.00	0.02	0.02	5.37
17147	illinois	piatt	5.39	0.05	3.71	0.05	0.00	0.01	0.01	0.00	0.00	0.01	0.03	0.00	0.04	0.33	0.00	0.02	9.65
17149	illinois	pike	5.33	0.31	2.55	0.19	0.00	0.07	0.01	0.02	0.01	0.01	0.02	0.00	0.32	5.62	0.01	0.04	14.51
17151	illinois	pope	0.22	0.10	0.27	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.00	0.16	0.00	0.00	0.05	0.85
17153	illinois	pulaski	0.52	0.07	0.75	0.07	0.00	0.00	0.00	0.02	0.00	0.01	0.11	0.00	0.09	0.00	0.00	0.02	1.65
17155	illinois	putnam	1.06	0.36	0.61	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.06	0.00	0.00	0.01	2.15
17157	illinois	randolph	5.03	0.42	4.12	1.36	0.00	0.74	0.01	0.77	0.01	0.01	0.04	0.00	0.87	0.08	0.00	0.11	13.58
17159	illinois	richland	3.09	0.08	1.90	0.22	0.00	0.11	0.01	0.16	0.00	0.00	0.07	0.00	0.20	1.31	0.35	0.02	7.54
17161	illinois	rock island	4.06	0.34	1.76	0.04	0.00	0.02	0.01	0.00	0.01	0.01	0.05	0.00	0.63	0.51	0.00	0.15	7.59
17165	illinois	saline	1.08	0.75	1.14	0.06	0.00	0.02	0.00	0.01	0.00	0.00	0.02	0.01	0.11	0.62	0.07	0.17	4.07
17167	illinois	sangamon	11.39	1.76	6.58	0.10	0.00	0.03	0.02	0.00	0.01	0.02	0.05	0.01	0.57	1.35	0.00	0.19	22.08
17169	illinois	schuyler	2.32	0.16	0.93	0.06	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.18	1.32	0.00	0.02	5.03
17171	illinois	scott	1.85	0.06	0.81	0.07	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.00	0.13	0.28	0.02	0.06	3.31
17173	illinois	shelby	7.25	0.23	4.59	0.24	0.00	0.06	0.01	0.05	0.00	0.01	0.03	0.02	0.71	1.89	0.00	0.07	15.18
17163	illinois	st clair	9.00	0.34	5.44	1.30	0.00	0.11	0.02	0.09	0.00	0.01	0.03	0.00	0.72	0.66	0.00	0.22	17.94
17175	illinois	stark	3.54	0.41	2.33	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.04	0.00	0.12	0.33	0.02	0.02	6.84

17177	illinois	stephenson	10.64	0.90	3.13	0.23	0.00	0.14	0.02	0.00	0.00	0.01	0.02	0.00	4.40	2.66	0.54	2.15	24.83
17179	illinois	tazewell	6.06	1.50	4.18	0.15	0.00	0.17	0.01	0.00	0.00	0.01	0.16	0.03	0.60	1.88	0.04	0.15	14.94
17181	illinois	union	0.43	0.18	0.66	0.04	0.00	0.01	0.00	0.02	0.00	0.00	0.03	0.00	0.17	0.11	0.00	0.05	1.70
17183	illinois	vermilion	10.04	0.19	6.67	0.14	0.00	0.02	0.02	0.01	0.00	0.02	0.04	0.02	0.34	0.37	0.00	0.10	17.97
17185	illinois	wabash	1.85	0.03	1.58	0.16	0.00	0.15	0.00	0.01	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.02	3.87
17187	illinois	warren	6.89	0.16	3.22	0.04	0.00	0.03	0.01	0.00	0.01	0.01	0.02	0.01	0.56	1.68	0.00	0.10	12.73
17189	illinois	washington	5.20	0.31	3.68	1.61	0.00	0.08	0.01	1.03	0.00	0.01	0.32	0.00	1.46	1.47	0.00	0.02	15.21
17191	illinois	wayne	3.66	1.21	2.85	0.39	0.00	0.13	0.01	0.08	0.00	0.01	0.20	0.01	0.47	1.38	0.32	0.07	10.78
17193	illinois	white	2.52	0.83	2.63	0.27	0.00	0.15	0.01	0.04	0.00	0.00	0.20	0.00	0.11	0.31	0.00	0.05	7.13
17195	illinois	whiteside	13.15	0.28	2.98	0.15	0.00	0.11	0.02	0.00	0.01	0.01	0.06	0.03	2.21	3.84	0.51	0.35	23.69
17197	illinois	will	17.80	7.58	14.94	1.17	0.00	0.11	0.04	0.06	0.01	0.03	0.07	0.04	1.03	2.49	0.04	1.34	46.77
17199	illinois	williamson	0.71	0.17	0.81	0.04	0.00	0.02	0.00	0.03	0.00	0.00	0.02	0.00	0.16	0.84	0.39	0.20	3.40
17201	illinois	winnebago	8.02	0.64	3.05	0.34	0.00	0.07	0.01	0.03	0.00	0.01	0.02	0.02	1.67	0.31	0.01	0.28	14.48
17203	illinois	woodford	7.15	0.23	4.90	0.16	0.00	0.06	0.01	0.00	0.00	0.01	0.02	0.01	0.35	3.83	0.55	1.86	19.17
18001	indiana	adams	4.07	0.37	2.69	0.42	0.00	0.12	0.01	0.00	0.00	0.00	0.00	0.00	1.70	5.98	1.23	0.90	17.48
18003	indiana	allen	5.00	0.67	4.57	0.81	0.00	0.06	0.01	0.00	0.00	0.00	0.01	0.00	1.04	2.71	0.11	0.58	15.57
18005	indiana	bartholomew	2.52	0.17	1.93	0.18	0.00	0.09	0.00	0.02	0.00	0.00	0.04	0.00	0.27	0.77	0.00	0.12	6.13
18007	indiana	benton	4.12	0.15	2.72	0.09	0.00	0.03	0.01	0.00	0.00	0.00	0.01	0.00	0.29	0.62	0.00	0.02	8.07
18009	indiana	blackford	1.34	0.22	1.49	0.09	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.07	1.81	0.00	0.04	5.10
18011	indiana	boone	5.23	0.32	5.02	0.20	0.00	0.12	0.01	0.00	0.00	0.00	0.01	0.00	0.32	1.80	0.01	0.25	13.28
18013	indiana	brown	0.31	0.09	0.06	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.86	0.01	0.06	1.54
18015	indiana	carroll	6.59	0.30	2.71	0.31	0.00	0.17	0.01	0.00	0.00	0.00	0.01	0.00	0.33	14.52	1.52	0.10	26.60
18017	indiana	cass	3.63	0.49	2.19	0.14	0.00	0.03	0.00	0.00	0.00	0.00	0.03	0.00	0.49	2.46	0.01	0.08	9.56
18019	indiana	clark	1.35	0.48	1.89	0.19	0.00	0.09	0.00	0.02	0.00	0.07	0.01	0.00	0.87	0.04	4.30	0.34	9.66
18021	indiana	clay	1.77	0.13	1.81	0.05	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.30	0.73	0.01	0.08	4.93
18023	indiana	clinton	6.60	0.40	4.29	0.20	0.00	0.13	0.01	0.00	0.00	0.00	0.02	0.00	0.21	11.08	1.73	0.11	24.78
18025	indiana	crawford	0.26	0.08	0.06	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.63	0.00	0.03	1.27
18027	indiana	daviess	3.75	0.26	1.43	0.16	0.00	0.17	0.00	0.01	0.00	0.01	0.01	0.00	0.56	3.13	0.19	0.28	9.95
18033	indiana	de kalb	1.61	0.31	1.78	0.29	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.75	0.54	0.01	0.13	5.45
18029	indiana	dearborn	0.41	0.19	0.23	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.42	0.02	0.00	0.14	1.46
18031	indiana	decatur	4.60	0.24	2.65	0.22	0.00	0.12	0.01	0.01	0.00	0.00	0.00	0.00	0.60	8.87	0.00	0.11	17.43
18035	indiana	delaware	3.98	0.71	3.43	0.12	0.00	0.09	0.00	0.00	0.00	0.00	0.04	0.00	0.47	1.02	0.02	0.16	10.04
18037	indiana	dubois	2.36	0.23	0.92	0.14	0.00	0.08	0.01	0.02	0.00	0.00	0.02	0.00	0.79	3.71	1.29	0.05	9.62
18039	indiana	elkhart	4.45	0.68	1.53	0.21	0.00	0.12	0.01	0.00	0.00	0.00	0.01	0.00	4.70	3.50	1.11	4.40	20.74
18041	indiana	fayette	1.45	0.15	0.88	0.06	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.86	0.00	0.08	3.85
18043	indiana	floyd	0.58	0.13	0.23	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.29	1.50	0.02	0.17	2.99
18045	indiana	fountain	3.03	0.15	2.19	0.07	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.63	0.00	0.06	6.43
18047	indiana	franklin	1.76	0.32	1.06	0.08	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.67	1.07	0.02	0.12	5.14
18049	indiana	fulton	2.53	0.18	1.44	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.01	0.00	0.57	1.44	0.00	0.10	6.40
18051	indiana	gibson	3.45	0.12	2.16	0.65	0.00	0.05	0.00	0.05	0.00	0.17	0.10	0.00	0.20	0.68	0.00	0.04	7.67
18053	indiana	grant	3.45	0.18	3.67	0.16	0.00	0.12	0.00	0.00	0.00	0.00	0.03	0.00	0.20	1.03	0.02	0.07	8.92
18055	indiana	greene	1.54	0.21	0.97	0.05	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.65	0.06	0.12	4.10
18057	indiana	hamilton	3.09	1.27	3.25	0.15	0.00	0.02	0.00	0.00	0.00	0.00	0.02	0.00	0.62	0.59	0.00	0.34	9.36
18059	indiana	hancock	3.96	0.27	3.55	0.18	0.00	0.04	0.01	0.00	0.00	0.02	0.00	0.00	0.27	3.36	0.00	0.21	11.90
18061	indiana	harrison	1.12	0.37	0.63	0.07	0.00	0.16	0.00	0.01	0.00	0.00	0.00	0.00	0.90	0.85	0.02	0.26	4.39
18063	indiana	hendricks	4.42	1.19	4.37	0.24	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	0.45	1.36	0.01	0.32	12.41
18065	indiana	henry	3.19	0.29	2.66	0.15	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.86	0.87	0.02	0.16	8.23
18067	indiana	howard	3.85	0.40	2.70	0.16	0.00	0.05	0.01	0.00	0.00	0.00	0.01	0.00	0.20	4.45	0.00	0.10	11.93
18069	indiana	huntington	3.26	0.21	2.78	0.22	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.66	2.36	0.34	0.14	10.09
18071	indiana	jackson	2.72	0.23	1.62	0.11	0.00	0.08	0.00	0.03	0.00	0.04	0.04	0.00	0.80	2.66	1.57	0.13	10.05
18073	indiana	jasper	5.49	0.37	1.97	0.12	0.00	0.21	0.01	0.02	0.00	0.01	0.01	0.00	3.28	1.53	0.01	0.06	13.09
18075	indiana	jay	4.07	0.42	2.52	0.40	0.00	0.10	0.01	0.00	0.00	0.00	0.01	0.00	0.57	8.81	3.69	0.12	20.74
18077	indiana	jefferson	0.60	0.40	0.69	0.04	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.35	0.81	0.00	0.17	3.11
18079	indiana	jennings	1.34	0.50	1.06	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.27	1.39	1.52	0.12	6.31
18081	indiana	johnson	3.11	0.24	2.24	0.13	0.00	0.05	0.00	0.01	0.00	0.00	0.02	0.00	0.43	0.42	0.01	0.27	6.94
18083	indiana	knox	4.26	0.12	2.84	0.38	0.00	0.08	0.00	0.04	0.00	0.19	0.07	0.00	0.21	1.08	0.03	0.04	9.35
18085	indiana	kosciusko	4.72	0.52	2.72	0.21	0.00	0.09	0.01	0.01	0.00	0.00	0.01	0.00	1.42	4.68	1.70	4.86	20.95
18091	indiana	la porte	7.26	0.47	2.75	0.43	0.00	0.18	0.02	0.06	0.00	0.01	0.03	0.00	1.63	13.20	0.00	0.21	26.25
18087	indiana	lagrange	2.59	1.15	0.79	0.13	0.00	0.08	0.01	0.00	0.00	0.00	0.02	0.00	3.24	1.61	1.38	4.18	15.17
18089	indiana	lake	3.42	0.22	2.24	0.12	0.00	0.01	0.01	0.00	0.00	0.01	0.02	0.00	0.42	0.02	0.01	0.28	6.78
18093	indiana	lawrence	0.71	0.33	0.44	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.58	0.00	0.01	0.13	2.25
18095	indiana	madison	4.86	0.65	4.49	0.10	0.00	0.02	0.01	0.00	0.00	0.00	0.03	0.00	0.52	0.99	0.00	0.22	11.88
18097	indiana	marion	0.59	0.47	0.35	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.27	0.00	0.03	0.28	2.04
18099	indiana	marshall	4.08	0.47	2.25	0.14	0.00	0.07	0.01	0.00	0.00	0.00	0.01	0.00	1.68	0.29	0.01	0.63	9.64
18101	indiana	martin	0.54	0.06	0.19	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.18	1.02	1.19	0.06	3.30
18103	indiana	miami	3.48	0.25	2.25	0.22	0.00	0.10	0.01	0.00	0.00	0.01	0.02	0.00	0.55	6.67	0.00	0.11	13.67
18105	indiana	monroe	0.34	0.19	0.20	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.31	0.01	0.02	0.15	1.23
18107	indiana	montgomery	5.76	0.19	3.92	0.13	0.00	0.06	0.01	0.00	0.00	0.00	0.01	0.00	0.30	3.76	0.01	0.12	14.27
18109	indiana	morgan	2.37	0.29	1.84	0.06	0.00	0.03	0.00	0.01	0.00	0.00	0.02	0.00	0.44	0.26	0.00	0.26	5.59
18111	indiana	newton	3.87	0.40	1.64	0.11	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	0.36	2.57	1.51	0.04	10.57
18113	indiana	noble	3.32	0.43	1.78	0.21	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	1.15	5.81	1.81	1.28	15.88
18115	indiana	ohio	0.38	0.07	0.10	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.93	0.01	0.05	1.97
18117	indiana	orange	0.78	0.20	0.36	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.30	0.66	1.31	0.08	3.74
18119	indiana	owen	0.83	0.18	0.49	0.04	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.25	0.73	0.00	0.12	2.69
18121	indiana	parke	1.89	0.18	1.39	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.02	0.01	0.09	4.06
18123	indiana	perry	0.52	0.10	0.15	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.87	0.01	0.04	1.99
18125	indiana	pike	0.92	0.06	0.51	0.05	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.05	0.88	0.02	0.02	2.56
18127	indiana	porter	2.96	0.25	1.94	0.10	0.00	0.03	0.01	0.00	0.00	0.00	0.02	0.00	0.31	0.64	0.00	0.17	6.44
18129	indiana	posey	3.68	0.09	2.03	0.70	0.00	0.03	0.00	0.10	0.00	0.17	0.18	0.00	0.15	1.76	0.01	0.03	8.94
18131	indiana	pulaski	2.74	0.17	1.49	0.04	0.00	0.46	0.01	0.03	0.00	0.02	0.00	0.00	0.35	1.41	1.35	0.06	8.14
18133	indiana	putnam	2.62	0.37	2.13	0.17	0.00	0.03	0.00	0.01	0.00	0.00	0.01	0.00	0.39	1.71	0.00	0.23	7.69
18135	indiana	randolph	6.93	0.26	4.72	0.41	0.00	0.14	0.02	0.00	0.00	0.00	0.05	0.00	0.59	12.66	0.03	0.15	25.97
18137	indiana	ripley	1.96	0.65	1.62	0.10	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.46	2.19	0.01	0.14	7.19
18139	indiana	rush	5.71	0.19	3.47	0.25	0.00	0.08	0.01	0.03	0.00	0.00	0.01	0.00	0.93	8.51	0.01	0.12	19.32
18143	indiana	scott	0.40	0.09	0.47	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.14	0.00	0.00	0.10	1.25
18145	indiana	shelby	4.87	0.42	4.06	0.13	0.00	0.07	0.01	0.01	0.00	0.00	0.03	0.00	0.28	2.36	0.02	0.14	12.41
18147	indiana	spencer	2.46	0.21	1.46	0.11	0.00	0.04	0.00	0.01	0.00	0.02	0.02	0.00	0.44	1.60	1.33	0.07	7.76
18141	indiana	st joseph	3.55	0.65	1.78	0.18	0.00	0.04	0.01	0.06	0.00	0.00	0.02	0.00	0.56	2.11	2.35	0.25	11.55
18149	indiana	starke	1.30	0.28	0.56	0.02	0.00	0.04	0.00	0.14	0.00	0.00	0.02	0.00	0.14	0.01	0.00	0.05	2.56
18151	indiana	steuben	1.09	0.73	0.68	0.12	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.78	0.00	0.11	4.14
18153	indiana	sullivan	1.88	0.64	1.57	0.07	0.00	0.03	0.00	0.01	0.00	0.05	0.01	0.00	0.15	1.67	0.01	0.05	6.16
18155	indiana	switzerland	0.47	0.13	0.21	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.20	0.89	0.00	0.07	2.03
18157	indiana	tippecanoe	4.16	0.24	3.12	0.13	0.00	0.11	0.00	0.01	0.00	0.00	0.02	0.00	0.28	2.06	0.00	0.16	10.29
18159	indiana	tipton	3.17	0.14	2.67	0.10	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	0.08	4.46	1.83	0.05	12.54
18161	indiana	union	1.62	0.10	1.04	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.98	0.00	0.03	4.06
18163	indiana	vanderburgh	1.19	0.15	1.00	0.22	0.00	0.01	0.00	0.03	0.00	0.02	0.04	0.00	0.18	0.17	1.95	0.07	5.02
18165	indiana	vermillion	1.88	0.10	1.11	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.00	0.14	1.91	0.00	0.05	5.28
18167	indiana	vigo	1.63	0.13	1.68	0.05	0.00	0.07	0.00	0.01	0.00	0.01	0.00	0.00	0.14	0.01	0.02	0.09	3.85
18169	indiana	wabash	4.53	0.22	2.69	0.27	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	1.06	8.19	1.58	0.11	18.79
18171	indiana	warren	2.49	0.13	1.75	0.11	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.63	0.00	0.07	5.37
18173	indiana	warrick	1.78	0.17	0.98	0.09	0.00	0.01	0.00	0.02	0.00	0.03	0.09	0.00	0.17	0.25	0.01	0.06	3.66
18175	indiana	washington	2.69	0.32	1.34	0.09	0.00	0.08	0.01	0.01	0.00	0.00	0.01	0.00	1.00	0.68	1.55	0.20	7.98
18177	indiana	wayne	3.06	0.37	1.99	0.12	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.00	0.86	0.97	0.01	0.17	7.61
18179	indiana	wells	4.22	0.16	3.30	0.32	0.00	0.10	0.01	0.00	0.00	0.00	0.02	0.00	0.57	6.08	0.52	0.08	15.37
18181	indiana	white	6.00	0.20	2.70	0.17	0.00	0.52	0.01	0.00	0.00	0.00	0.02	0.00	0.36	9.20	1.24	0.07	20.51
18183	indiana	whitley	2.51	0.25	1.65	0.27	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.51	2.67	1.62	0.25	9.76
19001	iowa	adair	1.87	0.50	1.47	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.03	0.78	0.68	0.04	6.43
19003	iowa	adams	1.34	0.36	0.84	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.57	1.13	0.01	0.02	4.33
19005	iowa	allamakee	2.86	0.64	0.61	0.08	0.00	0.12	0.01	0.00	0.00	0.00	0.00	0.00	2.50	3.34	0.98	0.07	11.22
19007	iowa	appanoose	0.65	0.50	0.63	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.12	0.00	0.04	2.64
19009	iowa	audubon	3.11	0.23	1.50	0.07	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.77	5.92	0.64	0.04	12.35
19011	iowa	benton	6.70	0.44	4.25	0.08	0.00	0.09	0.01	0.00	0.00	0.00	0.01	0.00	1.81	5.26	1.19	0.17	20.02
19013	iowa	black hawk	5.97	0.45	2.61	0.11	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	1.20	9.15	0.00	0.08	19.71
19015	iowa	boone	3.84	0.33	1.74	0.05	0.00	0.05	0.01	0.00	0.00	0.00	0.01	0.00	0.49	3.66	0.01	0.10	10.28
19017	iowa	bremer	3.92	0.20	1.37	0.09	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	1.15	6.64	0.00	0.09	13.56
19019	iowa	buchanan	5.29	0.18	1.30	0.14	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.00	1.21	11.89	0.03	0.16	20.35
19021	iowa	buena vista	4.24	0.10	2.05	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.79	8.70	0.72	0.07	16.87
19023	iowa	butler	4.72	0.34	1.67	0.12	0.00	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.79	10.40	0.00	0.09	18.26
19025	iowa	calhoun	4.38	0.15	2.07	0.09	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.50	7.79	0.00	0.02	15.12
19027	iowa	carroll	5.96	0.23	1.92	0.14	0.00	0.14	0.01	0.00	0.00	0.00	0.00	0.00	2.00	11.47	0.00	0.03	21.93
19029	iowa	cass	2.20	0.33	1.52	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.06	0.95	0.64	0.04	6.80
19031	iowa	cedar	5.48	0.63	2.67	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.01	0.00	0.99	7.64	0.00	0.09	17.71
19033	iowa	cerro gordo	4.44	0.32	2.06	0.05	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.34	3.90	0.00	0.04	11.20
19035	iowa	cherokee	3.39	0.15	1.82	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	1.23	5.29	0.03	0.07	12.12
19037	iowa	chickasaw	4.00	0.22	1.51	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	1.60	7.05	0.00	0.11	14.70
19039	iowa	clarke	0.94	0.37	0.45	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.63	2.02	0.00	0.06	4.53
19041	iowa	clay	2.93	0.12	1.68	0.05	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.71	3.71	0.65	0.06	9.96
19043	iowa	clayton	5.71	0.89	1.35	0.14	0.00	0.19	0.01	0.00	0.00	0.00	0.00	0.00	2.85	8.68	0.07	0.19	20.08
19045	iowa	clinton	8.20	0.66	4.04	0.10	0.00	0.10	0.01	0.00	0.00	0.00	0.01	0.00	4.20	2.91	0.00	0.14	20.37
19047	iowa	crawford	4.63	0.42	2.14	0.08	0.00	0.08	0.01	0.00	0.00	0.00	0.01	0.00	1.09	6.08	0.00	0.03	14.58
19049	iowa	dallas	4.50	0.52	2.60	0.04	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	0.89	2.15	0.52	0.09	11.38
19051	iowa	davis	1.80	0.54	0.76	0.07	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.85	4.20	0.03	0.17	8.50
19053	iowa	decatur	0.91	0.46	0.53	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.07	1.20	0.00	0.04	4.27
19055	iowa	delaware	8.01	0.54	1.46	0.19	0.00	0.22	0.01	0.00	0.00	0.00	0.00	0.00	4.24	12.26	0.01	0.09	27.05
19057	iowa	des moines	1.82	0.39	1.34	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.95	0.00	0.05	4.87
19059	iowa	dickinson	1.31	0.20	0.88	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.82	0.00	0.04	3.78
19061	iowa	dubuque	6.32	1.08	1.02	0.17	0.00	0.25	0.02	0.00	0.00	0.00	0.00	0.00	6.79	5.70	0.00	0.16	21.52
19063	iowa	emmet	2.26	0.16	1.13	0.05	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.55	4.04	0.67	0.03	8.92
19065	iowa	fayette	6.01	0.65	2.41	0.12	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.00	2.49	7.78	0.97	0.14	20.72
19067	iowa	floyd	3.80	0.29	1.86	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.90	4.84	0.94	0.07	12.83
19069	iowa	franklin	6.24	0.22	1.82	0.16	0.00	0.14	0.02	0.00	0.00	0.00	0.01	0.00	0.45	14.95	0.01	0.03	24.06
19071	iowa	fremont	1.95	0.42	1.99	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.32	0.01	0.02	5.05
19073	iowa	greene	4.23	0.24	2.10	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.55	4.65	0.00	0.03	11.93
19075	iowa	grundy	4.41	0.10	2.62	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.74	5.50	0.00	0.05	13.55
19077	iowa	guthrie	2.70	0.49	1.61	0.05	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.93	2.99	0.73	0.04	9.60
19079	iowa	hamilton	5.73	0.20	1.74	0.16	0.00	0.14	0.02	0.00	0.00	0.00	0.01	0.00	0.20	15.73	0.87	0.05	24.86
19081	iowa	hancock	4.75	0.21	1.70	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.41	8.87	3.42	0.03	19.59
19083	iowa	hardin	6.92	0.28	1.96	0.20	0.00	0.17	0.02	0.00	0.00	0.00	0.02	0.00	0.68	18.22	0.81	0.08	29.36
19085	iowa	harrison	3.69	0.61	2.03	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.73	0.00	0.04	7.87
19087	iowa	henry	2.57	0.51	1.77	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.64	2.13	0.05	0.10	7.85
19089	iowa	howard	3.89	0.24	1.80	0.08	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	1.24	5.61	0.00	0.06	13.03
19091	iowa	humboldt	2.44	0.13	1.11	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.28	3.41	0.40	0.07	7.92
19093	iowa	ida	2.36	0.22	1.30	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.69	2.59	0.01	0.03	7.28
19095	iowa	iowa	3.99	0.92	2.36	0.07	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.84	3.74	0.00	0.10	13.11
19097	iowa	jackson	4.39	1.17	1.32	0.09	0.00	0.14	0.01	0.00	0.00	0.00	0.01	0.00	4.80	1.86	0.00	0.18	13.96
19099	iowa	jasper	5.95	0.47	3.29	0.09	0.00	0.08	0.01	0.00	0.00	0.00	0.01	0.00	0.86	7.67	0.00	0.16	18.61
19101	iowa	jefferson	1.58	0.54	1.17	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.57	1.10	0.02	0.08	5.13
19103	iowa	johnson	4.52	1.12	2.84	0.09	0.00	0.13	0.01	0.00	0.00	0.00	0.01	0.00	1.71	4.93	0.17	0.34	15.87
19105	iowa	jones	4.35	0.48	1.53	0.07	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	2.56	2.52	0.00	0.08	11.66
19107	iowa	keokuk	3.41	0.66	1.68	0.09	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.90	6.97	0.01	0.08	13.89
19109	iowa	kossuth	7.55	0.31	3.42	0.15	0.00	0.13	0.01	0.00	0.00	0.00	0.01	0.00	0.75	13.39	0.01	0.03	25.78
19111	iowa	lee	2.12	0.56	1.29	0.07	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.75	2.26	0.00	0.07	7.15
19113	iowa	linn	6.04	0.58	3.26	0.07	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	1.65	4.07	0.00	0.20	16.01
19115	iowa	louisa	3.17	0.51	1.45	0.08	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.27	7.01	0.01	0.06	12.67
19117	iowa	lucas	0.78	0.44	0.46	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.92	0.00	0.06	3.59
19119	iowa	lyon	6.80	0.33	2.30	0.21	0.00	0.18	0.02	0.00	0.00	0.00	0.00	0.00	4.25	14.38	0.63	0.08	29.18
19121	iowa	madison	1.65	0.55	1.22	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.87	0.88	0.82	0.08	6.13
19123	iowa	mahaska	4.19	0.59	2.00	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.01	0.00	0.97	8.24	0.87	0.05	17.13
19125	iowa	marion	1.71	0.63	1.34	0.03	0.00	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.87	0.71	0.00	0.10	5.45
19127	iowa	marshall	4.15	0.24	2.37	0.07	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.89	5.34	0.00	0.10	13.25
19129	iowa	mills	1.99	0.51	1.70	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.13	0.00	0.03	4.58
19131	iowa	mitchell	4.94	0.22	1.99	0.10	0.00	0.11	0.01	0.01	0.00	0.00	0.00	0.00	2.22	6.74	0.21	0.09	16.65
19133	iowa	monona	2.62	0.47	1.64	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.34	0.00	0.02	5.59
19135	iowa	monroe	0.70	0.42	0.59	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.12	0.00	0.03	2.72
19137	iowa	montgomery	1.65	0.43	1.36	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.10	0.66	0.02	4.84
19139	iowa	muscatine	3.25	0.27	1.88	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.01	0.00	0.76	3.77	0.00	0.06	10.14
19141	iowa	obrien	4.55	0.09	1.95	0.12	0.00	0.11	0.01	0.00	0.00	0.00	0.00	0.00	1.75	9.39	0.60	0.06	18.65
19143	iowa	osceola	2.87	0.09	1.27	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.00	5.92	0.56	0.03	11.90
19145	iowa	page	1.99	0.54	1.74	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.74	0.38	0.01	0.03	5.47
19147	iowa	palo alto	4.07	0.16	1.49	0.10	0.00	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.68	9.12	0.62	0.02	16.40
19149	iowa	plymouth	7.53	0.76	3.64	0.20	0.00	0.18	0.02	0.00	0.00	0.00	0.01	0.00	2.31	16.03	0.00	0.13	30.82
19151	iowa	pocahontas	3.03	0.17	1.60	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.43	4.80	0.66	0.02	10.84
19153	iowa	polk	2.71	0.72	1.91	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.01	0.00	0.39	0.27	0.02	0.12	6.19
19155	iowa	pottawattamie	7.85	1.19	5.58	0.08	0.00	0.09	0.01	0.00	0.01	0.00	0.01	0.00	2.49	3.02	0.00	0.11	20.44
19157	iowa	poweshiek	2.91	0.63	1.75	0.04	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	1.14	2.41	0.87	0.08	9.91
19159	iowa	ringgold	1.34	0.58	0.76	0.04	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	1.33	2.01	0.01	0.04	6.18
19161	iowa	sac	4.72	0.27	2.16	0.10	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.99	8.41	0.06	0.04	16.90
19163	iowa	scott	9.42	0.41	3.90	0.16	0.00	0.15	0.02	0.01	0.00	0.00	0.01	0.00	1.59	12.37	0.01	0.15	28.19
19165	iowa	shelby	3.41	0.28	2.06	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.06	2.09	0.00	0.06	9.05
19167	iowa	sioux	10.26	0.17	2.89	0.35	0.00	0.30	0.03	0.00	0.01	0.00	0.00	0.00	8.77	21.33	3.03	0.27	47.39
19169	iowa	story	3.47	0.29	1.82	0.04	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	1.01	2.55	0.03	0.13	9.41
19171	iowa	tama	3.96	0.69	2.72	0.05	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	1.23	2.94	0.00	0.10	11.76
19173	iowa	taylor	1.54	0.43	1.09	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.66	1.53	0.69	0.04	6.06
19175	iowa	union	1.46	0.48	0.84	0.05	0.00	0.05	0.01	0.05	0.00	0.00	0.00	0.00	1.01	2.26	0.69	0.06	6.96
19177	iowa	van buren	1.40	0.52	0.87	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.66	2.44	0.00	0.10	6.09
19179	iowa	wapello	1.21	0.56	0.81	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.48	1.01	0.00	0.07	4.19
19181	iowa	warren	2.30	0.86	1.81	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.13	0.42	0.00	0.15	6.75
19183	iowa	washington	9.35	0.44	2.76	0.33	0.00	0.29	0.03	0.00	0.01	0.00	0.01	0.00	0.70	31.24	0.17	0.21	45.54
19185	iowa	wayne	1.03	0.52	1.08	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.85	0.22	0.00	0.05	3.81
19187	iowa	webster	4.58	0.40	2.29	0.10	0.00	0.06	0.02	0.13	0.00	0.00	0.01	0.00	0.28	4.57	0.78	0.03	13.25
19189	iowa	winnebago	3.24	0.28	1.26	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.30	3.75	0.77	0.02	9.73
19191	iowa	winneshiek	4.62	0.95	1.49	0.10	0.00	0.18	0.01	0.00	0.00	0.00	0.00	0.00	3.32	3.78	0.13	0.15	14.72
19193	iowa	woodbury	4.61	0.87	2.61	0.10	0.00	0.06	0.02	0.17	0.00	0.00	0.01	0.00	2.22	0.92	0.00	0.06	11.65
19195	iowa	worth	2.24	0.22	1.37	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.18	1.46	0.00	0.04	5.56
19197	iowa	wright	4.82	0.27	2.02	0.12	0.00	0.10	0.01	0.00	0.00	0.00	0.01	0.00	0.15	11.17	3.94	0.03	22.64
20001	kansas	allen	0.80	1.44	0.79	0.38	0.00	0.05	0.00	0.09	0.00	0.00	0.00	0.34	0.83	0.01	0.00	0.04	4.78
20003	kansas	anderson	1.65	1.42	1.49	0.50	0.00	0.07	0.00	0.13	0.00	0.00	0.01	0.28	1.06	0.94	0.00	0.08	7.63
20005	kansas	atchison	1.53	0.52	1.07	0.07	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.07	0.88	0.07	0.00	0.03	4.28
20007	kansas	barber	0.23	1.59	0.04	0.97	0.00	0.08	0.01	0.09	0.00	0.00	0.00	0.00	0.70	0.00	0.00	0.03	3.74
20009	kansas	barton	0.90	0.54	0.52	2.79	0.00	1.02	0.00	0.01	0.00	0.00	0.00	0.03	1.70	0.50	0.00	0.02	8.03
20011	kansas	bourbon	0.80	2.42	0.53	0.21	0.00	0.04	0.00	0.05	0.00	0.00	0.01	0.20	1.43	0.01	0.00	0.06	5.75
20013	kansas	brown	2.78	0.46	1.83	0.09	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.23	0.51	0.08	0.00	0.04	6.06
20015	kansas	butler	3.53	7.11	1.64	1.81	0.00	0.56	0.01	0.18	0.00	0.00	0.00	0.38	4.87	1.87	0.00	0.26	22.23
20017	kansas	chase	0.59	4.87	0.23	0.27	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.21	1.44	0.77	0.00	0.03	8.45
20019	kansas	chautauqua	0.29	2.91	0.08	0.07	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.09	0.66	0.01	0.00	0.05	4.20
20021	kansas	cherokee	1.50	1.20	1.88	1.53	0.00	0.10	0.00	0.51	0.05	0.00	0.01	0.22	0.76	0.95	0.01	0.06	8.79
20023	kansas	cheyenne	0.44	0.63	0.03	0.95	0.00	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.52	0.00	0.00	0.01	2.67
20025	kansas	clark	0.22	0.12	0.02	1.10	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.40	0.00	0.01	2.66

20027	kansas	clay	0.71	0.64	1.01	1.19	0.00	0.41	0.00	0.08	0.00	0.00	0.01	0.09	0.60	0.51	0.00	0.02	5.27
20029	kansas	cloud	0.51	0.45	0.56	1.34	0.00	0.43	0.00	0.06	0.00	0.00	0.01	0.05	0.57	0.57	0.00	0.01	4.57
20031	kansas	coffey	0.96	2.00	1.03	0.47	0.00	0.05	0.00	0.12	0.00	0.00	0.00	0.48	0.66	0.82	0.00	0.08	6.70
20033	kansas	comanche	0.18	0.17	0.02	0.88	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.46	0.44	0.00	0.01	2.31
20035	kansas	cowley	0.55	4.58	0.40	1.08	0.00	0.22	0.08	0.14	0.00	0.00	0.00	0.26	1.14	0.00	0.00	0.08	8.53
20037	kansas	crawford	1.41	1.51	1.04	0.63	0.00	0.05	0.00	0.29	0.01	0.00	0.01	0.19	1.13	0.01	0.00	0.05	6.33
20039	kansas	decatur	0.85	0.15	0.04	1.30	0.00	0.12	0.00	0.01	0.00	0.00	0.00	0.00	0.59	0.00	0.00	0.01	3.06
20041	kansas	dickinson	0.81	1.56	1.05	3.28	0.00	0.93	0.00	0.20	0.00	0.00	0.01	0.16	1.62	0.04	0.00	0.07	9.73
20043	kansas	doniphan	2.15	0.21	1.08	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.10	0.41	0.00	0.00	0.04	4.03
20045	kansas	douglas	1.31	1.61	1.35	0.29	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.23	1.03	0.01	0.01	0.19	6.10
20047	kansas	edwards	1.09	0.20	0.21	1.37	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.02	0.72	0.00	0.00	0.00	3.92
20049	kansas	elk	0.36	3.74	0.22	0.17	0.00	0.07	0.00	0.02	0.00	0.00	0.00	0.14	0.83	0.00	0.00	0.03	5.57
20051	kansas	ellis	0.13	0.28	0.03	1.29	0.00	0.83	0.00	0.01	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.02	2.97
20053	kansas	ellsworth	0.26	0.33	0.24	1.53	0.00	0.59	0.00	0.01	0.00	0.00	0.00	0.00	0.41	0.55	0.00	0.01	3.94
20055	kansas	finney	1.51	1.19	0.11	1.89	0.00	0.74	0.00	0.00	0.00	0.00	0.00	0.02	2.26	0.00	0.00	0.01	7.74
20057	kansas	ford	1.49	0.31	0.14	2.86	0.00	1.28	0.02	0.17	0.00	0.00	0.01	0.00	1.69	0.40	0.00	0.02	8.38
20059	kansas	franklin	1.54	1.83	2.24	0.56	0.00	0.07	0.00	0.11	0.00	0.00	0.00	0.30	1.56	0.43	0.00	0.12	8.79
20061	kansas	geary	0.38	0.96	0.22	0.23	0.00	0.08	0.00	0.02	0.00	0.00	0.01	0.01	0.38	0.84	0.00	0.02	3.14
20063	kansas	gove	0.73	0.59	0.05	1.04	0.00	0.39	0.00	0.01	0.00	0.00	0.00	0.00	0.75	0.00	0.00	0.04	3.60
20065	kansas	graham	0.38	0.15	0.07	1.17	0.00	0.81	0.00	0.06	0.00	0.00	0.00	0.00	0.24	0.38	0.00	0.01	3.21
20067	kansas	grant	1.06	0.13	0.06	1.17	0.00	0.21	0.00	0.01	0.00	0.00	0.00	0.00	2.10	0.33	0.00	0.01	5.08
20069	kansas	gray	1.81	0.60	0.12	1.49	0.00	0.90	0.01	0.01	0.00	0.00	0.00	0.00	3.08	0.37	0.00	0.01	8.41
20071	kansas	greeley	0.26	0.05	0.02	1.68	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.30	0.00	0.00	2.90
20073	kansas	greenwood	0.83	4.38	0.50	0.25	0.00	0.08	0.00	0.02	0.00	0.00	0.01	0.12	1.83	0.75	0.00	0.19	8.97
20075	kansas	hamilton	0.50	0.14	0.04	1.49	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.00	1.42	0.00	0.00	0.01	3.91
20077	kansas	harper	0.42	1.76	0.12	3.05	0.00	0.23	0.03	0.07	0.00	0.00	0.00	0.01	1.10	0.62	0.00	0.02	7.43
20079	kansas	harvey	1.39	0.55	0.92	1.99	0.00	0.54	0.02	0.14	0.00	0.00	0.01	0.15	0.99	0.44	0.00	0.07	7.20
20081	kansas	haskell	2.03	0.15	0.15	1.13	0.00	0.42	0.01	0.01	0.00	0.00	0.00	0.00	4.44	0.00	0.00	0.00	8.34
20083	kansas	hodgeman	0.35	0.93	0.03	1.14	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.89	0.00	0.00	0.01	3.66
20085	kansas	jackson	0.88	1.48	0.56	0.13	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.15	1.37	0.02	0.00	0.09	4.75
20087	kansas	jefferson	1.43	1.31	1.01	0.17	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.09	1.14	0.03	0.00	0.11	5.38
20089	kansas	jewell	0.45	0.36	0.79	1.59	0.00	0.66	0.00	0.01	0.00	0.00	0.00	0.02	0.50	0.49	0.00	0.03	4.90
20091	kansas	johnson	1.45	1.30	1.51	0.36	0.00	0.04	0.00	0.06	0.00	0.00	0.00	0.06	1.19	3.50	0.01	0.50	10.00
20093	kansas	kearny	0.54	0.35	0.04	1.32	0.00	0.37	0.00	0.01	0.00	0.00	0.00	0.00	0.70	0.32	0.00	0.01	3.68
20095	kansas	kingman	0.46	0.69	0.17	3.24	0.00	0.29	0.01	0.02	0.00	0.00	0.00	0.01	0.86	0.64	0.00	0.06	6.44
20097	kansas	kiowa	0.28	0.86	0.08	0.69	0.00	0.20	0.01	0.00	0.00	0.00	0.00	0.00	0.33	0.43	0.00	0.01	2.90
20099	kansas	labette	1.38	1.67	1.02	1.02	0.00	0.10	0.00	0.28	0.01	0.00	0.01	0.12	2.07	0.02	0.00	0.15	7.86
20101	kansas	lane	0.35	0.52	0.03	1.20	0.00	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.74	0.35	0.00	0.01	3.60
20103	kansas	leavenworth	1.01	1.33	1.02	0.18	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.07	1.36	0.12	0.01	0.19	5.34
20105	kansas	lincoln	0.24	0.31	0.33	1.46	0.00	0.55	0.00	0.02	0.00	0.00	0.00	0.00	0.54	0.53	0.00	0.01	4.01
20107	kansas	linn	0.75	1.67	1.21	0.32	0.00	0.03	0.00	0.09	0.00	0.00	0.00	0.32	1.24	0.01	0.00	0.15	5.78
20109	kansas	logan	0.37	0.64	0.02	1.03	0.00	0.24	0.00	0.01	0.00	0.00	0.00	0.00	0.24	0.32	0.00	0.01	2.88
20111	kansas	lyon	1.41	4.10	1.32	0.53	0.00	0.09	0.00	0.03	0.00	0.00	0.00	0.39	1.52	0.81	0.00	0.08	10.30
20115	kansas	marion	1.87	1.95	1.13	3.28	0.00	0.86	0.01	0.12	0.00	0.00	0.00	0.14	2.36	0.20	0.00	0.07	12.00
20117	kansas	marshall	2.66	0.90	2.05	1.15	0.00	0.17	0.00	0.01	0.00	0.00	0.00	0.21	0.64	0.12	0.00	0.02	7.95
20113	kansas	mcpherson	1.22	0.88	1.10	4.90	0.00	0.76	0.00	0.26	0.00	0.00	0.01	0.12	0.99	0.53	0.00	0.12	10.89
20119	kansas	meade	1.09	1.00	0.08	0.98	0.00	0.24	0.00	0.02	0.00	0.00	0.00	0.00	0.62	0.37	0.00	0.01	4.40
20121	kansas	miami	1.61	3.54	1.97	0.49	0.00	0.13	0.00	0.09	0.00	0.00	0.01	0.19	1.66	0.06	0.01	0.32	10.10
20123	kansas	mitchell	0.41	0.29	0.50	2.28	0.00	0.60	0.00	0.03	0.00	0.00	0.00	0.01	0.60	0.51	0.00	0.01	5.25
20125	kansas	montgomery	0.99	1.68	0.76	0.73	0.00	0.08	0.00	0.26	0.00	0.00	0.01	0.12	0.90	0.84	0.00	0.09	6.46
20127	kansas	morris	0.73	2.85	0.54	0.61	0.00	0.14	0.00	0.01	0.00	0.00	0.00	0.09	1.65	0.00	0.00	0.03	6.65
20129	kansas	morton	0.25	0.45	0.01	0.88	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.32	0.00	0.00	2.30
20131	kansas	nemaha	3.40	1.01	1.73	0.32	0.00	0.07	0.01	0.02	0.00	0.00	0.00	0.22	1.40	5.72	0.00	0.02	13.92
20133	kansas	neosho	1.00	1.43	0.84	0.55	0.00	0.07	0.00	0.17	0.00	0.00	0.01	0.21	1.17	0.07	0.00	0.06	5.60
20135	kansas	ness	0.13	0.82	0.01	1.33	0.00	0.52	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00	0.00	0.01	3.17
20137	kansas	norton	0.98	0.18	0.16	1.11	0.00	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.66	1.93	0.00	0.02	5.26
20139	kansas	osage	1.58	2.35	1.93	0.38	0.00	0.09	0.00	0.07	0.00	0.00	0.00	0.46	1.31	0.01	0.00	0.09	8.26
20141	kansas	osborne	0.23	0.24	0.29	1.46	0.00	0.57	0.00	0.01	0.00	0.00	0.00	0.02	0.43	0.00	0.00	0.01	3.25
20143	kansas	ottawa	0.42	0.48	0.58	2.04	0.00	0.57	0.00	0.19	0.00	0.00	0.00	0.04	0.86	0.02	0.00	0.04	5.25
20145	kansas	pawnee	0.89	0.33	0.25	2.51	0.00	0.84	0.00	0.00	0.00	0.00	0.01	0.00	1.21	0.00	0.00	0.00	6.05
20147	kansas	phillips	0.53	0.30	0.27	1.34	0.00	0.60	0.00	0.09	0.00	0.00	0.00	0.00	0.68	0.64	0.00	0.02	4.38
20149	kansas	pottawatomie	1.49	3.53	0.73	0.18	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.17	1.38	0.29	0.00	0.08	7.95
20151	kansas	pratt	0.86	0.25	0.17	1.68	0.00	0.24	0.04	0.02	0.00	0.00	0.00	0.03	0.87	0.48	0.00	0.03	4.68
20153	kansas	rawlins	0.54	0.16	0.05	1.22	0.00	0.22	0.00	0.01	0.00	0.00	0.00	0.03	0.32	0.35	0.00	0.01	2.90
20155	kansas	reno	1.28	1.56	1.29	4.48	0.00	1.04	0.01	0.06	0.00	0.00	0.00	0.13	2.02	0.42	0.00	0.17	12.49
20157	kansas	republic	1.05	0.46	0.93	1.02	0.00	0.25	0.00	0.02	0.00	0.00	0.00	0.04	0.59	0.00	0.00	0.03	4.40
20159	kansas	rice	0.63	0.44	0.45	1.86	0.00	0.44	0.00	0.01	0.00	0.00	0.00	0.05	0.88	0.29	0.00	0.03	5.09
20161	kansas	riley	0.97	2.71	0.96	0.63	0.00	0.29	0.00	0.01	0.00	0.00	0.00	0.17	0.91	1.16	0.00	0.06	7.87
20163	kansas	rooks	0.26	0.22	0.23	1.29	0.00	0.86	0.00	0.01	0.00	0.00	0.00	0.00	0.57	0.00	0.00	0.01	3.45
20165	kansas	rush	0.23	0.22	0.10	2.05	0.00	0.84	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.43	0.00	0.01	4.31
20167	kansas	russell	0.21	0.28	0.14	1.31	0.00	0.69	0.00	0.01	0.00	0.00	0.01	0.00	0.41	0.47	0.00	0.01	3.54
20169	kansas	saline	0.27	0.74	0.59	3.00	0.00	0.57	0.00	0.05	0.00	0.03	0.00	0.03	0.62	0.01	0.00	0.07	5.98
20171	kansas	scott	1.04	0.29	0.08	1.35	0.00	0.69	0.00	0.00	0.00	0.00	0.00	0.00	2.35	0.83	0.00	0.01	6.66
20173	kansas	sedgwick	1.99	1.64	2.10	8.35	0.00	1.91	0.01	0.32	0.00	0.00	0.02	0.15	1.55	0.12	0.00	0.23	18.40
20175	kansas	seward	0.84	0.58	0.06	0.54	0.00	0.21	0.01	0.01	0.00	0.00	0.00	0.00	1.02	0.00	0.00	0.01	3.28
20177	kansas	shawnee	1.55	1.53	1.07	0.26	0.00	0.03	0.00	0.01	0.00	0.00	0.01	0.27	0.53	0.00	0.00	0.13	5.40
20179	kansas	sheridan	1.71	0.15	0.16	1.12	0.00	0.40	0.00	0.01	0.00	0.00	0.00	0.00	1.23	0.34	0.00	0.00	5.14
20181	kansas	sherman	0.72	0.53	0.04	1.72	0.00	0.12	0.00	0.06	0.00	0.00	0.00	0.01	0.28	0.32	0.00	0.01	3.83
20183	kansas	smith	0.50	0.25	0.70	2.13	0.00	0.79	0.00	0.02	0.00	0.00	0.01	0.00	0.51	0.48	0.00	0.02	5.41
20185	kansas	stafford	1.01	0.39	0.34	2.00	0.00	0.31	0.01	0.01	0.00	0.00	0.00	0.01	0.90	0.17	0.00	0.02	5.16
20187	kansas	stanton	0.42	0.06	0.01	0.98	0.00	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.31	0.00	0.00	2.28
20189	kansas	stevens	1.59	0.12	0.05	0.98	0.00	0.43	0.04	0.04	0.00	0.00	0.00	0.00	0.43	0.35	0.00	0.01	4.05
20191	kansas	sumner	1.20	3.65	1.14	9.10	0.00	0.94	0.12	0.83	0.00	0.00	0.01	0.20	0.85	0.04	0.00	0.07	18.15
20193	kansas	thomas	1.42	0.28	0.09	1.73	0.00	0.22	0.00	0.03	0.00	0.00	0.00	0.01	0.55	0.34	0.00	0.01	4.69
20195	kansas	trego	0.17	0.14	0.03	1.12	0.00	0.61	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.01	2.41
20197	kansas	wabaunsee	1.53	4.98	0.70	0.22	0.00	0.05	0.00	0.03	0.00	0.00	0.01	0.40	1.39	0.00	0.00	0.11	9.42
20199	kansas	wallace	0.43	0.07	0.01	0.88	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.01	0.15	0.30	0.00	0.01	2.00
20201	kansas	washington	1.45	0.80	1.35	1.18	0.00	0.50	0.00	0.04	0.00	0.00	0.00	0.13	1.41	1.99	0.00	0.03	8.88
20203	kansas	wichita	0.70	0.07	0.05	1.52	0.00	0.39	0.00	0.00	0.00	0.00	0.00	0.00	1.31	0.31	0.00	0.01	4.36
20205	kansas	wilson	0.90	1.48	0.80	0.56	0.00	0.03	0.00	0.35	0.00	0.00	0.00	0.28	0.44	0.00	0.00	0.03	4.88
20207	kansas	woodson	0.74	1.97	0.59	0.38	0.00	0.04	0.00	0.14	0.00	0.00	0.00	0.17	0.91	0.00	0.00	0.02	4.96
20209	kansas	wyandotte	0.50	0.23	0.21	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.06	0.15	3.51	0.00	0.07	4.80
21001	kentucky	adair	0.55	0.41	0.11	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.38	0.02	0.30	0.26	3.12
21003	kentucky	allen	0.13	0.43	0.04	0.01	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	1.25	0.10	0.00	0.29	2.34
21005	kentucky	anderson	0.03	0.27	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.58	0.00	0.00	0.16	1.07
21007	kentucky	ballard	0.84	0.60	0.62	0.22	0.00	0.01	0.00	0.25	0.00	0.00	0.00	0.02	0.13	0.27	0.66	0.06	3.68
21009	kentucky	barren	1.12	0.73	0.50	0.09	0.00	0.03	0.00	0.08	0.00	0.00	0.00	0.03	2.92	0.01	0.48	0.28	6.28
21011	kentucky	bath	0.06	0.34	0.07	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.91	0.00	0.00	0.20	1.60
21013	kentucky	bell	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.38	0.00	0.01	0.45
21015	kentucky	boone	0.10	0.45	0.08	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.71	0.01	0.03	0.48	1.88
21017	kentucky	bourbon	0.19	1.00	0.20	0.05	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	2.03	0.01	0.02	0.89	4.44
21019	kentucky	boyd	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.09	0.20
21021	kentucky	boyle	0.11	0.48	0.05	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.02	1.25	0.00	0.00	0.17	2.10
21023	kentucky	bracken	0.06	0.31	0.04	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.47	0.01	0.00	0.11	1.04
21025	kentucky	breathitt	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.02	0.04	0.12
21027	kentucky	breckinridge	0.86	0.45	0.50	0.08	0.00	0.05	0.00	0.07	0.00	0.00	0.00	0.02	1.16	0.21	0.54	0.26	4.19
21029	kentucky	bullitt	0.43	0.36	0.24	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.53	0.27	0.01	0.31	2.18
21031	kentucky	butler	0.54	0.24	0.24	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.31	0.31	0.54	0.08	2.31
21033	kentucky	caldwell	0.64	0.19	0.41	0.21	0.00	0.19	0.00	0.08	0.00	0.00	0.00	0.00	0.23	0.01	0.00	0.07	2.02
21035	kentucky	calloway	2.42	0.34	1.54	0.91	0.00	0.31	0.01	0.22	0.00	0.00	0.00	0.06	0.31	0.05	1.76	0.20	8.13
21037	kentucky	campbell	0.05	0.43	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.77	0.04	0.01	0.36	1.69
21039	kentucky	carlisle	0.86	0.06	0.46	0.19	0.00	0.01	0.00	0.06	0.01	0.00	0.00	0.03	0.09	0.97	0.49	0.03	3.26
21041	kentucky	carroll	0.03	0.11	0.06	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.25	0.01	0.00	0.10	0.57
21043	kentucky	carter	0.02	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.03	0.00	0.23	0.86
21045	kentucky	casey	0.22	0.35	0.09	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	1.05	0.18	0.00	0.20	2.14
21047	kentucky	christian	2.35	0.44	1.29	1.16	0.00	0.02	0.00	0.76	0.00	0.00	0.00	0.00	0.96	0.01	0.26	0.20	7.48
21049	kentucky	clark	0.14	0.56	0.10	0.02	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	1.32	0.00	0.01	0.24	2.41
21051	kentucky	clay	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.34	0.00	0.05	0.51
21053	kentucky	clinton	0.28	0.18	0.04	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.49	0.02	0.69	0.09	1.82
21055	kentucky	crittenden	0.30	0.31	0.23	0.07	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.02	0.28	0.28	0.01	0.12	1.64
21057	kentucky	cumberland	0.14	0.10	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.01	0.19	0.07	0.73
21059	kentucky	daviess	3.16	0.90	2.25	0.27	0.00	0.06	0.00	0.16	0.00	0.02	0.00	0.03	0.50	0.39	1.09	0.13	8.98
21061	kentucky	edmonson	0.07	0.12	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.00	0.06	0.10	0.82
21063	kentucky	elliott	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.10	0.31
21065	kentucky	estill	0.03	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.05	0.00	0.09	0.44
21067	kentucky	fayette	0.15	0.74	0.16	0.02	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.98	0.74	0.02	2.38	5.25
21069	kentucky	fleming	0.18	0.59	0.22	0.03	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	1.71	0.11	0.00	0.23	3.11
21071	kentucky	floyd	0.02	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.35	0.00	0.02	0.50
21073	kentucky	franklin	0.05	0.36	0.04	0.01	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.20	1.34
21075	kentucky	fulton	0.74	0.18	0.92	0.16	0.00	0.01	0.00	0.09	0.08	0.00	0.00	0.10	0.03	0.25	0.42	0.02	2.98
21077	kentucky	gallatin	0.05	0.09	0.05	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.17	0.00	0.03	0.06	0.46
21079	kentucky	garrard	0.10	0.47	0.02	0.01	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	1.78	0.00	0.00	0.20	2.63
21081	kentucky	grant	0.03	0.41	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.63	0.00	0.00	0.30	1.41
21083	kentucky	graves	4.17	0.87	1.54	0.69	0.00	0.16	0.01	0.22	0.00	0.00	0.01	0.04	0.49	0.30	4.63	0.16	13.32
21085	kentucky	grayson	0.38	0.41	0.23	0.03	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.01	0.93	0.02	0.26	0.16	2.46
21087	kentucky	green	0.31	0.34	0.13	0.02	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.91	0.01	0.01	0.12	1.92
21089	kentucky	greenup	0.01	0.28	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.02	0.00	0.23	0.84
21091	kentucky	hancock	0.09	0.11	0.09	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.14	0.33	0.00	0.04	0.83
21093	kentucky	hardin	0.97	0.80	0.77	0.12	0.00	0.07	0.00	0.04	0.00	0.00	0.00	0.03	1.31	0.61	0.03	0.46	5.21
21095	kentucky	harlan	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.02	0.01	0.44
21097	kentucky	harrison	0.21	0.74	0.11	0.01	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	1.34	0.01	0.00	0.27	2.78
21099	kentucky	hart	0.14	0.36	0.07	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.98	0.02	0.02	0.28	1.91
21101	kentucky	henderson	2.82	0.93	2.21	0.14	0.00	0.11	0.00	0.03	0.00	0.00	0.00	0.14	0.24	0.43	0.02	0.10	7.17
21103	kentucky	henry	0.28	0.71	0.29	0.02	0.00	0.01	0.00	0.06	0.00	0.00	0.00	0.00	1.01	0.01	0.00	0.27	2.65
21105	kentucky	hickman	1.86	0.24	1.07	0.39	0.00	0.04	0.00	0.09	0.10	0.00	0.00	0.03	0.08	0.76	1.76	0.02	6.46
21107	kentucky	hopkins	1.57	0.43	0.80	0.10	0.00	0.12	0.00	0.01	0.00	0.00	0.00	0.07	0.23	0.31	1.77	0.09	5.52
21109	kentucky	jackson	0.04	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.01	0.00	0.11	0.60
21111	kentucky	jefferson	0.20	0.31	0.26	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.30	0.00	0.07	0.68	1.85
21113	kentucky	jessamine	0.10	0.60	0.06	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	1.34	0.76	0.00	0.63	3.54
21115	kentucky	johnson	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.02	0.06	0.18
21117	kentucky	kenton	0.07	0.38	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.01	0.06	0.37	1.52
21119	kentucky	knott	0.03	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.34	0.01	0.03	0.47
21121	kentucky	knox	0.04	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.41	0.02	0.12	0.74
21123	kentucky	larue	0.57	0.31	0.43	0.06	0.00	0.01	0.00	0.07	0.00	0.00	0.00	0.00	0.61	0.34	0.00	0.19	2.61
21125	kentucky	laurel	0.04	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.28	0.00	0.18	1.20
21127	kentucky	lawrence	0.01	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.01	0.07	0.25
21129	kentucky	lee	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.06	0.12
21131	kentucky	leslie	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.04
21133	kentucky	letcher	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.04
21135	kentucky	lewis	0.03	0.18	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.01	0.00	0.07	0.64
21137	kentucky	lincoln	0.34	0.62	0.18	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	2.30	0.05	0.02	0.25	3.81
21139	kentucky	livingston	0.23	0.48	0.25	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.33	0.28	0.28	0.05	1.99
21141	kentucky	logan	2.09	0.50	1.23	1.02	0.00	0.06	0.00	0.79	0.00	0.00	0.00	0.00	1.37	0.28	0.00	0.13	7.47
21143	kentucky	lyon	0.08	0.13	0.08	0.02	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.10	0.29	0.00	0.02	0.75
21151	kentucky	madison	0.09	1.15	0.02	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.01	3.24	0.04	0.00	0.30	4.88
21153	kentucky	magoffin	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.16	0.28
21155	kentucky	marion	0.56	0.60	0.28	0.04	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.00	1.74	0.71	0.02	0.16	4.17
21157	kentucky	marshall	0.49	0.25	0.36	0.12	0.00	0.13	0.00	0.01	0.00	0.00	0.00	0.00	0.26	0.30	0.30	0.10	2.33
21159	kentucky	martin	0.03	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.02	0.00	0.48
21161	kentucky	mason	0.24	0.63	0.16	0.01	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.96	0.02	0.00	0.17	2.24
21145	kentucky	mccracken	0.44	0.36	0.41	0.11	0.00	0.01	0.00	0.05	0.01	0.00	0.00	0.00	0.08	0.00	0.33	0.08	1.88
21147	kentucky	mccreary	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.03	0.10
21149	kentucky	mclean	2.39	0.30	1.01	0.24	0.00	0.07	0.01	0.10	0.00	0.00	0.00	0.03	0.13	0.29	2.59	0.03	7.19
21163	kentucky	meade	0.63	0.34	0.35	0.10	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.66	0.01	0.41	0.22	2.80
21165	kentucky	menifee	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.08	0.22
21167	kentucky	mercer	0.18	0.90	0.12	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	2.36	0.02	0.00	0.46	4.08
21169	kentucky	metcalfe	0.26	0.27	0.05	0.01	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.91	0.01	0.14	0.15	1.85
21171	kentucky	monroe	0.65	0.33	0.10	0.04	0.00	0.02	0.00	0.06	0.00	0.00	0.00	0.00	1.49	0.00	0.98	0.06	3.74
21173	kentucky	montgomery	0.08	0.36	0.02	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	1.00	0.38	0.00	0.15	2.02
21175	kentucky	morgan	0.01	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.16	0.58
21177	kentucky	muhlenberg	0.49	0.57	0.30	0.05	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.31	0.31	0.79	0.08	2.96
21179	kentucky	nelson	0.89	0.70	0.52	0.21	0.00	0.04	0.00	0.11	0.00	0.00	0.00	0.00	2.08	0.99	0.00	0.36	5.92
21181	kentucky	nicholas	0.06	0.44	0.05	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.73	0.01	0.00	0.13	1.47
21183	kentucky	ohio	1.23	0.90	0.40	0.05	0.00	0.05	0.00	0.01	0.00	0.00	0.00	0.02	0.35	0.37	1.17	0.12	4.67
21185	kentucky	oldham	0.21	0.42	0.17	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.42	0.00	0.03	0.39	1.68
21187	kentucky	owen	0.04	0.44	0.02	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.78	0.01	0.00	0.21	1.54
21189	kentucky	owsley	0.02	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.34	0.00	0.04	0.49
21191	kentucky	pendleton	0.04	0.43	0.03	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.44	0.01	0.00	0.31	1.28
21193	kentucky	perry	0.03	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.32	0.01	0.03	0.49
21195	kentucky	pike	0.01	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.02	0.03	0.24
21197	kentucky	powell	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.06	0.22
21199	kentucky	pulaski	0.28	0.50	0.17	0.04	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	2.16	0.01	0.02	0.33	3.57
21201	kentucky	robertson	0.05	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.37	0.00	0.04	0.74
21203	kentucky	rockcastle	0.03	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.11	0.65
21205	kentucky	rowan	0.03	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.34	0.00	0.09	0.69
21207	kentucky	russell	0.17	0.23	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.31	0.00	0.02	0.12	1.97
21209	kentucky	scott	0.11	0.66	0.06	0.03	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	1.40	0.49	0.00	0.65	3.45
21211	kentucky	shelby	1.05	0.86	0.95	0.12	0.00	0.01	0.00	0.14	0.00	0.00	0.00	0.02	1.72	0.01	0.01	0.78	5.68
21213	kentucky	simpson	0.89	0.20	0.56	0.48	0.00	0.03	0.00	0.38	0.00	0.00	0.00	0.01	0.34	0.00	0.00	0.12	3.02
21215	kentucky	spencer	0.15	0.29	0.21	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.60	0.00	0.00	0.16	1.44
21217	kentucky	taylor	0.48	0.30	0.20	0.05	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.00	0.88	0.07	0.19	0.19	2.42
21219	kentucky	todd	1.37	0.19	0.75	0.54	0.00	0.03	0.00	0.41	0.00	0.00	0.00	0.00	0.83	0.22	0.43	0.10	4.89
21221	kentucky	trigg	0.53	0.21	0.25	0.19	0.00	0.03	0.00	0.19	0.00	0.00	0.00	0.00	0.30	0.01	0.01	0.07	1.80
21223	kentucky	trimble	0.04	0.13	0.10	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.08	0.61
21225	kentucky	union	2.28	0.43	1.19	0.22	0.00	0.09	0.00	0.05	0.00	0.00	0.00	0.00	0.25	0.03	0.00	0.05	4.59
21227	kentucky	warren	1.53	0.67	0.93	0.51	0.00	0.02	0.00	0.28	0.00	0.00	0.00	0.05	1.96	3.10	0.41	0.40	9.86
21229	kentucky	washington	0.22	0.53	0.10	0.01	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	1.30	0.01	0.00	0.18	2.40
21231	kentucky	wayne	0.37	0.23	0.09	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.81	0.01	0.59	0.13	2.29
21233	kentucky	webster	2.01	0.35	1.02	0.07	0.00	0.10	0.00	0.03	0.00	0.01	0.00	0.04	0.22	0.31	1.85	0.05	6.08
21235	kentucky	whitley	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.01	0.01	0.15	0.53
21237	kentucky	wolfe	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.07	0.15
21239	kentucky	woodford	0.10	0.61	0.09	0.01	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	1.05	0.00	0.00	1.26	3.16
22001	louisiana	acadia	0.01	1.08	1.40	0.03	0.14	5.14	0.00	0.00	0.00	0.00	0.00	0.07	0.31	0.01	0.00	0.09	8.30
22003	louisiana	allen	0.00	0.28	0.04	0.01	0.00	0.22	0.00	0.01	0.00	0.00	0.00	0.03	0.12	0.01	0.00	0.04	0.77
22005	louisiana	ascension	7.98	0.22	0.32	2.02	0.75	0.55	0.47	6.25	0.01	0.00	0.13	0.04	0.28	0.02	0.00	0.07	19.11
22007	louisiana	assumption	0.01	0.17	0.03	0.00	2.47	0.00	0.00	0.01	0.00	0.00	0.00	0.26	0.02	0.00	0.00	0.01	2.97
22009	louisiana	avoyelles	0.17	1.50	1.29	0.25	0.36	1.19	0.20	0.03	0.00	0.00	0.09	0.19	0.54	0.01	0.00	0.04	5.86
22011	louisiana	beauregard	0.01	0.40	0.01	0.00	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.34	0.02	0.00	0.13	0.95

22013	louisiana	bienville	0.13	0.07	0.01	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.11	0.00	0.36	0.02	0.73
22015	louisiana	bossier	0.10	0.58	0.04	0.03	0.00	0.01	0.01	0.02	0.00	0.00	0.00	0.02	0.32	0.02	0.05	0.22	1.43
22017	louisiana	caddo	0.44	1.56	0.19	0.12	0.00	0.02	0.49	0.03	0.00	0.01	0.00	0.04	0.45	0.01	0.00	0.19	3.55
22019	louisiana	calcasieu	0.01	1.52	0.12	0.01	0.01	0.44	0.00	0.00	0.01	0.00	0.00	0.01	1.08	0.03	0.00	0.22	3.45
22021	louisiana	caldwell	0.08	0.05	0.15	0.02	0.00	0.14	0.04	0.00	0.03	0.00	0.00	0.00	0.07	0.01	0.00	0.02	0.61
22023	louisiana	cameron	0.00	1.03	0.06	0.00	0.00	0.88	0.00	0.00	0.00	0.00	0.00	0.01	0.62	0.00	0.00	0.04	2.66
22025	louisiana	catahoula	0.44	0.18	1.32	0.21	0.03	0.61	0.79	0.03	0.09	0.00	0.00	0.08	0.16	0.00	0.00	0.02	3.96
22027	louisiana	claiborne	0.52	0.07	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.02	0.97	0.04	1.88
22029	louisiana	concordia	0.36	0.13	1.65	0.20	0.01	0.84	0.53	0.01	0.01	0.00	0.00	0.08	0.11	0.00	0.00	0.02	3.97
22031	louisiana	de soto	0.08	0.57	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.70	0.00	0.11	0.12	1.61
22033	louisiana	east baton rouge	0.08	0.78	0.02	0.01	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.53	0.01	0.12	0.24	1.84
22035	louisiana	east carroll	0.86	0.03	2.11	0.18	0.00	0.22	0.22	0.01	0.02	0.00	0.01	0.16	0.03	0.00	0.00	0.01	3.87
22037	louisiana	east feliciana	0.00	0.30	0.01	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.28	0.01	0.00	0.06	0.70
22039	louisiana	evangeline	0.02	0.60	0.39	0.01	0.01	1.22	0.01	0.00	0.00	0.00	0.00	0.08	0.46	0.01	0.00	0.10	2.92
22041	louisiana	franklin	1.71	0.09	1.04	0.48	0.00	0.18	0.44	0.08	0.11	0.00	0.11	0.09	0.27	0.00	0.00	0.04	4.66
22043	louisiana	grant	0.00	0.24	0.03	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.01	0.08	0.01	0.00	0.04	0.44
22045	louisiana	iberia	0.01	0.32	0.08	0.00	2.55	0.01	0.00	0.00	0.00	0.00	0.00	0.18	0.07	0.00	0.00	0.03	3.26
22047	louisiana	iberville	0.02	0.19	0.16	0.01	2.77	0.01	0.00	0.00	0.00	0.00	0.00	0.38	0.17	0.01	0.00	0.03	3.74
22049	louisiana	jackson	0.25	0.04	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.49	0.02	0.89
22051	louisiana	jefferson	0.02	0.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.32	0.05	0.82
22053	louisiana	jefferson davis	0.01	0.66	0.48	0.03	0.03	3.44	0.00	0.00	0.00	0.00	0.00	0.03	0.37	0.00	0.00	0.07	5.13
22059	louisiana	la salle	0.02	0.04	0.09	0.05	0.00	0.17	0.01	0.01	0.01	0.00	0.00	0.00	0.03	0.01	0.00	0.01	0.47
22055	louisiana	lafayette	0.02	1.28	0.20	0.06	0.67	0.25	0.00	0.00	0.00	0.00	0.00	0.05	0.29	0.01	0.00	0.22	3.05
22057	louisiana	lafourche	0.01	1.06	0.01	0.00	1.79	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.35	0.01	0.00	0.06	3.39
22061	louisiana	lincoln	1.00	0.09	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.01	1.79	0.04	3.29
22063	louisiana	livingston	0.00	0.30	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.04	0.00	0.15	0.69
22065	louisiana	madison	1.19	0.03	1.86	0.12	0.00	0.39	0.77	0.03	0.05	0.00	0.00	0.12	0.04	0.00	0.07	0.00	4.68
22067	louisiana	morehouse	1.19	0.20	1.31	0.24	0.00	1.20	0.11	0.03	0.02	0.00	0.06	0.49	0.26	0.00	0.00	0.03	5.16
22069	louisiana	natchitoches	0.60	0.61	0.13	0.03	0.00	0.03	0.04	0.00	0.02	0.00	0.00	0.05	0.50	0.00	0.94	0.07	3.03
22071	louisiana	orleans	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.00	0.44
22073	louisiana	ouachita	0.38	0.39	0.48	0.17	0.00	0.11	0.19	0.05	0.13	0.00	0.01	0.08	0.14	0.01	0.26	0.09	2.48
22075	louisiana	plaquemines	0.02	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.01	0.03	0.01	0.48
22077	louisiana	pointe coupee	0.21	0.55	0.90	0.34	1.20	0.07	0.05	0.11	0.01	0.00	0.00	0.49	0.27	0.00	0.00	0.04	4.23
22079	louisiana	rapides	0.12	1.21	0.40	0.06	0.27	0.43	0.18	0.04	0.00	0.00	0.01	0.22	0.59	0.03	0.00	0.14	3.70
22081	louisiana	red river	0.03	0.27	0.03	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.53	0.00	0.00	0.04	0.94
22083	louisiana	richland	1.25	0.08	1.29	0.71	0.00	0.72	0.40	0.05	0.09	0.00	0.06	0.12	0.47	0.00	0.02	0.03	5.31
22085	louisiana	sabine	1.02	0.12	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.01	1.81	0.04	3.34
22087	louisiana	st bernard	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.13
22089	louisiana	st charles	0.08	0.20	0.00	0.02	0.23	0.01	0.00	0.05	0.00	0.00	0.00	0.00	0.12	0.00	0.22	0.02	0.96
22091	louisiana	st helena	0.10	0.15	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.01	0.09	0.02	0.85
22093	louisiana	st james	2.09	0.12	0.09	0.53	1.18	0.14	0.12	1.63	0.00	0.00	0.03	0.14	0.02	0.00	0.00	0.00	6.10
22095	louisiana	st john the baptist	0.00	0.07	0.00	0.00	0.42	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.01	0.00	0.00	0.00	0.54
22097	louisiana	st landry	0.24	2.81	2.49	0.29	0.66	1.93	0.03	0.01	0.00	0.00	0.01	0.27	0.82	0.03	0.00	0.24	9.84
22099	louisiana	st martin	0.17	1.35	0.42	0.04	1.37	0.57	0.09	0.02	0.00	0.00	0.00	0.16	0.08	0.01	0.00	0.05	4.35
22101	louisiana	st mary	0.02	0.15	0.08	0.01	2.66	0.00	0.00	0.00	0.00	0.00	0.01	0.08	0.12	0.01	0.00	0.03	3.17
22103	louisiana	st tammany	0.01	0.39	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.02	0.01	0.39	1.15
22105	louisiana	tangipahoa	0.36	0.64	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.55	0.03	0.00	0.21	2.87
22107	louisiana	tensas	0.95	0.01	1.11	0.30	0.00	0.37	1.66	0.14	0.19	0.00	0.01	0.02	0.01	0.00	0.07	0.00	4.84
22109	louisiana	terrebonne	0.00	0.26	0.00	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.01	0.00	0.02	0.78
22111	louisiana	union	0.99	0.12	0.14	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.02	1.78	0.06	3.46
22113	louisiana	vermilion	0.02	2.77	0.52	0.01	1.92	3.16	0.00	0.00	0.00	0.00	0.01	0.13	0.82	0.01	0.00	0.12	9.50
22115	louisiana	vernon	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01	0.02	0.06	0.32
22117	louisiana	washington	0.22	0.35	0.02	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.92	0.02	0.04	0.11	1.73
22119	louisiana	webster	0.02	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.09	0.01	0.13	0.08	0.47
22121	louisiana	west baton rouge	0.01	0.28	0.15	0.01	1.06	0.00	0.00	0.03	0.00	0.00	0.00	0.17	0.06	0.00	0.13	0.03	1.93
22123	louisiana	west carroll	0.62	0.38	1.13	0.46	0.00	0.19	0.05	0.02	0.00	0.00	0.15	0.12	0.13	0.01	0.00	0.03	3.29
22125	louisiana	west feliciana	0.02	0.17	0.02	0.01	0.02	0.00	0.00	0.01	0.00	0.00	0.00	0.05	0.19	0.00	0.00	0.03	0.54
22127	louisiana	winn	0.04	0.07	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.29	0.02	0.47
23001	maine	androscoggin	0.10	0.10	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.00	0.14	0.02	1.10	0.05	1.80
23003	maine	aroostook	0.01	0.07	0.00	0.01	0.00	0.21	0.00	0.03	0.00	0.00	0.14	0.00	0.04	0.00	0.00	0.02	0.52
23005	maine	cumberland	0.04	0.21	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.02	0.00	0.17	0.59
23007	maine	franklin	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.03	0.17
23009	maine	hancock	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.00	0.00	0.02	0.10
23011	maine	kennebec	0.21	0.19	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.01	0.92	0.10	1.81
23013	maine	knox	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.03	0.00	0.00	0.03	0.12
23015	maine	lincoln	0.02	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.04	0.01	0.01	0.03	0.16
23017	maine	oxford	0.02	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.04	0.01	0.00	0.05	0.34
23019	maine	penobscot	0.14	0.16	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.08	0.00	0.17	0.01	0.01	0.07	0.66
23021	maine	piscataquis	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.02	0.00	0.00	0.01	0.14
23023	maine	sagadahoc	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.03	0.10
23025	maine	somerset	0.11	0.15	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.01	0.00	0.03	0.47
23027	maine	waldo	0.08	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.13	0.00	0.00	0.03	0.33
23029	maine	washington	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.01	0.11
23031	maine	york	0.06	0.23	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.11	0.02	0.00	0.22	0.66
24001	maryland	allegany	0.03	0.14	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.14	0.01	0.00	0.07	0.40
24003	maryland	anne arundel	0.22	0.28	0.22	0.06	0.00	0.02	0.00	0.13	0.00	0.00	0.00	0.00	0.12	0.01	0.03	0.49	1.59
24005	maryland	baltimore	2.17	1.32	0.96	0.28	0.00	0.13	0.01	0.18	0.00	0.10	0.06	0.00	1.02	0.29	0.08	1.94	8.53
24510	maryland	baltimore city	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04
24009	maryland	calvert	0.15	0.06	0.12	0.17	0.00	0.04	0.00	0.03	0.00	0.01	0.00	0.00	0.05	0.00	0.00	0.08	0.72
24011	maryland	caroline	4.67	0.38	1.67	0.85	0.00	0.36	0.02	0.13	0.00	0.03	0.03	0.00	0.34	0.00	5.98	0.10	14.57
24013	maryland	carroll	3.11	1.23	1.49	0.54	0.00	0.30	0.01	0.19	0.00	0.13	0.04	0.00	3.20	0.14	1.96	1.17	13.51
24015	maryland	cecil	1.74	0.57	1.24	0.46	0.00	0.12	0.00	0.32	0.00	0.42	0.04	0.00	1.05	0.24	1.58	0.73	8.52
24017	maryland	charles	0.40	0.41	0.38	0.16	0.00	0.17	0.00	0.37	0.00	0.03	0.00	0.00	0.13	0.03	0.01	0.20	2.30
24019	maryland	dorchester	1.74	0.18	1.18	0.46	0.00	0.18	0.01	0.22	0.00	0.20	0.21	0.00	0.00	0.10	2.04	0.05	6.57
24021	maryland	frederick	6.01	2.52	2.16	0.99	0.00	0.52	0.02	0.57	0.00	0.12	0.02	0.00	7.32	1.21	0.18	1.39	23.03
24023	maryland	garrett	0.42	0.24	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.80	0.02	0.01	0.15	1.71
24025	maryland	harford	1.61	0.56	0.51	0.13	0.00	0.08	0.00	0.07	0.00	0.06	0.15	0.00	1.27	0.02	0.07	0.97	5.51
24027	maryland	howard	0.71	0.47	0.29	0.17	0.00	0.08	0.00	0.10	0.00	0.02	0.01	0.00	0.41	0.04	0.05	0.82	3.17
24029	maryland	kent	2.90	0.44	2.11	1.01	0.00	0.33	0.01	0.74	0.00	0.01	0.37	0.00	0.98	0.18	0.64	0.13	9.83
24031	maryland	montgomery	0.95	1.77	0.78	0.31	0.00	0.04	0.01	0.20	0.00	0.00	0.03	0.00	0.89	0.09	0.02	1.74	6.83
24033	maryland	prince georges	0.74	0.44	0.46	0.12	0.00	0.02	0.00	0.19	0.00	0.00	0.01	0.00	0.27	0.52	0.02	0.63	3.43
24035	maryland	queen annes	3.51	0.62	2.33	1.07	0.00	0.14	0.01	0.18	0.00	0.00	0.02	0.00	0.39	0.01	2.34	0.16	10.79
24039	maryland	somerset	2.86	0.16	0.68	0.32	0.00	0.12	0.01	0.02	0.00	0.00	0.04	0.00	0.03	0.08	5.01	0.02	9.36
24037	maryland	st marys	0.41	0.17	0.55	0.18	0.00	0.07	0.00	0.17	0.00	0.03	0.03	0.00	0.19	0.08	0.02	0.19	2.07
24041	maryland	talbot	1.84	0.24	1.88	1.03	0.00	0.18	0.00	0.13	0.00	0.00	0.00	0.00	0.13	0.12	0.82	0.06	6.44
24043	maryland	washington	3.35	0.85	0.89	0.42	0.00	0.27	0.01	0.02	0.00	0.16	0.01	0.00	4.57	0.60	0.21	0.61	11.97
24045	maryland	wicomico	3.18	0.28	0.67	0.19	0.00	0.13	0.01	0.14	0.00	0.16	0.04	0.00	0.05	0.00	4.63	0.09	9.57
24047	maryland	worcester	3.17	0.13	0.78	0.29	0.00	0.13	0.01	0.14	0.00	0.03	0.04	0.00	0.03	0.01	5.31	0.04	10.10
25001	massachusetts	barnstable	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.06	0.11
25003	massachusetts	berkshire	0.23	0.18	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.05	0.01	0.24	1.23
25005	massachusetts	bristol	0.18	0.28	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.05	0.02	0.00	0.43	0.14	0.02	0.62	1.79
25007	massachusetts	dukes	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.01	0.01	0.00	0.02	0.16
25009	massachusetts	essex	0.07	0.23	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.17	0.09	0.01	0.62	1.24
25011	massachusetts	franklin	0.22	0.28	0.02	0.20	0.00	0.01	0.00	0.00	0.00	0.01	0.05	0.00	0.42	0.01	0.01	0.30	1.55
25013	massachusetts	hampden	0.16	0.31	0.01	0.01	0.00	0.01	0.00	0.02	0.00	0.01	0.04	0.00	0.29	0.01	0.01	0.39	1.27
25015	massachusetts	hampshire	0.28	0.29	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.01	0.12	0.00	0.44	0.15	0.02	0.43	1.81
25017	massachusetts	middlesex	0.20	0.35	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.03	0.02	0.00	0.41	0.08	0.04	1.34	2.51
25019	massachusetts	nantucket	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.08
25021	massachusetts	norfolk	0.01	0.11	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.01	0.01	0.42	0.64
25023	massachusetts	plymouth	0.04	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.06	0.00	0.07	0.07	0.01	0.51	1.56
25025	massachusetts	suffolk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
25027	massachusetts	worcester	0.38	1.01	0.04	0.02	0.00	0.02	0.00	0.01	0.00	0.03	0.03	0.00	0.83	0.30	0.29	1.71	4.67
26001	michigan	alcona	0.05	0.07	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.03	0.27
26003	michigan	alger	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.01	0.08
26005	michigan	allegan	5.88	0.53	1.19	0.47	0.00	0.19	0.02	0.00	0.00	0.03	0.06	0.00	3.22	12.86	6.19	0.44	31.10
26007	michigan	alpena	0.10	0.16	0.05	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.18	0.00	0.00	0.04	0.60
26009	michigan	antrim	0.08	0.18	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.03	0.05	0.00	0.09	0.00	0.00	0.05	0.53
26011	michigan	arenac	0.26	0.12	0.19	0.15	0.04	0.01	0.00	0.00	0.00	0.00	0.07	0.07	0.24	0.01	0.00	0.02	1.19
26013	michigan	baraga	0.02	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.39	0.00	0.45
26015	michigan	barry	1.97	0.71	1.00	0.35	0.00	0.06	0.00	0.00	0.00	0.00	0.01	0.00	2.02	0.32	0.01	0.33	6.79
26017	michigan	bay	1.68	0.26	1.30	0.57	0.54	0.02	0.00	0.00	0.00	0.00	0.24	0.66	0.32	0.03	0.00	0.09	5.72
26019	michigan	benzie	0.04	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.02	0.08	0.00	0.02	0.27
26021	michigan	berrien	1.59	0.26	1.07	0.13	0.00	0.02	0.01	0.01	0.01	0.50	0.05	0.01	0.30	1.10	0.00	0.20	5.27
26023	michigan	branch	3.03	0.59	2.04	0.19	0.00	0.06	0.01	0.01	0.00	0.01	0.13	0.00	0.77	2.83	0.09	0.33	10.08
26025	michigan	calhoun	2.87	1.05	2.28	0.35	0.00	0.09	0.01	0.00	0.00	0.00	0.01	0.00	1.03	1.96	0.01	0.40	10.07
26027	michigan	cass	3.90	0.38	1.28	0.18	0.00	0.08	0.01	0.00	0.00	0.14	0.03	0.02	0.29	7.97	0.00	0.19	14.48
26029	michigan	charlevoix	0.04	0.12	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.04	0.28
26031	michigan	cheboygan	0.05	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.02	0.31
26033	michigan	chippewa	0.03	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.06	0.40
26035	michigan	clare	0.18	0.21	0.05	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.01	0.00	0.11	0.98
26037	michigan	clinton	4.03	0.81	2.60	1.06	0.00	0.11	0.01	0.06	0.00	0.00	0.20	0.04	4.35	0.44	0.01	0.29	14.07
26039	michigan	crawford	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.01	0.00	0.00	0.02	0.13
26041	michigan	delta	0.06	0.10	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.12	0.00	0.00	0.02	0.34
26043	michigan	dickinson	0.02	0.07	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.01	0.17
26045	michigan	eaton	2.40	0.70	2.55	0.84	0.00	0.03	0.00	0.00	0.00	0.00	0.17	0.01	0.73	0.71	0.01	0.41	8.58
26047	michigan	emmet	0.04	0.14	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.02	0.30
26049	michigan	genesee	1.42	0.68	2.07	0.59	0.02	0.04	0.00	0.01	0.00	0.00	0.23	0.04	0.70	0.01	0.01	0.67	6.50
26051	michigan	gladwin	0.18	0.24	0.11	0.04	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.19	0.01	0.00	0.11	0.93
26053	michigan	gogebic	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.06	0.00	0.00	0.14
26055	michigan	grand traverse	0.14	0.16	0.02	0.02	0.00	0.02	0.00	0.01	0.00	0.11	0.01	0.00	0.09	0.16	0.00	0.06	0.80
26057	michigan	gratiot	3.80	0.61	2.23	0.49	0.33	0.07	0.01	0.00	0.00	0.00	0.09	0.20	2.53	2.97	0.00	0.13	13.49
26059	michigan	hillsdale	2.69	1.13	2.31	0.49	0.00	0.06	0.01	0.00	0.00	0.00	0.14	0.00	1.69	1.17	0.01	0.38	10.07
26061	michigan	houghton	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.09
26063	michigan	huron	3.10	0.34	1.03	1.48	1.18	0.10	0.01	0.00	0.00	0.00	0.09	1.43	3.69	0.89	0.65	0.05	14.04
26065	michigan	ingham	2.96	0.71	2.20	1.05	0.01	0.05	0.01	0.01	0.00	0.00	0.17	0.00	1.68	0.25	0.02	0.57	9.69
26067	michigan	ionia	3.94	0.65	1.87	0.61	0.01	0.11	0.01	0.00	0.00	0.01	0.04	0.01	3.22	3.75	1.45	0.23	15.90
26069	michigan	iosco	0.09	0.11	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.04	0.46
26071	michigan	iron	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.11
26073	michigan	isabella	1.31	0.65	1.20	0.52	0.01	0.06	0.00	0.00	0.00	0.00	0.02	0.08	1.42	0.06	0.00	0.19	5.53
26075	michigan	jackson	2.07	0.84	1.41	0.38	0.00	0.05	0.00	0.00	0.00	0.00	0.01	0.00	1.45	0.05	0.01	0.69	6.98
26077	michigan	kalamazoo	3.00	0.58	1.12	0.24	0.00	0.06	0.01	0.00	0.00	0.03	0.01	0.01	1.00	3.16	0.01	0.52	9.75
26079	michigan	kalkaska	0.03	0.08	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.04	0.00	0.00	0.03	0.23
26081	michigan	kent	2.31	0.79	0.87	0.34	0.00	0.08	0.01	0.00	0.00	0.28	0.02	0.06	2.83	0.30	0.01	0.52	8.43
26083	michigan	keweenaw	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
26085	michigan	lake	0.06	0.08	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.06	0.00	0.00	0.04	0.26
26087	michigan	lapeer	1.73	0.97	2.00	0.59	0.06	0.06	0.00	0.00	0.00	0.00	0.04	0.03	1.35	0.06	0.01	0.63	7.53
26089	michigan	leelanau	0.04	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.01	0.00	0.05	0.01	0.00	0.02	0.28
26091	michigan	lenawee	7.12	0.93	5.92	3.14	0.00	0.10	0.01	0.01	0.00	0.04	0.14	0.00	3.30	0.48	0.01	0.49	21.69
26093	michigan	livingston	1.64	0.75	1.06	0.67	0.00	0.03	0.00	0.01	0.00	0.00	0.01	0.00	1.07	0.16	0.01	0.68	6.09
26095	michigan	luce	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04
26097	michigan	mackinac	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.03	0.00	0.00	0.01	0.12
26099	michigan	macomb	1.02	0.31	1.77	0.15	0.01	0.02	0.00	0.01	0.00	0.01	0.02	0.01	0.63	0.05	0.01	0.48	4.49
26101	michigan	manistee	0.08	0.08	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.08	0.00	0.00	0.05	0.01	0.00	0.04	0.36
26103	michigan	marquette	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.01	0.00	0.02	0.12
26105	michigan	mason	0.26	0.13	0.06	0.06	0.00	0.02	0.00	0.00	0.00	0.04	0.03	0.00	0.28	0.10	0.00	0.06	1.05
26107	michigan	mecosta	1.19	0.50	0.14	0.09	0.00	0.07	0.00	0.00	0.00	0.00	0.07	0.04	0.63	2.63	0.01	0.21	5.58
26109	michigan	menominee	0.22	0.16	0.02	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.00	0.00	0.03	0.86
26111	michigan	midland	0.62	0.19	0.49	0.16	0.08	0.01	0.00	0.00	0.00	0.00	0.02	0.08	0.31	0.12	0.00	0.12	2.19
26113	michigan	missaukee	0.55	0.25	0.04	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.98	0.01	0.00	0.05	2.01
26115	michigan	monroe	3.80	0.42	4.64	1.82	0.00	0.03	0.01	0.03	0.00	0.01	0.43	0.01	0.39	0.02	0.01	0.43	12.01
26117	michigan	montcalm	2.34	0.72	0.62	0.64	0.02	0.10	0.01	0.00	0.00	0.02	0.33	0.25	1.85	0.71	0.01	0.31	7.93
26119	michigan	montmorency	0.03	0.05	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.17
26121	michigan	muskegon	0.71	0.27	0.19	0.14	0.00	0.03	0.00	0.00	0.00	0.05	0.04	0.00	0.97	0.18	0.01	0.17	2.75
26123	michigan	newaygo	1.03	0.38	0.17	0.10	0.00	0.04	0.00	0.01	0.00	0.02	0.02	0.00	1.37	0.29	0.01	0.25	3.68
26125	michigan	oakland	0.47	0.46	0.42	0.64	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.18	0.04	0.02	1.42	3.68
26127	michigan	oceana	0.76	0.27	0.12	0.06	0.00	0.05	0.00	0.01	0.00	0.19	0.15	0.00	0.29	1.49	0.01	0.11	3.52
26129	michigan	ogemaw	0.33	0.25	0.04	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.05	0.00	0.50	0.01	0.00	0.04	1.32
26131	michigan	ontonagon	0.01	0.08	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.02	0.00	0.00	0.01	0.13
26133	michigan	osceola	0.37	0.39	0.03	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.02	0.00	0.66	0.02	0.00	0.13	1.68
26135	michigan	oscoda	0.04	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.07	0.00	0.00	0.04	0.22
26137	michigan	otsego	0.03	0.07	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.04	0.00	0.00	0.02	0.21
26139	michigan	ottawa	2.77	0.63	0.83	0.28	0.00	0.08	0.01	0.00	0.00	0.18	0.02	0.01	2.89	2.03	2.26	0.49	12.48
26141	michigan	presque isle	0.06	0.11	0.04	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.08	0.00	0.00	0.01	0.37
26143	michigan	roscommon	0.04	0.03	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.11	0.00	0.01	0.23
26145	michigan	saginaw	4.20	0.33	4.29	1.44	0.85	0.04	0.01	0.31	0.00	0.00	0.28	0.38	0.72	1.02	0.00	0.18	14.05
26151	michigan	sanilac	3.27	0.87	3.29	1.77	0.78	0.08	0.01	0.00	0.00	0.00	0.08	0.65	2.73	0.19	0.00	0.16	13.84
26153	michigan	schoolcraft	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.35	0.00	0.43
26155	michigan	shiawassee	2.18	0.71	3.02	1.21	0.06	0.04	0.00	0.01	0.00	0.00	0.01	0.03	1.35	0.07	0.00	0.34	9.04
26147	michigan	st clair	0.93	0.44	1.62	0.59	0.02	0.02	0.00	0.00	0.00	0.00	0.02	0.02	0.86	0.01	0.01	0.43	4.97
26149	michigan	st joseph	3.07	0.37	1.30	0.15	0.00	0.04	0.01	0.00	0.00	0.01	0.16	0.01	0.79	1.41	0.23	0.31	7.85
26157	michigan	tuscola	2.68	0.57	2.07	1.46	0.69	0.12	0.00	0.00	0.00	0.00	0.19	1.17	1.05	0.25	1.28	0.23	11.76
26159	michigan	van buren	1.34	0.28	0.43	0.05	0.00	0.02	0.00	0.00	0.00	0.43	0.03	0.00	0.75	0.22	0.01	0.22	3.78
26161	michigan	washtenaw	2.69	1.19	2.32	1.19	0.00	0.05	0.01	0.06	0.00	0.00	0.02	0.00	1.36	0.49	0.02	1.70	11.12
26163	michigan	wayne	0.43	0.13	0.59	0.62	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.19	0.03	0.01	0.58	2.62
26165	michigan	wexford	0.07	0.11	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.04	0.34
27001	minnesota	aitkin	0.08	0.23	0.03	0.02	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.04	0.59
27003	minnesota	anoka	0.67	0.49	0.34	0.17	0.00	0.08	0.00	0.03	0.00	0.00	0.00	0.00	0.21	0.03	0.06	0.47	2.56
27005	minnesota	becker	0.90	0.49	1.05	0.47	0.08	0.06	0.00	0.00	0.00	0.00	0.05	0.04	0.59	0.35	0.22	0.06	4.36
27007	minnesota	beltrami	0.07	0.24	0.07	0.06	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.04	0.70
27009	minnesota	benton	1.79	0.42	0.68	0.07	0.00	0.09	0.00	0.00	0.00	0.00	0.10	0.01	1.38	0.92	0.25	0.12	5.84
27011	minnesota	big stone	1.41	0.10	1.26	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.64	0.01	0.02	3.69
27013	minnesota	blue earth	7.87	0.28	3.57	0.25	0.00	0.20	0.02	0.00	0.00	0.00	0.14	0.00	0.39	18.03	0.02	0.13	30.90
27015	minnesota	brown	4.08	0.21	2.08	0.16	0.01	0.09	0.01	0.00	0.00	0.00	0.14	0.03	1.09	5.64	0.05	0.04	13.63
27017	minnesota	carlton	0.08	0.26	0.01	0.04	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.04	0.65
27019	minnesota	carver	3.22	0.63	1.63	0.09	0.00	0.09	0.01	0.00	0.00	0.01	0.01	0.00	1.52	1.88	0.00	0.21	9.32
27021	minnesota	cass	0.18	0.28	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.43	0.00	0.01	0.08	1.07
27023	minnesota	chippewa	3.05	0.20	1.79	0.11	0.68	0.02	0.00	0.00	0.00	0.00	0.03	0.12	0.59	1.11	0.01	0.03	7.75
27025	minnesota	chisago	0.60	0.35	0.45	0.09	0.00	0.03	0.00	0.01	0.00	0.00	0.09	0.00	0.41	0.02	0.02	0.14	2.23
27027	minnesota	clay	2.07	0.54	3.37	1.45	0.75	0.06	0.00	0.06	0.00	0.00	0.02	0.03	0.24	0.72	0.22	0.04	9.58
27029	minnesota	clearwater	0.11	0.23	0.07	0.06	0.00	0.02	0.00	0.02	0.00	0.00	0.02	0.00	0.23	0.02	0.15	0.04	0.97
27031	minnesota	cook	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.02
27033	minnesota	cottonwood	3.35	0.26	2.14	0.09	0.01	0.06	0.01	0.00	0.00	0.00	0.07	0.00	0.53	4.90	0.07	0.08	11.57
27035	minnesota	crow wing	0.16	0.23	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.01	0.00	0.09	0.72
27037	minnesota	dakota	5.64	0.57	2.44	0.11	0.00	0.10	0.01	0.12	0.00	0.00	0.47	0.13	2.01	2.62	0.05	0.39	14.66
27039	minnesota	dodge	4.68	0.20	1.85	0.08	0.00	0.09	0.01	0.00	0.00	0.00	0.22	0.01	0.70	5.56	0.04	0.07	13.52
27041	minnesota	douglas	0.79	0.58	0.71	0.20	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.01	0.56	0.03	0.00	0.05	2.98
27043	minnesota	faribault	4.90	0.15	2.88	0.15	0.00	0.09	0.01	0.00	0.00	0.00	0.15	0.01	0.29	6.22	0.00	0.03	14.89
27045	minnesota	fillmore	5.14	1.20	1.68	0.10	0.00	0.19	0.01	0.02	0.00	0.00	0.02	0.01	2.59	4.66	0.02	0.33	15.97

27047	minnesota	freeborn	6.09	0.41	2.73	0.21	0.00	0.12	0.01	0.00	0.00	0.00	0.13	0.01	0.53	9.95	0.01	0.09	20.29
27049	minnesota	goodhue	8.49	1.10	3.79	0.15	0.00	0.22	0.01	0.01	0.00	0.00	0.30	0.05	2.97	7.01	0.07	0.27	24.46
27051	minnesota	grant	1.15	0.14	1.26	0.18	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.08	0.03	0.00	0.02	3.06
27053	minnesota	hennepin	1.76	0.94	1.28	0.29	0.00	0.07	0.00	0.00	0.00	0.00	0.01	0.00	1.33	0.01	0.01	0.86	6.57
27055	minnesota	houston	1.59	0.55	0.41	0.10	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.01	1.43	0.78	0.00	0.09	5.04
27057	minnesota	hubbard	0.11	0.16	0.04	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.07	0.08	0.00	0.00	0.04	0.60
27059	minnesota	isanti	1.12	0.43	0.87	0.04	0.00	0.04	0.00	0.01	0.00	0.00	0.14	0.00	0.43	0.13	0.02	0.24	3.49
27061	minnesota	itasca	0.04	0.18	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.09	0.00	0.00	0.04	0.41
27063	minnesota	jackson	3.86	0.24	2.33	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.01	0.00	0.23	5.26	0.00	0.07	12.15
27065	minnesota	kanabec	0.30	0.39	0.16	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.05	0.00	0.35	0.01	0.00	0.11	1.42
27067	minnesota	kandiyohi	3.68	0.84	1.82	0.15	0.37	0.06	0.01	0.00	0.00	0.00	0.16	0.09	0.69	2.41	0.30	0.13	10.71
27069	minnesota	kittson	0.08	0.32	0.68	0.90	0.15	0.04	0.00	0.06	0.00	0.00	0.01	0.01	0.12	0.02	0.00	0.01	2.39
27071	minnesota	koochiching	0.02	0.11	0.01	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.01	0.21
27073	minnesota	lac qui parle	2.69	0.19	2.27	0.10	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.01	0.25	1.78	0.01	0.05	7.40
27075	minnesota	lake	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.02	0.00	0.00	0.07
27077	minnesota	lake of the woods	0.00	0.03	0.07	0.03	0.00	0.00	0.00	0.02	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.18
27079	minnesota	le sueur	3.81	0.46	2.23	0.11	0.00	0.08	0.01	0.00	0.00	0.00	0.03	0.01	0.78	5.04	0.01	0.08	12.66
27081	minnesota	lincoln	1.54	0.43	1.12	0.10	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.37	1.56	0.01	0.04	5.21
27083	minnesota	lyon	3.34	0.16	2.19	0.09	0.01	0.05	0.01	0.00	0.00	0.00	0.00	0.00	1.09	3.07	0.25	0.07	10.33
27087	minnesota	mahnomen	0.36	0.10	0.60	0.17	0.03	0.00	0.00	0.00	0.00	0.00	0.03	0.02	0.08	0.00	0.00	0.01	1.42
27089	minnesota	marshall	0.25	0.38	2.00	2.20	0.40	0.23	0.00	0.10	0.00	0.00	0.01	0.13	0.10	0.00	0.01	0.02	5.82
27091	minnesota	martin	8.16	0.17	2.97	0.23	0.00	0.20	0.02	0.00	0.00	0.00	0.11	0.01	0.31	18.82	0.01	0.04	31.05
27085	minnesota	mcleod	3.54	0.38	2.47	0.07	0.04	0.04	0.00	0.01	0.00	0.00	0.11	0.13	1.18	0.35	0.00	0.10	8.42
27093	minnesota	meeker	3.05	0.37	1.97	0.11	0.02	0.05	0.00	0.00	0.00	0.00	0.05	0.02	1.30	0.53	0.37	0.11	7.99
27095	minnesota	mille lacs	0.57	0.41	0.29	0.09	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.01	0.48	0.01	0.33	0.10	2.33
27097	minnesota	morrison	3.37	1.17	0.60	0.13	0.00	0.17	0.01	0.01	0.00	0.00	0.07	0.05	2.71	2.37	1.44	0.17	12.28
27099	minnesota	mower	7.23	0.44	3.31	0.23	0.00	0.15	0.01	0.00	0.00	0.00	0.14	0.05	0.89	10.13	0.31	0.10	22.99
27101	minnesota	murray	2.85	0.22	2.13	0.10	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.92	2.57	0.01	0.09	8.97
27103	minnesota	nicollet	5.31	0.15	2.13	0.15	0.01	0.12	0.01	0.00	0.00	0.00	0.04	0.01	1.05	9.16	0.33	0.04	18.51
27105	minnesota	nobles	4.48	0.38	2.18	0.10	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	1.52	6.03	0.02	0.05	14.86
27107	minnesota	norman	0.69	0.16	1.60	1.08	0.38	0.06	0.00	0.04	0.00	0.00	0.03	0.05	0.10	0.10	0.01	0.01	4.32
27109	minnesota	olmsted	3.64	0.73	1.46	0.05	0.00	0.10	0.01	0.00	0.00	0.00	0.28	0.03	1.70	1.73	0.02	0.22	9.95
27111	minnesota	otter tail	2.26	1.14	1.62	0.51	0.03	0.16	0.01	0.01	0.00	0.00	0.08	0.10	1.69	0.68	0.10	0.14	8.54
27113	minnesota	pennington	0.09	0.29	0.71	0.79	0.00	0.07	0.00	0.03	0.00	0.00	0.03	0.01	0.15	0.02	0.01	0.01	2.21
27115	minnesota	pine	0.42	0.57	0.16	0.06	0.00	0.03	0.00	0.01	0.00	0.00	0.02	0.00	0.63	0.01	0.26	0.12	2.26
27117	minnesota	pipestone	2.41	0.24	1.16	0.14	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.97	4.57	0.01	0.09	9.67
27119	minnesota	polk	1.03	0.85	3.31	4.31	1.24	0.12	0.00	0.11	0.00	0.00	0.04	0.57	0.19	0.28	0.01	0.03	12.12
27121	minnesota	pope	2.18	0.41	1.21	0.16	0.04	0.04	0.00	0.02	0.00	0.00	0.07	0.06	0.59	1.78	0.02	0.06	6.64
27123	minnesota	ramsey	0.17	0.06	0.02	0.24	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.19	0.19	0.04	0.01	0.96
27125	minnesota	red lake	0.15	0.14	0.50	0.58	0.01	0.04	0.00	0.01	0.00	0.00	0.03	0.03	0.16	0.02	0.01	0.01	1.70
27127	minnesota	redwood	4.54	0.21	3.24	0.09	0.11	0.06	0.01	0.00	0.00	0.00	0.09	0.02	0.66	4.02	0.04	0.04	13.14
27129	minnesota	renville	6.02	0.10	3.17	0.16	0.82	0.09	0.01	0.00	0.00	0.00	0.39	0.17	0.84	7.03	0.01	0.11	18.93
27131	minnesota	rice	3.88	0.65	2.27	0.08	0.00	0.09	0.01	0.00	0.00	0.00	0.03	0.01	1.14	2.81	0.12	0.20	11.30
27133	minnesota	rock	3.39	0.31	1.48	0.15	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	1.34	5.28	0.00	0.05	12.10
27135	minnesota	roseau	0.11	0.34	0.65	0.68	0.00	0.07	0.00	0.17	0.00	0.00	0.02	0.01	0.16	0.08	0.01	0.02	2.32
27139	minnesota	scott	3.53	0.96	2.41	0.09	0.00	0.11	0.00	0.00	0.00	0.00	0.02	0.01	1.66	1.10	0.00	0.23	10.12
27141	minnesota	sherburne	1.05	0.40	0.57	0.09	0.00	0.08	0.00	0.00	0.00	0.00	0.16	0.02	0.61	0.54	0.60	0.12	4.25
27143	minnesota	sibley	5.72	0.22	3.26	0.11	0.09	0.09	0.01	0.00	0.00	0.00	0.29	0.06	1.22	3.81	1.81	0.10	16.80
27137	minnesota	st louis	0.06	0.25	0.01	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.16	0.01	0.00	0.10	0.62
27145	minnesota	stearns	8.11	2.11	2.19	0.41	0.06	0.55	0.02	0.00	0.00	0.00	0.10	0.07	8.52	3.90	1.73	0.27	28.04
27147	minnesota	steele	3.75	0.23	1.72	0.18	0.00	0.08	0.01	0.00	0.00	0.00	0.18	0.02	0.37	5.64	0.07	0.05	12.29
27149	minnesota	stevens	2.74	0.13	1.34	0.22	0.06	0.05	0.01	0.01	0.00	0.00	0.00	0.12	1.03	3.37	0.00	0.02	9.10
27151	minnesota	swift	2.55	0.21	1.56	0.14	0.11	0.02	0.00	0.00	0.00	0.00	0.01	0.09	0.34	0.28	0.18	0.03	5.53
27153	minnesota	todd	1.45	0.88	0.54	0.14	0.00	0.12	0.00	0.00	0.00	0.00	0.01	0.01	1.58	0.33	0.17	0.16	5.42
27155	minnesota	traverse	1.52	0.03	1.78	0.22	0.13	0.01	0.00	0.03	0.00	0.00	0.00	0.01	0.09	0.26	0.00	0.02	4.10
27157	minnesota	wabasha	2.79	0.76	0.87	0.17	0.00	0.13	0.01	0.00	0.00	0.00	0.20	0.01	2.58	0.34	0.00	0.09	7.95
27159	minnesota	wadena	0.30	0.27	0.10	0.01	0.00	0.05	0.00	0.01	0.00	0.00	0.03	0.02	0.34	0.12	0.01	0.04	1.30
27161	minnesota	waseca	4.19	0.11	1.85	0.19	0.00	0.09	0.01	0.00	0.00	0.00	0.04	0.00	0.59	8.04	0.02	0.08	15.22
27163	minnesota	washington	0.83	0.87	0.46	0.01	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.01	0.00	0.47	3.20
27165	minnesota	watonwan	2.95	0.05	1.53	0.12	0.00	0.05	0.01	0.00	0.00	0.00	0.03	0.00	0.20	4.27	0.01	0.03	9.25
27167	minnesota	wilkin	0.95	0.06	1.80	0.49	0.60	0.02	0.00	0.02	0.00	0.00	0.05	0.01	0.06	0.00	0.00	0.01	4.05
27169	minnesota	winona	3.09	0.95	0.71	0.08	0.00	0.15	0.01	0.00	0.00	0.00	0.02	0.01	2.94	1.31	0.05	0.23	9.56
27171	minnesota	wright	4.55	1.20	3.27	0.20	0.00	0.20	0.01	0.00	0.00	0.01	0.02	0.00	3.59	0.46	1.08	0.54	15.16
27173	minnesota	yellow medicine	2.78	0.36	2.09	0.12	0.04	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.56	2.40	0.02	0.06	8.48
28001	mississippi	adams	0.02	0.15	0.10	0.00	0.00	0.00	0.01	0.02	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.02	0.39
28003	mississippi	alcorn	0.08	0.24	0.20	0.02	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.22	0.00	0.16	0.08	1.04
28005	mississippi	amite	0.54	0.17	0.06	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	0.99	0.03	2.19
28007	mississippi	attala	0.05	0.32	0.02	0.01	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.01	0.14	0.01	0.03	0.05	0.70
28009	mississippi	benton	0.05	0.21	0.12	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.02	0.14	0.00	0.00	0.04	0.61
28011	mississippi	bolivar	1.00	0.03	4.17	0.67	0.00	1.84	0.57	0.06	0.35	0.00	0.01	0.74	0.03	0.00	0.10	0.01	9.58
28013	mississippi	calhoun	0.12	0.27	0.17	0.02	0.00	0.01	0.19	0.01	0.00	0.00	0.14	0.03	0.21	1.10	0.00	0.04	2.29
28015	mississippi	carroll	0.10	0.43	0.17	0.06	0.00	0.00	0.20	0.08	0.01	0.00	0.00	0.01	0.26	0.01	0.00	0.04	1.38
28017	mississippi	chickasaw	0.31	0.66	0.26	0.03	0.00	0.01	0.03	0.00	0.00	0.00	0.13	0.02	0.35	4.42	0.03	0.06	6.31
28019	mississippi	choctaw	0.17	0.13	0.02	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.07	3.36	0.00	0.03	3.81
28021	mississippi	claiborne	0.07	0.10	0.02	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.15	0.89	0.00	0.02	1.28
28023	mississippi	clarke	0.34	0.16	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.01	0.68	0.04	1.40
28025	mississippi	clay	0.12	0.54	0.06	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.04	0.49	1.25	0.00	0.04	2.58
28027	mississippi	coahoma	0.86	0.02	1.85	0.45	0.00	0.50	1.76	0.11	0.36	0.00	0.01	0.20	0.01	0.00	0.11	0.00	6.26
28029	mississippi	copiah	0.40	0.25	0.04	0.02	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.24	0.01	0.69	0.07	1.74
28031	mississippi	covington	1.23	0.19	0.13	0.07	0.00	0.05	0.01	0.00	0.01	0.00	0.00	0.00	0.98	0.00	2.33	0.03	5.02
28033	mississippi	de soto	0.33	0.48	1.35	0.19	0.00	0.44	0.19	0.24	0.03	0.00	0.00	0.08	0.25	0.00	0.06	0.15	3.80
28035	mississippi	forrest	0.34	0.23	0.03	0.02	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.23	0.01	0.56	0.07	1.51
28037	mississippi	franklin	0.05	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.11	0.01	0.36
28039	mississippi	george	0.01	0.07	0.01	0.01	0.00	0.00	0.07	0.00	0.11	0.04	0.00	0.00	0.25	0.01	0.00	0.07	0.66
28041	mississippi	greene	0.11	0.17	0.01	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.10	0.02	0.25	0.05	0.74
28043	mississippi	grenada	0.04	0.25	0.06	0.01	0.00	0.00	0.10	0.01	0.01	0.00	0.01	0.00	0.19	0.00	0.03	0.02	0.72
28045	mississippi	hancock	0.01	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.01	0.00	0.06	0.31
28047	mississippi	harrison	0.01	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.15	0.36
28049	mississippi	hinds	0.30	1.17	0.20	0.07	0.00	0.01	0.17	0.03	0.01	0.00	0.00	0.00	0.92	1.92	0.22	0.23	5.25
28051	mississippi	holmes	0.35	0.47	0.38	0.06	0.00	0.01	0.64	0.08	0.05	0.01	0.01	0.06	0.16	0.01	0.03	0.03	2.35
28053	mississippi	humphreys	0.51	0.00	1.33	0.18	0.00	0.18	0.71	0.02	0.06	0.00	0.00	0.08	0.00	0.00	0.11	0.00	3.21
28055	mississippi	issaquena	0.32	0.01	0.88	0.03	0.00	0.06	0.30	0.02	0.07	0.00	0.00	0.04	0.00	0.00	0.09	0.00	1.82
28057	mississippi	itawamba	0.11	0.10	0.10	0.02	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.01	0.12	1.33	0.05	0.04	1.89
28059	mississippi	jackson	0.03	0.38	0.01	0.00	0.00	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.44	0.00	0.06	0.14	1.10
28061	mississippi	jasper	1.00	0.12	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	1.82	0.04	3.35
28063	mississippi	jefferson	0.11	0.18	0.08	0.01	0.00	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.10	0.01	0.23	0.03	0.83
28065	mississippi	jefferson davis	0.30	0.15	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.18	0.00	0.62	0.04	1.35
28067	mississippi	jones	2.17	0.22	0.20	0.11	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.50	0.02	3.79	0.08	7.20
28069	mississippi	kemper	0.02	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.13	0.03	0.54
28071	mississippi	lafayette	0.09	0.32	0.05	0.01	0.00	0.01	0.05	0.00	0.00	0.00	0.02	0.00	0.20	1.54	0.00	0.06	2.34
28073	mississippi	lamar	0.34	0.28	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.01	0.57	0.09	1.60
28075	mississippi	lauderdale	0.03	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.17	0.05	0.57
28077	mississippi	lawrence	0.60	0.13	0.06	0.03	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.19	0.00	1.09	0.04	2.18
28079	mississippi	leake	2.50	0.22	0.24	0.13	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.28	0.00	4.55	0.07	8.11
28081	mississippi	lee	0.17	0.46	0.70	0.07	0.00	0.02	0.14	0.03	0.01	0.00	0.03	0.02	0.28	0.00	0.04	0.08	2.05
28083	mississippi	leflore	1.32	0.03	2.29	0.20	0.00	0.36	0.82	0.06	0.10	0.01	0.01	0.17	0.01	0.00	0.00	0.00	5.38
28085	mississippi	lincoln	0.60	0.24	0.06	0.03	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.61	0.01	0.99	0.06	2.64
28087	mississippi	lowndes	0.19	0.58	0.13	0.04	0.00	0.00	0.09	0.02	0.02	0.00	0.00	0.01	0.24	1.22	0.03	0.04	2.62
28089	mississippi	madison	0.27	0.98	0.30	0.05	0.00	0.01	0.23	0.01	0.00	0.00	0.00	0.02	0.48	0.00	0.00	0.13	2.50
28091	mississippi	marion	0.93	0.20	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.00	1.89	0.06	3.80
28093	mississippi	marshall	0.13	0.56	0.37	0.05	0.00	0.03	0.04	0.08	0.01	0.00	0.00	0.06	0.52	0.00	0.00	0.08	1.95
28095	mississippi	monroe	0.29	0.57	0.42	0.09	0.00	0.01	0.06	0.03	0.04	0.00	0.01	0.01	0.51	1.25	0.03	0.05	3.38
28097	mississippi	montgomery	0.08	0.15	0.02	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.01	0.00	0.17	1.04	0.00	0.02	1.56
28099	mississippi	neshoba	2.15	0.21	0.20	0.11	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.32	0.00	3.89	0.08	7.06
28101	mississippi	newton	0.73	0.15	0.08	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.00	1.35	0.07	2.87
28103	mississippi	noxubee	0.58	0.62	0.21	0.04	0.00	0.01	0.15	0.01	0.01	0.00	0.00	0.02	0.31	1.12	0.47	0.03	3.59
28105	mississippi	oktibbeha	0.08	0.32	0.02	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.30	1.06	0.03	0.06	1.89
28107	mississippi	panola	0.29	0.76	0.78	0.31	0.00	0.34	0.34	0.07	0.09	0.00	0.02	0.05	0.42	0.01	0.03	0.07	3.59
28109	mississippi	pearl river	0.02	0.29	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.78	0.02	0.00	0.23	1.37
28111	mississippi	perry	0.15	0.19	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.28	0.02	0.79
28113	mississippi	pike	0.43	0.20	0.04	0.02	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.36	0.02	0.78	0.07	1.96
28115	mississippi	pontotoc	0.09	0.69	0.35	0.02	0.00	0.01	0.03	0.00	0.01	0.00	0.05	0.06	0.36	0.00	0.00	0.14	1.80
28117	mississippi	prentiss	0.03	0.29	0.19	0.01	0.00	0.01	0.03	0.01	0.00	0.00	0.01	0.00	0.18	0.00	0.00	0.06	0.81
28119	mississippi	quitman	0.34	0.03	1.46	0.15	0.00	0.97	0.65	0.02	0.10	0.00	0.00	0.09	0.00	0.00	0.00	0.00	3.82
28121	mississippi	rankin	1.70	0.44	0.23	0.09	0.00	0.07	0.04	0.00	0.01	0.00	0.00	0.02	0.61	0.01	3.02	0.17	6.43
28123	mississippi	scott	1.91	0.19	0.19	0.10	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.01	0.39	0.00	3.39	0.07	6.35
28125	mississippi	sharkey	0.54	0.00	0.96	0.08	0.00	0.03	0.31	0.02	0.05	0.00	0.00	0.18	0.00	0.00	0.00	0.00	2.18
28127	mississippi	simpson	1.60	0.23	0.16	0.09	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.30	0.88	2.98	0.06	6.38
28129	mississippi	smith	2.01	0.14	0.20	0.10	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.29	0.03	3.55	0.08	6.50
28131	mississippi	stone	0.02	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.26	0.01	0.04	0.09	0.58
28133	mississippi	sunflower	1.09	0.03	3.51	0.54	0.00	0.97	0.28	0.05	0.11	0.01	0.01	0.70	0.00	0.01	0.03	0.01	7.35
28135	mississippi	tallahatchie	0.79	0.28	2.04	0.48	0.00	0.37	0.43	0.06	0.10	0.00	0.01	0.26	0.11	0.00	0.00	0.02	4.95
28137	mississippi	tate	0.18	0.41	0.39	0.12	0.00	0.07	0.13	0.04	0.02	0.00	0.03	0.03	0.73	0.00	0.00	0.11	2.27
28139	mississippi	tippah	0.11	0.27	0.10	0.04	0.00	0.01	0.02	0.00	0.00	0.00	0.00	0.01	0.23	1.33	0.00	0.08	2.22
28141	mississippi	tishomingo	0.01	0.14	0.03	0.01	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.04	0.32
28143	mississippi	tunica	0.79	0.03	1.43	0.30	0.00	1.22	0.64	0.19	0.15	0.00	0.01	0.28	0.02	0.00	0.00	0.00	5.06
28145	mississippi	union	0.13	0.36	0.24	0.01	0.00	0.01	0.03	0.00	0.00	0.00	0.01	0.00	0.28	1.34	0.00	0.07	2.47
28147	mississippi	walthall	0.66	0.23	0.08	0.04	0.00	0.03	0.00	0.00	0.01	0.00	0.00	0.01	0.73	0.01	1.14	0.07	3.00
28149	mississippi	warren	0.20	0.26	0.29	0.08	0.00	0.00	0.07	0.02	0.05	0.00	0.00	0.03	0.02	0.00	0.03	0.03	1.07
28151	mississippi	washington	1.06	0.01	2.86	0.39	0.00	0.63	0.39	0.04	0.15	0.00	0.01	0.54	0.01	0.00	0.10	0.01	6.19
28153	mississippi	wayne	1.73	0.17	0.16	0.09	0.00	0.07	0.01	0.00	0.00	0.01	0.00	0.00	0.18	0.00	3.05	0.04	5.51
28155	mississippi	webster	0.12	0.20	0.03	0.01	0.00	0.00	0.08	0.00	0.00	0.00	0.04	0.00	0.07	1.85	0.12	0.02	2.54
28157	mississippi	wilkinson	0.01	0.17	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.12	0.00	0.02	0.01	0.36
28159	mississippi	winston	0.66	0.21	0.07	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.16	5.80	0.78	0.05	7.78
28161	mississippi	yalobusha	0.05	0.22	0.05	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.02	0.01	0.16	0.01	0.03	0.03	0.67
28163	mississippi	yazoo	2.15	0.71	0.79	0.55	0.00	0.14	0.90	1.21	0.10	0.03	0.03	0.12	0.27	0.00	0.00	0.05	7.05
29001	missouri	adair	0.38	0.58	0.51	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.91	1.54	0.01	0.08	4.06
29003	missouri	andrew	1.18	0.38	1.16	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.01	0.00	0.08	3.39
29005	missouri	atchison	1.66	0.24	1.47	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.00	0.13	1.11	0.01	0.02	4.73
29007	missouri	audrain	2.67	0.44	3.33	0.45	0.00	0.19	0.00	0.12	0.00	0.00	0.00	0.00	0.88	2.69	0.00	0.13	10.91
29009	missouri	barry	1.87	0.47	0.19	0.10	0.00	0.08	0.01	0.03	0.00	0.00	0.01	0.00	1.56	0.01	3.41	0.16	7.87
29011	missouri	barton	1.14	1.04	1.20	0.69	0.00	0.04	0.00	0.38	0.00	0.00	0.01	0.00	0.81	1.72	0.01	0.08	7.12
29013	missouri	bates	1.32	1.06	2.12	0.36	0.00	0.02	0.00	0.20	0.01	0.01	0.00	0.00	1.62	1.54	0.00	0.13	8.40
29015	missouri	benton	0.46	0.40	0.30	0.06	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.79	1.35	0.35	0.07	3.83
29017	missouri	bollinger	0.12	0.35	0.26	0.04	0.00	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.48	0.04	0.01	0.07	1.44
29019	missouri	boone	0.54	0.59	0.89	0.15	0.00	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.51	0.57	0.01	0.24	3.59
29021	missouri	buchanan	1.50	0.36	1.56	0.07	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.53	2.10	0.01	0.08	6.24
29023	missouri	butler	0.49	0.20	1.10	0.11	0.00	1.27	0.04	0.11	0.00	0.00	0.00	0.00	0.16	1.63	0.01	0.06	5.19
29025	missouri	caldwell	0.37	0.59	0.85	0.11	0.00	0.05	0.00	0.02	0.00	0.00	0.01	0.00	0.49	1.38	0.01	0.07	3.95
29027	missouri	callaway	0.62	0.49	0.59	0.04	0.00	0.03	0.00	0.07	0.00	0.00	0.00	0.00	0.83	2.42	0.00	0.20	5.29
29029	missouri	camden	0.11	0.11	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	1.41	0.00	0.07	1.98
29031	missouri	cape girardeau	1.04	0.77	1.37	0.41	0.00	0.09	0.00	0.09	0.00	0.01	0.02	0.01	1.27	0.08	0.01	0.14	5.32
29033	missouri	carroll	1.70	0.64	2.06	0.12	0.00	0.02	0.00	0.02	0.00	0.00	0.01	0.00	0.93	1.34	0.00	0.05	6.89
29035	missouri	carter	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.02	0.22
29037	missouri	cass	2.02	1.81	3.43	0.38	0.00	0.04	0.00	0.07	0.00	0.00	0.00	0.00	2.03	3.36	0.01	0.44	13.59
29039	missouri	cedar	0.10	0.60	0.10	0.03	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.85	0.01	0.00	0.10	1.82
29041	missouri	chariton	1.40	0.66	2.07	0.19	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.93	0.27	0.01	0.04	5.62
29043	missouri	christian	0.09	0.62	0.01	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.00	1.00	0.01	0.01	0.24	2.03
29045	missouri	clark	1.00	0.33	1.05	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.02	0.00	0.03	3.13
29047	missouri	clay	0.84	0.86	1.51	0.11	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	2.32	0.11	0.03	0.33	6.14
29049	missouri	clinton	1.07	0.48	1.08	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.80	1.83	0.02	0.09	5.41
29051	missouri	cole	0.21	0.34	0.18	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.01	0.00	0.76	1.43	0.01	0.07	3.05
29053	missouri	cooper	0.89	0.49	1.19	0.43	0.00	0.01	0.00	0.07	0.00	0.00	0.01	0.00	1.03	0.20	0.07	0.05	4.44
29055	missouri	crawford	0.04	0.16	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.51	0.00	0.00	0.10	0.83
29057	missouri	dade	0.43	0.65	0.41	0.34	0.00	0.06	0.00	0.14	0.00	0.00	0.02	0.00	1.30	1.28	0.01	0.09	4.73
29059	missouri	dallas	0.12	0.35	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.91	0.01	0.01	0.18	1.62
29061	missouri	daviess	1.01	0.60	1.29	0.08	0.00	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.39	4.79	0.00	0.11	8.34
29063	missouri	de kalb	0.77	0.52	1.00	0.07	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.65	1.47	0.01	0.06	4.59
29065	missouri	dent	0.04	0.23	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.45	0.01	0.00	0.11	0.86
29067	missouri	douglas	0.09	0.24	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.64	0.00	0.00	0.11	1.10
29069	missouri	dunklin	0.52	0.15	1.44	0.32	0.00	1.22	2.49	0.20	0.01	0.25	0.10	0.02	0.02	0.00	0.01	0.02	6.77
29071	missouri	franklin	0.79	0.75	0.51	0.06	0.00	0.04	0.00	0.02	0.00	0.00	0.00	0.00	1.42	2.04	0.01	0.27	5.91
29073	missouri	gasconade	0.09	0.26	0.07	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.51	0.28	0.00	0.09	1.33
29075	missouri	gentry	0.57	0.59	0.77	0.06	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.67	1.25	0.01	0.05	3.99
29077	missouri	greene	0.22	1.35	0.11	0.08	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.00	2.48	0.02	0.01	0.36	4.70
29079	missouri	grundy	0.42	0.37	0.84	0.08	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.36	1.23	0.00	0.06	3.37
29081	missouri	harrison	0.90	0.67	1.11	0.05	0.00	0.01	0.00	0.03	0.00	0.00	0.01	0.00	0.65	1.22	0.01	0.07	4.74
29083	missouri	henry	0.45	0.90	1.29	0.32	0.00	0.03	0.00	0.10	0.00	0.00	0.00	0.00	1.27	1.18	0.01	0.09	5.65
29085	missouri	hickory	0.07	0.30	0.03	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.54	0.00	0.00	0.05	1.02
29087	missouri	holt	1.62	0.34	1.15	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.07	1.22	0.01	0.01	4.44
29089	missouri	howard	0.67	0.37	0.69	0.09	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.59	1.48	0.01	0.08	4.02
29091	missouri	howell	0.08	0.56	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.24	0.02	0.00	0.16	2.08
29093	missouri	iron	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.06	0.28
29095	missouri	jackson	1.62	0.77	2.27	0.24	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	1.20	0.00	0.01	0.40	6.56
29097	missouri	jasper	0.65	0.82	0.74	0.57	0.00	0.02	0.00	0.27	0.00	0.00	0.00	0.00	1.26	0.81	0.06	0.13	5.35
29099	missouri	jefferson	0.18	0.36	0.14	0.02	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.41	0.07	0.02	0.26	1.50
29101	missouri	johnson	0.91	1.34	1.46	0.07	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	2.08	0.03	1.00	0.23	7.19
29103	missouri	knox	0.98	0.38	1.45	0.10	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.75	0.03	0.03	4.44
29105	missouri	laclede	0.24	0.46	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.12	1.39	0.00	0.14	3.41
29107	missouri	lafayette	2.36	0.55	2.59	0.12	0.00	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.90	0.87	0.00	0.10	7.52
29109	missouri	lawrence	0.68	1.06	0.27	0.16	0.00	0.03	0.00	0.04	0.00	0.00	0.00	0.00	2.70	0.05	0.91	0.19	6.10
29111	missouri	lewis	1.30	0.31	1.50	0.12	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.69	1.63	0.00	0.10	5.68
29113	missouri	lincoln	1.43	0.57	2.05	0.37	0.00	0.08	0.00	0.05	0.00	0.00	0.02	0.00	0.88	0.58	1.38	0.13	7.55
29115	missouri	linn	0.45	0.58	0.97	0.09	0.00	0.01	0.00	0.02	0.00	0.00	0.01	0.00	1.04	0.04	0.00	0.04	3.24
29117	missouri	livingston	0.66	0.40	1.58	0.12	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.43	0.33	0.00	0.03	3.61
29121	missouri	macon	0.57	0.72	1.11	0.10	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	1.00	0.73	0.00	0.14	4.39
29123	missouri	madison	0.15	0.18	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.31	1.96	0.00	0.04	2.68
29125	missouri	maries	0.10	0.30	0.05	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.84	0.59	0.00	0.08	1.99
29127	missouri	marion	1.49	0.34	1.33	0.17	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.43	2.46	0.00	0.05	6.31
29119	missouri	mcdonald	0.87	0.22	0.08	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.01	1.66	0.09	3.65
29129	missouri	mercer	0.33	0.39	0.39	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.47	1.20	0.01	0.03	2.85
29131	missouri	miller	0.29	0.31	0.07	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.80	3.73	0.02	0.10	5.34
29133	missouri	mississippi	1.15	0.03	2.22	0.60	0.00	0.09	0.01	0.69	0.00	0.21	0.01	0.00	0.03	0.00	0.96	0.01	6.02
29135	missouri	moniteau	0.45	0.54	0.33	0.08	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	1.69	1.09	0.27	0.09	4.59
29137	missouri	monroe	1.46	0.49	1.75	0.15	0.00	0.06	0.00	0.01	0.00	0.00	0.01	0.00	0.68	2.31	0.00	0.07	7.00
29139	missouri	montgomery	1.16	0.31	1.17	0.24	0.00	0.06	0.00	0.08	0.00	0.00	0.01	0.00	0.64	0.49	0.00	0.10	4.29
29141	missouri	morgan	0.59	0.35	0.17	0.05	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.84	1.23	0.58	0.14	3.99
29143	missouri	new madrid	1.49	0.03	2.96	0.33	0.00	0.76	2.98	0.17	0.00	0.00	0.01	0.00	0.01	0.00	0.87	0.01	9.64

29145	missouri	newton	1.52	0.85	0.22	0.18	0.00	0.06	0.01	0.09	0.00	0.00	0.00	0.00	2.36	0.01	2.65	0.21	8.14
29147	missouri	nodaway	1.47	1.12	1.74	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.92	0.27	0.00	0.11	5.70
29149	missouri	oregon	0.07	0.28	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.01	0.00	0.14	1.22
29151	missouri	osage	0.24	0.34	0.10	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.98	1.03	0.02	0.04	2.76
29153	missouri	ozark	0.05	0.25	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.00	0.00	0.09	1.07
29155	missouri	pemiscot	0.72	0.02	2.71	0.30	0.00	2.12	1.77	0.29	0.01	0.02	0.02	0.00	0.00	0.00	0.00	0.01	7.98
29157	missouri	perry	1.32	0.88	1.34	0.52	0.00	0.01	0.00	0.27	0.00	0.00	0.03	0.00	2.42	0.33	0.00	0.11	7.24
29159	missouri	pettis	1.85	0.83	1.84	0.33	0.00	0.06	0.00	0.06	0.00	0.00	0.00	0.00	1.35	1.11	1.27	0.12	8.82
29161	missouri	phelps	0.03	0.18	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.01	0.00	0.14	0.72
29163	missouri	pike	2.19	0.59	1.68	0.35	0.00	0.06	0.04	0.51	0.00	0.00	0.01	0.00	1.04	1.35	0.00	0.13	7.95
29165	missouri	platte	1.42	0.57	1.94	0.07	0.00	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.50	3.34	0.00	0.13	8.02
29167	missouri	polk	0.23	0.91	0.11	0.05	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	2.31	0.22	0.03	0.18	4.08
29169	missouri	pulaski	0.11	0.12	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	1.43	0.00	0.05	1.97
29171	missouri	putnam	0.26	0.47	0.34	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.99	1.29	0.01	0.05	3.45
29173	missouri	ralls	1.57	0.34	1.78	0.22	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.38	1.73	0.01	0.05	6.13
29175	missouri	randolph	0.32	0.40	0.67	0.10	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.51	1.41	0.00	0.09	3.52
29177	missouri	ray	1.08	0.79	1.71	0.06	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.90	0.16	0.00	0.16	4.91
29179	missouri	reynolds	0.10	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	1.45	0.00	0.07	1.78
29181	missouri	ripley	0.14	0.22	0.14	0.02	0.00	0.25	0.00	0.01	0.00	0.00	0.00	0.00	0.30	1.50	0.00	0.05	2.63
29195	missouri	saline	2.89	0.41	2.72	0.11	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.55	3.04	0.00	0.04	9.80
29197	missouri	schuyler	0.18	0.34	0.32	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.51	0.33	0.00	0.09	1.77
29199	missouri	scotland	0.85	0.42	0.88	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.72	2.21	0.00	0.06	5.21
29201	missouri	scott	1.81	0.34	1.56	0.60	0.00	0.33	0.15	0.50	0.00	0.04	0.02	0.00	0.31	0.01	1.18	0.04	6.89
29203	missouri	shannon	0.02	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.07	0.42
29205	missouri	shelby	1.07	0.36	1.90	0.12	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.50	0.91	0.81	0.05	5.77
29183	missouri	st charles	3.14	0.45	2.67	0.26	0.00	0.02	0.00	0.12	0.00	0.00	0.03	0.00	0.58	2.50	0.03	0.14	9.95
29185	missouri	st clair	0.17	0.56	0.29	0.07	0.00	0.01	0.00	0.16	0.00	0.00	0.00	0.00	0.81	0.10	0.00	0.10	2.28
29187	missouri	st francois	0.10	0.38	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.02	1.90	0.21	3.32
29189	missouri	st louis	0.51	0.16	0.69	0.13	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.20	1.91
29510	missouri	st louis city	0.09	0.07	0.01	0.01	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23
29186	missouri	ste genevieve	0.18	0.35	0.16	0.03	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.48	0.03	0.00	0.09	1.41
29207	missouri	stoddard	3.00	0.60	2.81	0.62	0.00	1.85	2.67	0.31	0.00	0.01	0.02	0.00	0.30	0.01	0.87	0.07	13.14
29209	missouri	stone	0.21	0.17	0.04	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.56	0.00	0.24	0.09	1.35
29211	missouri	sullivan	1.06	0.60	0.53	0.07	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.00	0.86	14.15	0.00	0.04	17.36
29213	missouri	taney	0.02	0.16	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.02	0.00	0.05	0.48
29215	missouri	texas	0.14	0.45	0.01	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	1.00	0.01	0.00	0.19	1.83
29217	missouri	vernon	1.75	1.02	1.03	0.35	0.00	0.05	0.00	0.13	0.06	0.00	0.00	0.00	1.31	14.65	0.02	0.08	20.45
29219	missouri	warren	0.57	0.24	0.45	0.05	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.33	0.28	0.01	0.10	2.10
29221	missouri	washington	0.04	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.01	0.00	0.10	0.67
29223	missouri	wayne	0.14	0.17	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.20	1.57	0.00	0.05	2.18
29225	missouri	webster	0.24	0.72	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.69	0.21	0.03	0.24	3.22
29227	missouri	worth	0.19	0.30	0.22	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	1.15	0.00	0.02	2.18
29229	missouri	wright	0.22	0.44	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.14	0.01	0.00	0.14	2.01
30001	montana	beaverhead	0.01	0.37	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.01	0.00	0.05	0.98
30003	montana	big horn	0.03	0.49	0.00	0.69	0.04	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.01	0.00	0.06	1.77
30005	montana	blaine	0.01	0.43	0.00	0.65	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.01	0.22	0.01	0.00	0.03	1.41
30007	montana	broadwater	0.00	0.18	0.00	0.13	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.01	0.08	0.00	0.00	0.01	0.44
30009	montana	carbon	0.02	0.22	0.00	0.07	0.01	0.04	0.00	0.00	0.00	0.00	0.01	0.01	0.33	0.01	0.00	0.05	0.76
30011	montana	carter	0.02	0.20	0.00	0.16	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.40	0.07	0.00	0.11	1.00
30013	montana	cascade	0.02	0.81	0.00	0.91	0.00	0.20	0.00	0.00	0.00	0.00	0.00	0.03	0.45	0.01	0.01	0.10	2.56
30015	montana	chouteau	0.01	0.77	0.00	1.66	0.00	0.13	0.00	0.02	0.00	0.00	0.01	0.03	0.12	0.01	0.00	0.01	2.77
30017	montana	custer	0.04	0.32	0.00	0.11	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.52	0.01	0.00	0.04	1.07
30019	montana	daniels	0.00	0.35	0.00	1.24	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.21	0.06	0.00	0.00	0.01	1.90
30021	montana	dawson	0.02	0.37	0.00	1.13	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.07	0.20	0.01	0.00	0.03	1.90
30023	montana	deer lodge	0.00	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.00	0.01	0.08
30025	montana	fallon	0.02	0.33	0.00	0.20	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.01	0.27	0.00	0.00	0.02	0.87
30027	montana	fergus	0.01	0.71	0.00	0.51	0.00	0.09	0.00	0.01	0.00	0.00	0.01	0.01	0.28	0.00	0.00	0.03	1.65
30029	montana	flathead	0.00	0.15	0.00	0.11	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.05	0.39
30031	montana	gallatin	0.02	0.48	0.00	0.25	0.00	0.18	0.00	0.00	0.00	0.00	0.02	0.02	0.25	0.01	0.00	0.06	1.30
30033	montana	garfield	0.02	0.38	0.00	0.21	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.31	0.07	0.00	0.07	1.09
30035	montana	glacier	0.01	0.25	0.00	0.33	0.00	0.24	0.00	0.02	0.00	0.00	0.00	0.04	0.13	0.01	0.01	0.04	1.07
30037	montana	golden valley	0.00	0.13	0.00	0.07	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.09	0.01	0.00	0.04	0.38
30039	montana	granite	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.01	0.12
30041	montana	hill	0.00	0.27	0.00	1.31	0.00	0.04	0.00	0.01	0.00	0.00	0.01	0.04	0.07	0.00	0.00	0.01	1.77
30043	montana	jefferson	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.02	0.23
30045	montana	judith basin	0.01	0.40	0.00	0.25	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.01	0.23	0.00	0.00	0.02	0.97
30047	montana	lake	0.01	0.23	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.13	0.01	0.00	0.06	0.51
30049	montana	lewis and clark	0.01	0.16	0.00	0.07	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.06	0.00	0.05	0.54
30051	montana	liberty	0.00	0.12	0.00	0.75	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.04	0.05	0.00	0.00	0.00	1.05
30053	montana	lincoln	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.01	0.04
30057	montana	madison	0.01	0.38	0.00	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.01	0.00	0.04	0.75
30055	montana	mccone	0.01	0.27	0.00	1.06	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.08	0.14	0.01	0.00	0.03	1.63
30059	montana	meagher	0.00	0.16	0.00	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.02	0.39
30061	montana	mineral	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.02
30063	montana	missoula	0.01	0.12	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.02	0.00	0.09	0.40
30065	montana	musselshell	0.01	0.25	0.00	0.14	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.01	0.00	0.04	0.66
30067	montana	yellowstone national park	0.00	0.24	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.04	0.54
30069	montana	petroleum	0.00	0.15	0.00	0.16	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.01	0.44
30071	montana	phillips	0.02	0.36	0.00	0.41	0.00	0.03	0.00	0.01	0.00	0.00	0.01	0.02	0.30	0.07	0.00	0.03	1.25
30073	montana	pondera	0.00	0.20	0.00	0.32	0.00	0.16	0.00	0.00	0.00	0.00	0.00	0.01	0.05	0.00	0.00	0.02	0.77
30075	montana	powder river	0.01	0.22	0.00	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.01	0.00	0.05	0.70
30077	montana	powell	0.00	0.11	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.01	0.00	0.01	0.25
30079	montana	prairie	0.02	0.14	0.00	0.14	0.01	0.02	0.00	0.00	0.00	0.00	0.01	0.02	0.16	0.00	0.00	0.01	0.53
30081	montana	ravalli	0.01	0.12	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.06	0.29
30083	montana	richland	0.05	0.55	0.01	1.35	0.08	0.13	0.00	0.04	0.00	0.00	0.04	0.02	0.34	0.01	0.00	0.03	2.61
30085	montana	roosevelt	0.01	0.54	0.00	1.93	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.15	0.11	0.01	0.00	0.03	2.82
30087	montana	rosebud	0.03	0.38	0.00	0.16	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.00	0.00	0.03	1.06
30089	montana	sanders	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.01	0.00	0.02	0.13
30091	montana	sheridan	0.01	0.41	0.01	1.75	0.00	0.03	0.00	0.05	0.00	0.00	0.01	0.37	0.13	0.00	0.00	0.01	2.77
30093	montana	silver bow	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.01	0.04
30095	montana	stillwater	0.01	0.26	0.00	0.16	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.19	0.01	0.00	0.04	0.71
30097	montana	sweet grass	0.00	0.12	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.00	0.03	0.32
30099	montana	teton	0.01	0.56	0.00	0.30	0.00	0.20	0.00	0.00	0.00	0.00	0.00	0.01	0.11	0.04	0.00	0.03	1.28
30101	montana	toole	0.00	0.18	0.00	0.65	0.00	0.24	0.00	0.01	0.00	0.00	0.00	0.04	0.06	0.00	0.00	0.02	1.21
30103	montana	treasure	0.02	0.09	0.00	0.07	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.01	0.00	0.00	0.36
30105	montana	valley	0.02	0.64	0.01	1.33	0.00	0.03	0.00	0.04	0.00	0.00	0.00	0.20	0.26	0.04	0.00	0.02	2.58
30107	montana	wheatland	0.00	0.22	0.00	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.03	0.52
30109	montana	wibaux	0.01	0.22	0.00	0.27	0.00	0.02	0.00	0.03	0.00	0.00	0.01	0.03	0.12	0.01	0.00	0.01	0.72
30111	montana	yellowstone	0.22	0.98	0.01	1.30	0.08	0.31	0.00	0.05	0.00	0.00	0.00	0.07	1.11	0.00	0.00	0.15	4.29
31001	nebraska	adams	3.11	0.13	1.23	0.11	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	1.08	0.17	0.00	0.02	5.91
31003	nebraska	antelope	2.16	0.31	0.71	0.05	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	1.74	2.28	0.00	0.02	7.39
31005	nebraska	arthur	0.12	0.07	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.18	0.00	0.01	0.65
31007	nebraska	banner	0.28	0.10	0.02	0.43	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.03	0.60	0.19	0.00	0.01	1.74
31009	nebraska	blaine	0.19	0.08	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.20	0.00	0.01	0.98
31011	nebraska	boone	2.55	0.32	0.87	0.05	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	1.54	2.73	0.00	0.01	8.13
31013	nebraska	box butte	0.41	0.34	0.02	0.52	0.09	0.02	0.00	0.03	0.00	0.00	0.01	0.13	0.65	0.00	0.00	0.01	2.24
31015	nebraska	boyd	0.36	0.35	0.15	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.01	0.00	0.01	1.57
31017	nebraska	brown	0.67	0.23	0.08	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.33	0.20	0.00	0.05	2.65
31019	nebraska	buffalo	4.61	0.77	1.22	0.14	0.00	0.05	0.00	0.00	0.00	0.00	0.03	0.00	2.16	0.06	0.00	0.10	9.15
31021	nebraska	burt	2.56	0.47	1.89	0.23	0.00	0.02	0.00	0.03	0.00	0.00	0.04	0.00	0.60	0.57	0.00	0.03	6.45
31023	nebraska	butler	3.22	0.37	1.77	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.78	0.91	0.23	0.04	7.39
31025	nebraska	cass	3.46	0.28	3.09	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.05	0.00	0.33	0.09	0.00	0.04	7.40
31027	nebraska	cedar	3.34	0.55	1.58	0.06	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	2.37	1.32	0.00	0.03	9.34
31029	nebraska	chase	1.31	0.43	0.06	0.26	0.00	0.06	0.00	0.00	0.00	0.00	0.01	0.04	1.00	0.18	0.00	0.01	3.38
31031	nebraska	cherry	0.82	0.67	0.07	0.07	0.00	0.04	0.00	0.01	0.00	0.00	0.01	0.01	2.47	0.00	0.00	0.04	4.22
31033	nebraska	cheyenne	0.44	0.20	0.02	0.74	0.01	0.24	0.00	0.01	0.00	0.00	0.00	0.06	0.56	0.00	0.00	0.01	2.29
31035	nebraska	clay	3.33	0.20	1.10	0.10	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.91	2.90	0.01	0.04	8.66
31037	nebraska	colfax	3.00	0.29	1.22	0.07	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	1.82	2.87	0.00	0.02	9.34
31039	nebraska	cuming	4.50	0.36	1.91	0.12	0.00	0.13	0.01	0.01	0.00	0.00	0.00	0.00	6.39	2.61	0.00	0.03	16.08
31041	nebraska	custer	3.39	1.02	0.61	0.20	0.00	0.08	0.01	0.00	0.00	0.00	0.02	0.00	3.73	1.64	0.00	0.08	10.77
31043	nebraska	dakota	1.38	0.10	0.89	0.24	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.48	0.00	0.02	3.33
31045	nebraska	dawes	0.19	0.23	0.02	0.18	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.01	0.56	0.00	0.00	0.02	1.24
31047	nebraska	dawson	3.51	0.73	0.69	0.11	0.00	0.05	0.01	0.00	0.00	0.00	0.00	0.00	3.18	0.75	0.00	0.05	9.09
31049	nebraska	deuel	0.21	0.07	0.01	0.38	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.17	0.00	0.00	0.01	0.89
31051	nebraska	dixon	2.01	0.23	1.14	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.98	0.69	0.23	0.04	5.40
31053	nebraska	dodge	3.15	0.20	1.58	0.03	0.00	0.05	0.00	0.01	0.00	0.00	0.01	0.00	1.07	1.16	0.00	0.03	7.29
31055	nebraska	douglas	2.18	1.04	1.46	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.23	1.37	0.04	0.16	6.55
31057	nebraska	dundy	0.72	0.51	0.04	0.23	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.02	0.65	0.00	0.11	0.01	2.34
31059	nebraska	fillmore	3.01	0.13	1.35	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.56	0.57	0.01	0.01	5.77
31061	nebraska	franklin	0.81	0.16	0.35	0.16	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.01	0.00	0.01	1.85
31063	nebraska	frontier	0.92	0.15	0.18	0.43	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.66	0.03	0.00	0.01	2.47
31065	nebraska	furnas	1.21	0.21	0.33	0.92	0.00	0.10	0.00	0.01	0.00	0.00	0.02	0.01	0.43	0.22	0.01	0.01	3.46
31067	nebraska	gage	3.96	0.57	2.58	0.42	0.00	0.09	0.02	0.22	0.00	0.00	0.01	0.00	0.78	1.02	0.25	0.04	9.97
31069	nebraska	garden	0.42	0.23	0.02	0.40	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.02	0.58	0.00	0.00	0.02	1.71
31071	nebraska	garfield	0.31	0.21	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.23	0.00	0.02	1.40
31073	nebraska	gosper	0.90	0.10	0.36	0.15	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.22	0.00	0.01	2.10
31075	nebraska	grant	0.08	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.10	0.01	0.59
31077	nebraska	greeley	1.01	0.30	0.32	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.87	0.27	0.17	0.01	2.99
31079	nebraska	hall	3.75	0.31	0.57	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.65	0.08	0.01	0.05	6.52
31081	nebraska	hamilton	3.15	0.13	1.01	0.04	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.16	0.00	0.02	5.30
31083	nebraska	harlan	1.48	0.15	0.57	0.51	0.00	0.07	0.00	0.00	0.00	0.00	0.02	0.00	0.60	0.24	0.01	0.02	3.67
31085	nebraska	hayes	0.61	0.13	0.05	0.29	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.01	0.56	0.19	0.12	0.01	2.02
31087	nebraska	hitchcock	0.44	0.13	0.03	0.71	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.01	0.28	0.00	0.00	0.01	1.71
31089	nebraska	holt	2.51	1.32	0.50	0.14	0.00	0.14	0.01	0.00	0.00	0.00	0.02	0.02	3.09	2.50	0.00	0.05	10.29
31091	nebraska	hooker	0.06	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.01	0.31
31093	nebraska	howard	1.35	0.42	0.34	0.13	0.00	0.02	0.00	0.00	0.00	0.00	0.02	0.00	1.31	0.07	0.00	0.02	3.69
31095	nebraska	jefferson	2.46	0.32	1.43	0.26	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.59	1.77	0.23	0.02	7.14
31097	nebraska	johnson	0.92	0.31	0.57	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.12	0.01	0.01	2.42
31099	nebraska	kearney	2.18	0.35	0.83	0.11	0.00	0.03	0.00	0.00	0.00	0.00	0.02	0.00	1.23	0.05	0.01	0.02	4.82
31101	nebraska	keith	1.05	0.20	0.11	0.38	0.01	0.04	0.00	0.00	0.00	0.00	0.01	0.02	0.57	0.18	0.00	0.02	2.59
31103	nebraska	keya paha	0.32	0.30	0.04	0.07	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.71	0.00	0.00	0.02	1.49
31105	nebraska	kimball	0.17	0.19	0.01	0.66	0.01	0.10	0.00	0.02	0.00	0.00	0.00	0.01	0.18	0.00	0.00	0.01	1.38
31107	nebraska	knox	2.32	1.02	0.87	0.17	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	1.99	1.09	0.18	0.04	7.76
31109	nebraska	lancaster	5.39	1.04	4.03	0.42	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	0.97	0.58	0.01	0.15	12.64
31111	nebraska	lincoln	2.41	0.65	0.28	0.20	0.00	0.06	0.00	0.00	0.00	0.00	0.01	0.00	2.96	0.00	0.00	0.07	6.66
31113	nebraska	logan	0.26	0.10	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.00	0.00	0.01	0.71
31115	nebraska	loup	0.17	0.14	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.02	0.01	0.01	0.74
31119	nebraska	madison	1.98	0.20	0.99	0.13	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.77	0.36	0.00	0.03	5.49
31117	nebraska	mcpherson	0.17	0.04	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.18	0.00	0.01	0.80
31121	nebraska	merrick	2.51	0.32	0.62	0.03	0.00	0.05	0.00	0.00	0.00	0.00	0.02	0.00	0.79	0.51	0.00	0.02	4.87
31123	nebraska	morrill	0.87	0.28	0.04	0.19	0.03	0.03	0.00	0.00	0.00	0.00	0.01	0.08	1.48	0.00	0.00	0.02	3.03
31125	nebraska	nance	1.32	0.23	0.54	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.47	1.08	0.00	0.01	3.73
31127	nebraska	nemaha	1.67	0.45	1.48	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.25	0.00	0.02	4.18
31129	nebraska	nuckolls	1.65	0.20	0.77	0.26	0.00	0.06	0.00	0.00	0.00	0.00	0.02	0.00	0.62	0.15	0.01	0.02	3.78
31131	nebraska	otoe	2.81	0.39	2.39	0.09	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.36	1.22	0.00	0.04	7.32
31133	nebraska	pawnee	1.03	0.35	0.84	0.10	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.67	0.17	0.00	0.05	3.26
31135	nebraska	perkins	1.51	0.09	0.07	0.66	0.00	0.08	0.00	0.02	0.00	0.00	0.01	0.01	0.31	0.18	0.00	0.01	2.97
31137	nebraska	phelps	3.46	0.16	1.04	0.09	0.00	0.04	0.00	0.00	0.00	0.00	0.02	0.00	2.32	0.24	0.01	0.01	7.40
31139	nebraska	pierce	1.49	0.21	0.73	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.97	0.76	0.21	0.03	4.47
31141	nebraska	platte	4.77	0.24	1.68	0.27	0.00	0.09	0.01	0.00	0.00	0.00	0.03	0.00	2.43	5.74	0.00	0.02	15.28
31143	nebraska	polk	2.99	0.19	1.18	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.36	0.95	0.22	0.02	6.98
31145	nebraska	red willow	0.92	0.16	0.14	0.75	0.00	0.13	0.00	0.02	0.00	0.00	0.02	0.00	0.72	0.20	0.00	0.03	3.10
31147	nebraska	richardson	2.04	0.69	1.57	0.06	0.00	0.02	0.00	0.00	0.00	0.00	0.04	0.00	0.78	0.68	0.00	0.04	5.94
31149	nebraska	rock	0.46	0.29	0.08	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.21	0.00	0.01	2.18
31151	nebraska	saline	2.85	0.26	1.73	0.19	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.58	0.00	0.02	6.24
31153	nebraska	sarpy	2.46	0.29	1.74	0.04	0.00	0.03	0.00	0.01	0.00	0.00	0.09	0.00	0.59	1.65	0.00	0.11	7.01
31155	nebraska	saunders	5.56	0.61	3.48	0.09	0.00	0.04	0.01	0.00	0.00	0.00	0.01	0.00	2.56	0.18	0.00	0.09	12.64
31157	nebraska	scotts bluff	1.00	0.39	0.03	0.12	0.07	0.03	0.00	0.01	0.00	0.00	0.00	0.29	1.11	0.18	0.00	0.04	3.28
31159	nebraska	seward	3.72	0.33	1.77	0.07	0.00	0.06	0.00	0.01	0.00	0.00	0.03	0.00	1.05	1.05	0.00	0.04	8.13
31161	nebraska	sheridan	0.44	0.32	0.04	0.24	0.01	0.04	0.00	0.03	0.00	0.00	0.01	0.05	0.96	0.16	0.00	0.02	2.32
31163	nebraska	sherman	0.93	0.35	0.23	0.14	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.66	0.26	0.00	0.03	2.64
31165	nebraska	sioux	0.37	0.20	0.03	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.03	0.88	0.17	0.00	0.02	1.78
31167	nebraska	stanton	1.69	0.33	0.90	0.19	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.96	0.69	0.01	0.03	4.83
31169	nebraska	thayer	2.15	0.22	1.04	0.29	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.56	0.09	0.01	0.02	4.43
31171	nebraska	thomas	0.12	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.18	0.00	0.01	0.61
31173	nebraska	thurston	1.81	0.44	1.03	0.19	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.25	0.01	0.02	4.80
31175	nebraska	valley	0.86	0.27	0.23	0.10	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.03	0.02	0.01	0.02	2.57
31177	nebraska	washington	3.65	0.49	2.74	0.04	0.00	0.03	0.00	0.02	0.00	0.00	0.01	0.00	1.28	1.15	0.00	0.11	9.53
31179	nebraska	wayne	2.26	0.23	1.24	0.18	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.85	0.30	0.22	0.02	5.35
31181	nebraska	webster	1.45	0.29	0.63	0.40	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	1.27	0.30	0.00	0.01	4.42
31183	nebraska	wheeler	0.69	0.26	0.10	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.54	0.26	0.00	0.02	2.92
31185	nebraska	york	3.76	0.09	1.23	0.18	0.00	0.06	0.00	0.01	0.00	0.00	0.03	0.00	0.81	0.52	0.20	0.03	6.92
32510	nevada	carson city	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.02	0.29
32001	nevada	churchill	0.11	0.23	0.01	0.11	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.18	0.00	0.00	0.06	1.71
32003	nevada	clark	0.09	0.03	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	0.00	0.19	1.34
32005	nevada	douglas	0.06	0.17	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.72	0.00	0.00	0.04	1.00
32007	nevada	elko	0.26	0.33	0.02	0.08	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	3.25	0.00	0.00	0.08	4.05
32009	nevada	esmeralda	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.11
32011	nevada	eureka	0.03	0.20	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.02	0.75
32013	nevada	humboldt	0.20	0.76	0.02	0.19	0.00	0.03	0.00	0.01	0.00	0.00	0.04	0.00	2.27	0.00	0.00	0.04	3.56
32015	nevada	lander	0.04	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.00	0.00	0.03	0.77
32017	nevada	lincoln	0.05	0.12	0.01	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.60	0.00	0.00	0.01	0.89
32019	nevada	lyon	0.23	0.29	0.02	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	2.79	0.02	0.00	0.12	3.57
32021	nevada	mineral	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.01	0.11
32023	nevada	nye	0.12	0.12	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.04	0.00	1.48	0.00	0.00	0.02	1.81
32027	nevada	pershing	0.12	0.36	0.01	0.12	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.40	0.00	0.00	0.01	2.03
32029	nevada	storey	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.82	0.00	0.00	0.00	0.91
32031	nevada	washoe	0.19	0.14	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	2.29	0.00	0.01	0.39	3.06
32033	nevada	white pine	0.04	0.07	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.49	0.00	0.00	0.03	0.70
33001	new hampshire	belknap	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.01	0.07	0.21
33003	new hampshire	carroll	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.00	0.06	0.16
33005	new hampshire	cheshire	0.09	0.10	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.02	0.11	0.14	0.64
33007	new hampshire	coos	0.04	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.20
33009	new hampshire	grafton	0.08	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.01	0.06	0.09	0.53
33011	new hampshire	hillsborough	0.04	0.34	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.02	0.11	0.05	0.03	0.47	1.12
33013	new hampshire	merrimack	0.10	0.22	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.03	0.01	0.26	0.86
33015	new hampshire	rockingham	0.06	0.22	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.13	0.02	0.15	0.48	1.10
33017	new hampshire	strafford	0.02	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.02	0.01	0.12	0.32

33019	new hampshire	sullivan	0.08	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.00	0.09	0.39
34001	new jersey	atlantic	0.08	0.32	0.05	0.02	0.00	0.01	0.01	0.01	0.00	0.37	0.10	0.00	0.01	0.09	0.01	0.24	1.31
34003	new jersey	bergen	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.36	0.00	0.00	0.41	0.00	0.01	0.77	0.07	0.50	2.23
34005	new jersey	burlington	0.58	1.16	1.89	0.11	0.00	0.05	0.01	0.13	0.00	0.26	0.05	0.00	0.18	0.24	0.02	1.62	6.29
34007	new jersey	camden	0.02	0.06	0.09	0.00	0.00	0.00	0.00	0.01	0.00	0.16	0.10	0.00	0.01	0.00	0.02	0.47	0.95
34009	new jersey	cape may	0.01	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.08	0.29
34011	new jersey	cumberland	0.32	0.36	0.47	0.19	0.00	0.03	0.01	0.44	0.00	0.04	0.11	0.00	0.06	0.02	0.19	0.14	2.36
34013	new jersey	essex	0.09	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.73	5.56	0.11	6.53
34015	new jersey	gloucester	0.47	0.40	0.66	0.23	0.00	0.03	0.01	0.04	0.00	0.18	0.41	0.00	0.33	0.41	0.01	0.63	3.81
34017	new jersey	hudson	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
34019	new jersey	hunterdon	0.84	2.66	0.67	0.26	0.00	0.08	0.00	0.00	0.00	0.00	0.02	0.02	0.50	0.14	0.04	1.97	7.21
34021	new jersey	mercer	0.24	0.60	0.63	0.05	0.00	0.01	0.00	0.02	0.00	0.00	0.01	0.00	0.12	0.25	0.02	0.46	2.41
34023	new jersey	middlesex	0.30	0.27	0.82	0.03	0.00	0.08	0.00	0.03	0.00	0.01	0.06	0.00	0.02	0.16	0.01	0.57	2.35
34025	new jersey	monmouth	0.20	0.64	0.56	0.08	0.00	0.15	0.00	0.05	0.00	0.01	0.04	0.02	0.10	0.01	1.86	3.05	6.77
34027	new jersey	morris	0.14	0.75	0.10	0.06	0.00	0.01	0.00	0.01	0.00	0.00	0.03	0.00	0.03	0.12	0.02	1.02	2.30
34029	new jersey	ocean	0.09	0.11	0.14	0.01	0.00	0.01	0.00	0.05	0.00	0.06	0.01	0.00	0.08	0.07	0.01	0.32	0.94
34031	new jersey	passaic	0.00	0.30	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.03	0.53	0.88
34033	new jersey	salem	2.04	1.07	2.12	0.52	0.00	0.09	0.01	0.37	0.00	0.13	0.77	0.00	0.81	0.03	1.17	1.60	10.73
34035	new jersey	somerset	0.24	0.97	0.24	0.12	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.43	0.06	0.06	0.84	3.03
34037	new jersey	sussex	0.91	1.89	0.05	0.02	0.00	0.05	0.00	0.01	0.00	0.00	0.04	0.00	0.74	0.16	0.05	1.37	5.29
34039	new jersey	union	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.07
34041	new jersey	warren	1.50	0.87	0.44	0.09	0.00	0.04	0.01	0.01	0.00	0.01	0.02	0.00	0.69	0.11	1.38	0.77	5.94
35001	new mexico	bernalillo	0.05	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.00	0.00	0.21	0.72
35003	new mexico	catron	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.01	0.20
35005	new mexico	chaves	0.77	0.87	0.07	0.07	0.00	0.19	0.04	0.00	0.22	0.01	0.00	0.00	4.09	0.00	0.00	0.18	6.50
35006	new mexico	cibola	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.04	0.15
35007	new mexico	colfax	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.02	0.24
35009	new mexico	curry	0.50	0.07	0.04	0.78	0.00	0.27	0.00	0.01	0.00	0.01	0.01	0.00	3.01	0.00	0.00	0.03	4.73
35011	new mexico	de baca	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.01	0.17
35013	new mexico	dona ana	0.59	0.53	0.06	0.04	0.00	0.04	0.23	0.00	0.66	0.01	0.19	0.01	3.00	0.00	0.00	0.09	5.45
35015	new mexico	eddy	0.09	0.10	0.01	0.01	0.00	0.05	0.11	0.00	0.15	0.00	0.00	0.00	0.51	0.00	0.00	0.05	1.09
35017	new mexico	grant	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.02	0.26
35019	new mexico	guadalupe	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.02	0.21
35021	new mexico	harding	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.01	0.17
35023	new mexico	hidalgo	0.02	0.05	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.00	0.00	0.01	0.29
35025	new mexico	lea	0.25	0.07	0.01	0.09	0.00	0.01	0.20	0.03	0.04	0.01	0.00	0.00	0.87	0.00	0.00	0.04	1.62
35027	new mexico	lincoln	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.06	0.26
35028	new mexico	los alamos	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03
35029	new mexico	luna	0.03	0.10	0.01	0.08	0.00	0.01	0.01	0.00	0.00	0.00	0.02	0.00	0.13	0.00	0.00	0.01	0.42
35031	new mexico	mckinley	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.36	0.58
35033	new mexico	mora	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.02	0.16
35035	new mexico	otero	0.02	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.20	0.00	0.00	0.07	0.40
35037	new mexico	quay	0.02	0.04	0.00	0.15	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.02	0.41
35039	new mexico	rio arriba	0.02	0.17	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.07	0.49
35041	new mexico	roosevelt	0.30	0.22	0.03	0.48	0.00	0.46	0.02	0.01	0.01	0.00	0.01	0.00	1.51	0.00	0.00	0.02	3.06
35045	new mexico	san juan	0.03	0.17	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.18	0.58
35047	new mexico	san miguel	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.18	0.00	0.00	0.03	0.29
35043	new mexico	sandoval	0.05	0.26	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.42	0.00	0.00	0.17	0.97
35049	new mexico	santa fe	0.02	0.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.06	0.40
35051	new mexico	sierra	0.02	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.01	0.24
35053	new mexico	socorro	0.07	0.08	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.03	0.64
35055	new mexico	taos	0.01	0.09	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.03	0.29
35057	new mexico	torrance	0.14	0.14	0.01	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.01	0.48	0.00	0.00	0.08	0.93
35059	new mexico	union	0.06	0.46	0.00	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.02	0.96
35061	new mexico	valencia	0.19	0.31	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.22	0.01	0.00	0.13	1.93
36001	new york	albany	0.34	0.83	0.02	0.05	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.68	0.03	0.01	0.36	2.35
36003	new york	allegany	0.63	0.53	0.06	0.04	0.00	0.04	0.00	0.01	0.00	0.00	0.21	0.00	0.89	0.06	0.00	0.18	2.66
36005	new york	bronx	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
36007	new york	broome	0.54	0.53	0.05	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.85	0.08	0.01	0.25	2.35
36009	new york	cattaraugus	1.25	0.85	0.16	0.07	0.00	0.08	0.00	0.01	0.00	0.01	0.00	0.01	1.77	0.02	0.00	0.33	4.58
36011	new york	cayuga	3.56	0.94	1.19	0.22	0.00	0.20	0.01	0.01	0.00	0.00	0.05	0.00	4.31	0.08	0.07	0.23	10.87
36013	new york	chautauqua	1.47	0.75	0.18	0.08	0.00	0.07	0.01	0.00	0.00	0.28	0.01	0.00	1.96	0.03	0.01	0.30	5.15
36015	new york	chemung	0.24	0.32	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.02	0.00	0.17	1.08
36017	new york	chenango	0.94	0.73	0.08	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.31	0.05	0.01	0.22	3.45
36019	new york	clinton	0.59	0.19	0.05	0.02	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.75	0.01	0.19	0.05	1.88
36021	new york	columbia	0.71	0.66	0.14	0.04	0.00	0.04	0.00	0.00	0.00	0.01	0.01	0.00	0.88	0.07	0.06	0.45	3.08
36023	new york	cortland	0.76	0.46	0.10	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.03	0.00	0.16	2.68
36025	new york	delaware	0.66	0.74	0.06	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.05	0.01	0.23	2.92
36027	new york	dutchess	0.55	1.00	0.09	0.06	0.00	0.03	0.00	0.00	0.00	0.01	0.01	0.00	0.75	0.07	0.08	1.23	3.89
36029	new york	erie	2.50	1.65	0.47	0.23	0.00	0.16	0.01	0.04	0.00	0.07	0.08	0.01	2.85	0.11	0.86	0.71	9.74
36031	new york	essex	0.08	0.25	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.05	0.54
36033	new york	franklin	0.49	0.24	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.01	0.00	0.05	1.51
36035	new york	fulton	0.18	0.30	0.02	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.32	0.00	0.20	0.00	0.00	0.11	1.18
36037	new york	genesee	2.66	0.79	0.55	0.40	0.01	0.11	0.01	0.02	0.00	0.00	0.71	0.09	3.22	0.04	0.43	0.21	9.24
36039	new york	greene	0.09	0.36	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.12	0.01	0.60	0.16	1.40
36041	new york	hamilton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
36043	new york	herkimer	0.91	0.69	0.10	0.04	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.18	0.02	0.02	0.18	3.19
36045	new york	jefferson	1.18	0.82	0.16	0.06	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	1.60	0.02	0.23	0.10	4.24
36047	new york	kings	0.14	0.01	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	7.37	0.00	7.54
36049	new york	lewis	1.06	0.37	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.50	0.01	0.00	0.05	3.19
36051	new york	livingston	2.37	0.66	0.70	0.22	0.00	0.11	0.01	0.01	0.00	0.00	0.05	0.09	2.17	0.02	0.00	0.23	6.62
36053	new york	madison	1.76	1.06	0.32	0.10	0.00	0.11	0.01	0.00	0.00	0.00	0.00	0.00	2.22	0.02	0.01	0.25	5.87
36055	new york	monroe	1.11	0.44	0.54	0.42	0.00	0.03	0.00	0.00	0.00	0.03	0.17	0.09	0.41	0.01	0.01	0.56	3.82
36057	new york	montgomery	2.02	1.60	0.24	0.13	0.00	0.12	0.01	0.01	0.00	0.00	0.00	0.00	2.89	0.03	0.01	0.56	7.63
36059	new york	nassau	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.44	0.46
36061	new york	new york	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
36063	new york	niagara	1.65	0.80	0.63	0.22	0.00	0.09	0.00	0.00	0.00	0.09	0.06	0.01	1.30	0.05	0.06	0.26	5.24
36065	new york	oneida	1.80	1.01	0.30	0.09	0.00	0.09	0.01	0.00	0.00	0.00	0.00	0.00	2.01	0.03	0.01	0.29	5.64
36067	new york	onondaga	2.63	0.97	0.58	0.26	0.00	0.15	0.01	0.00	0.00	0.01	0.01	0.00	3.13	0.03	0.82	0.37	8.99
36069	new york	ontario	2.01	0.80	0.76	0.24	0.00	0.13	0.01	0.02	0.00	0.01	0.05	0.03	2.23	0.07	0.01	0.21	6.56
36071	new york	orange	1.15	2.38	0.14	0.13	0.00	0.05	0.01	0.06	0.00	0.04	0.29	0.00	1.40	0.02	1.57	1.37	8.61
36073	new york	orleans	0.89	0.29	0.40	0.15	0.00	0.02	0.00	0.00	0.00	0.08	0.17	0.01	0.29	0.00	0.00	0.13	2.44
36075	new york	oswego	0.29	0.35	0.09	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.33	0.02	0.00	0.15	1.29
36077	new york	otsego	0.90	0.92	0.09	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.02	0.01	0.29	3.43
36079	new york	putnam	0.03	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.17	0.50
36081	new york	queens	0.15	0.01	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.05	0.01	0.01	1.25
36083	new york	rensselaer	0.62	0.43	0.05	0.02	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.76	0.04	0.05	0.31	2.31
36085	new york	richmond	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.05
36087	new york	rockland	0.07	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	2.86	0.07	3.15
36091	new york	saratoga	0.82	0.61	0.07	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.11	0.01	0.65	0.38	3.72
36093	new york	schenectady	0.09	0.24	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01	0.05	0.15	0.64
36095	new york	schoharie	0.58	0.66	0.04	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.73	0.06	0.01	0.21	2.36
36097	new york	schuyler	0.54	0.31	0.08	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.71	0.08	0.02	0.18	2.00
36099	new york	seneca	1.28	0.53	0.76	0.14	0.00	0.11	0.00	0.01	0.00	0.01	0.00	0.00	1.17	0.70	0.08	0.15	4.97
36089	new york	st lawrence	1.05	0.56	0.11	0.05	0.00	0.05	0.00	0.00	0.00	0.01	0.00	0.00	1.50	0.02	0.00	0.15	3.50
36101	new york	steuben	1.97	1.63	0.24	0.09	0.00	0.27	0.01	0.05	0.00	0.01	0.06	0.04	2.57	0.34	0.42	0.41	8.11
36103	new york	suffolk	0.19	0.76	0.02	0.15	0.00	0.10	0.00	0.02	0.00	0.02	0.28	0.01	0.02	0.19	0.19	2.09	4.03
36105	new york	sullivan	0.22	0.40	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.02	0.77	1.22	2.97
36107	new york	tioga	0.56	0.53	0.05	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.79	0.07	0.01	0.24	2.33
36109	new york	tompkins	0.88	0.44	0.13	0.07	0.00	0.06	0.00	0.00	0.00	0.00	0.01	0.00	0.96	0.04	0.00	0.42	3.02
36111	new york	ulster	0.21	0.53	0.04	0.05	0.00	0.01	0.01	0.00	0.00	0.27	0.02	0.00	0.25	0.03	0.01	0.41	1.86
36113	new york	warren	0.01	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.05	0.21
36115	new york	washington	1.39	0.84	0.12	0.06	0.00	0.06	0.01	0.00	0.00	0.00	0.01	0.00	2.16	0.07	0.03	0.25	5.01
36117	new york	wayne	1.19	0.36	0.63	0.12	0.00	0.06	0.00	0.01	0.00	0.13	0.07	0.00	0.78	0.56	0.34	0.15	4.41
36119	new york	westchester	0.03	0.16	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.05	0.03	0.71	1.10
36121	new york	wyoming	3.25	0.82	0.33	0.19	0.00	0.17	0.01	0.00	0.00	0.00	0.15	0.01	4.69	0.02	0.00	0.17	9.81
36123	new york	yates	1.26	0.55	0.27	0.18	0.00	0.11	0.00	0.01	0.00	0.02	0.04	0.02	1.64	0.01	0.02	0.18	4.30
37001	north carolina	alamance	0.29	0.19	0.10	0.09	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.02	1.01	1.03	0.45	0.23	3.48
37003	north carolina	alexander	2.68	0.13	0.27	0.14	0.00	0.11	0.01	0.02	0.00	0.00	0.00	0.00	1.09	0.88	5.73	0.15	11.23
37005	north carolina	alleghany	0.19	0.10	0.01	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.91	0.69	0.15	0.06	2.21
37007	north carolina	anson	1.98	0.37	0.23	0.12	0.00	0.08	0.03	0.07	0.00	0.02	0.00	0.04	0.28	4.08	3.18	0.04	10.54
37009	north carolina	ashe	0.07	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.75	0.00	0.23	1.84
37011	north carolina	avery	0.07	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.80	0.16	0.04	1.17
37013	north carolina	beaufort	1.11	0.04	0.57	0.34	0.00	0.04	0.17	0.54	0.01	0.00	0.04	0.04	0.02	2.23	0.08	0.03	5.26
37015	north carolina	bertie	1.36	0.19	0.30	0.16	0.00	0.06	0.24	0.19	0.10	0.00	0.01	0.04	0.01	0.42	2.19	0.01	5.28
37017	north carolina	bladen	2.74	0.10	0.35	0.14	0.00	0.11	0.05	0.04	0.05	0.01	0.01	0.04	0.06	36.62	0.65	0.04	41.02
37019	north carolina	brunswick	0.51	0.22	0.11	0.04	0.00	0.02	0.01	0.03	0.00	0.01	0.00	0.02	0.05	6.37	0.00	0.06	7.46
37021	north carolina	buncombe	0.16	0.27	0.02	0.06	0.00	0.01	0.00	0.00	0.00	0.02	0.01	0.00	0.84	1.19	0.02	0.47	3.07
37023	north carolina	burke	0.52	0.16	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.93	1.05	0.20	3.12
37025	north carolina	cabarrus	0.72	0.42	0.19	0.11	0.00	0.05	0.01	0.02	0.00	0.00	0.07	0.04	1.33	1.82	0.93	0.50	6.21
37027	north carolina	caldwell	0.20	0.09	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.97	0.39	0.17	2.10
37029	north carolina	camden	1.26	0.07	1.72	0.51	0.00	0.00	0.05	0.24	0.00	0.00	0.17	0.00	0.01	0.00	0.39	0.02	4.46
37031	north carolina	carteret	0.15	0.01	0.28	0.02	0.00	0.01	0.06	0.01	0.00	0.00	0.04	0.00	0.01	0.40	0.00	0.02	1.00
37033	north carolina	caswell	0.12	0.22	0.03	0.03	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.32	0.83	0.47	0.08	2.14
37035	north carolina	catawba	1.62	0.21	0.26	0.14	0.00	0.07	0.01	0.02	0.00	0.00	0.00	0.00	0.86	1.33	2.68	0.45	7.66
37037	north carolina	chatham	1.85	0.42	0.19	0.10	0.00	0.09	0.01	0.01	0.00	0.00	0.00	0.00	1.62	1.12	3.41	0.38	9.20
37039	north carolina	cherokee	0.06	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.11	0.65	0.14	0.08	1.10
37041	north carolina	chowan	0.16	0.05	0.19	0.11	0.00	0.02	0.22	0.07	0.08	0.01	0.01	0.01	0.03	0.20	0.17	0.01	1.34
37043	north carolina	clay	0.07	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.69	0.15	0.03	1.06
37045	north carolina	cleveland	2.24	0.18	0.40	0.23	0.00	0.10	0.03	0.04	0.00	0.01	0.00	0.10	1.16	0.03	3.90	0.38	8.81
37047	north carolina	columbus	1.44	0.09	0.51	0.11	0.00	0.06	0.08	0.27	0.10	0.01	0.09	0.09	0.20	15.07	0.26	0.06	18.43
37049	north carolina	craven	0.49	0.18	0.24	0.07	0.00	0.02	0.07	0.06	0.02	0.00	0.03	0.00	0.04	4.71	0.12	0.06	6.10
37051	north carolina	cumberland	0.95	0.29	0.42	0.10	0.00	0.06	0.16	0.26	0.01	0.01	0.11	0.04	0.09	8.23	0.45	0.14	11.30
37053	north carolina	currituck	0.74	0.28	0.90	0.64	0.00	0.01	0.03	0.23	0.03	0.04	0.00	0.09	0.02	0.00	0.40	0.06	3.49
37055	north carolina	dare	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.10
37057	north carolina	davidson	0.99	0.29	0.25	0.11	0.00	0.05	0.01	0.03	0.00	0.00	0.00	0.05	0.89	1.16	1.45	0.42	5.72
37059	north carolina	davie	0.24	0.21	0.14	0.08	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.64	1.17	0.26	0.31	3.09
37061	north carolina	duplin	7.57	0.31	1.04	0.52	0.00	0.36	0.16	0.27	0.07	0.02	0.05	0.10	0.40	83.11	4.22	0.07	98.27
37063	north carolina	durham	0.12	0.12	0.02	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.15	1.33	0.03	0.25	2.05
37065	north carolina	edgecombe	0.87	0.16	0.61	0.15	0.00	0.04	0.38	0.29	0.11	0.00	0.09	0.06	0.05	6.54	0.68	0.03	10.05
37067	north carolina	forsyth	0.13	0.18	0.07	0.03	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.27	1.50	0.01	0.41	2.65
37069	north carolina	franklin	0.32	0.38	0.21	0.09	0.00	0.02	0.01	0.07	0.00	0.00	0.01	0.02	0.55	2.78	0.35	0.25	5.05
37071	north carolina	gaston	0.36	0.12	0.07	0.04	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.60	0.08	0.52	0.35	2.18
37073	north carolina	gates	0.40	0.12	0.19	0.07	0.00	0.02	0.16	0.02	0.04	0.00	0.00	0.00	0.02	1.64	0.51	0.03	3.22
37075	north carolina	graham	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.13	0.02	0.22
37077	north carolina	granville	0.07	0.32	0.04	0.03	0.00	0.01	0.01	0.04	0.00	0.00	0.01	0.00	0.36	0.13	0.00	0.23	1.24
37079	north carolina	greene	1.72	0.22	0.68	0.23	0.00	0.08	0.32	0.28	0.05	0.01	0.16	0.15	0.05	22.82	0.43	0.03	27.22
37081	north carolina	guilford	0.46	0.50	0.24	0.16	0.00	0.05	0.01	0.08	0.00	0.00	0.01	0.00	1.39	1.69	1.13	0.60	6.31
37083	north carolina	halifax	0.31	0.26	0.36	0.05	0.00	0.03	0.50	0.24	0.10	0.00	0.02	0.03	0.25	3.05	0.16	0.04	5.41
37085	north carolina	harnett	2.17	0.45	0.70	0.22	0.00	0.16	0.41	0.31	0.01	0.01	0.08	0.13	0.25	7.02	2.83	0.20	14.97
37087	north carolina	haywood	0.14	0.18	0.01	0.05	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.46	0.82	0.02	0.24	1.94
37089	north carolina	henderson	0.28	0.12	0.04	0.09	0.00	0.01	0.00	0.02	0.00	0.13	0.01	0.00	0.48	1.06	0.00	0.20	2.45
37091	north carolina	hertford	0.60	0.06	0.16	0.07	0.00	0.03	0.13	0.09	0.06	0.00	0.02	0.01	0.48	0.93	0.00	0.00	2.66
37093	north carolina	hoke	0.84	0.24	0.21	0.08	0.00	0.04	0.16	0.11	0.00	0.00	0.00	0.03	0.04	6.30	0.71	0.09	8.87
37095	north carolina	hyde	0.38	0.00	0.20	0.13	0.00	0.00	0.16	0.13	0.04	0.00	0.03	0.00	0.00	0.20	0.04	0.00	1.23
37097	north carolina	iredell	1.91	0.31	0.46	0.24	0.00	0.10	0.01	0.08	0.00	0.00	0.00	0.00	5.17	1.21	3.08	0.48	13.05
37099	north carolina	jackson	0.05	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.84	0.00	0.05	1.04
37101	north carolina	johnston	2.39	0.71	1.21	0.31	0.00	0.17	0.20	0.51	0.05	0.03	0.26	0.16	0.58	20.91	1.68	0.36	29.53
37103	north carolina	jones	1.67	0.14	0.28	0.10	0.00	0.07	0.22	0.15	0.01	0.00	0.02	0.02	0.03	21.49	0.52	0.03	24.75
37105	north carolina	lee	0.40	0.10	0.12	0.04	0.00	0.03	0.02	0.07	0.00	0.00	0.00	0.02	0.12	0.75	0.71	0.08	2.46
37107	north carolina	lenoir	1.10	0.13	0.31	0.10	0.00	0.06	0.14	0.29	0.01	0.01	0.04	0.12	0.05	13.54	0.28	0.03	16.18
37109	north carolina	lincoln	1.48	0.17	0.29	0.16	0.00	0.08	0.01	0.02	0.00	0.00	0.01	0.03	1.59	0.07	1.95	0.35	6.21
37113	north carolina	macon	0.05	0.06	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.75	0.02	0.09	1.08
37115	north carolina	madison	0.09	0.10	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	1.03	0.03	0.27	1.86
37117	north carolina	martin	0.19	0.13	0.16	0.06	0.00	0.02	0.35	0.15	0.10	0.00	0.01	0.01	0.02	0.41	0.26	0.01	1.86
37111	north carolina	mcdowell	0.20	0.09	0.02	0.04	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.02	0.09	0.87	0.35	0.11	1.81
37119	north carolina	mecklenburg	0.05	0.14	0.02	0.02	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.20	0.00	0.35	0.23	1.03
37121	north carolina	mitchell	0.07	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.83	0.16	0.05	1.21
37123	north carolina	montgomery	1.65	0.25	0.16	0.09	0.00	0.07	0.01	0.02	0.00	0.00	0.00	0.03	0.13	1.60	2.77	0.05	6.84
37125	north carolina	moore	1.81	0.39	0.23	0.10	0.00	0.09	0.02	0.05	0.00	0.00	0.01	0.02	0.32	0.70	3.05	0.23	7.03
37127	north carolina	nash	0.94	0.71	0.57	0.10	0.00	0.05	0.18	0.21	0.04	0.01	0.16	0.02	0.26	5.16	1.47	0.13	10.03
37129	north carolina	new hanover	0.08	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.84	0.18	0.02	1.21
37131	north carolina	northampton	0.91	0.07	0.37	0.11	0.00	0.04	0.15	0.15	0.06	0.00	0.00	0.00	0.39	6.89	0.85	0.03	9.90
37133	north carolina	onslow	0.82	0.08	0.16	0.06	0.00	0.04	0.07	0.12	0.01	0.00	0.02	0.00	0.03	8.62	0.50	0.06	10.60
37135	north carolina	orange	0.17	0.20	0.05	0.04	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.66	0.75	0.41	0.34	2.66
37137	north carolina	pamlico	0.18	0.03	0.16	0.14	0.00	0.00	0.01	0.03	0.00	0.00	0.03	0.01	0.00	0.00	0.06	0.01	0.67
37139	north carolina	pasquotank	1.18	0.03	1.50	0.38	0.00	0.04	0.09	0.14	0.00	0.01	0.12	0.17	0.02	1.07	0.21	0.03	5.01
37141	north carolina	pender	1.28	0.10	0.19	0.07	0.00	0.05	0.01	0.04	0.00	0.02	0.00	0.03	0.04	13.46	0.80	0.04	16.13
37143	north carolina	perquimans	1.56	0.12	1.09	0.59	0.00	0.12	0.24	0.27	0.08	0.24	0.08	0.24	0.04	1.07	1.77	0.04	7.80
37145	north carolina	person	0.11	0.21	0.16	0.13	0.00	0.01	0.00	0.10	0.00	0.00	0.00	0.00	0.25	0.44	0.16	0.10	1.68
37147	north carolina	pitt	1.78	0.32	1.04	0.30	0.00	0.10	0.37	0.39	0.16	0.01	0.06	0.07	0.08	18.72	0.83	0.07	24.30
37149	north carolina	polk	0.01	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.15	0.39
37151	north carolina	randolph	5.20	0.56	0.65	0.36	0.00	0.23	0.02	0.04	0.00	0.00	0.00	0.00	2.94	4.81	8.67	0.63	24.13
37153	north carolina	richmond	2.53	0.43	0.25	0.14	0.00	0.11	0.03	0.03	0.00	0.00	0.00	0.03	0.04	3.79	3.94	0.07	11.40
37155	north carolina	robeson	4.48	0.15	1.07	0.25	0.00	0.17	0.23	0.91	0.08	0.03	0.06	0.19	0.24	25.18	3.52	0.14	36.70
37157	north carolina	rockingham	0.11	0.27	0.06	0.09	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.56	0.30	0.25	0.38	2.14
37159	north carolina	rowan	0.98	0.40	0.44	0.19	0.00	0.06	0.01	0.03	0.00	0.00	0.00	0.03	1.35	1.29	1.10	0.49	6.38
37161	north carolina	rutherford	0.50	0.10	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.92	0.75	0.30	2.96
37163	north carolina	sampson	7.59	0.44	1.23	0.59	0.00	0.37	0.33	0.42	0.07	0.02	0.20	0.09	0.55	95.76	2.78	0.10	110.54
37165	north carolina	scotland	0.65	0.24	0.15	0.07	0.00	0.03	0.07	0.07	0.01	0.00	0.00	0.03	0.03	0.54	1.05	0.05	2.97
37167	north carolina	stanly	0.94	0.20	0.32	0.22	0.00	0.05	0.19	0.04	0.00	0.00	0.00	0.00	0.54	0.54	1.40	0.20	4.66
37169	north carolina	stokes	0.16	0.19	0.04	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.67	1.22	0.73	0.34	3.42
37171	north carolina	surry	2.38	0.27	0.34	0.15	0.00	0.13	0.01	0.05	0.00	0.00	0.00	0.02	0.86	0.85	4.84	0.36	10.25
37173	north carolina	swain	0.05	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.75	0.15	0.01	1.00
37175	north carolina	transylvania	0.07	0.07	0.01	0.03	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.10	0.93	0.00	0.08	1.30
37177	north carolina	tyrrell	0.33	0.01	0.21	0.15	0.00	0.01	0.02	0.11	0.02	0.00	0.03	0.05	0.00	0.27	0.05	0.00	1.26
37179	north carolina	union	9.55	0.29	2.42	1.76	0.00	0.39	0.09	0.49	0.01	0.00	0.01	0.09	1.29	1.35	18.45	0.55	36.75
37181	north carolina	vance	0.06	0.11	0.07	0.03	0.00	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.05	0.72	0.15	0.07	1.31
37183	north carolina	wake	0.13	0.42	0.32	0.11	0.00	0.03	0.03	0.15	0.00	0.00	0.03	0.00	0.22	1.29	0.01	0.55	3.28
37185	north carolina	warren	0.06	0.13	0.06	0.02	0.00	0.01	0.01	0.02	0.00	0.00	0.00	0.01	0.20	0.61	0.12	0.04	1.29
37187	north carolina	washington	0.48	0.11	0.33	0.11	0.00	0.01	0.14	0.21	0.03	0.01	0.02	0.07	0.02	0.34	0.07	0.01	1.94
37189	north carolina	watauga	0.07	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.82	0.02	0.11	1.58
37191	north carolina	wayne	5.50	0.40	1.90	0.38	0.00	0.33	0.43	0.75	0.16	0.02	0.22	0.31	0.38	58.34	2.51	0.21	71.85
37193	north carolina	wilkes	4.19	0.21	0.40	0.24	0.00	0.17	0.02	0.01	0.00	0.00	0.00	0.00	1.43	0.82	7.54	0.26	15.30
37195	north carolina	wilson	0.35	0.28	0.59	0.14	0.00	0.02	0.26	0.29	0.06	0.01	0.16	0.03	0.03	2.48	0.15	0.06	4.88
37197	north carolina	yadkin	1.07	0.19	0.37	0.13	0.00	0.07	0.01	0.06	0.00	0.00	0.00	0.02	1.09	0.94	3.22	0.28	7.46
37199	north carolina	yancey	0.06	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.85	0.00	0.09	1.18
38001	north dakota	adams	0.07	0.22	0.00	0.95	0.00	0.03	0.00	0.13	0.00	0.00	0.00	0.02	0.15	0.00	0.00	0.03	1.60

38003	north dakota	barnes	1.47	0.35	3.31	1.12	0.00	0.08	0.00	0.05	0.00	0.00	0.09	0.35	0.16	0.00	0.00	0.04	7.02
38005	north dakota	benson	0.45	0.81	1.09	1.21	0.00	0.29	0.00	0.16	0.00	0.00	0.00	0.52	0.22	0.00	0.00	0.02	4.79
38007	north dakota	billings	0.01	0.29	0.00	0.31	0.00	0.02	0.00	0.04	0.00	0.00	0.05	0.00	0.21	0.00	0.00	0.01	0.95
38009	north dakota	bottineau	0.08	0.49	0.35	1.96	0.00	0.43	0.00	0.73	0.00	0.00	0.00	0.06	0.07	0.02	0.00	0.02	4.22
38011	north dakota	bowman	0.05	0.36	0.00	0.85	0.00	0.04	0.00	0.13	0.00	0.00	0.00	0.04	0.22	0.02	0.00	0.02	1.72
38013	north dakota	burke	0.01	0.31	0.01	1.19	0.00	0.18	0.00	0.33	0.00	0.00	0.00	0.03	0.08	0.00	0.00	0.00	2.15
38015	north dakota	burleigh	0.44	1.00	0.32	1.41	0.00	0.20	0.00	0.60	0.00	0.00	0.11	0.11	0.71	0.00	0.00	0.10	5.00
38017	north dakota	cass	4.95	0.26	9.15	2.21	0.44	0.22	0.01	0.15	0.00	0.01	0.20	0.61	0.25	0.08	0.00	0.06	18.59
38019	north dakota	cavalier	0.06	0.74	0.51	3.46	0.00	0.56	0.00	2.40	0.00	0.00	0.08	0.48	0.02	0.02	0.00	0.01	8.35
38021	north dakota	dickey	0.94	0.37	1.22	0.20	0.00	0.04	0.00	0.01	0.00	0.00	0.07	0.04	0.38	0.03	0.00	0.02	3.33
38023	north dakota	divide	0.01	0.29	0.01	1.48	0.00	0.04	0.00	0.10	0.00	0.00	0.00	0.20	0.06	0.02	0.00	0.01	2.22
38025	north dakota	dunn	0.07	0.62	0.00	0.87	0.00	0.06	0.00	0.10	0.00	0.00	0.04	0.00	0.42	0.02	0.00	0.04	2.23
38027	north dakota	eddy	0.19	0.33	0.53	0.36	0.00	0.14	0.00	0.04	0.00	0.00	0.07	0.12	0.21	0.00	0.00	0.02	1.99
38029	north dakota	emmons	0.52	0.42	0.32	0.76	0.00	0.08	0.00	0.43	0.00	0.00	0.06	0.02	0.39	0.00	0.00	0.02	3.03
38031	north dakota	foster	0.41	0.18	0.96	0.49	0.00	0.18	0.00	0.04	0.00	0.00	0.08	0.22	0.22	0.03	0.00	0.02	2.83
38033	north dakota	golden valley	0.03	0.23	0.00	0.65	0.00	0.04	0.00	0.07	0.00	0.00	0.00	0.05	0.11	0.02	0.00	0.02	1.21
38035	north dakota	grand forks	1.87	0.84	3.14	3.48	0.56	0.35	0.01	0.32	0.00	0.01	0.22	1.98	0.23	0.38	0.00	0.03	13.42
38037	north dakota	grant	0.20	0.54	0.01	1.04	0.00	0.09	0.00	0.25	0.00	0.00	0.00	0.01	0.46	0.00	0.00	0.03	2.63
38039	north dakota	griggs	0.21	0.43	0.81	0.60	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.17	0.16	0.00	0.00	0.02	2.47
38041	north dakota	hettinger	0.15	0.32	0.00	1.92	0.00	0.06	0.00	0.42	0.00	0.00	0.00	0.01	0.11	0.02	0.00	0.01	3.02
38043	north dakota	kidder	0.21	0.85	0.23	0.40	0.00	0.14	0.00	0.14	0.00	0.00	0.03	0.01	0.59	0.03	0.00	0.03	2.66
38045	north dakota	la moure	0.94	0.29	1.56	0.30	0.01	0.03	0.00	0.01	0.00	0.00	0.08	0.21	0.27	0.03	0.00	0.02	3.73
38047	north dakota	logan	0.29	0.42	0.50	0.34	0.00	0.06	0.00	0.08	0.00	0.00	0.01	0.05	0.44	0.03	0.00	0.01	2.19
38049	north dakota	mchenry	0.12	0.69	0.23	0.85	0.00	0.10	0.00	0.25	0.00	0.00	0.00	0.04	0.63	0.00	0.00	0.05	2.96
38051	north dakota	mcintosh	0.26	0.50	0.69	0.66	0.00	0.04	0.00	0.16	0.00	0.00	0.06	0.01	0.38	0.00	0.00	0.02	2.79
38053	north dakota	mckenzie	0.02	0.53	0.01	1.12	0.03	0.09	0.00	0.06	0.00	0.00	0.04	0.05	0.29	0.02	0.00	0.03	2.29
38055	north dakota	mclean	0.36	0.70	0.31	3.12	0.00	0.23	0.00	1.09	0.00	0.00	0.07	0.38	0.29	0.02	0.00	0.03	6.62
38057	north dakota	mercer	0.09	0.35	0.01	0.53	0.02	0.08	0.00	0.22	0.00	0.00	0.05	0.02	0.23	0.00	0.00	0.02	1.60
38059	north dakota	morton	0.39	1.03	0.05	2.21	0.00	0.24	0.00	0.47	0.00	0.00	0.00	0.02	0.97	0.00	0.00	0.07	5.47
38061	north dakota	mountrail	0.03	0.64	0.02	1.88	0.00	0.14	0.00	0.53	0.00	0.00	0.00	0.18	0.18	0.02	0.00	0.02	3.64
38063	north dakota	nelson	0.20	0.37	0.91	1.17	0.00	0.13	0.00	0.18	0.00	0.00	0.00	0.28	0.07	0.03	0.00	0.01	3.36
38065	north dakota	oliver	0.13	0.35	0.03	0.64	0.00	0.06	0.00	0.15	0.00	0.00	0.07	0.06	0.37	0.00	0.00	0.01	1.88
38067	north dakota	pembina	0.30	0.18	1.14	2.15	0.56	0.09	0.00	0.16	0.00	0.00	0.19	0.79	0.06	0.03	0.00	0.01	5.65
38069	north dakota	pierce	0.17	0.36	0.50	0.89	0.00	0.13	0.00	0.17	0.00	0.00	0.00	0.11	0.18	0.00	0.00	0.01	2.51
38071	north dakota	ramsey	0.40	0.40	0.94	1.67	0.00	0.50	0.00	0.58	0.00	0.00	0.08	0.61	0.03	0.03	0.00	0.01	5.25
38073	north dakota	ransom	0.77	0.37	0.98	0.37	0.00	0.01	0.00	0.00	0.00	0.00	0.08	0.24	0.30	0.03	0.00	0.03	3.19
38075	north dakota	renville	0.03	0.20	0.15	1.69	0.00	0.41	0.00	0.70	0.00	0.00	0.00	0.06	0.04	0.00	0.00	0.01	3.29
38077	north dakota	richland	2.24	0.27	3.23	0.70	0.32	0.01	0.00	0.05	0.00	0.00	0.09	0.21	0.32	0.00	0.00	0.07	7.51
38079	north dakota	rolette	0.05	0.35	0.18	0.85	0.00	0.14	0.00	0.36	0.00	0.00	0.04	0.05	0.11	0.02	0.00	0.04	2.19
38081	north dakota	sargent	0.88	0.20	1.43	0.29	0.00	0.03	0.00	0.01	0.00	0.00	0.09	0.12	0.18	0.03	0.00	0.01	3.27
38083	north dakota	sheridan	0.10	0.32	0.29	0.87	0.00	0.13	0.00	0.24	0.00	0.00	0.00	0.03	0.13	0.00	0.00	0.01	2.12
38085	north dakota	sioux	0.06	0.22	0.01	0.27	0.00	0.02	0.00	0.05	0.00	0.00	0.00	0.00	0.36	0.02	0.00	0.01	1.03
38087	north dakota	slope	0.04	0.25	0.00	0.84	0.00	0.03	0.00	0.09	0.00	0.00	0.00	0.03	0.12	0.02	0.00	0.01	1.42
38089	north dakota	stark	0.13	0.61	0.01	1.85	0.00	0.07	0.00	0.22	0.00	0.00	0.00	0.01	0.32	0.02	0.00	0.03	3.27
38091	north dakota	steele	0.84	0.21	1.44	0.61	0.00	0.06	0.00	0.02	0.00	0.00	0.00	0.66	0.05	0.03	0.00	0.00	3.94
38093	north dakota	stutsman	1.37	0.76	3.37	0.98	0.00	0.13	0.00	0.10	0.00	0.00	0.08	0.27	0.49	0.03	0.00	0.04	7.62
38095	north dakota	towner	0.10	0.68	0.50	1.99	0.00	0.28	0.00	0.92	0.00	0.00	0.07	0.27	0.04	0.02	0.00	0.01	4.90
38097	north dakota	traill	1.38	0.09	2.60	1.27	0.38	0.12	0.00	0.05	0.00	0.00	0.03	0.60	0.06	0.04	0.00	0.03	6.65
38099	north dakota	walsh	0.43	0.37	1.08	2.92	0.50	0.21	0.00	0.30	0.00	0.00	0.35	1.19	0.10	0.03	0.00	0.02	7.50
38101	north dakota	ward	0.15	0.91	0.57	5.03	0.00	0.45	0.00	1.57	0.00	0.00	0.00	0.20	0.38	0.03	0.00	0.04	9.35
38103	north dakota	wells	0.53	0.37	1.28	1.47	0.00	0.20	0.00	0.12	0.00	0.00	0.00	0.45	0.14	0.00	0.00	0.02	4.58
38105	north dakota	williams	0.02	0.46	0.02	2.67	0.01	0.16	0.00	0.16	0.00	0.00	0.05	0.38	0.13	0.00	0.00	0.02	4.10
39001	ohio	adams	0.67	0.38	0.34	0.04	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	1.12	0.22	0.00	0.27	3.09
39003	ohio	allen	10.31	0.19	3.41	1.96	0.00	0.51	0.36	4.61	0.01	0.00	0.12	0.01	0.47	6.72	1.71	0.15	30.53
39005	ohio	ashland	2.79	0.68	1.72	0.38	0.00	0.11	0.01	0.01	0.00	0.00	0.00	0.00	2.19	1.92	0.28	0.46	10.56
39007	ohio	ashtabula	1.14	0.52	0.94	0.08	0.00	0.08	0.00	0.00	0.00	0.01	0.00	0.00	1.24	0.04	0.01	0.34	4.41
39009	ohio	athens	0.20	0.23	0.06	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.01	0.00	0.16	1.05
39011	ohio	auglaize	4.81	0.31	3.24	0.71	0.00	0.13	0.01	0.00	0.00	0.00	0.01	0.00	1.62	8.05	1.57	0.12	20.58
39013	ohio	belmont	0.37	0.65	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.10	0.01	0.00	0.25	2.45
39015	ohio	brown	1.29	0.56	2.46	0.06	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.00	1.19	0.38	0.01	0.32	6.34
39017	ohio	butler	4.01	0.99	3.38	0.33	0.00	0.08	0.01	0.01	0.00	0.00	0.01	0.00	2.47	2.16	0.01	0.92	14.39
39019	ohio	carroll	0.94	0.65	0.34	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.41	0.09	0.05	0.33	3.93
39021	ohio	champaign	4.64	0.43	3.39	0.26	0.00	0.04	0.01	0.02	0.00	0.00	0.03	0.00	0.82	2.29	0.01	0.37	12.32
39023	ohio	clark	5.17	0.47	3.80	0.20	0.00	0.03	0.01	0.02	0.00	0.00	0.03	0.00	1.76	1.31	0.01	0.32	13.12
39025	ohio	clermont	1.52	0.37	2.14	0.05	0.00	0.01	0.00	0.02	0.00	0.00	0.03	0.00	0.52	0.01	0.01	0.48	5.17
39027	ohio	clinton	4.18	0.25	4.22	0.14	0.00	0.05	0.01	0.06	0.00	0.00	0.01	0.00	0.36	2.41	0.00	0.22	11.92
39029	ohio	columbiana	2.77	0.81	0.97	0.30	0.00	0.13	0.01	0.00	0.00	0.00	0.01	0.00	3.55	0.66	0.53	0.59	10.32
39031	ohio	coshocton	2.11	0.59	0.52	0.11	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.60	2.62	1.97	0.40	10.02
39033	ohio	crawford	4.33	0.39	3.74	0.64	0.00	0.07	0.01	0.00	0.00	0.00	0.03	0.00	0.59	5.13	0.15	0.09	15.16
39035	ohio	cuyahoga	0.26	0.11	0.14	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.59	0.01	0.25	1.43
39037	ohio	darke	10.71	0.54	6.18	0.92	0.00	0.24	0.03	0.01	0.00	0.01	0.01	0.00	2.88	18.52	17.34	0.29	57.69
39039	ohio	defiance	2.47	0.24	3.59	0.77	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.82	0.40	0.04	0.08	8.46
39041	ohio	delaware	2.69	0.40	3.26	0.25	0.00	0.05	0.01	0.00	0.00	0.00	0.03	0.00	0.32	2.63	0.00	0.36	10.01
39043	ohio	erie	1.19	0.14	1.05	0.29	0.00	0.09	0.00	0.00	0.00	0.00	0.01	0.00	0.20	0.01	0.01	0.12	3.10
39045	ohio	fairfield	3.63	0.53	2.59	0.34	0.00	0.08	0.01	0.01	0.00	0.03	0.01	0.00	1.03	3.01	0.01	0.40	11.67
39047	ohio	fayette	3.08	0.18	3.50	0.19	0.00	0.01	0.00	0.04	0.00	0.00	0.02	0.00	0.34	0.04	0.00	0.21	7.61
39049	ohio	franklin	1.33	0.25	1.92	0.19	0.00	0.01	0.00	0.05	0.00	0.00	0.04	0.00	0.36	0.13	0.01	0.50	4.79
39051	ohio	fulton	3.43	0.14	2.07	0.55	0.00	0.06	0.01	0.00	0.00	0.00	0.02	0.00	2.25	1.84	0.00	0.12	10.51
39053	ohio	gallia	0.40	0.30	0.09	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.74	0.02	0.00	0.28	1.87
39055	ohio	geauga	0.73	0.53	0.19	0.06	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.97	0.07	0.02	0.81	3.46
39057	ohio	greene	4.40	0.44	3.50	0.21	0.00	0.03	0.01	0.04	0.00	0.00	0.01	0.00	0.47	1.36	0.01	0.35	10.83
39059	ohio	guernsey	0.48	0.53	0.10	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.98	0.17	0.00	0.29	2.60
39061	ohio	hamilton	0.90	0.17	0.53	0.03	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.45	0.01	0.14	0.41	2.69
39063	ohio	hancock	3.87	0.20	4.23	0.94	0.00	0.03	0.01	0.00	0.00	0.00	0.02	0.00	0.31	1.08	1.56	0.15	12.42
39065	ohio	hardin	4.59	0.19	3.35	0.38	0.00	0.12	0.01	0.00	0.00	0.00	0.02	0.00	1.12	5.59	1.42	0.13	16.93
39067	ohio	harrison	0.41	0.61	0.09	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.11	0.05	0.31	2.31
39069	ohio	henry	3.41	0.23	3.23	1.11	0.00	0.11	0.00	0.01	0.00	0.00	0.03	0.00	0.47	0.39	0.04	0.06	9.11
39071	ohio	highland	1.81	0.56	2.75	0.20	0.00	0.03	0.00	0.04	0.00	0.00	0.02	0.00	1.10	0.87	0.01	0.41	7.79
39073	ohio	hocking	0.13	0.11	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.01	0.00	0.12	0.60
39075	ohio	holmes	4.25	1.33	0.90	0.24	0.00	0.29	0.01	0.00	0.00	0.00	0.00	0.00	4.78	0.94	2.14	1.34	16.23
39077	ohio	huron	4.39	0.33	4.01	0.58	0.00	0.11	0.01	0.02	0.00	0.00	0.13	0.00	0.86	1.74	0.37	0.18	12.73
39079	ohio	jackson	0.16	0.18	0.06	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.41	0.02	0.00	0.11	0.96
39081	ohio	jefferson	0.43	0.56	0.05	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.05	0.00	0.23	2.32
39083	ohio	knox	2.60	0.69	1.69	0.19	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	1.32	2.22	0.36	0.81	9.99
39085	ohio	lake	0.18	0.20	0.05	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.01	0.06	0.10	0.74
39087	ohio	lawrence	0.16	0.11	0.04	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.02	0.00	0.24	0.88
39089	ohio	licking	3.43	0.80	2.27	0.25	0.00	0.06	0.01	0.03	0.00	0.01	0.02	0.00	1.80	1.89	1.92	0.64	13.14
39091	ohio	logan	2.92	0.39	2.71	0.22	0.00	0.05	0.00	0.01	0.00	0.00	0.02	0.00	0.68	0.83	0.01	0.23	8.07
39093	ohio	lorain	2.97	0.59	3.08	0.27	0.00	0.10	0.01	0.01	0.00	0.01	0.01	0.00	1.74	2.66	0.01	0.48	11.95
39095	ohio	lucas	1.32	0.44	1.16	0.27	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.00	0.03	1.60	0.00	0.13	5.00
39097	ohio	madison	6.47	0.35	5.96	0.45	0.00	0.05	0.01	0.02	0.00	0.00	0.03	0.00	1.15	2.71	0.01	0.21	17.42
39099	ohio	mahoning	1.97	0.49	0.87	0.21	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	2.14	0.10	0.48	0.42	6.80
39101	ohio	marion	3.88	0.58	3.50	0.39	0.00	0.07	0.01	0.00	0.00	0.00	0.02	0.00	0.61	5.76	0.00	0.19	15.03
39103	ohio	medina	2.07	0.89	2.04	0.21	0.00	0.07	0.00	0.01	0.00	0.00	0.01	0.00	1.68	0.10	0.02	1.02	8.14
39105	ohio	meigs	0.23	0.14	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.01	0.01	0.11	0.99
39107	ohio	mercer	9.73	0.50	4.05	1.05	0.00	0.30	0.03	0.00	0.01	0.00	0.02	0.00	5.15	17.00	13.32	0.20	51.37
39109	ohio	miami	4.88	0.49	4.51	0.40	0.00	0.05	0.01	0.01	0.00	0.00	0.01	0.00	1.31	1.78	0.02	1.39	14.86
39111	ohio	monroe	0.25	0.25	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.59	0.02	0.00	0.17	1.32
39113	ohio	montgomery	3.58	0.48	2.96	0.18	0.00	0.04	0.01	0.01	0.00	0.00	0.01	0.00	1.28	1.31	0.02	0.51	10.38
39115	ohio	morgan	0.29	0.25	0.05	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.02	0.00	0.15	1.32
39117	ohio	morrow	3.00	0.46	2.86	0.49	0.00	0.07	0.01	0.00	0.00	0.00	0.01	0.00	0.81	3.47	0.01	0.40	11.59
39119	ohio	muskingum	1.07	0.71	0.35	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.56	0.66	0.04	0.44	4.92
39121	ohio	noble	0.21	0.31	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.56	0.02	0.00	0.16	1.30
39123	ohio	ottawa	0.87	0.19	1.95	0.18	0.00	0.01	0.00	0.00	0.00	0.01	0.02	0.00	0.12	0.08	0.01	0.07	3.52
39125	ohio	paulding	2.55	0.13	3.11	0.87	0.00	0.04	0.01	0.03	0.00	0.00	0.00	0.00	0.88	1.97	0.00	0.06	9.66
39127	ohio	perry	0.77	0.30	0.49	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.51	0.20	0.00	0.19	2.53
39129	ohio	pickaway	6.19	0.38	5.11	0.56	0.00	0.13	0.01	0.01	0.00	0.00	0.01	0.00	0.77	4.80	0.00	0.24	18.23
39131	ohio	pike	0.48	0.16	0.33	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.02	0.00	0.13	1.47
39133	ohio	portage	1.38	0.56	1.08	0.17	0.00	0.05	0.00	0.00	0.00	0.00	0.01	0.00	1.02	0.08	0.01	0.63	4.99
39135	ohio	preble	8.04	0.54	5.08	0.45	0.00	0.15	0.01	0.01	0.00	0.01	0.01	0.00	1.88	6.74	0.02	0.47	23.42
39137	ohio	putnam	4.48	0.26	5.01	1.46	0.00	0.10	0.01	0.02	0.00	0.00	0.03	0.00	1.05	5.83	1.51	0.08	19.85
39139	ohio	richland	3.21	0.60	1.87	0.36	0.00	0.11	0.01	0.02	0.00	0.00	0.01	0.00	2.08	2.61	0.99	0.36	12.22
39141	ohio	ross	1.85	0.46	1.48	0.21	0.00	0.06	0.00	0.02	0.00	0.00	0.00	0.00	0.76	0.10	0.00	0.21	5.18
39143	ohio	sandusky	3.12	0.22	3.11	0.41	0.00	0.03	0.00	0.00	0.00	0.01	0.04	0.00	0.37	0.50	0.00	0.12	7.94
39145	ohio	scioto	0.45	0.21	0.28	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.45	0.02	1.46	0.21	3.11
39147	ohio	seneca	4.80	0.56	4.63	1.01	0.00	0.08	0.01	0.00	0.00	0.00	0.03	0.00	0.65	3.87	0.01	0.19	15.85
39149	ohio	shelby	5.03	0.35	3.25	0.60	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	2.00	7.75	1.55	0.13	20.81
39151	ohio	stark	5.58	1.43	2.47	0.72	0.00	0.26	0.02	0.00	0.00	0.00	0.01	0.00	4.61	0.73	2.00	0.77	18.59
39153	ohio	summit	0.34	0.16	0.17	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.02	0.02	0.28	1.32
39155	ohio	trumbull	1.56	0.59	1.29	0.13	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00	1.26	0.06	0.10	0.47	5.57
39157	ohio	tuscarawas	1.96	0.61	0.47	0.13	0.00	0.07	0.01	0.00	0.00	0.00	0.01	0.00	2.65	0.25	0.40	0.44	6.99
39159	ohio	union	4.14	0.56	5.30	0.57	0.00	0.05	0.01	0.01	0.00	0.00	0.01	0.00	0.73	2.87	1.92	0.27	16.42
39161	ohio	van wert	3.72	0.09	3.53	0.64	0.00	0.30	0.01	0.01	0.00	0.00	0.02	0.00	0.53	5.19	0.48	0.05	14.57
39163	ohio	vinton	0.11	0.14	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.03	0.06	0.53
39165	ohio	warren	1.47	0.42	2.10	0.14	0.00	0.02	0.00	0.03	0.00	0.00	0.01	0.00	0.44	0.11	0.01	0.63	5.37
39167	ohio	washington	0.56	0.36	0.17	0.04	0.01	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.98	0.09	0.01	0.20	2.44
39169	ohio	wayne	12.55	2.07	3.42	0.93	0.00	0.76	0.04	0.03	0.01	0.00	0.02	0.00	11.40	7.25	6.30	1.16	45.94
39171	ohio	williams	2.02	0.30	2.35	0.66	0.00	0.03	0.00	0.00	0.00	0.00	0.02	0.00	1.02	0.70	0.01	0.14	7.25
39173	ohio	wood	4.69	0.76	5.17	1.30	0.00	0.10	0.01	0.03	0.00	0.01	0.02	0.00	0.56	0.36	2.05	0.26	15.31
39175	ohio	wyandot	3.61	0.36	3.23	0.50	0.00	0.05	0.01	0.00	0.00	0.00	0.02	0.00	0.25	4.34	1.42	0.07	13.87
40001	oklahoma	adair	1.05	0.46	0.10	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.87	0.67	1.60	0.13	4.99
40003	oklahoma	alfalfa	0.13	1.02	0.07	3.27	0.00	0.14	0.00	0.06	0.00	0.00	0.00	0.00	0.87	0.40	0.02	0.03	6.01
40005	oklahoma	atoka	0.13	0.81	0.01	0.01	0.00	0.01	0.00	0.00	0.04	0.02	0.02	0.00	0.74	0.57	0.00	0.10	2.47
40007	oklahoma	beaver	0.14	0.15	0.01	1.67	0.00	0.28	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.27	0.01	0.02	3.27
40009	oklahoma	beckham	0.06	0.17	0.01	0.82	0.00	0.11	0.07	0.00	0.03	0.00	0.00	0.00	0.66	0.01	0.02	0.04	2.00
40011	oklahoma	blaine	0.15	1.01	0.03	2.23	0.00	0.18	0.02	0.07	0.01	0.00	0.00	0.00	1.36	0.47	0.02	0.04	5.58
40013	oklahoma	bryan	0.31	1.46	0.11	0.25	0.00	0.03	0.01	0.00	0.03	0.00	0.00	0.03	2.00	0.02	0.00	0.19	4.42
40015	oklahoma	caddo	0.40	2.76	0.10	2.58	0.00	0.19	0.17	0.11	0.10	0.00	0.00	0.02	2.14	2.32	0.00	0.12	11.01
40017	oklahoma	canadian	0.26	2.82	0.10	5.07	0.00	0.16	0.10	0.04	0.00	0.00	0.00	0.00	2.77	0.13	0.00	0.20	11.67
40019	oklahoma	carter	0.11	1.43	0.01	0.06	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.32	0.01	0.00	0.17	3.14
40021	oklahoma	cherokee	0.23	0.42	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.65	0.69	0.17	0.18	2.40
40023	oklahoma	choctaw	0.32	1.03	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.07	0.63	0.01	0.15	3.28
40025	oklahoma	cimarron	0.16	0.12	0.01	0.95	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.97	0.22	0.01	0.01	2.56
40027	oklahoma	cleveland	0.15	1.26	0.02	0.28	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.16	0.01	0.54	3.19
40029	oklahoma	coal	0.11	0.35	0.01	0.01	0.00	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.60	0.59	0.03	0.05	1.80
40031	oklahoma	comanche	0.14	3.32	0.01	2.03	0.00	0.02	0.10	0.01	0.01	0.00	0.00	0.00	1.73	0.03	0.00	0.15	7.56
40033	oklahoma	cotton	0.18	0.97	0.02	3.64	0.00	0.04	0.06	0.34	0.00	0.00	0.00	0.00	1.38	0.71	0.03	0.04	7.41
40035	oklahoma	craig	0.68	1.85	0.41	0.32	0.00	0.09	0.00	0.01	0.00	0.00	0.00	0.08	2.54	0.62	0.34	0.26	7.22
40037	oklahoma	creek	0.15	2.29	0.02	0.03	0.00	0.01	0.00	0.00	0.11	0.00	0.00	0.03	1.43	0.05	0.03	0.51	4.66
40039	oklahoma	custer	0.12	1.30	0.02	2.06	0.00	0.06	0.05	0.11	0.01	0.00	0.00	0.00	1.03	0.42	0.02	0.04	5.25
40041	oklahoma	delaware	1.49	0.57	0.22	0.11	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	1.48	0.01	2.63	0.19	6.77
40043	oklahoma	dewey	0.09	0.83	0.01	1.36	0.00	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.65	0.39	0.02	0.02	3.44
40045	oklahoma	ellis	0.08	0.20	0.01	0.65	0.00	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.61	0.32	0.02	0.02	1.96
40047	oklahoma	garfield	1.63	3.27	0.30	9.33	0.00	1.00	0.07	0.93	0.00	0.00	0.09	0.02	2.07	0.72	0.00	0.08	19.49
40049	oklahoma	garvin	0.24	2.08	0.08	0.28	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.69	0.02	0.03	0.24	4.69
40051	oklahoma	grady	0.35	1.74	0.06	2.27	0.00	0.15	0.09	0.00	0.00	0.00	0.00	0.00	3.17	0.82	0.00	0.30	8.97
40053	oklahoma	grant	0.28	1.62	0.23	5.52	0.00	0.48	0.02	0.10	0.00	0.00	0.00	0.05	0.66	0.56	0.03	0.04	9.60
40055	oklahoma	greer	0.07	0.07	0.01	1.26	0.00	0.04	0.04	0.01	0.01	0.00	0.00	0.01	0.36	0.40	0.02	0.03	2.31
40057	oklahoma	harmon	0.07	0.06	0.01	1.02	0.00	0.07	0.18	0.04	0.01	0.00	0.00	0.00	0.45	0.34	0.02	0.01	2.27
40059	oklahoma	harper	0.10	0.66	0.01	0.46	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.31	0.01	0.02	2.52
40061	oklahoma	haskell	0.54	0.57	0.06	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.05	0.35	0.56	0.06	3.25
40063	oklahoma	hughes	0.47	1.08	0.06	0.04	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.31	3.91	0.00	0.12	7.01
40065	oklahoma	jackson	0.07	0.06	0.01	3.22	0.00	0.03	0.35	0.20	0.02	0.00	0.00	0.00	0.33	0.40	0.02	0.04	4.74
40067	oklahoma	jefferson	0.19	0.78	0.02	0.43	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.75	0.58	0.03	0.04	3.85
40069	oklahoma	johnston	0.14	0.88	0.01	0.02	0.00	0.02	0.00	0.00	0.04	0.00	0.00	0.00	0.81	0.76	0.04	0.17	2.91
40071	oklahoma	kay	0.60	2.06	0.75	4.15	0.00	0.26	0.04	0.00	0.00	0.00	0.00	0.14	0.72	0.62	0.00	0.13	9.47
40073	oklahoma	kingfisher	0.30	0.97	0.07	4.53	0.00	0.92	0.03	0.05	0.00	0.00	0.00	0.00	2.66	0.71	0.03	0.09	10.38
40075	oklahoma	kiowa	0.11	0.41	0.02	3.47	0.00	0.09	0.06	0.12	0.01	0.00	0.00	0.00	0.87	0.42	0.02	0.03	5.62
40077	oklahoma	latimer	0.10	0.39	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.40	0.55	0.00	0.09	1.55
40079	oklahoma	le flore	1.94	0.83	0.35	0.15	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.08	1.22	0.12	3.07	0.21	8.06
40081	oklahoma	lincoln	0.13	1.35	0.01	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.43	0.03	0.00	0.34	3.35
40083	oklahoma	logan	0.21	1.54	0.07	3.03	0.00	0.47	0.00	0.00	0.00	0.00	0.00	0.00	1.99	0.03	0.00	0.48	7.83
40085	oklahoma	love	0.13	0.63	0.02	0.04	0.00	0.04	0.00	0.00	0.02	0.00	0.00	0.00	0.64	0.70	0.03	0.12	2.37
40093	oklahoma	major	0.15	1.55	0.03	1.45	0.00	0.30	0.00	0.08	0.02	0.00	0.00	0.00	1.03	0.41	0.02	0.04	5.09
40095	oklahoma	marshall	0.12	0.67	0.01	0.06	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.72	0.00	0.10	2.34
40097	oklahoma	mayes	0.51	0.62	0.15	0.26	0.00	0.04	0.00	0.03	0.00	0.00	0.00	0.00	1.61	0.72	0.37	0.19	4.50
40087	oklahoma	mcclain	0.42	1.19	0.12	0.94	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	1.89	1.91	0.00	0.30	6.84
40089	oklahoma	mccurtain	1.00	0.72	0.13	0.08	0.00	0.04	0.00	0.00	0.01	0.00	0.00	0.03	0.75	0.53	1.34	0.17	4.80
40091	oklahoma	mcintosh	0.16	0.91	0.02	0.01	0.00	0.01	0.00	0.00	0.03	0.01	0.00	0.02	0.98	0.74	0.05	0.13	3.06
40099	oklahoma	murray	0.15	0.55	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.67	0.85	0.04	0.07	2.38
40101	oklahoma	muskogee	0.33	1.23	0.24	0.28	0.00	0.03	0.01	0.00	0.00	0.00	0.01	0.02	1.59	0.02	0.12	0.30	4.18
40103	oklahoma	noble	0.29	2.16	0.15	2.32	0.00	0.05	0.00	0.02	0.00	0.00	0.00	0.07	1.16	0.64	0.03	0.07	6.97
40105	oklahoma	nowata	0.23	2.80	0.12	0.13	0.00	0.05	0.00	0.04	0.00	0.00	0.00	0.05	1.55	0.68	0.00	0.10	5.76
40107	oklahoma	okfuskee	0.17	0.88	0.02	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.97	0.84	0.00	0.09	3.01
40109	oklahoma	oklahoma	0.24	0.99	0.11	1.12	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	1.12	0.08	0.00	0.74	4.44
40111	oklahoma	okmulgee	0.18	1.35	0.03	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.99	0.78	0.00	0.21	3.60
40113	oklahoma	osage	0.35	12.55	0.11	0.25	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.21	4.13	0.02	0.00	0.35	18.01
40115	oklahoma	ottawa	0.73	0.61	0.34	0.34	0.00	0.06	0.00	0.00	0.01	0.00	0.00	0.02	0.98	0.01	0.62	0.12	3.83
40117	oklahoma	pawnee	0.14	1.76	0.06	0.19	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.08	0.85	0.69	0.03	0.12	3.92
40119	oklahoma	payne	0.14	1.21	0.06	0.74	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.02	1.26	0.06	0.00	0.27	3.83
40121	oklahoma	pittsburg	0.14	0.98	0.02	0.01	0.00	0.01	0.00	0.00	0.08	0.01	0.00	0.00	1.38	0.01	0.01	0.18	2.84
40123	oklahoma	pontotoc	0.14	0.96	0.02	0.01	0.00	0.01	0.00	0.00	0.04	0.00	0.00	0.00	1.03	0.66	0.00	0.24	3.10
40125	oklahoma	pottawatomie	0.29	0.75	0.06	0.28	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.99	1.29	0.00	0.26	3.95
40127	oklahoma	pushmataha	0.10	0.43	0.01	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.36	0.55	0.03	0.10	1.66
40129	oklahoma	roger mills	0.08	0.25	0.01	0.60	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.64	0.35	0.02	0.03	2.02
40131	oklahoma	rogers	2.13	1.55	0.23	0.75	0.00	0.15	0.10	1.35	0.01	0.00	0.03	0.00	1.98	0.02	0.37	0.33	9.00
40133	oklahoma	seminole	0.15	0.72	0.02	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.56	1.01	0.00	0.14	2.63
40135	oklahoma	sequoyah	0.27	0.27	0.12	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.64	0.08	0.18	2.22
40137	oklahoma	stephens	0.12	1.00	0.01	0.30	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.45	0.01	0.00	0.15	3.05
40139	oklahoma	texas	2.49	0.23	0.17	2.36	0.00	0.48	0.01	0.01	0.00	0.00	0.00	0.00	2.18	17.60	0.01	0.04	25.59
40141	oklahoma	tillman	0.24	0.21	0.03	4.39	0.00	0.14	0.97	0.01	0.01	0.00	0.00	0.03	1.24	0.54	0.03	0.04	7.88
40143	oklahoma	tulsa	0.29	1.71	0.10	0.11	0.00	0.02	0.00	0.00	0.01	0.00	0.00	0.00	0.80	2.08	0.01	0.61	5.74
40145	oklahoma	wagoner	0.16	0.66	0.54	0.45	0.00	0.04	0.00	0.00	0.02	0.00	0.00	0.18	1.00	0.02	0.00	0.19	3.26
40147	oklahoma	washington	0.14	1.88	0.13	0.15	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.10	0.93	0.68	0.00	0.19	4.21
40149	oklahoma	washita	0.14	1.09	0.03	3.29	0.00	0.15	0.14	0.03	0.01	0.00	0.00	0.00	1.20	0.43	0.02	0.03	6.56
40151	oklahoma	woods	0.10	1.73	0.01	1.42	0.00	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.93	0.36	0.02	0.03	4.65
40153	oklahoma	woodward	0.28	1.17	0.02	0.52	0.00	0.04	0.01	0.18	0.00	0.00	0.00	0.00	0.67	0.34	0.02	0.04	3.30
41001	oregon	baker	0.04	0.31	0.01	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.36	0.00	0.00	0.04	0.87

41003	oregon	benton	0.04	0.41	0.00	0.05	0.00	0.01	0.00	0.11	0.00	0.00	0.16	0.00	0.31	0.01	0.01	0.11	1.22
41005	oregon	clackamas	0.28	0.72	0.03	0.12	0.00	0.05	0.00	0.52	0.00	0.03	0.05	0.01	0.79	0.02	1.13	0.76	4.52
41007	oregon	clatsop	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.03	0.10
41009	oregon	columbia	0.03	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.01	0.00	0.12	0.58
41011	oregon	coos	0.03	0.21	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.20	0.00	0.00	0.07	0.52
41013	oregon	crook	0.03	0.17	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.41	0.00	0.00	0.04	0.76
41015	oregon	curry	0.01	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.06	0.20
41017	oregon	deschutes	0.02	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.14	0.51
41019	oregon	douglas	0.05	0.93	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.40	0.00	0.00	0.27	1.69
41021	oregon	gilliam	0.01	0.04	0.00	0.86	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.01	1.05
41023	oregon	grant	0.01	0.09	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.13	0.00	0.00	0.01	0.29
41025	oregon	harney	0.03	0.57	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.08	0.00	0.53	0.00	0.00	0.04	1.29
41027	oregon	hood river	0.09	0.05	0.05	0.01	0.00	0.01	0.02	0.00	0.00	0.33	0.03	0.00	0.03	0.00	0.00	0.03	0.64
41029	oregon	jackson	0.06	0.75	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.21	0.01	0.39	0.24	1.70
41031	oregon	jefferson	0.01	0.39	0.00	0.09	0.00	0.00	0.00	0.01	0.00	0.00	0.11	0.00	0.16	0.00	0.00	0.06	0.83
41033	oregon	josephine	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.08	0.25
41035	oregon	klamath	0.10	1.09	0.03	0.19	0.00	0.13	0.01	0.00	0.00	0.01	0.04	0.00	0.58	0.00	0.00	0.10	2.28
41037	oregon	lake	0.03	0.37	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.02	0.80
41039	oregon	lane	0.06	0.92	0.01	0.02	0.00	0.00	0.00	0.10	0.00	0.00	0.01	0.00	0.34	0.01	0.04	0.43	1.96
41041	oregon	lincoln	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.03	0.19
41043	oregon	linn	0.18	1.31	0.02	0.10	0.00	0.02	0.00	0.26	0.00	0.00	0.03	0.00	0.44	0.01	0.22	0.63	3.22
41045	oregon	malheur	0.26	0.94	0.02	0.29	0.07	0.05	0.00	0.01	0.00	0.01	0.16	0.05	1.39	0.00	0.00	0.08	3.33
41047	oregon	marion	0.39	2.70	0.05	0.49	0.01	0.05	0.01	1.14	0.00	0.04	0.25	0.02	1.88	0.03	0.86	0.50	8.43
41049	oregon	morrow	0.21	0.15	0.02	0.94	0.00	0.04	0.00	0.02	0.00	0.02	0.15	0.01	1.11	0.00	0.00	0.04	2.72
41051	oregon	multnomah	0.04	0.44	0.01	0.05	0.00	0.01	0.00	0.08	0.00	0.00	0.05	0.00	0.13	0.01	0.01	0.20	1.06
41053	oregon	polk	0.08	0.78	0.01	0.20	0.00	0.04	0.00	0.32	0.00	0.02	0.06	0.01	0.49	0.01	0.00	0.17	2.18
41055	oregon	sherman	0.00	0.02	0.00	1.02	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	1.16
41057	oregon	tillamook	0.24	0.08	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.23	0.00	0.00	0.02	1.62
41059	oregon	umatilla	0.13	0.41	0.02	1.77	0.00	0.03	0.01	0.06	0.00	0.02	0.12	0.08	0.40	0.00	0.00	0.08	3.14
41061	oregon	union	0.01	0.25	0.00	0.15	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.03	0.61
41063	oregon	wallowa	0.01	0.68	0.00	0.08	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.12	0.00	0.00	0.02	0.96
41065	oregon	wasco	0.02	0.25	0.00	0.36	0.00	0.02	0.00	0.00	0.00	0.04	0.00	0.00	0.12	0.01	0.00	0.05	0.88
41067	oregon	washington	0.11	0.85	0.02	0.58	0.00	0.13	0.00	0.97	0.01	0.07	0.11	0.01	0.43	0.06	0.01	0.37	3.74
41069	oregon	wheeler	0.01	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.01	0.15
41071	oregon	yamhill	0.54	0.78	0.05	0.27	0.00	0.12	0.00	0.63	0.00	0.07	0.09	0.01	0.84	0.03	0.69	0.32	4.45
42001	pennsylvania	adams	2.75	1.57	1.32	0.42	0.00	0.19	0.02	0.01	0.00	0.21	0.02	0.00	2.38	0.77	0.87	0.94	11.47
42003	pennsylvania	allegheny	0.29	0.71	0.08	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.34	0.04	0.03	0.64	2.19
42005	pennsylvania	armstrong	0.70	0.65	0.16	0.05	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.75	0.02	0.00	0.24	2.63
42007	pennsylvania	beaver	0.78	0.46	0.17	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.69	1.28	0.00	0.50	4.00
42009	pennsylvania	bedford	1.95	0.75	0.22	0.11	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	2.16	0.57	0.37	0.32	6.57
42011	pennsylvania	berks	11.92	2.46	2.67	1.18	0.00	0.68	0.04	0.06	0.01	0.03	0.03	0.05	10.70	9.42	9.42	2.24	50.92
42013	pennsylvania	blair	2.48	0.42	0.28	0.11	0.00	0.11	0.01	0.00	0.00	0.00	0.01	0.00	2.85	0.38	0.01	0.24	6.89
42015	pennsylvania	bradford	2.15	1.93	0.19	0.09	0.00	0.10	0.01	0.00	0.00	0.00	0.01	0.00	2.67	0.61	0.01	0.25	8.00
42017	pennsylvania	bucks	2.92	2.76	1.45	0.45	0.00	0.13	0.01	0.01	0.00	0.00	0.01	0.00	1.63	0.13	0.06	1.98	11.54
42019	pennsylvania	butler	1.25	1.23	0.56	0.12	0.00	0.11	0.00	0.00	0.00	0.00	0.02	0.00	1.11	0.06	0.02	0.55	5.05
42021	pennsylvania	cambria	0.49	0.26	0.10	0.04	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.44	0.04	0.00	0.21	1.63
42023	pennsylvania	cameron	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.06
42025	pennsylvania	carbon	0.17	0.20	0.05	0.02	0.00	0.02	0.00	0.06	0.00	0.00	0.01	0.00	0.12	0.01	0.05	0.09	0.79
42027	pennsylvania	centre	1.53	0.66	0.30	0.13	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.52	0.22	0.01	0.44	4.88
42029	pennsylvania	chester	7.78	2.43	1.62	0.73	0.00	0.35	0.03	0.12	0.01	0.00	0.05	0.00	7.04	4.60	0.56	3.96	29.28
42031	pennsylvania	clarion	0.49	0.44	0.12	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.05	0.00	0.55	0.07	0.00	0.21	2.00
42033	pennsylvania	clearfield	0.30	0.27	0.02	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.01	0.02	0.07	0.97
42035	pennsylvania	clinton	0.70	0.13	0.09	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.99	0.15	0.25	0.14	2.52
42037	pennsylvania	columbia	1.45	0.55	0.47	0.23	0.00	0.06	0.00	0.01	0.00	0.00	0.02	0.00	0.88	1.40	0.66	0.24	5.98
42039	pennsylvania	crawford	1.89	0.92	0.62	0.12	0.00	0.13	0.01	0.02	0.00	0.00	0.02	0.00	1.87	0.07	0.41	0.53	6.58
42041	pennsylvania	cumberland	4.96	1.48	1.17	0.47	0.00	0.37	0.02	0.02	0.00	0.12	0.01	0.00	5.83	1.97	1.57	0.92	18.93
42043	pennsylvania	dauphin	2.60	0.59	0.67	0.29	0.00	0.16	0.01	0.01	0.00	0.00	0.13	0.00	1.66	0.78	3.28	1.24	11.42
42045	pennsylvania	delaware	0.22	0.35	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.04	0.01	0.04	0.21	1.18
42047	pennsylvania	elk	0.07	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01	0.00	0.05	0.27
42049	pennsylvania	erie	0.94	0.64	0.33	0.08	0.00	0.06	0.00	0.02	0.00	0.24	0.05	0.00	0.70	0.04	0.02	0.27	3.41
42051	pennsylvania	fayette	0.50	0.55	0.10	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.03	0.00	0.34	2.27
42053	pennsylvania	forest	0.03	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.04	0.12
42055	pennsylvania	franklin	7.39	1.63	1.18	0.60	0.00	0.50	0.03	0.02	0.01	0.11	0.01	0.00	8.67	4.25	2.96	0.74	28.09
42057	pennsylvania	fulton	1.17	0.44	0.11	0.09	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.98	2.24	0.01	0.17	5.29
42059	pennsylvania	greene	0.17	0.49	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.07	0.00	0.52	0.01	0.00	0.34	1.64
42061	pennsylvania	huntingdon	1.69	0.53	0.19	0.09	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	1.77	0.62	0.20	0.19	5.36
42063	pennsylvania	indiana	1.34	0.61	0.39	0.07	0.00	0.10	0.00	0.00	0.00	0.00	0.01	0.00	1.31	0.06	0.01	0.57	4.50
42065	pennsylvania	jefferson	0.34	0.31	0.08	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.42	0.05	0.01	0.13	1.40
42067	pennsylvania	juniata	1.99	0.33	0.29	0.11	0.00	0.10	0.01	0.00	0.00	0.00	0.00	0.00	1.26	1.74	1.14	0.25	7.20
42069	pennsylvania	lackawanna	0.25	0.32	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.01	0.05	0.14	1.09
42071	pennsylvania	lancaster	42.86	4.85	5.96	2.71	0.00	2.22	0.15	0.64	0.03	0.02	0.08	0.00	37.04	44.31	35.53	7.64	184.03
42073	pennsylvania	lawrence	1.20	0.55	0.45	0.11	0.00	0.08	0.00	0.01	0.00	0.00	0.00	0.00	1.24	0.06	0.00	0.38	4.09
42075	pennsylvania	lebanon	10.50	0.97	1.77	0.75	0.00	0.59	0.04	0.06	0.01	0.00	0.01	0.00	7.12	12.15	8.64	0.75	43.35
42077	pennsylvania	lehigh	1.93	0.88	1.16	0.62	0.00	0.10	0.00	0.01	0.00	0.00	0.03	0.00	0.64	0.37	0.03	0.46	6.24
42079	pennsylvania	luzerne	0.52	0.48	0.07	0.05	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.29	0.02	0.02	0.23	1.74
42081	pennsylvania	lycoming	1.14	0.57	0.24	0.06	0.00	0.05	0.00	0.01	0.00	0.00	0.01	0.00	0.84	1.06	0.03	0.24	4.25
42083	pennsylvania	mckean	0.07	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.01	0.00	0.07	0.37
42085	pennsylvania	mercer	1.89	0.77	0.66	0.11	0.00	0.11	0.01	0.00	0.00	0.00	0.00	0.00	1.67	0.10	0.01	0.56	5.91
42087	pennsylvania	mifflin	1.67	0.33	0.23	0.09	0.00	0.07	0.01	0.00	0.00	0.00	0.05	0.00	1.58	1.63	0.12	0.36	6.12
42089	pennsylvania	monroe	0.28	0.25	0.12	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01	0.01	0.27	1.06
42091	pennsylvania	montgomery	1.18	2.19	0.32	0.11	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.97	0.90	0.11	1.42	7.28
42093	pennsylvania	montour	0.61	0.22	0.24	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.09	0.24	0.54	2.60
42095	pennsylvania	northampton	3.63	1.00	1.51	0.56	0.00	0.19	0.00	0.01	0.00	0.00	0.01	0.00	0.80	0.05	0.01	0.49	8.26
42097	pennsylvania	northumberland	2.66	0.41	0.67	0.20	0.00	0.12	0.01	0.02	0.00	0.00	0.03	0.00	1.76	2.59	1.73	0.39	10.59
42099	pennsylvania	perry	2.95	0.59	0.48	0.22	0.00	0.15	0.01	0.01	0.00	0.01	0.01	0.00	2.25	5.41	0.50	0.39	12.99
42101	pennsylvania	philadelphia	1.18	0.02	0.11	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	5.91	0.00	0.12	7.45
42103	pennsylvania	pike	0.13	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.04	0.04	0.30
42105	pennsylvania	potter	0.35	0.24	0.03	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.47	0.01	0.00	0.10	1.27
42107	pennsylvania	schuylkill	2.51	0.45	0.42	0.20	0.00	0.14	0.01	0.10	0.00	0.00	0.02	0.00	1.12	2.52	4.13	0.73	12.36
42109	pennsylvania	snyder	2.93	0.52	0.52	0.19	0.00	0.14	0.01	0.01	0.00	0.00	0.01	0.00	1.34	3.09	2.43	0.74	11.93
42111	pennsylvania	somerset	2.15	1.10	0.32	0.10	0.00	0.15	0.01	0.00	0.00	0.00	0.00	0.00	2.57	0.72	0.01	0.38	7.54
42113	pennsylvania	sullivan	0.23	0.19	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.23	0.00	0.02	0.06	0.79
42115	pennsylvania	susquehanna	0.99	1.23	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.55	0.03	0.00	0.28	4.27
42117	pennsylvania	tioga	1.14	1.13	0.11	0.05	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	1.17	1.22	0.00	0.22	5.11
42119	pennsylvania	union	2.07	0.23	0.42	0.12	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	1.70	1.45	1.49	0.25	7.82
42121	pennsylvania	venango	0.30	0.23	0.07	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.04	0.00	0.15	1.12
42123	pennsylvania	warren	0.30	0.17	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.02	0.00	0.16	1.05
42125	pennsylvania	washington	1.43	2.72	0.22	0.13	0.00	0.09	0.02	0.23	0.00	0.00	0.01	0.00	2.24	0.07	0.02	1.94	9.13
42127	pennsylvania	wayne	0.65	0.55	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.03	0.03	0.01	0.22	2.62
42129	pennsylvania	westmoreland	1.87	1.13	0.43	0.11	0.00	0.10	0.01	0.00	0.00	0.00	0.02	0.00	2.18	0.10	0.02	0.98	6.94
42131	pennsylvania	wyoming	0.43	0.88	0.03	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.01	0.01	0.19	2.01
42133	pennsylvania	york	8.52	1.98	3.42	1.44	0.00	0.51	0.02	0.14	0.00	0.16	0.02	0.00	3.94	5.59	1.46	1.89	29.09
44001	rhode island	bristol	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.01	0.01	0.05	0.14
44003	rhode island	kent	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.01	0.00	0.17	0.31
44005	rhode island	newport	0.03	0.07	0.00	0.06	0.00	0.00	0.00	0.01	0.00	0.00	0.04	0.00	0.25	0.01	0.01	0.11	0.60
44007	rhode island	providence	0.01	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.18	0.28	0.28	0.59	1.63
44009	rhode island	washington	0.03	0.13	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.01	0.00	0.19	0.01	0.02	0.17	0.58
45001	south carolina	abbeville	0.11	0.21	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.54	0.01	0.11	0.12	1.15
45003	south carolina	aiken	1.58	1.72	0.20	0.11	0.00	0.08	0.07	0.01	0.02	0.03	0.01	0.00	0.30	0.15	2.93	0.45	7.65
45005	south carolina	allendale	0.11	0.56	0.06	0.03	0.00	0.01	0.06	0.01	0.04	0.02	0.00	0.00	0.07	0.00	0.07	0.02	1.06
45007	south carolina	anderson	1.25	0.53	0.15	0.11	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	1.73	0.17	2.13	0.71	6.88
45009	south carolina	bamberg	0.21	1.11	0.04	0.01	0.00	0.01	0.09	0.01	0.06	0.00	0.01	0.00	0.40	0.05	0.07	0.04	2.12
45011	south carolina	barnwell	0.37	1.04	0.06	0.02	0.00	0.02	0.11	0.01	0.05	0.01	0.00	0.00	0.13	0.01	0.72	0.06	2.61
45013	south carolina	beaufort	0.01	0.31	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.01	0.04	0.42
45015	south carolina	berkeley	0.06	0.91	0.03	0.01	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.06	0.10	0.01	0.12	1.33
45017	south carolina	calhoun	0.56	0.39	0.08	0.03	0.00	0.02	0.35	0.03	0.25	0.00	0.02	0.00	0.07	0.89	0.62	0.06	3.38
45019	south carolina	charleston	0.09	0.68	0.02	0.02	0.00	0.01	0.01	0.00	0.00	0.00	0.01	0.00	0.08	0.01	0.12	0.12	1.16
45021	south carolina	cherokee	0.09	0.42	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.32	0.00	0.15	0.14	1.18
45023	south carolina	chester	0.06	0.22	0.01	0.01	0.00	0.00	0.01	0.00	0.00	0.04	0.01	0.00	0.53	0.01	0.23	0.11	1.24
45025	south carolina	chesterfield	0.55	0.82	0.10	0.04	0.00	0.02	0.01	0.03	0.00	0.03	0.00	0.00	0.32	0.07	1.22	0.12	3.34
45027	south carolina	clarendon	1.26	1.19	0.43	0.30	0.00	0.05	0.21	0.10	0.09	0.01	0.01	0.00	0.11	1.74	1.70	0.03	7.23
45029	south carolina	colleton	0.06	1.94	0.04	0.02	0.00	0.03	0.06	0.01	0.04	0.00	0.00	0.00	0.09	0.02	0.01	0.07	2.39
45031	south carolina	darlington	0.55	0.52	0.37	0.13	0.00	0.07	0.37	0.13	0.08	0.01	0.01	0.00	0.16	0.01	0.91	0.07	3.41
45033	south carolina	dillon	1.21	1.54	0.33	0.16	0.00	0.07	0.14	0.16	0.17	0.00	0.00	0.00	0.07	7.19	0.93	0.04	12.02
45035	south carolina	dorchester	0.41	0.79	0.09	0.02	0.00	0.02	0.07	0.01	0.04	0.00	0.00	0.00	0.08	0.63	0.66	0.10	2.95
45037	south carolina	edgefield	0.23	0.23	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.06	0.00	0.00	0.20	0.01	0.71	0.06	1.58
45039	south carolina	fairfield	0.00	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.10	0.00	0.01	0.04	0.41
45041	south carolina	florence	0.21	1.40	0.47	0.10	0.00	0.01	0.21	0.12	0.04	0.00	0.01	0.04	0.09	0.02	0.00	0.10	2.83
45043	south carolina	georgetown	0.06	0.46	0.03	0.00	0.00	0.00	0.02	0.01	0.01	0.00	0.00	0.00	0.01	0.02	0.31	0.02	0.96
45045	south carolina	greenville	0.08	0.38	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.63	0.13	0.01	0.61	1.91
45047	south carolina	greenwood	0.04	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.00	0.42	0.12	0.91
45049	south carolina	hampton	0.07	0.74	0.03	0.01	0.00	0.01	0.14	0.03	0.07	0.00	0.00	0.00	0.02	0.02	0.07	0.02	1.24
45051	south carolina	horry	0.53	1.25	0.43	0.11	0.00	0.03	0.11	0.19	0.19	0.01	0.02	0.00	0.20	3.29	0.19	0.17	6.72
45053	south carolina	jasper	0.03	0.61	0.01	0.00	0.00	0.00	0.01	0.02	0.01	0.00	0.00	0.00	0.02	0.40	0.00	0.01	1.13
45055	south carolina	kershaw	0.10	0.68	0.02	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.10	0.03	0.59	0.14	1.71
45057	south carolina	lancaster	0.26	0.21	0.05	0.02	0.00	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.31	0.01	0.46	0.17	1.52
45059	south carolina	laurens	0.63	0.31	0.06	0.04	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.68	0.01	1.44	0.19	3.39
45061	south carolina	lee	0.59	0.66	0.33	0.17	0.00	0.04	0.15	0.12	0.05	0.02	0.01	0.00	0.07	0.02	0.64	0.05	2.90
45063	south carolina	lexington	3.16	1.05	0.34	0.18	0.00	0.14	0.09	0.00	0.02	0.01	0.01	0.00	0.39	0.07	5.55	0.37	11.39
45067	south carolina	marion	0.17	0.42	0.16	0.03	0.00	0.01	0.05	0.06	0.10	0.01	0.01	0.00	0.13	0.42	0.08	0.05	1.70
45069	south carolina	marlboro	0.42	0.56	0.22	0.08	0.00	0.02	0.23	0.20	0.08	0.01	0.00	0.00	0.05	1.01	0.43	0.02	3.34
45065	south carolina	mccormick	0.08	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.44	0.08	0.01	0.77
45071	south carolina	newberry	1.16	0.21	0.12	0.07	0.00	0.06	0.01	0.03	0.00	0.01	0.00	0.00	1.18	0.53	1.91	0.13	5.40
45073	south carolina	oconee	3.20	0.16	0.31	0.17	0.00	0.13	0.01	0.04	0.00	0.00	0.00	0.00	0.59	0.02	5.55	0.23	10.42
45075	south carolina	orangeburg	1.97	1.46	0.33	0.16	0.00	0.08	0.42	0.07	0.27	0.01	0.02	0.00	0.73	2.17	2.58	0.14	10.40
45077	south carolina	pickens	0.07	0.19	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.08	0.01	0.36	1.25
45079	south carolina	richland	0.08	0.64	0.04	0.04	0.00	0.01	0.02	0.01	0.03	0.00	0.00	0.00	0.08	0.01	0.01	0.18	1.16
45081	south carolina	saluda	1.45	0.20	0.17	0.10	0.00	0.07	0.02	0.00	0.01	0.08	0.02	0.00	0.74	0.46	2.40	0.08	5.79
45083	south carolina	spartanburg	0.22	0.43	0.05	0.03	0.00	0.03	0.00	0.01	0.00	0.08	0.01	0.00	1.01	0.03	0.22	0.54	2.67
45085	south carolina	sumter	0.48	0.79	0.21	0.10	0.00	0.03	0.06	0.09	0.03	0.01	0.01	0.00	0.22	0.04	0.74	0.10	2.91
45087	south carolina	union	0.07	0.12	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.57	0.49	0.06	1.49
45089	south carolina	williamsburg	0.23	1.28	0.28	0.04	0.00	0.01	0.24	0.06	0.05	0.00	0.01	0.01	0.07	0.40	0.08	0.05	2.81
45091	south carolina	york	0.13	0.49	0.04	0.04	0.00	0.01	0.04	0.01	0.00	0.02	0.00	0.00	0.95	0.99	0.09	0.52	3.34
46003	south dakota	aurora	0.63	0.61	0.66	0.20	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.01	0.56	0.07	0.08	0.03	2.89
46005	south dakota	beadle	1.30	0.80	1.22	0.15	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.06	0.97	0.49	0.02	0.05	5.11
46007	south dakota	bennett	0.15	0.36	0.00	0.28	0.00	0.15	0.00	0.06	0.00	0.00	0.00	0.01	0.48	0.00	0.00	0.02	1.51
46009	south dakota	bon homme	0.95	0.53	0.98	0.09	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.04	0.59	0.23	0.00	0.03	3.45
46011	south dakota	brookings	1.72	0.58	1.36	0.14	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.03	1.17	0.67	0.00	0.10	5.85
46013	south dakota	brown	2.62	0.78	2.84	0.29	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.08	0.52	0.15	0.00	0.16	7.48
46015	south dakota	brule	0.71	0.66	0.50	0.38	0.00	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.63	0.26	0.00	0.04	3.29
46017	south dakota	buffalo	0.16	0.18	0.10	0.11	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.01	0.86
46019	south dakota	butte	0.09	0.43	0.01	0.08	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.40	0.00	0.00	0.18	1.21
46021	south dakota	campbell	0.25	0.25	0.20	0.29	0.00	0.01	0.00	0.08	0.00	0.00	0.00	0.00	0.28	0.06	0.06	0.01	1.51
46023	south dakota	charles mix	1.28	0.91	1.32	0.52	0.00	0.05	0.00	0.07	0.00	0.00	0.00	0.00	0.95	0.68	0.08	0.09	5.97
46025	south dakota	clark	1.46	0.53	1.35	0.38	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.02	0.84	0.90	0.00	0.03	5.59
46027	south dakota	clay	1.22	0.20	1.01	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.03	0.14	0.15	0.00	0.05	2.88
46029	south dakota	codington	1.31	0.53	1.06	0.48	0.00	0.05	0.00	0.01	0.00	0.00	0.00	0.04	0.84	0.08	0.00	0.13	4.53
46031	south dakota	corson	0.24	0.51	0.01	0.53	0.00	0.04	0.00	0.19	0.00	0.00	0.00	0.01	0.69	0.05	0.00	0.03	2.31
46033	south dakota	custer	0.02	0.15	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.03	0.38
46035	south dakota	davison	0.83	0.44	0.91	0.12	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.01	0.26	0.33	0.00	0.05	2.96
46037	south dakota	day	0.84	0.35	1.11	0.48	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.03	0.44	0.00	0.00	0.02	3.30
46039	south dakota	deuel	0.75	0.72	0.62	0.14	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.78	0.09	0.10	0.05	3.31
46041	south dakota	dewey	0.13	0.28	0.01	0.26	0.00	0.02	0.00	0.10	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.07	1.30
46043	south dakota	douglas	0.65	0.37	0.66	0.07	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.53	0.31	0.08	0.02	2.73
46045	south dakota	edmunds	0.91	0.48	1.11	0.40	0.00	0.02	0.00	0.05	0.00	0.00	0.00	0.00	0.53	0.27	0.00	0.01	3.78
46047	south dakota	fall river	0.04	0.13	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.37	0.00	0.00	0.03	0.61
46049	south dakota	faulk	0.96	0.54	1.12	0.56	0.00	0.02	0.00	0.08	0.00	0.00	0.00	0.00	0.43	0.39	0.07	0.02	4.18
46051	south dakota	grant	1.22	0.79	1.07	0.35	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.01	1.24	0.10	0.00	0.03	4.87
46053	south dakota	gregory	0.45	0.79	0.29	0.18	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.80	0.00	0.00	0.03	2.60
46055	south dakota	haakon	0.14	0.25	0.01	0.52	0.00	0.10	0.00	0.09	0.00	0.00	0.00	0.00	0.49	0.00	0.00	0.02	1.61
46057	south dakota	hamlin	1.16	0.21	1.04	0.14	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.01	0.62	0.09	0.00	0.03	3.33
46059	south dakota	hand	1.13	0.79	1.17	0.66	0.00	0.05	0.00	0.18	0.00	0.00	0.00	0.02	0.86	0.06	0.00	0.03	4.95
46061	south dakota	hanson	0.86	0.24	0.81	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.44	0.34	0.02	0.02	2.81
46063	south dakota	harding	0.05	0.30	0.00	0.14	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.01	0.41	0.00	0.00	0.13	1.06
46065	south dakota	hughes	0.31	0.17	0.18	0.83	0.00	0.04	0.00	0.28	0.00	0.00	0.00	0.01	0.20	0.06	0.00	0.03	2.11
46067	south dakota	hutchinson	1.67	0.48	1.46	0.10	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.81	1.56	0.02	0.03	6.20
46069	south dakota	hyde	0.33	0.36	0.21	0.49	0.00	0.03	0.00	0.18	0.00	0.00	0.00	0.00	0.47	0.00	0.00	0.03	2.10
46071	south dakota	jackson	0.08	0.31	0.00	0.36	0.00	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.33	0.05	0.00	0.03	1.28
46073	south dakota	jerauld	0.50	0.45	0.28	0.15	0.00	0.03	0.00	0.08	0.00	0.00	0.00	0.00	0.47	0.13	0.00	0.02	2.13
46075	south dakota	jones	0.09	0.28	0.01	0.48	0.00	0.15	0.00	0.14	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.00	1.46
46077	south dakota	kingsbury	1.59	0.56	1.57	0.22	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.99	0.09	0.09	0.05	5.21
46079	south dakota	lake	1.62	0.29	1.13	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.45	0.43	0.00	0.03	4.03
46081	south dakota	lawrence	0.03	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.05	0.00	0.02	0.27
46083	south dakota	lincoln	3.18	0.30	2.49	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.07	0.76	0.73	0.00	0.09	7.70
46085	south dakota	lyman	0.41	0.54	0.06	0.95	0.00	0.52	0.00	0.32	0.00	0.00	0.00	0.02	0.50	0.06	0.00	0.03	3.40
46091	south dakota	marshall	0.61	0.32	0.47	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.02	0.91	0.41	0.00	0.03	2.85
46087	south dakota	mccook	1.58	0.33	1.34	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.53	0.71	0.00	0.05	4.62
46089	south dakota	mcpherson	0.43	0.58	0.42	0.31	0.00	0.03	0.00	0.05	0.00	0.00	0.00	0.01	0.59	0.35	0.01	0.04	2.82
46093	south dakota	meade	0.12	0.99	0.01	0.44	0.00	0.04	0.00	0.08	0.00	0.00	0.00	0.00	1.02	0.00	0.00	0.12	2.82
46095	south dakota	mellette	0.08	0.40	0.01	0.21	0.00	0.06	0.00	0.05	0.00	0.00	0.00	0.00	0.39	0.05	0.00	0.02	1.27
46097	south dakota	miner	0.96	0.59	0.82	0.10	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.02	0.59	0.32	0.00	0.03	3.47
46099	south dakota	minnehaha	3.54	1.02	2.63	0.04	0.00	0.07	0.01	0.00	0.00	0.00	0.01	0.03	1.66	1.25	0.00	0.17	10.42
46101	south dakota	moody	1.60	0.31	0.95	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.56	0.27	0.11	0.04	3.87
46103	south dakota	pennington	0.14	0.43	0.01	0.46	0.00	0.05	0.00	0.07	0.00	0.00	0.00	0.00	0.45	0.00	0.00	0.06	1.69
46105	south dakota	perkins	0.12	0.65	0.01	0.43	0.00	0.07	0.00	0.09	0.00	0.00	0.00	0.01	0.63	0.04	0.00	0.10	2.18
46107	south dakota	potter	0.45	0.16	0.31	0.87	0.00	0.01	0.00	0.32	0.00	0.00	0.00	0.00	0.24	0.06	0.00	0.00	2.43
46109	south dakota	roberts	1.36	0.96	1.53	0.30	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.06	0.63	0.29	0.00	0.07	5.21
46111	south dakota	sanborn	0.44	0.38	0.36	0.04	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.04	0.54	0.08	0.00	0.05	1.95
46113	south dakota	shannon	0.02	0.19	0.00	0.11	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.41
46115	south dakota	spink	2.17	0.80	2.60	0.71	0.00	0.04	0.00	0.03	0.00	0.00	0.00	0.03	0.99	0.32	0.01	0.04	7.75
46117	south dakota	stanley	0.07	0.18	0.01	0.74	0.00	0.08	0.00	0.10	0.00	0.00	0.00	0.00	0.17	0.00	0.00	0.01	1.37
46119	south dakota	sully	0.66	0.17	0.26	1.86	0.00	0.02	0.00	0.68	0.00	0.00	0.00	0.01	0.16	0.06	0.00	0.02	3.91
46121	south dakota	todd	0.10	0.46	0.02	0.08	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.01	0.49	0.00	0.00	0.03	1.25
46123	south dakota	tripp	0.71	1.19	0.33	0.57	0.00	0.25	0.00	0.05	0.00	0.00	0.00	0.00	1.37	0.09	0.00	0.07	4.63
46125	south dakota	turner	2.56	0.49	1.89	0.08	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.02	0.75	0.88	0.00	0.13	6.84
46127	south dakota	union	2.21	0.22	1.64	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.05	0.40	0.35	0.00	0.02	4.95
46129	south dakota	walworth	0.28	0.20	0.25	0.37	0.00	0.01	0.00	0.06	0.00	0.00	0.00	0.00	0.28	0.06	0.00	0.01	1.54
46135	south dakota	yankton	1.16	0.52	1.03	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.04	0.53	0.15	0.00	0.04	3.56

46137	south dakota	ziebach	0.09	0.22	0.01	0.26	0.00	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.37	0.05	0.00	0.04	1.09
47001	tennessee	anderson	0.01	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.02	0.00	0.19	0.70
47003	tennessee	bedford	1.69	0.69	0.45	0.17	0.00	0.08	0.02	0.15	0.00	0.00	0.00	0.02	1.73	0.03	2.41	0.64	8.07
47005	tennessee	benton	0.05	0.25	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.01	0.00	0.06	0.66
47007	tennessee	bledsoe	0.12	0.24	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.65	0.03	0.47	0.14	1.72
47009	tennessee	blount	0.09	0.49	0.12	0.03	0.00	0.01	0.00	0.07	0.00	0.00	0.00	0.00	1.31	0.03	0.01	0.45	2.60
47011	tennessee	bradley	2.36	0.42	0.24	0.13	0.00	0.10	0.01	0.03	0.00	0.00	0.00	0.00	1.10	0.01	4.04	0.22	8.65
47013	tennessee	campbell	0.03	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.04	0.10	0.44
47015	tennessee	cannon	0.13	0.15	0.18	0.01	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.01	0.41	0.01	0.00	0.36	1.31
47017	tennessee	carroll	0.50	0.66	0.30	0.06	0.00	0.00	0.25	0.12	0.00	0.00	0.00	0.01	0.19	0.01	0.00	0.13	2.23
47019	tennessee	carter	0.02	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.01	0.00	0.16	0.72
47021	tennessee	cheatham	0.05	0.16	0.10	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.24	0.01	0.00	0.18	0.78
47023	tennessee	chester	0.04	0.32	0.08	0.01	0.00	0.00	0.04	0.01	0.00	0.00	0.00	0.02	0.11	0.00	0.00	0.05	0.70
47025	tennessee	claiborne	0.03	0.26	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.90	0.05	0.04	0.17	1.46
47027	tennessee	clay	0.34	0.11	0.03	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.46	0.01	0.62	0.11	1.76
47029	tennessee	cocke	0.35	0.20	0.05	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.54	0.02	0.51	0.17	1.89
47031	tennessee	coffee	0.71	0.27	0.53	0.06	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.04	0.84	0.01	0.09	0.24	2.89
47033	tennessee	crockett	0.63	0.43	0.77	0.22	0.00	0.01	0.68	0.35	0.00	0.00	0.00	0.04	0.20	0.00	0.00	0.08	3.40
47035	tennessee	cumberland	0.08	0.22	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.59	0.07	0.02	0.22	1.22
47037	tennessee	davidson	0.12	0.29	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.20	0.01	0.00	0.37	1.02
47041	tennessee	de kalb	0.08	0.18	0.10	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.34	0.05	0.00	0.17	0.94
47039	tennessee	decatur	0.03	0.25	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.19	0.01	0.02	0.06	0.65
47043	tennessee	dickson	0.03	0.39	0.01	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.62	0.03	0.00	0.30	1.41
47045	tennessee	dyer	0.83	0.48	2.28	0.27	0.00	0.16	0.44	0.17	0.00	0.00	0.00	0.49	0.21	0.16	0.00	0.07	5.56
47047	tennessee	fayette	0.48	1.25	0.75	0.13	0.00	0.02	0.52	0.08	0.00	0.00	0.00	0.07	0.41	0.18	0.00	0.19	4.08
47049	tennessee	fentress	0.36	0.17	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.01	0.60	0.16	1.83
47051	tennessee	franklin	0.87	0.29	0.39	0.08	0.00	0.04	0.05	0.13	0.00	0.00	0.00	0.00	0.69	0.48	0.52	0.25	3.81
47053	tennessee	gibson	1.99	0.71	2.09	1.05	0.00	0.02	0.71	0.49	0.00	0.00	0.01	0.12	0.32	0.22	0.02	0.10	7.85
47055	tennessee	giles	0.31	0.50	0.17	0.03	0.00	0.01	0.01	0.05	0.00	0.00	0.00	0.03	1.51	0.01	0.08	0.43	3.13
47057	tennessee	grainger	0.03	0.26	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.81	0.00	0.00	0.24	1.35
47059	tennessee	greene	1.65	0.97	0.16	0.07	0.00	0.06	0.01	0.03	0.00	0.00	0.01	0.00	3.27	0.06	1.90	0.87	9.07
47061	tennessee	grundy	0.16	0.11	0.04	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.03	0.16	0.01	0.24	0.07	0.82
47063	tennessee	hamblen	0.41	0.25	0.09	0.02	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.72	0.02	0.53	0.14	2.23
47065	tennessee	hamilton	0.19	0.44	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.04	0.37	0.37	1.94
47067	tennessee	hancock	0.03	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.00	0.04	0.08	0.64
47069	tennessee	hardeman	0.10	0.70	0.16	0.04	0.00	0.01	0.15	0.01	0.00	0.00	0.00	0.03	0.15	0.01	0.00	0.12	1.50
47071	tennessee	hardin	0.14	0.20	0.29	0.03	0.00	0.01	0.01	0.03	0.00	0.00	0.00	0.01	0.21	0.00	0.02	0.08	1.02
47073	tennessee	hawkins	0.04	0.47	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.96	0.01	0.04	0.36	1.91
47075	tennessee	haywood	0.77	0.55	1.13	0.45	0.00	0.02	1.49	0.39	0.01	0.00	0.01	0.10	0.09	0.00	0.00	0.04	5.07
47077	tennessee	henderson	0.28	0.31	0.42	0.04	0.00	0.00	0.05	0.03	0.00	0.01	0.00	0.00	0.35	0.02	0.00	0.14	1.65
47079	tennessee	henry	0.92	0.59	0.52	0.18	0.00	0.04	0.01	0.07	0.00	0.00	0.00	0.04	0.51	4.25	0.05	0.12	7.30
47081	tennessee	hickman	0.07	0.22	0.07	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.37	0.03	0.00	0.13	0.90
47083	tennessee	houston	0.03	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.01	0.02	0.05	0.46
47085	tennessee	humphreys	0.17	0.17	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.01	0.00	0.09	0.84
47087	tennessee	jackson	0.02	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.03	0.11	0.46
47089	tennessee	jefferson	0.11	0.27	0.04	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	1.37	0.06	0.04	0.29	2.22
47091	tennessee	johnson	0.01	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.09	0.50
47093	tennessee	knox	0.01	0.46	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.80	0.01	0.00	0.62	1.92
47095	tennessee	lake	0.48	0.03	1.11	0.10	0.00	0.03	0.22	0.25	0.05	0.00	0.00	0.06	0.01	0.00	0.00	0.00	2.35
47097	tennessee	lauderdale	0.67	1.05	1.60	0.16	0.00	0.02	0.73	0.19	0.12	0.01	0.00	0.05	0.14	0.01	0.00	0.07	4.84
47099	tennessee	lawrence	0.54	0.36	0.26	0.09	0.00	0.08	0.01	0.01	0.00	0.00	0.00	0.00	1.08	0.10	0.26	0.48	3.30
47101	tennessee	lewis	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.03	0.05	0.23
47103	tennessee	lincoln	1.17	0.57	0.35	0.13	0.00	0.07	0.14	0.08	0.00	0.00	0.05	0.03	1.62	0.03	1.20	0.36	5.81
47105	tennessee	loudon	0.31	0.32	0.13	0.05	0.00	0.01	0.00	0.02	0.00	0.04	0.00	0.00	1.13	0.04	0.00	0.24	2.30
47111	tennessee	macon	0.23	0.29	0.07	0.02	0.00	0.01	0.00	0.19	0.00	0.00	0.00	0.00	0.58	0.23	0.28	0.21	2.11
47113	tennessee	madison	0.52	0.80	0.44	0.15	0.00	0.01	0.35	0.16	0.00	0.00	0.00	0.04	0.16	0.01	0.00	0.11	2.76
47115	tennessee	marion	0.11	0.19	0.05	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.01	0.16	0.10	0.84
47117	tennessee	marshall	0.60	0.47	0.15	0.06	0.00	0.12	0.00	0.02	0.00	0.00	0.00	0.00	1.31	0.05	0.40	0.56	3.74
47119	tennessee	maury	0.35	0.64	0.39	0.08	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	1.72	0.02	0.00	0.49	3.77
47107	tennessee	mcminn	0.46	0.32	0.09	0.03	0.00	0.02	0.00	0.05	0.00	0.00	0.00	0.02	1.10	0.02	0.42	0.21	2.73
47109	tennessee	mcnairy	0.15	0.18	0.25	0.01	0.00	0.00	0.13	0.01	0.00	0.00	0.00	0.04	0.19	0.02	0.02	0.10	1.10
47121	tennessee	meigs	0.06	0.18	0.01	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.42	0.29	0.00	0.07	1.05
47123	tennessee	monroe	0.23	0.28	0.12	0.02	0.00	0.01	0.00	0.06	0.00	0.00	0.00	0.00	1.35	0.01	0.00	0.20	2.29
47125	tennessee	montgomery	0.62	0.43	0.39	0.20	0.00	0.01	0.00	0.23	0.00	0.00	0.00	0.00	0.67	0.03	0.04	0.22	2.87
47127	tennessee	moore	0.21	0.14	0.04	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.57	0.20	0.28	0.08	1.55
47129	tennessee	morgan	0.12	0.10	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.01	0.23	0.09	0.79
47131	tennessee	obion	2.05	0.17	2.07	0.34	0.00	0.03	0.02	0.13	0.01	0.06	0.00	0.01	0.34	0.16	0.33	0.08	5.80
47133	tennessee	overton	0.05	0.57	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.76	0.01	0.00	0.13	1.55
47135	tennessee	perry	0.02	0.11	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.08	0.01	0.00	0.04	0.31
47137	tennessee	pickett	0.15	0.12	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.18	0.22	0.03	0.98
47139	tennessee	polk	0.63	0.04	0.07	0.04	0.00	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.36	0.02	0.91	0.08	2.20
47141	tennessee	putnam	0.04	0.64	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.55	0.02	0.00	0.26	1.54
47143	tennessee	rhea	0.05	0.21	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.27	0.06	0.11	0.10	0.85
47145	tennessee	roane	0.01	0.19	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.01	0.00	0.14	0.70
47147	tennessee	robertson	1.64	0.72	1.42	1.06	0.00	0.06	0.00	0.81	0.00	0.00	0.00	0.06	1.56	0.09	0.01	0.31	7.74
47149	tennessee	rutherford	0.31	0.72	0.42	0.11	0.00	0.02	0.02	0.15	0.00	0.00	0.00	0.02	1.21	0.09	0.03	0.72	3.82
47151	tennessee	scott	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.03	0.05	0.23
47153	tennessee	sequatchie	0.03	0.22	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.02	0.25	0.34	0.00	0.08	0.98
47155	tennessee	sevier	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.37	0.01	0.00	0.20	0.75
47157	tennessee	shelby	0.45	2.82	1.52	0.12	0.00	0.05	0.61	0.21	0.02	0.06	0.01	0.01	0.22	0.00	0.07	0.44	6.60
47159	tennessee	smith	0.04	0.28	0.05	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.62	0.06	0.00	0.27	1.36
47161	tennessee	stewart	0.09	0.21	0.08	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.14	0.01	0.00	0.08	0.63
47163	tennessee	sullivan	0.06	0.31	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.01	0.34	0.01	0.64	2.37
47165	tennessee	sumner	0.69	0.57	0.40	0.19	0.00	0.01	0.00	0.36	0.00	0.00	0.00	0.00	1.75	0.01	0.05	0.60	4.63
47167	tennessee	tipton	0.68	1.05	1.55	0.20	0.00	0.02	0.48	0.11	0.11	0.03	0.00	0.06	0.16	0.03	0.00	0.11	4.62
47169	tennessee	trousdale	0.02	0.14	0.01	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.08	0.60
47171	tennessee	unicoi	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.33	0.00	0.03	0.43
47173	tennessee	union	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.10	0.47
47175	tennessee	van buren	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.03	0.05	0.41
47177	tennessee	warren	0.27	0.32	0.28	0.03	0.00	0.01	0.00	0.09	0.00	0.00	0.00	0.00	0.99	0.02	0.00	0.23	2.23
47179	tennessee	washington	0.25	0.64	0.03	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	2.05	0.03	0.05	0.47	3.56
47181	tennessee	wayne	0.07	0.27	0.04	0.01	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.48	0.19	0.10	0.09	1.27
47183	tennessee	weakley	1.98	0.63	1.53	0.47	0.00	0.03	0.02	0.21	0.00	0.00	0.00	0.03	0.39	1.97	0.65	0.16	8.07
47185	tennessee	white	0.16	0.33	0.05	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	1.08	0.01	0.05	0.23	1.94
47187	tennessee	williamson	0.16	0.80	0.29	0.04	0.00	0.01	0.00	0.13	0.00	0.00	0.00	0.00	1.28	0.01	0.01	0.96	3.67
47189	tennessee	wilson	0.05	1.15	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.63	0.04	0.01	0.97	3.91
48001	texas	anderson	0.52	1.36	0.05	0.06	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.00	1.94	1.19	0.00	0.23	5.39
48003	texas	andrews	0.07	0.08	0.01	0.03	0.00	0.02	0.29	0.00	0.04	0.00	0.00	0.00	0.20	0.53	0.01	0.03	1.31
48005	texas	angelina	0.72	0.30	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.55	0.02	1.43	0.16	3.32
48007	texas	aransas	0.08	0.11	0.01	0.00	0.00	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.08	0.92	0.01	0.01	1.25
48009	texas	archer	0.42	0.95	0.04	1.00	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.00	2.24	0.00	0.01	0.06	4.76
48011	texas	armstrong	0.08	0.12	0.01	0.72	0.00	0.13	0.02	0.00	0.00	0.00	0.00	0.00	0.23	0.43	0.00	0.02	1.76
48013	texas	atascosa	0.88	3.46	0.08	0.12	0.00	0.16	0.10	0.00	0.05	0.00	0.01	0.00	4.28	0.03	0.00	0.23	9.40
48015	texas	austin	0.63	1.23	0.06	0.07	0.00	0.06	0.03	0.00	0.00	0.00	0.00	0.00	3.08	0.01	0.03	0.25	5.45
48017	texas	bailey	0.40	0.16	0.04	0.32	0.00	0.32	0.46	0.03	0.01	0.00	0.00	0.00	2.02	0.00	0.01	0.01	3.78
48019	texas	bandera	0.04	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.01	0.00	0.27	0.60
48021	texas	bastrop	0.59	2.04	0.06	0.04	0.00	0.05	0.01	0.00	0.01	0.00	0.00	0.00	2.66	0.04	0.02	0.36	5.90
48023	texas	baylor	0.19	0.27	0.02	1.46	0.00	0.02	0.04	0.00	0.00	0.00	0.00	0.00	0.81	0.54	0.01	0.02	3.37
48025	texas	bee	0.21	0.99	0.01	0.03	0.00	0.14	0.05	0.00	0.00	0.00	0.00	0.00	0.71	0.00	0.00	0.07	2.21
48027	texas	bell	2.67	4.11	0.05	0.64	0.00	0.67	0.23	0.00	0.01	0.00	0.00	0.00	1.94	0.07	0.02	0.72	11.14
48029	texas	bexar	1.24	3.64	0.09	0.40	0.00	0.49	0.08	0.02	0.00	0.00	0.03	0.00	4.00	0.35	0.02	1.20	11.54
48031	texas	blanco	0.11	0.25	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.01	0.02	0.33	1.22
48033	texas	borden	0.03	0.06	0.00	0.09	0.00	0.00	0.37	0.00	0.00	0.00	0.02	0.00	0.13	0.00	0.00	0.02	0.71
48035	texas	bosque	0.33	1.51	0.03	0.14	0.00	0.09	0.02	0.01	0.00	0.00	0.02	0.00	1.54	0.01	0.00	0.29	4.00
48037	texas	bowie	0.80	1.11	0.13	0.14	0.00	0.05	0.00	0.01	0.01	0.00	0.00	0.00	2.17	0.02	0.47	0.21	5.12
48039	texas	brazoria	2.76	5.47	0.41	0.10	0.00	1.37	0.22	0.04	0.00	0.00	0.01	0.00	9.00	0.86	1.28	1.17	22.69
48041	texas	brazos	0.57	0.65	0.06	0.17	0.00	0.10	0.50	0.06	0.00	0.00	0.00	0.00	2.33	0.04	0.52	0.24	5.25
48043	texas	brewster	0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.23	0.00	0.02	0.47
48045	texas	briscoe	0.04	0.12	0.01	0.64	0.00	0.05	0.25	0.03	0.00	0.00	0.00	0.00	0.20	0.00	0.00	0.01	1.36
48047	texas	brooks	0.14	0.21	0.01	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.57	0.54	0.00	0.02	1.52
48049	texas	brown	0.21	0.52	0.02	0.17	0.00	0.06	0.01	0.00	0.01	0.00	0.00	0.00	1.12	0.02	0.00	0.31	2.46
48051	texas	burleson	0.77	1.64	0.06	0.14	0.00	0.08	0.30	0.02	0.00	0.00	0.02	0.00	2.12	0.01	0.36	0.14	5.65
48053	texas	burnet	0.16	0.30	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.01	0.02	0.43	1.58
48055	texas	caldwell	0.55	1.10	0.04	0.09	0.00	0.13	0.14	0.00	0.00	0.00	0.00	0.00	1.71	0.04	0.65	0.21	4.67
48057	texas	calhoun	0.40	0.69	0.02	0.01	0.00	0.29	0.21	0.00	0.00	0.00	0.00	0.00	0.43	0.82	0.00	0.04	2.92
48059	texas	callahan	0.17	0.34	0.02	0.33	0.00	0.03	0.01	0.00	0.00	0.00	0.01	0.00	0.89	0.01	0.00	0.14	1.94
48061	texas	cameron	0.56	1.44	0.04	0.15	0.33	2.91	2.21	0.07	0.00	0.09	0.04	0.00	0.86	0.05	0.01	0.19	8.95
48063	texas	camp	0.96	0.33	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.85	0.04	1.77	0.05	4.18
48065	texas	carson	0.20	0.19	0.01	1.14	0.00	0.20	0.29	0.01	0.00	0.00	0.02	0.00	0.32	0.01	0.01	0.02	2.41
48067	texas	cass	0.97	0.51	0.09	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.01	1.49	0.12	3.83
48069	texas	castro	2.41	0.27	0.14	1.84	0.00	0.57	0.47	0.04	0.01	0.00	0.01	0.02	8.09	0.00	0.00	0.01	13.89
48071	texas	chambers	0.34	1.45	0.04	0.02	0.00	0.66	0.00	0.01	0.00	0.00	0.00	0.00	1.60	0.04	0.00	0.14	4.32
48073	texas	cherokee	0.65	0.90	0.06	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.61	0.01	0.47	0.17	3.94
48075	texas	childress	0.07	0.02	0.01	0.62	0.00	0.05	0.43	0.01	0.00	0.00	0.00	0.00	0.20	0.45	0.01	0.01	1.88
48077	texas	clay	0.52	1.26	0.05	0.70	0.00	0.03	0.00	0.00	0.01	0.00	0.00	0.00	2.68	0.02	0.00	0.09	5.37
48079	texas	cochran	0.11	0.05	0.01	0.09	0.00	0.08	1.14	0.01	0.08	0.00	0.00	0.01	0.36	0.41	0.00	0.01	2.36
48081	texas	coke	0.04	0.07	0.00	0.07	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.01	0.25	0.66
48083	texas	coleman	0.14	0.32	0.02	0.68	0.00	0.18	0.06	0.00	0.00	0.00	0.00	0.00	0.72	0.00	0.00	0.18	2.31
48085	texas	collin	1.86	1.90	0.13	3.45	0.00	1.17	0.04	0.00	0.01	0.00	0.00	0.00	3.99	0.09	0.02	1.04	13.71
48087	texas	collingsworth	0.07	0.14	0.01	0.35	0.00	0.04	0.60	0.00	0.05	0.00	0.00	0.01	0.35	0.00	0.00	0.01	1.65
48089	texas	colorado	0.58	1.45	0.06	0.06	0.00	0.50	0.05	0.01	0.00	0.00	0.01	0.00	1.97	0.01	0.01	0.09	4.81
48091	texas	comal	0.13	0.32	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.03	0.01	0.36	1.43
48093	texas	comanche	0.64	1.06	0.06	0.11	0.00	0.12	0.05	0.00	0.13	0.00	0.01	0.00	3.66	0.01	0.00	0.29	6.16
48095	texas	concho	0.05	0.06	0.01	0.54	0.00	0.12	0.36	0.01	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.40	1.80
48097	texas	cooke	0.57	2.08	0.06	0.58	0.00	0.23	0.01	0.00	0.00	0.00	0.02	0.00	2.72	0.03	0.01	0.39	6.71
48099	texas	coryell	0.61	2.43	0.05	0.46	0.00	0.20	0.01	0.00	0.00	0.00	0.02	0.00	2.45	0.08	0.05	0.61	6.97
48101	texas	cottle	0.06	0.05	0.01	0.27	0.00	0.04	0.43	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.01	0.01	1.14
48103	texas	crane	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.09
48105	texas	crockett	0.04	0.06	0.00	0.05	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.66	1.04
48107	texas	crosby	0.10	0.09	0.02	0.47	0.00	0.09	4.05	0.03	0.00	0.00	0.01	0.00	0.19	0.61	0.01	0.02	5.69
48109	texas	culberson	0.06	0.05	0.01	0.00	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.19	0.31	0.12	0.00	0.76
48111	texas	dallam	1.17	0.30	0.05	1.27	0.00	0.14	0.03	0.02	0.00	0.00	0.01	0.00	2.68	0.33	0.00	0.02	6.01
48113	texas	dallas	0.66	1.94	0.13	0.58	0.00	0.07	0.03	0.00	0.00	0.00	0.00	0.00	2.48	0.09	0.02	0.75	6.75
48115	texas	dawson	0.02	0.08	0.00	0.04	0.00	0.03	1.99	0.09	0.01	0.00	0.00	0.00	0.04	0.01	0.18	0.01	2.51
48123	texas	de witt	0.50	1.17	0.04	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	2.29	0.01	0.28	0.07	4.43
48117	texas	deaf smith	1.80	0.21	0.14	2.56	0.00	0.68	0.10	0.01	0.00	0.00	0.00	0.00	7.96	0.00	0.01	0.03	13.51
48119	texas	delta	0.39	0.90	0.12	0.31	0.00	0.11	0.14	0.01	0.00	0.00	0.00	0.00	0.69	0.00	0.00	0.06	2.74
48121	texas	denton	0.93	2.29	0.10	1.70	0.00	0.48	0.03	0.00	0.00	0.00	0.00	0.00	4.02	0.05	0.94	1.47	12.02
48125	texas	dickens	0.09	0.10	0.01	0.03	0.00	0.02	0.42	0.00	0.01	0.00	0.00	0.00	0.30	0.44	0.01	0.03	1.45
48127	texas	dimmit	0.07	0.17	0.01	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.25	0.40	0.00	0.02	0.93
48129	texas	donley	0.16	0.13	0.01	0.18	0.00	0.04	0.14	0.00	0.05	0.00	0.00	0.00	0.77	0.00	0.00	0.02	1.50
48131	texas	duval	0.11	0.14	0.01	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.00	0.00	0.04	0.88
48133	texas	eastland	0.21	0.77	0.02	0.11	0.00	0.03	0.04	0.00	0.01	0.00	0.00	0.00	1.08	0.01	0.00	0.14	2.41
48135	texas	ector	0.02	0.02	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.07	0.30
48137	texas	edwards	0.06	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.39	0.01	0.68	1.34
48141	texas	el paso	0.09	0.96	0.01	0.23	0.00	0.02	1.11	0.00	1.15	0.00	0.01	0.00	0.27	0.02	0.00	0.28	4.14
48139	texas	ellis	2.04	1.81	0.14	2.45	0.00	0.85	1.33	0.81	0.00	0.00	0.01	0.00	3.32	0.02	0.01	0.56	13.35
48143	texas	erath	1.13	1.38	0.11	0.10	0.00	0.09	0.02	0.00	0.01	0.00	0.00	0.00	6.47	0.02	0.00	0.43	9.76
48145	texas	falls	2.95	2.78	0.10	1.01	0.00	1.18	0.26	0.00	0.00	0.00	0.00	0.00	3.73	0.01	0.43	0.19	12.64
48147	texas	fannin	0.81	1.79	0.16	0.96	0.00	0.23	0.02	0.08	0.00	0.00	0.00	0.02	2.64	0.03	0.01	0.32	7.06
48149	texas	fayette	0.63	1.56	0.05	0.10	0.00	0.03	0.01	0.00	0.01	0.00	0.00	0.00	2.63	0.06	0.89	0.13	6.10
48151	texas	fisher	0.09	0.41	0.01	0.76	0.00	0.05	1.11	0.01	0.02	0.00	0.00	0.00	0.40	0.00	0.01	0.03	2.91
48153	texas	floyd	0.37	0.15	0.03	1.00	0.00	0.15	1.66	0.06	0.01	0.00	0.01	0.00	1.22	0.00	0.01	0.03	4.70
48155	texas	foard	0.05	0.05	0.00	0.70	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.01	1.07
48157	texas	fort bend	3.06	9.56	0.34	0.50	0.01	3.76	5.04	0.08	0.00	0.00	0.01	0.00	6.13	0.22	0.01	0.84	29.57
48159	texas	franklin	0.81	0.58	0.08	0.14	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.21	0.00	1.14	0.07	4.09
48161	texas	freestone	0.52	1.11	0.05	0.12	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	2.49	0.02	0.02	0.15	4.51
48163	texas	frio	0.26	0.96	0.02	0.12	0.00	0.08	0.06	0.01	0.08	0.01	0.03	0.00	0.85	0.00	0.01	0.03	2.51
48165	texas	gaines	0.03	0.15	0.01	0.13	0.00	0.11	1.54	0.03	0.24	0.00	0.01	0.00	0.12	0.01	0.00	0.01	2.38
48167	texas	galveston	0.22	0.42	0.03	0.07	0.00	0.05	0.05	0.01	0.00	0.00	0.04	0.00	0.94	0.06	0.01	0.27	2.16
48169	texas	garza	0.05	0.08	0.01	0.03	0.00	0.00	0.55	0.00	0.00	0.00	0.00	0.00	0.11	0.47	0.00	0.02	1.32
48171	texas	gillespie	0.23	0.42	0.02	0.15	0.00	0.06	0.00	0.01	0.00	0.01	0.00	0.00	0.98	0.01	0.00	1.00	2.90
48173	texas	glasscock	0.04	0.04	0.01	0.09	0.00	0.01	1.50	0.01	0.01	0.00	0.00	0.00	0.16	0.00	0.00	0.07	1.93
48175	texas	goliad	0.22	0.78	0.02	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.98	0.01	0.00	0.04	2.10
48177	texas	gonzales	3.71	1.52	0.34	0.19	0.00	0.17	0.02	0.00	0.00	0.00	0.01	0.00	4.08	0.03	9.20	0.15	19.44
48179	texas	gray	0.36	0.18	0.03	0.45	0.00	0.09	0.13	0.00	0.00	0.00	0.00	0.00	1.63	0.00	0.00	0.02	2.89
48181	texas	grayson	0.97	1.83	0.05	2.23	0.00	0.29	0.03	0.00	0.00	0.00	0.00	0.00	2.38	0.06	0.01	0.62	8.47
48183	texas	gregg	0.22	0.21	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.41	2.26	0.00	0.14	3.28
48185	texas	grimes	0.85	0.99	0.07	0.04	0.00	0.04	0.06	0.03	0.00	0.00	0.01	0.00	2.49	0.09	0.36	0.25	5.26
48187	texas	guadalupe	0.82	1.75	0.05	0.29	0.00	0.45	0.08	0.00	0.00	0.00	0.00	0.00	2.50	0.16	0.06	0.50	6.67
48189	texas	hale	1.08	0.22	0.06	0.90	0.00	0.47	2.93	0.22	0.00	0.00	0.01	0.00	3.21	0.01	0.00	0.02	9.14
48191	texas	hall	0.05	0.07	0.01	0.24	0.00	0.04	0.81	0.00	0.01	0.00	0.00	0.00	0.25	0.00	0.00	0.01	1.48
48193	texas	hamilton	0.33	0.89	0.04	0.14	0.00	0.18	0.00	0.00	0.00	0.00	0.00	0.00	1.79	0.01	0.00	0.31	3.69
48195	texas	hansford	1.17	0.17	0.08	2.33	0.00	0.13	0.09	0.00	0.00	0.00	0.00	0.00	3.62	0.40	0.01	0.02	8.02
48197	texas	hardeman	0.10	0.03	0.01	1.34	0.00	0.03	0.09	0.02	0.00	0.00	0.00	0.00	0.36	0.51	0.00	0.02	2.52
48199	texas	hardin	0.10	0.29	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.02	0.00	0.11	0.93
48201	texas	harris	1.81	4.29	0.18	0.47	0.00	0.19	0.01	0.40	0.00	0.00	0.00	0.00	7.25	0.16	0.03	1.93	16.73
48203	texas	harrison	0.30	0.47	0.03	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.13	0.03	0.02	0.17	2.19
48205	texas	hartley	1.54	0.28	0.08	0.63	0.00	0.15	0.03	0.01	0.00	0.00	0.04	0.00	4.43	0.00	0.00	0.01	7.20
48207	texas	haskell	0.10	0.55	0.01	2.68	0.00	0.14	0.75	0.10	0.02	0.00	0.00	0.00	0.31	0.52	0.00	0.02	5.20
48209	texas	hays	0.26	0.63	0.02	0.06	0.00	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.82	0.02	0.01	0.58	2.45
48211	texas	hemphill	0.20	0.16	0.02	0.19	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	1.01	0.00	0.00	0.05	1.65
48213	texas	henderson	0.54	1.60	0.06	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	2.50	0.05	0.01	0.28	5.10
48215	texas	hidalgo	1.13	6.07	0.11	0.14	1.14	4.14	1.71	1.18	0.01	0.37	0.38	0.01	2.62	0.05	0.01	0.52	19.59
48217	texas	hill	3.18	4.42	0.07	2.54	0.00	1.21	0.81	0.20	0.00	0.00	0.01	0.00	3.50	0.01	0.01	0.53	16.49
48219	texas	hockley	0.09	0.14	0.01	0.10	0.00	0.13	2.79	0.00	0.01	0.00	0.00	0.00	0.39	0.00	0.00	0.02	3.71
48221	texas	hood	0.20	0.62	0.02	0.10	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	1.02	0.01	0.01	0.24	2.25
48223	texas	hopkins	1.73	1.67	0.16	0.20	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.00	5.48	0.01	1.39	0.21	10.95
48225	texas	houston	0.53	1.14	0.05	0.06	0.00	0.02	0.02	0.00	0.01	0.00	0.00	0.00	1.87	0.46	0.16	0.21	4.52
48227	texas	howard	0.04	0.09	0.01	0.01	0.00	0.00	0.63	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.04	1.04
48229	texas	hudspeth	0.05	0.32	0.01	0.05	0.00	0.02	0.08	0.00	0.00	0.00	0.03	0.00	0.29	0.00	0.00	0.05	0.90
48231	texas	hunt	0.89	2.43	0.37	1.48	0.00	0.42	0.02	0.03	0.01	0.00	0.00	0.06	3.64	0.08	0.02	0.81	10.28
48233	texas	hutchinson	0.35	0.15	0.01	0.72	0.00	0.06	0.07	0.11	0.00	0.00	0.00	0.00	0.30	0.00	0.01	0.02	1.80
48235	texas	irion	0.04	0.01	0.00	0.03	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.01	0.24	0.53
48237	texas	jack	0.23	0.62	0.02	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.18	0.00	0.00	0.09	2.21
48239	texas	jackson	1.25	1.39	0.06	0.08	0.00	0.27	0.32	0.00	0.00	0.00	0.00	0.01	1.12	0.71	0.00	0.04	5.25
48241	texas	jasper	0.10	0.25	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.39	0.02	0.00	0.10	0.88
48243	texas	jeff davis	0.05	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.28	0.00	0.01	0.51

48245	texas	jefferson	0.60	2.05	0.07	0.03	0.00	1.05	0.02	0.04	0.00	0.00	0.00	0.00	2.89	0.02	0.00	0.21	6.97
48247	texas	jim hogg	0.11	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.46	0.17	0.01	1.26
48249	texas	jim wells	0.25	0.57	0.02	0.05	0.00	0.43	0.15	0.06	0.00	0.00	0.00	0.00	0.81	0.00	0.00	0.05	2.41
48251	texas	johnson	1.23	2.39	0.11	0.73	0.00	0.46	0.02	0.00	0.01	0.00	0.00	0.00	5.93	0.12	0.01	1.03	12.05
48253	texas	jones	0.14	0.84	0.02	4.15	0.00	0.15	1.75	0.23	0.00	0.00	0.01	0.00	0.64	0.00	0.00	0.10	8.04
48255	texas	karnes	0.40	1.19	0.03	0.11	0.00	0.09	0.07	0.00	0.00	0.00	0.00	0.00	1.45	0.01	0.00	0.08	3.42
48257	texas	kaufman	0.99	2.41	0.27	0.76	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.07	4.54	0.07	0.02	0.69	9.97
48259	texas	kendall	0.12	0.25	0.01	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.52	0.03	0.01	0.64	1.67
48261	texas	kenedy	0.13	0.85	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.68	0.00	0.00	0.01	1.69
48263	texas	kent	0.07	0.06	0.01	0.02	0.00	0.02	0.19	0.00	0.00	0.00	0.00	0.00	0.20	0.46	0.00	0.02	1.04
48265	texas	kerr	0.05	0.09	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.02	0.00	0.35	0.79
48267	texas	kimble	0.04	0.05	0.00	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.01	0.47	0.79
48269	texas	king	0.05	0.04	0.00	0.29	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.13	0.47	0.00	0.04	1.07
48271	texas	kinney	0.05	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.36	0.13	0.21	0.95
48273	texas	kleberg	0.19	1.61	0.02	0.05	0.00	0.41	0.42	0.00	0.00	0.00	0.01	0.00	0.96	0.01	0.00	0.03	3.70
48275	texas	knox	0.12	0.02	0.01	1.87	0.00	0.01	0.16	0.02	0.01	0.00	0.00	0.00	0.61	0.00	0.01	0.02	2.86
48283	texas	la salle	0.06	0.43	0.01	0.01	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.02	0.81
48277	texas	lamar	0.83	2.00	0.22	0.57	0.00	0.18	0.03	0.02	0.00	0.00	0.02	0.05	2.67	0.01	0.02	0.18	6.79
48279	texas	lamb	1.18	0.27	0.08	0.62	0.00	0.43	1.98	0.02	0.01	0.00	0.01	0.01	4.13	0.01	0.01	0.01	8.77
48281	texas	lampasas	0.13	0.45	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.63	0.01	0.00	0.31	1.58
48285	texas	lavaca	0.54	1.09	0.05	0.05	0.00	0.05	0.00	0.00	0.01	0.00	0.00	0.00	2.47	0.03	0.33	0.07	4.70
48287	texas	lee	0.40	0.90	0.04	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	2.06	0.05	0.00	0.11	3.61
48289	texas	leon	1.87	1.40	0.18	0.11	0.00	0.09	0.02	0.00	0.00	0.00	0.00	0.00	2.56	0.01	2.44	0.18	8.87
48291	texas	liberty	0.64	1.88	0.12	0.04	0.00	0.18	0.00	0.03	0.00	0.00	0.01	0.00	2.63	0.07	0.01	0.33	5.96
48293	texas	limestone	0.70	2.12	0.06	0.15	0.00	0.07	0.04	0.00	0.00	0.00	0.01	0.00	2.92	0.01	0.39	0.14	6.63
48295	texas	lipscomb	0.17	0.19	0.01	0.42	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.55	0.40	0.01	0.01	1.82
48297	texas	live oak	0.18	0.79	0.01	0.01	0.00	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.76	0.00	0.00	0.05	1.86
48299	texas	llano	0.18	0.09	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.78	0.00	0.01	0.08	1.21
48301	texas	loving	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.03
48303	texas	lubbock	0.30	0.14	0.05	0.30	0.00	0.40	7.32	0.02	0.02	0.01	0.03	0.00	1.23	0.04	0.40	0.14	10.39
48305	texas	lynn	0.04	0.11	0.01	0.06	0.00	0.14	4.94	0.06	0.01	0.00	0.01	0.00	0.09	0.00	0.00	0.03	5.50
48313	texas	madison	0.58	0.83	0.05	0.03	0.00	0.03	0.00	0.00	0.02	0.00	0.01	0.03	1.64	0.17	0.75	0.11	4.26
48315	texas	marion	0.03	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.02	0.03	0.35
48317	texas	martin	0.02	0.04	0.00	0.02	0.00	0.00	0.67	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.02	0.86
48319	texas	mason	0.14	0.16	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.67	0.00	0.01	0.14	1.18
48321	texas	matagorda	0.68	1.83	0.07	0.03	0.00	0.31	0.41	0.07	0.00	0.00	0.00	0.00	1.59	0.00	0.01	0.06	5.05
48323	texas	maverick	0.07	0.13	0.01	0.04	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.06	0.71
48307	texas	mcculloch	0.09	0.19	0.01	0.29	0.00	0.02	0.07	0.01	0.00	0.00	0.00	0.00	0.46	0.00	0.00	0.30	1.44
48309	texas	mclennan	3.19	4.24	0.11	2.24	0.00	0.92	0.30	0.01	0.01	0.00	0.01	0.00	4.62	0.04	0.76	0.72	17.16
48311	texas	mcmullen	0.09	0.20	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.47	0.00	0.01	1.12
48325	texas	medina	0.56	0.97	0.02	0.38	0.00	0.22	0.30	0.03	0.01	0.00	0.00	0.00	0.97	0.04	0.00	0.17	3.69
48327	texas	menard	0.05	0.02	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.42	0.78
48329	texas	midland	0.08	0.08	0.01	0.05	0.00	0.01	0.32	0.00	0.01	0.00	0.00	0.00	0.42	0.02	0.00	0.15	1.16
48331	texas	milam	1.25	2.83	0.05	0.41	0.00	0.41	0.24	0.00	0.03	0.00	0.00	0.00	2.33	0.04	0.77	0.19	8.54
48333	texas	mills	0.17	0.44	0.02	0.07	0.00	0.05	0.00	0.00	0.01	0.00	0.00	0.00	0.89	0.00	0.00	0.81	2.47
48335	texas	mitchell	0.08	0.19	0.01	0.13	0.00	0.01	0.43	0.00	0.00	0.00	0.00	0.00	0.38	0.01	0.00	0.06	1.30
48337	texas	montague	0.43	1.06	0.04	0.19	0.00	0.03	0.01	0.00	0.00	0.00	0.02	0.00	2.18	0.03	0.00	0.18	4.17
48339	texas	montgomery	0.40	0.61	0.04	0.04	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	1.59	0.08	0.01	0.95	3.78
48341	texas	moore	0.85	0.19	0.05	1.03	0.00	0.37	0.11	0.03	0.00	0.00	0.00	0.00	2.31	0.00	0.15	0.01	5.09
48343	texas	morris	0.66	0.28	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.78	0.01	0.85	0.09	2.79
48345	texas	motley	0.05	0.07	0.01	0.01	0.00	0.01	0.25	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.01	0.69
48347	texas	nacogdoches	3.69	0.68	0.34	0.18	0.00	0.15	0.02	0.00	0.00	0.00	0.00	0.00	1.30	0.00	6.02	0.13	12.52
48349	texas	navarro	0.87	2.87	0.07	0.22	0.00	0.28	0.50	0.03	0.00	0.00	0.02	0.02	3.49	0.02	0.01	0.30	8.71
48351	texas	newton	0.04	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.05	0.41
48353	texas	nolan	0.06	0.06	0.01	0.25	0.00	0.02	0.74	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.00	0.04	1.44
48355	texas	nueces	0.53	1.51	0.03	0.12	0.00	3.98	4.26	0.03	0.00	0.00	0.01	0.00	0.79	0.03	0.00	0.12	11.42
48357	texas	ochiltree	0.44	0.20	0.04	2.31	0.00	0.27	0.08	0.00	0.00	0.00	0.00	0.00	1.28	0.39	0.00	0.02	5.04
48359	texas	oldham	0.18	0.12	0.02	0.24	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.79	0.38	0.00	0.01	1.82
48361	texas	orange	0.13	0.21	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.59	0.01	0.01	0.16	1.15
48363	texas	palo pinto	0.27	1.02	0.03	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.22	0.02	0.00	0.16	2.79
48365	texas	panola	1.14	0.49	0.11	0.07	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	1.08	0.02	1.49	0.11	4.56
48367	texas	parker	0.64	1.64	0.06	0.05	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	3.31	0.04	0.01	1.08	6.88
48369	texas	parmer	1.84	0.18	0.13	1.96	0.00	0.66	0.32	0.08	0.00	0.00	0.01	0.01	6.89	0.39	0.15	0.02	12.65
48371	texas	pecos	0.07	0.09	0.01	0.01	0.00	0.01	0.05	0.00	0.01	0.00	0.00	0.00	0.39	0.00	0.00	0.35	0.99
48373	texas	polk	0.13	0.40	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.02	0.03	0.09	1.17
48375	texas	potter	0.19	0.21	0.02	0.33	0.00	0.04	0.01	0.01	0.01	0.00	0.02	0.00	0.96	0.10	0.00	0.05	1.94
48377	texas	presidio	0.04	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.27	0.00	0.01	0.50
48379	texas	rains	0.18	0.43	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.89	0.00	0.02	0.07	1.64
48381	texas	randall	1.04	0.24	0.10	1.84	0.00	0.40	0.04	0.01	0.00	0.00	0.02	0.00	5.36	0.01	0.00	0.12	9.17
48383	texas	reagan	0.03	0.01	0.00	0.11	0.00	0.00	0.34	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.12	0.74
48385	texas	real	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.41	0.00	0.07	0.57
48387	texas	red river	0.64	1.16	0.16	0.31	0.00	0.06	0.01	0.01	0.00	0.00	0.00	0.00	1.91	0.01	0.27	0.16	4.70
48389	texas	reeves	0.06	0.06	0.01	0.04	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.30	0.00	0.00	0.01	0.50
48391	texas	refugio	0.14	0.71	0.01	0.04	0.00	0.32	0.33	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.01	0.03	2.10
48393	texas	roberts	0.07	0.08	0.01	0.25	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.29	0.00	0.00	0.01	0.74
48395	texas	robertson	1.32	1.55	0.13	0.13	0.00	0.13	0.22	0.00	0.00	0.00	0.01	0.00	2.64	0.05	1.58	0.14	7.90
48397	texas	rockwall	0.12	0.28	0.01	0.27	0.00	0.07	0.00	0.00	0.00	0.00	0.04	0.00	0.59	0.01	0.00	0.16	1.56
48399	texas	runnels	0.11	0.12	0.01	1.37	0.00	0.10	0.48	0.00	0.00	0.00	0.00	0.00	0.57	0.00	0.00	0.18	2.96
48401	texas	rusk	1.00	0.73	0.09	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.38	0.01	1.60	0.15	5.07
48403	texas	sabine	0.20	0.12	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.27	0.02	0.84
48405	texas	san augustine	0.74	0.16	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.00	1.47	0.03	2.87
48407	texas	san jacinto	0.18	0.28	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.69	0.01	0.00	0.15	1.35
48409	texas	san patricio	0.51	1.19	0.02	0.01	0.00	2.19	3.06	0.00	0.00	0.00	0.02	0.00	0.68	0.01	0.00	0.06	7.74
48411	texas	san saba	0.18	0.52	0.02	0.18	0.00	0.02	0.01	0.00	0.02	0.00	0.01	0.00	0.85	0.00	0.00	0.18	1.99
48413	texas	schleicher	0.07	0.01	0.01	0.02	0.00	0.01	0.10	0.00	0.00	0.00	0.00	0.00	0.24	0.49	0.00	0.62	1.57
48415	texas	scurry	0.08	0.13	0.01	0.10	0.00	0.02	0.84	0.00	0.00	0.00	0.01	0.00	0.38	0.00	0.00	0.04	1.60
48417	texas	shackelford	0.15	0.35	0.01	0.11	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.56	0.58	0.01	0.02	1.82
48419	texas	shelby	4.70	0.49	0.44	0.23	0.00	0.20	0.02	0.00	0.00	0.00	0.00	0.00	0.98	0.01	8.50	0.08	15.65
48421	texas	sherman	0.94	0.20	0.04	0.94	0.00	0.21	0.12	0.01	0.00	0.00	0.00	0.00	2.04	0.00	0.00	0.01	4.52
48423	texas	smith	0.52	1.28	0.05	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	2.27	0.04	0.01	0.50	4.77
48425	texas	somervell	0.07	0.17	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.36	0.00	0.00	0.08	0.77
48427	texas	starr	0.63	1.28	0.06	0.03	0.04	0.58	0.08	0.12	0.00	0.00	0.01	0.00	3.01	0.02	0.00	0.11	5.96
48429	texas	stephens	0.09	0.49	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.48	0.01	0.00	0.05	1.18
48431	texas	sterling	0.04	0.03	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.19	0.52
48433	texas	stonewall	0.08	0.17	0.01	0.28	0.00	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.30	0.47	0.00	0.04	1.42
48435	texas	sutton	0.04	0.01	0.00	0.07	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.22	0.00	0.00	0.67	1.03
48437	texas	swisher	0.65	0.15	0.06	1.47	0.00	0.18	0.60	0.18	0.00	0.00	0.00	0.00	2.94	0.02	0.01	0.02	6.28
48439	texas	tarrant	0.50	1.54	0.05	0.21	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	2.48	0.02	0.01	1.01	5.96
48441	texas	taylor	0.22	0.27	0.02	1.49	0.00	0.10	0.37	0.02	0.01	0.00	0.01	0.00	1.11	0.02	0.00	0.17	3.81
48443	texas	terrell	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.11	0.23	0.41
48445	texas	terry	0.08	0.07	0.01	0.09	0.00	0.14	2.25	0.01	0.09	0.00	0.00	0.01	0.35	0.00	0.00	0.03	3.13
48447	texas	throckmorton	0.14	0.29	0.01	0.88	0.00	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.51	0.58	0.00	0.02	2.45
48449	texas	titus	1.03	0.53	0.10	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.01	0.00	0.82	0.02	1.51	0.09	4.22
48451	texas	tom green	0.22	0.06	0.03	0.76	0.00	0.28	1.76	0.02	0.01	0.00	0.02	0.00	0.89	0.04	0.00	1.55	5.63
48453	texas	travis	1.10	3.35	0.04	0.37	0.00	0.45	0.09	0.00	0.00	0.00	0.00	0.00	1.47	0.02	0.03	0.68	7.62
48455	texas	trinity	0.14	0.36	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.55	0.01	0.00	0.07	1.16
48457	texas	tyler	0.08	0.20	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.03	0.00	0.09	0.72
48459	texas	upshur	0.80	0.72	0.08	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.01	0.00	1.82	0.01	0.89	0.25	4.65
48461	texas	upton	0.03	0.01	0.00	0.09	0.00	0.00	0.26	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.10	0.66
48463	texas	uvalde	0.31	0.31	0.02	0.27	0.00	0.17	0.15	0.03	0.00	0.00	0.01	0.00	0.66	0.00	0.00	0.31	2.25
48465	texas	val verde	0.01	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.50	0.66
48467	texas	van zandt	0.71	1.84	0.07	0.04	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	3.67	0.02	0.01	0.42	6.81
48469	texas	victoria	0.76	1.86	0.06	0.12	0.00	0.30	0.23	0.00	0.00	0.00	0.00	0.04	2.12	0.03	0.00	0.10	5.64
48471	texas	walker	0.49	0.67	0.05	0.04	0.00	0.02	0.00	0.03	0.00	0.00	0.00	0.00	1.79	0.56	0.10	0.22	3.98
48473	texas	waller	0.71	1.54	0.07	0.04	0.00	0.22	0.02	0.01	0.00	0.00	0.00	0.00	3.03	0.03	0.01	0.45	6.13
48475	texas	ward	0.01	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.01	0.01	0.11
48477	texas	washington	0.46	1.08	0.04	0.04	0.00	0.02	0.02	0.00	0.00	0.00	0.00	0.00	2.37	0.07	0.37	0.17	4.66
48479	texas	webb	0.50	0.42	0.05	0.02	0.00	0.03	0.01	0.00	0.00	0.00	0.00	0.00	2.16	1.13	0.00	0.11	4.43
48481	texas	wharton	1.90	2.03	0.22	0.12	0.00	1.86	2.03	0.10	0.00	0.00	0.02	0.01	1.82	0.01	0.31	0.11	10.51
48483	texas	wheeler	0.23	0.24	0.02	0.22	0.00	0.06	0.09	0.00	0.00	0.00	0.00	0.00	1.04	0.44	0.01	0.01	2.37
48485	texas	wichita	0.21	1.38	0.02	2.89	0.00	0.04	0.06	0.02	0.00	0.00	0.02	0.00	1.04	0.01	0.00	0.12	5.82
48487	texas	wilbarger	0.12	0.29	0.01	3.73	0.00	0.13	0.27	0.01	0.05	0.00	0.00	0.00	0.59	0.00	0.00	0.05	5.25
48489	texas	willacy	0.14	1.06	0.01	0.01	0.04	2.31	1.02	0.10	0.00	0.01	0.02	0.00	0.63	0.01	0.00	0.03	5.38
48491	texas	williamson	4.88	5.59	0.11	1.00	0.00	1.00	0.82	0.00	0.00	0.00	0.01	0.00	5.21	0.02	0.01	1.05	19.71
48493	texas	wilson	0.94	1.93	0.08	0.15	0.00	0.19	0.06	0.00	0.01	0.00	0.02	0.00	4.12	0.05	0.00	0.31	7.86
48495	texas	winkler	0.04	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.41	0.00	0.01	0.54
48497	texas	wise	0.57	1.52	0.05	0.15	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	2.97	0.05	0.01	0.63	6.00
48499	texas	wood	1.05	0.74	0.10	0.06	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	2.34	0.01	1.06	0.20	5.61
48501	texas	yoakum	0.04	0.03	0.01	0.06	0.00	0.03	1.00	0.01	0.12	0.00	0.00	0.00	0.07	0.41	0.00	0.01	1.81
48503	texas	young	0.15	0.89	0.01	0.39	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.77	0.00	0.00	0.06	2.31
48505	texas	zapata	0.08	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.35	0.00	0.01	0.87
48507	texas	zavala	0.15	0.51	0.01	0.24	0.00	0.14	0.09	0.03	0.01	0.00	0.01	0.00	0.59	0.00	0.00	0.07	1.85
49001	utah	beaver	0.11	0.16	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.12	1.35	0.00	0.01	1.78
49003	utah	box elder	0.17	0.76	0.01	0.54	0.00	0.05	0.00	0.11	0.00	0.00	0.01	0.00	0.83	0.00	0.00	0.22	2.71
49005	utah	cache	0.25	0.92	0.02	0.35	0.00	0.15	0.00	0.09	0.00	0.00	0.01	0.00	0.90	0.09	0.37	0.17	3.32
49007	utah	carbon	0.01	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.06	0.22
49009	utah	daggett	0.10	0.02	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	1.51	0.00	0.00	1.67
49011	utah	davis	0.04	0.31	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.12	0.00	0.00	0.14	0.69
49013	utah	duchesne	0.04	0.27	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.06	0.71
49015	utah	emery	0.01	0.09	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.04	0.32
49017	utah	garfield	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.01	0.16
49019	utah	grand	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.03	0.01	0.07
49021	utah	iron	0.17	0.34	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.33	1.81	0.00	0.10	2.79
49023	utah	juab	0.01	0.11	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.04	0.29
49025	utah	kane	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.01	0.07
49027	utah	millard	0.11	0.45	0.01	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.00	0.15	0.04	1.28
49029	utah	morgan	0.01	0.12	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.13	0.39
49031	utah	piute	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.01	0.14
49033	utah	rich	0.02	0.16	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.31	0.00	0.00	0.04	0.54
49035	utah	salt lake	0.03	0.26	0.00	0.10	0.00	0.01	0.00	0.00	0.00	0.00	0.02	0.00	0.18	0.00	0.00	0.28	0.89
49037	utah	san juan	0.01	0.06	0.00	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.05	0.23
49039	utah	sanpete	0.05	0.36	0.01	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.04	0.25	1.10
49041	utah	sevier	0.03	0.16	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.00	0.00	0.03	0.49
49043	utah	summit	0.01	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.16	0.39
49045	utah	tooele	0.04	0.22	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.01	0.31	0.05	0.94
49047	utah	uintah	0.02	0.27	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.10	0.67
49049	utah	utah	0.43	1.28	0.04	0.38	0.00	0.08	0.00	0.01	0.00	0.06	0.01	0.00	1.78	0.04	0.66	0.55	5.32
49051	utah	wasatch	0.01	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.11	0.38
49053	utah	washington	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.04	0.26
49055	utah	wayne	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.02	0.19
49057	utah	weber	0.15	0.58	0.01	0.08	0.00	0.02	0.00	0.00	0.00	0.00	0.02	0.00	0.79	0.01	0.18	0.24	2.11
50001	vermont	addison	1.07	0.64	0.12	0.05	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	1.66	0.01	0.04	0.14	3.76
50003	vermont	bennington	0.10	0.14	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.08	0.48
50005	vermont	caledonia	0.17	0.12	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.01	0.00	0.06	0.70
50007	vermont	chittenden	0.19	0.24	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.01	0.00	0.14	0.94
50009	vermont	essex	0.08	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.26
50011	vermont	franklin	0.85	0.20	0.08	0.04	0.00	0.03	0.00	0.01	0.00	0.00	0.00	0.00	1.31	0.01	0.03	0.05	2.61
50013	vermont	grand isle	0.08	0.08	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.01	0.33
50015	vermont	lamoille	0.09	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.04	0.37
50017	vermont	orange	0.30	0.19	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.50	0.02	0.00	0.09	1.16
50019	vermont	orleans	0.43	0.17	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.01	0.00	0.05	1.44
50021	vermont	rutland	0.29	0.27	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.47	0.01	0.00	0.18	1.27
50023	vermont	washington	0.09	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.01	0.01	0.07	0.44
50025	vermont	windham	0.18	0.17	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.01	0.00	0.15	0.83
50027	vermont	windsor	0.13	0.18	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.01	0.00	0.20	0.80
51001	virginia	accomack	1.48	0.23	0.57	0.23	0.00	0.08	0.01	0.06	0.00	0.00	0.03	0.02	0.02	0.49	2.25	0.02	5.49
51003	virginia	albemarle	0.07	0.29	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.03	0.01	0.36	1.40
51005	virginia	alleghany	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.04	0.20
51007	virginia	amelia	0.85	0.13	0.12	0.05	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.02	0.42	0.01	1.44	0.07	3.16
51009	virginia	amherst	0.04	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.42	0.00	0.00	0.07	0.71
51011	virginia	appomattox	0.04	0.13	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.01	0.02	0.03	0.66
51013	virginia	arlington	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
51015	virginia	augusta	1.66	0.65	0.18	0.10	0.00	0.08	0.01	0.00	0.00	0.00	0.00	0.02	2.69	0.10	1.76	0.45	7.70
51017	virginia	bath	0.02	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.02	0.21
51019	virginia	bedford	0.22	0.65	0.03	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.73	0.05	0.01	0.37	3.10
51021	virginia	bland	0.05	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.00	0.03	0.06	0.57
51023	virginia	botetourt	0.19	0.26	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.04	0.00	0.22	1.70
51025	virginia	brunswick	0.10	0.26	0.04	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.12	0.64	0.00	0.05	1.24
51027	virginia	buchanan	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.03	0.03	0.12
51029	virginia	buckingham	0.40	0.16	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.34	1.36	0.48	0.09	2.91
51031	virginia	campbell	0.14	0.31	0.04	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	1.04	0.02	0.00	0.16	1.75
51033	virginia	caroline	0.30	0.15	0.16	0.01	0.00	0.01	0.00	0.08	0.00	0.02	0.07	0.05	0.08	0.01	0.05	0.12	1.09
51035	virginia	carroll	0.13	0.16	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.22	0.00	0.00	0.16	1.71
51036	virginia	charles city	0.17	0.06	0.08	0.05	0.00	0.01	0.00	0.05	0.00	0.00	0.00	0.00	0.01	0.85	0.00	0.03	1.30
51037	virginia	charlotte	0.16	0.19	0.05	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.03	0.43	0.66	0.00	0.06	1.62
51041	virginia	chesterfield	0.07	0.20	0.06	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.05	0.06	0.07	0.19	0.73
51043	virginia	clarke	0.11	0.28	0.05	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.02	0.52	0.06	0.01	0.44	1.52
51045	virginia	craig	0.03	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.02	0.35
51047	virginia	culpeper	0.37	0.52	0.21	0.02	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	1.12	0.04	0.01	0.32	2.65
51049	virginia	cumberland	0.45	0.10	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.93	0.04	1.84
51051	virginia	dickenson	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.06	0.11
51053	virginia	dinwiddie	0.11	0.23	0.13	0.01	0.00	0.00	0.01	0.05	0.00	0.00	0.02	0.00	0.16	0.17	0.03	0.10	1.02
51057	virginia	essex	0.31	0.06	0.24	0.05	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.03	0.01	0.00	0.02	0.77
51059	virginia	fairfax	0.01	0.64	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.30	1.05
51061	virginia	fauquier	0.80	1.15	0.31	0.07	0.00	0.04	0.00	0.02	0.00	0.00	0.00	0.03	2.61	0.09	0.03	0.96	6.12
51063	virginia	floyd	0.15	0.25	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.11	0.04	0.00	0.15	1.75
51065	virginia	fluvanna	0.02	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.08	0.38
51067	virginia	franklin	0.91	0.29	0.10	0.06	0.00	0.04	0.00	0.01	0.00	0.00	0.00	0.00	2.50	0.03	0.01	0.16	4.12
51069	virginia	frederick	0.17	0.42	0.03	0.02	0.00	0.01	0.01	0.00	0.00	0.04	0.01	0.00	0.62	0.29	0.01	0.21	1.82
51071	virginia	giles	0.03	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.01	0.00	0.09	0.55
51073	virginia	gloucester	0.12	0.05	0.13	0.02	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.02	0.04	0.01	0.04	0.06	0.51
51075	virginia	goochland	0.19	0.14	0.04	0.02	0.00	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.40	0.00	0.05	0.19	1.08
51077	virginia	grayson	0.13	0.17	0.01	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.85	0.00	0.00	0.18	1.36
51079	virginia	greene	0.02	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.04	0.33
51081	virginia	greensville	0.02	0.17	0.10	0.01	0.00	0.00	0.08	0.01	0.02	0.00	0.02	0.00	0.02	0.00	0.17	0.02	0.66
51083	virginia	halifax	0.14	0.51	0.04	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.54	0.71	0.02	0.12	2.13
51650	virginia	hampton	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
51085	virginia	hanover	0.43	0.45	0.38	0.10	0.00	0.03	0.00	0.06	0.00	0.02	0.08	0.00	0.44	0.08	0.07	0.40	2.55

51087	virginia	henrico	0.20	0.30	0.14	0.03	0.00	0.01	0.00	0.08	0.00	0.02	0.04	0.00	0.11	0.01	0.00	0.15	1.09
51089	virginia	henry	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.08	0.32
51091	virginia	highland	0.02	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.23	0.00	0.00	0.07	0.40
51093	virginia	isle of wight	0.34	0.27	0.35	0.10	0.00	0.02	0.23	0.05	0.06	0.08	0.01	0.02	0.12	1.06	0.05	0.10	2.86
51095	virginia	james city	0.21	0.08	0.03	0.03	0.00	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.04	1.12	0.05	0.05	1.63
51097	virginia	king and queen	0.16	0.15	0.17	0.03	0.00	0.00	0.00	0.06	0.00	0.01	0.00	0.03	0.04	0.01	0.00	0.04	0.71
51099	virginia	king george	0.05	0.22	0.06	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.00	0.07	0.00	0.00	0.07	0.52
51101	virginia	king william	0.21	0.09	0.11	0.03	0.00	0.01	0.00	0.10	0.00	0.00	0.00	0.00	0.18	0.00	0.04	0.03	0.79
51103	virginia	lancaster	0.15	0.05	0.06	0.02	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.71	0.00	0.01	1.02
51105	virginia	lee	0.06	0.13	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.70	0.01	0.00	0.18	1.10
51107	virginia	loudoun	0.25	1.65	0.14	0.05	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	1.13	0.16	0.02	1.56	5.00
51109	virginia	louisa	0.09	0.32	0.05	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.34	0.02	0.04	0.14	1.03
51111	virginia	lunenburg	0.11	0.18	0.03	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.22	0.67	0.00	0.07	1.30
51113	virginia	madison	0.19	0.23	0.08	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.06	0.00	0.13	1.60
51115	virginia	mathews	0.02	0.02	0.03	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.26
51117	virginia	mecklenburg	0.06	0.28	0.11	0.03	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.05	0.33	0.00	0.03	0.08	1.05
51119	virginia	middlesex	0.18	0.06	0.10	0.04	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.04	0.72	0.00	0.03	1.18
51121	virginia	montgomery	0.20	0.16	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.06	0.00	0.22	1.57
51125	virginia	nelson	0.03	0.11	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.24	0.01	0.00	0.08	0.48
51127	virginia	new kent	0.17	0.08	0.06	0.01	0.00	0.01	0.00	0.04	0.00	0.00	0.00	0.02	0.04	0.88	0.00	0.05	1.36
51700	virginia	newport news	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02
51710	virginia	norfolk	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01
51131	virginia	northampton	0.16	0.02	0.41	0.29	0.00	0.02	0.01	0.05	0.00	0.00	0.04	0.04	0.00	0.00	0.02	0.01	1.08
51133	virginia	northumberland	0.19	0.02	0.22	0.14	0.00	0.02	0.01	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.64
51135	virginia	nottoway	0.35	0.16	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.01	0.87	0.08	1.87
51137	virginia	orange	0.24	0.47	0.12	0.04	0.00	0.02	0.00	0.01	0.00	0.02	0.00	0.00	0.76	0.03	0.01	0.22	1.94
51139	virginia	page	1.50	0.20	0.15	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.61	0.11	2.75	0.11	5.59
51141	virginia	patrick	0.05	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.01	0.00	0.10	0.71
51143	virginia	pittsylvania	0.74	0.62	0.10	0.09	0.00	0.04	0.01	0.06	0.00	0.00	0.01	0.00	2.50	0.41	0.08	0.32	4.99
51145	virginia	powhatan	0.07	0.15	0.02	0.01	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.15	0.00	0.27	0.10	0.78
51147	virginia	prince edward	0.16	0.16	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.69	0.19	0.07	1.64
51149	virginia	prince george	0.09	0.16	0.11	0.02	0.00	0.00	0.00	0.02	0.00	0.00	0.02	0.00	0.03	0.03	0.04	0.06	0.58
51153	virginia	prince william	0.08	0.49	0.05	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.42	0.01	0.01	0.54	1.63
51155	virginia	pulaski	0.11	0.19	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.87	0.00	0.00	0.11	1.31
51157	virginia	rappahannock	0.04	0.14	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.04	0.00	0.11	0.71
51159	virginia	richmond	0.28	0.10	0.25	0.05	0.00	0.01	0.00	0.02	0.00	0.00	0.00	0.02	0.04	0.00	0.00	0.01	0.78
51161	virginia	roanoke	0.06	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.06	0.10	0.50
51163	virginia	rockbridge	0.16	0.28	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.93	0.06	0.00	0.18	1.66
51165	virginia	rockingham	5.57	0.60	0.60	0.29	0.00	0.25	0.02	0.02	0.01	0.00	0.00	0.00	5.01	0.03	7.13	0.46	20.00
51167	virginia	russell	0.10	0.21	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	1.30	0.01	0.00	0.28	1.93
51169	virginia	scott	0.07	0.28	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.82	0.01	0.00	0.44	1.64
51171	virginia	shenandoah	1.17	0.27	0.14	0.07	0.00	0.06	0.01	0.00	0.00	0.00	0.00	0.00	1.14	0.14	1.64	0.29	4.93
51173	virginia	smyth	0.21	0.20	0.02	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.51	0.00	0.03	0.22	2.22
51175	virginia	southampton	0.40	0.31	0.34	0.07	0.00	0.02	0.33	0.11	0.11	0.02	0.01	0.03	0.09	2.04	0.23	0.06	4.16
51177	virginia	spotsylvania	0.10	0.27	0.06	0.01	0.00	0.01	0.00	0.01	0.00	0.00	0.01	0.00	0.34	0.06	0.00	0.15	1.03
51179	virginia	stafford	0.05	0.13	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.00	0.00	0.13	0.02	0.00	0.10	0.47
51800	virginia	suffolk	0.25	0.30	0.36	0.10	0.00	0.01	0.29	0.07	0.09	0.00	0.00	0.00	0.03	0.13	0.15	0.16	1.95
51181	virginia	surry	0.22	0.11	0.21	0.07	0.00	0.01	0.03	0.03	0.03	0.04	0.00	0.00	0.04	0.85	0.04	0.02	1.71
51183	virginia	sussex	0.13	0.31	0.19	0.02	0.00	0.01	0.05	0.06	0.02	0.01	0.00	0.01	0.02	0.69	0.00	0.03	1.56
51185	virginia	tazewell	0.13	0.17	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.90	0.01	0.03	0.28	1.53
51810	virginia	virginia beach	0.45	0.06	0.60	0.34	0.00	0.02	0.02	0.02	0.00	0.00	0.00	0.00	0.02	1.84	0.08	0.14	3.60
51187	virginia	warren	0.04	0.13	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.00	0.05	0.17	0.66
51191	virginia	washington	0.32	0.44	0.03	0.03	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	2.27	0.01	0.22	0.70	4.04
51193	virginia	westmoreland	0.40	0.09	0.26	0.03	0.00	0.05	0.00	0.07	0.00	0.00	0.13	0.02	0.07	0.89	0.00	0.01	2.02
51195	virginia	wise	0.13	0.03	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.79	0.03	0.06	1.14
51197	virginia	wythe	0.27	0.45	0.03	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	1.79	0.01	0.00	0.28	2.87
51199	virginia	york	0.03	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.05	0.03	0.18
53001	washington	adams	0.26	0.34	0.04	3.61	0.00	0.06	0.01	0.06	0.00	0.04	0.41	0.14	0.46	0.01	0.20	0.01	5.66
53003	washington	asotin	0.01	0.18	0.00	0.17	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.01	0.44
53005	washington	benton	0.76	0.57	0.17	1.56	0.02	0.04	0.05	0.08	0.02	0.61	1.31	0.02	0.86	0.02	0.00	0.10	6.20
53007	washington	chelan	0.05	0.17	0.02	0.02	0.00	0.01	0.01	0.00	0.00	0.13	0.01	0.00	0.00	0.00	0.00	0.03	0.45
53009	washington	clallam	0.01	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.04	0.19
53011	washington	clark	0.46	0.54	0.05	0.04	0.00	0.03	0.00	0.01	0.00	0.03	0.00	0.00	1.05	0.07	0.36	0.64	3.28
53013	washington	columbia	0.01	0.41	0.00	2.37	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.09	0.02	0.00	0.00	0.00	2.92
53015	washington	cowlitz	0.11	0.03	0.01	0.01	0.00	0.01	0.00	0.01	0.00	0.05	0.02	0.00	0.10	0.00	0.14	0.08	0.58
53017	washington	douglas	0.03	0.14	0.01	2.04	0.00	0.06	0.00	0.04	0.00	0.11	0.01	0.00	0.08	0.00	0.00	0.02	2.54
53019	washington	ferry	0.00	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.01	0.00	0.00	0.02	0.18
53021	washington	franklin	1.30	1.26	0.16	1.56	0.01	0.08	0.05	0.05	0.01	0.30	0.77	0.10	1.68	0.00	0.36	0.04	7.72
53023	washington	garfield	0.01	0.44	0.00	1.63	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.00	2.17
53025	washington	grant	1.15	1.76	0.10	1.78	0.00	0.05	0.03	0.16	0.01	0.72	0.87	0.32	1.54	0.00	0.22	0.05	8.76
53027	washington	grays harbor	0.02	0.12	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00	0.08	0.01	0.00	0.04	0.30
53029	washington	island	0.03	0.04	0.00	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.07	0.31
53031	washington	jefferson	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.03	0.09
53033	washington	king	1.04	0.08	0.10	0.05	0.00	0.04	0.01	0.07	0.00	0.02	0.02	0.00	3.65	0.15	0.05	1.79	7.08
53035	washington	kitsap	0.04	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.12	0.03	0.01	0.34	0.60
53037	washington	kittitas	0.04	0.24	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.14	0.01	0.00	0.09	0.56
53039	washington	klickitat	0.02	0.20	0.00	0.19	0.00	0.01	0.00	0.00	0.00	0.03	0.01	0.00	0.09	0.00	0.00	0.04	0.61
53041	washington	lewis	1.61	0.13	0.15	0.08	0.00	0.07	0.01	0.10	0.00	0.00	0.00	0.01	0.68	0.06	2.42	0.24	5.55
53043	washington	lincoln	0.04	0.36	0.01	4.89	0.00	0.39	0.00	0.02	0.00	0.00	0.66	0.03	0.15	0.01	0.00	0.04	6.58
53045	washington	mason	0.02	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.08	0.21
53047	washington	okanogan	0.05	0.13	0.01	0.04	0.00	0.01	0.00	0.00	0.00	0.03	0.01	0.00	0.12	0.00	0.00	0.08	0.49
53049	washington	pacific	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.08	0.03	0.00	0.01	0.17
53051	washington	pend oreille	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.01	0.07
53053	washington	pierce	0.23	0.09	0.02	0.04	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.58	0.06	2.09	0.85	3.99
53055	washington	san juan	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.03	0.09
53057	washington	skagit	0.34	0.07	0.05	0.05	0.00	0.03	0.01	0.01	0.00	0.07	0.08	0.00	0.96	0.01	0.61	0.11	2.40
53059	washington	skamania	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.02	0.00	0.00	0.04	0.00	0.00	0.02	0.13
53061	washington	snohomish	0.65	0.16	0.06	0.04	0.01	0.04	0.00	0.05	0.00	0.05	0.20	0.00	2.03	0.05	1.46	0.68	5.47
53063	washington	spokane	0.09	1.43	0.01	5.07	0.00	0.58	0.00	0.17	0.00	0.00	1.24	0.88	0.45	0.02	0.01	0.31	10.27
53065	washington	stevens	0.02	0.22	0.00	0.05	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.02	0.00	0.07	0.52
53067	washington	thurston	0.38	0.11	0.03	0.01	0.00	0.01	0.00	0.03	0.00	0.01	0.00	0.00	0.82	0.05	1.99	0.31	3.75
53069	washington	wahkiakum	0.01	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.01	0.08
53071	washington	walla walla	0.16	1.55	0.03	4.33	0.00	0.02	0.01	0.06	0.00	0.09	0.16	0.19	0.37	0.00	0.00	0.03	7.02
53073	washington	whatcom	0.50	0.09	0.05	0.05	0.00	0.02	0.00	0.00	0.00	0.06	0.02	0.00	1.64	0.01	0.02	0.06	2.51
53075	washington	whitman	0.07	2.31	0.02	10.11	0.00	1.02	0.01	0.04	0.00	0.01	0.70	1.83	0.19	0.08	0.00	0.03	16.41
53077	washington	yakima	2.85	0.95	0.35	0.43	0.01	0.13	0.07	0.36	0.02	1.35	0.18	0.01	7.18	0.02	0.55	0.36	14.81
54001	west virginia	barbour	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.07	0.42
54003	west virginia	berkeley	0.18	0.56	0.04	0.02	0.00	0.01	0.00	0.00	0.00	0.06	0.01	0.00	0.44	0.05	0.01	0.37	1.75
54005	west virginia	boone	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10
54007	west virginia	braxton	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.09	0.28
54009	west virginia	brooke	0.01	0.10	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.03	0.25
54011	west virginia	cabell	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.13	0.28
54013	west virginia	calhoun	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.04	0.15
54015	west virginia	clay	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.08
54017	west virginia	doddridge	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.09	0.27
54019	west virginia	fayette	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.04	0.22
54021	west virginia	gilmer	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.04	0.23
54023	west virginia	grant	0.57	0.11	0.05	0.03	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	1.19	0.09	2.23
54025	west virginia	greenbrier	0.04	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.01	0.01	0.18	1.10
54027	west virginia	hampshire	0.45	0.14	0.04	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.01	0.87	0.20	2.09
54029	west virginia	hancock	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.09	0.20
54031	west virginia	hardy	1.58	0.12	0.15	0.08	0.00	0.07	0.01	0.00	0.00	0.00	0.00	0.00	0.44	0.01	3.22	0.15	5.82
54033	west virginia	harrison	0.01	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.00	0.00	0.23	0.72
54035	west virginia	jackson	0.02	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.25	0.64
54037	west virginia	jefferson	0.43	0.40	0.23	0.09	0.00	0.03	0.00	0.01	0.00	0.02	0.00	0.03	1.02	0.04	0.01	0.62	2.95
54039	west virginia	kanawha	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.01	0.00	0.11	0.23
54041	west virginia	lewis	0.01	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.09	0.39
54043	west virginia	lincoln	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.05	0.14
54045	west virginia	logan	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05
54049	west virginia	marion	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.17	0.41
54051	west virginia	marshall	0.01	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.01	0.00	0.19	0.61
54053	west virginia	mason	0.11	0.32	0.09	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.02	0.00	0.13	1.07
54047	west virginia	mcdowell	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02
54055	west virginia	mercer	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.01	0.00	0.13	0.34
54057	west virginia	mineral	0.29	0.13	0.03	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.57	0.11	1.28
54059	west virginia	mingo	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.08
54061	west virginia	monongalia	0.01	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.15	0.52
54063	west virginia	monroe	0.04	0.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.53	0.01	0.01	0.13	0.92
54065	west virginia	morgan	0.01	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.06	0.22
54067	west virginia	nicholas	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.07	0.31
54069	west virginia	ohio	0.02	0.17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.10	0.47
54071	west virginia	pendleton	0.63	0.11	0.06	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.30	0.00	1.25	0.15	2.56
54073	west virginia	pleasants	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.05	0.12
54075	west virginia	pocahontas	0.01	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.10	0.44
54077	west virginia	preston	0.07	0.28	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.51	0.02	0.01	0.24	1.14
54079	west virginia	putnam	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.01	0.00	0.14	0.37
54081	west virginia	raleigh	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.01	0.00	0.09	0.21
54083	west virginia	randolph	0.01	0.15	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.01	0.00	0.14	0.44
54085	west virginia	ritchie	0.01	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.08	0.36
54087	west virginia	roane	0.01	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.15	0.45
54089	west virginia	summers	0.00	0.08	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.10	0.00	0.00	0.05	0.27
54091	west virginia	taylor	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.00	0.00	0.09	0.30
54093	west virginia	tucker	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.02	0.12
54095	west virginia	tyler	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.05	0.19
54097	west virginia	upshur	0.01	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.06	0.33
54099	west virginia	wayne	0.00	0.11	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.08	0.24
54101	west virginia	webster	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.02	0.07
54103	west virginia	wetzel	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.08	0.19
54105	west virginia	wirt	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.04	0.18
54107	west virginia	wood	0.04	0.16	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.01	0.00	0.22	0.62
54109	west virginia	wyoming	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.03
55001	wisconsin	adams	0.41	0.20	0.14	0.03	0.00	0.01	0.00	0.00	0.00	0.01	0.18	0.02	0.15	0.00	0.00	0.03	1.21
55003	wisconsin	ashland	0.04	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.01	0.17
55005	wisconsin	barron	1.31	0.61	0.43	0.07	0.00	0.06	0.00	0.00	0.00	0.00	0.11	0.03	1.17	0.01	0.12	0.14	4.08
55007	wisconsin	bayfield	0.04	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00	0.04	0.28
55009	wisconsin	brown	3.05	1.22	0.67	0.49	0.00	0.17	0.01	0.00	0.00	0.00	0.00	0.00	5.13	0.03	0.00	0.18	10.96
55011	wisconsin	buffalo	1.71	0.52	0.40	0.06	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	1.36	0.04	0.80	0.12	5.07
55013	wisconsin	burnett	0.15	0.13	0.04	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.15	0.00	0.00	0.06	0.55
55015	wisconsin	calumet	1.93	0.78	0.75	0.41	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.02	2.74	0.01	0.00	0.09	6.82
55017	wisconsin	chippewa	2.05	0.90	0.67	0.05	0.00	0.12	0.00	0.01	0.00	0.00	0.00	0.00	1.62	0.05	0.02	0.14	5.66
55019	wisconsin	clark	2.84	1.17	0.65	0.17	0.00	0.22	0.01	0.03	0.00	0.00	0.00	0.00	3.36	0.04	0.02	0.30	8.81
55021	wisconsin	columbia	4.97	1.01	1.37	0.45	0.00	0.08	0.01	0.02	0.00	0.00	0.03	0.01	1.95	0.21	0.48	1.06	11.63
55023	wisconsin	crawford	0.81	0.46	0.24	0.02	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.62	0.11	0.00	0.12	2.42
55025	wisconsin	dane	10.75	2.73	3.37	0.98	0.00	0.23	0.02	0.02	0.00	0.00	0.04	0.00	6.97	1.35	0.09	0.99	27.55
55027	wisconsin	dodge	7.25	1.94	2.61	0.48	0.00	0.23	0.01	0.00	0.00	0.00	0.11	0.04	4.79	0.83	0.02	0.38	18.70
55029	wisconsin	door	0.34	0.23	0.09	0.12	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.43	0.00	0.00	0.06	1.31
55031	wisconsin	douglas	0.02	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.06	0.00	0.00	0.04	0.27
55033	wisconsin	dunn	1.69	0.53	0.62	0.03	0.00	0.10	0.00	0.00	0.00	0.00	0.08	0.06	1.39	0.06	0.01	0.19	4.77
55035	wisconsin	eau claire	1.11	0.46	0.32	0.04	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.76	0.06	0.16	0.17	3.17
55037	wisconsin	florence	0.01	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.08
55039	wisconsin	fond du lac	5.18	1.33	1.57	0.78	0.00	0.18	0.01	0.00	0.00	0.00	0.14	0.06	5.53	0.19	0.00	0.20	15.19
55041	wisconsin	forest	0.03	0.06	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.01	0.00	0.01	0.07	0.00	0.03	0.23
55043	wisconsin	grant	5.17	1.88	1.38	0.18	0.00	0.19	0.01	0.01	0.00	0.00	0.00	0.00	4.35	1.45	0.06	0.95	15.64
55045	wisconsin	green	3.53	1.72	1.41	0.30	0.00	0.12	0.01	0.00	0.00	0.00	0.03	0.00	3.12	0.11	0.01	0.42	10.78
55047	wisconsin	green lake	1.24	0.22	0.29	0.14	0.00	0.03	0.00	0.00	0.00	0.00	0.14	0.02	0.70	0.01	0.01	0.11	2.92
55049	wisconsin	iowa	2.11	0.88	0.62	0.12	0.00	0.07	0.00	0.01	0.00	0.00	0.12	0.00	2.39	0.09	0.34	0.50	7.25
55051	wisconsin	iron	0.01	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.09
55053	wisconsin	jackson	0.92	0.29	0.21	0.03	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.83	0.29	0.01	0.12	2.75
55055	wisconsin	jefferson	4.06	1.01	2.05	0.40	0.00	0.08	0.01	0.02	0.00	0.00	0.39	0.01	1.99	0.22	0.59	0.91	11.74
55057	wisconsin	juneau	0.73	0.25	0.27	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.08	0.00	0.58	0.01	0.28	0.08	2.38
55059	wisconsin	kenosha	2.61	0.45	1.28	0.55	0.00	0.02	0.00	0.00	0.00	0.00	0.09	0.00	0.81	0.08	0.00	0.31	6.21
55061	wisconsin	kewaunee	1.24	0.42	0.21	0.23	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	2.10	0.05	0.00	0.03	4.42
55063	wisconsin	la crosse	0.99	0.40	0.27	0.02	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.69	0.22	0.01	0.13	2.78
55065	wisconsin	lafayette	4.20	1.33	1.34	0.13	0.00	0.12	0.01	0.00	0.00	0.00	0.00	0.00	3.19	0.41	0.01	0.36	11.11
55067	wisconsin	langlade	0.28	0.16	0.07	0.02	0.00	0.10	0.00	0.02	0.00	0.00	0.07	0.01	0.33	0.00	0.00	0.05	1.11
55069	wisconsin	lincoln	0.18	0.20	0.05	0.01	0.00	0.02	0.00	0.01	0.00	0.00	0.02	0.00	0.19	0.01	0.00	0.06	0.74
55071	wisconsin	manitowoc	2.36	0.97	0.62	0.47	0.00	0.16	0.01	0.00	0.00	0.00	0.00	0.01	3.43	0.01	0.31	0.20	8.55
55073	wisconsin	marathon	3.06	1.40	0.81	0.17	0.00	0.23	0.01	0.08	0.00	0.00	0.04	0.00	3.60	0.02	0.00	0.27	9.68
55075	wisconsin	marinette	0.57	0.22	0.10	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.01	0.00	0.60	0.02	0.00	0.08	1.64
55077	wisconsin	marquette	0.82	0.22	0.20	0.05	0.00	0.02	0.00	0.05	0.00	0.00	0.02	0.00	0.41	0.13	0.01	0.12	2.05
55078	wisconsin	menominee	0.04	0.12	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.31	0.00	0.61
55079	wisconsin	milwaukee	0.09	0.33	0.08	0.08	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.05	0.05	0.72
55081	wisconsin	monroe	1.54	0.79	0.36	0.07	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	1.55	0.03	0.02	0.35	4.78
55083	wisconsin	oconto	1.00	0.40	0.28	0.10	0.00	0.04	0.00	0.00	0.00	0.00	0.06	0.00	1.04	0.01	0.00	0.13	3.08
55085	wisconsin	oneida	0.02	0.05	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.03	0.14
55087	wisconsin	outagamie	3.27	0.96	1.53	0.38	0.00	0.12	0.01	0.00	0.00	0.00	0.26	0.00	4.08	0.08	0.00	0.19	10.90
55089	wisconsin	ozaukee	0.98	0.87	0.47	0.26	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.01	1.13	0.01	0.64	0.17	4.62
55091	wisconsin	pepin	0.54	0.16	0.18	0.02	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.54	0.00	0.01	0.04	1.53
55093	wisconsin	pierce	2.51	0.87	0.80	0.03	0.00	0.13	0.00	0.01	0.00	0.00	0.01	0.00	1.43	0.03	0.00	0.32	6.14
55095	wisconsin	polk	1.07	0.50	0.28	0.03	0.00	0.05	0.00	0.01	0.00	0.00	0.00	0.00	0.93	0.04	0.02	0.21	3.14
55097	wisconsin	portage	1.09	0.57	0.20	0.03	0.00	0.06	0.01	0.01	0.00	0.01	0.43	0.06	1.11	0.07	0.01	0.14	3.79
55099	wisconsin	price	0.11	0.17	0.01	0.01	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.16	0.00	0.00	0.06	0.53
55101	wisconsin	racine	2.12	0.50	1.55	0.31	0.00	0.02	0.00	0.02	0.00	0.00	0.15	0.00	0.63	0.15	0.01	0.85	6.30
55103	wisconsin	richland	1.11	0.55	0.27	0.03	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	1.09	0.12	0.00	0.44	3.67
55105	wisconsin	rock	9.43	1.38	4.18	0.68	0.00	0.09	0.01	0.03	0.00	0.00	0.15	0.01	2.84	0.73	0.01	0.55	20.09
55107	wisconsin	rusk	0.38	0.23	0.07	0.02	0.00	0.03	0.00	0.01	0.00	0.00	0.03	0.00	0.46	0.00	0.00	0.07	1.31
55111	wisconsin	sauk	2.79	1.00	0.79	0.13	0.00	0.10	0.01	0.01	0.00	0.00	0.01	0.00	2.29	1.39	0.07	0.67	9.27
55113	wisconsin	sawyer	0.09	0.07	0.02	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.02	0.30
55115	wisconsin	shawano	2.26	0.79	0.49	0.15	0.00	0.13	0.01	0.00	0.00	0.00	0.11	0.00	2.39	0.05	0.01	0.22	6.61
55117	wisconsin	sheboygan	2.04	0.91	0.83	0.48	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.02	2.38	0.03	0.00	0.25	7.05
55109	wisconsin	st croix	2.29	0.91	0.84	0.07	0.00	0.12	0.00	0.02	0.00	0.00	0.03	0.00	1.54	0.12	0.01	0.30	6.25
55119	wisconsin	taylor	0.60	0.41	0.16	0.02	0.00	0.04	0.00	0.01	0.00	0.00	0.05	0.00	0.71	0.02	0.00	0.12	2.14
55121	wisconsin	trempealeau	2.34	0.59	0.47	0.07	0.00	0.08	0.01	0.00	0.00	0.00	0.02	0.01	1.39	0.09	0.97	0.13	6.17
55123	wisconsin	vernon	1.95	1.05	0.55	0.07	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	1.67	0.06	0.08	0.47	6.00
55125	wisconsin	vilas	0.02	0.03	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.01	0.00	0.00	0.07	0.17	0.01	0.33
55127	wisconsin	walworth	5.36	0.81	1.70	0.40	0.00	0.07	0.01	0.03	0.00	0.00	0.04	0.01	2.35	1.13	0.73	0.46	13.11
55129	wisconsin	washburn	0.10	0.09	0.03	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.10	0.00	0.00	0.06	0.41
55131	wisconsin	washington	2.04	0.90	0.86	0.34	0.00	0.16	0.00	0.00	0.00	0.00	0.00	0.00	2.18	0.02	0.61	0.24	7.36

55133	wisconsin	waukesha	2.33	0.94	1.17	0.32	0.00	0.04	0.00	0.00	0.01	0.00	0.00	0.36	0.00	0.70	0.07	0.01	0.60	6.54
55135	wisconsin	waupaca	1.99	0.77	0.55	0.16	0.00	0.06	0.00	0.00	0.01	0.00	0.00	0.14	0.00	1.53	0.03	0.00	0.20	5.44
55137	wisconsin	waushara	0.79	0.29	0.23	0.09	0.00	0.03	0.00	0.00	0.03	0.00	0.00	0.28	0.04	0.34	0.02	0.00	0.16	2.31
55139	wisconsin	winnebago	1.93	0.66	1.18	0.64	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.01	0.01	1.54	0.01	0.00	0.18	6.23
55141	wisconsin	wood	0.91	0.63	0.28	0.08	0.00	0.06	0.00	0.00	0.01	0.00	0.00	0.01	0.00	0.98	0.01	0.00	0.14	3.14
56001	wyoming	albany	0.02	0.22	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.45	0.03	0.00	0.07	0.80
56003	wyoming	big horn	0.07	0.33	0.00	0.04	0.04	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.23	0.02	0.00	0.09	1.03
56005	wyoming	campbell	0.02	0.12	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.40	0.02	0.00	0.14	0.74
56007	wyoming	carbon	0.02	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.52	0.02	0.00	0.06	0.85
56009	wyoming	converse	0.01	0.11	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.27	0.02	0.00	0.22	0.70
56011	wyoming	crook	0.02	0.22	0.00	0.02	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.35	0.02	0.00	0.08	0.71
56013	wyoming	fremont	0.03	0.58	0.00	0.07	0.01	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.41	0.02	0.00	0.19	1.36
56015	wyoming	goshen	0.19	0.52	0.00	0.15	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.70	0.03	0.00	0.05	1.72
56017	wyoming	hot springs	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.03	0.24
56019	wyoming	johnson	0.01	0.14	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.02	0.00	0.12	0.61
56021	wyoming	laramie	0.14	0.44	0.01	0.79	0.01	0.07	0.00	0.00	0.07	0.00	0.00	0.00	0.04	0.76	0.03	0.00	0.15	2.52
56023	wyoming	lincoln	0.01	0.43	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.02	0.00	0.16	0.86
56025	wyoming	natrona	0.02	0.18	0.00	0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.03	0.00	0.18	0.84
56027	wyoming	niobrara	0.01	0.11	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.02	0.00	0.03	0.51
56029	wyoming	park	0.03	0.30	0.00	0.09	0.05	0.09	0.00	0.00	0.01	0.00	0.00	0.00	0.04	0.17	0.02	0.00	0.07	0.85
56031	wyoming	platte	0.06	0.23	0.00	0.07	0.01	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.45	0.02	0.00	0.04	0.89
56033	wyoming	sheridan	0.01	0.24	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.29	0.02	0.00	0.06	0.71
56035	wyoming	sublette	0.01	0.24	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.34	0.02	0.00	0.08	0.69
56037	wyoming	sweetwater	0.01	0.19	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.02	0.00	0.05	0.42
56039	wyoming	teton	0.00	0.09	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.00	0.03	0.20
56041	wyoming	uinta	0.01	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.18	0.02	0.00	0.14	0.47
56043	wyoming	washakie	0.02	0.13	0.00	0.07	0.02	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.12	0.00	0.00	0.09	0.52
56045	wyoming	weston	0.01	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.25	0.02	0.00	0.05	0.42

ATTACHMENT 17

Technical Stakeholder Working Group Meeting Summary – Fall 2021

General permit overview

The Division of Water Resources has been directed by the North Carolina General Assembly to develop general permits by July 1, 2022, for existing Animal Feeding Operations (AFO) that intend to build and operate a digester system. Senate Bill 605, known as the 2021 Farm Act, describes the general permit requirements. Currently, digester systems are processed as modifications to a facility's existing permit.

The new general permits will apply to currently permitted facilities seeking to add digester systems. The Department, under the 2021 NC Farm Act, maintains the authority to require an individual permit for a facility adding a digester when necessary.

Input process

Event	Type of Feedback	Timeframe
Stakeholder Process	Technical Workgroup Sessions	Fall 2021
Stakeholder Process	Public Meeting/Forum - oral, written and electronic comments accepted	Fall/Winter 2021
Public Comment Period	Public Meetings - oral, written and electronic comments accepted	Winter 2021/Spring 2022

Existing permits

As of Sept. 27, 2021, out of 2,083 total permitted swine facilities in North Carolina, there are 15 facilities which have one or more animal waste digester systems.

List of Permits

Stakeholder process

DWR invited a group of approximately 60 stakeholders to gather input on the general permits content over two meetings in November-December 2021. The stakeholders represented community groups and environmental non-governmental organizations, state agencies and universities, and permittee and industry representatives.

Meeting #1: November 19, 2021

28 attendees met via Zoom to learn about the parameters of the general permits and then participated in breakout sessions discussing these sections of the existing general permit:

- Performance Standards
- Operations and Maintenance
- Monitoring and Reporting

- Other Conditions, including Inspections and Entry, General Conditions, Penalties and Definitions

Staff gathered policy and technical questions from attendees to answer in the second session.

Key themes

Participants raised the following issues:

- Monitoring for methane, nitrous oxide and ammonia from constructed biogas operations.
- The potential economic opportunity of the digester systems that could also remove potential pollutants from farm properties.
- More frequent monitoring of groundwater nearby digester operations
- Having the new general permits carry over the same monitoring conditions as the existing general permits.
- Potential expanded alert system for breaches.
- Better explanations of technical components of the general permits and digester systems for the communities near digester facilities.
- Avoiding overly regulating operations that are already regulated under existing permits.
- Further study of cumulative impacts on properties near farming operations.
- Addressing cattle and poultry operations as unique from swine operations.

Meeting #2: December 7, 2021

39 attendees rejoined on Zoom for the second work group session. They heard responses to some questions raised to staff from the first session, and then added feedback on discussion prompts from DWR staff on these items needing further input:

- Notification requirements on construction, completion and startup operations
- Operational parameters
- Priorities for recordkeeping requirements of operations and maintenance
- Parameters, analytical components, frequency and duration of influent and effluent monitoring
- Monitoring and reporting of failures, bypasses and other operational disruptions
- Safety precautions during severe weather or other emergency events
- Requirements for design and equipment in the digester system
- Requirements for infrastructure associated with the transport of biogas
- Conditions for use or re-introduction of tail gas in digester operations
- Requirements for inspection conditions and frequency specific to digester systems

Key themes

Participants raised the following issues:

- Ensuring pipelines carrying gas from digesters are properly sealed and monitored
- General permits capturing the wide range of digester system types that could be used.
- Improving understanding of the reporting process and where public records can be found.
- Potential for expanded notification of neighbors of construction of a digester system.

- Providing more details on responsibility for addressing gas transportation issues beyond the farm property.
- More frequent monitoring of effluent from facilities.
- Not adding additional requirements from existing general permits to new digester system operations.
- Potential need for an expanded emergency alert system for potential spills or leaks.
- Fully defining what elements of the digester and transportation system are covered under the general permits and regulated by DWR.
- Making sure other area of DEQ (particularly the Division of Air Quality) are involved in the general permit development.

Next steps

After draft permits are produced by DWR staff, the Department will hold three public hearings in the spring of 2022. Information on meeting locations and comment procedures for these hearings will be released early next year.

ATTACHMENT 18

DRAFT ENVIRONMENTAL JUSTICE REPORT

Biogas Digester General Permit Development

North Carolina Department of Environmental Quality

2/2/2022

Contents

1 Introduction

In July of 2021, the North Carolina General Assembly passed the Farm Act of 2021, Session Law 2021 - 78. Section 11 of the legislation required the North Carolina Department of Environmental Quality (NCDEQ or Department) to develop a General Permit for facilities that construct and operate a farm digester system. This analysis will evaluate the three types of general permits separately: Swine, Cattle, and Poultry with a Liquid Waste Management System (Wet Poultry.)

Animal operations are defined by General Statute 143-215.10B as feedlots involving more than 250 swine, 100 confined cattle, 75 horses, 1,000 sheep, or 30,000 poultry with a liquid waste management system. The general permits contain the required performance standards, operation and maintenance requirements, monitoring and reporting requirements, policy for inspections and entry to the farms, general conditions and the penalty policy. Each general permit is issued with a Certificate of Coverage that is permittee-specific and designates the permitted number and type of animals.

Based on the goal and scope of this analysis, several methodologies were considered which built upon a previously conducted, smaller scale community analysis which focused exclusively on swine AFOs. All of the methodologies used steady state live weight (SSLW), or the number of animal feeding operations (AFO), combined with proximity either to land or people to determine where (based on permitted facility reporting), the greatest number of animals are likely located near the greatest number of people.

It is important to note that this is an analysis of the facilities with current 2019 general permits and it is not anticipated that every facility covered under the 2019 general permits will apply for the new digester general permits. Additionally, the new digester general permits will replace the existing general permits only if the facility applies for the digester general permit. As it is not possible to predict which facilities will install digesters under the new general permits, this analysis relies on general information and is not a site-specific examination.

2 Methodology

Previously, NCDEQ developed five potential methods from which to select the communities with the highest potential exposure to AFOs (See Table 1). In most methods, SSLW was used as the indicator, assuming that higher SSLW values result in the generation of more waste. Higher amounts of waste may lead to externalities typically reported as complaints associated with facility operations (i.e. odor).

Using the fifth methodology outlined below, NCDEQ was able to effectively identify the areas across the state with the highest amount of SSLW per square mile. SSLW was separated out by the type of certificate of coverage: Cattle, Wet Poultry, and Swine. Due to availability of data, only the types of facilities required to have an NCDEQ-issued certificate of coverage under state law were included in this analysis. The 5 counties with the highest SSLW per square mile for each of the three types of coverage were selected for the analysis. Select demographic data was overlaid with the SSLW per square mile across the state at the county and census tract level. The datasets included in this analysis are poverty (Table S1701, American Community Survey 2019 5-year estimates), race and ethnicity (Table P2, 2020 Census), and limited English proficiency; Spanish (Table C16002, American Community Survey 2019 5-year estimates).

Table 1. Potential Methodologies for Analysis

Method	Description of Methodology
Land Proximity by SSLW: This method calculates the locational proximity of all land parcels (regardless of the parcel's use classification) to SSLW	1) A 2km buffer was placed around each COC 2) All land parcels within 2km of a COC were assigned the SSLW value of that COC. If a land parcel was proximate to more than one COC, the parcel was assigned the total SSLW from all COCs within 2km
Number of Residential Parcels Within 2km of a Swine State COC Per Census Tract: This method calculates the number of parcels in each Census Tract that are within 2km of a COC, without factoring SSLW	1) Residential parcels were selected from county parcel data, and any residential parcels within 2km of a COC were selected 2) The residential parcels were joined to the census tract to calculate the number of residential parcels within each census tract that were located proximate to at least one COC
Average SSLW Near Residential Parcels: This method calculates the average SSLW of residential parcels located within 2km of a COC in each Census Tract	1) Residential parcels were selected from county parcel data 2) Each residential parcel was assigned the SSLW amount of SSLW of each COC located within 2km of the parcel 3) To compare census tracts across the counties, the SSLW values assigned to each residential parcel within each census tract were averaged across the entire tract to produce a single number per census tract.
Overall SSLW: This method calculates the value of SSLW per Census Tract	1) The total SSLW value of every COC would be calculated
Pounds of SSLW/Square Mile: This method calculates pounds of SSLW per square mile per Census Tract	1) The total SSLW value of every State COC located within each census tract in the state was calculated 2) This total number was divided by square miles per census tract 3) The 5 counties with the highest values of SSLW per square mile were included in the analysis, separated by type of animal.

3 Environmental Justice Analysis

Environmental justice (EJ) is the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income, with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies (US EPA). The primary goal of this Draft EJ Report is to encourage comments and suggestions from the surrounding community, industry, and environmental groups throughout the comment period. Public comments will be considered throughout the remainder of the comment period to inform the Final EJ Report.

The following components are included within this Draft EJ Report:

- Potentially Underserved Communities as defined by NCDEQ
- Existing locations of biogas digester permits
- Existing general permits SSLW distribution
- Comparison of local area demographics to the county and statewide census data (from the US Census; https://data.census.gov/cedsci/advanced)
- Limited English Proficiency
- Tribal Counties
- County Health Rankings

3.1 Potentially Underserved Communities

NCDEQ has selected specific block groups across the state that meet a certain threshold for both race and ethnicity and poverty when compared to the County and State percentages. This is the only portion of the analysis that is on the block group level. Block groups are statistical divisions of census tracts, are generally defined to contain between 600 and 3,000 people and are used to present data and control block numbering. A block group consists of clusters of blocks within the same census tract that have the same first digit of their four-digit census block number (US Census Bureau).

NCDEQ defines potentially Underserved Communities by examining the Race/Ethnicity and Poverty criteria of each block group. The block group is then compared to both the County and the State and selected as a potentially underserved block group if it meets the following criteria for Race/Ethnicity and Poverty:

Racial/Ethnic composition:
Share of nonwhites and Hispanic or Latino (of any race) is over fifty percent **OR**
Share of nonwhites and Hispanic or Latino (of any race) is at least ten percent higher than County or State share.
AND
Poverty rate:
Share of population experiencing poverty is over twenty percent **AND**
Share of households in poverty is at least five percent higher than the County or State share.

Approximately 25% of North Carolina's block groups meet this definition of potentially underserved.

This dataset is a selection of the 2019 ACS data from the data tables B03002—Hispanic or Latino Origin by Race—and S1701—Poverty Status in the Past 12 Months. Learn more about NC DEQ's Potentially Underserved Block Groups 2019 - Overview (arcgis.com).

3.2 Existing biogas permits

The NCDEQ has issued 17 individual permits to date for biogas digesters. These are located across 7 counties in North Carolina:

- Bladen
- Duplin
- Harnett
- Sampson
- Johnston
- Wayne
- Yadkin

Of the existing 17 permits, 4 are located within NCDEQ selected potentially underserved block groups (Figure 1).

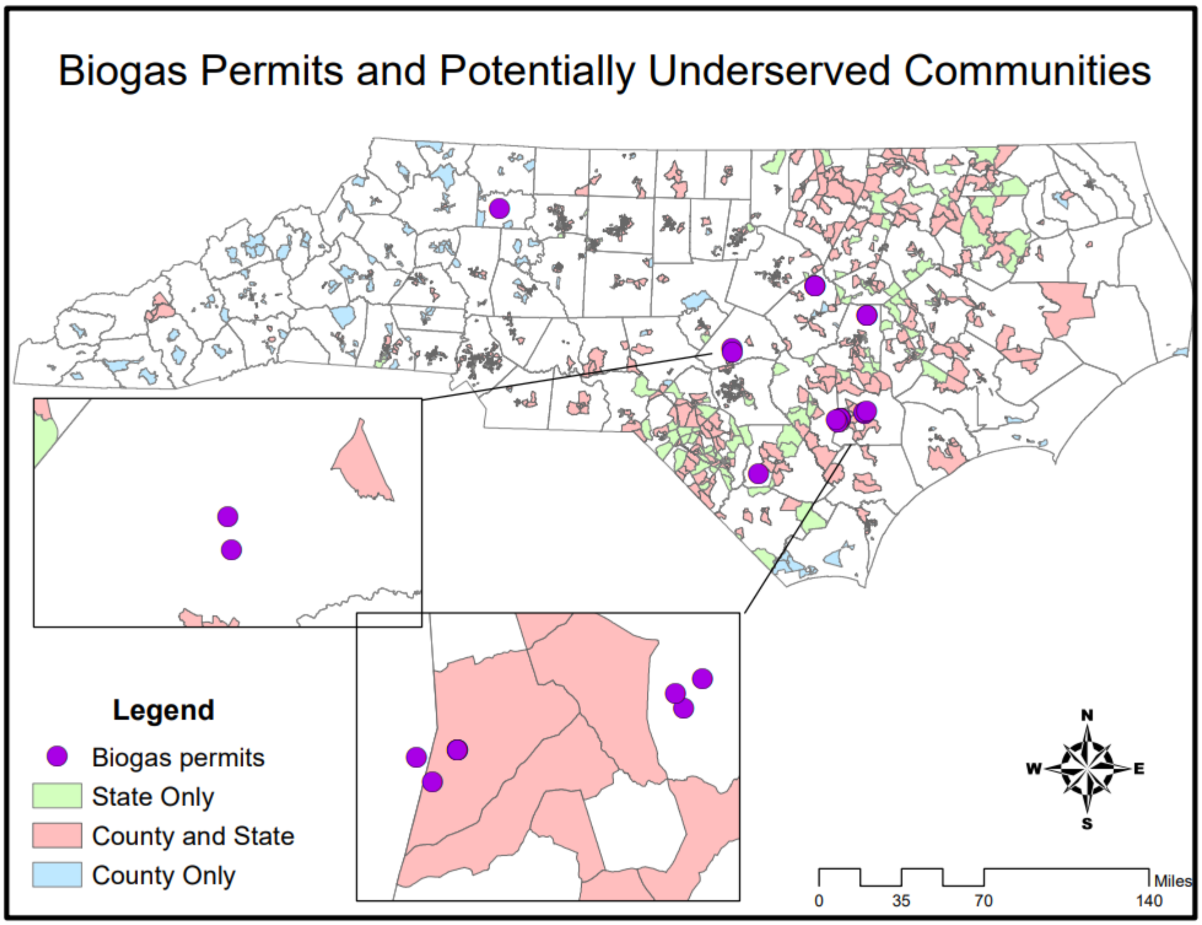


Figure 1. Existing biogas digester permits in North Carolina overlaid with the potentially underserved block group selection.

4 Swine

Across North Carolina, there are 2,161 swine permits covered under the 2019 general permit. The following table (Table 2) outlines the 5 counties in North Carolina that have the highest amount of SSLW per square mile for Swine. Two portions of the analysis are included below. The first portion includes the entire state overlaying the SSLW at the block group level for potentially underserved communities and at the census tract level with Limited English Proficiency for Spanish speakers. The second portion of the analysis is at the census tract level and includes race and ethnicity, and poverty for the top five counties only. For Swine, the certificates of coverage are located across 60 counties.

Table 2. Counties with the highest swine SSLW per square mile

County	SSLW/Sq mile
Sampson County	6,421,962.46
Duplin County	5,876,928.48
Wayne County	4,169,531.19
Bladen County	2,206,113.71
Robeson County	1,734,305.98

4.1 Potentially Underserved Communities

The following figure (Figure 2) shows the potentially underserved block group selection overlaid with the swine certificates of coverage averaged out to show SSLW per square mile.

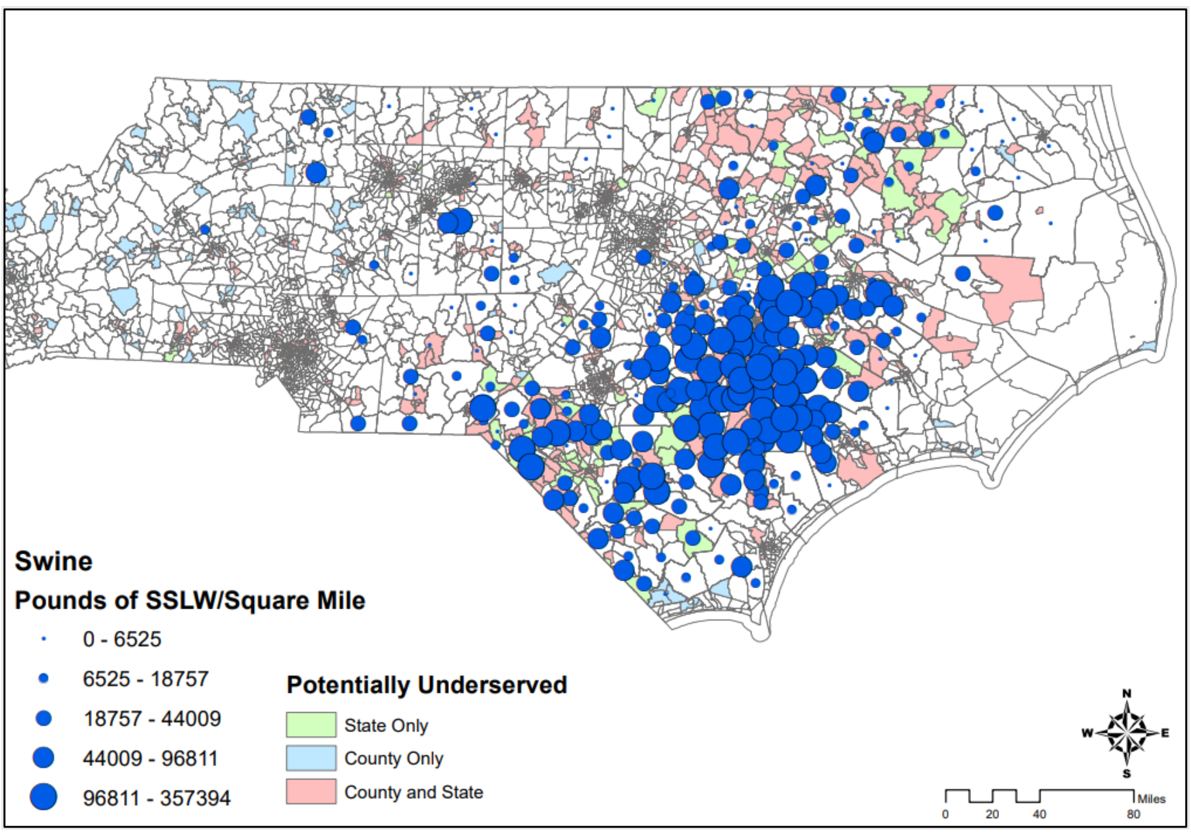


Figure 2. Swine SSLW/Square Mile (census tracts) overlaid with Potentially Underserved Communities (Block Group).

4.2 Limited English Proficiency- Spanish

Per the Safe Harbor Guidelines, should an LEP Group be identified during the permit application process, written translations of vital documents for each eligible LEP language group that constitutes 5% or includes 1,000 members (whichever is less) of the population of persons eligible to be served or likely to be affected or encountered. If there are fewer than 50 persons in a language group that reaches the 5% trigger, then NCDEQ will not translate vital written materials, but instead will provide written notice in the primary language of the LEP language group of the right to receive competent oral interpretation of those written materials, free of cost. The safe harbor provisions apply to the translation of written documents only. Safe harbor guidelines are based on EPA guidance for LEP persons and implemented by NCDEQ when deemed appropriate.

The following figure (Figure 3) shows the census tracts across North Carolina with a population who speaks English less than very well for Spanish greater than 5% and the Swine SSLW per square mile.

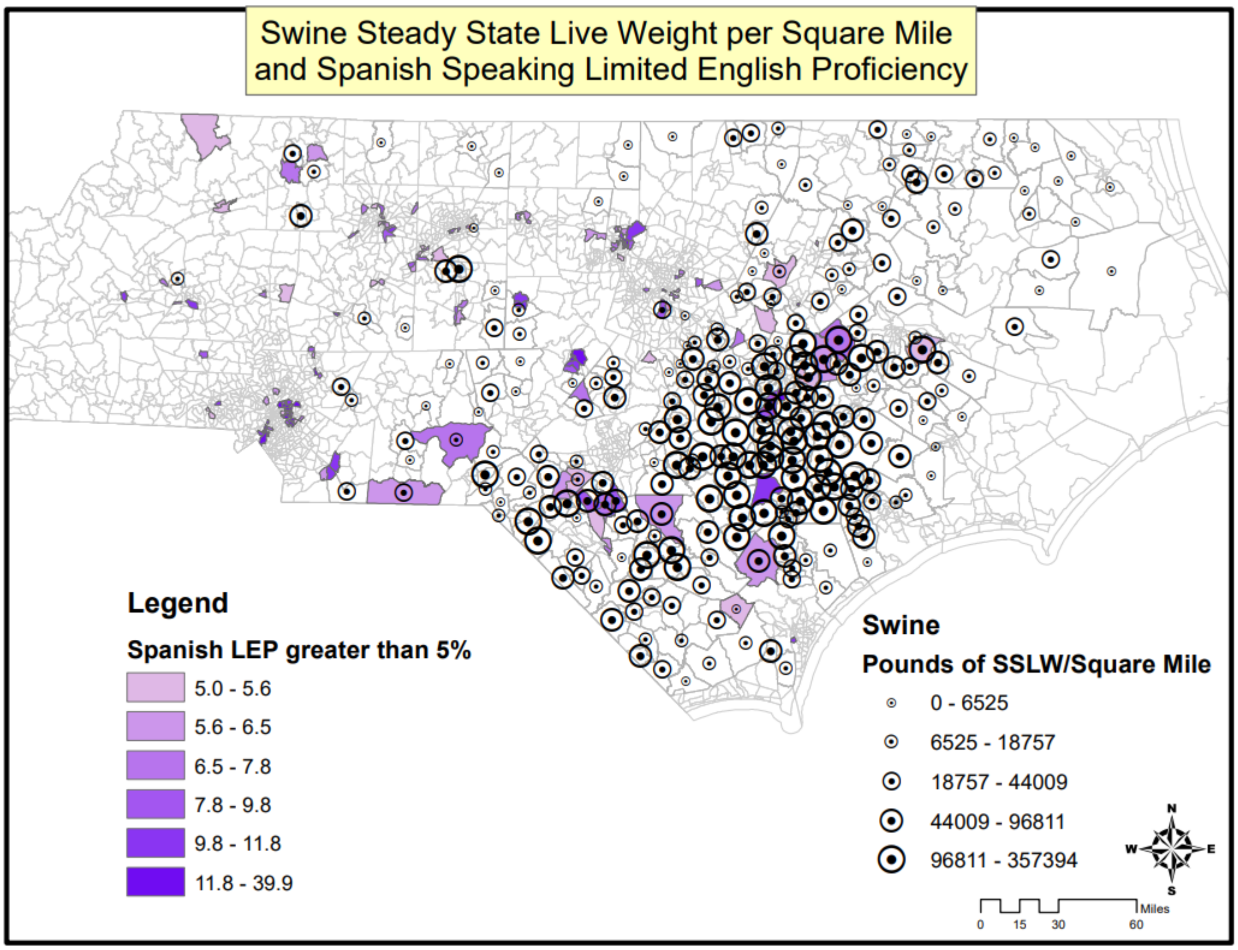


Figure 3. Census tracts with Spanish speaking populations who speak English less than very well and SSLW averages for swine.

4.3 Five County Analysis

Using standard environmental justice guidelines from the EPA and NEPA documentation, the following conditions will be flagged as communities with the potential for having environmental justice concerns:

1. 10% or more in comparison to the county or state average
2. 50% or more minority
3. 5% or more in comparison to the county or state average for poverty

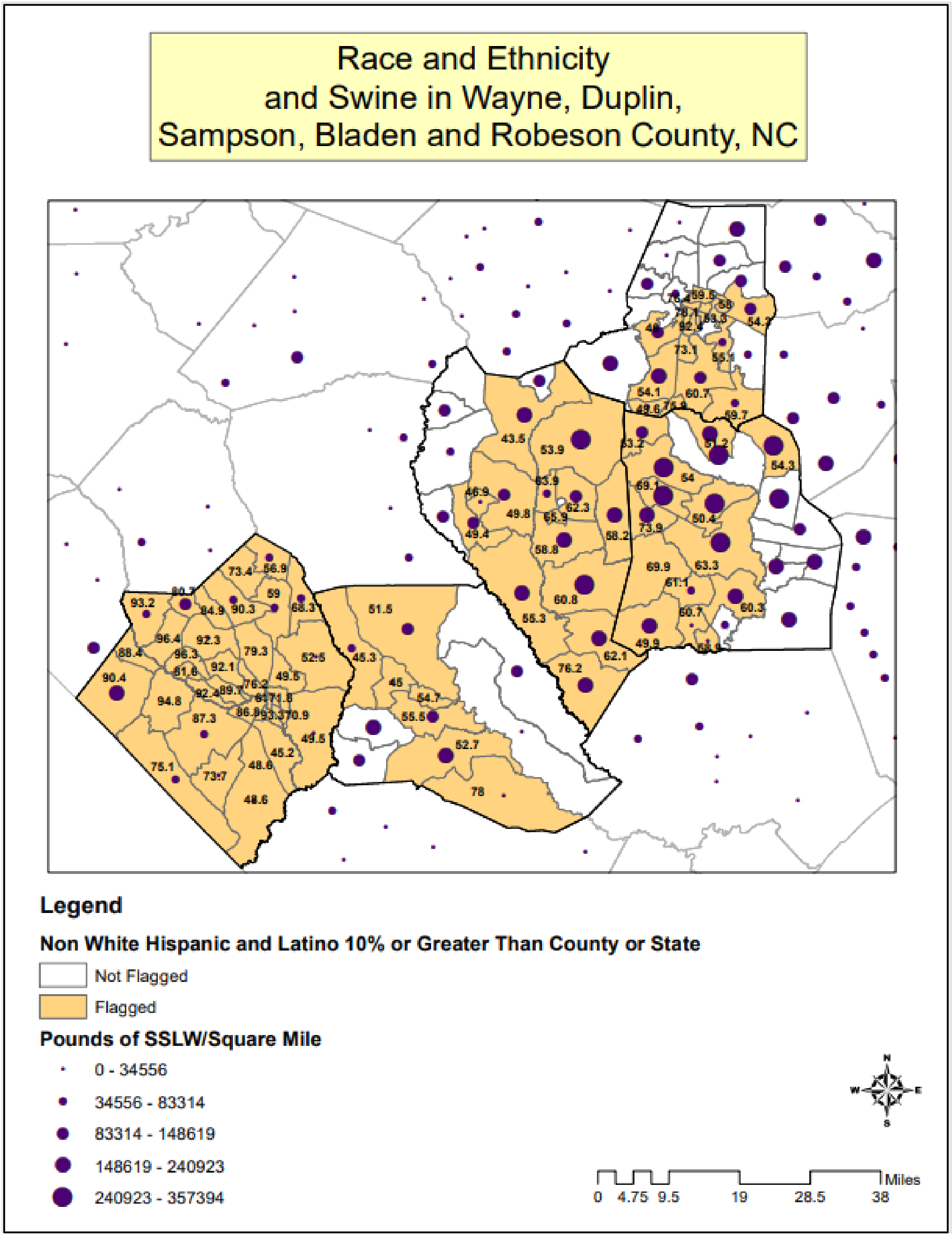
For example, if a census tract has 35% of the population classified as low income but the county consists of 30% low income, the census tract would exceed the county average by 16.7% and thus be flagged as an area with the potential for having EJ concerns. 2020 Census Bureau data is real data gathered every ten years, whereas the estimates from the more recent years are modeled based on the real data.

Race and Ethnicity

The following maps show the top 5 counties as ranked by the SSLW per square mile. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 3 shows the 2020 Census data for the 5 counties and North Carolina.

Table 3. Race and ethnicity for the five counties with the highest SSLW per Square mile for swine and North Carolina

Geography	Total Population	Hispanic or Latino	White	Percent Nonwhite and Hispanic or Latino	Black or African American	American Indian or Alaska Native	Asian	Native Hawaiian or Pacific Islander	Some other Race	Two or more Races
North Carolina	10,439,388	1,118,596	6,312,148	39.5%	2,107,526	100,886	340,059	6,980	46,340	406,853
Bladen County	29,606	2,546	15,830	46.5%	9,505	701	47	8	67	902
Duplin County	48,715	10,813	24,945	48.8%	11,437	154	155	4	120	1,087
Robeson County	116,530	11,757	29,159	75.0%	26,218	43,536	897	63	411	4,489
Sampson County	59,036	12,249	29,729	49.6%	13,944	1002	216	18	156	1,722
Wayne County	117,333	14,927	60,199	48.7%	35,329	335	1,542	71	454	4,476


Race and Ethnicity
and Swine in Wayne, Duplin,
Sampson, Bladen and Robeson County, NC
Legend
Non White Hispanic and Latino 10% or Greater Than County or State
Not Flagged
Flagged
Pounds of SSLW/Square Mile
0 - 34556
34556 - 83314
83314 - 148619
148619 - 240923
240923 - 357394
Miles
0 4.75 9.5 19 28.5 38
N
S
W
E

Poverty

The following maps show the top 5 counties as ranked by the SSLW per square mile. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 4 shows the 2020 Census data for the 5 counties and North Carolina.

Table 4. Poverty levels for the five counties with the highest SSLW per Square mile for Swine and North Carolina

Geography	Total Population	MOE +/-	Percent below Poverty	MOE +/-
North Carolina	9,984,891	1,988	14.7%	0.2%
Bladen County	32,918	214	24.3%	3.2%
Duplin County	58,374	229	21.2%	2.6%
Robeson County	128,917	494	27.7%	1.1%
Sampson County	62,511	272	20.9%	2.5%
Wayne County	120,420	494	20.2%	1.2%

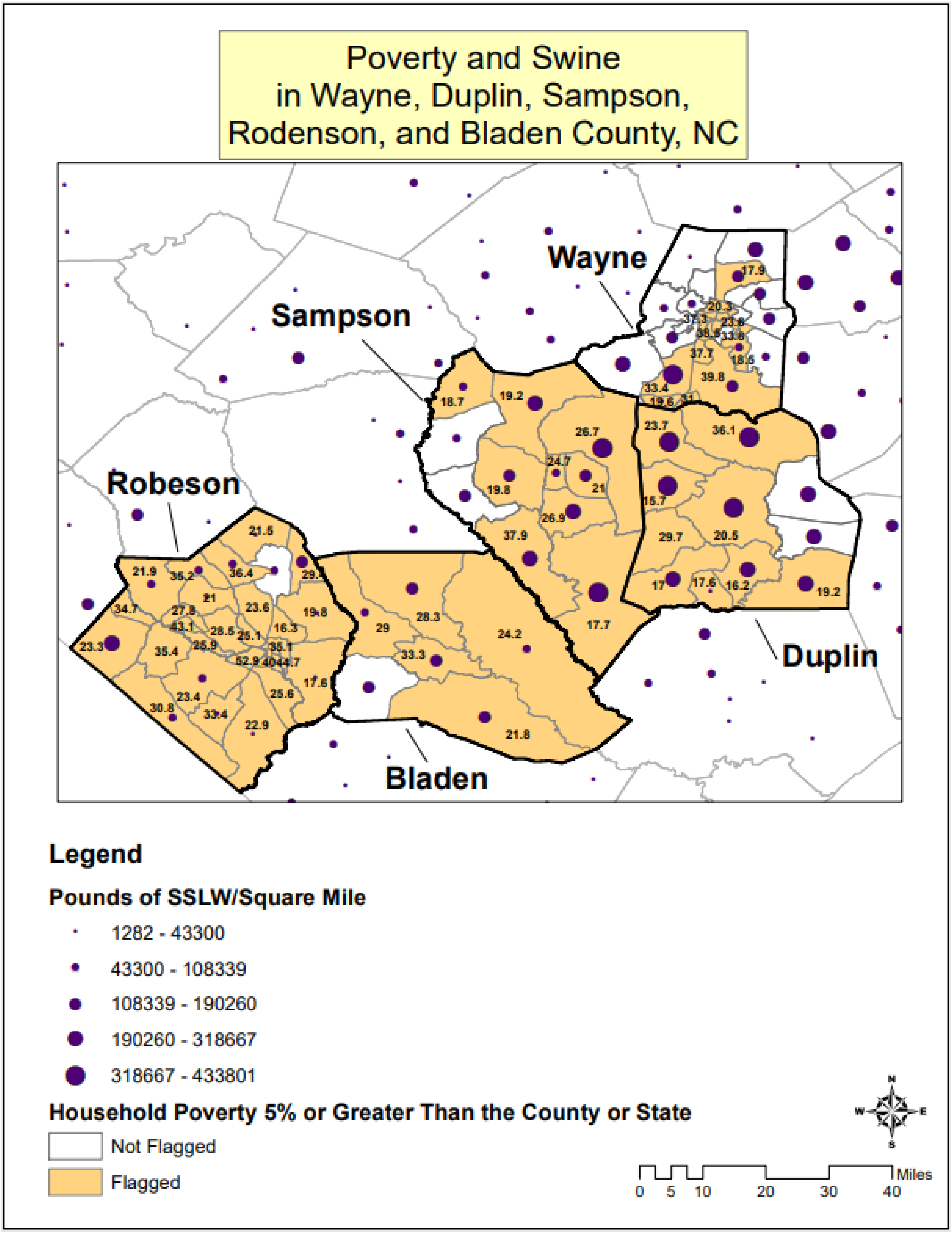


Figure 5. Map for the five counties and flagged census tracts for poverty.

5 Cattle

Across North Carolina, there are 222 cattle permits covered under the 2019 general permit. The following table (Table 5) outlines the 5 counties in North Carolina that have the highest amount of SSLW per square mile for cattle. Two portions of the analysis are included below. The first portion includes the entire state overlaying the SSLW at the block group level for potentially underserved communities and at the census tract level with Limited English Proficiency for Spanish speakers. The second portion of the analysis is at the census tract level and includes race and ethnicity, and poverty for the top five counties only. For cattle, the certificates of coverage are located across 34 counties.

Table 5. Counties with the highest Cattle SSLW per square mile

County	SSLW/Sq mile
Iredell County	1,409,478.90
Randolph County	919,199.30
Lincoln County	431,896.80
Davidson County	408,691.25
Gaston County	351,827.30

5.1 Potentially Underserved Communities

The following figure (Figure 6) shows the potentially underserved block group selection overlaid with the cattle certificates of coverage averaged out to show SSLW per square mile.

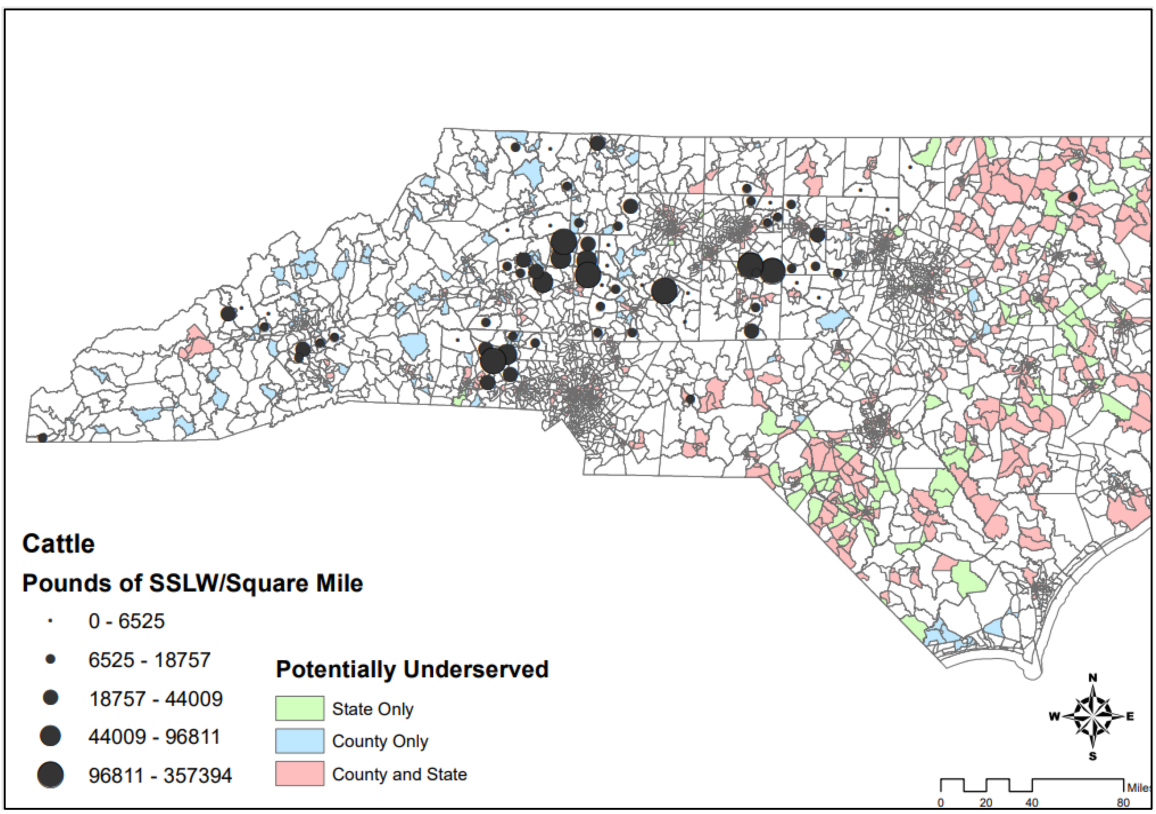


Figure 6. Cattle SSLW/Square Mile (census tracts) overlaid with Potentially Underserved Communities (Block Group).

5.2 Limited English Proficiency- Spanish

Per the Safe Harbor Guidelines, should an LEP Group be identified during the permit application process, written translations of vital documents for each eligible LEP language group that constitutes 5% or includes 1,000 members (whichever is less) of the population of persons eligible to be served or likely to be affected or encountered. If there are fewer than 50 persons in a language group that reaches the 5% trigger, then NCDEQ will not translate vital written materials, but instead will provide written notice in the primary language of the LEP language group of the right to receive competent oral interpretation of those written materials, free of cost. The safe harbor provisions apply to the translation of written documents only. Safe harbor guidelines are based on EPA guidance for LEP persons and implemented by NCDEQ when deemed appropriate.

The following figure (Figure 7) shows the census tracts across North Carolina with a population who speaks English less than very well for Spanish greater than 5% and the cattle SSLW per square mile.

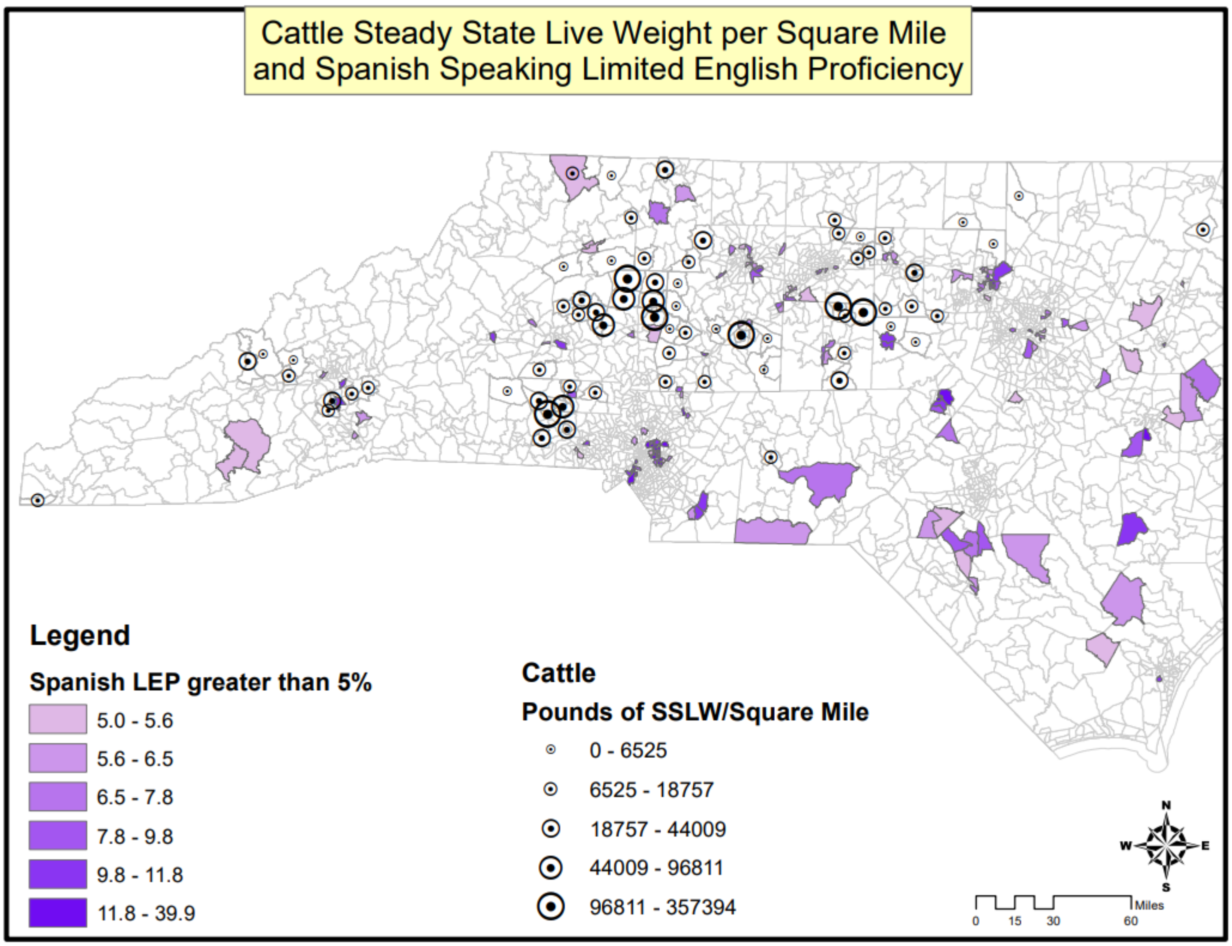


Figure 7. Census Tracts with Spanish speaking populations who speak English less than very well and SSLW averages for cattle.

5.3 Five County Analysis

Using standard environmental justice guidelines from the EPA and NEPA documentation, the following conditions will be flagged as communities with the potential for having environmental justice concerns:

2. 10% or more in comparison to the county or state average
3. 50% or more minority
4. 5% or more in comparison to the county or state average for poverty

For example, if a census tract has 35% of the population classified as low income but the county consists of 30% low income, the census tract would exceed the county average by 16.7% and thus be flagged as an area with the potential for having EJ concerns. 2020 Census Bureau data is real data gathered every ten years, whereas the estimates from the more recent years are modeled based on the real data.

Race and Ethnicity

The following maps show the top 5 counties as ranked by the SSLW per square mile for cattle. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 6 shows the 2020 Census data for the 5 counties and North Carolina.

Table 6. Race and ethnicity for the five counties with the highest SSLW per Square mile for cattle and North Carolina

Geography	Total population	Hispanic or Latino	White	Percent Non-white and Hispanic or Latino	Black or African American	American Indian and Alaska Native	Asian	Native Hawaiian and Pacific Islander	Some other Race	Two or more Races
North Carolina	10,439,388	1,118,596	6,312,148	39.54%	2,107,526	100,886	340,059	6,980	46,340	406,853
Davidson County	168,930	13,902	129,487	23.35%	15,839	665	2,440	43	491	6,063
Gaston County	227,943	20,068	153,653	32.59%	39,762	753	3,509	59	844	9,295
Iredell County	186,693	15,777	136,393	26.94%	21,255	437	4,718	58	656	7,399
Lincoln County	86,810	6,412	71,661	17.45%	4,405	237	692	15	208	3,180
Randolph County	144,171	19,051	108,354	24.84%	8,592	666	2,158	10	412	4,928

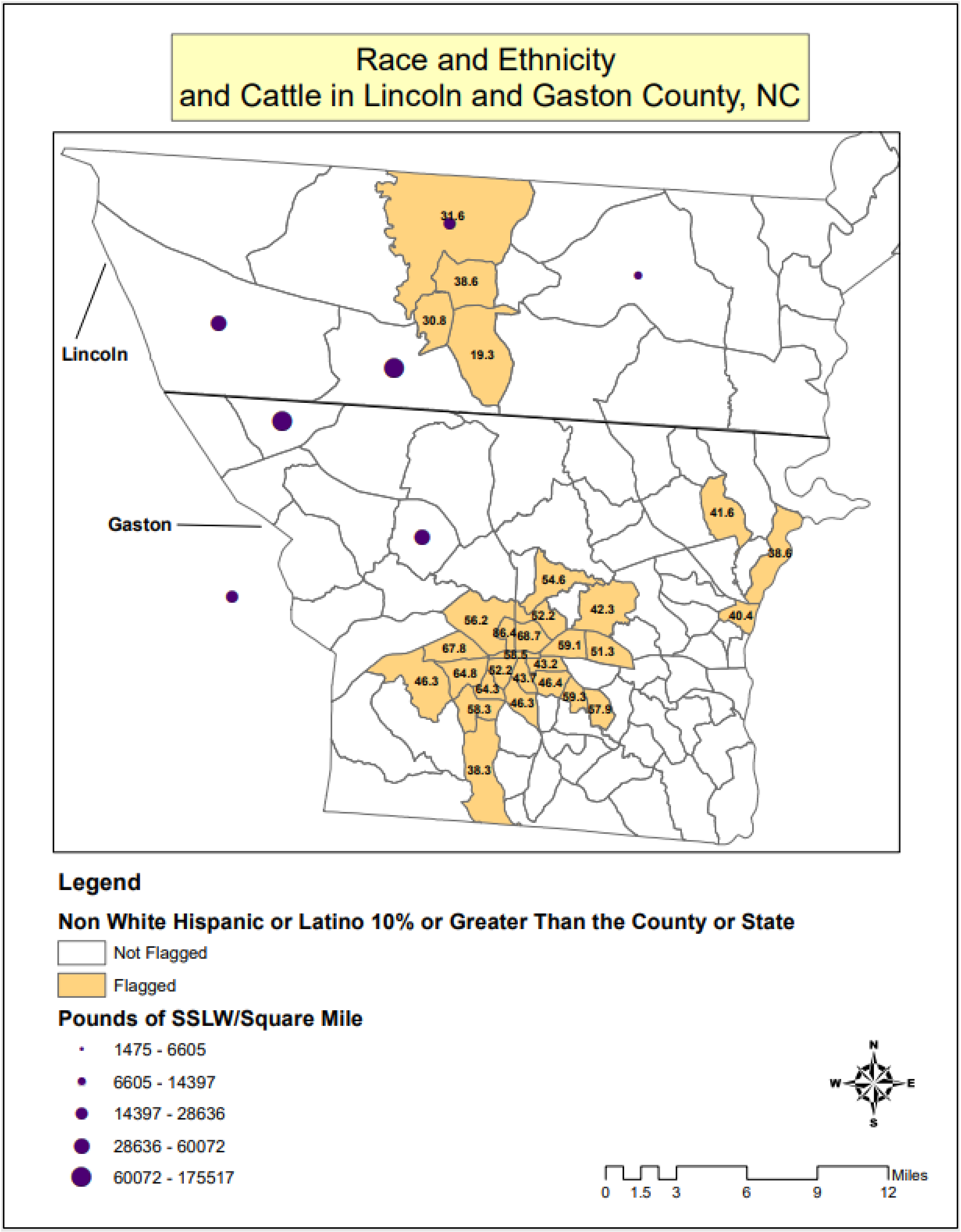


Figure 8. Map of Lincoln and Gaston Counties flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

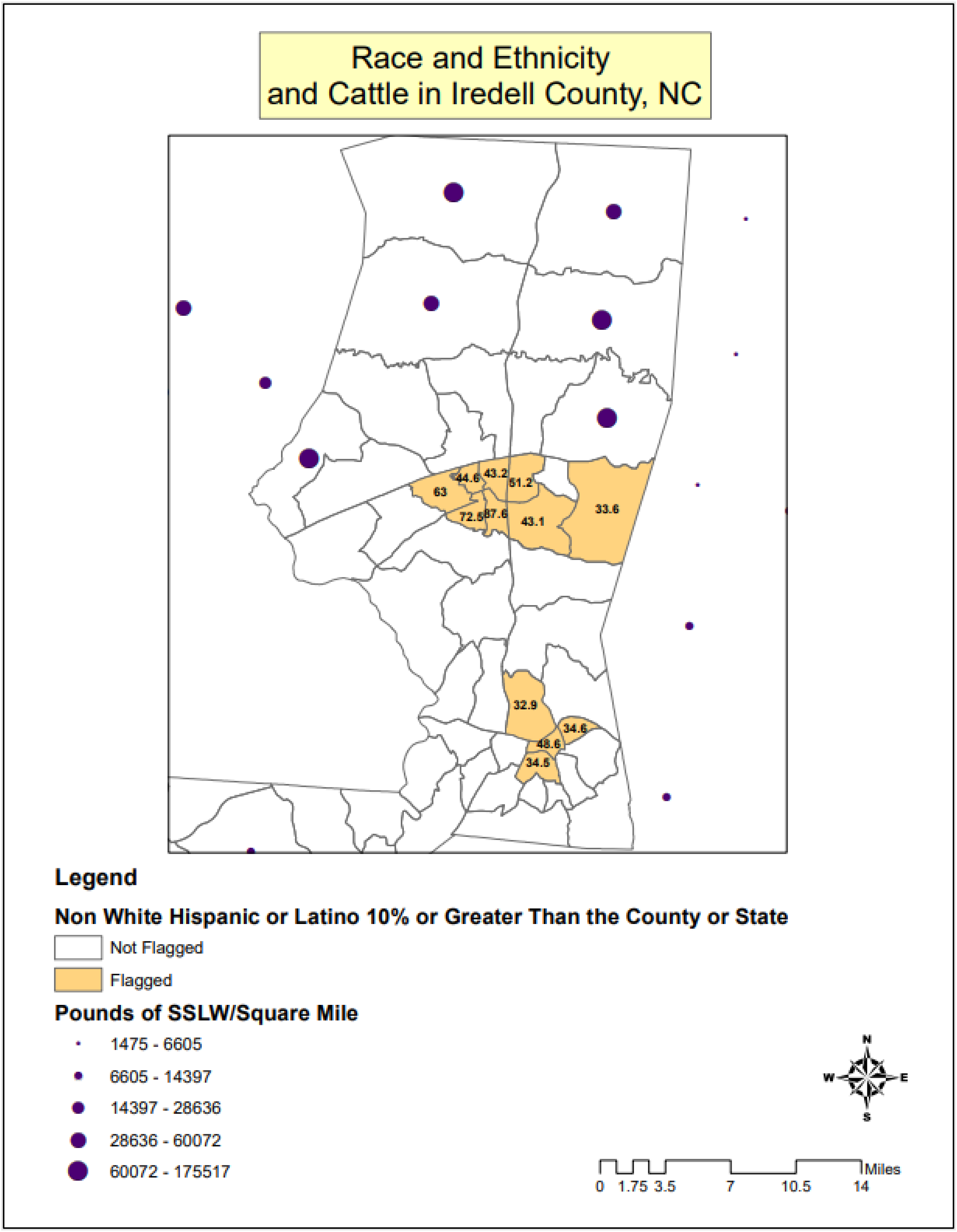


Figure 9. Map of Iredell County flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

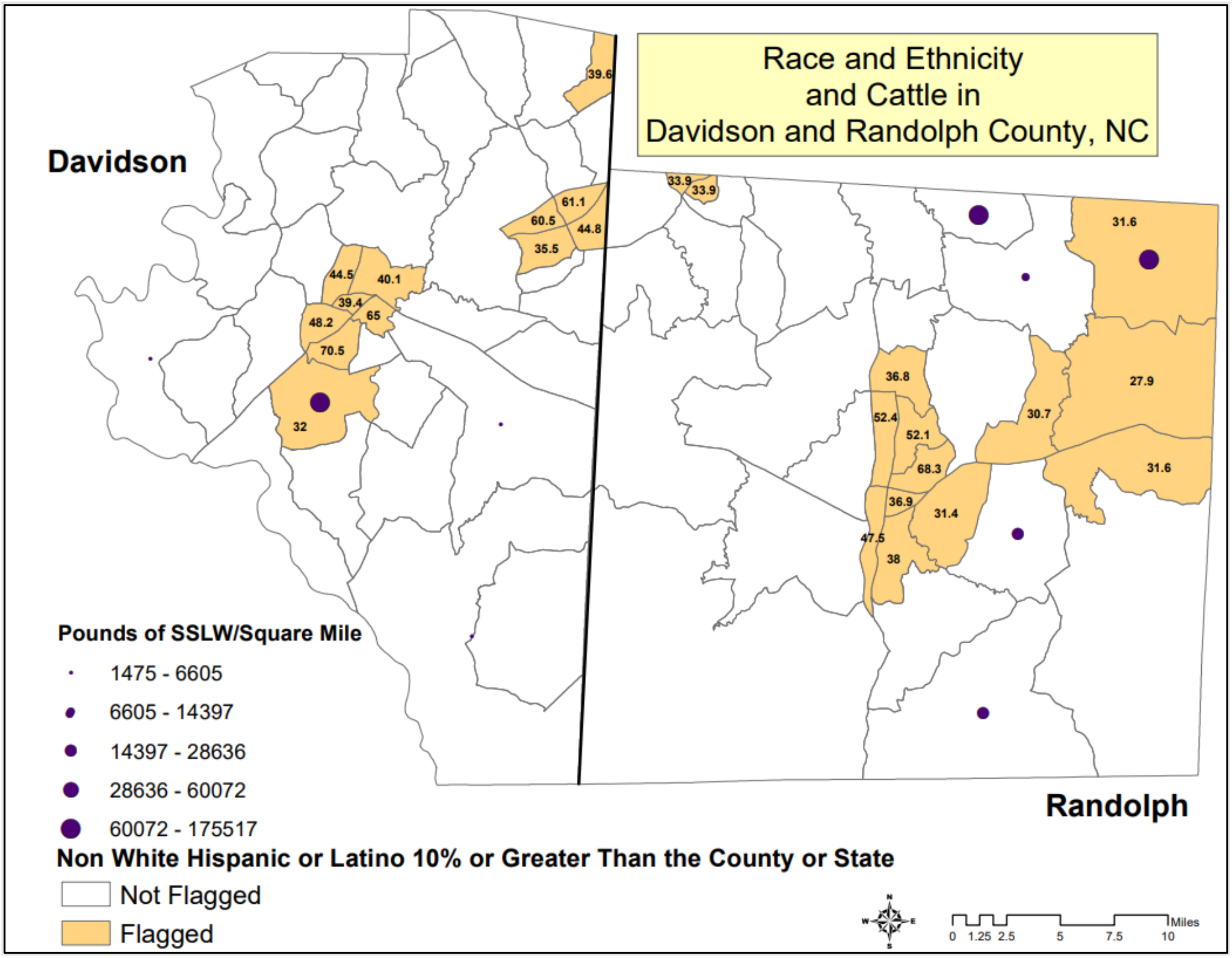


Figure 10. Map of Davidson and Randolph Counties flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

Poverty

The following maps show the top 5 counties as ranked by the SSLW per square mile. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 7 shows the 2020 Census data for the 5 counties and North Carolina.

Table 7. Poverty levels for the five counties with the highest SSLW per Square mile for cattle and North Carolina

Geography	Total Population	MOE +/-	Percent below Poverty	MOE +/-
North Carolina	9,984,891	1,988	14.7%	0.2%
Davidson County	162926	490	15.4%	1.2%
Gaston County	215978	399	14.5%	0.9%
Iredell County	173761	316	10.9%	0.9%
Lincoln County	82082	211	12.1%	1.4%
Randolph County	141274	345	15.2%	1.3%

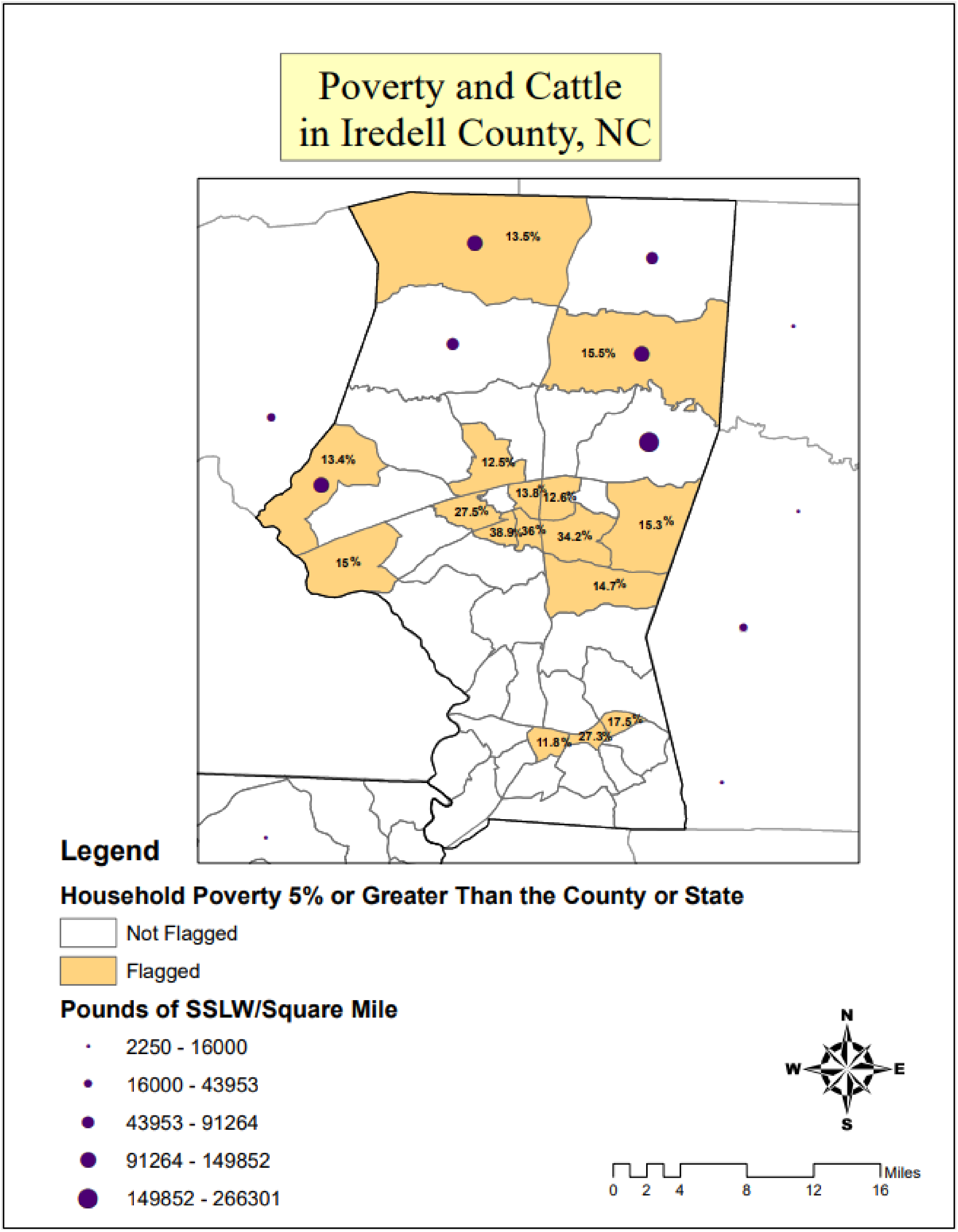


Figure 11. Map for Iredell County and flagged census tracts for poverty.

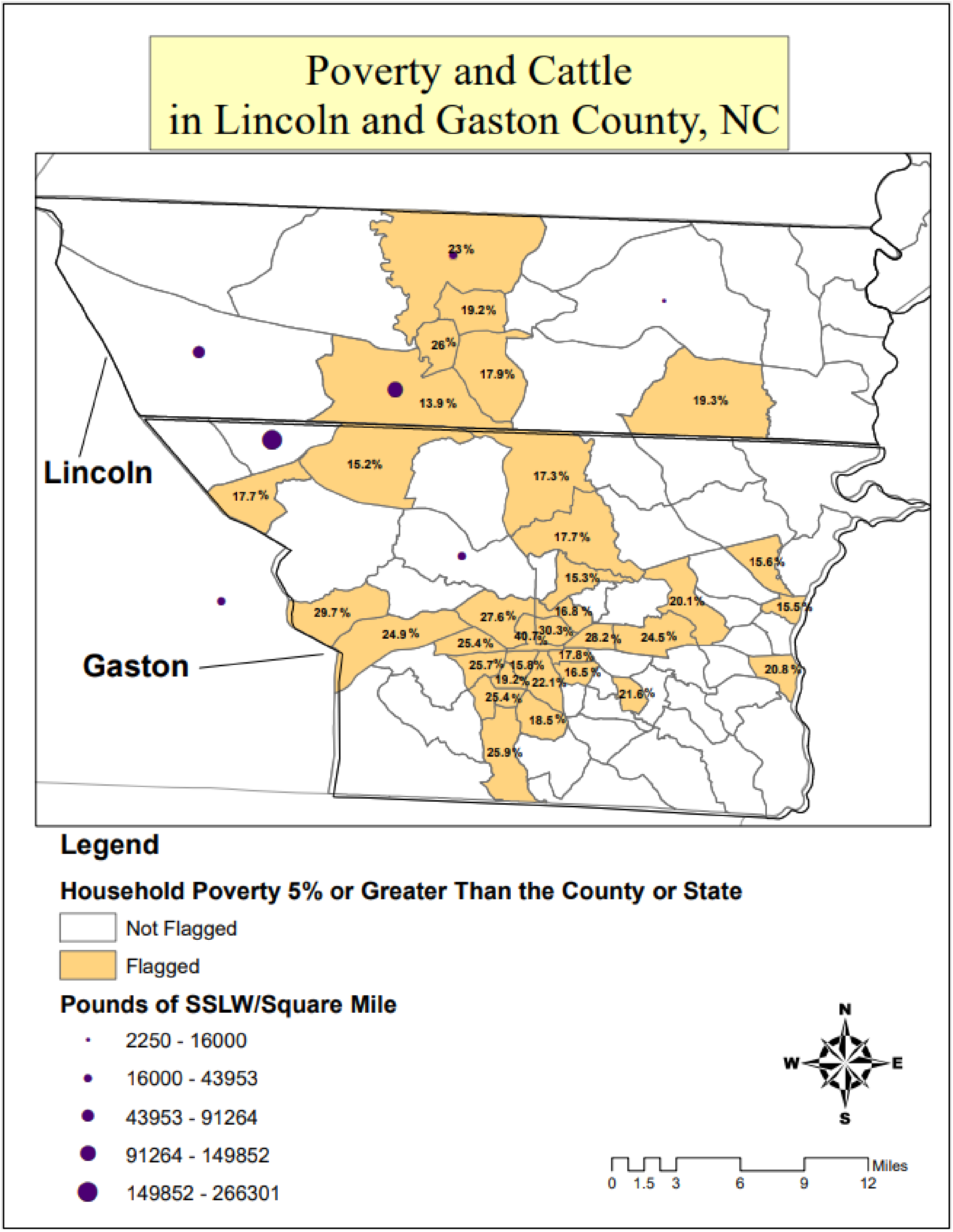


Figure 12. Map for Lincoln and Gaston Counties and flagged census tracts for poverty.

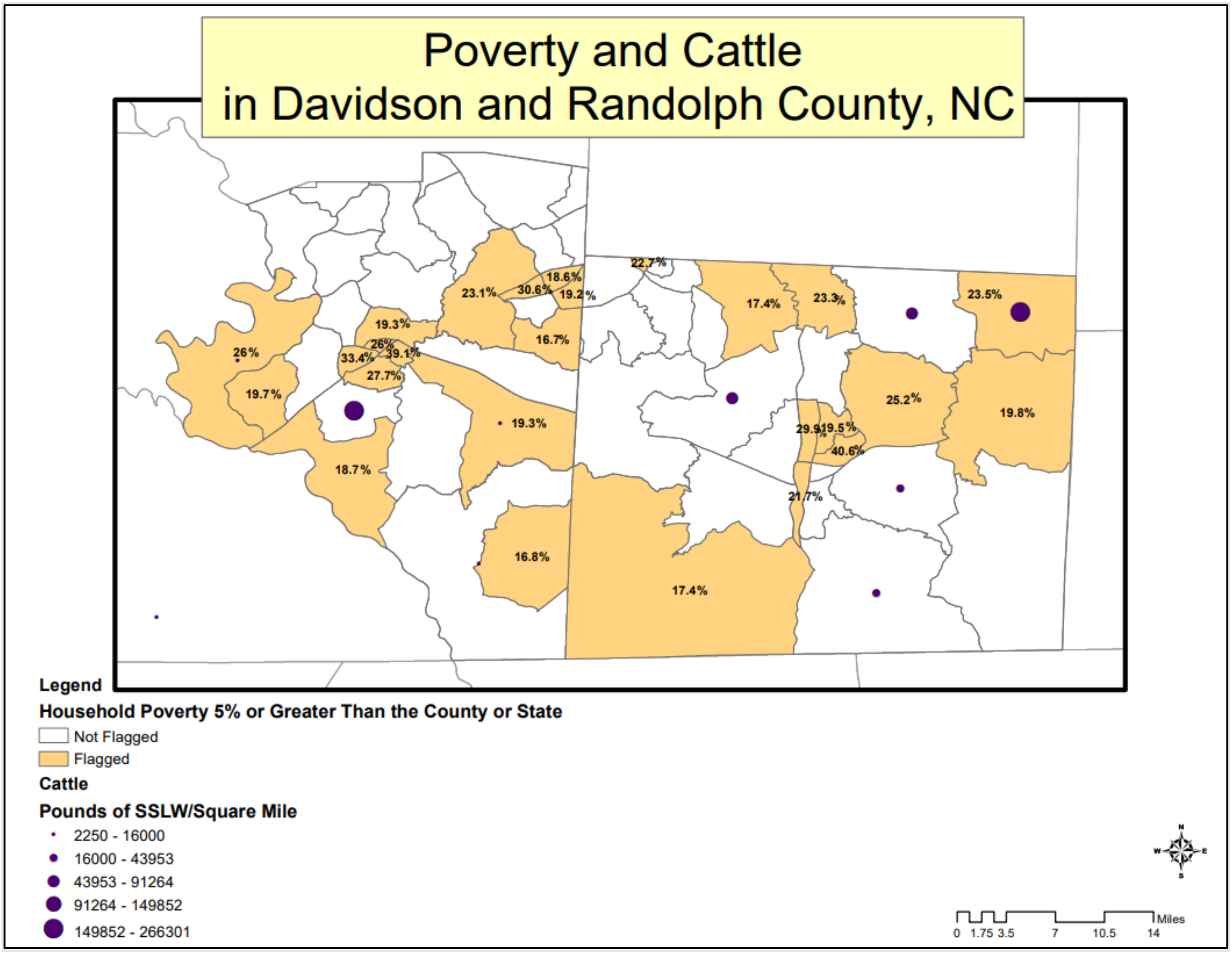


Figure 13. Map for Davidson and Randolph Counties and flagged census tracts for poverty.

6 Wet Poultry

Across North Carolina, there are 9 wet poultry permits covered under the 2019 general permit. The following table (Table 8) outlines the 5 counties in North Carolina that have the highest amount of SSLW per square mile for Wet Poultry. Two portions of the analysis are included below. The first portion includes the entire state overlaying the SSLW at the block group level for potentially underserved communities and at the census tract level with Limited English Proficiency for Spanish speakers. The second portion of the analysis is at the census tract level and includes race and ethnicity, and poverty for the top five counties only. For poultry, the certificates of coverage are located across 5 counties.

Table 8. SSLW per Square Mile: top 5 counties for wet poultry

County	SSLW /sq Mile
Union County	119,149.00
Hyde County	34,756.40
Nash County	24,577.80
Orange County	8,571.43
Halifax County	4,528.30

6.1 Potentially Underserved Communities

The following figure (Figure 14) shows the potentially underserved block group selection overlaid with the wet poultry certificates of coverage averaged out to show SSLW per square mile.

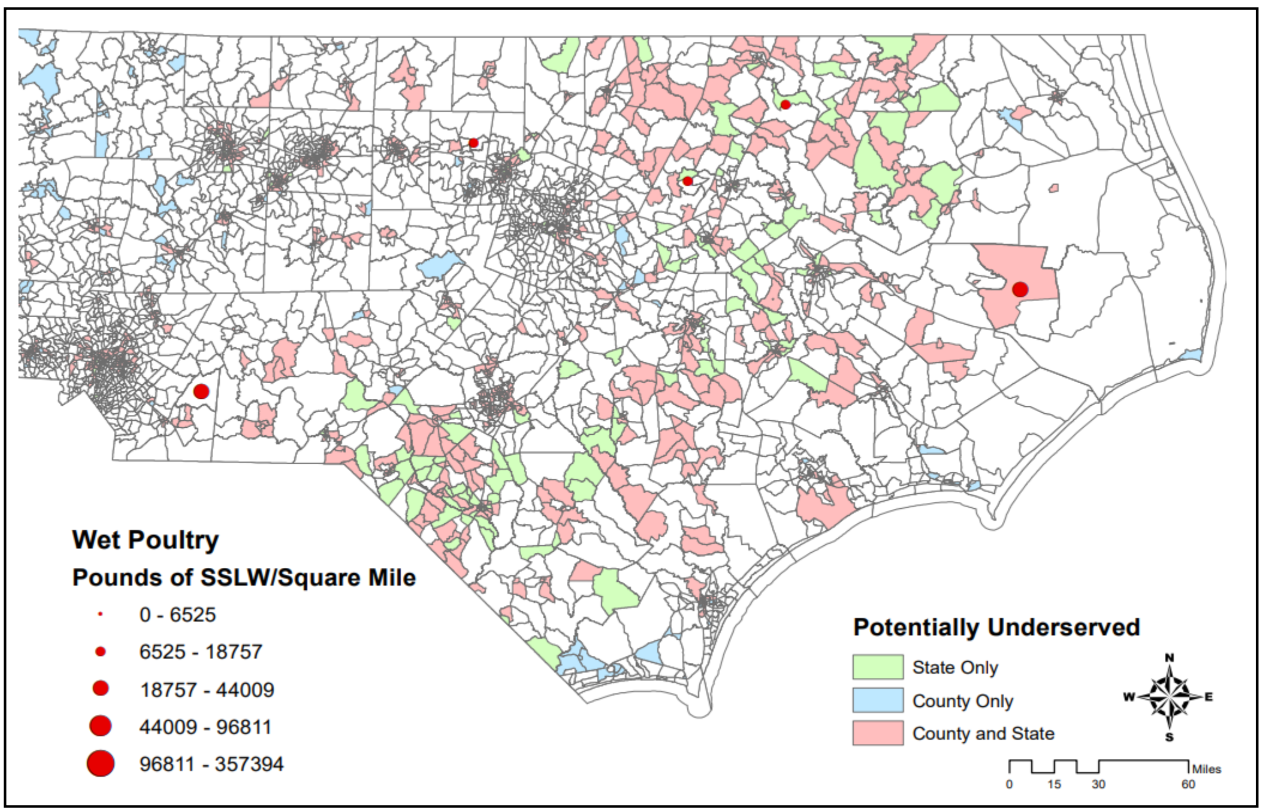


Figure 14. Wet poultry SSLW/Square Mile (census tracts) overlaid with Potentially Underserved Communities (Block Group).

6.2 Limited English Proficiency- Spanish

Per the Safe Harbor Guidelines, should an LEP Group be identified during the permit application process, written translations of vital documents for each eligible LEP language group that constitutes 5% or includes 1,000 members (whichever is less) of the population of persons eligible to be served or likely to be affected or encountered. If there are fewer than 50 persons in a language group that reaches the 5% trigger, then NCDEQ will not translate vital written materials, but instead will provide written notice in the primary language of the LEP language group of the right to receive competent oral interpretation of those written materials, free of cost. The safe harbor provisions apply to the translation of written documents only. Safe harbor guidelines are based on EPA guidance for LEP persons and implemented by NCDEQ when deemed appropriate.

The following figure (Figure 15) shows the census tracts across North Carolina with a population who speaks English less than very well for Spanish greater than 5%.

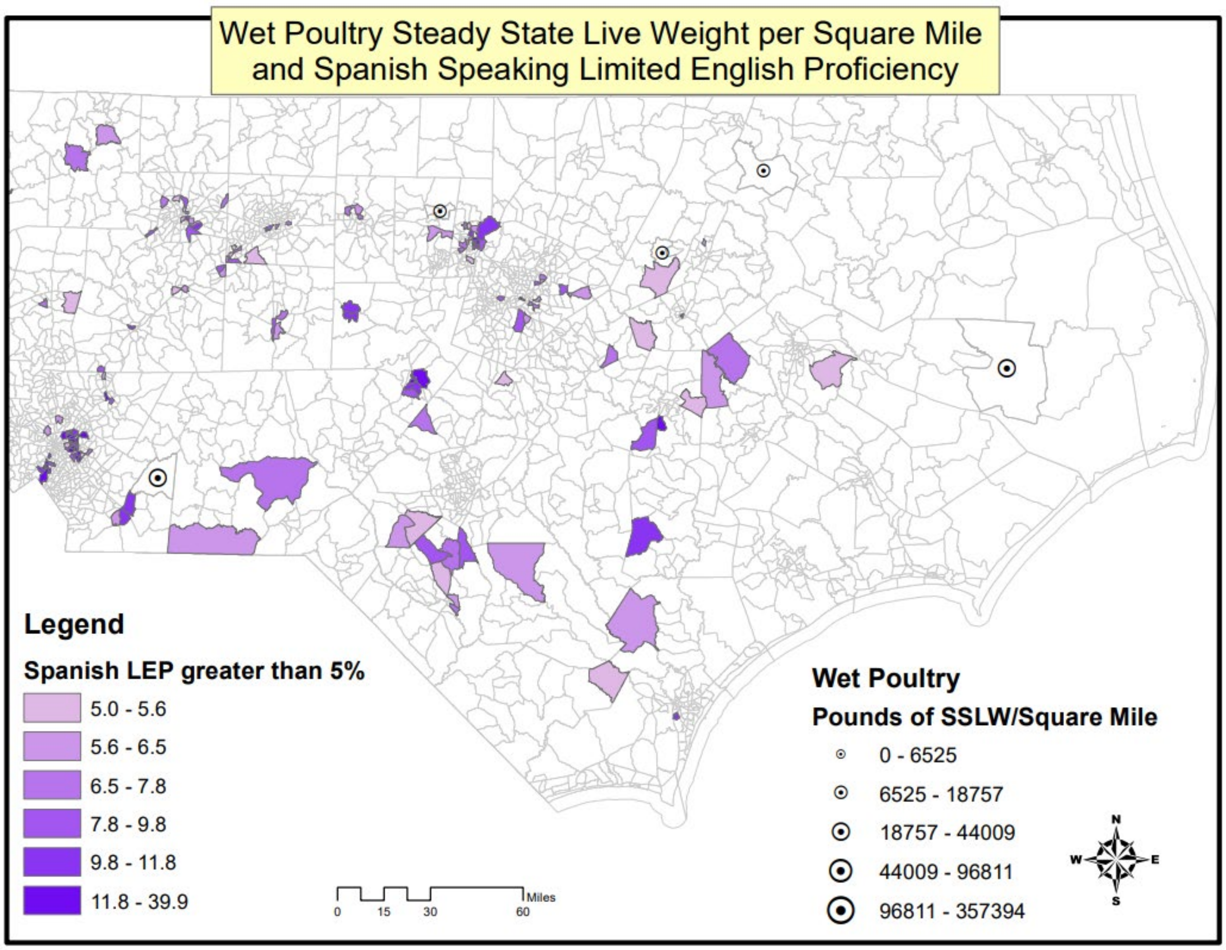


Figure 15. Census tracts with Spanish speaking populations who speak English less than very well and SSLW averages for wet poultry.

6.3 Five County Analysis

Using standard environmental justice guidelines from the EPA and NEPA documentation, the following conditions will be flagged as communities with the potential for having environmental justice concerns:

3. 10% or more in comparison to the county or state average
4. 50% or more minority
5. 5% or more in comparison to the county or state average for poverty

For example, if a census tract has 35% of the population classified as low income but the county consists of 30% low income, the census tract would exceed the county average by 16.7% and thus be flagged as an area with the potential for having EJ concerns. 2020 Census Bureau data is real data gathered every ten years, whereas the estimates from the more recent years are modeled based on the real data.

Race and Ethnicity

The following maps show the top 5 counties as ranked by the SSLW per square mile. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 9 shows the 2020 Census data for the 5 counties and North Carolina.

Table 9. Race and Ethnicity for the five counties with the highest SSLW per Square mile for Poultry and North Carolina.

Geography	Total Population	Hispanic or Latino	White	Percent non-White and Hispanic or Latino	Black or African American	American Indian and Alaska Native	Asian	Native Hawaiian and Pacific Islander	Some other race	Two or more races
North Carolina	10,439,388	1,118,596	6,312,148	39.5%	2,107,526	100,886	340,059	6,980	46,340	406,853
Halifax County	48,622	1454	19,070	60.8%	24737	1593	281	11	142	1334
Hyde County	4,589	347	2,928	36.2%	1152	7	7	2	15	131
Nash County	94,970	7322	46,317	51.2%	36679	615	904	28	407	2698
Orange County	148,696	15812	96,537	35.1%	15571	334	12615	43	798	6986
Union County	23,8267	30110	161,113	32.4%	26500	641	9516	90	1199	9098

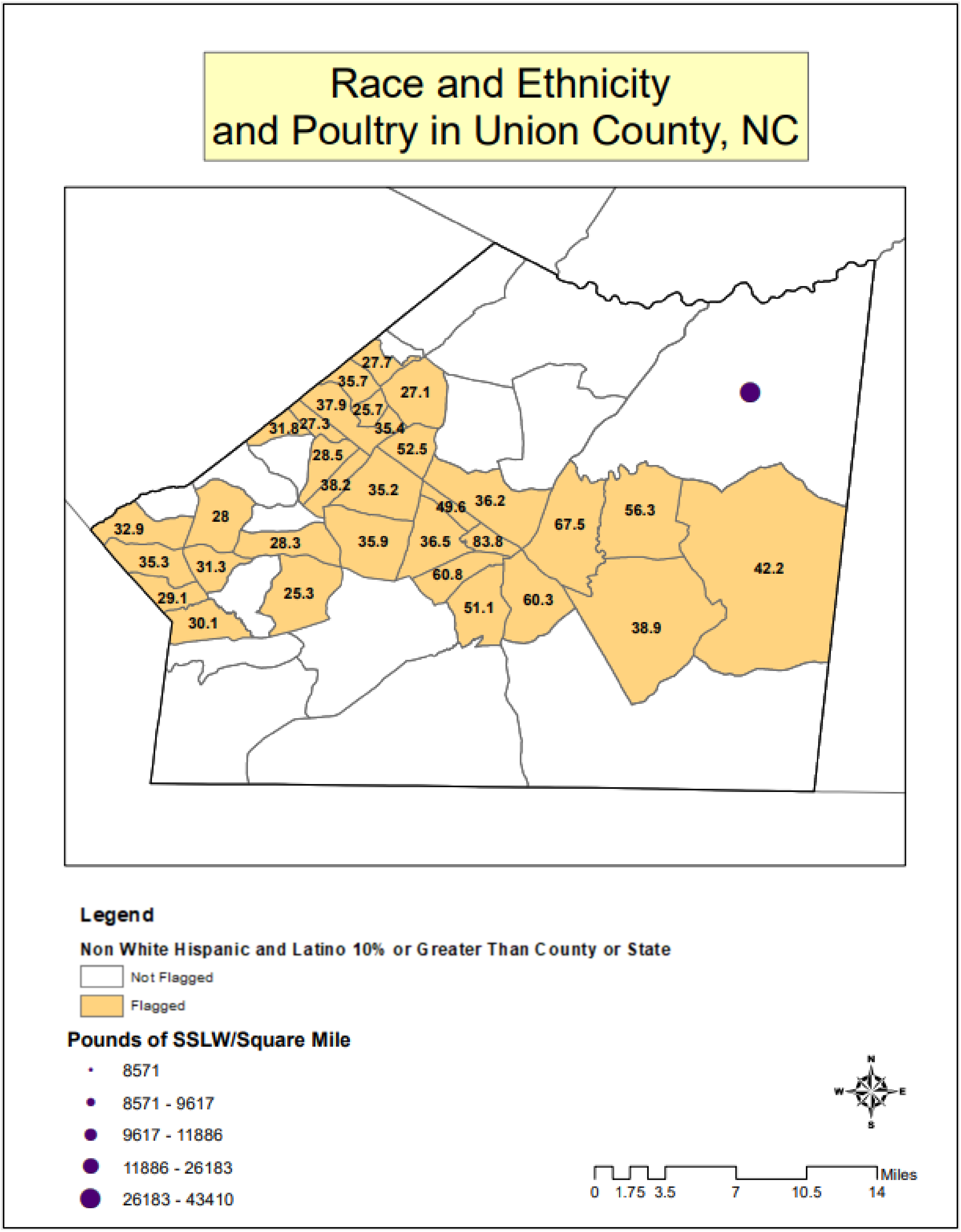


Figure 16. Map of Union County flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

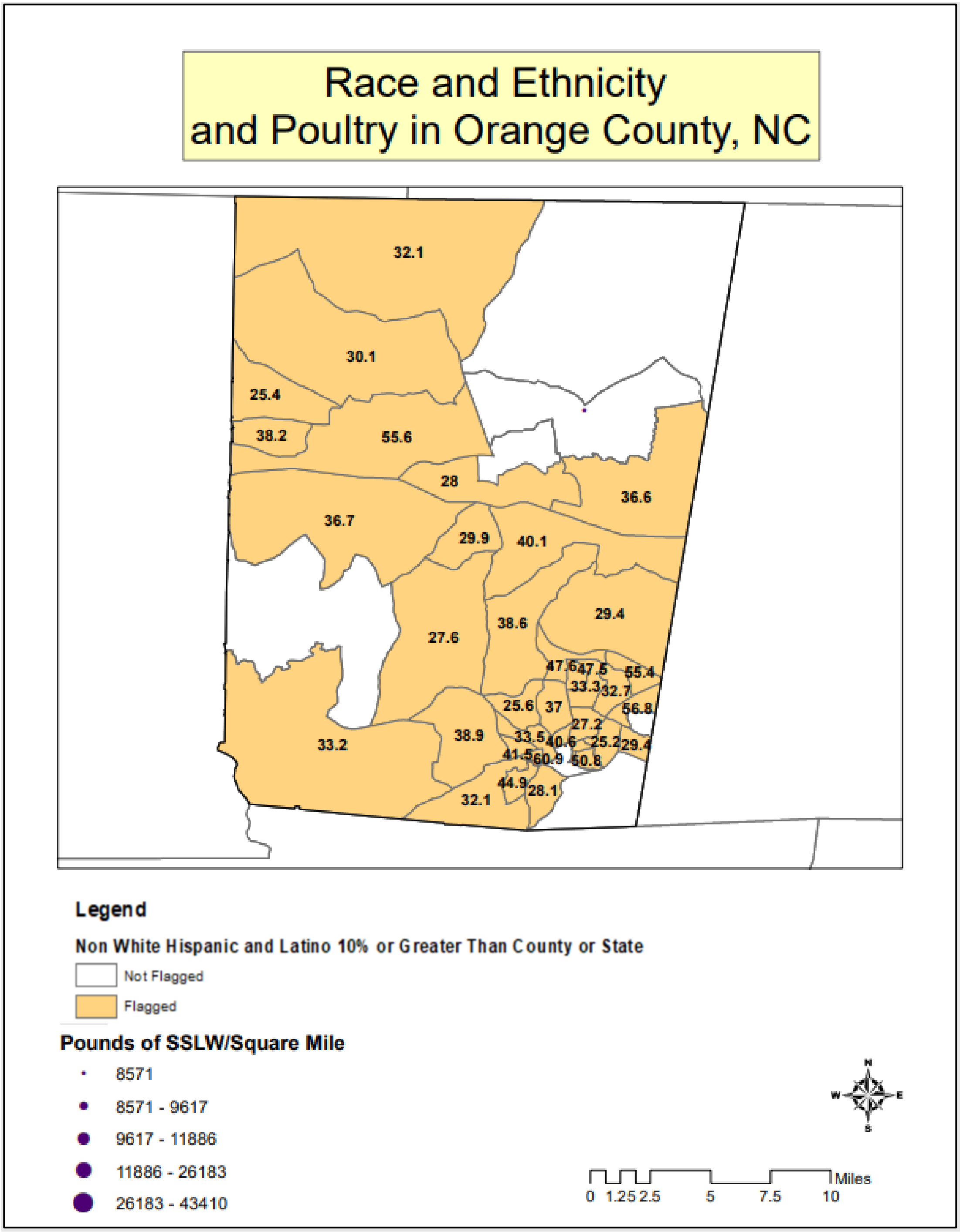


Figure 17. Map of Orange County flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

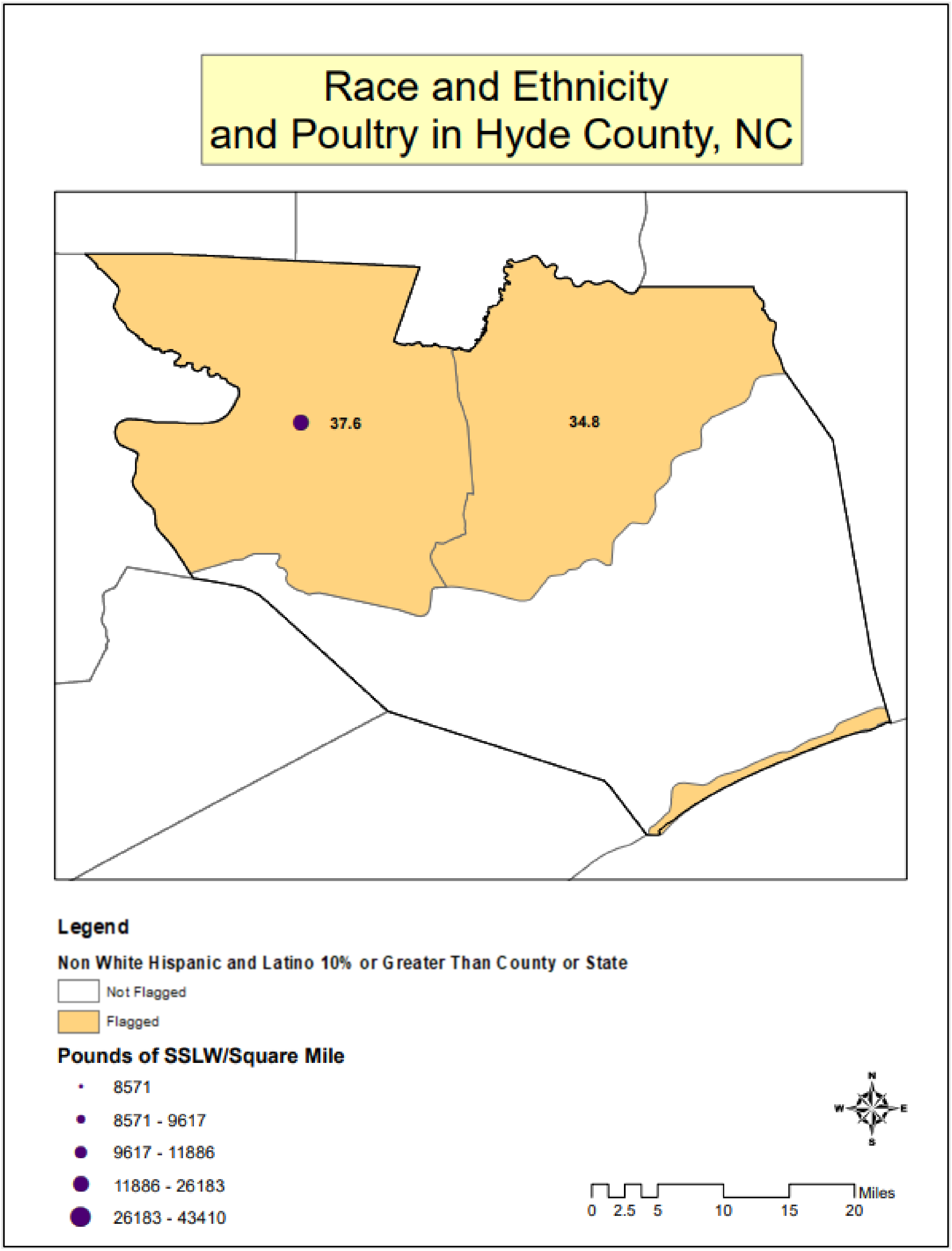


Figure 18. Map of Hyde County flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

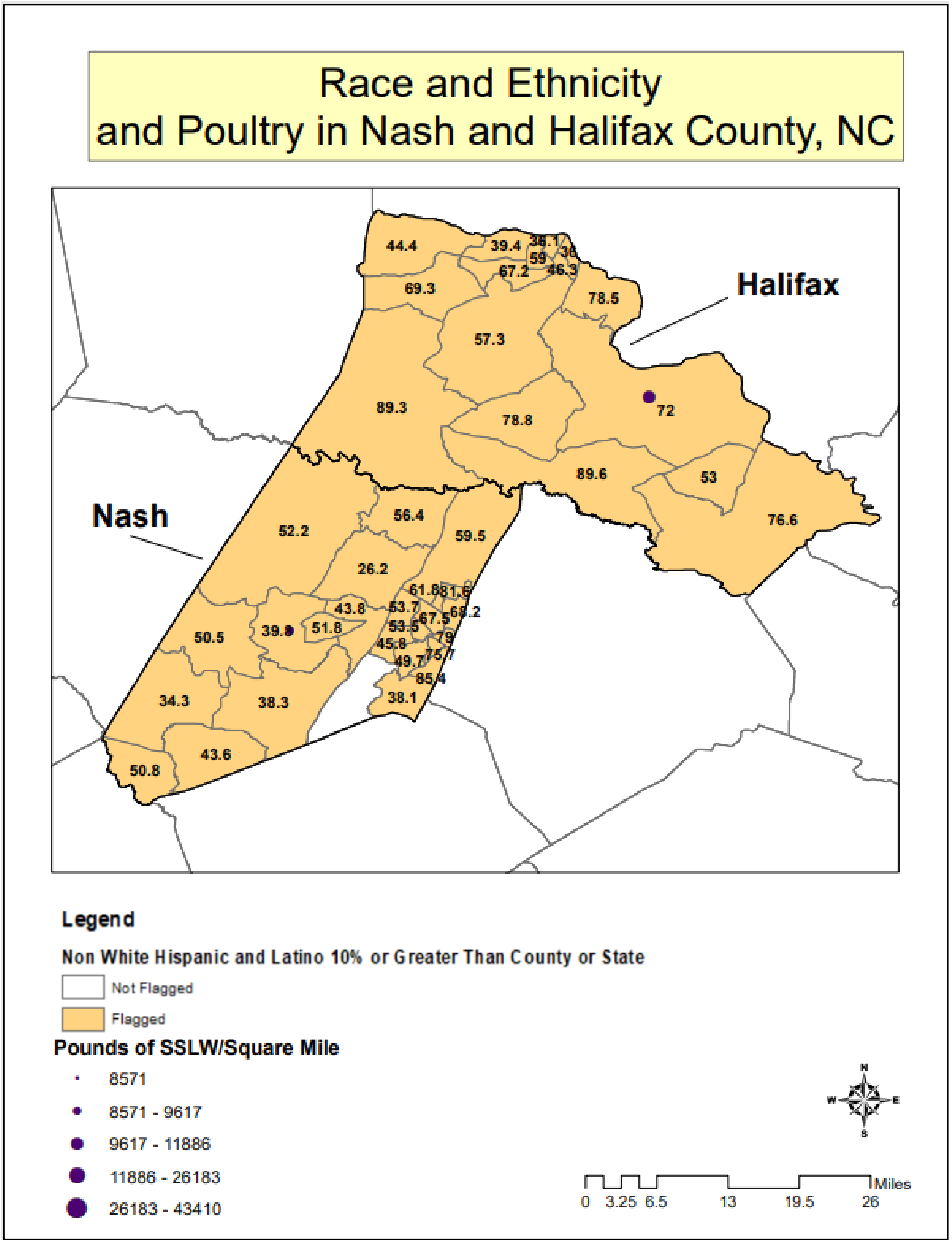


Figure 19. Map of Nash and Halifax Counties flagged census tracts for Nonwhite and Hispanic or Latino greater than 10% compared to the county or state.

Poverty

The following maps show the top 5 counties as ranked by the SSLW per square mile. This was calculated on the census tract level and summed up to the county level. All census tracts that are flagged in comparison to either the state or county as laid out above are highlighted in yellow in the maps. Table 10 shows the 2020 Census data for the 5 counties and North Carolina.

Table 10. Poverty levels for the five counties with the highest SSLW per Square mile for Poultry and North Carolina

Geography	Total Population	MOE +/-	Percent below Poverty	MOE +/-
North Carolina	9,984,891	1,988	13.6%	0.3%
Halifax County	49,855	255	25.8%	2.2%
Hyde County	4,624	152	24.3%	9.2%
Nash County	92,009	374	15.2%	1.5%
Orange County	133,298	744	13.7%	0.9%
Union County	227,980	366	8.2%	0.7%

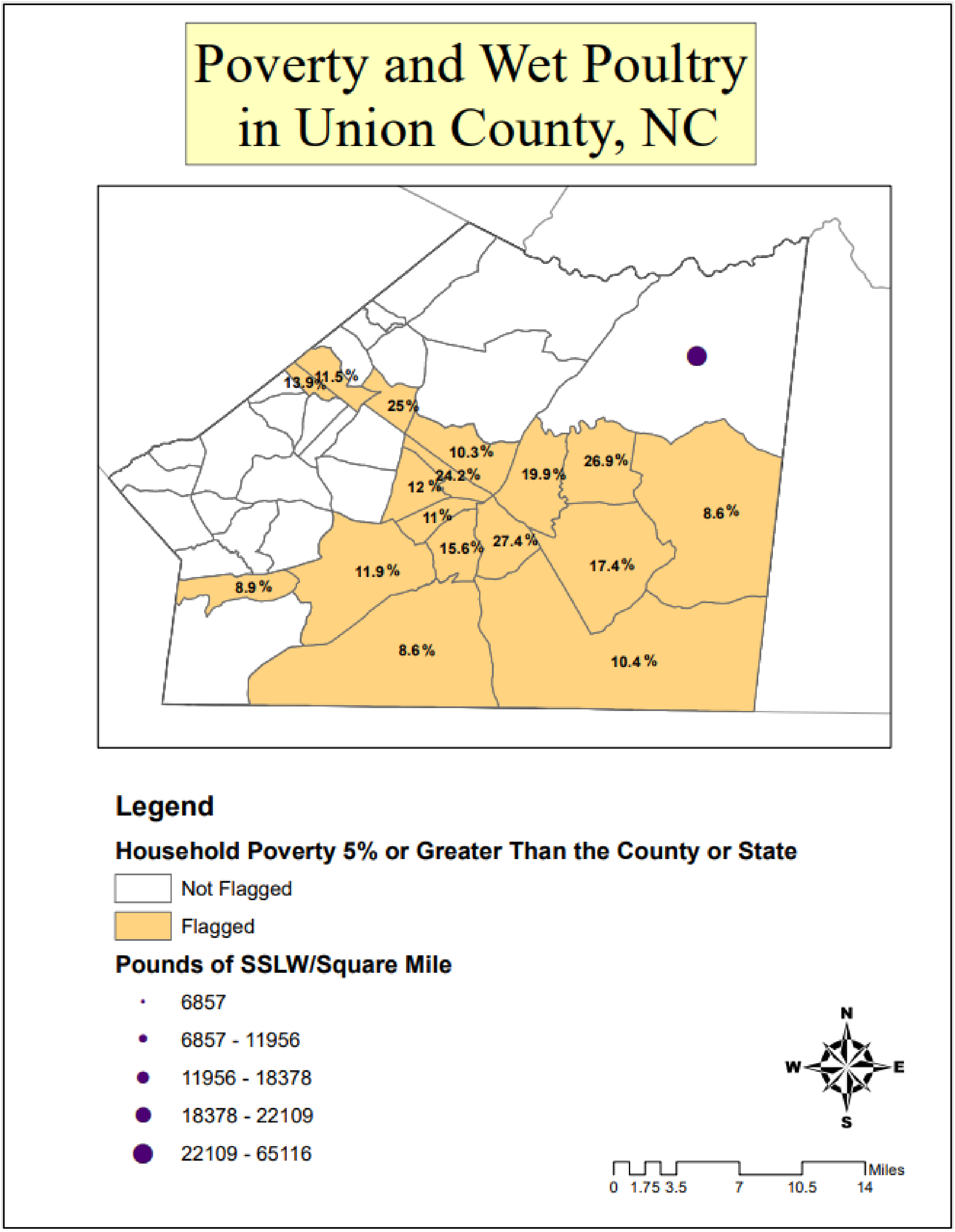


Figure 20. Map of Union County flagged census tracts for poverty.

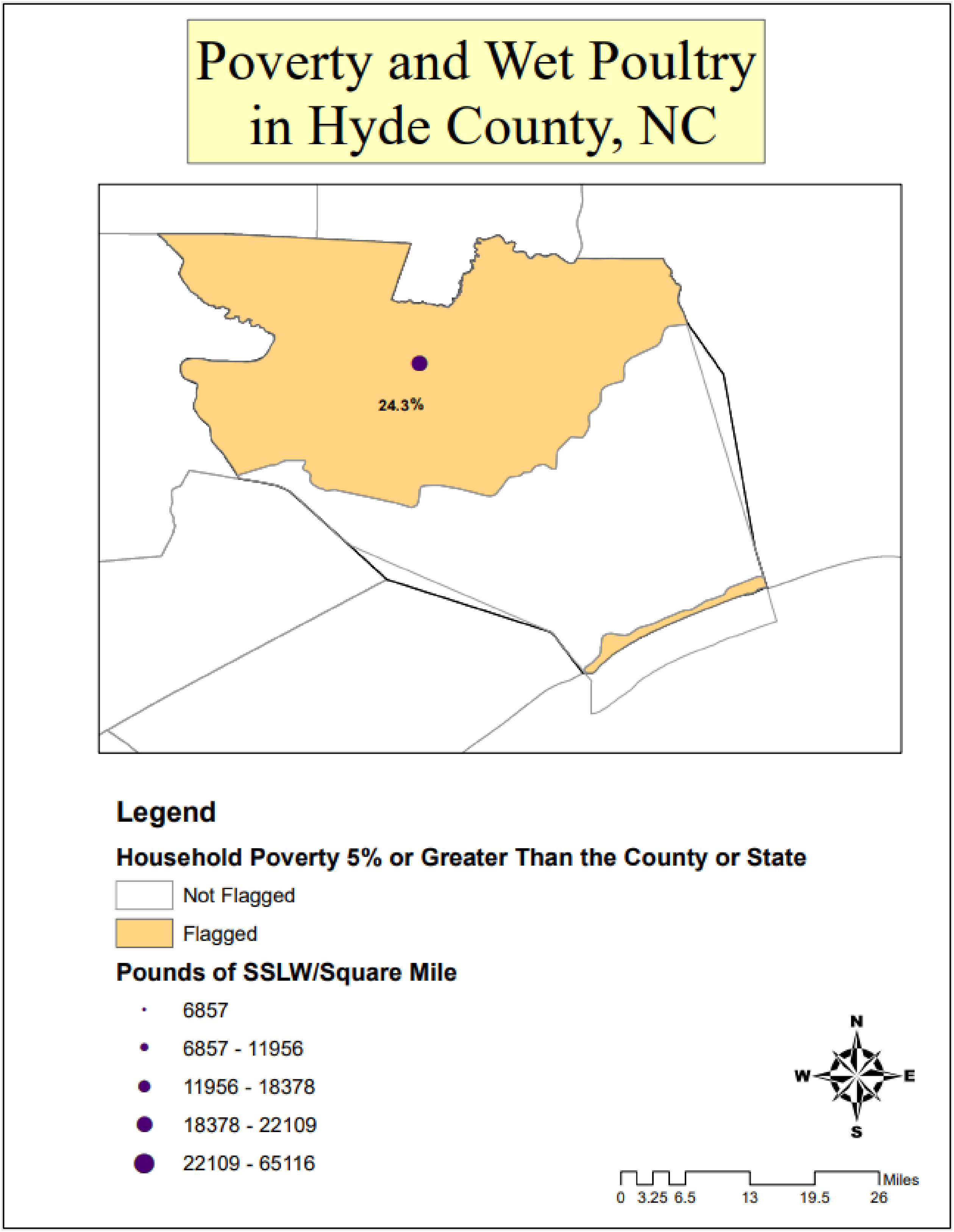


Figure 21. Map of Hyde County flagged census tracts for poverty.

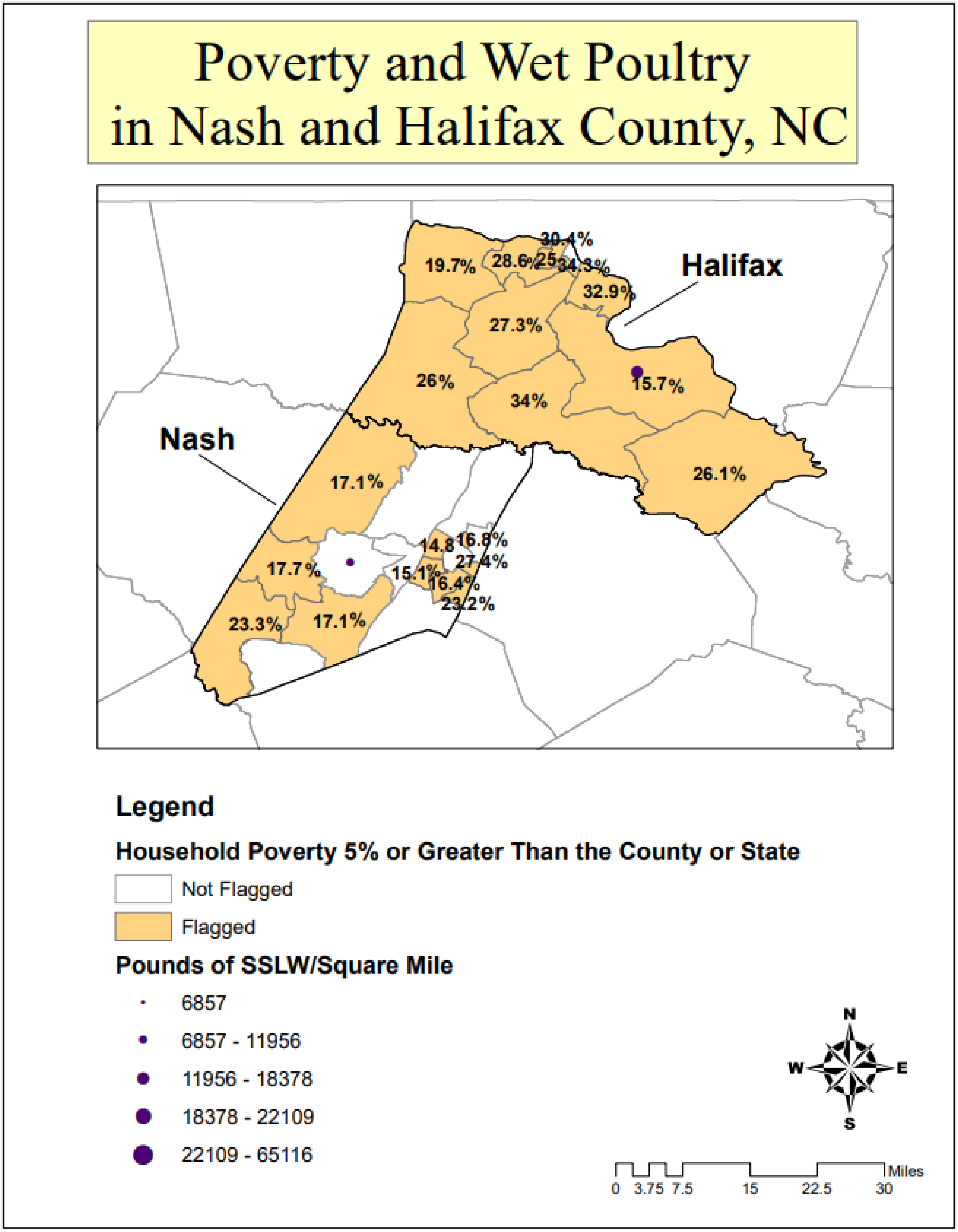


Figure 22. Map of Nash and Halifax Counties flagged census tracts for poverty.

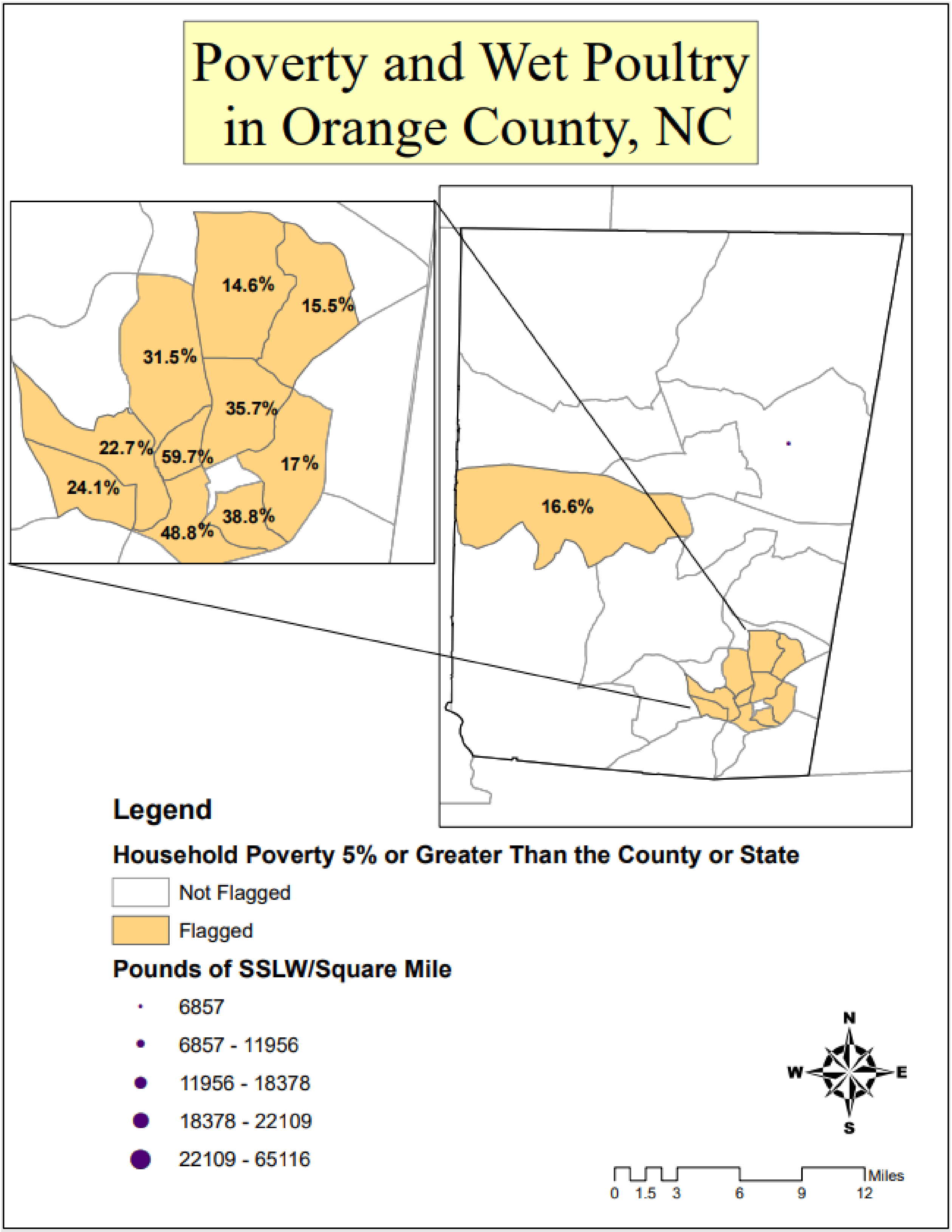


Figure 23. Map of Orange County flagged census tracts for poverty.

7 Tribal Communities

Across North Carolina, there are 7 state recognized tribes and 1 federally recognized tribe. Additionally, there are 4 Urban Indian Organizations. According to the Commission of Indian Affairs, these tribes and tribal organizations reside in 27 counties across North Carolina (Figure 24).

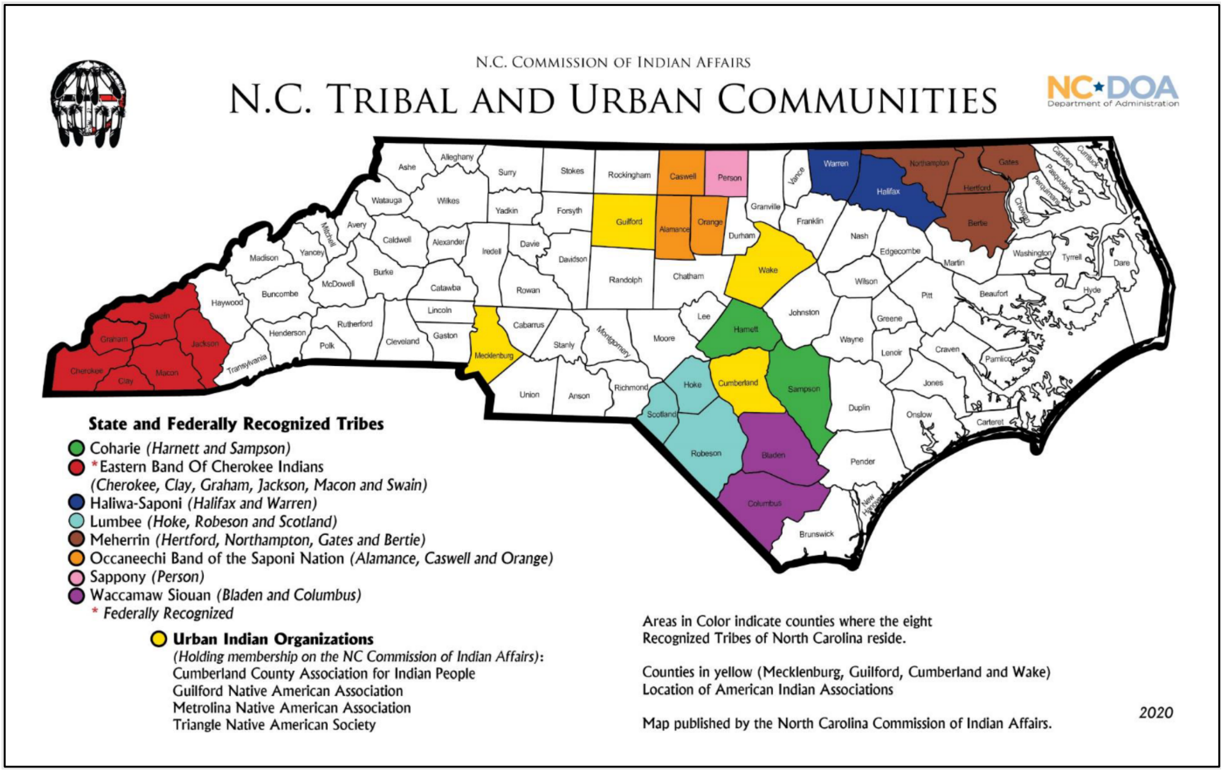


Figure 24. Map of North Carolina Tribal Communities (https://ncadmin.nc.gov/public/american-indians/map-nc-tribal-communities) .

8 County Health Rankings

The University of Wisconsin Population Health Institute, in collaboration with the Robert Wood Johnson Foundation, calculated County Health Rankings for all the States in the United States (www.countyhealthrankings.org). This ranking is based on health outcomes (such as lifespan and self-reported health status) and health factors (such as environmental, social and economic conditions). The following, Figure 25, ranks all 100 counties in North Carolina, with 1 indicating the healthiest. Tables 11-13 outline the health rankings for the 5 counties with the highest SSLW for each permit type included in the above analysis.

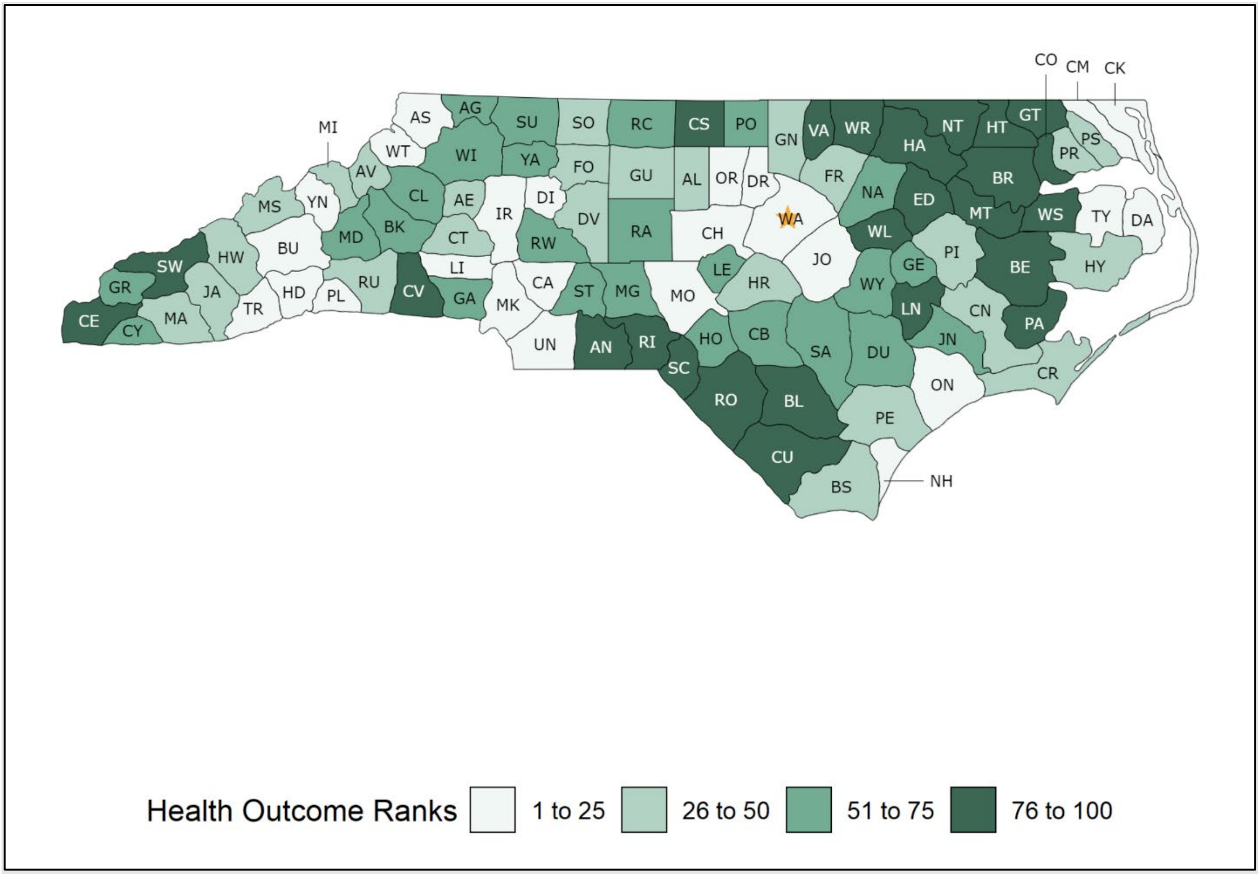


Figure 25. County Health Outcome Rankings for Health Factors in North Carolina provided by University of Wisconsin Public Health Institute

Table 11. Health information for the five counties with highest amount of SSLW per Square Mile for swine

Geography	Health Factors Ranking	Health Outcomes Ranking
Bladen County	93	86
Duplin County	85	58
Robeson County	100	100
Sampson County	80	67
Wayne County	70	64

Table 12. Health information for the five counties with highest amount of SSLW per Square Mile for cattle

Geography	Health Factors Ranking	Health Outcomes Ranking
Davidson County	47	49
Gaston County	40	51
Iredell County	17	15
Lincoln County	26	25
Randolph County	49	52

Table 13. Health information for the five counties with highest amount of SSLW per Square Mile four wet poultry

Geography	Health Factors Ranking	Health Outcomes Ranking
Union County	7	3
Hyde County	86	46
Nash County	63	62
Orange County	1	2
Halifax County	98	95

9 Conclusion

Environmental justice is the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income, with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies (US EPA). This Draft EJ report examined the SSLW per square mile for the three types of permits in North Carolina; wet poultry, swine, and cattle. Demographic data for poverty, race and ethnicity, and limited English proficiency was also analyzed.

It is important to note that this is an analysis of the facilities with current 2019 general permits and it is not anticipated that every facility covered under the 2019 general permits will apply for the new digester general permits. Additionally, the new digester general permits will replace the existing general permits only if the facility applies for the digester general permit. As it is not possible to predict which facilities will install digesters under the new general permits, this analysis relies on general information and is not a site-specific examination.

Based on the results from this analysis, the following outreach recommendations are recommended:

- Spanish translation for vital documents;
- Include the option for comments via phone lines or mail for the potential for lack of internet access;
- Conduct a mix of virtual and in person meetings, centrally located in the counties included in this report
- Communicate the information and process with the tribes, tribal organizations, and Commission of Indian Affairs across the state;
- Work with known community leaders across the state to distribute information to hard-to-reach communities, with a focus on the counties included in this analysis.

In addition, the NCDEQ Secretary's Environmental Justice and Equity Advisory Board (EJEAB) recommended specific actions in a letter dated August 26, 2021. As a result, NCDEQ has taken several actions, including:

- Scheduling a 90-day comment period on the draft permits;
- Scheduling four public meetings to accept input on the draft permits, including one in Duplin County and Sampson County with 60 days of notice for the public meetings;
- Arranging for Spanish translations of the draft permits and public notices;
- Conducting the fall stakeholder workgroup sessions with an independent moderator;
- Updating the EJEAB members throughout the permit development process.

ATTACHMENT 19

GENERAL ASSEMBLY OF NORTH CAROLINA
SESSION 2021

SESSION LAW 2021-78
SENATE BILL 605

AN ACT TO MAKE VARIOUS CHANGES TO THE LAWS CONCERNING AGRICULTURE AND FORESTRY.

The General Assembly of North Carolina enacts:

VOLUNTARY AGRICULTURAL DISTRICT TECHNICAL CHANGES

SECTION 1. Article 61 of Chapter 106 of the General Statutes reads as rewritten:

"Article 61.

"Agricultural Development and Preservation of Farmland.

…

"Part 2. Voluntary Agricultural Districts.

"§ 106-737. Qualifying farmland.

In order for farmland to qualify for inclusion in a voluntary agricultural district or an enhanced voluntary agricultural district under Part 1 or Part 2 of this Article, it must be real property that:

(1) ~~Is engaged in agriculture as that word is defined in G.S. 106-581.1.~~Is used for bona fide farm purposes, as that term is defined in G.S. 106-743.4(a) and G.S. 160D-903.

(2) Repealed by Session Laws 2005-390, s. 11 effective September 13, 2005.

(3) Is managed in accordance with the Soil Conservation Service defined erosion control practices that are addressed to highly erodable land; and

(4) Is the subject of a conservation agreement, as defined in G.S. 121-35, between the ~~county~~ local government administering the voluntary agricultural district program and the owner of such land that prohibits nonfarm use or development of such land for a period of at least 10 years, except for the creation of not more than three lots that meet applicable county and municipal zoning and subdivision regulations. The form of the conservation agreement shall be approved by the agricultural advisory board created under G.S. 106-739.

"§ 106-737.1. Revocation of conservation agreement.

By written notice to the ~~county,~~ local government administering the voluntary agricultural district program, the landowner may revoke this conservation agreement. Such revocation shall result in loss of qualifying farm status.

"§ 106-738. Voluntary agricultural districts.

(a) An ordinance adopted under this Part shall provide:

(1) For the establishment of voluntary agricultural districts ~~consisting initially of at least the number of contiguous acres of agricultural land, and forestland or horticultural land that is part of a qualifying farm or the number of qualifying farms deemed appropriate by the governing board of the county or city adopting the ordinance;~~upon the execution of a conservation agreement as provided in G.S. 106-737(4).

S605-V-7

(2) ~~For the formation of such districts upon the execution by the owners of the requisite acreage of an agreement to sustain agriculture in the district;~~

(3) ~~That the form of this agreement must be reviewed and approved by an agricultural advisory board established under G.S. 106-739 or some other county board or official;~~

(4) ~~That each such district have a representative on the agricultural advisory board established under G.S. 106-739.~~

(5) The minimum size, including acreage; number of tracts; and appropriate proximity of multiple tracts of agricultural land, forestland, or horticultural land that may comprise a voluntary agricultural district.

(b) The purpose of such agricultural districts shall be to increase identity and pride in the agricultural community and its way of life and to ~~increase protection from nuisance suits~~ decrease the likelihood of legal disputes, such as nuisance actions between farm owners and their neighbors, and other negative impacts on properly managed farms. The county or city that adopted an ordinance under this Part may take such action as it deems appropriate to encourage the formation of such districts and to further their purposes and objectives.

(c) A county ordinance adopted pursuant to this Part is effective within the unincorporated areas of the county. A city ordinance adopted pursuant to this Part is effective within the corporate limits of the city. A city may amend its ordinances in accordance with G.S. 160A-383.2 with regard to agricultural districts within its planning jurisdiction.

"§ 106-739. Agricultural advisory board.

(a) An ordinance adopted under this Part or Part 3 of this Article shall provide for the establishment of an agricultural advisory board, organized and appointed ~~as the county or city that adopted the ordinance shall deem appropriate.~~ by the board of county commissioners or the city council adopting the ordinance. The county or city that adopted the ordinance may confer upon this advisory board authority to:

(1) Review and make recommendations or decisions concerning the establishment and modification of agricultural ~~districts;~~districts. The board of county commissioners or the city council may make decisions regarding the establishment and modification of voluntary agricultural districts or may delegate that authority to the agricultural advisory board. If the authority is delegated to the agricultural advisory board, the agricultural advisory board's decisions shall be appealable to the board of county commissioners or city council by an owner of land that has been denied enrollment in a voluntary agricultural district or has been removed from a voluntary agricultural district by the agricultural advisory board.

(1a) Execute agreements with landowners necessary for enrollment of land in a voluntary agricultural district.

(2) Review and make recommendations concerning any ordinance or amendment adopted or proposed for adoption under this Part or Part 3 of this ~~Article;~~Article.

(3) Hold public hearings on public projects likely to have an impact on agricultural operations, particularly if such projects involve condemnation of all or part of any qualifying ~~farm;~~farm.

(4) Advise the governing board of the county or city that adopted the ordinance on projects, programs, or issues affecting the agricultural economy or way of life within the ~~county;~~county.

(5) Perform other related tasks or duties assigned by the governing board of the county or city that adopted the ordinance.

(b) The members of the agricultural advisory board shall be chosen to provide the broadest possible representation of the geographical regions of the local government and to represent, to the extent possible, all segments of agricultural production existing within the local government. A majority of the members of the agricultural advisory board shall be actively engaged in agriculture.

(c) The agricultural advisory board may, at the discretion of the board of county commissioners or the city council, utilize an existing local government agency for the purpose of administration, record keeping, and other related tasks or duties.

…

"§ 106-741. Record notice of proximity to farmlands.

(a) All counties shall require that land records include some form of notice reasonably calculated to alert a person researching the title of a particular tract that such tract is located within one-half mile of ~~a poultry, swine, or dairy qualifying farm or within 600 feet of any other qualifying farm or within one-half mile of a voluntary agricultural district.~~the property line of any tract of land enrolled in a voluntary agricultural district.

…

"§ 106-743. Local ordinances.

A county or a city adopting an ordinance under this Part or Part 3 of this Article may consult with the North Carolina Commissioner of Agriculture or ~~his~~ the Commissioner's staff before adoption, and shall record the ordinance with the Commissioner's office after adoption. Thereafter, the county or city shall submit to the Commissioner at least once a year, a written report including the status, ~~progress~~ progress, number of enrolled farms and acres, and activities of its farmland preservation program under this Part or Part 3 of this Article.

"Part 3. Enhanced Voluntary Agricultural Districts.

"§ 106-743.1. Enhanced voluntary agricultural districts.

(a) A county or a municipality may adopt an ordinance establishing an enhanced voluntary agricultural district. An ordinance adopted pursuant to this Part shall provide:

(1) For the establishment of an enhanced voluntary agricultural district that initially consists of at least the number of contiguous acres of agricultural land, and forestland and horticultural land that is part of a qualifying farm under G.S. 106-737 or the number of qualifying farms deemed appropriate by the governing board of the county or city adopting the ordinance.

(2) For the formation of the enhanced voluntary agricultural district upon the execution of a conservation agreement, as defined in G.S. 121-35, that meets the condition set forth in G.S. 106-743.2 by the landowners of the requisite acreage to sustain agriculture in the enhanced voluntary agricultural district.

(3) That the form of the agreement under subdivision (2) of this subsection be reviewed and approved by an agricultural advisory board established under G.S. 106-739, or other governing board of the county or city that adopted the ordinance.

~~(4) That each enhanced voluntary agricultural district have a representative on the agricultural advisory board established under G.S. 106-739.~~

(b) The purpose of establishing an enhanced voluntary agricultural district is to allow a county or a city to provide additional benefits to farmland beyond that available in a voluntary agricultural district established under Part 2 of this Article, when the owner of the farmland agrees to the condition imposed under G.S. 106-743.2. The county or city that adopted the ordinance may take any action it deems appropriate to encourage the formation of these districts and to further their purposes and objectives.

(c) A county ordinance adopted pursuant to this Part is effective within the unincorporated areas of the county. A city ordinance adopted pursuant to this Part is effective within the corporate limits of the city. A city may amend its ordinances in accordance with G.S. 160A-383.2 with regard to agricultural districts within its planning jurisdiction.

(d) A county or city ordinance adopted pursuant to this Part may be adopted simultaneously with the creation of a voluntary agricultural district pursuant to G.S. 106-738.
…."

ALLOW MAGISTRATES TO WAIVE TRIALS FOR STATE FOREST RULE OFFENSES

SECTION 2.(a) G.S. 7A-273 reads as rewritten:

"**§ 7A-273. Powers of magistrates in infractions or criminal actions.**

In criminal actions or infractions, any magistrate has power:

…

(2) In misdemeanor or infraction cases involving alcohol offenses under Chapter 18B of the General Statutes, traffic offenses, hunting, fishing, State park and recreation area rule offenses under Chapters 113 and 143B of the General Statutes, <u>State forest rule offenses under Articles 74 and 75 of Chapter 106 of the General Statutes, </u>boating offenses under Chapter 75A of the General Statutes, open burning offenses under Article 78 of Chapter 106 of the General Statutes, and littering offenses under G.S. 14-399(c) and G.S. 14-399(c1), to accept written appearances, waivers of trial or hearing and pleas of guilty or admissions of responsibility, in accordance with the schedule of offenses and fines or penalties promulgated by the Conference of Chief District Judges pursuant to G.S. 7A-148, and in such cases, to enter judgment and collect the fines or penalties and costs;

…."

SECTION 2.(b) This section becomes effective December 1, 2021, and applies to offenses committed on or after that date.

EXEMPT CERTAIN FIRES FROM OPEN BURNING LAWS

SECTION 3.(a) G.S. 106-950 is amended by adding a new subsection to read:

"<u>(a2) Except in cases where the Commissioner has prohibited all open burning during periods of hazardous forest fire conditions or during air pollution episodes declared pursuant to Article 21B of Chapter 143 of the General Statutes, this Article does not apply to any fires started, or caused to be started, for cooking, warming, or ceremonial events, if the fire is confined (i) within an enclosure from which burning material may not escape or (ii) within a protected area upon which a watch is being maintained and which is provided with adequate fire protection equipment.</u>"

SECTION 3.(b) This section becomes effective December 1, 2021, and applies to offenses committed on or after that date.

FOREST SERVICE OVERTIME MODIFICATION

SECTION 4.(a) G.S. 106-903 reads as rewritten:

"**§ 106-903. Overtime compensation for forest fire fighting.**

The Department shall, within funds appropriated to the Department, provide <u>either monetary</u> overtime compensation <u>or compensatory leave at an hour-for-hour rate, at its discretion, </u>to the ~~professional~~ employees of the North Carolina Forest Service <u>who are exempt from the Fair Labor Standards Act and </u>involved in fighting forest ~~fires.~~<u>fires for overtime earned while conducting fire suppression duties as defined in G.S. 106-955. If the Department provides compensatory leave for overtime earned, it shall be provided in a manner consistent with the State's general compensatory time policy for exempt employees established by the Office of State Human Resources.</u>"

SECTION 4.(b) This section is effective when it becomes law and applies to overtime earned on or after that date.

INCREASE PUNISHMENT FOR TIMBER LARCENY AND INCREASE CIVIL PENALTIES FOR DAMAGING TIMBER OR AGRICULTURAL COMMODITIES

SECTION 5.(a) G.S. 14-135 reads as rewritten:

"**§ 14-135. ~~Cutting, injuring, or removing another's~~ Larceny of timber.**

(a) Offense. – Except as otherwise provided in subsection (b) of this section, a person commits the offense of larceny of timber if the person does any of the following:

(1) ~~If any person not being the bona fide owner thereof, shall knowingly and willfully cut down, injure or remove any standing, growing or fallen tree or log off the property of another, the person shall be punished the same as in G.S. 14-72.~~Knowingly and willfully cuts down, injures, or removes any timber owned by another person, without the consent of the owner of the land or the owner of the timber, or without a lawful easement running with the land.

(2) Buys timber directly from the owner of the timber and fails to make payment in full to the owner by (i) the date specified in the written timber sales agreement or (ii) if there is no such agreement, 60 days from the date that the buyer removes the timber from the property.

(b) Exceptions. – The following are exceptions to the offense set forth in subsection (a) of this section:

(1) A person is not guilty of an offense under subdivision (1) of subsection (a) of this section if the person is an employee or agent of an electric power supplier, as defined in G.S. 62-133.8, and either of the following conditions is met:

a. The person believed in good faith that consent of the owner had been obtained prior to cutting down, injuring, or removing the timber.

b. The person believed in good faith that the cutting down, injuring, or removing of the timber was permitted by a utility easement or was necessary to remove a tree hazard. For purposes of this sub-subdivision, the term "tree hazard" includes a dead or dying tree, dead parts of a living tree, or an unstable living tree that is within striking distance of an electric transmission line, electric distribution line, or electric equipment and constitutes a hazard to the line or equipment in the event of a tree failure.

(2) A person is not guilty of an offense under subdivision (2) of subsection (a) of this section if either of the following conditions is met:

a. The person remitted payment in full within the time period set in subdivision (2) of subsection (a) of this section to a person he or she believed in good faith to be the rightful owner of the timber.

b. The person remitted payment in full to the owner of the timber within the 10-day period set forth in subsection (c) of this section.

(c) Prima Facie Evidence. – An owner of timber who does not receive payment in full within the time period set in subdivision (2) of subsection (a) of this section may notify the timber buyer in writing of the owner's demand for payment at the timber buyer's last known address by certified mail or by personal delivery. The timber buyer's failure to make payment in full within 10 days after the mailing or personal delivery authorized under this subsection shall constitute prima facie evidence of the timber buyer's intent to commit an offense under subdivision (2) of subsection (a) of this section.

(d) Penalty; Restitution. – A person who commits an offense under subsection (a) of this section is guilty of a Class G felony. Additionally, a defendant convicted of an offense under subsection (a) of this section shall be ordered to make restitution to the timber owner in an amount equal to either of the following:

(1) Three times the value of the timber cut down, injured, or removed in violation of subdivision (1) of subsection (a) of this section.

(2) Three times the value of the timber bought but not paid for in violation of subdivision (2) of subsection (a) of this section.

Restitution shall also include the cost incurred by the owner to determine the value of the timber. For purposes of subdivisions (1) and (2) of this subsection, "value of the timber" shall be based on the stumpage rate of the timber.

(e) Civil Remedies. – Nothing in this section shall affect any civil remedies available for a violation of subsection (a) of this section."

SECTION 5.(b) G.S. 1-539.1 reads as rewritten:

"**§ 1-539.1. Damages for unlawful cutting, removal or burning of timber; misrepresentation of property lines.**

(a) Any person, firm or corporation not being the bona fide owner thereof or agent of the owner who shall without the consent and permission of the bona fide owner enter upon the land of another and injure, cut or remove any valuable wood, timber, shrub or tree therefrom, shall be liable to the owner of said land for ~~double~~ triple the value of such wood, timber, shrubs or trees so injured, cut or removed.

(b) If any person, firm or corporation shall willfully and intentionally set on fire, or cause to be set on fire, in any manner whatever, any valuable wood, timber or trees on the lands of another, such person, firm or corporation shall be liable to the owner of said lands for ~~double~~ triple the value of such wood, timber or trees damaged or destroyed thereby.

…."

SECTION 5.(c) G.S. 1-539.2B reads as rewritten:

"**§ 1-539.2B. ~~Double~~ Triple damages for injury to agricultural commodities or production systems; define value of agricultural commodities grown for educational, testing, or research purposes.**

(a) Any person who unlawfully and willfully injures or destroys any other person's agricultural commodities or production system is liable to the owner for ~~double~~ triple the value of the commodities or production system injured or destroyed.

…."

SECTION 5.(d) Subsection (a) of this section becomes effective December 1, 2021, and applies to offenses committed on or after that date. Subsections (b) and (c) of this section become effective December 1, 2021, and apply to civil actions filed on or after that date.

REQUIRE TIMBER BUYERS AND TIMBER OPERATORS TO PROVIDE A WOOD LOAD TICKET TO SELLERS OF CERTAIN WOOD PRODUCTS

SECTION 6.(a) Article 22 of Chapter 14 of the General Statutes is amended by adding a new section to read:

"**§ 14-135.1. Wood load tickets required for certain wood product sales; exceptions; penalties.**

(a) Definition. – For purposes of this section, the term "wood product" means trees, timber, wood, or any combination thereof.

(b) Requirement. – Except as provided in this section, whenever a timber buyer or timber operator purchases wood product by the load directly from a timber grower or seller and the load is sold by weight, cord, or measure of board feet, the timber buyer or operator shall furnish the timber grower or seller, within 30 days of the completion of the wood product harvest, a separate, true, and accurate wood load ticket for each load of wood product removed from the timber grower's or seller's property. At a minimum, each wood load ticket shall include all of the following information provided by the timber grower or seller who sold the wood product:

(1) The name of the timber grower or seller.

(2) The county from which the wood product was severed.

(3) The amount of wood product severed.

(4) The date the wood product was delivered to the timber buyer or timber operator.

(c) Applicability. – The provisions of this section do not apply to the following:

(1) The sale of wood for firewood only.

(2) A landowner harvesting and processing their own timber.

(3) Bulk or lump sum sales for an agreed total price for all timber purchased and sold in one transaction.

(d) Punishment. – Any person who violates this section is guilty of a Class 2 misdemeanor."

SECTION 6.(b) This section becomes effective December 1, 2021, and applies to offenses committed on or after that date.

EXPAND THE LAWS ENFORCED BY DEPARTMENT OF AGRICULTURE AND CONSUMER SERVICES LAW ENFORCEMENT OFFICERS

SECTION 7.(a) G.S. 106-897 reads as rewritten:

"**§ 106-897. Forest laws defined.**

The forest laws consist ~~of:~~of all of the following:

(1) ~~G.S. 14-136 to G.S. 14-140;~~G.S. 14-135 to G.S. 14-140.1.

(2) Articles 74 through 84 of this ~~Chapter;~~Chapter.

(3) G.S. 77-13 and ~~G.S. 77-14;~~G.S. 77-14.

(4) Other statutes enacted for the protection of forests and woodlands from fire, insects, or disease and concerning obstruction of streams and ditches in forests and ~~woodlands; and~~woodlands.

(5) Regulations and ordinances adopted under the authority of the above statutes."

SECTION 7.(b) This section becomes effective December 1, 2021, and applies to offenses committed on or after that date.

REQUIRE PRODUCTION OF ELECTRONIC RECORDS FOR DEPARTMENT OF AGRICULTURE AND CONSUMER SERVICES RECORD AUDITS

SECTION 8.(a) G.S. 106-92.8 reads as rewritten:

"**§ 106-92.8. Tonnage fees: reporting system.**

For the purpose of defraying expenses connected with the registration, inspection and analysis of the materials coming under this Article, each manufacturer or registrant shall pay to the Department of Agriculture and Consumer Services tonnage fees in addition to registration fees as follows: for agricultural liming material, fifty cents (50¢) per ton; for landplaster, fifty cents (50¢) per ton; excepting that these fees shall not apply to materials which are sold to fertilizer manufacturers for the sole purpose for use in the manufacture of fertilizer or to materials when sold in packages of 10 pounds or less.

Any manufacturer, importer, jobber, firm, corporation or person who distributes materials coming under this Article in this State shall make application for a permit to report the materials sold and pay the tonnage fees as set forth in this section.

The Commissioner of Agriculture shall grant such permits on the following conditions: The applicant's agreement ~~that he will~~ to keep such records as may be necessary to indicate accurately the tonnage of liming materials, etc., sold in the State and ~~his~~ the applicant's agreement for the Commissioner or ~~this~~ the Commissioner's authorized representative to examine such records to verify the tonnage statement. If the records are available electronically, the electronic records shall be made available to the Commissioner or the Commissioner's authorized representative. The registrant shall report quarterly and pay the applicable tonnage fees quarterly, on or before the tenth day of October, January, April, and July of each year. The report and payment shall cover the tonnage of liming materials, etc., sold during the preceding quarter. The report shall be on forms furnished by the Commissioner. If the report is not filed and the tonnage fees paid by the last day of the month in which it is due, or if the report be false, the amount due shall bear a penalty of ten percent (10%) which shall be added to the tonnage fees due. If the report is not filed and the tonnage fees paid within 60 days of the date due, or if the report or tonnage be false, the Commissioner may revoke the permit and cancel the registration."

SECTION 8.(b) G.S. 106-277.12 reads as rewritten:

"**§ 106-277.12. Records.**

All persons transporting or delivering for transportation, selling, offering or exposing for sale agricultural or vegetable seeds if their name appears on the label shall keep for a period of two years a file sample and a complete record of such seed, including invoices showing lot number, kind and variety, origin, germination, purity, treatment, and the labeling of each lot. The Commissioner or ~~his~~ the Commissioner's duly authorized agents shall have the right to inspect such records in connection with the administration of this Article at any time during customary business hours. If the records are available electronically, the electronic records shall be made available to the Commissioner or the Commissioner's authorized representative."

SECTION 8.(c) G.S. 106-284.40(c)(2) reads as rewritten:

"(2) Keep such records as may be necessary or required by the Commissioner to indicate accurately the tonnage of commercial feed distributed in this State, and the Commissioner or ~~his~~ the Commissioner's duly designated agent shall have the right to examine such records during normal business hours, to verify statements of tonnage. If the records are available electronically, the electronic records shall be made available to the Commissioner or the Commissioner's authorized representative. Failure to make an accurate statement of tonnage or to pay the inspection fee or comply as provided herein shall constitute sufficient cause for the cancellation of all registrations on file for the distributor."

SECTION 8.(d) G.S. 106-671(b) reads as rewritten:

"(b) Reporting System. – Each manufacturer, importer, jobber, firm, corporation or person who distributes commercial fertilizers in this State shall make application to the Commissioner for a permit to report the tonnage of commercial fertilizer sold and shall pay to the North Carolina Department of Agriculture and Consumer Services an inspection fee of fifty cents (50¢) per ton. The Commissioner is authorized to require each such distributor to keep such records as may be necessary to indicate accurately the tonnage of commercial fertilizers sold in the State, and as are satisfactory to the Commissioner. Such records shall be available to the Commissioner, or ~~his~~ the Commissioner's duly authorized representative, at any and all reasonable hours for the purpose of making such examination as is necessary to verify the tonnage statement and the inspection fees paid. If the records are available electronically, the electronic records shall be made available to the Commissioner or the Commissioner's authorized representative. Each registrant shall report monthly the tonnage sold to non-registrants on forms furnished by the Commissioner. Such reports shall be made and inspection fees shall be due and payable monthly on the fifteenth of each month covering the tonnage and kind of commercial fertilizers sold during the past month. If the report is not filed and the inspection fee paid by the last day of the month it is due, the amount due shall bear a penalty of ten percent (10%), which shall be added to the inspection fee due. If the report is not filed and the inspection fee paid within 60 days of the date due, or if the report or tonnage be false, the Commissioner may revoke the permit."

TOBACCO TRUST FUND COMMISSION ADMIN EXPENSES

SECTION 9. G.S. 143-717(i) reads as rewritten:

"(i) Limit on Operating and Administrative Expenses. – All administrative expenses of the Commission shall be paid from the Fund. No more than ~~three hundred fifty thousand dollars ($350,000)~~ three hundred seventy-five thousand dollars ($375,000) may be used each fiscal year for administrative and operating expenses of the Commission and its staff, provided that the Commission may annually adjust the administrative expense cap imposed by this subsection, so long as that any cap increase does not exceed the amount necessary to provide for statewide salary and benefit adjustments enacted by the General Assembly."

WORKERS' COMPENSATION DEFINITION CLARIFICATION

SECTION 10. G.S. 97-2 reads as rewritten:

"**§ 97-2. Definitions.**

When used in this Article, unless the context otherwise requires:

(1) Employment. – The term "employment" includes employment by the State and all political subdivisions thereof, and all public and quasi-public corporations therein and all private employments in which three or more employees are regularly employed in the same business or establishment or in which one or more employees are employed in activities which involve the use or presence of radiation, except agriculture and domestic services, unless 10 or more full-time nonseasonal agricultural workers are regularly employed by the employer and an individual sawmill and logging operator with less than 10 employees, who saws and logs less than 60 days in any six consecutive months and whose principal business is unrelated to sawmilling or logging. For purposes of this section, "agriculture" has the same meaning as in G.S. 106-581.1.

…."

CREATE A NEW GENERAL PERMIT FOR FARMS WITH FARM DIGESTER SYSTEMS

SECTION 11.(a) G.S. 143-213 reads as rewritten:

"**§ 143-213. Definitions.**

Unless the context otherwise requires, the following terms as used in this Article and Articles 21A of this Chapter are defined as follows:

…

(5a) The terms "animal waste" and "animal waste management system" have the same meaning as in G.S. 143-215.10B.

…

(12a) The term "farm digester system" means a system, including all associated equipment and lagoon covers, by which gases are collected and processed from an animal waste management system for the digestion of animal biomass for use as a renewable energy resource. A farm digester system shall be considered an agricultural feedlot activity within the meaning of "animal operation" and shall also be considered a part of an "animal waste management system" as those terms are defined in G.S. 143-215.10B.

(12b) The term "lagoon cover" means a structure or material that covers a lagoon receiving animal waste as part of an animal waste management system. For purposes of this subdivision, the term "lagoon" includes a lagoon as defined in G.S. 106-802(1) or a storage pond.

…

(14a) The term "renewable animal biomass energy resource" means any renewable energy resource, as defined in G.S. 62-133.8(a)(8), that utilizes animal waste as a biomass resource, including a farm digester system.

…."

SECTION 11.(b) G.S. 143-215.10C reads as rewritten:

"**§ 143-215.10C. Applications and permits.**

(a) No person shall construct or operate an animal waste management system for an animal operation or operate an animal waste management system for a dry litter poultry facility that is required to be permitted under 40 Code of Federal Regulations § 122, as amended at 73 Federal Register 70418 (November 20, 2008), without first obtaining an individual permit or a general permit under this Article. The Commission shall develop a system of individual and general permits for animal operations and dry litter poultry facilities based on species, number of animals, and other relevant factors. <u>The Commission shall develop a general permit for animal operations that includes authorization for the permittee to construct and operate a farm digester system.</u> It is the intent of the General Assembly that most animal waste management systems be permitted under a general permit. The Commission, in its discretion, may require that an animal waste management ~~system~~ <u>system, including an animal waste management system that utilizes a farm digester system,</u> be permitted under an individual permit if the Commission determines that an individual permit is necessary to protect water quality, public health, or the environment. <u>After the general permit for animal operations that includes authorization for the permittee to construct and operate a farm digester system has been issued, the decision to require an individual permit shall not be based solely on the fact that the animal waste management system utilizes a farm digester system.</u> The owner or operator of an animal operation shall submit an application for a permit at least 180 days prior to construction of a new animal waste management system or expansion of an existing animal waste management system and shall obtain the permit prior to commencement of the construction or expansion. The owner or operator of a dry litter poultry facility that is required to be permitted under 40 Code of Federal Regulations § 122, as amended at 73 Federal Register 70418 (November 20, 2008), shall submit an application for a permit at least 180 days prior to operation of a new animal waste management system.

…

(c) The Commission shall act on a permit application as quickly as possible and may conduct any inquiry or investigation it considers necessary before acting on an application.

<u>(c1) Failure of the Commission to make a final permitting decision involving a notice of intent for a certificate of coverage under a general permit for animal operations that includes authorization for the permittee to construct and operate a farm digester system within 90 days of the Commission's receipt of a completed notice of intent shall result in the deemed approval of coverage under the permit. If the Commission fails to act within 90 days of the Commission's receipt of a completed notice of intent, the permittee may request that the Commission provide written confirmation that the notice of intent is deemed approved. Failure to provide this written confirmation within 10 days of the request shall serve as a basis to seek a contested case hearing pursuant to Article 3 of Chapter 150B of the General Statutes. Unless all parties to the case agree otherwise in writing, the administrative law judge shall issue a final decision or order in the contested case no later than 120 days after its commencement pursuant to G.S. 150B-23; provided that, upon written request of the administrative law judge or any party to the hearing, the Chief Administrative Law Judge may extend this deadline for good cause shown, no more than two times, for not more than 30 days per extension. Upon review of a failure to act on a notice of intent, the administrative law judge may either (i) direct the Commission to issue a written certificate of coverage under the general permit or (ii) deny the petition.</u>

…."

SECTION 11.(c) For purposes of this section, the following definitions apply:

(1) "Certificate of coverage" means an approval granted to a person who meets the requirements of coverage under a general permit as provided in 15A NCAC 02T .0111 (Conditions for Issuing General Permits).

(2) "Commission" means the Environmental Management Commission.

(3) "Notice of intent" means a request for coverage under a general permit using forms approved by the Division of Water Resources of the Department of Environmental Quality.

SECTION 11.(d) The Commission shall immediately initiate the process of developing and issuing a general permit for animal operations that includes authorization for the permittee to construct and operate a farm digester system. In addition to conditions required to describe and authorize the construction, monitoring, and proper operation of farm digester systems, the general permit shall contain the same conditions that are included in the currently existing general permits for animal operations. The general permit shall become effective no later than 12 months after the effective date of this section and shall expire on the later of September 30, 2024, or the effective date of the next version of the currently existing general permit for animal operations.

SECTION 11.(e) Until the general permit issued under subsection (d) of this section becomes effective, any animal operation that holds a general or individual permit that (i) is in effect on the effective date of this section and (ii) authorizes the construction and operation of a farm digester system may construct and continue to operate the farm digester system as authorized by that permit. For any animal operation that holds a general or individual permit that is in effect on the effective date of this section, but that does not authorize the construction and operation of a farm digester system, an operator may submit a notice of intent to be covered under the general permit to be developed under subsection (d) of this section. If the submitted notice of intent is incomplete, the Commission shall notify the applicant of the deficiency in the notice of intent. When an operator submits a completed notice of intent, the Commission shall, within 90 days of receipt of the completed notice of intent, either issue a certificate of coverage allowing the operator to construct and operate the farm digester system or notify the operator of the basis for the denial of the certificate of coverage. If the Commission fails to take action on the notice of intent within 90 days, authorization to construct and operate a farm digester system under the existing general permit shall be deemed approved.

SECTION 11.(f) Nothing in this section shall apply to permits for facilities that are required to be permitted under 40 C.F.R. § 122, as amended at 73 Federal Register 70418 (November 20, 2008).

SECTION 11.(g) G.S. 106-806 reads as rewritten:

"**§ 106-806. Construction or renovation of swine houses at preexisting swine farms.**

(a) As used in this section, the following definitions apply:

(1) "Farm digester system" means a farm digester system as defined in G.S. 143-213(12a).

(2) "New swine farm" means any swine farm the operations of which were sited on or after October 1, 1995. "New swine farm" does not include any preexisting swine farm, even if a subsequent site evaluation is performed on or after October 1, 1995, at the preexisting swine farm.

~~(2)~~(3) "Preexisting swine farm" means any swine farm either the operations of which were begun prior to October 1, 1995, or the site evaluation of which was approved prior to October 1, 1995, by the Department of Environmental Quality under Part 1A of Article 21 of Chapter 143 of the General Statutes.

~~(3)~~(4) "Renovation or construction," "renovated or constructed," and any similar phrase mean any activity to renovate, construct, reconstruct, rebuild, modify, alter, change, restructure, upgrade, improve, enlarge, reduce, move, or otherwise perform construction work on a swine house that is a component of a swine farm.

…

(e) Notwithstanding any other provision of this Article, a farm digester system that is a component of a preexisting swine farm may be constructed or renovated if the construction or renovation of the farm digester system satisfies all of the following requirements:

(1) The construction or renovation of the farm digester system does not result in an increase in the permitted capacity of the swine farm, as measured by the annual steady state live weight capacity of the swine farm.

(2) The construction or renovation of the farm digester system does not result in requiring an increase in the total permitted capacity of the animal waste management system or systems located at the swine farm.

(3) The construction or renovation of the farm digester system shall comply with the siting requirements set out in G.S. 106-803 to the maximum extent practicable. Except as provided in subsection (c) of this section, construction or renovation of the farm digester system shall not result in any portion of the constructed or renovated farm digester system being located closer to the building, property, or well that is the object of the siting requirement than any existing component of the animal waste management system that fails to meet the siting requirements of G.S. 106-803.

(4) Renovation or construction of a farm digester system shall not be allowed in the 100-year floodplain."

SECTION 11.(h) G.S. 105-275(8) is amended by adding a new sub-subdivision to read:

"a2. Notwithstanding sub-subdivision a1. of this subdivision, sub-subdivision a. of this subdivision applies to a farm digester system as defined in G.S. 143-213(12a)."

SECTION 11.(i) This section is effective when it becomes law.

CLARIFY THE DURATION OF DRIVERS LICENSES FOR H-2A WORKERS

SECTION 12.(a) G.S. 20-7(f)(3) reads as rewritten:

"(3) Duration of license for certain other drivers. – The durations listed in subdivisions (1), (2) and (2a) of this subsection are valid unless the Division determines that a license of shorter duration should be issued when the applicant holds valid documentation issued by, or under the authority of, the United States government that demonstrates the applicant's legal presence of limited duration in the United States. In no event shall a license of limited duration expire later than the expiration of the authorization for the applicant's legal presence in the United States. A drivers license issued to an H-2A worker expires three years after the date of issuance of the H-2A worker's visa; provided, if at any time during that three-year period an H-2A worker's visa duration is not extended by United States Citizenship and Immigration Services, the license expires on the date the H-2A worker's visa expires. For purposes of this subdivision, the term "H-2A worker" means a foreign worker who holds a valid H-2A visa pursuant to the Immigration and Nationality Act (8 U.S.C. § 1101(a)(15)(H)(ii)(a)) and who is legally residing in this State."

SECTION 12.(b) This section is effective when it becomes law and applies to applications for licenses submitted on or after that date.

AG COST SHARE TECHNICAL CORRECTION

SECTION 13A. G.S. 106-850(b)(2) reads as rewritten:

"(2) The program shall ~~initially include the present 16 nutrient sensitive watershed counties and 17 additional counties.~~include the entire State."

SEVERABILITY CLAUSE AND EFFECTIVE DATE

SECTION 14.(a) If any provision of this act or the application thereof to any person or circumstances is held invalid, such invalidity shall not affect other provisions or applications of this act that can be given effect without the invalid provision or application, and, to this end, the provisions of this act are declared to be severable.

SECTION 14.(b) Except as otherwise provided, this act is effective when it becomes law.

In the General Assembly read three times and ratified this the 30th day of June, 2021.

s/ Phil Berger
President Pro Tempore of the Senate

s/ Destin Hall
Presiding Officer of the House of Representatives

s/ Roy Cooper
Governor

Approved 12:05 p.m. this 2nd day of July, 2021

ATTACHMENT 20

TECHNICAL PAPER

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Characterizing Ammonia Emissions from Swine Farms in Eastern North Carolina: Part 2—Potential Environmentally Superior Technologies for Waste Treatment

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ABSTRACT

The need for developing environmentally superior and sustainable solutions for managing the animal waste at commercial swine farms in eastern North Carolina has been recognized in recent years. Program OPEN (Odor, Pathogens, and Emissions of Nitrogen), funded by the North Carolina State University Animal and Poultry Waste Management Center (APWMC), was initiated and charged with the evaluation of potential environmentally superior technologies (ESTs) that have been developed and implemented at selected swine farms or facilities. The OPEN program has demonstrated the effectiveness of a new paradigm for policy-relevant environmental research related to North Carolina's animal waste management programs. This new paradigm is based on a commitment to improve scientific understanding associated with a wide array of environmental issues (i.e., issues related to the movement of N from animal waste into air, water, and soil media; the transmission of odor and odorants; disease-transmitting vectors; and airborne pathogens). The primary focus of this paper is on emissions of ammonia (NH_3) from some potential ESTs that were being evaluated at full-scale swine facilities. During 2-week-long periods in two different seasons (warm and cold), NH_3 fluxes from water-holding structures and NH_3 emissions from animal houses or barns were measured at six potential EST sites: (1) Barham farm—in-ground ambient temperature anaerobic digester/energy recovery/greenhouse vegetable production system; (2) BOC #93 farm—upflow biofiltration system—EKOKAN ; (3) Carrolls farm—aerobic blanket system—ISSUES-ABS; (4) Corbett #1 farm—solids separation/gasification for energy and ash recovery centralized system—BEST; (5) Corbett #2 farm—solid separation/reciprocating water technology—ReCip; and (6) Vestal farm—Recycling of Nutrient, Energy and Water System—ISSUES-RENEW. The ESTs were compared with similar measurements made at two conventional lagoon and spray technology (LST) farms (Moore farm and Stokes farm). A flow-through dynamic chamber system and two sets of open-path Fourier transform infrared (OP-FTIR) spectrometers measured NH_3 fluxes continuously from water-holding structures and emissions from housing units at the EST and conventional LST sites. A statistical-observational model for lagoon NH_3 flux was developed using a multiple linear regression analysis of 15-min averaged NH_3 flux data against the relevant environmental parameters measured at the two conventional farms during two different seasons of the year. This was used to

IMPLICATIONS

Current estimates indicate that atmospheric NH_3 emitted from North Carolina swine facilities account for approximately 46% of the state's atmospheric NH_3 emission. As part of an agreement between the state of North Carolina and two animal production agriculture companies, some potential ESTs were evaluated for NH_3 emissions. This paper describes the evaluation of six potential ESTs using the statistical-observational model developed for NH_3 emissions from the conventional LST currently in use for managing swine waste. The evaluated alternative technologies may require additional technical modifications to be qualified as unconditional ESTs relative to NH_3 emissions reductions.

compare the water-holding structures at ESTs with those from lagoons at conventional sites under similar environmental conditions. Percentage reductions in NH_3 emissions from different components of each potential EST, as well as the whole farm on which the EST was located were evaluated from the estimated emissions from water-holding structures, barns, etc., all normalized by the appropriate nitrogen excretion rate at the potential EST farm, as well as from the appropriate conventional farm. This study showed that ammonia emissions were reduced by all but one potential EST for both experimental periods. However, on the basis of our evaluation results and analysis and available information in the scientific literature, the evaluated alternative technologies may require additional technical modifications to be qualified as unconditional ESTs relative to NH_3 emissions reductions.

INTRODUCTION

The scientific attention given to atmospheric ammonia (NH_3) and its roles in both atmospheric chemistry and eutrophication of ecosystems has grown during the last decade.[1–7] It has been recognized that NH_3 is responsible for neutralizing the acids produced by the oxidation of sulfur (S) and nitrogen (N). This neutralization process results in the formation of atmospheric aerosol containing ammonium (NH_4^+), which may be of concern in increasing fine particulate matter concentration.[8–12] Swine farms are a significant source of NH_3 in eastern North Carolina. Lagoon and spray technology (LST) is the conventional and current system used in North Carolina to manage swine waste. It consists of anaerobic lagoons that store and biologically treat the swine waste (~98% liquid), and the effluent from these lagoons is periodically pumped and sprayed as a nutrient source on surrounding crop fields.[1] Many sensitive ecosystems lie within approximately 100 km of NH_3 area sources in North Carolina. Ecosystems in proximity to high NH_3 emission sources and NH_3/NH_4^+ deposition are subject to potential environmental consequences, including aquatic eutrophication and soil acidification.

The need for developing environmentally superior and sustainable solutions for the management of animal waste is vital for the future of animal farms in North Carolina, the United States, and the world. In addressing that need, the North Carolina Attorney General initiated the development, implementation, and evaluation of environmentally superior swine waste management technologies (ESTs) that would be appropriate to each category of swine farms in North Carolina. This evaluation was done through agreements between the Attorney General of North Carolina with Smithfield Foods, Inc. and Premium Standard Farm, Inc. Those agreements provided funds for research to develop and evaluate ESTs through the Animal and Poultry Waste Management Center (APWMC) at North Carolina State University, Raleigh, NC.[13] The agreements define “Environmentally Superior Technology or Technologies” as any technology, or combination of technologies that (1) is permittable by the appropriate governmental authority; (2) is determined to be technically, operationally, and economically feasible for an identified category or categories of farms (to be described in a technology determination); and (3) meets the following performance standards:

- Eliminates the discharge of animal waste to surface waters and groundwater through direct discharge, seepage, or runoff;
- Substantially eliminates atmospheric emission of NH_3;
- Substantially eliminates the emission of odor that is detectable beyond the boundaries of the parcel or tract of land on which the swine farm is located;
- Substantially eliminates the release of disease-transmitting vectors and airborne pathogens; and
- Substantially eliminates nutrient and heavy metal contamination of soil and groundwater.

Program OPEN (Odor, Pathogens, and Emissions of Nitrogen) was initiated as an integrated study of the emissions of NH_3, odor and odorants, and pathogens from potential ESTs for swine facilities. Its main purpose was to evaluate potential ESTs that have been developed and implemented under an agreement between the North Carolina Attorney General and the participating companies that own approximately 10% of the swine farms in North Carolina, using the conventional LST. Under this program, ESTs implemented at selected swine facilities were evaluated to determine if they would be able to substantially reduce atmospheric emissions of NH_3, odor, and pathogens. This study focuses on the emissions of N in the form of NH_3 from different components/processes involved in swine waste handling and treatment, including waste storage lagoons, swine houses, and spray fields at six selected EST sites. These are described below in the following format: name of the farm where the potential EST was used, type of technology, and brand name where applicable.

(1) Barham farm: in-ground ambient temperature anaerobic digester/energy recovery/greenhouse vegetable production system;
(2) BOC #93 farm: upflow biofiltration system—EKOKAN;
(3) Carrolls farm: aerobic blanket system—ISSUES-ABS;
(4) Corbett #1 farm: solids separation/gasification for energy and ash recovery centralized system—BEST;
(5) Corbett #2 farm: solid separation/reciprocating water technology—ReCip;
(6) Vestal farm: Recycling of Nutrient, Energy and Water System—ISSUES-RENEW.

These potential ESTs were evaluated during two seasons (cool and warm), and the results are compared and contrasted with data from two conventional LST swine farms (Moore farm and Stokes farm), which have been described in the companion paper.[1] The evaluation of two other potential ESTs, qualified as unconditional ESTs relative to NH_3 emissions reductions, are described in another paper.[14]

NH_3 fluxes from water-holding structures and other area sources and NH_3 emissions from animal houses at all of the ESTs and conventional farms were measured by a dynamic flow-through chamber system and open-path Fourier transform infrared (OP-FTIR) spectroscopy.

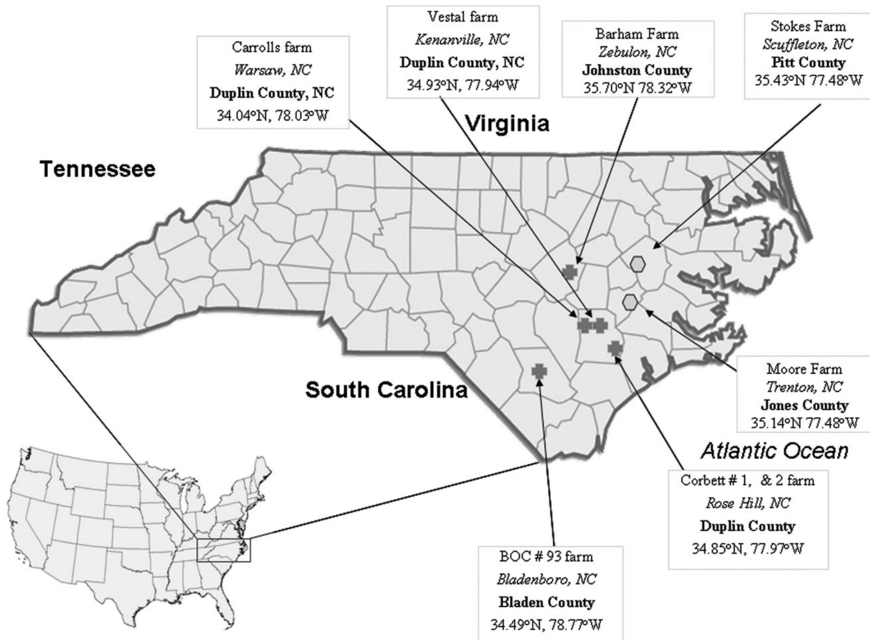

Figure 1. Map of North Carolina indicating the location of the ESTs and LSTs.

Recent studies, using a mass balance approach to estimate NH_3 emission rates, found that swine houses represent a more significant source than previously hypothesized.[15] On the basis of a review of published data, the loss of NH_3 from swine houses was estimated to be around 15% of total N excreted.[16] Griffing et al.[17] used the mass balance method to estimate that approximately 80% of NH_3 loss was due to volatilization from liquid waste storage systems. In this study also, the mass balance approach was used to estimate NH_3-N emissions from different components of the EST and LST farms, as well as N excretion rates, on the basis of swine population and feed data. Normalizing emissions by N excretion rate, percentage reductions in NH_3-N emissions are determined for water-holding structures, barns, and the whole farm for each EST facility from their estimated values for the appropriate LST farm.[1]

EXPERIMENTAL SETUPS AT EST SITES

Sampling Sites and Periods

NH_3 flux measurements were conducted during two different seasons (warm and cool) at eight swine farms (two conventional sites and six EST sites) in eastern North Carolina (for location see Figure 1). Two conventional sites (Stokes farm and Moore farm, i.e., LSTs) are also referred to as "baseline" sites for comparison with EST sites. The six EST sites were Barham, BOC #93, Carrolls, Corbett #1, Corbett #2, and Vestal farms, respectively. Aneja et al.[1] have given a detailed description of the two baseline farms with LST, as well as instrumentation and sampling techniques and scheme, therefore this information will not be repeated.

OP-FTIR measurements were conducted to measure the NH_3 flux from the ventilation systems at the swine houses. Of the swine houses measured, there were two different types of ventilation systems, namely, mechanical or tunnel ventilation and natural ventilation. At the Barham, BOC #93, and Carrolls farms, mechanical ventilation was used. At the Corbett #1 and #2 farms and the Vestal farm, the swine houses had natural ventilation. The methodologies for the measurement of NH_3-N emissions from the swine barns are described in Aneja et al.[1]

A brief description of each of the potential ESTs that have been evaluated is provided here. Williams,[18] and Williams,[19] contain more detailed information including site plans, design schematics, and projected operational characteristics.

Barham Farm. Barham Farm (35.70 °N, 78.32 °W, 130 m mean sea level [MSL]) is located near Zebulon, NC, in Johnston county. Field campaigns were conducted during April 1–12, 2002 and November 11–22, 2002, at this farm site. However, during the first measurement period in April we were notified that the EST was not fully functioning as designed, because the biofilters were not operational during that time. A schematic layout of the EST at Barham farm including the various sampling locations is given in Figure 2. This potential EST has an in-ground ambient digester comprised of a covered anaerobic waste lagoon. The primary lagoon was covered by an impermeable layer of 40-mm thick high-density polypropylene that prevented gaseous methane and other gases and odor from escaping into the atmosphere during the digestion process. Methane gas that is produced during the digestive process was extracted and burned into a biogas generator to produce electricity. Heat from the generator was captured and used to produce hot water that was used by the farm in its production activities. Effluent from the digester (covered lagoon) flowed into a storage pond with a surface area of 4459 m^2. This storage pond was formerly part of the primary anaerobic lagoon before the digester was built. A portion of this effluent was further treated via biofilters, the purpose of which was to convert NH_4^+ to nitrate in the effluent. This nitrified effluent was then used to flush out the swine production facilities, and the

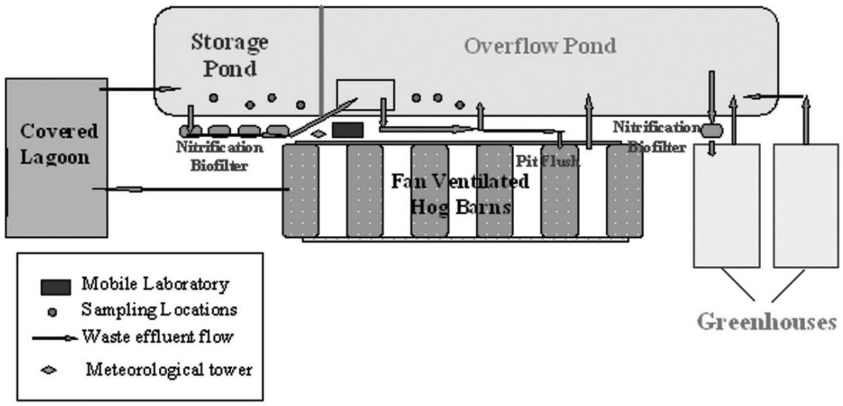

Figure 2. A schematic layout of the EST at Barham farm.

excess effluent was channeled into the larger overflow pond with a surface area of 19,398 m^2. A heavy polymer baffle separated the overflow and storage ponds. The overflow pond was used to store rainwater and overspills from the storage pond. Water from the overflow pond was also pumped into a nitrification biofiltration system where the nutrients in the treated effluent were used to fertilize vegetables grown in greenhouses adjacent to the swine production facility.

In this study NH_3 flux measurements were made from the surfaces of the storage pond, the overflow pond, and from the covered anaerobic primary lagoon. Average NH_3 concentrations were measured using the OP-FTIR spectroscopy system across the forced ventilation fan openings, as well as along the sides of swine houses (barns) to estimate barn emissions during the experimental periods.

BOC #93 Farm. BOC #93 farm (34.49 °N, 78.77 °W) is located near Bladenboro, NC, in Bladen county. NH_3 measurements were conducted on March 31, April 11, 2003, and June 16–27, 2003. A schematic layout of the EST at BOC #93 farm including the various sampling points is given in Figure 3.

The EKOKAN waste treatment system consisted of solids/liquid separation and biofiltration of the liquid with upflow aerated biological filters. Five finishing barns were connected to the waste treatment system, and the barn pits were emptied automatically in sequence. Wastewater from the barn pits was released to a solids separation unit. Coarse solids were separated from the wastewater using a screen separator (TR separator). After the solids/liquid separation process, the liquid was pumped to a 40,000-gal equalization tank. Liquid flowed from the equalization tank by gravity and passed through first- and second-stage aerated biofilters connected in series (two sets). Wastewater flowed upward through the biofilters, and air was supplied at the bottom of each biofilter with blowers. The biofilter tanks were covered, and air and any excess foam from the aerated treatment were routed through polyvinyl chloride pipes to exit points over an anaerobic lagoon. The biofilters were backwashed periodically to remove excess biosolids. Treated effluent from the biofilters flowed by gravity to a storage basin, with a portion of the treated effluent being recycled to the solids separation basin, from which it was pumped to the equalization tank, which had a surface area of 28.3 m^2. Water was pumped from the storage basin to the barns to refill the pits. At this site, the anaerobic lagoon that received manure from 10 barns was partitioned using plastic curtains into three sections, with one section much larger than the other two. The larger section received manure from five barns not connected to the EKOKAN treatment system. One of the smaller sections received any overflow from the solids separation basin, the separated solids, and

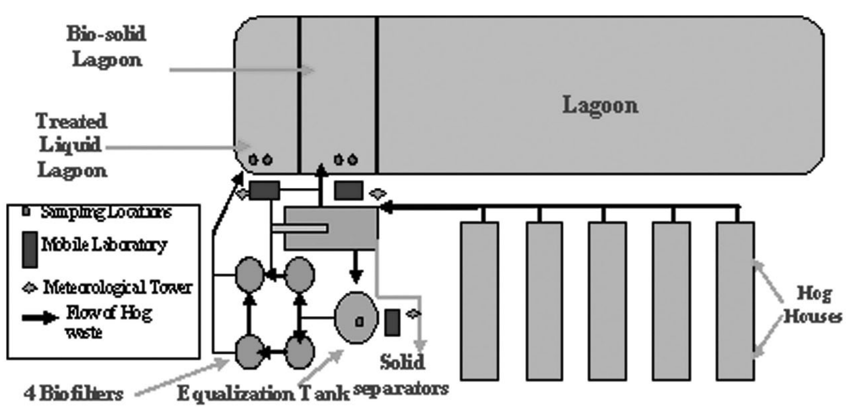

Figure 3. A schematic layout of the potential EST at BOC #93 farm.

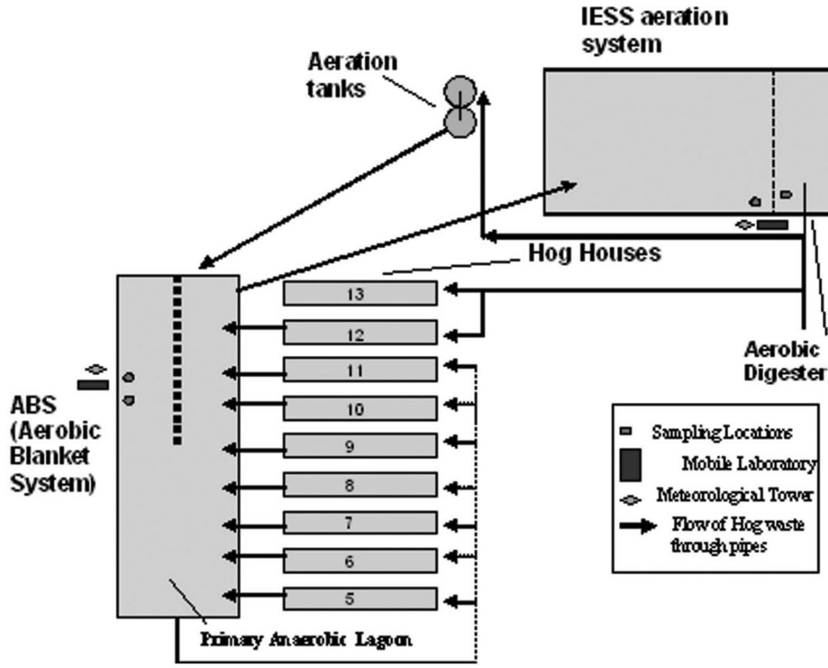

Figure 4. A schematic layout of the potential EST at Carrolls farm.

the backwashed biosolids that were removed from the biofilters. This was known as the biosolids lagoon and had a surface area of 3229.2 m^2. The other small section received the treated effluent from the biofilters. This was known as the treated effluent lagoon and had a surface area of 1614.6 m^2. NH_3 fluxes were measured from the treated effluent storage, the biosolids storage lagoon, and the equalization tank during the experimental periods.

Carrolls Farm. Carrolls (34.04 °N, 78.03 °W) is located near Warsaw, NC, in Duplin county. NH_3 flux measurements were conducted on this farm from March 29 to April 2, 2004 and from June 28 to July 2, 2004, respectively. A schematic layout of the EST at Carrolls farm including the various sampling points is given in Figure 4.

The waste stream in the proposed EST flows from the houses to a primary anaerobic lagoon equipped with the ABS. This is known as the ABS lagoon and has a surface area of 3304.8 m^2.

The ABS consists of a fine mist of treated swine waste that is applied every 15 min to the surface of the anaerobic lagoon. During both evaluation periods, only half of the anaerobic lagoon was being treated by the ABS. The treated swine waste arises from an aeration treatment that takes place in an adjoining water-holding structure (aerobic digester). Waste from the anaerobic lagoon flows into an aerobic digester (IESS aeration system). This is referred to as the west side of the aerated lagoon and has a surface area of 5068.8 m^2. This portion of the basin is sectioned off with a plastic barrier. The aerated waste eventually flows into the sectioned-off portion of the aeration treatment basin. This is known as the east side of the aerated lagoon, and has a surface area of 6010.2 m^2. The waste is then used to flush the animal houses and supplies the treated water for the ABS. During the first evaluation period, the IESS aeration system was not functioning and treated waste for the ABS was derived by using two aeration treatment tanks. For the second evaluation, the aeration treatment basin was operating as designed. Only waste from finishing houses 5–13 flowed into the ABS-equipped anaerobic lagoon. Waste from the remaining farrow and weaning houses flowed into a separate lagoon. These houses and their accompanying lagoon were not included in the evaluation of the EST.

Corbett #1 Farm. Corbett #1 farm (34.85 °N, 77.97 °W) is located near Rose Hill, NC, in Duplin county. NH_3 flux measurements were conducted during October 1–8, 2003, and December 4–7, 2003, respectively. A schematic layout of the EST at Corbett #1 farm including the various sampling points is given in Figure 5.

Manure flushed from the barns flows first to a collection pit, then to an aboveground feed tank, and then to a screw-press separator on a raised platform. The separator has a screen with 0.25-mm openings. The liquid that flows through the screw-press separator screen flows to a second feed tank, which has a surface area of 27.1 m^2, and then to two tangential-flow gravity settling tanks sited parallel to each other. Each tangential-flow settling system consists of a 2.2-m diameter tank with a cone bottom followed by a 1.2-m diameter sludge thickening tank, also with a cone bottom. Tangential flow in the first tank causes solids to concentrate in the center of the tank and settle down to the bottom. This settled slurry is then pumped to the second tank for sludge thickening. For approximately 10 min every hour the settled slurry from the second tangential-flow settling tank is pumped back to the tank that feeds the screw-press separator, where the settled slurry is combined with the flushed manure that is being pumped to the screw-press separator. The treated

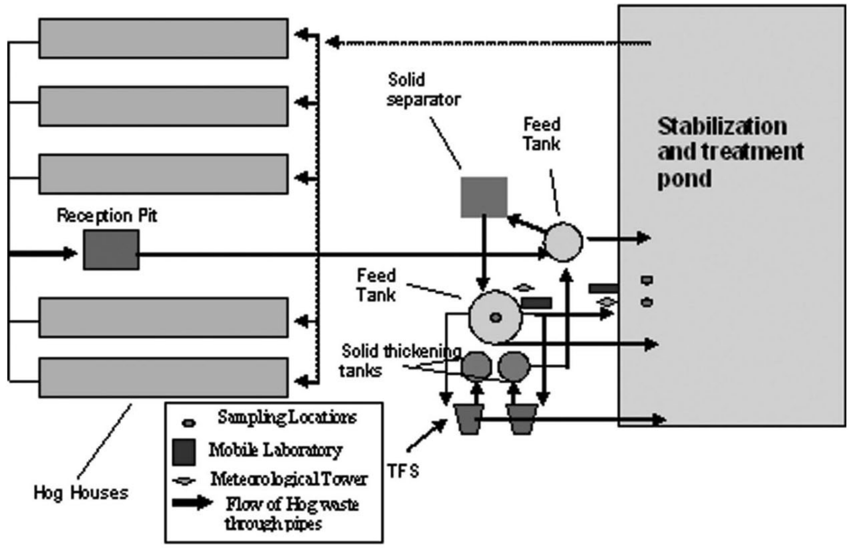

Figure 5. A schematic layout of the potential EST at Corbett #1 farm.

waste and any overflow go to a stabilization and treatment pond, which has an area of 8291.9 m^2.

Corbett #2 Farm. Corbett #2 farm (34.84 °N, 77.96 °W) is located near Rose Hill, NC, in Duplin county. Measurement campaigns were conducted during March 10–21 and June 2–13, 2003 at this farm site. A schematic layout of this potential EST including locations of flux measurements is given in Figure 6.

The ReCip encompasses two cells, or treatment basins, filled with media (proprietary technology), that would alternately drain and fill on a cyclic basis. The draining and filling cycles created aerobic, anaerobic, and anoxic conditions within the cells, providing both biotic and abiotic treatment processes to promote nitrification and denitrification. The treatment process was preceded by a solids separation step. The solid waste and the treated liquid waste went into individual lagoons, which had surface areas of 2601 m^2 and 2717 m^2, respectively. The ReCip project at the evaluation time was designed to treat only the liquid portion of the swine waste.

Vestal Farm. Vestal farm (34.93 °N, 77.94 °W) is located near Kenansville, NC, in Duplin county. NH_3 flux measurements were conducted during March 16–18, 2004, and August 4–12, 2004, respectively. A schematic layout of the EST at the Vestal Farm including the various sampling points is given in Figure 7.

The RENEW system uses a mesophilic digester as well as aeration and wastewater filtering and disinfection systems. This project also incorporated a microturbine generator. For this system, the waste first flows from the pig

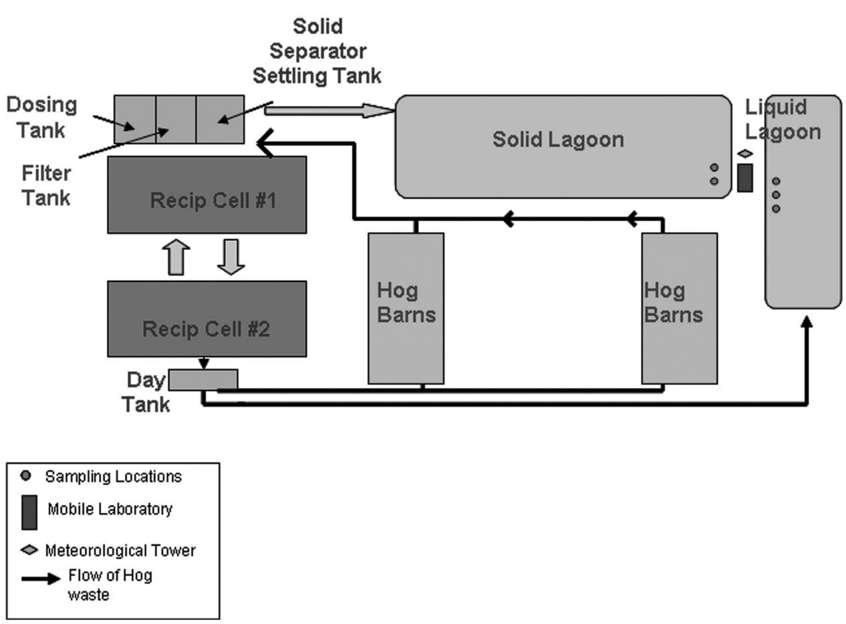

Figure 6. A schematic layout of the EST at Corbett #2 farm.

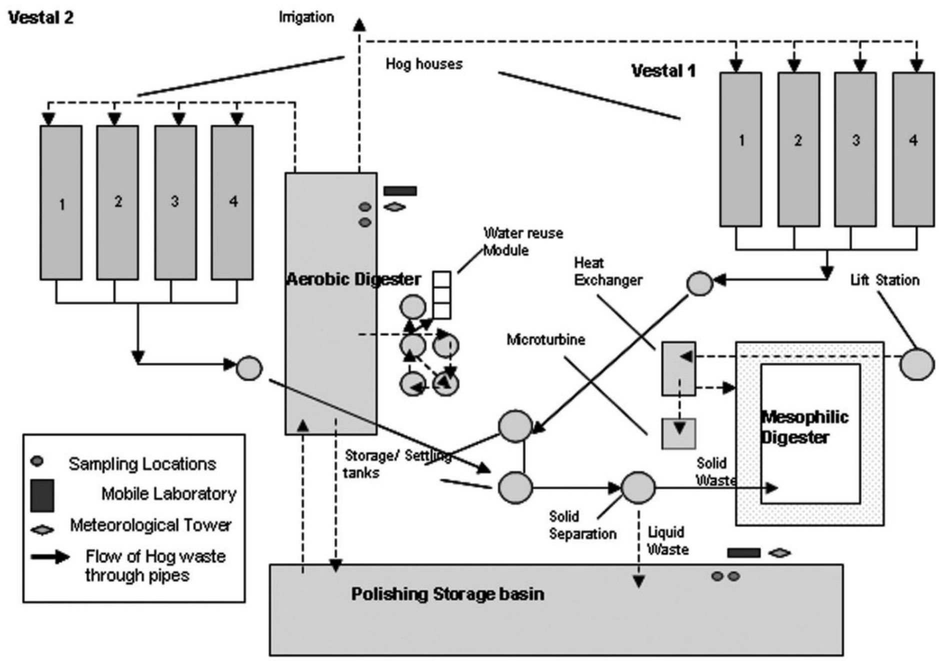

Figure 7. A schematic layout of the potential EST at Vestal farm.

barns to equalization and concentrator tanks, which serve to produce a thickened liquid. This liquid then flows to a mesophilic digester. The digester, which operates at a temperature of 95 °F, produces biogas, which is used to fuel the microturbine generator. The generator produces electricity, which is sold and used on the electric power grid. The waste stream then flows to a polishing storage basin, which has a surface area of 22,636 m^2, and then to an aerobic digester, also called a nitrification pond, which has a surface area of 1880.6 m^2. A portion of the waste stream then flows back to the polishing storage basin where it is used to flush the pig barns and is sprayed on cropland if necessary. The remaining portion of the waste stream flows through a filtration system. The filtration system consists of sand carbon filters and reverse osmosis. The water is then disinfected using ozonation and ultraviolet light. Filtered and disinfected water is then returned to the pig barns where it is used as drinking water for the pigs.

Approach to Evaluate NH_3 Emissions Reduction at EST Farms

At each EST and conventional site, the monitoring of NH_3 emissions were limited to about two 2-week periods, representing both a warm and a cool season. It was suggested that the estimated emissions from an EST for each measurement period be compared with the estimated emissions from conventional sites. However, because measurements at different sites were made at different times of the year, environmental conditions are likely to be different at different sites, even during a representative warm or cool season. Thus, there is a need for accounting for these differences in our relative comparisons of the various alternative and conventional technologies.

A rational basis for this adjustment for somewhat different environmental conditions is the development of a statistical-observational model based on multiple regression. This is developed between NH_3 emissions and measured environmental parameters at the two conventional sites.[1] Such a comparison does not require highly uncertain extrapolations of emissions at EST sites beyond the two measurement periods. It also provides a sound basis for ranking the various ESTs on the basis of their comparisons with conventional sites for each of the warm and cold seasons.

Relationships between NH_3 flux and lagoon temperature, pH, and TKN, as well as certain environmental parameters were examined in the accompanying paper, Aneja et al.[1] over a relatively wide range of lagoon temperatures (~2–35 °C) and lagoon air temperature differences that were observed during the fall and winter field campaigns at both conventional farms.

The multiple regression equation based on flux measurement data from two conventional farms is given by Aneja et al.[1]

$$\mathrm{Log}_{10}\mathrm{F} = 3.8655 + 0.04491(\mathrm{T}_1) - 0.05946(D). \quad (1)$$

Here, F denotes the average NH_3-N emission from the conventional lagoon in $\mu g \cdot min^{-1}$/1000 kg-lw (lw = live weight), where T_l is the lagoon temperature in °C, and D is a hot-air variable that is equal to zero if lagoon is warmer than air, but is equal to $\Delta T = T_a - T_l$ when $T_a > T_l$ and T_a is air temperature in °C at 10-m height. This statistical-observational model was used to estimate the projected NH_3-N flux from lagoons at the LST baseline farms to compare with the measured NH_3-N flux from water-holding structures at an EST site, for the average values of T_l and D observed at the latter.

Aneja et al.[1] describes the development of the statistical-observational model in more detail.

Table 1. The summary of animal mass, feed consumed, N content, and N excretion at EST farms.

Farm Information	Number of Pigs	Average Pig Mass (kg/pig)	Total Pig Mass (kg)	Feed Consumed (kg/pig/week)	N Content (%)	*E* (kg-N/week/1000 kg-lw)
Barham						
April 2002	4,000	238.1	952,560	12.84	2.25[a]/3.09[b]	1.65
November 2002	4,000	238.1	952,560	15.92	2.38[a]/3.43[b]	1.77
BOC #93						
April 2003	4,221	82.7	348,994	11.93	2.78	2.82
June 2003	4,373	48.0	209,952	14.41	3.24	5.25
Carrolls						
March–April 2004	6,332	59.2	374,854	12.89	2.56	3.90
June 2004	6,095	59.7	363,872	13.21	2.67	4.13
Corbett #1						
October 2003	3,386	55.4	187,584	15.44	3.01	5.86
December 2003	2,680	104.7	280,596	16.27	2.15	2.34
Corbett #2						
March 2003	1,249	98.5	123,054	16.27	2.76	3.19
June 2003	1,485	70.3	104,396	14.47	3.08	4.50
Vestal						
March 2004	9,507	38.3	364,118	10.03	2.79	5.03
August 2004	10,248	44.7	458,086	11.02	3.17	5.47

Notes: All farms are finishing operations except Barham farm, which is a farrow-to-wean operation with a mixture of sows and mature pigs. [a]N content of the feed in gestation houses; [b]N content of the feed in farrowing houses.

Estimated NH_3 emission from animal houses at a potential EST were compared with the estimated NH_3 emissions from similar houses at a conventional farm (either Moore farm—tunnel ventilated, or Stokes farm—naturally ventilated), depending on the type of the house ventilation used at the EST farm, for the same season.

Both EST emissions and conventional NH_3 emissions were normalized by the N excretion rate (E) for the farm, and are called %E. On the basis of the N mass balance equation with the given animal feed information (Table 1), E in units of kg-N week^{-1} (1000 kg-lw)$^{-1}$ was determined using the following equation:

$$E = \frac{F_c \times N_f \times (1 - e_r)}{\bar{w}} \times 1000, \qquad (2)$$

where F_c is the feed consumed (kg pig^{-1} · week^{-1}), N_f is the fraction of N content in feed, e_r is the feed efficiency rate (ratio of average gain of N to N intake),[20] and $\bar{w}$ is the average live animal mass (kg/pig). The N excretion data are presented in Table 1. N excretion and NH_3-N emissions at each farm was calculated in the same units (kg-N · week^{-1} (1000 kg-lw)$^{-1}$) , thus, %E represents the loss rate of NH_3 from a source as a percentage of N-excretion rate. A potential EST was evaluated by comparison of %E value from the EST (%E_{EST}) farm to %E value from a baseline conventional farm (%E_{CONV}), and percent reduction of NH_3-N can be estimated as

$$\%\ reduction = \frac{(\%E_{CONV} - \%E_{EST})}{\%E_{CONV}} \times 100 \qquad (3)$$

Such percentage reductions can be estimated, separately for water-holding structures, animal houses/barns, etc., as well as for the whole EST farm. An algorithmic flow diagram for the evaluation of NH_3 emissions from water-holding structures at the EST farms is shown in Figure 8.

RESULTS AND DISCUSSION

Temporal Variations of Lagoon NH_3 Fluxes

Average NH_3 fluxes from water-holding structures at EST farms during the measurement periods are summarized in Table 2. The NH_3 flux results from all water-holding structures showed strong seasonal variation with significantly higher flux during the warm season than during the cold season at the EST farms. Seasonal differences in water-holding structure NH_3 fluxes are revealed from the composite hourly average fluxes measured at all of the EST farms, in which higher NH_3 fluxes with more clear diurnal variations were found during the warm season. Typical diurnal trends for the water-holding structure NH_3 fluxes showed low fluxes during the morning hours that increased with time during the early part of the day as the air and lagoon temperatures increased after sun rise, attaining maximum values around 3:00 p.m. and then decreasing during the evening hours. This trend was found to follow approximately the diurnal trends of air and lagoon temperatures at the experimental farms. An example of these patterns is shown in the composite hourly averaged NH_3 fluxes from the storage lagoons at Barham and Corbett #2 farms (Figure 9, a and b).

NH_3 Emissions from Water-Holding Structures

Average fluxes and total estimated emissions from water-holding structures are presented in Table 2. Flux measurements of NH_3 at the Barham farm were conducted from storage and overflow lagoons at the farm. For the April 2002 measurement campaign, the EST at the farm was not fully functioning and did not achieve steady state. The fluxes from the water-holding structures were found to be

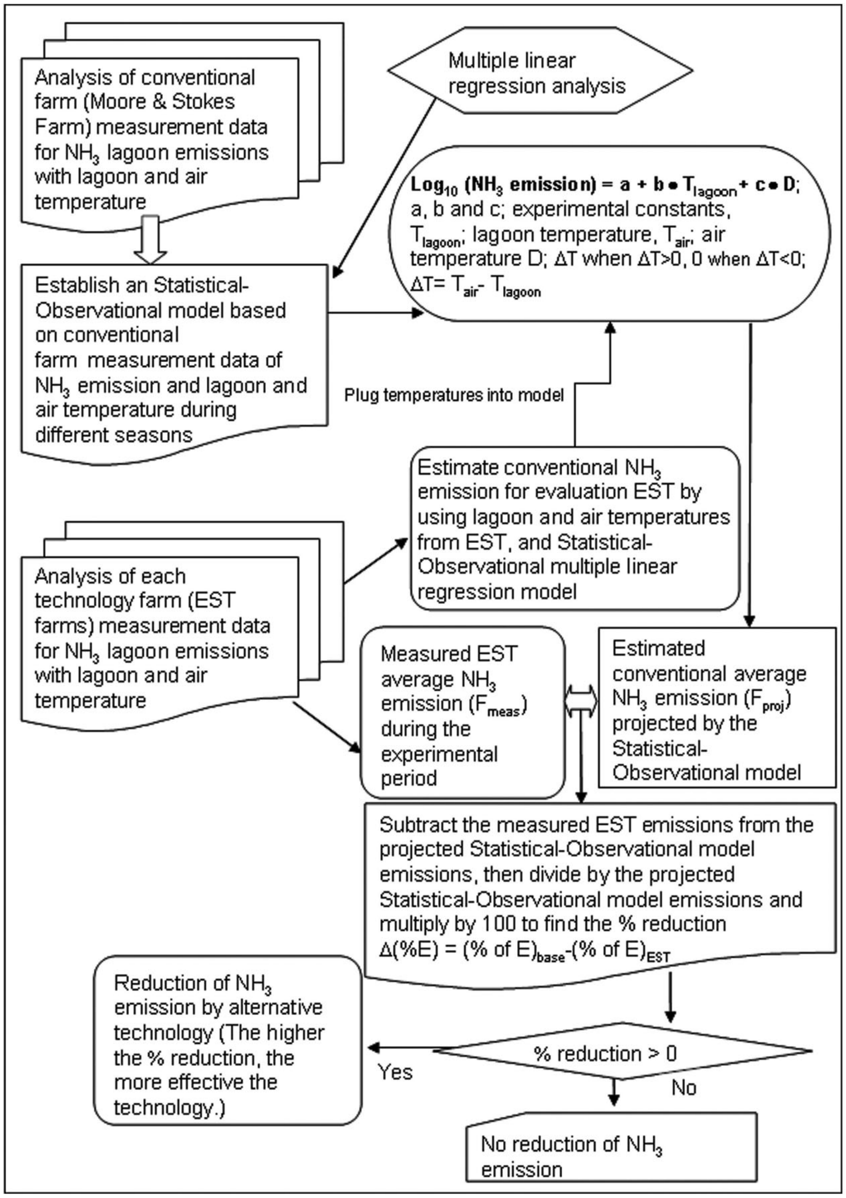

Figure 8. Algorithm flow chart for evaluation of EST NH_3 emissions from water-holding structures.

higher in the warm season than in the cold season. In the warm season, the overflow lagoon had a slightly higher flux, whereas in the cool season the storage lagoon had a higher flux. Overall, the weekly emissions were three to five times higher in the overflow lagoon than in the storage lagoon. This is a result of the overflow lagoon's area being approximately five times larger.

At BOC #93 farm, fluxes were measured from three water-holding structures: (1) the treated effluent lagoon, (2) the biosolids lagoon, and (3) and the equalization tank. In the warm season, the fluxes were higher for both the treated effluent lagoon and the biosolids lagoon than in the cool season, but the flux was lower in the warm season for the equalization tank. This is because the tank has a controlled temperature. For the cool season evaluation, the average fluxes from the three water-holding structures are very similar, with the treated effluent lagoon having the highest flux. Overall the emissions were highest from the biosolids lagoon, which had the largest surface area. The equalization tank had the smallest (negligible) emissions of the water-holding structures because of its small surface area.

At Carrolls farm, NH_3 fluxes were measured from the ABS lagoon, and from the east and west sides of the aeration lagoon. For both the warm season and the cool season, the ABS lagoon had the highest flux. Flux was higher in the warm season than the cool season for the ABS lagoon. A different pattern was observed for the aeration lagoon, where the fluxes were unusually higher in the cool season. Emissions followed the same pattern as fluxes, with the ABS having significantly larger emissions than the aeration lagoon.

Corbett #1 contained two water-holding structures: a stabilization lagoon and a feed tank. For the warm season evaluation, the flux was four times higher in the feed tank than in the stabilization lagoon; however, emissions were

Table 2. Estimated NH_3 emissions from water-holding structures at six EST farms during the experimental periods.

Farm Name and Sampling Period	Water-Holding Structure	Average 15-min Flux (μg NH_3-N m^{-2} min^{-1})	Water-Holding Structure Surface Area (m^2)	Weekly NH_3 Emissions (kg-N/week)	Total Emissions from Water-Holding Structures (kg-N/week)	Total Emission/Pig (kg-N/pig/week)	Total Emission/1000 kg-lw (kg-N/1000 kg-lw/week)
Barham April 2002	Storage lagoon	1101.9 ± 64.2	4,459	49.53	293.08	0.073	0.31
	Overflow lagoon	1245.6 ± 175.1	19,398	243.55			
Barham November 2002	Storage lagoon	435.8 ± 39.2	4,459	19.59	65.36	0.016	0.07
	Overflow lagoon	234.1 ± 34.0	19,398	45.77			
BOC #93 April 2003	Treated effluent lagoon	1711.0 ± 329.6	1614.6	27.92	79.21	0.019	0.23
	Biosolids lagoon	1556.5 ± 430.1	3229.2	50.81			
	Equalization tank	1673.4 ± 515.8	28.3	0.48			
BOC #93 June 2003	Treated effluent lagoon	2473.3 ± 928.8	1614.6	40.36	122.1	0.028	0.58
	Biosolids lagoon	2491.5 ± 537.1	3229.2	81.32			
	Equalization tank	1474.5 ± 643.6	28.3	0.42			
Carrolls March–April 2004	East side of aeration lagoon	480.2 ± 93.0	3304.8	14.9	80.4	0.013	0.21
	West side of aeration lagoon	446.7 ± 123.1	6010.2	29.1			
	ABS lagoon	713.1 ± 106.3	5068.8	36.4			
Carrolls June–July 2004	East side of aeration lagoon	209.4 ± 14.4	3304.8	4.2	83.1	0.014	0.23
	West side of aeration lagoon	127.5 ± 32.9	6010.2	12.7			
	ABS lagoon	1295.8 ± 135.6	5068.8	66.2			
Corbett #1 October 2003	Stabilization lagoon	734.0 ± 246.7	8291.8	61.51	62.33	0.018	0.33
	Feed tank	2992.6 ± 109.0	27.1	0.82			
Corbett #1 December 2003	Stabilization lagoon	415.2 ± 84.2	8291.8	34.8	34.8	0.013	0.12
Corbett #2 March 2003	Solid storage lagoon	472.6 ± 174.6	2601	12.39	42.51	0.034	0.35
	Liquid storage lagoon	1100.5 ± 457.6	2717	30.12			
Corbett #2 June 2003	Solid storage lagoon/	1624.3 ± 558.9	2601	42.59	84.36	0.057	0.81
	liquid storage lagoon	1525.2 ± 469.2	2717	41.77			
Vestal March 2004	Aerobic digester	1010.7 ± 60.7	1880.6	19.2	149.9	0.016	0.39
	polishing storage basin	573.1 ± 136.7	22,636.0	130.8			
Vestal August 2004	Aerobic digester	840.6 ± 284.8	1880.6	15.9	490.7	0.048	1.07
	polishing storage basin	2080.7 ± 340.8	22,636.0	474.8			

two orders of magnitude higher from the stabilization lagoon. This is likely due to the stabilization lagoon having a significantly larger surface area. For the December measurement period, measurements were only made at the stabilization lagoon. This was because of the malfunction of the dynamic flow-through chamber used for the feed tanks. This is not thought to be a problem because of the relatively small emissions observed in the October evaluation period. For the stabilization lagoon, the flux was lower in the cool season than in the warm season.

NH_3 fluxes at Corbett #2 farm were measured from the solid storage and the liquid storage lagoons. Emissions for both of these water-holding structures were lower in the cool season than the warm season. In the cool season, the liquid storage lagoon had the higher flux. Conversely, in the warm season, the solid storage lagoon had the higher flux. Both water-holding structures have similar surface areas, and therefore the same pattern was repeated for the emissions.

Vestal farm contained two water-holding structures, an aerobic digester, and a polishing storage basin. For the polishing storage basin, the flux was higher in the warm season than the cool season. This pattern was not observed for the aerobic digester, where the emissions in the digester were slightly lower in the warm season than in the cool season. The larger area of the polishing storage basin resulted in much higher emissions for both the cool and warm periods.

Total emissions (kg-N/1000 kg-lw/week), normalized by live animal mass at the farm, were calculated for each experimental period for each EST farm (see Table 2). The emissions for the cool season for all EST farms ranged from 0.07 to 0.39 kg-N/1000 kg-lw/week, with a mean value of 0.23 kg-N/1000 kg-lw/week. The emissions for

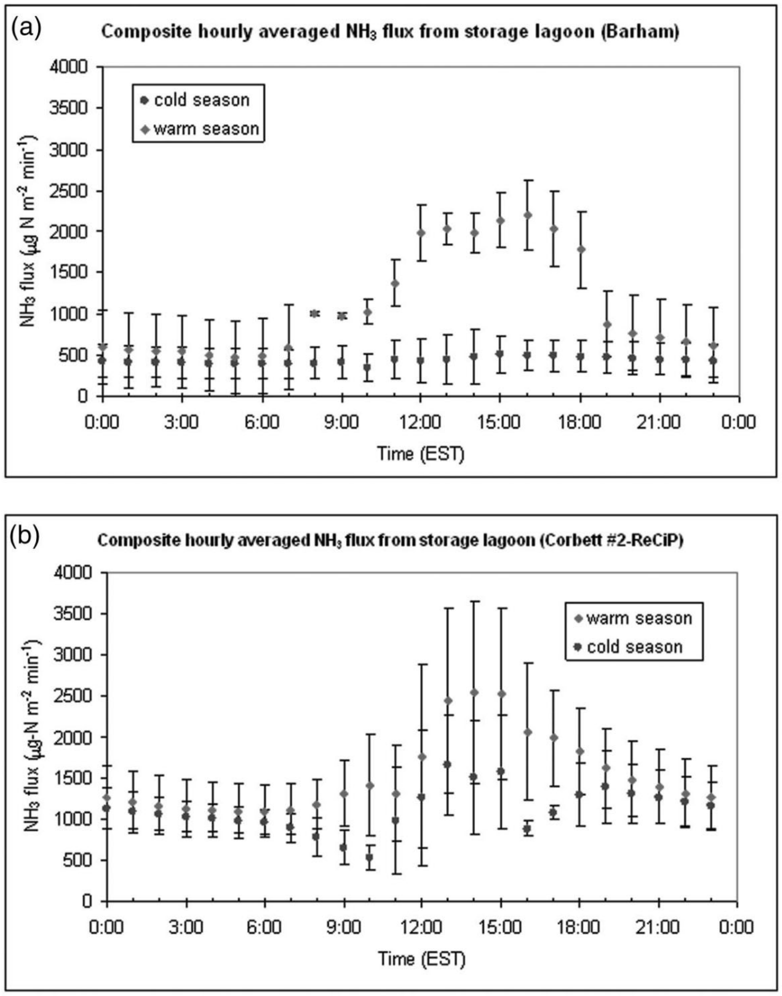

Figure 9. (a) Composite hourly averaged NH_3 flux from storage lagoon during April (warm season) and November (cold season), 2002 measurement periods at Barham farm. Error bar indicates ±1 standard deviation. (b) Composite hourly averaged NH_3 flux from storage lagoon during March (cold season) 2002 and June (warm season) 2003 measurement periods at Corbett #2 farm. Error bar indicates ±1 standard deviation.

the warm season for all EST farms ranged from 0.31 to 1.07 kg-N/1000 kg-lw/week, with an average value of 0.56 kg-N/1000 kg-lw/week. For all six EST farms, the total emissions were much higher in the warm season than the cold season, with the exception of the Carrolls farm where the seasonal difference was small.

NH_3 Emissions from Housing Units (Barns)

Table 3 shows the overall averages for the NH_3 emissions estimated by OP-FTIR measurements made during the sampling periods from the swine barns at the six EST farms. Emissions for the cool season for all of the EST farms ranged from 0.008 to 0.98 kg-N/1000 kg-lw/week, with an average value of 0.37 kg-N/1000 kg-lw/week. For the warm season, emissions ranged from 0.16 to 1.29 kg-N/1000 kg-lw/week, with a mean value of 0.70 kg-N/1000 kg-lw/week. Higher emissions from the barns were experienced during the warm period at five of six EST farms; the exception was the Barham farm. Emissions from naturally ventilated barns were noticeably lower on days when the curtains were closed to block the wind and maintain heat in the barn.

Evaluation of Total NH_3 Emissions from EST Farms

To calculate the total percent reduction, the sum of projected emissions and measured emissions were taken for

Table 3. Estimated NH_3 emission from the swine houses at EST farms during the experimental periods (OP-FTIR measurements).

EST Farms	Sampling Periods	Barn Emissions (kg-N/1000 kg-lw/week)
Barham	April 2002	0.34
	November 2002	0.49
BOC # 93	April 2003	0.57
	June 2003	1.29
Carrolls	March–April 2004	0.98
	June–July 2004	1.15
Corbett # 1	October 2003	0.16
	December 2003	0.008
Corbett # 2	March 2003	0.12
	June 2003	0.52
Vestal	March 2004	0.07
	August 2004	0.75

Table 4. Summary of NH_3 emissions from the EST farms and percent reduction during the experimental periods.

EST Farms	Sampling Periods	Emission Sources	Measured Emission (F_{meas}; kg-N/week/ 1000 kg-lw)	$\%E_{EST}$	$\%E_{EST}$ (WHS + House)	EST Average Lagoon Temperature (°C)	EST Average D (°C)	Conventional Lagoon Emission (model/estimated; kg-N/week/ 1000 kg-lw(F_{proj})	$\%E_{CONV}$	$\%E_{CONV}$ (Lagoon + House)	Percent Reduction
Barham	April 2002	WHS	0.31	18.8	39.4	17.2	0.7	0.4	11.3	35.2	−11.9
		House	0.34	20.6				1.05	23.9[a]		
	November 2002	WHS	0.07	4.0	31.7	14.2	0.3	0.31	9.7	32.5	2.5
		House	0.49	27.7				0.89	22.8[b]		
BOC #93	April 2003	WHS	0.23	8.2	28.4	18.5	0.7	0.46	14.3	37.1	23.5
		House	0.57	20.2				0.89	22.8 [b]		
	June 2003	WHS	0.58	11.0	35.6	28.6	0.3	1.38	38.9	62.8	43.3
		House	1.29	24.6				1.05	23.9		
Carrolls farm	March–April 2004	WHS	0.21	5.4	30.5	15.0	0.0	0.34	10.6	33.4	8.7
		House	0.98	25.1				0.89	22.8[b]		
	June–July 2004	WHS	0.23	5.6	33.4	29.1	0.0	1.50	42.2	66.1	49.5
		House	1.15	27.8				1.05	23.9		
Corbett #1 farm	October 2003	WHS	0.33	5.6	8.3	21.8	0.2	0.69	19.4	29.4	71.8
		House	0.16	2.7				0.25	10.0[c]		
	December 2003	WHS	0.12	5.1	5.4	9.3	0	0.19	5.9	15.9	66.0
		House	0.008	0.3				0.25	10.0[c]		
Corbett #2 farm	September 2003	WHS	0.35	11.0	14.8	14.9	1.6	0.28	8.7	18.7	20.9
		House	0.12	3.8				0.25	10.0[c]		
	December 2003	WHS	0.81	18.0	28.9	24.1	1.0	0.78	22.0	32.0	9.7
		House	0.49	10.9				0.25	10.0[c]		
Vestal	March 2004	WHS	0.39	7.8	9.2	14.8	0.6	0.32	10.0	20.0	54.0
		House	0.07	1.4				0.25	10.0[c]		
	August 2004	WHS	1.07	19.6	33.3	28.5	0.3	1.36	38.3	48.3	31.1
		House	0.75	13.7				0.25	10.0[c]		

Notes: [a]NH_3 emission measured from barns at tunnel (fan) ventilated conventional farm (Moore farm) during October 2002; [b]NH_3 emission measured from barns at tunnel (fan) ventilated conventional farm (Moore farm) during February 2003; [c]NH_3 emission measured from barns at naturally ventilated conventional farm (Stokes farm) during January 2003. WHS = water-holding structures.

the water-holding structures and barns. These numbers were then used to calculate total percent reduction using the same process that was applied individually for water-holding structures and barns.

Table 4 shows the summary of the total NH_3 emissions measured from all six EST farms, along with the projected emissions from the LST farms and the percent reduction values for their evaluation of potential N reduction.

Out of six EST farms, five show percent reduction in NH_3 emissions for both experimental periods. The BEST technology used at Corbett #1 was the most successful, with a reduction of 71.8% in the warm season, and 66% in the cool season. The next largest percent reduction was the ISSUES-RENEW system used at Vestal farm, with percent reductions of 54 and 31.1% for cool and warm seasons, respectively. The next most effective was the EKO-KAN technology at BOC #93 farm, which had percent reductions of 23.5 and 43.3% for the cool and warm seasons, respectively. A further technology with reduction in both seasons was the ISSUES-PBS, which was located at Carrolls farm. There was a small reduction of 8.7% in the cool season, and a larger reduction of 49.5% in the warm season. The ReCip technology was only slightly effective, with small reductions in both seasons of 20.9 and 9.7%, respectively.

The technology at Barham farm was less effective with mixed results, reducing emissions slightly in one experimental period, but enhancing in the other. This could be the result of the EST at Barham farm not being fully functional. The technology did not achieve steady state until the April 2002 sampling period. However, this does not explain the insignificant percent reduction during the November 2002 experimental period.

CONCLUSIONS

A rational and functional approach to study NH_3 emissions at commercial-scale animal production agricultural farms was developed.

Six potential ESTs were evaluated to determine if they would substantially reduce atmospheric emissions of NH_3 at swine facilities from their estimated or projected emissions in comparison to what had been observed on two conventional LST swine farms during two different (warm and cold) experimental periods. Five of six farms showed varying amounts of percent reductions in NH_3 emissions for both experimental periods.

One of the five ESTs showed an appreciable percent reduction in NH_3 emissions for both periods. The technology used at Corbett #1 had the highest percent reductions of 71.8 and 66% for the warm and cool seasons, respectively.

However, on the basis of our evaluation results, analysis, and available information in the scientific literature, the evaluated alternative technologies may require additional technical modifications to be qualified as unconditional ESTs relative to NH_3 emissions reductions.

This study did not address the potential reductions in odor and pathogens by the potential ESTs. Other scientists in the OPEN project evaluated those environmental factors.

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ATTACHMENT 21

Compiled by Dr. Viney P. Aneja, PhD

September 3, 2021

There is abundant scientific evidence that swine concentrated animal feeding operations (CAFOs) operated in North Carolina have adverse impacts on the environment. One of these well-documented impacts are emissions of ammonia (NH_3) from hog waste lagoons, spraying and fields where the waste is land-applied. The ammonia emitted by these sources also deposits on the ecosystem, including surface waters where it can cause increased oxygen demand and eutrophication and can impact aquatic ecosystems and harm fish and other aquatic organisms. Airborne ammonia also deposits on land, where it can overload soil with nitrogen and increase nitrate leaching into groundwater, and make well water unsafe to drink. There is a concern that swine-waste biogas production could exacerbate these problems by emitting more reactive nitrogen, including ammonia, into the atmosphere. This report considers this risk and concludes that storing digestate in open lagoons and land-applying it to field may increase ammonia emissions from hog CAFOs and contribute to further degradation of local air and water quality.

I. Nitrogen cycle in North Carolina hog animal waste management systems

In North Carolina, Concentrated Animal Feeding Operations (CAFOs) are used extensively for meat production. Though the term Concentrated Animal Feeding Operation bears a technical definition under the Clean Water Act, here it is used to refer generally to a production model that raises large numbers of animals in confinement where they are fed and watered until they are slaughtered. Unlike traditional models of livestock husbandry, animals raised in CAFOs do not roam to forage and the feed is produced off-site. The CAFO model of production is used to produce beef, dairy, hogs, poultry, milk, and eggs. The majority of CAFOs in North Carolina produce are either broiler chickens or hogs.

North Carolina witnessed intense growth in its hog industry during the 1990s (Aneja et al., 2000). Due to the large number of animals raised in a concentrated location, CAFOs produce large volumes of waste (US EPA, 2004). In traditional animal production models, animal waste is deposited throughout the environment as the livestock forage. However, in the CAFO model of production, the waste accumulates within the barn. The most common system for disposing of this massive amount of waste is known as the lagoon and sprayfield system. In general, the floor of the swine barns is made of concrete with slats, allowing the urine and feces excreted by the hogs to fall into an underground storage pit below the barn. Depending on the design of the CAFO, the waste either remains in the pit for months before it is scraped out or is flushed out with lagoon water periodically. In North Carolina, there has been a significant shift towards flush systems that remove the waste from barns more frequently. At least three of the four hog operations at issue in this case use the flushing method of removing hog urine and feces from below the barn.[1]

In the conventional lagoon and sprayfield system, once waste is removed from the storage pit, it is transferred into an open-air retention pond or "lagoon" that stores millions of gallons of animal waste. This waste contains bacteria, nutrients such as phosphorus and nitrogen, and heavy metals including arsenic, cadmium, copper and zinc. The pH of waste in the lagoon is manipulated to favor bacteria that anaerobically digest the waste. The liquid waste rises to the top, and nutrient and elemental rich sludge forms at the bottom. The sludge is periodically removed and applied to land. The liquid waste is frequently applied as fertilizer to growing fields, known as spray-fields, via high-pressure sprayers. The waste may also be applied through other methods such as drag-hose application of waste to the surface of the land. Strategies such as injection, which incorporate the waste into the soil and help limit emissions, have not been widely adopted in North Carolina. The spray-fields grow crops such as hay and Bermuda grass in order to absorb the nutrients contained in the waste. In North Carolina, over 2,200 swine operations are permitted to use this kind of animal waste management system. Hog production in North Carolina is overwhelmingly centered in the Eastern Coastal Plain, particularly in Robeson, Columbus, Bladen, Sampson, Pender, Duplin, Onslow, Wayne, Lenoir, Greene, and Pitt counties.

One of the main drawbacks of managing waste in this manner is the effect on air and water quality. CAFOs are significant contributors to air pollution, which often disproportionately impacts low-income and minority communities (Wing et al., 2000). Hog barns, lagoons, land application (i.e. spraying) of animal waste, and land biogenic emissions all emit large quantities of ammonia and other pollutants into the atmosphere (Aneja et al. 2001, 2008, 2009).

Ammonia (NH3), a form of reactive nitrogen, is the most abundant gas-phase alkaline species in the atmosphere. Ammonia emissions from animal agriculture result from the degradation of urea by the ubiquitous enzyme urease, which results in ammonium (NH4+) formation Urea is mainly excreted in the animal urine and once it is hydrolyzed it is much more prone to ammonia emissions than organic nitrogen excreted in the feces.

Ammonia emitted by hog operations is transported and dispersed by wind and is deposited on surface waters or land through dry deposition or wet deposition (Figure 1) (Aneja et al., 2001). Multiple studies have modelled the dispersion patterns of ammonia from CAFOs in Eastern North Carolina (Walker et al., 2000; Costanza et al., 2008; Bajwa et al., 2008). These studies have established that ammonia produced by hog CAFOs deposits a significant amount of nitrogen into the Cape Fear River Basin.

When ammonia directly or indirectly deposits into surface waters it can cause algal blooms and eutrophication (Costanza et al., 2008; Aneja et al., 2001). These conditions in turn cause hypoxia—low oxygen levels—in rivers and streams that alters aquatic ecosystems and harms fish and other species (Costanza et al., 2008).

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[1] *See, e.g.,* Letter from Jeff P. Cappadona, Cavanaugh & Associates, to Christine Lawson, DEQ, Att. 1 Anaerobic Digester System O&M 3-4 (Jan. 30, 2020).

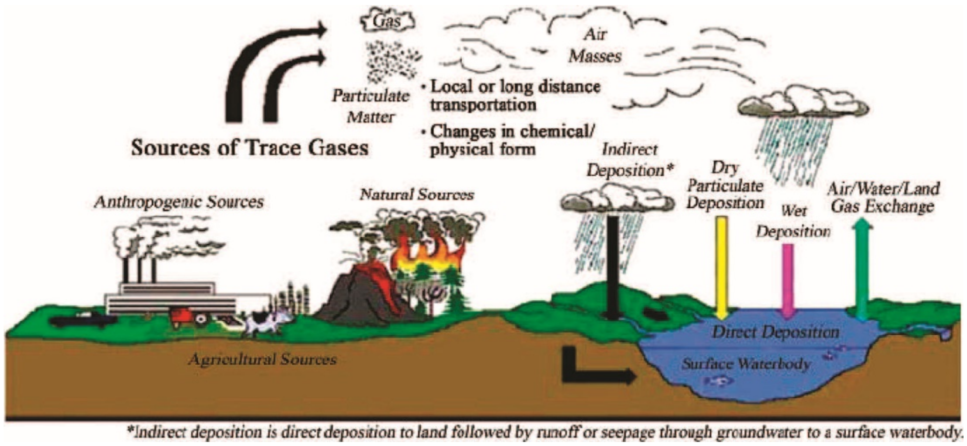

Fig. 1. Atmospheric emissions, transport, transformation and deposition of trace gases. (Aneja et al., 2001)

NH3 can neutralize sulfuric acid and nitric acid in the atmosphere to form fine particulate matter with an aerodynamic diameter <2.5 µm (PM2.5), which is closely linked to health and climatic effects. PM2.5 can penetrate deep into people's lungs and bloodstream and affect respiratory and cardiovascular health. PM2.5 exposure has been linked to a variety of health problems including heart attacks, irregular heartbeat, aggravated asthma, decreased lung function, premature death in people with heart or lung disease, and increased respiratory symptoms such as irritation of the airways, coughing, and difficulty breathing (EPA 2021). PM2.5 also has significant environmental effects, including formation of haze. PM2.5 can be carried over long distances by wind and settle on land or surface waters. When the PM2.5 containing ammonium/ammonia settles in surface waters it can increase the acidity or cause nutrient overloading, which leads to algal blooms and eutrophication.

High concentrations of ammonia, PM2.5, and other pollutants associated with hog CAFOs have a significant detrimental impact on the health and wellbeing of people living nearby. A study published by researchers from Duke University in 2018 found, after controlling for other factors, that for North Carolinians who live near hog CAFOs that use lagoons and sprayfields, mortality rates were substantially higher from causes such as anemia, kidney disease, tuberculosis, and lower birth rates than people who live further away from these operations (Kravchenko et al., 2018). Another recent study linked ammonia and particulate matter created by hog CAFOs to increased mortality rates in nearby communities (Domingo et al., 2021). CAFOs create areas of highly concentrated air pollution and odor that impairs the quality of life of nearby communities. Vulnerable populations are most at risk from the health impacts of CAFO-produced pollutants. Children, for instance, inhale 20-50% more air than adults, and air pollution can exacerbate existing health conditions in the elderly. In addition to affecting health, CAFO-produced pollution has substantial social impacts. Odor, for instance, may be detected several miles from CAFOs.

The waste treatment lagoons that CAFOs use to manage animal waste in North Carolina are a public hazard in times of extreme weather events e.g. hurricanes (Aneja et al., 2001). These events can cause lagoon overflows, which highly contaminate soil, surface water and groundwater used for wells with nutrients and pathogens impacting human health and the environment. Environmentally, these events cause extreme nutrient overload in waterways, which can have a negative impact on entire ecosystems by causing events like algae blooms.

II. Impact of retrofitting existing animal waste management systems to produce biogas on ammonia emissions

Anaerobic digestion (AD) is a method for converting biomass into bioenergy. Livestock manure is a commonly used biomass material for production of bioenergy.

Many livestock (hog and cattle) manure treatment systems rely on open lagoons where the CH_4, CO_2, NH_3 (ammonia) and other gases, such as reduced sulfur compounds, volatile organic compounds (VOCs) are emitted into the atmosphere. When these open systems are covered, gaseous emissions except ammonia are reduced, which results in the effluent leaving the anaerobic digester, known as digestate, with a modified chemical content (e.g. total solids, carbon, ammonia, ammonium (NH_4^+), and pH), relative to waste from a conventional open lagoon system. TAN content and pH in digested slurry are higher than in untreated slurry. Thus, potential for ammonia emissions during subsequent slurry storage are increased (Baines, 2021). The digestate contains more ammonium (NH_4^+) due to reduction in ammonia emissions from the anaerobic digester (i.e. covered lagoon) to the atmosphere, and has less degradable biomass carbon than the substrate in an open lagoon resulting in changes in GHG and NH_3 emissions (Baines, 2021).

The production of biogas through AD of livestock manure is a complex process. It involves a variety of physiological and biochemical metabolic pathways, the essence of which is the material and energy metabolism of microorganisms under anaerobic conditions. AD may be classified typically into three stages according to the utilization and transformation of organic matter (Baines, 2021):

1. Hydrolysis;

2. Acidogenesis; and

3. Methanogenesis.

In the hydrolysis step, macromolecular organic matters (fat, carbohydrate, protein, etc.) are hydrolyzed into small molecules such as monosaccharides, amino acids, fatty acids and so on by the action of extracellular enzymes. In the Acidogenesis step, the small molecule organic compounds are converted to a volatile organic acid, ethanol etc. by the acidified bacteria. H_2, CO_2 and acetic acid are then formed under the action of hydrogenic bacteria and acetogenic bacteria. Finally, in the Methanogenesis step, the methanogenic bacteria synthesize methane using acetic acid, H_2, CO_2 etc. in the methanogenesis stage.

In general, when the digestion process is complete, the pH of the digestate hovers between 7.5 and 8. The pH value in the course of anaerobic digestion is the result of the acid-alkali balance; the pH value decreasing with the increasing of organic acids, and increasing with the increases of Total Ammoniacal Nitrogen (TAN) (TAN = ammonium + ammonia), which is the product of the decomposition of nitrogenous organics (Lorimor and Sawyer, 2004; Grabow; Baines, 2021). The mass of total N is not significantly changed by anaerobic digestion, however the mass of organic nitrogen decreases and the mass of ammoniacal nitrogen increases. Organic N decreases as it is mineralized to TAN (Total Ammoniacal Nitrogen=NH4+N + NH3); i.e. NH3 N expressed as a percentage of TKN increases as manure is digested. Both the TAN and the mineral N (mineral N= TAN + NO3- N) increase. The impact of N transformations is to increase the fraction of total-N in the total ammoniacal form. This is important since the ammoniacal form is composed of both dissolved ammonium (NH4+) and ammonia (NH3) gas. Ammonia gas is emitted into the atmosphere from the open lagoon, i.e. secondary lagoon, during land application (i.e. spraying) of animal waste, and land biogenic emissions of the digested effluent. The amount of NH3 lost during land application of the digestate depends on multiple factors, including the method of application (generally high-pressure spray versus low pressure application or injection) and temperature. (Nyord et al. 2012; Dari et al., 2019).

Anaerobic digesters do not significantly change the nutrient quantity as nitrogen and phosphorus are retained, only the carbon is reduced through conversion and degassing of methane and carbon dioxide. The mass of organic nitrogen is decreased, and it is mineralized to TAN. Thus, ammonia and ammonium are found at higher concentrations in liquid digestates than raw manure (Nkoa, 2014). Therefore, anaerobic digestates stored in open secondary lagoons and land-applied to fields have higher NH3 emission potential than undigested animal manures and slurries (Aneja et al., 2008; Alburquerque et al., 2012; Nkoa, 2014) especially as the temperature increases. An increase in NH3 emissions during the summer from a secondary lagoon filled with digestate relative to conventional open lagoons was documented in North Carolina in 2008 as part of the North Carolina State University Animal and Poultry Waste Management Center's evaluation of potentially environmentally superior technologies. (Aneja 2008). Factors such as temperature and pH may alter the equilibrium between ammonia and ammonium. For example, increasing temperature and pH will enhance ammonia emissions. (Angelidaki et al., 2003; Weaver et al., 2012; Nkoa, 2014). Air movement across an open lagoon surface also enhances NH3 loss. (Dari et al., 2019).

Once digestate is removed from lagoons and applied to fields it has the potential to emit ammonia at a greater rate than conventional hog waste, as depicted in Figure 2. (Nyord et al., 2012). Most studies evaluating ammonia emissions from land-applied digestate have evaluated land-application methods such as drag-hose irrigation or injection. (Nyord et al. 2012; Chantigny et al. 2009). It is well-established that these methods of land-application produce substantially less ammonia emissions than high-pressure spray hoses, (Grabow 2007), which are the land-application method of choice on hog operations in North Carolina. Some studies evaluating ammonia emissions from land-applied digestate have also used data from much colder climates than North Carolina. (Chantigny et al. 2009). It is well-established that low temperatures inhibit NH3 volatilization. (Dari et al., 2019). Therefore, it is reasonable to believe that land-application of digestate effluent through spraying in North Carolina, particularly during the summer, is likely to cause greater ammonia emissions than documented in these studies.

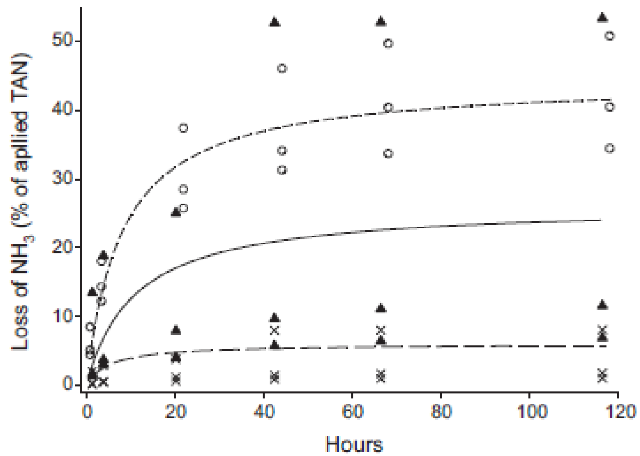

Fig. 3. Ammonia emissions 2008. Cumulative percentage loss of total applied TAN through emission over 116 h after application of pig slurry by trailing hoses. Regression lines and symbols represent cumulative losses of NH_3 emission from each measuring period; ---- and ○: digested slurry; — and ▲: untreated slurry; — — and ×: separated slurry.

Figure 2: Cumulative Ammonia loss by digestate, cattle, and pig slurry spread onto land under identical conditions (Nyord et al. 2012)

III. Impact of DEQ's issuance of permits to M-B to incorporate anaerobic digesters (covered lagoon) into the lagoon & sprayfield system

The permits issued by DEQ for the Waters, Kilpatrick, Benson, and Farm 2037/2038 hog operations authorize Murphy-Brown to modify its existing waste management systems to produce biogas, store digestate in open secondary lagoons, and apply digestate to fields. Instead of flushing hog urine and feces from barns into open lagoons the permits allow the hog operations to store the urine, feces, and water in covered lagoons where, as described in the previous section, biogas will be produced through the process of methanogenesis. The biogas along with other gases trapped under the lagoon cover will be siphoned off to be conditioned and then processed at a central plant.

The digestate will have a different composition from conventional waste, including a significantly higher concentration of NH3/NH4+. The digestate will be transferred to a secondary open lagoon. Ammonia emissions from the secondary lagoon are likely to exceed those expected from a similarly situated conventional lagoon system. (Aneja et al., 2008; Alburquerque et al., 2012; Nkoa, 2014). When the digestate effluent is land-applied, primarily through spraying, this higher rate of ammonia emissions is likely to continue. (Nyord et al., 2012).

The increased ammonia emissions from the secondary open lagoons and land-applied digestate effluent relative to a conventional lagoon and sprayfield system means that a technology intended to benefit the environment, biogas production and capture, may worsens air and water pollution coming from these four hog operations (Harper et al., 2010). The anticipated increase in ammonia emissions could cause more ammonia to be deposited onto nearby soil, where it can seep into groundwater or runoff into surface waters, and directly into surface waters, where it causes algal blooms, low oxygen conditions, and harms aquatic ecosystems.

IV. Availability of technology that mitigates ammonia emissions from lagoons & sprayfield system

Emissions of reactive nitrogen in animal waste can be reduced through various technologies and practices. (Aneja et al., 2009; Szogi et al., 2014). Farm-specific feed management practices cannot be enforced by regulators, but waste management technology requirements can. For example, in the Netherlands, Denmark, and UK, manure injection into soil, rather than spraying, has long been mandated to reduce ammonia emissions. However, technological fixes must be assessed for potential pollutant swapping, i.e., the increased emission of one pollutant resulting from abating another. One example of pollution swapping is increased nitrate leaching that may result from switching to manure injection without reducing the nitrogen application rate. (Aneja et al., 2009).

Over a decade ago, an engineered waste management system known as “Super Soils” was developed and tested in North Carolina. (Aneja et al., 2008). The system included a module that removed nitrogen from waste through a process known as nitrification-denitrification. The system reported a 73% reduction in ammonia emissions from hog operations. (Szogi et al., 2006). The Super Soils system also created an additional income stream for farmers by allowing them to sell concentrated nutrient byproducts of the system as fertilizer. The Super Soils technology has been improved upon twice. (Szogi et al., 2014). Though there is no lagoon in the Super Soil System for storage of the animal manure, the system’s modules have been adapted elsewhere to work alongside covered lagoons producing biogas. (Schmidt, 2009).

V. Conclusion

In sum, hog waste lagoons and sprayfields are a significant source of ammonia emissions, which harm local air and water quality and enhance formation of PM2.5, which is harmful to human health. The system permitted by DEQ, which includes a covered lagoon anaerobic digester, secondary uncovered lagoons that store digestate, and land application of the digestate through spraying, is likely to increase those ammonia emissions. The process of anaerobic digestion alters the composition of hog waste such that the digestate that comes out of the covered lagoon has a higher concentration of total ammoniacal nitrogen (TAN) than conventional waste. This change in composition increases ammonia emissions from lagoons and sprayfields fertilized with digestate relative to conventional lagoon and spray waste management systems. The expected increase in ammonia emissions, may in turn exacerbate air and water quality impacts. There are technologies available that reduce the nitrogen content of digestate and can therefore decrease ammonia emissions, avoiding significant environmental and public health impacts.

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ATTACHMENT 22

PERGAMON

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Atmospheric transport and wet deposition of ammonium in North Carolina

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Abstract

Wet deposition and transport analysis has been performed for ammonium (NH_4^+) in North Carolina, USA. Multiple regression analysis is employed to model the temporal trend and seasonality in monthly volume-weighted mean NH_4^+ concentrations in precipitation from 1983 to 1996 at six National Atmospheric Deposition Program/National Trends Network (NADP/NTN) sites. A significant ($p < 0.01$) increasing trend beginning in 1990, which corresponds to an annual concentration increase of approximately 9.5%, is detected at the rural Sampson County site (NC35), which is located within a densely populated network of swine and poultry operations. This trend is positively correlated with increasing ammonia (NH_3) emissions related to the vigorous growth of North Carolina's swine population since 1990, particularly in the state's Coastal Plain region. A source–receptor regression model, which utilizes weekly NH_4^+ concentrations in precipitation in conjunction with boundary layer air mass back trajectories, is developed to statistically test for the influence of a particular NH_3 source region on NH_4^+ concentrations at surrounding NADP/NTN sites for the years 1995–1996. NH_3 emissions from this source region, primarily evolving from swine and poultry operations, are found to increase NH_4^+ concentration in precipitation at sites up to ≈ 80 km away. At the Scotland County (NC36) and Wake County (NC41) sites, mean NH_4^+ concentrations show increases of at least 44% for weeks during which 25% or more back trajectories are influenced by this source region.

Keywords: Ammonium; Ammonia; Multiple regression; Wet deposition; Back trajectories; Source–receptor

1. Introduction

1.1. Background

Interest in atmospheric ammonia (NH_3) has increased substantially over the past several years as its roles in both atmospheric chemistry and nutrient cycling have become better understood. Atmospheric NH_3 is an abundant alkaline responsible for the neutralization of a substantial fraction of strong acids in the atmosphere (Asman et al., 1982). The adverse effects of excess reduced nitrogen in forest systems are well documented (Nihlgard, 1985; Reuss and Johnson, 1986). Atmospheric NH_x (NH_x = ammonia + ammonium + amines) is now thought to be an important component of total atmospheric nitrogen (TAN) input to nitrogen-sensitive ecosystems such as coastal and estuarine waters (Aneja et al., 1998; Aneja, 1997). Such systems may receive nitrogen by direct wet and dry atmospheric deposition, as well as surface and ground water transport. North Carolina's Albemarle-Pamlico Sound system is one of many North American, European and Asian estuarine and coastal ecosystems impacted by atmospheric nitrogen deposition which are exhibiting advanced signs of eutrophication in the form of recurring toxic and non-toxic phytoplankton

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blooms (Paerl, 1991,1995; Paerl et al., 1993). Such phytoplankton blooms are associated with varying degrees of oxygen depletion in water and fish losses (Paerl, 1995).

Atmospheric nitrogen deposition may be responsible for a substantial amount of nitrogen input across North Carolina's Coastal Plain region, owing to NH_3 emissions from the large number of animal operations across the region. Such sources will have a local impact on NH_3 and NH_4^+ deposition, and densely populated groups of such sources may have a regional influence on NH_4^+ deposition. The purpose of this study is to investigate the temporal characteristics of wet NH_4^+ deposition across North Carolina and to investigate the possible influence of NH_3 derived from a region of Coastal Plain animal operations on wet NH_4^+ deposition across the state. The analysis presented uses multiple linear regression to estimate the seasonality and long term trend in NH_4^+ concentration in precipitation at six National Atmospheric Deposition Program/National Trends Network (NADP/NTN) sites across North Carolina. Ammonium transport analysis is performed to investigate the influence of NH_3 emissions in southeastern North Carolina, where livestock population density is the largest, on NH_4^+ concentration in precipitation across the state. This is accomplished using boundary layer air mass back trajectory analysis coupled with multiple linear regression modeling. Incorporation of a weekly *influence factor* into a linear regression model allows for a formal statistical test of this source region's influence.

1.2. Ammonia emissions

Many studies have shown domestic animals to be the largest global source of atmospheric NH_3, with emission estimates ranging from 20–35 Tg N yr^{-1} (Schlesinger and Hartley, 1992; Warneck, 1988). Nitrogen emission estimates for the state of North Carolina (NC) show domestic animals to be the largest statewide contributor of NH_3, with swine operations present as the primary domestic animal source (Wooten, 1997; Aneja, 1997). Swine operations account for approximately 48% (68,450 t yr^{-1}, NH_3–N) of all North Carolina NH_3 emissions and approximately 21% of total nitrogen emissions.

Prior to 1990, the number of hogs in the state was relatively stable at near 2.5 million, however, this number began to rapidly increase within the period 1989–1990 (NCDA, 1998). In 1996, North Carolina contained approximately 9.3 million hogs, roughly 93% of which were located in the Coastal Plain region shown in Fig. 1. Area I in this figure, designated by shading, is defined as the six individual NC counties with the largest hog population densities. This collection of counties has an average hog population density of ~ 528 hogs km^{-2}. The average county hog population density for the remaining Coastal Plain is ~ 65 hogs km^{-2}. Area I contains approximately 66% of the Coastal Plain hog population and only 17% of the total Coastal Plain land area. Area I also contains approximately 68% of the Coastal Plain's domestic turkey population. These factors make area I a region of strong NH_3 emission relative to the rest of the state. Table 1 shows estimated area I NH_3 emissions by domestic animal type. Swine operations account for 77% of total NH_3 emissions from domestic animals within area I. Animal population statistics used in this study were supplied by the North Carolina Department of Agriculture (NCDA, 1998). Emissions are based on emission factors given by Battye et al. (1994).

1.3. Atmospheric ammonia

Estimates of the atmospheric lifetime of NH_3 range from approximately 0.5 h to 5 d (Fowler et al., 1997; Aneja et al., 1998). This short lifetime is the result of rapid gas-to-particle conversion of NH_3 to NH_4^+ and deposition of NH_3 to natural surfaces, particularly wet surfaces and vegetation which have low NH_3 compensation points. Once into the atmosphere, NH_3 which is not dry deposited or scavenged by raindrops will undergo conversion to NH_4^+ aerosol. The lifetime of NH_4^+ aerosol is typically 5–10 d (Crutzen, 1983). The rate of this conversion, which is largely unknown, will have an important bearing on the regional impact of NH_3 sources or source regions such as area I in this study. If this conversion proceeds slowly, area I emissions will primarily be deposited locally; thus, less NH_4^+ will be made available for long-range transport.

Conversion of NH_3 to NH_4^+ aerosol depends on the concentrations of strong acids and water vapor in the atmosphere. Ammonia reacts with sulfuric, nitric, and hydrochloric acids to form ammonium sulphate, ammonium bisulphate, ammonium nitrate and ammonium chloride aerosols (RGAR, 1997; INDTE, 1994; Finlayson-Pitts and Pitts, 1986). Ammonium aerosol formed in these reactions can exist as a solid particle or a liquid droplet depending on relative humidity (Finlayson-Pitts and Pitts, 1986). While the principal chemical transformation of NH_3 in the atmosphere is incorporation into NH_4^+ aerosol, approximately 10% is oxidized via the hydroxyl radical (OH) (Roberts, 1995).

1.4. Wet removal processes

The processes by which gases and aerosols are removed by precipitation can be divided into in-cloud and below-cloud regimes. The below-cloud processes include inertial removal by precipitation and diffusive removal on precipitation (Twomey, 1977). The in-cloud processes which govern wet deposition of aerosols and gases include inertial removal by cloud drops, nucleation, and diffusion to cloud drops (Twomey, 1977). It is generally agreed that in-cloud removal processes are more efficient

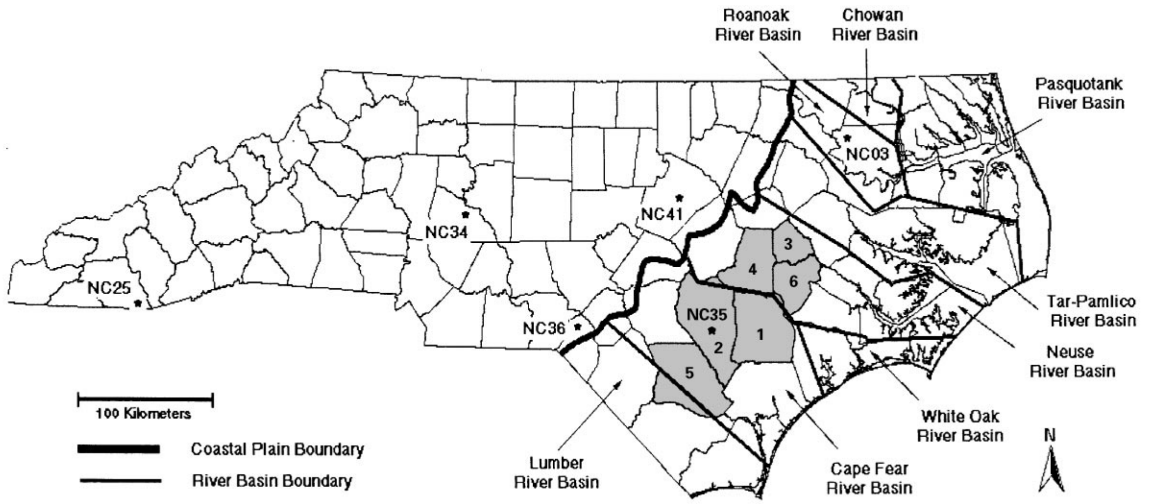

Fig. 1. NADP/NTN sites (*), area I NH_3 source region, and Coastal Plain river basins. The shaded area represents area I, a region defined as having an average hog population density of $\sim$ 528 hogs km^{-2}. Numbers within area I represent the following individual counties with corresponding estimated hog population densities (hogs km^{-2}): (1) Duplin County, 991; (2) Sampson County, 735; (3) Greene County, 503; (4) Wayne County, 349; (5) Bladen County, 316; and (6) Lenoir County, 274.

than below-cloud processes (Twomey, 1977; Asman, 1995). This is due to the large total surface area of droplets within the cloud compared to raindrops below cloud base. The contribution of below-cloud processes should not be ignored, however, especially in the case where the concentration of the compound of interest is larger in air below cloud base compared to the concentration within the cloud. It should be noted that some portion of the NH_4^+ measured in precipitation is the result of dry deposition of NH_4^+ aerosol to the precipitation collector. It is also important to point out that a fraction of the NH_4^+ measured in rainfall originates as NH_3 gas which is wet and dry deposited to the precipitation collector, resulting in the formation of NH_4^+ when NH_3 reacts with water (Warneck, 1988).

Table 1
Estimated area I NH_3–N emissions from domestic animals for the years 1996–1997[a]

Animal	NH_3–N emissions (t yr^{-1})
Hogs	47,679
Turkeys	9,585
Broilers	2,435
Cattle	2,154
All Chickens	181

[a] NH_3–N tons $= \frac{14}{17}$(NH_3 tons). Emissions calculated using emission factors given by Battye et al. (1994). Animal population statistics were provided by the North Carolina Department of Agriculture (NCDA, 1998). Calculations reflect hog population as of 1 December 1996; turkey population for 1996; cattle population as of 1 January 1997; broiler population for 1996; chicken population as of 1 December 1996. Emissions from turkeys and broilers were calculated by dividing the total population by the average number of flocks per year, 5.75 for broilers and 3.5 for turkeys.

2. Methods

2.1. Data selection

Precipitation chemistry data used in this study was provided by the National Atmospheric Deposition Program/National Trends Network, a nationwide precipitation collection network which began in 1978 and now operates over 200 sites (NADP/NTN, 1998). NADP/NTN samples are collected at 9:00 a.m. every Tuesday and sent to the Illinois State Water Survey, Central Analytical Laboratory (CAL) for chemical analysis (Peden, 1986). All samples are subject to the same field handling protocol and analytical procedures at CAL.

Only species concentrations in precipitation were used in this study. The term precipitation includes liquid, solid and mixed phase. Additionally, only samples that were considered valid and complete by NADP/NTN standards were used. Information on the data validation and completeness criteria used in this analysis can be found at the NADP/NTN website: *http://nadp.sws.uiuc.edu* (NADP/NTN, 1998). This study also employed the use of NADP/NTN daily precipitation information. The Rowan County (NC34) and Wake County (NC41) sites should be considered suburban. The remaining sites are located in rural areas. Fig. 1 shows the general location of the six NADP/NTN sites used in this study.

2.2. Trends analysis

Given the drastic increase in NH_3 emissions within area I beginning during the period 1989–1990, we hypothesize that a positive trend in NH_4^+ concentration in precipitation should be present at site NC35, located within area I, beginning in 1990 and that this trend may also be present at additional sites. To test this hypothesis, multiple linear regression is used to illustrate seasonality and trend in 14 yr worth of monthly volume-weighted mean NH_4^+ concentrations in precipitation. The 14 yr record is split into two periods. Period 1 includes the years 1983–1989 and period 2 includes the years 1990–1996. This was done in an effort to facilitate the correlation of any increasing NH_4^+ trends during period 2 with the growth of the North Carolina swine industry and the resulting increase in area I NH_3 emissions beginning in 1990. Complete analyses were performed for both periods at all sites.

Multiple linear regression analysis to investigate the temporal variation of precipitation chemistry has been widely used in the past (Buishand et al., 1988; Dana and Easter, 1987; MAP3S/RAINE, 1982). The present analysis employs the following regression model for separate analysis of both 7 yr periods at all sites:

$$Y_i = a_0 + a\cos(2\pi i/12 - \phi) + bi + cP_i + e_i,$$
$$i = 1, \ldots, 12N. \tag{1}$$

where N represents the number of years in the time series ($N = 7$). In this case, Y_i is the natural logarithm of the volume-weighted mean NH_4^+ concentration (mg l^{-1}) in precipitation for the ith month. The use of the natural log transform of concentrations has proven useful in improving the fit of parametric models such as the one above (Dana and Easter, 1987). The approximate lognormal distribution of species concentrations in precipitation has been illustrated (MAP3S/RAINE, 1982). Sirois (1991) points out that this transformation will help to achieve the regression modeling assumption of constant error variance. The term a_0 represents the intercept.

P_i represents the natural logarithm of the precipitation amount (ml) for the ith month. The inverse relationship between precipitation amount and concentration of ammonium in precipitation, which is the result of several processes occurring simultaneously, is well illustrated by Prado-Fiedler (1990). It should be pointed out that the use of volume-weighted concentrations will effectively reduce the amount of variation imposed on the concentration by precipitation amount. The precipitation term in the presently addressed regression model is included to capture remaining precipitation effects.

The term bi, where i (month) goes from 1 to $12N$, represents the monotonic trend in NH_4^+ concentration in precipitation over time. It should be pointed out that a systematic trend in precipitation amount will have some degree of collinearity with the trend term for concentration, thus the precision in the estimate of the trend magnitude will be reduced. Additionally, the assumption of a linear trend will simplify the structure of the actual trend which may have non-linear characteristics. In addressing the detectability of possible trends in the 7 yr records analyzed, emphasis was placed on the value of b in Eq. (1) for which the null hypothesis H_0: $b = 0$ is rejected with 80% probability. Buishand et al. (1988) used this approach in addressing the detectability of NH_4^+ trends in 5 yr records of monthly bulk precipitation samples.

The probability of detecting a trend, or power, is a function of degrees of freedom v, significance level (SL), and the noncentrality parameter δ where

$$\delta = b/\sigma_b. \tag{2}$$

Power increases with δ and v, though power varies minimally above $v = 30$. Power has a much stronger relationship with δ, which is a function of trend magnitude. Note that $v = n - c$ where n is the total number of observations and c is the number of regression coefficients in the model. As pointed out by Buishand et al. (1988), it is usually necessary to have at least 10 degrees of freedom in order to obtain a powerful t-test. For this reason it is advantageous to use monthly concentration values in this analysis rather than annual values. Another way to increase the power of trend detectability is to reduce the error (e_i) standard deviation. This illustrates the usefulness of adding additional explanatory variables, such as precipitation amount, to regression models designed to detect trend and seasonality. In this analysis, adding explanatory variables such as concentrations of other analytes (NO_3^- and SO_4^{2-}) resulted in a reduction of error standard deviation at the cost of severe multicollinearity among model independent variables. To avoid such multicollinearity, only the most parsimonious group of variables was chosen for the final model.

The use of a cosine or sine term to model the annual cycle in concentrations is well documented (Buishand et al., 1988; MAP3S/RAINE, 1982). The cosine term in model (1) represents the seasonal component of the variation in concentration where a is the amplitude and ϕ is the phase angle. This term achieves a maximum at $i \approx 6\phi/\pi$, thus the location of the maximum in the annual cycle of concentration can be readily identified. It should be noted that the seasonality term will have some degree of collinearity with seasonality present in the precipitation term.

To test the hypothesis that presence of seasonality and trend in precipitation amount exist, the following regression model was used for analysis of both periods at all sites:

$$P_i = a_0 + a\cos(2\pi i/12 - \phi) + bi + e_i, \quad i = 1, \ldots, 12N. \tag{3}$$

P_i represents the natural logarithm of precipitation amount and bi is the trend term. The cosine term models the seasonal cycle in precipitation amount.

2.2.1. Reparameterized models

To facilitate the estimation of the coefficients $a_0, a, b, c,$ and ϕ following Buishand et al. (1988), the final form of model (1) becomes

$$Y_i = a_0 + \alpha\cos(2\pi i/12) + \beta\sin(2\pi i/12) + bi + cP_i + e_i,$$

$$i = 1, \ldots'' \, 12N \tag{4}$$

where $\alpha = a\cos\phi$ and $\beta = a\sin\phi$. The cosine term in Eq. (1) is thus decomposed into the cosine and sine terms in Eq. (4) to determine a and ϕ. Using the Proc Reg procedure within SAS® statistical analysis software, estimates $\hat{a}_0, \hat{\alpha}, \hat{\beta}, \hat{b}$, and $\hat{c}$ of the regression coefficients in Eq. (4) were calculated by the method of ordinary least squares (SAS Institute Inc., 1990). It follows that estimates of a and ϕ can be derived from the relations:

$$\hat{a} = \sqrt{\hat{\alpha}^2 + \hat{\beta}^2}, \tag{5}$$

$$\hat{\phi} = \begin{cases} \arctan(\hat{\beta}/\hat{\alpha}) & \text{if } \hat{\alpha} \geqslant 0, \\ \arctan(\hat{\beta}/\hat{\alpha}) + \pi & \text{if } \hat{\alpha} < 0. \end{cases} \tag{6}$$

Proc Reg also provides estimates of the standard errors of the above regression coefficients as well as an estimate of σ_e. Buishand et al. (1988) give the following relationships for $\hat{a}$ and $\hat{\phi}$ which are derived from a Taylor expansion of Eqs. (5) and (6):

$$\sigma^2(\hat{a}) = \operatorname{var}\hat{a} \approx (\alpha^2/a^2)\operatorname{var}\hat{\alpha} + (\beta^2/a^2)\operatorname{var}\hat{\beta} + 2(\alpha\beta/a^2)\operatorname{cov}(\hat{\alpha}, \hat{\beta}), \tag{7}$$

$$\sigma^2(\hat{\phi}) = \operatorname{var}\hat{\phi} \approx (\beta^2/a^4)\operatorname{var}\hat{\alpha} + (\alpha^2/a^4)\operatorname{var}\hat{\beta} - 2(\alpha\beta/a^4)\operatorname{cov}(\hat{\alpha}, \hat{\beta}). \tag{8}$$

Estimated values of the variances and covariances of $\hat{\alpha}$ and $\hat{\beta}$ are calculated by Proc Reg. By using estimates of a, α and β on the right-hand sides of (7) and (8), the standard errors $\hat{\sigma}(\hat{a})$ and $\hat{\sigma}(\hat{\phi})$ can be calculated.

The final form of model (3) is:

$$P_i = a_0 + \alpha\cos(2\pi i/12) + \beta\sin(2\pi i/12) + bi + e_i,$$

$$i = 1, \ldots, 12N. \tag{9}$$

Model variables are defined as in model (3). Estimates of a, α, β, ϕ and b and their standard errors are obtained as in model (4).

The Student's t-statistic is used to test the statistical significance of the regression coefficients in the model under the null hypothesis (H_0) that the regression coefficient being tested is zero. Probability values (p-values) are also calculated for each coefficient. A small p-value suggests a disagreement between the data and the null hypothesis. In this study, the significance level SL $= 0.1$ is chosen as the value above which a given p-value results in failure to reject H_0.

The validity of the t-test is subject to the basic assumptions of the linear regression model. These assumptions are that the errors (e_i): have constant variance; are uncorrelated with each other in time; and have a normal distribution. The Durbin–Watson test was used to test for first-order autocorrelation among the residuals. At sites where the correlation assumption was violated or considered inconclusive, a regression technique, Proc Autoreg, which accounts for correlation in residuals was used to estimate regression coefficients. The validity of the normality assumption was assessed by visual inspection of residual frequency distributions and quantile–quantile (Q–Q) plots of ordered residuals against normal quantiles. In observing Q–Q plots, a linear relationship suggests a normal population of residuals. For a more detailed treatment of residual frequency distributions and Q–Q plots, the reader is referred to *SAS® System for Statistical Graphics* (1991).

In this analysis, outliers that were not considered influential were left in the model. The level of outlier influence was estimated by using the difference in fits (DFFITS) and difference in beta (DFBETAS) statistics produced by the Proc Reg procedure. The commonly used cutoff value of 2 was used for both statistics (Bowerman and O'Connell, 1990). When the value of DFFITS exceeds 2, removing the corresponding observation from the data set substantially changes the point prediction of Y_i (model (4)). If the value of DFBETAS exceeds 2, then removing the corresponding observation from the data set substantially changes the point estimate of the corresponding regression coefficient. Those observations with DFFITS and DFBETAS values less than 2 after the initial model run were retained.

2.3. Source–receptor analysis

The source–receptor analysis developed here combines air mass back trajectory analysis with multiple linear regression to investigate the influence of area I NH_3 emissions on NH_4^+ concentration in precipitation at surrounding NADP/NTN sites. An air mass trajectory within the boundary layer which arrives at a site after having traversed any portion of area I for any duration of time is considered to possibly contain elevated concentrations of NH_3 and NH_4^+, and is consequently labeled as influenced. The distances of individual sites from the perimeter of area I are listed in Table 2. All sites in

Table 2
Distance from the perimeter of area I to NADP sites

Site	Distance (km)
NC41	56
NC03	76
NC36	60
NC34	170
NC25	387

Fig. 1 except site NC35 are included in the source–receptor analysis. This analysis uses weekly NH_4^+ concentrations in precipitation and daily precipitation information. The period of analysis includes the years 1995 and 1996.

Back trajectory models are commonly used to link receptors to source regions in wet deposition studies (Billman-Stunder et al., 1986; Moody and Samson, 1989; Henderson and Weingartner, 1982; Ruijgrok and Romer, 1993; RGAR, 1997). Such studies typically utilize chemistry data from samples collected on an event basis. This analysis uses weekly concentration values and back trajectories for days on which precipitation was measured. Therefore, a weekly sample can be comprised of a maximum of seven separate precipitation events.

Back trajectories were calculated using version 4 of the Hybrid Single Particle Lagrangian Integrated Trajectory (HYSPLIT) model developed by the National Oceanic and Atmospheric Administration's Air Resources Laboratory (NOAA/ARL). The model was accessed and run from the World Wide Web at *http://www.arl.noaa.gov/ready/hysplit4.html* (HYSPLIT4, 1997). The HYSPLIT model calculates three-dimensional trajectories from previously gridded horizontal (u and v) and vertical (ω) wind fields output and archived every 2 h from NOAA's National Center for Environmental Prediction's Nested Grid Model (NGM) (Draxler, 1997). The trajectories in this analysis used the vertical motion prescribed by the NGM and are therefore kinematic.

A single 24 h back trajectory starting at 2300 h was run for each day on which precipitation was measured at the NADP/NTN site. Given its distance from area I, 36 h back trajectories were calculated for site NC25. When initial trajectories exhibited significant curvature, additional trajectories were also run at 1800 and 1200 h. The trajectory level was chosen as 150 m. The 150 m level is expected to be within the boundary layer even during stable nocturnal conditions, when boundary layer depth is typically 100–200 m (Arya, 1998). The HYSPLIT wind fields at this level are interpolated from the nearest NGM sigma level. There are about 4 NGM sigma levels within the boundary layer (Draxler, 1996).

In our analysis it is assumed that most of the NH_3 emitted at ground level within area I will likely arrive at surrounding NADP/NTN sites, aside from NC25, via transport within the boundary layer. We are, by consequence, characterizing flow conditions which result in the wet deposition of area I NH_3 and NH_4^+ at a site via below-cloud mechanisms. Only when the cloud-level trajectory is the same as the 150 m level will we describe those conditions which result in wet deposition of NH_3 or NH_4^+, which has originated from area I, by both in-cloud and below-cloud mechanisms.

The methodology presented here also assumes that the 150 m trajectory characterizes the general flow of the boundary layer. Within the well-mixed daytime boundary layer, trajectories at different levels should not deviate from each other significantly. Arya (1988) points out that wind direction across the moderately unstable and convective boundary layer typically changes by less than 15°. Under stable conditions, the horizontal deviation of trajectories at different heights in the boundary layer will be greater. The largest change in trajectory imposed by diurnal variation in boundary layer stability may take place in the transition from unstable daytime conditions to more stable nighttime conditions. In this case however, the layers that begin to form and fan out as stratification proceeds will roughly have the same concentration of NH_3 or NH_4^+, a result of the uniform concentration within the well-mixed daytime boundary layer.

During the time required for transport from area I to NC25, a considerable amount of NH_4^+ may make it out of the boundary layer and into clouds associated with precipitating systems transported by synoptic scale flow. For this reason, a 2000 m trajectory was calculated in addition to the 150 m trajectory.

In order to facilitate the use of daily back trajectories with weekly NADP samples, an influence factor (I) is introduced. I is defined as the ratio of the number of influenced trajectories during week i to the total number of trajectories during week i. Note that $I = 1$ when all the trajectories during a week are considered influenced. The maximum possible number of trajectories for a week is 7.

We hypothesize that area I NH_3 emissions may be influencing NH_4^+ concentrations at NADP sites surrounding area I. To set up this hypothesis, we first use the non-parametric Wilcoxon Rank Sums (WRS) test to compare the average concentration for weekly samples where $I \geqslant 0.25$ to the average concentration of those weeks where $I < 0.25$. $I \geqslant 0.25$ represents a week during which 25% or more of all back trajectories were considered to be influenced. H_0 for the WRS test is that the means of the groups being compared are equal. The WRS test is also performed to test for the equality of mean precipitation amounts of the two I groups. If the two group mean precipitation amounts are equal, then the physical meaning of different group mean NH_4^+ concentrations is maximized with respect to the dependence of concentration on I. Using the information gained in this means comparison as the foundation for our hypothesis, we then use regression analysis to conduct a formal hypothesis test.

In order to formally test for a statistically significant influence of area I NH_3 emissions on NH_4^+ concentration in precipitation collected at sites other than NC35, the influence factor (I) is incorporated into the following source–receptor regression model:

$$Z_i = a_0 + \alpha\cos(2\pi i/52) + \beta\sin(2\pi i/52) + dP_i + fI_i + e_i, \quad i = 1,\ldots,52N. \tag{10}$$

In this model, Z_i represents the natural logarithm of NH_4^+ concentration in precipitation ($mg\,l^{-1}$) for the ith week, P_i is the precipitation amount (ml) for the ith week, and I_i represents the influence factor for the ith week. The sine and cosine terms model the annual cycle which may be present in the weekly concentration values. Coefficients α, β, and corresponding phase angle ϕ are described in Section 2.2.1. $N(N = 2)$ represents the number of years present in the time series. This model was used to perform separate analyses at each site for the entire 2 yr period. Tests for regression coefficients, recognition of model assumptions and treatment of outliers are described in Section 2.2.1.

3. Results and discussion

3.1. Trends analysis

Table 3 summarizes the results of the trends analysis [model (4)]. The average R^2 for all sites is 0.45, meaning that the collection of variables in the regression model explains about 45% of the variation in the monthly volume-weighted mean NH_4^+ concentrations in precipitation. Mean observed concentration values for time period 1 range from 0.16 to 0.28 $mg\,l^{-1}$, and from 0.15 to 0.31 $mg\,l^{-1}$ for period 2. Mean predicted concentrations for time period 1 range from 0.11 to 0.24 $mg\,l^{-1}$, and from 0.11 to 0.26 $mg\,l^{-1}$ during period 2. The two highest concentration values, both observed and predicted, for period 1 are found at the two suburban sites, NC34 and NC41. It should be noted that NH_4^+ concentrations at NC34 and NC41 are likely influenced by NH_3 emissions from local livestock throughout both periods. The highest value for period 2 is shared by site NC35, located within area I, and NC41. The mean predicted value is an average of 0.045 $mg\,l^{-1}$ lower than the corresponding mean observed value at all sites during both periods. Precipitation amount is a significant predictor variable at several sites.

Table 3 shows a statistically significant (SL = 0.01) seasonal cycle at all sites, with maximum NH_4^+ concentrations in precipitation occurring in the summer, except for a spring maximum at site NC25 during period 1. Figs. 2 and 3 illustrate the typical seasonal cycle. Standard deviations of the phase angle, which illustrate the accuracy of i_{max}, range from 6.6 to 16.6°, or approximately 1 to 2.5 weeks (Table 3).

The increased ambient concentration of NH_4^+ during summer comes partly from the fact that mineralization in soil, which drives natural production of NH_3, is governed partially by temperature dependent microbial activity. A 10°C increase in soil temperature approximately doubles the rate of ammonification (Addiscott, 1983). As pointed out by Davies et al. (1986,1991), seasonality in

Table 3

Estimated regression coefficients in model (4), multiple coefficients of determination (R^2), and mean volume-weighted monthly NH_4^+ ion concentrations in precipitation for all sites. Time period 1: 1983–1989. Time period 2: 1990–1996. The value i_{max} is the month corresponding to the seasonal maximum ($i_{max} = 1$ corresponds to January). The phase angle $\hat{\phi}$ and its standard deviation (SD) are in degrees. $\hat{y}$ and y represent predicted and observed concentrations ($mg\,l^{-1}$), respectively

Site	Period	$\hat{a}$	SD $\hat{a}$	$\hat{\phi}$	SD $\hat{\phi}$	i_{max}	$\hat{b}$	$\hat{c}$	R^2	mean $\hat{y}$	$\bar{y}$
NC03	1	0.748[a]	0.108	189	8.3	6	− 0.0023	− 0.59[a]	0.47	0.13	0.18
	2	0.712[a]	0.082	181	6.6	6	0.0009	− 0.41[a]	0.54	0.16	0.21
NC34	1	0.708[a]	0.088	181	6.9	6	0.0005	− 0.58[a]	0.62	0.24	0.28
	2	0.545[a]	0.077	213	7.9	7	0.0050[b]	− 0.17	0.49	0.23	0.27
NC41	1	0.489[a]	0.134	201	16.4	7	0.0062	− 0.37[a]	0.27	0.22	0.27
	2	0.433[a]	0.071	194	9.6	7	0.0018	− 0.46[a]	0.47	0.26	0.31
NC35	1	0.407[a]	0.122	183	16.6	6	− 0.0003	− 0.23	0.16	0.16	0.19
	2	0.516[a]	0.068	187	7.7	6	0.0079[a]	− 0.32[a]	0.53	0.26	0.31
NC36	1	0.820[a]	0.163	244	11.6	8	0.0008	− 0.53[a]	0.57	0.11	0.16
	2	0.487[a]	0.088	263	10.7	9	0.0012	− 0.46[a]	0.45	0.16	0.20
NC25	1	0.780[a]	0.130	161	8.7	5	− 0.0012	− 0.22	0.41	0.13	0.17
	2	0.658[a]	0.111	198	9.9	7	0.0003	− 0.29[b]	0.41	0.11	0.15

[a]Significant at the 1% level.

[b]Significant at the 5% level, but not at the 1% level.

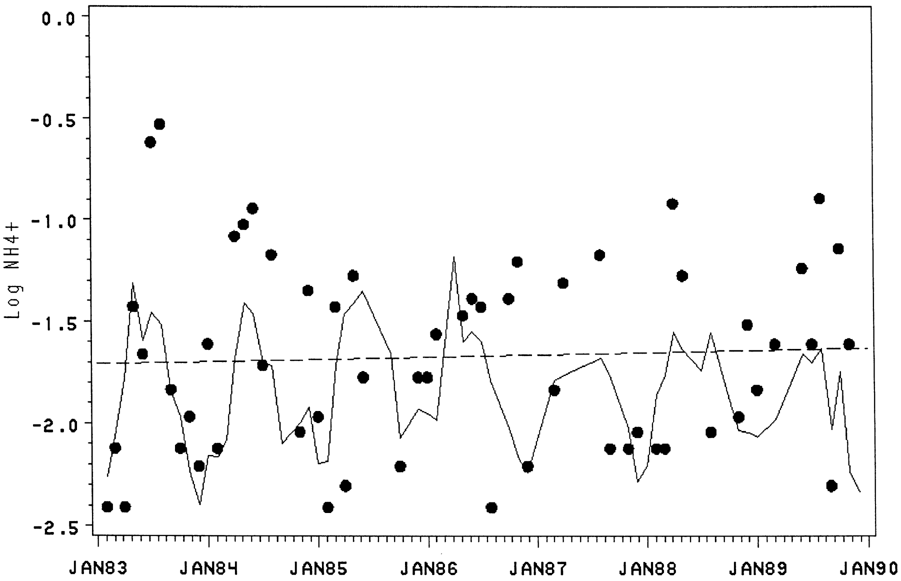

Fig. 2. Seasonality and insignificant trend ($p > 0.10$) in the natural log (log NH_4^+) of monthly volume-weighted NH_4^+ concentration in precipitation (mg l^{-1}) at Sampson County site NC35 during period 1. Solid dots (●) represent observed values. The solid and dashed lines respectively represent predicted values and estimated trend given by model (4).

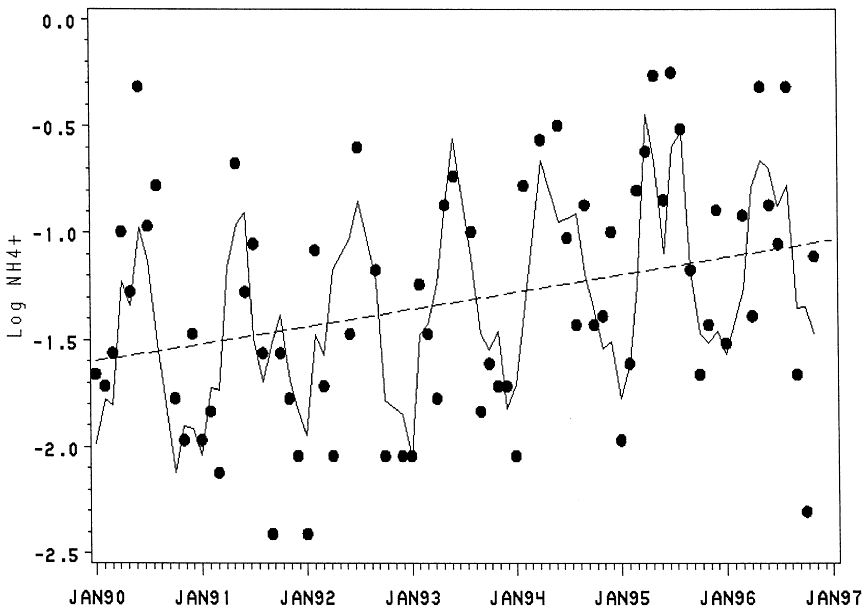

Fig. 3. Seasonality and significant trend ($p < 0.01$) in the natural log (log NH_4^+) of monthly volume-weighted NH_4^+ concentration in precipitation (mg l^{-1}) at Sampson County site NC35 during period 2. Solid dots (●) represent observed values. The solid and dashed lines respectively represent predicted values and estimated trend given by model (4).

synoptic scale meteorological conditions may also influence the seasonal pattern of NH_4^+ concentration in precipitation. This may involve a seasonal redistribution of upwind sources due to changing prevailing flow regimes and seasonal changes in precipitation characteristics such as intensity and duration.

As mentioned above, area I is believed to be a significant source of atmospheric NH_3 in the North Carolina Coastal Plain region. Harper and Sharpe (1997) have shown that volatilization of NH_3 from waste lagoons has a positive correlation with lagoon surface temperature, thus such sources will have peak emission strengths during summer. This may contribute to the summertime maximum NH_4^+ concentration in precipitation at sites such as NC35 which are located within a dense population of swine operations. These sources are randomly located among croplands, thus increased concentrations of NH_4^+ in rainfall during summer may also be related to emissions of NH_3 resulting from summer fertilizer application.

It is hypothesized that seasonal patterns in precipitation may have some influence on NH_4^+ concentration in precipitation. The use of regression model (9) to detect seasonal patterns in precipitation amount revealed statistically significant (SL $= 0.05$) annual cycles during period 1 at all sites except NC25. Maximum values of precipitation amount occurred during the summer in all cases. Significant annual cycles during period 2 were found at NC36, NC35 and NC03. Maximum precipitation amounts were again found during the summer at all sites. The inverse relationship between precipitation amount and NH_4^+ concentration in precipitation suggests that the summer maximum in NH_4^+ concentration is unrelated to the seasonal pattern in precipitation amount.

As stated earlier, the hypothesis behind the trends analysis is that increasing NH_3 emissions in area I may be imposing a positive trend in NH_4^+ concentrations in precipitation at site NC35, located within area I, and that this trend may also be present at additional sites. The trends should follow the temporal pattern of NH_3 emissions related to the NC hog industry growth which was generally stable within the period 1983–1989 (period 1) and has experienced drastic growth during the period 1990–1996 (period 2). Regression analysis shows a highly significant (SL $= 0.01$) increasing trend present at Sampson County site NC35 during period 2 (Table 3) and lack of trend during period 1. Figs. 2 and 3 show predicted value trendlines for both periods at NC35. The estimated monthly increase in NH_4^+ concentration in precipitation during period 2 corresponds to an annual increase of approximately 9.5%. The location of site NC35 among such a large number of animal operations and the correlation of the NH_4^+ trend with hog population growth suggests that the increasing magnitude of local NH_3 emission is the most likely explanation for the positive NH_4^+ trend. Furthermore, Cornelius (1997) showed that hog population density may be a significant predictor of NH_4^+ concentration in precipitation for sites within densities > 140 hog mi^{-2} such as area I. Site NC35 is specifically included in his analysis. It is important to realize that the trend model addressed here and the analytical methods employed by Cornelius (1997) do not identify sources of NH_4^+ in precipitation. This analysis illustrates the apparent correlation between the increasing trend in NH_4^+ concentration in precipitation at site NC35 and the increase in local NH_3 emissions resulting from rapid growth in the number of neighboring swine operations. A less significant (SL $= 0.1$) trend is found at site NC34 for period 2 (Table 3) and is most likely related to the increase in NH_3 emissions associated with a 33% countywide (Rowan County) increase in the cattle population for the period. The remaining sites, NC03, NC41, NC36, and NC25 do not show significant temporal trends during either period (Table 3).

Buishand et al. (1988) show that $\delta = 2.85$ for an 80% chance of rejecting H_0: $b = 0$ where $v > 30$ (two-sided test at the 5% level). In this analysis, $v = n - 5$, where the average n for all sites is ≈ 69. It follows from Eq. (2) that $b_{80} = 2.85\sigma_{\hat{b}}$. Using this relationship, the estimate of b for site NC34 during period 2 falls slightly below $b_{80} = 0.006$. At site NC35 for period 2, $b_{80} = 0.005$ and $b_{90} = 0.007$, indicating that the estimated trend ($b = 0.0079$) was detected with 80 and 90% probability.

In this analysis it is found that precipitation volume has a statistically significant inverse relationship with monthly volume-weighted NH_4^+ concentration at some sites. Any trend in precipitation volume over time may then impose a trend of opposite sign in NH_4^+ concentration. Regression model (9) was used to test the hypothesis of no trend in precipitation amount at each site. A single significant ($p < 0.05$) positive trend at site NC41 was found during period 2. This trend was not strong enough to impose a significant decreasing trend in NH_4^+ concentration in precipitation during the period. No other trends were detected at any site during either period.

During the analyses of concentrations [model (4)] and precipitation amount [model (9)], no departures from normality were detected in residuals. The number of influential outliers was less than 5 in all cases.

3.2. Source–receptor analysis

Table 4 shows the results of the Wilcoxon rank sums (WRS) test. At site NC36, the mean NH_4^+ concentration in precipitation for weeks where $I \geqslant 0.25$ is approximately 0.08 mg l^{-1} or 44% higher than the mean value for weeks where $I < 0.25$. The WRS test suggests that these means are significantly different at SL $= 0.10$. It is important to understand that the difference between these means could be entirely due to differences in group mean precipitation amounts. However, the WRS test for equality of mean precipitation amounts fails to show unequal means for the two I groups at the 10% level. This suggests that the difference in mean NH_4^+ values is likely not arising simply from a difference in group mean precipitation amounts. At site NC41, the mean NH_4^+ concentration in precipitation for weeks where $I \geqslant 0.25$ is approximately 0.17 mg l^{-1} or 50% higher than the

Table 4
Group means for weekly NH_4^+ (mg l^{-1}) concentrations in precipitation and precipitation amount (ml) and p-values (Prob. $> |Z|$) for the Wilcoxon Rank Sums test. $\mu_{1NH_4^+}$ is the mean of the observed weekly NH_4^+ values for which $I < 0.25$. $\mu_{2NH_4^+}$ is the mean of the observed weekly NH_4^+ values for which $I \geqslant 0.25$. p-value$_A$ is the probability of falsely rejecting H_0 for the WRS test for equality of $\mu_{1NH_4^+}$ and $\mu_{2NH_4^+}$. μ_{1P} is the mean observed precipitation amount of weekly samples with $I < 0.25$. μ_{2P} is the mean observed precipitation amount of weekly samples with $I \geqslant 0.25$. p-value$_B$ is the probability of falsely rejecting H_0 for the WRS test for equality of μ_{1P} and μ_{2P}

Site	$\mu_{1NH_4^+}$	$\mu_{2NH_4^+}$	p-value$_A$	μ_{1P}	μ_{2P}	p-value$_B$
NC03	0.24	0.31	0.1600	1608	2437	0.1525
NC36	0.18	0.26	0.0586[a]	2033	2395	0.6411
NC41	0.34	0.51	0.0747[a]	1560	2337	0.0621[a]

[a]Significant at the 10% level, but not at the 5% or 1% levels.

mean value for weeks where $I < 0.25$. Here the group mean concentrations are again deemed significantly different at SL = 0.10. The mean precipitation amounts are also significantly different at SL = 0.10. The higher mean precipitation value for $I \geqslant 0.25$ shows that in spite of the inverse relationship between precipitation amount and concentration, the concentration for those weeks where $I \geqslant 0.25$ is still greater than those weeks where $I < 0.25$. At NC03, the mean NH_4^+ concentration for weeks where $I \geqslant 0.25$ is approximately 0.07 mg l^{-1} or 29% higher than the mean value for weeks where $I < 0.25$. In this case the results were not statistically significant (SL < 0.20). The mean precipitation amounts for this site where not shown to be significantly different (SL = 0.10). It should be noted that first order autocorrelation between NH_4^+ concentration and I was found at both NC41 and NC36 ($\rho < 0.24$ in both cases). For this reason the p-values used to test H_0 in the WRS test should be used with caution, since the WRS test assumes independence in the data. The WRS was chosen here for its resistance to outliers in comparing means.

Table 5 summarizes the results of model (10) in the source–receptor analysis. The average R^2 is 0.45, meaning that precipitation amount and boundary layer air mass transport explain approximately 45% of the variation in weekly NH_4^+ concentration in precipitation. The influence of precipitation amount is highly significant ($p < 0.01$) at all sites. Statistically significant ($p < 0.01$) annual cycles are also found at all sites. The cycles reach a maximum during the summer at all sites, showing good agreement with the cyclic pattern predicted by the trends model for monthly values. An important result from this analysis is the sitewise estimates of f. Statistically significant ($p < 0.10$) positive values of $\hat{f}$ are found for NC03, NC36 and NC41. A positive value of $\hat{f}$ suggests that NH_4^+ concentration in precipitation increases with the percentage of influenced trajectories during a week. These results suggest that transport of NH_4^+ and or NH_3 originating from area I is detected for distances up to approximately 80 km. The influence of area I NH_3 emissions on NH_4^+ concentration in precipitation at sites greater than approximately 80 km away was not detected. We do not wish to suggest that the spatial extent of wet deposited NH_3 and NH_4^+ originating from area I is limited to 80 km. A larger data set is needed to provide more information for sites NC34 and NC25. The average number of observations used in this analysis for all sites is 80. Q–Q plots and frequency distributions of residuals suggest that the normality assumption was satisfied at all sites.

Table 5
Influence factor regression coefficients ($\hat{f}$) and multiple coefficients of determination (R^2) for model (10). Number of influenced trajectories and percent of total back trajectories (BT) labeled as influenced for the entire period are included for each site

Site	$\hat{f}$	R^2	No. influenced BT	% influenced BT
NC03	0.441[a]	0.52	53	22.0
NC25	− 1.475[b]	0.32	11	4.0
NC34	− 0.579[a]	0.39	29	9.0
NC36	0.495[b]	0.56	56	27.0
NC41	0.323[a]	0.44	89	38.0

[a]Significant at the 10% level, but not at the 5 or 1% levels.
[b]Significant at the 5% level, but not at the 1% level.

4. Conclusions

In this study, multiple linear regression has been used to illustrate the temporal characteristics of NH_4^+ concentration in precipitation at six North Carolina NADP/NTN sites. Seasonality, with maximum concentrations during warm months, is observed at all sites. A significant ($p < 0.01$) increasing trend in NH_4^+ concentration in precipitation beginning in 1990 is found at site NC35. This trend, which corresponds to an average annual increase in NH_4^+ concentration in precipitation of 9.5%, is correlated with the increasing number of local swine operations since 1990. A less significant trend at Rowan County site NC34 is found, which is also likely the result of increasing local NH_3 emissions and not the result of increasing area I emissions.

A source–receptor relationship has been developed for five NADP/NTN sites surrounding an area of strong NH_3 emission located over southeast North Carolina. For this analysis, boundary layer air mass back trajectories based on daily precipitation information are used in conjunction with weekly NH_4^+ concentrations in precipitation. This information is incorporated into a source–receptor regression model to test for the dependence of weekly NH_4^+ concentration in precipitation on the percentage of back trajectories during a week which are possibly influenced by this area of strong NH_3 emission.

Results show that NH_4^+ concentration in precipitation is positively correlated with the percentage of influenced trajectories during a week at sites up to approximately 80 km away. For those weeks during which 25% or more boundary layer air mass trajectories traversed area I, NH_4^+ concentration in precipitation at Scotland County site NC36 is approximately 44% higher than other weeks. At Wake County site NC41, NH_4^+ concentration in precipitation is at least 50% higher for those weeks during which 25% or more trajectories traversed area I.

Results from this analysis show that NH_3 emitted from area I is being transported over distances which would allow direct wet deposition as NH_4^+ or NH_3 to nitrogen sensitive coastal and estuarine waters. This also suggests that NH_3 emitted from area I is being wet deposited as NH_4^+ or NH_3 to all river basins in the North Carolina Coastal Plain region.

The swine industry accounts for roughly 21% of all NC nitrogen emissions and 93% of the total hog population resides in the Coastal Plain region of the state. The fraction of TAN being deposited to NC coastal and estuarine waters which can be attributed to agricultural NH_3 emissions over the southeast part of the state may therefore be significant. This study points to the need for broader regulations governing nitrogen emissions. As Paerl (1995) points out, atmospheric nitrogen emissions have increased in a largely unregulated manner globally over the last four decades. In the US, nitrogen emission restrictions primarily consist of NO_x regulations. Over the past several years it has become apparent that excess nitrogen can have acute and chronic detrimental effects on coastal and estuarine ecosystems. In coastal states such as North Carolina, whose NH_3 emissions represent a significant portion of overall nitrogen emissions, regulations on primary NH_3 sources, in addition to NO_x restrictions, may be warranted. In the future, it may be possible for us to mitigate the influence of ammonia emissions through chemical or physical process controls.

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ATTACHMENT 23

Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution

Michael A. Mallin, Matthew R. McIver, Anna R. Robuck & Amanda Kahn Dickens

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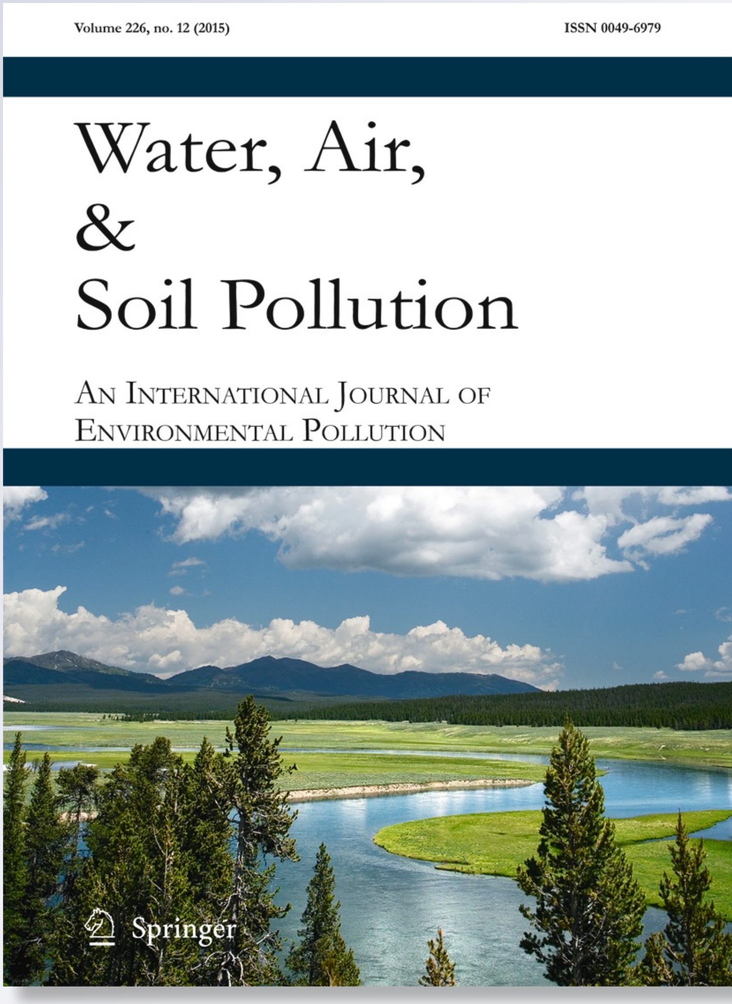

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Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution

Michael A. Mallin · Matthew R. McIver · Anna R. Robuck · Amanda Kahn Dickens

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Abstract Concentrated animal feeding operations (CAFOs) are the principal means of livestock production in the USA and Europe, and these industrial-scale facilities have a high potential to pollute nearby waterways. Chemical and biological stream water quality of a swine and poultry CAFO-rich watershed was investigated on 10 dates during 2013. Geometric mean fecal coliform counts were in the thousands at five of seven sites, especially in locations near swine waste sprayfields. Nitrate concentrations were very high and widespread throughout the watershed, with some individual samples yielding >10 mg-N/L. Ammonium concentrations were likewise high, but greatest near swine waste sprayfields, ranging up to 38 mg-N/L. Five-day biochemical oxygen demand (BOD5) concentrations exceeded 10 mg/L in 11 of 70 stream samples, reaching as high as 88 mg/L. BOD5 concentrations were significantly correlated with components of animal waste including total organic carbon, ammonium, and phosphorus, as well as the nutrient response variable chlorophyll *a*. The degree of nutrient and fecal contamination did not significantly differ between rainy and dry periods, indicating that surface and groundwater pollution occurs independently of stormwater runoff. This research shows that industrial-scale swine and poultry production leads to chronic pollution that is both a human health and ecosystem hazard. There are approximately 450,000 CAFOs currently operating in the USA, with the majority located in watersheds feeding major riverine and estuarine systems with known water quality problems. Current US waste management protocols for this widespread system of livestock production fail to protect freshwater and estuarine ecosystems along the US Mid-Atlantic, Southeast and Gulf coasts, and expansion into industrializing nations will likely bring severe pollution with it.

Keywords CAFO · Nutrients · Fecal bacteria · BOD · Algal blooms

1 Introduction

Industrial-scale livestock production is the most common and widespread means of swine and poultry production in the USA and Europe (Thu and Durrenberger 1998; Thorne 2007) and occurs within facilities known as concentrated, or confined, animal feeding operations (CAFOs). The USEPA (2014) defines large CAFOs as containing ≥1000 head of beef cattle, 2500 swine >25 kg, 10,000 swine <25 kg, 125,000 chickens, 82,000 laying hens, or 55,000 turkeys. Large-scale production of livestock in CAFOs involves shipping in vast quantities of feed from elsewhere (often from other states) which results in the deposition of large amounts of excretory nitrogen, phosphorus, organic matter, and fecal microbes in the watershed where the CAFO is located (Cahoon et al. 1999; Mallin and Cahoon 2003). Waste generated by swine in CAFOs is hosed

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through slats in the floor of confinement buildings and is drained or pumped outdoors into a cesspit which the industry calls a "lagoon." Periodically, the liquid waste supernatant of lagoons is pumped out and sprayed onto surrounding fields (i.e., sprayfields), which are planted with a cover crop such as Bermuda grass to absorb excess nutrients. Poultry waste is usually collected as dry litter, mixed with straw, and spread on neighboring fields; some CAFOs containing laying hens use the lagoon system. However, the production of vast quantities of animal manure within watersheds can overload the ecosystem's capacity to dilute and process such waste (Carpenter et al. 1998; Weldon and Hornbuckle 2006). According to the USEPA (2014), there are approximately 450,000 CAFOs in the USA, unevenly distributed with the vast majority concentrated in a few states. While inventory and sales vary year to year, the largest swine-producing states include Iowa, North Carolina, Minnesota, Illinois, and Nebraska; the largest broiler chicken producers include Arkansas, Alabama, Mississippi, North Carolina, and Texas; and the largest turkey producers include North Carolina, Arkansas, Minnesota, Missouri, and South Carolina (USDA 2014a). This investigation did not include cattle CAFOs; however, cattle production is especially large in Texas, Nebraska, Kansas, Oklahoma, Iowa, Mississippi, and Wisconsin (USDA 2014a). The CAFO system has currently expanded beyond the USA and northern Europe into Eastern Europe and South America (Thorne 2007).

Serious human health (Cole et al. 2000; Wing and Wolf 2000) and environmental impacts (Campagnolo et al. 2002; Burkholder et al. 2007) of industrial animal production have been documented by researchers, including water quality impacts. The acute polluting impacts of large-scale accidents and hurricanes involving CAFO waste on freshwater streams and estuaries have been well documented (Burkholder et al. 1997; Mallin 2000; Mallin et al. 1999). The location of many CAFOs on river and stream floodplains renders receiving water vulnerable to such accidents, especially during major storms (Wing et al. 2002). CAFOs have been cited as supplying excessive nitrate to Midwestern streams in Ohio and Iowa (Weldon and Hornbuckle 2006; Hoorman et al. 2008). Karr et al. (2001) were able to trace nitrogen that was derived from CAFOs several kilometers downstream using isotopic techniques. A broad-scale study found that streams whose watersheds contained swine and poultry CAFOs had significantly higher concentrations of ammonium, nitrate, and total N than streams whose watersheds lacked CAFOs (Harden 2015). Additionally, the deposition of manure and urine in storage lagoons and on surrounding fields has caused ammonium and/or nitrate pollution of groundwater on or near production facilities (Ritter and Chirnside 1990; Westerman et al. 1995; Liebhardt et al. 1979).

However, chronic pollution of surface waters by a suite of potential pollutants has not been comprehensively assessed for a CAFO-rich watershed. This research investigated physical, chemical, and biological pollution of stream waters in a watershed containing numerous swine and poultry CAFOs while lacking industrial or municipal point sources of pollution and containing little traditional crop agriculture. The degree of pollution was determined from two perspectives: first, since this stream consists of public waters, it was of interest to investigate whether or not these waters were impaired based on state chemical and biological standards. Second, comparative water quality conditions based on parameter concentrations were made using the published literature on coastal plain streams as well as broad-scale literature analyses of streams in general.

Stocking Head Creek (Fig. 1) is a second-order blackwater stream located in the Northeast Cape Fear River basin in eastern North Carolina. Catchment area is 1980 ha (4893 acres), and stream length to where it enters the Northeast Cape Fear River is 22.1 km (13.7 mi). The watershed soils are dominated by Noboco loamy fine sand, Johns fine sandy loam, Autryville loamy fine sand, Pactolus fine sand, Lumbee sandy loam, and Marvyn and Gritney soils (NRCS 2014a). There is some traditional row crop agriculture within this watershed, but aerial photography indicates that coverage by such is small in comparison to CAFO sprayfields and forest cover. The Northeast Cape Fear River is a fifth-order tributary of the sixth-order Cape Fear River, the watershed of which contains approximately half of the 9,000,000 plus swine in North Carolina as well as vast numbers of confined poultry. Cahoon et al. (1999) estimated that the Cape Fear River basin annually received 82,700 metric tons of nitrogen and 26,000 metric tons of phosphorus as animal waste in this watershed from CAFOs alone. Thus, CAFO-rich subwatersheds are likely to be sources of considerable nutrient pollution to larger rivers and estuaries (Mallin and Cahoon 2003; Burkholder et al. 2007). As such, our primary objective was to investigate potential environmental impacts to a stream draining a swine and poultry

CAFO-rich watershed. From July to October 2013, we sampled four main channels and three tributary stations for a broad selection of physical, chemical, and biological water quality parameters.

The location of swine CAFOs and the permitted numbers of swine for each are available from the North Carolina Department of Environment and Natural Resources, and we used these data to map and enumerate swine CAFOs. However, the N.C. Department of Agriculture and Consumer Services (NCDA & CS) does not provide such watershed-specific information on poultry CAFOs to the public or other agencies, although counts of each type of livestock are available on a county-by-county basis from NCDA & CS. Thus, a second objective was to obtain the locations of poultry CAFOs within this basin and estimate poultry numbers confined within them using alternative GIS and aerial photography-based techniques.

2 Methods

2.1 Sample Locations and Frequency

Seven stations were sampled, four on the main stream and two on first-order tributaries (Table 1, Fig. 1). The seventh station (MC-50) was collected on second-order Maxwell Creek, which joins lower Stocking Head Creek before it enters the Northeast Cape Fear River. The Maxwell Creek watershed also contains CAFOs, but they are not as concentrated near stream waters as those in the Stocking Head Creek watershed and they are not quantified in this investigation; MC-50 is used as a comparison site herein. All sites were sampled from bridges on public right-of-ways. The sampling design included five sample trips each taken during two different 30-day periods, one in mid-summer and one in fall. This was planned in accordance with the North Carolina Department of Environment and Natural Resources (NCDENR)'s protocol for fecal coliform sampling for assessment of whether a given stream supported its designated use or if it belongs on the state's 303(d) list for impaired waters. Sampling of Stocking Head Creek occurred during both dry and wet periods. Rainfall data were obtained from the NC CRONOS data set, using Station #319026 Wallace, latitude 34.72, longitude 77.97778, in Duplin County. Rainfall amount was computed for the day of sampling, the day of sampling plus the previous 24-h period, and the day of sampling plus the previous 48-h period.

2.2 Sample Parameters and Methodology

To obtain a full perspective of the stream's physical and chemical qualities, a suite of parameters was sampled. Field measurements were made on-site using YSI field meters calibrated and checked according to standard procedures for water temperature, pH, dissolved oxygen, turbidity, and specific conductance. Samples were collected from surface waters by bucket haul and distributed into pre-cleaned bottles for nutrient (ammonium, nitrate, total nitrogen, orthophosphate, total phosphorus), total suspended solids (TSS), chlorophyll *a*, and 5-day biochemical oxygen demand (BOD5) analyses. Samples were kept on ice and delivered to the laboratory for subsequent analysis within proper holding times. Chain of custody records was maintained.

Analytical methods used (see APHA 1995; USEPA 1983, 1997) were as follows: TSS, SM 2540D; ammonium, EPA 350.1; nitrate+nitrite, EPA 353.2; TKN, EPA 351.2, total nitrogen as calculation of TKN+nitrate; orthophosphate, SM 4500PE; total phosphorous, SM 4500 PE; BOD5, SM 5210B; and fecal coliform bacteria, SM 9222D MF. Total organic carbon concentrations (TOC) were obtained during the fall sampling in an effort to better understand causes of high BOD5 levels that were seen in the summer; the analytical method used for TOC was SM 5310B. Chlorophyll *a* measurements were performed using EPA method 445.0, based on the Welschmeyer (1994) fluorometry method.

2.3 Statistical Analyses

Summary statistics were derived for each parameter (means, standard deviations, medians, minimum, maximum for all data, also geometric means for fecal coliform analysis). Data were tested for normality using the Shapiro-Wilk test, with most chemical and biological parameters requiring log-transformation prior to further statistical analysis. An important consideration was whether or not the pollutant concentrations measured in Stocking Head Creek were the result of acute stormwater-driven surface runoff into the creek or a result of chronic, long-term pollution impacting groundwater as well as surface waters. As such, we analyzed whether or not rainfall produced higher pollutant

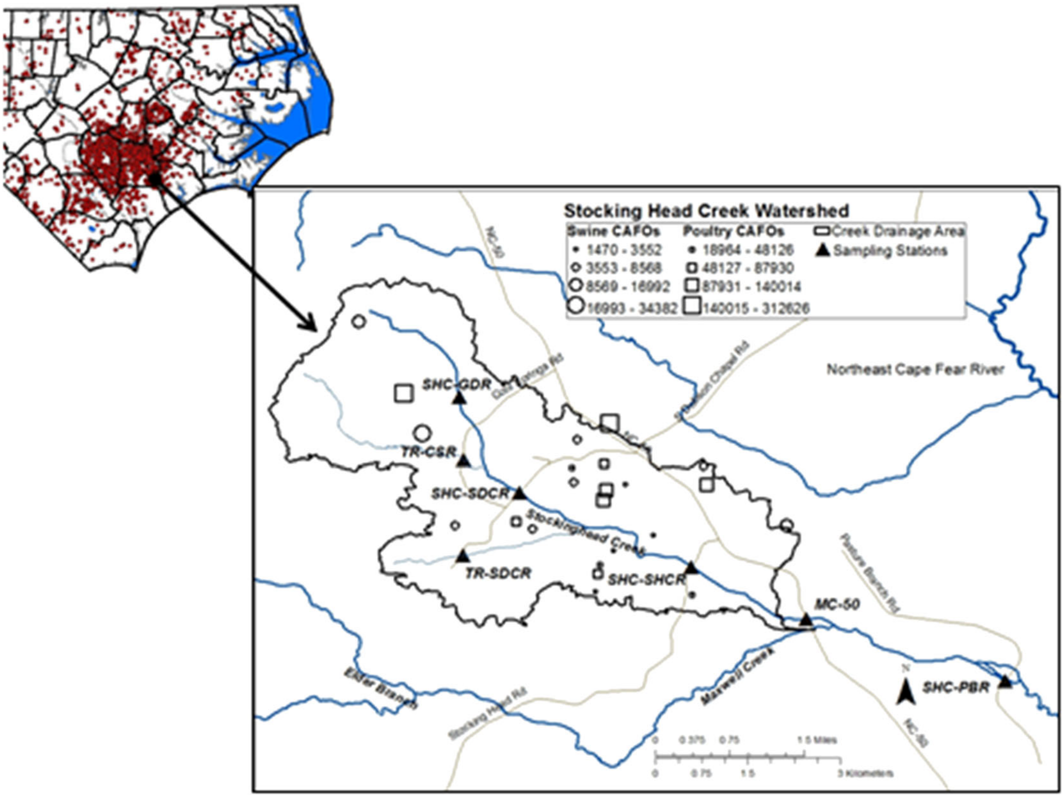

Fig. 1 Map of the Stocking Head Creek watershed, eastern North Carolina, USA, including streams, roads, sampling sites, and swine and poultry CAFOs sorted by size groups. *Inset* is the North Carolina Coastal Plain with *red dots* indicating location of swine CAFOs

parameter concentrations than occurred on non-rain periods. Measurable rainfall occurred either on the day of sampling or within the 48 h preceding the sample day on five of 10 sampling occasions. They were August 1 and 13, September 24, and October 8 and 10. *T* tests were used to test selected parameter concentrations between wet and dry periods (α=0.05). Parameters tested for wet-dry differences included ammonium, nitrate, and fecal coliform bacterial concentrations. To assess potential chemical and biological parameters influencing BOD, correlation and regression analyses were performed using SAS (Schlotzhauer and Littell 1987).

2.4 CAFO Map Construction

A digital elevation model was downloaded from the USGS geospatial portal and used as the data input to delineate the watershed boundaries of Stocking Head Creek. The ArcMap 10.1 Hydrology toolset was utilized, and the catchment area of Stocking Head Creek system was identified. A shapefile including all of the documented animal operations from the NC OneMap geospatial portal was clipped to only display those CAFOs within the newly defined watershed area. After establishing these boundaries and existing CAFOs (primarily swine operations), 2012 orthophotography from the North Carolina OneMap service was analyzed for undocumented CAFOs. The signature shape of the farm buildings (long rectangles side by side) was used to identify these locations, which were presumed to be poultry CAFOs. These were manually digitized as polygons superimposed on the aerial photos, and added to the existing CAFO location data to provide a more accurate assessment of the total number of animal operations within the Stocking Head Creek watershed. Besides the lack of inclusion on the NCDENR database, other characteristics that distinguish North Carolina poultry from swine CAFOs from the air include a lack of waste lagoons for poultry (all swine CAFOs contain waste lagoons but few poultry CAFOs do—egg-laying facilities only). Also, based on aerial photography, poultry CAFO structures are generally longer than swine

Table 1 Description and location of sampled sites, roughly in descending order from headwaters downstream in Stocking Head Creek. See Fig. 1 for locations, nearby roads, and CAFOs

SHC-GDR (Stocking Head Creek at Graham Dobson Road): N 34.91197, W 77.94507. This location collects the uppermost branch of Stocking Head Creek; a swine CAFO and sprayfields are present several hundred of meters upstream from the sampling site. Aerial photos indicate that this stream initiates in what is now a swine CAFO sprayfield.

TR-CSR (unnamed first-order tributary at Cool Springs Road): N 34.90279, W 77.94440. This site had no immediately adjoining CAFOs or sprayfields, but the GIS map (Fig. 1) indicates a large swine CAFO and a large poultry CAFO upstream.

TR-SDCR (unnamed first-order tributary entering Stocking Head Creek at South Dobson Chapel Road): N 34.88878, W 77.94453. Grazing cattle were occasionally present upstream within 75 m of this site, and waste spraying equipment was stationed within 50 m upstream on several sampling dates.

SHC-SDCR (Stocking Head Creek at South Dobson Chapel Rd.): N 34.89796, W 77.93628. Numerous CAFOs, sprayfields, and grazing cattle were located within 200 m of the stream at this site.

SHC-SHCR (Stocking Head Creek at Stocking Head Road): N 34.88710, W 77.91124. A large CAFO sprayfield was located within 50 m of the stream, with a ditch carrying sprayfield runoff that emptied directly into the stream at this location.

MC-50 (Maxwell Creek at SR 50): N 34.87950, W 77.89438. This sampling site adjoined a wetland area which was hydrologically connected to Stocking Head Creek. The Maxwell Creek watershed also contains CAFOs, although fewer than those in Stocking Head Creek watershed (Fig. 1).

SHC-PBR (Stocking Head Creek at Pasture Branch Road): N 34.87043, W 77.86539. This location was the farthest downstream site sampled. This downstream area also likely receives inputs from CAFOs in the Maxwell Creek drainage. There was also an adjoining forested wetland that supplied flow to the stream here.

CAFOs and have the feeding silo located at the building midpoint, while swine CAFOs have feeding silos located at the building's end.

The dimensions of each poultry building were computed from the digitized aerial photographs. Maximum bird (considered as broiler chickens) populations per building were estimated by assuming 743 cm^2 (0.80 ft^2) of space allotted per bird as is standard for a major poultry producer (Sanderson Farms 2007). The United Egg Producers (2010) recommend 436–557 cm^2 (0.47 to 0.60 ft^2) of space per egg-laying chicken; thus, we feel our counts are conservative. As there is no way to distinguish chicken from turkey operations from the air, for the purposes of this study, we assumed all broiler chickens. As a comparative reference regarding livestock manure production, the National Resources Conservation Service within the US Department of Agriculture uses animal units, i.e., 1 cow=9 hogs=455 broiler chickens=67 turkeys (NRCS 2014b).

3 Results

The Stocking Head Creek watershed (excluding Maxwell Creek) contains 13 swine CAFOs that are permitted by NCDENR for collectively 108,068 heads of swine. This watershed also contains 11 poultry CAFOs, consisting of a total of 42 individual buildings. Average poultry house size was approximately 2323 m^2 (25,000 ft^2), with an average capacity of 31,250 birds. Thus, the watershed can house a maximum of 1,312,500 broiler chickens or equivalent turkeys using the animal unit conversions above. Grazing cattle were visibly present in this watershed (with some photographed grazing directly under swine waste sprays), but an accurate count is beyond the capability of this study. Confined swine, poultry, and cattle produce large quantities of manure; conversion factors of excreted waste into total nitrogen, phosphorus, and fecal bacteria for these three livestock types can be found in Mallin and Cahoon (2003). While the local human population relies on septic systems for sewage treatment, aerial photography revealed only 67 human dwellings in the watershed, yielding a scant 0.03 septic systems/ha.

Summer water temperatures ranged between 22.0 and 28.0 °C, and fall water temperatures ranged from 16.1 to 22.6 °C. Most sampling events reflected neutral stream pH conditions ranging from 6.5 to 7.3. Dissolved oxygen concentrations ranged from mildly hypoxic (3.5 mg/L) to supersaturation (17.0 mg/L) during an algal bloom. Average turbidity by station ranged from 1.1 to 21.0 NTU (nephelometric turbidity units). Total suspended solids (TSS) in most cases were less than 25 mg/L (Table 2). Elevated TSS concentrations occurred a few times primarily at the tributary station TR-SDCR.

Ammonium in Stocking Head Creek during the 10 sample trips ranged from the detection limit (0.05 mg/L) to 37.8 mg/L (Table 2). Highest ammonium concentrations were found at station TR-SDCR, followed by station SHC-SHCR. These stations are both within 50 m of swine CAFO sprayfields (Table 1, Fig. 1). Nitrate concentrations in Stocking Head Creek were very high (Table 2). Whereas the highest ammonium

Table 2 Water quality parameter concentrations in Stocking Head Creek, 2013, given as mean±standard deviation/median/range; fecal coliforms as geometric mean/range. n=70, except for TOC where n=35

Parameter	Station						
	TR-SDCR	SHC-GDR	TR-CSR	SHC-SDCR	SHC-SHCR	MC-50	SHC-PBR
Turbidity	25.6+20.1	4.5±2.2	11.4±9.2	9.5±5.4	12.4±5.8	1.0±2.3	4.0±2.5
NTU	21.0	4.1	9.5	10.0	12.0	1.1	3.8
	4.0–72.0	2.0–10.0	1.0–31.0	2.0–22.1	6.3–22.0	0.0–8.0	1.0–8.0
TSS	23.9±18.2	5.6±2.4	15.0±28.3	9.5±3.6	10.9±4.4	3.2±1.6	4.2±3.5
mg/L	21.3	5.4	6.4	9.9	11.3	3.4	3.4
	5.8–56.7	3.3–10.8	1.4–94.0	2.9–14.9	5.2–19.0	1.4–6.1	1.3–12.1
DO	7.4±4.0	5.9±0.8	7.2±0.5	7.9±0.8	7.4±0.3	6.0±0.9	6.4±0.8
mg/L	5.8	5.8	6.9	7.7	7.4	5.9	6.1
	3.5–17.0	4.7–7.3	6.8–8.4	7.0–9.3	6.9–7.9	5.0–7.7	5.6–7.9
Ammonium	10.5±13.6	0.1±0.1	0.3±0.2	0.2±0.2	3.3±4.1	0.1±0.1	0.4±0.5
mg-N/L	3.6	0.1	0.2	0.1	1.0	0.1	0.1
	0.2–37.8	0.1–0.2	0.1–0.7	0.1–0.8	0.2–10.9	0.1–0.1	0.1–1.6
Nitrate	2.9±3.6	6.8±4.2	7.9±2.2	3.6±2.2	6.1±2.1	0.3±0.2	1.3±0.6
mg-N/L	1.4	6.0	8.4	3.9	6.3	0.2	1.1
	0.1–10.0	1.6–13.6	3.0–10.5	0.6–7.4	1.1–8.4	0.1–0.7	0.8–2.4
TN	15.7±16.7	7.2±4.2	8.4±1.8	3.9±2.2	8.7±4.1	0.5±0.3	1.8±0.9
mg-N/L	7.6	6.2	8.6	4.2	7.9	0.5	1.6
	0.5–46.6	2.1–13.6	4.3–10.8	0.8–7.4	2.1–16.1	0.1–0.9	0.8–3.5
OP	1.8±1.7	0.1±0.0	0.1±0.1	0.2±0.1	0.4±0.2	0.3±0.1	0.3±0.1
mg-P/L	1.6	0.1	0.1	0.2	0.3	0.3	0.3
	0.1–5.5	0.1–0.2	0.1–0.4	0.1–0.3	0.2–0.6	0.2–0.4	0.2–0.4
TP	2.8±8.3	0.2±0.1	0.2±0.2	0.2±0.1	0.5±0.2	0.3±0.1	0.4±0.1
mg-P/L	1.7	0.2	0.1	0.2	0.4	0.3	0.4
	0.2–10.7	0.1–0.3	0.1–0.6	0.1–0.3	0.3–0.8	0.2–0.4	0.3–0.6
TOC	36.1±13.9	14.2±1.2	10.9±0.9	13.7±1.6	19.0±3.9	17.5±1.4	16.2±0.8
mg-C/L	39.9	13.5	11.2	13.0	17.4	18.1	15.7
	13.1–50.4	13.2–15.5	9.6–11.9	12.5–16.3	14.3–23.1	15.1–18.4	15.5–17.4
Chlorophyll *a*	12.3±11.8	10.1±8.4	2.9±2.1	5.2±2.8	10.7±11.9	6.1±8.2	1.2±0.6
μg/L	8.5	8.0	2.0	5.0	7.0	2.0	1.0
	1.0–40.0	2.0–28.0	1.0–8.0	1.0–12.0	3.0–44.0	1.0–25.0	0.0–2.0
BOD5	18.7+25.7	2.0±1.2	1.7±0.8	2.7±3.5	7.4±8.3	3.2±6.3	1.4±0.5
mg/L	11.0	1.5	1.5	1.0	3.5	1.0	1.0
	1.0–88.0	1.0–4.0	1.0–3.0	1.0–12.0	1.0–25.0	1.0–21.0	1.1–2.0
Fecal col.	9126	1184	1470	1772	5863	220	391
CFU/100 mL	455–60,000	330–3000	546–5000	728–8000	1182–60,000	91–1360	55–4000

concentrations were found at the two sites located closest to waste sprayfields, several sites showed high nitrate, including sites distant from sprayfields (Table 2; Fig. 1). Concentrations ranged from 0.08 to 13.60 mg-N/L, with station means ranging from 0.30 to 7.94 mg-N/L. Particularly high nitrate concentrations were documented at these four sites: SHC-GDR, TR-CSR, SHC-SDCR, and SHC-SHCR. On 12 of 70 sampling occasions, stream nitrate concentrations exceeded 8 mg-N/L. Total nitrogen (TN) concentrations ranged from 0.11 to 46.70 mg-N/L, while highest individual station average TN concentrations occurred at TR-SDCR, TR-CSR, and SHC-SHCR. The TN values in this stream were dominated by inorganic nitrogen (i.e., nitrate and ammonium) rather than organic nitrogen, with average percent inorganic N ranging from 80 to 100 % of TN, depending upon station.

Orthophosphate concentrations in Stocking Head Creek ranged from 0.04 to 5.45 mg-P/L, with station means ranging from 0.11 to 1.78 mg-P/L (Table 2). Highest concentrations were found at station TR-SDCR, followed by SHC-SHCR. Total phosphorus ranged from 0.040 to 10.70 mg-P/L, and station means ranged from 0.15 at SHC-GDR to 2.83 mg-P/L at TR-SDCR, with SHC-SHCR second highest at 0.50 mg-P/L (Table 2). Most TOC concentrations were in the 10–20 mg/L range; however, higher TOC concentrations occurred at TR-SDCR and SHC-SHCR, the stations nearest to sprayfields (Table 2).

Chlorophyll *a* represents the amount of suspended live microalgal biomass found in a sample of water (Wetzel 2001). Elevated chlorophyll *a* concentrations (algal blooms) occurred at TR-SDCR on July 29 (40 μg/L) and at this same site on September 18 (44 μg/L), with smaller blooms occurring several times at other sites. Thus, algal blooms occurred within Stocking Head Creek, but were inconsistent in time and among sampling sites (Table 2).

Biochemical oxygen demand (BOD) is a measure of the organic matter available for consumption by the bacteria in a body of water during respiration; excessive BOD can lead to hypoxia. Five-day BOD (BOD5) varied widely (Table 2), from background concentrations of 1.0 mg/L up to high values of 21 mg/L at MC-50, 25 mg/L at SHC-SHCR, and the maximum of 88 mg/L at station TR-SDCR. That station maintained the highest overall BOD5 concentrations (Table 2), reaching 10 mg/L or more on six of 10 dates, while station SHC-SHCR exceeded 10 mg/L on three dates. The stream stations with highest average BOD concentrations were those in closest proximity to swine waste sprayfields (Table 1; Fig. 1).

North Carolina uses fecal coliform bacteria counts as a proxy for potentially pathogenic bacteria in fresh water bodies; this standard is commonly used throughout the southeast of the USA (EPA Region 4) for freshwater contact. The NC protocol (NCDENR 1999) for sampling and the means for determining fecal impairment of a water body state that fecal coliform counts shall not exceed a geometric mean of 200 CFU/100 mL based on at least five consecutive samples during any 30-day period nor exceed 400 CFU/100 mL in more than 20 % of the samples examined during such period. Fecal coliform counts for Stocking Head Creek were generally very high (Fig. 2). During summer 2013, the upper five stations exceeded 400 CFU/100 ml on 100 % of the time sampled, and the geometric means for all seven stations exceeded 200 CFU/10 mL for five samples in 30 days. Fecal coliform counts for Stocking Head Creek in fall 2013 were even higher than those in summer; the upper five stations exceeded 400 CFU/100 ml on 96 % of the time sampled, and the geometric means for six of the seven stations exceeded 200 CFU/10 mL for five samples in 30 days. During both the summer and the fall sampling periods, fecal coliform criteria for impaired waters were well exceeded. Elevated fecal coliform counts occurred during both wet and dry periods. Highest fecal coliform counts occurred at TR-SDCR and SHC-SHCR, the stations nearest to sprayfields. Most other stations also had high counts, with geometric means exceeding 1000 CFU/100 mL (Table 2, Fig. 2). While lower Maxwell Creek (MC-50) maintained the lowest counts, it still exceeded state criteria for impaired waters.

An important consideration is whether or not the high pollutant values measured in Stocking Head Creek were the result of acute stormwater-driven surface runoff into the creek or a result of chronic, long-term pollution impacting groundwater as well as surface waters. As such, we analyzed whether or not rainfall produced higher pollutant parameter concentrations than occurred

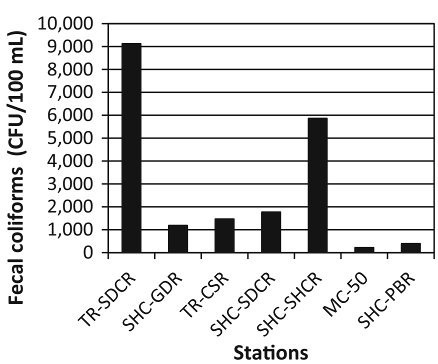

Fig. 2 Fecal coliform bacteria counts (as colony-forming units/100 mL) for Stocking Head Creek watershed sampling sites, presented as geometric mean of 10 samples per site. Note that the NC freshwater recreational standard is 200 CFU/100 mL

on non-rain periods. For all non-rain sample dates and stations, the fecal coliform geometric mean was 1455 CFU/100 mL, and counts exceeded 200 CFU/100 mL on 31 of 35 samples with 89 % exceedence of the state standard. For all rain periods and stations combined, the fecal coliform geometric mean was 1467 CFU/100 mL, and counts exceeded 200 CFU/10 mL on 30 of 35 samples with 86 % exceedence of the state standard. *T* test results showed no significant difference in means between wet and dry periods (p=0.49; df=68). Thus, fecal coliform pollution of Stocking Head Creek was not rain dependent; rather, fecal coliform pollution was a *chronic* condition.

Ammonium concentrations during non-rain periods were 2.67±7.59 mg-N/L (mean±standard deviation) vs. rain period concentrations of 1.56±4.65 mg-N/L. *T* test results on log-transformed data showed no significant difference in means (p=0.64; df=68). Nitrate concentrations during non-rain periods were 4.45±3.68 mg-N/L vs. rain concentrations of 3.82±3.56 mg-N/L. *T* test on log-transformed nitrate data showed no significant difference in means (p=0.38, df=68). Thus, inorganic nitrogen concentrations were not increased by rainfall-driven surface runoff, but instead they were a chronic condition in Stocking Head Creek, indicating groundwater pollution.

4 Discussion

Ammonium is a reduced form of inorganic nitrogen that is a major component of fresh human sewage or animal excreta (Clark et al. 1977). It is readily used by algae and bacteria, and an overabundance of ammonium can stimulate eutrophication (Wetzel 2001). Research in coastal plain blackwater streams and rivers has indicated that ammonium concentrations of 0.5 mg/L (ppm) or higher can stimulate algal blooms (Mallin et al. 2004). Additionally, ammonium exerts a significant chemical oxygen demand in sewage treatment plant discharges (Clark et al. 1977), as it is oxidized to nitrate. Thus, excess ammonium can lead to aquatic ecosystem deterioration through more than one pathway. The ammonium concentrations found in Stocking Head Creek, particularly at the sites nearest to sprayfields, greatly exceeded ammonium concentrations typically found in other coastal plain streams and rivers (Smock and Gilinsky 1992; Mallin et al. 2004, 2006). Previously, only during swine or poultry lagoon breaches have such concentrations been found in receiving streams in this region (Burkholder et al. 1997; Mallin 2000).

Ammonium pollution from CAFOs is not limited to runoff or seepage entering waterways. Volatilization of ammonia from CAFOs releases vast amounts of inorganic nitrogen to the atmosphere. On the North Carolina Coastal Plain alone, annual ammonia emissions from swine and poultry have been estimated as 73,500 and 22,900 metric tons (Costanza et al. 2008). Regionally, ammonia volatilization from waste lagoons, sprayfields, and litterfields carries ammonia well outside of the watershed of origination (Walker et al. 2000; Costanza et al. 2008) and is reflected in elevated ammonium concentrations in rainfall measurements downwind (Willey et al. 2006). In a study of the two most CAFO-rich North Carolina watersheds, ammonium concentrations were found to have increased approximately 500 % in the Neuse River and 315 % in the Northeast Cape Fear River between 1995 and 2005 (Burkholder et al. 2006).

Ammonium at the concentrations found in Stocking Head Creek can have other impacts besides increasing algal blooms and chemical oxygen demand. Recent studies (see review by Glibert et al. 2015) have indicated that ammonium at these concentrations can stimulate cyanobacterial production as well as increase the production of toxic microcystin, while suppressing growth of diatoms which are generally benign organisms that support the higher trophic levels. Ammonium has been found to preferentially enhance cyanobacterial growth in locales as diverse as San Francisco Bay (Glibert et al. 2014) to the South Carolina Coastal Plain (Siegel et al. 2011). Interestingly, in July 2011, an unprecedented (during 23 years of monitoring) bloom of cyanobacteria (mostly *Anabaena planctonica*) occurred in the blackwater Northeast Cape Fear River downstream from its confluence with Stocking Head Creek (and several other CAFO-impacted watersheds). The bloom lasted for several weeks and, upon decomposition, resulted in a BOD that decreased river DO from 5.3 to 0.7 mg/L (S. Petter Garrett, NCDENR, personal communication August 4, 2011).

Nitrate is likewise readily used by visible plants and algae for growth. It is mobile in soils and readily moves through the water table to enter streams (Keeney 1986). Average nitrate concentrations at six of seven stations well exceeded levels known to stimulate algal production and lead to elevated BOD in blackwater streams (Mallin et al. 2004). Nitrate concentrations in this stream were well in excess of those reported from other streams in the southeast USA (Edwards and Meyer 1987; Smock and Gilinsky 1992; Mallin et al. 2004, 2006; Carey et al. 2007), but in line with nitrate concentrations reported from Ohio watersheds impacted by runoff from combined row crop and dense CAFO presence (Hoorman et al. 2008). Regarding human health, there is a US EPA drinking well nitrate standard of 10 mg-N/L to prevent blue baby syndrome (also called methemoglobinemia). It is notable that on three occasions, even the 10-mg/L standard for drinking well water was exceeded, and in seven of 70 samples collected, stream nitrate concentrations exceeded 9 mg/L, close to the methemoglobinemia standard.

There are no point-source discharges entering this creek. The local human population uses septic systems, but the 67 human dwellings in the watershed are spaced well away from the creek, with sprayfields located between human dwellings and riparian areas. Thus, the principal sources of nitrate are swine CAFO waste (either runoff from sprayfields, or subsurface movement into the stream), poultry litter spread on fields, and cattle manure deposited on land. Nitrate concentrations of similar magnitude have been documented from subsurface waters draining sprayfields and surface streams passing through or near swine sprayfields (Evans et al. 1984; Stone et al. 1995). Total nitrogen concentrations in Stocking Head Creek were very high compared to available data from other blackwater coastal plain streams (Smock and Gilinsky 1992; Mallin et al. 2004, 2006). To provide a wider perspective, using a large data set of 1070 streams, Dodds et al. (1998) determined that average TN concentrations >1.5 mg/L were characteristic of eutrophic conditions; this level was well exceeded according to average station TN at six of the seven sites (Table 2).

Orthophosphate is the most common form of inorganic phosphorus directly used by algae. It is not very mobile in soils and adsorbs readily to soil particles (Wetzel 2001). Orthophosphate station means were generally 2–10 times the average levels found in a selection of less-impacted blackwater coastal plain streams (Edwards and Meyer 1987; Mallin et al. 2004, 2006; Carey et al. 2007). Average orthophosphate concentrations in Stocking Head Creek were similar to those in streams draining mixed row crop and CAFO watersheds in Ohio (Hoorman et al. 2008).

Concentrations of TP≥0.50 mg-P/L or greater can increase BOD in blackwater streams by serving as a substrate assimilated by ambient bacteria (Mallin et al. 2004). In the present study, TP was higher than 0.50 mg-P/L in 11 of 70 samples. Using data from 1366 streams, Dodds et al. (1998) concluded that TP concentrations >0.075 mg/L were characteristic of eutrophic streams; average TP at all sites exceeded this threshold (Table 2). Additionally, a study of soils in eastern North Carolina using a soil phosphorus index developed by the N.C. Division of Agronomy found that the soils in counties where CAFOs were abundant (including Duplin County) contained excessively high index values (Cahoon and Ensign 2004). We note that besides direct eutrophication impacts, highly variable nutrient ratios (such as seen with this impacted stream) can lead to changes in taxonomic structure for resident phytoplankton and higher trophic levels (Glibert et al. 2015).

Bacteria require phosphorus both structurally and energetically (Kirchman 1994), and fecal bacteria growth in stream sediments can be stimulated by inputs of phosphate (Toothman et al. 2009). Also, fecal coliform bacteria within the water column can be significantly stimulated by additions of organic or inorganic P inputs >0.100 mg/L, increasing survival and reproduction (Chudoba et al. 2013). Mean and median concentrations of TP in all Stocking Head Creek stations exceeded 0.100 mg-P/L. The data suggest that high phosphorus inputs to stream waters polluted by fecal bacteria can magnify human health risks, as well as ecosystem impacts.

BOD5 concentrations at times were very high in Stocking Head Creek (Table 2). Comparison of BOD5 from many streams and rivers in coastal North Carolina indicates that concentrations of 1 to 2 mg/L are background for minimally impacted streams (Mallin et al. 2006). Elevated BOD can be stimulated by several causes. One common cause of elevated BOD is the introduction of organic materials such as human sewage or animal waste into the water; thus, dissolved organic carbon, if labile, can stimulate BOD (Clark et al. 1977). Another cause is algal blooms, which upon death and decay create a source of labile organic matter available for bacterial consumption. In a variety of coastal plain freshwater streams, tidal creeks, and lakes, Mallin et al. (2006) found strong statistical correlations between BOD and chlorophyll *a*. Ammonium pollution can exert a significant chemical oxygen demand in waterways, and elevated phosphorus concentrations can lead to elevated BOD by directly stimulating bacteria growth. In Stocking Head Creek, BOD5 was positively correlated with TOC (r=0.833. p<0.0001), ammonium (r=0.666, p<0.0001), TP (r=0.626, p<0.0001), orthophosphate (r=0.569, p<0.0001), chlorophyll a (r=0.316, p=0.008), and TN (r=0.284, p=0.017). Multiple regression analyses indicated that the best predictive linear model for BOD5 in this stream was (as log-transformed data)

$$\mathrm{BOD5} = 0.952(\mathrm{TOC}) + 0.367(\mathrm{AMM})–3.961,\ r^2 = 0.85,\ p < 0.0001$$

Thus, the high BOD in Stocking Head Creek is directly related to common components of animal waste (TOC, ammonium, phosphorus) as well as to chlorophyll a, a response variable to nutrient inputs. As mentioned, the lower Cape Fear River and its estuary are on the 303(d) list due to DO violations. Stocking Head Creek enters the Northeast Cape Fear River, which enters the lower Cape Fear River at Wilmington. The high levels of BOD observed in Stocking Head Creek as well as the high nitrate, ammonium, and fecal bacteria loads contribute to the low DO concentrations frequently occurring in summer in the Northeast Cape Fear River.

Most troublesome from a human health perspective is the data indicating that Stocking Head Creek is highly polluted by fecal bacteria, by both measures of the NC criteria for impaired waters. The upper five stations exceeded 400 CFU/100 ml on 96–100 % of the time sampled, and six of seven stations exceeded a geometric mean of 200 CFU/10 mL for five samples in both 30-day periods. Importantly, elevated fecal coliform counts occurred during both wet and dry periods; this creek is chronically polluted by fecal bacteria. The stimulatory effect of phosphorus loading on fecal bacteria (Chudoba et al. 2013) further exacerbates the potential human health issues. Fecal bacteria generated by livestock within the watershed are not confined to the immediate watershed but are likely to be carried downstream into higher order streams. A bacterial source-tracking study using molecular techniques demonstrated swine waste contamination at Cape Fear River system sites well downstream from swine CAFOs (Arfken et al. 2013).

In addition to surface waters, groundwater under and near swine and poultry CAFOs can contain very high inorganic nitrogen concentrations. Ritter and Chirnside (1990) found ammonium concentrations up to 960 mg-N/L and nitrate up to 50 mg-N/L in test wells in close proximity to swine waste lagoons on the Delmarva Peninsula. In North Carolina, Westerman et al. (1995) analyzed seepage from two swine waste lagoons and found ammonium concentrations in nearby wells averaged more than 50 mg-N/L, with nitrate of 6–15 mg-N/L. In other areas, it has also been observed that both spreading and spraying of livestock waste on the landscape will lead to excessive nitrate in groundwater (Liebhardt et al. 1979).

The water table in this area varies seasonally but is relatively near the land surface on average. No groundwater level monitoring wells are immediately on-site, but the US Geological Survey operates a well 10 km southwest at Rose Hill (Well DU-157). At that well, annual average depth to the surficial water table from 2004 to 2014 ranged from 1.3 to 2.7 m, with an average of 2.2 m. The North Carolina Division of Water Resources operates a monitoring well 10 km southeast at Chinquapin (Well W29D9), and 2013 data showed depth to surficial water table ranging from 1.4 to 1.8 m below the land surface. The local predominating sandy loam soils (NRCS 2014a) have moderate to rapid permeability with permeability rates of 0.5–1.5 m/day and 1.5–3.0 m/day, respectively (USDA 2014b). Thus, following a swine waste spray event, nitrate (and likely some portion of the fecal bacteria load) could migrate into the water table in as little as a day or two and from there move laterally to the nearest stream. Thus, groundwater in a CAFO-rich watershed such as that of Stocking Head Creek is a source of nitrogen and fecal bacterial pollution to the stream waters, and continual (including non-storm event) groundwater inputs into the stream at selected locations results in chronic pollution. Simply considering overland runoff will underestimate the N flux to aquatic systems as this ignores infiltration and leaching (Carpenter et al. 1998). The lack of concentration differences in fecal coliform, ammonium, and nitrate concentrations between rainy and dry periods shows that the stream pollution is chronic and a result of normal CAFO operations and presently accepted waste disposal techniques.

This research has demonstrated that drainage basins rich in CAFOs cause chronic pollution that has both human health and ecosystem impacts. However, the scope of US confined animal operations is nationwide. Many CAFO-rich watersheds pollute freshwater streams and rivers, which eventually enter estuaries located on the Atlantic and Gulf coasts. For instance, poultry CAFO-rich Delaware, Maryland, and Virginia, and poultry and swine CAFO-rich North Carolina drain into middle-Atlantic estuaries. Major swine, poultry, and cattle-producing states such as Arkansas, Alabama, Iowa, Illinois, Minnesota, Missouri, and Texas in the Mississippi River drainage feed the Gulf of Mexico. As such, Weldon and Hornbuckle (2006) determined that for four major agricultural watersheds in Iowa, nitrate was strongly correlated with CAFO densities, and these watersheds made an outsized contribution to nitrate loading to the Mississippi River.

An extensive study covering 90 % of estuarine surface area in the USA (Bricker et al. 1999) concluded that severe eutrophication conditions (toxic algal blooms, bottom-water hypoxia, losses of submersed aquatic vegetation) were most prevalent along the middle Atlantic and Gulf Coast estuaries. Howarth et al. (2012) have demonstrated that estuarine nitrogen discharge from a wide selection of rivers in Europe and North America is positively correlated with net watershed nitrogen inputs. That study showed that for watersheds that have positive increases in animal feed from outside the system, there is a strong correlation with riverine N flux. Many of those rivers drain watersheds rich in poultry CAFOs, swine CAFOs, or both. The magnitude of industrial livestock production indicates that not only are immediate watersheds severely polluted but the collective impacts of the numerous subwatersheds draining CAFO-rich areas contribute to major ecosystem impacts far downstream as well. As the magnitude of the CAFO style of industrial livestock production grows beyond the USA and Europe into developing nations (Thorne 2007), highly concentrated nutrient and fecal microbial pollution from these sources will similarly expand.

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ATTACHMENT 24

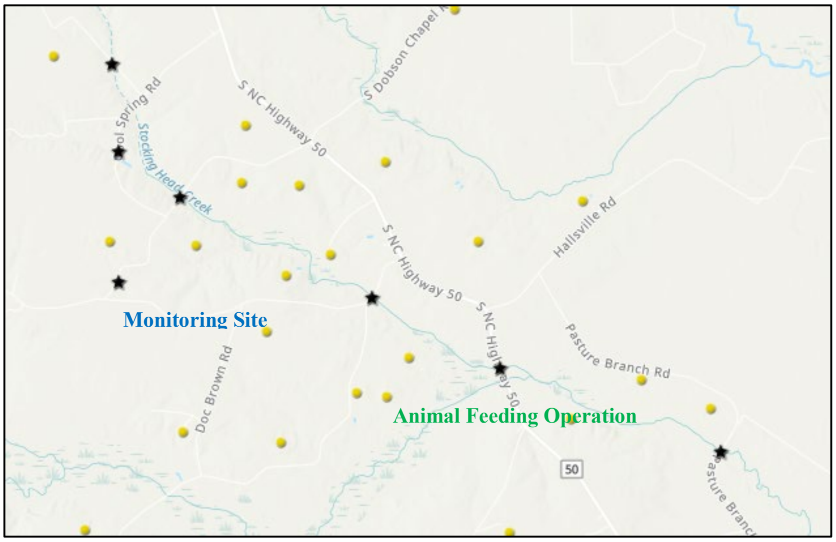
North Carolina Department of Environmental Quality

May 4, 2020

Title VI: Increasing Equity, Transparency, and Environmental Protection in the Permitting of Swine Operations in North Carolina

Attachment G: Cape Fear River Animal Feeding Operations Monitoring Study: Preliminary Report

NORTH CAROLINA
Department of Environmental Quality



CAPE FEAR RIVER ANIMAL FEEDING OPERATIONS MONITORING STUDY: PRELIMINARY REPORT

North Carolina Department of Environmental Quality

May 2020

Contents

Figures

Tables

Introduction

The Cape Fear River Animal Feeding Operations Monitoring Study (CFRAFOMS) is an ongoing surface water quality monitoring study that evaluates water quality in watersheds adjacent to high concentrations of permitted animal feeding operations utilizing lagoon and spray fields for waste management. This report evaluates the analytical data obtained from water samples collected from surface water quality monitoring stations. The data presented in this report is from eleven monitoring stations in Duplin and Pender Counties (see Figure 1 and Table 1) and represents monitoring results from April 2018 to October 2019. Ten of the eleven monitoring stations are test stations located in Duplin County in watersheds with high concentrations of animal feeding operations. The monitoring station in Pender County is a reference/background station and no registered animal feeding operations are present in the drainage area of this station.

The CFRAFOMS was conducted as part of a settlement agreement between North Carolina Department of Environmental Quality (NCDEQ) and multiple parties that include the North Carolina Environmental Justice Network (NCEJN), The Rural Empowerment Association for Community Help (REACH), and The Waterkeeper Alliance, Inc. The CFRAFOMS was intended to provide NCDEQ an opportunity to evaluate surface water conditions in areas with a high concentration of animal feeding operations, and if surface water impacts were found, then to evaluate potential sources. The study also helps the NCEJN, REACH, and Waterkeeper Alliance Inc. to evaluate the terms discussed in the settlement agreement (Settlement Agreement.pdf).

Program Background

Animal waste management systems in North Carolina (NC) are regulated by the Animal Feeding Operations (AFO program in the Department of Environmental Quality's. The AFO Program is responsible for issuing permits and enforcing compliance activities on animal feeding operation facilities across the state. Animal operations are defined by General Statute 143-215.10B as feedlots involving more than 250 swine, 100 confined cattle, 75 horses, 1,000 sheep, or 30,000 poultry with a liquid waste management system. NCDEQ AFO Program has some of the most stringent permit requirements for AFOs in the country and is one of the few states that requires annual inspections of every permitted facility. Permitting requirements for animal feeding operations in North Carolina can be found at:
/about/divisions/water-resources/water-resources-permits/wastewater-branch/animal-feeding-operation-permits/permits.

The majority of NC swine AFOs are covered by the N.C. Swine State General Permit. The general permit contains performance standards, operation and maintenance requirements, monitoring and reporting requirements, policy for inspections and entry to the farms, general conditions and the penalty assessment policy. A Certificate of Coverage (CoC) is issued with each permit that is permittee-specific and designates the permitted number of animals and type of animal operation. All permitted animal operations are required to have a Certified Animal Waste Management Plan (CAWMP) that has been developed by a Certified Technical Specialist. The CAWMP identifies the fields to which the waste is applied, the crops to be grown and other operational details of the waste management system. Animal waste must be applied at no greater than agronomic rates – an amount that can be used productively by the crops planted.

North Carolina contains approximately 2,100 permitted swine farms and is the nation's second highest producer of swine. The CFRAFOMS was conducted primarily in Duplin County where there are 483 permitted animal farming facilities, which accounts for approximately 23% of the state's permitted swine facilities.

Methods

Surface Water Quality Monitoring Study Plan

The CFRAFOMS was designed to investigate potential water quality impacts in highly concentrated areas of AFOs. Ten (10) water quality monitoring stations and one reference/background station were included in this study (Figure 1). General information on the selected monitoring stations including their location, stream index number, and watershed characteristics can be found in Table 1. The station locations were selected to provide a picture of the surface water quality adjacent to animal feeding operations.

The Stocking Head Creek watershed has 22,353 acres of land mass and is located in the Cape Fear River basin. Seven of the eleven water quality monitoring stations are located in this watershed. Murphey's Creek monitoring site is located in Rockfish Creek watershed which has 30,981 acres. Muddy creek monitoring site is located in the Muddy Creek watershed with 30,718 acres. Sikes Mill Run monitoring site is located in Six Runs Creek watershed with 14,548 acres. The background monitoring site is located in Harrisons Creek watershed located in Pender county with 23,433 acres.

Water quality parameters most commonly analyzed to investigate water quality impacts from AFOs are nutrients and pathogens. These parameters along with a suite of other physical and chemical water quality parameters were monitored on a monthly basis between April 2018 and October 2019. The full list of parameters and the sample type are listed in Table 2.

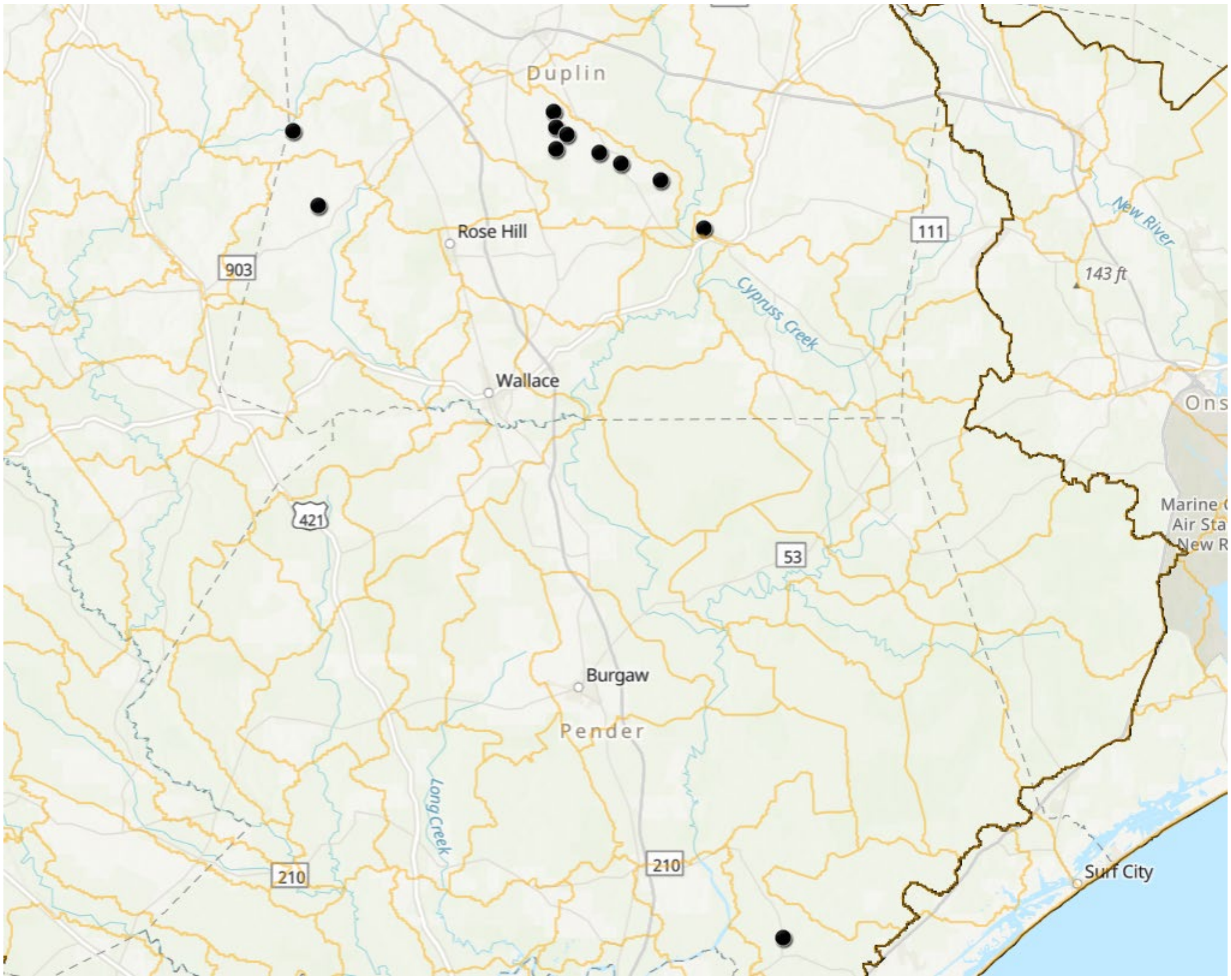


Figure 1. Map showing the watersheds and water quality monitoring sites.

Stream Name	Stream Index	Monitoring Location	Watershed Characteristics	County
Stocking Head Creek	18-74-24	Graham Dobson Road (SHC_GDR)	Several Crossroads throughout this high density CAFO watershed. This is a headwaters stream crossing.	Duplin
		Cool Spring Road (Unnamed Tributary) (TR_CSR)		
		S. Dobson Chapel Road (SHC_SDCR)		
		S. Dobson Chapel Road (Unnamed Tributary) (TR_SDCR)		
		Stocking Head Road (SHC_SHCR)		
		S NC Highway 50 (SHC_50)	Fish station at this crossroads.	
		Pasture Branch Road (SHC_PBR)	Benthic macroinvertebrate station at this crossroads.	
Murpheys Creek	18-74-29-0.5	Waycross Road (MC_WR)	High density CAFO watershed.	Duplin
Muddy Creek	18-74-25	Durwood Evan Road (MC_DER)	Medium density CAFO watershed. Impaired biological station at this location.	Duplin
Sikes Mill Run	18-68-2-10-4	Beasley Mill Road (SMR_BMR)	High density CAFO watershed with stream originating on hog farm.	Duplin
Harrisons Creek	18-74-49	Hwy 210 (HC_210)	Largely undeveloped watershed with some row crop and silviculture operations.	Pender

Table 1. Water quality monitoring locations with stream index number, watershed description and county name.

Parameter	Sample Type
Dissolved Oxygen (mg/L and percent saturation)	Surface
pH (SU)	Surface
Specific Conductance (µmhos/cm)	Surface
Temperature (°C)	Surface
Ammonia as N (NH_3) (mg/L)	Grab Sample
Nitrate+Nitrite (NO_3+NO_2) (mg/L)	Grab Sample
Total Kjeldahl Nitrogen (TKN) (mg/L)	Grab Sample
Total Phosporus (TP) (mg/L)	Grab Sample
Turbidity (mg/L)	Grab Sample
Fecal coliform (CFU/100 mL)	Grab Sample
Colored Dissolved Organic Matter (CDOM) (mg/L)	Grab sample

Table 2. Water quality analysis parameters and sampling type used for collecting and analyzing water samples.

Water samples were collected, stored, and transported from monitoring stations following approved monitoring standard operating procedures (SOPs) (AMS QAPP, 2017). Chemical analyses of all parameters except CDOM were conducted by the NC Water Sciences Section Chemistry Laboratory using EPA-approved methods (40CFR Part 136).

In April 2019, DWR began collecting samples for the stable isotope and excitation-emission matrix (EEM) fluorescence analysis. Samples were collected monthly for six months from all monitoring stations in accordance with the NC State University Osburn Biogeochemistry Laboratory (Osburn Lab) sampling protocol (CDOM Sampling SOP_DWR.pdf) and analyzed by the Osburn Lab.

NCDEQ also investigated any potential impacts that underground drain tiles located in fields receiving animal waste may have on water quality. Regional office field staff conducted intensive inspections on farms located in the study area to identify fields that had drain tiles (See Inspection Notes in Appendix 1).

Results

The mean and median concentrations for nutrient parameters and pathogens were calculated for the data between April 2018 and October 2019. The results are shown below in Figures 2-6 by parameter for each monitoring station.

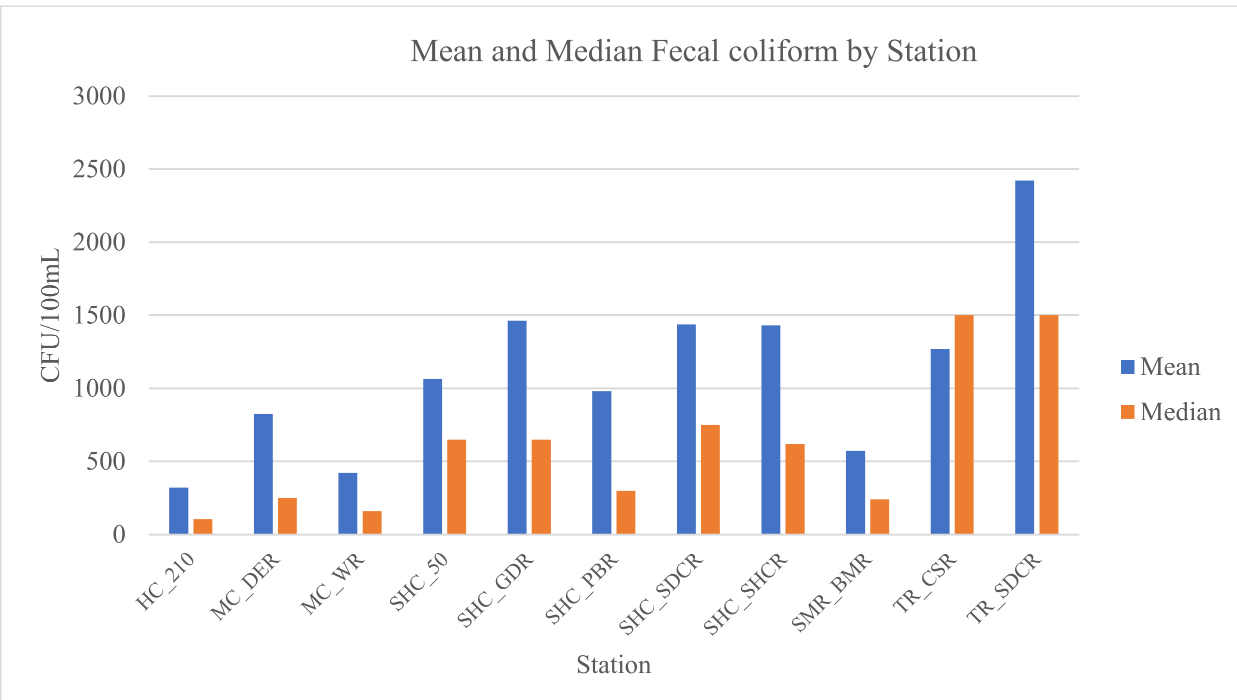


Figure 2. Mean and median concentrations for Fecal coliform (April 2018-October 2019).

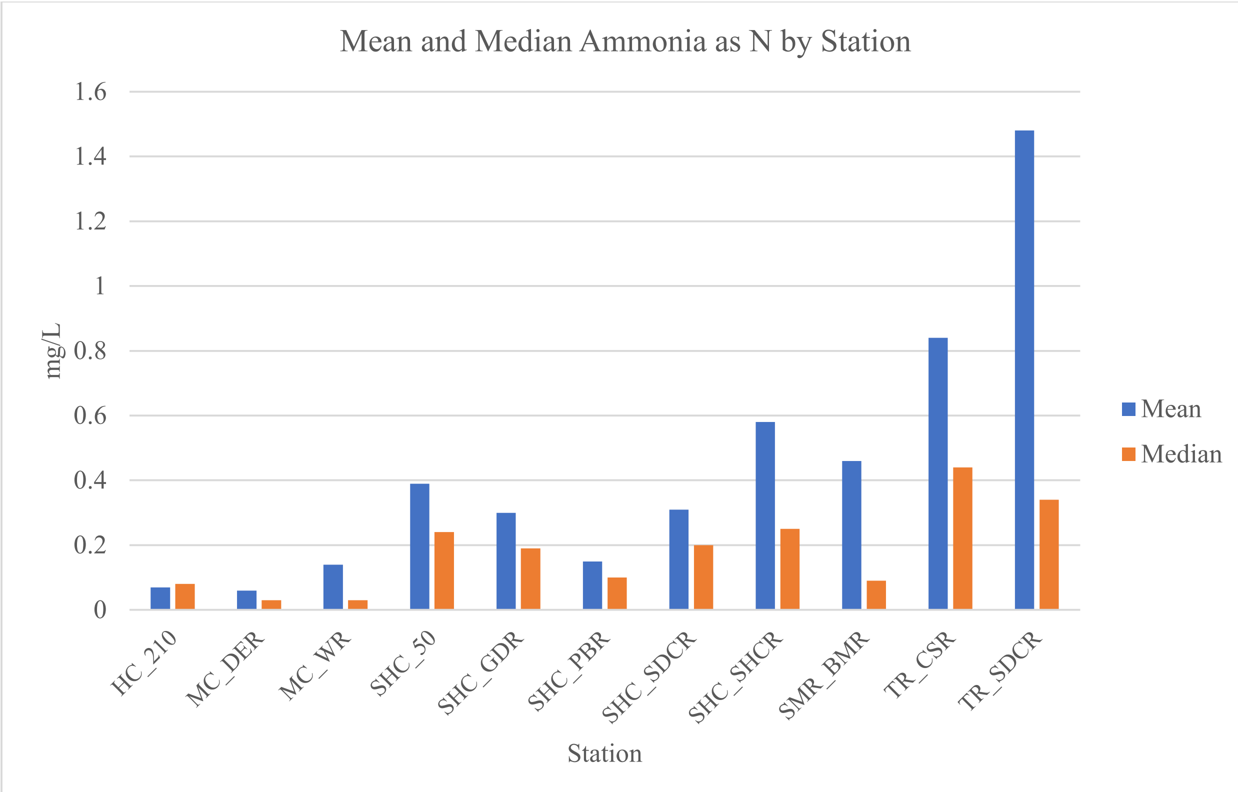


Figure 3. Mean and median concentrations for Ammonia as N (April 2018-October 2019).

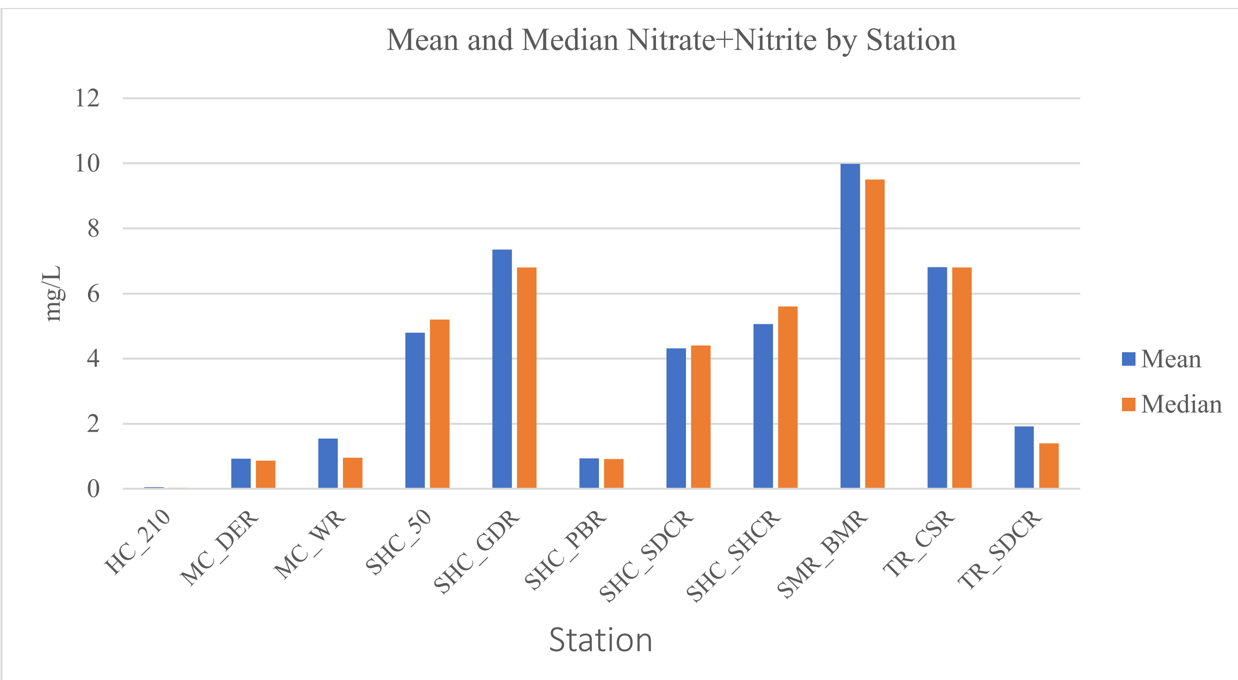


Figure 4. Mean and median concentrations for Nitrate+Nitrite (April 2018-October 2019).

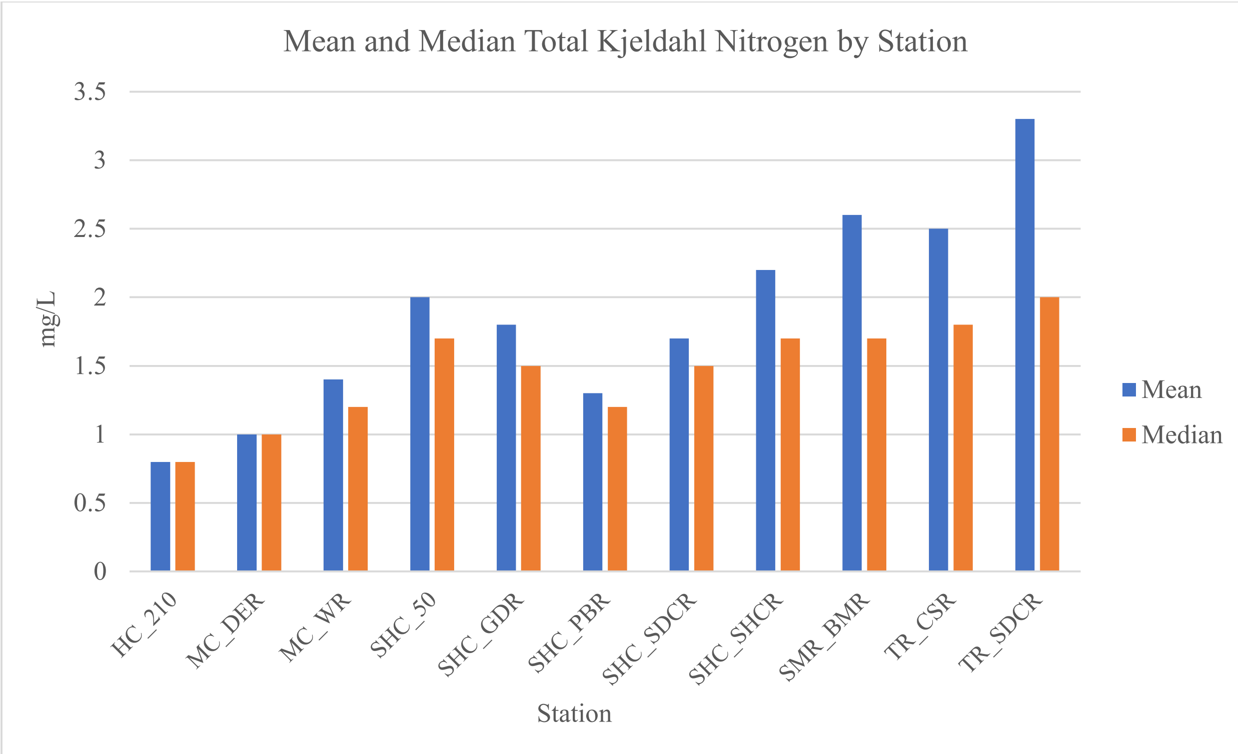


Figure 5. Mean and median concentrations for Total Kjeldahl Nitrogen (April 2018-October 2019).

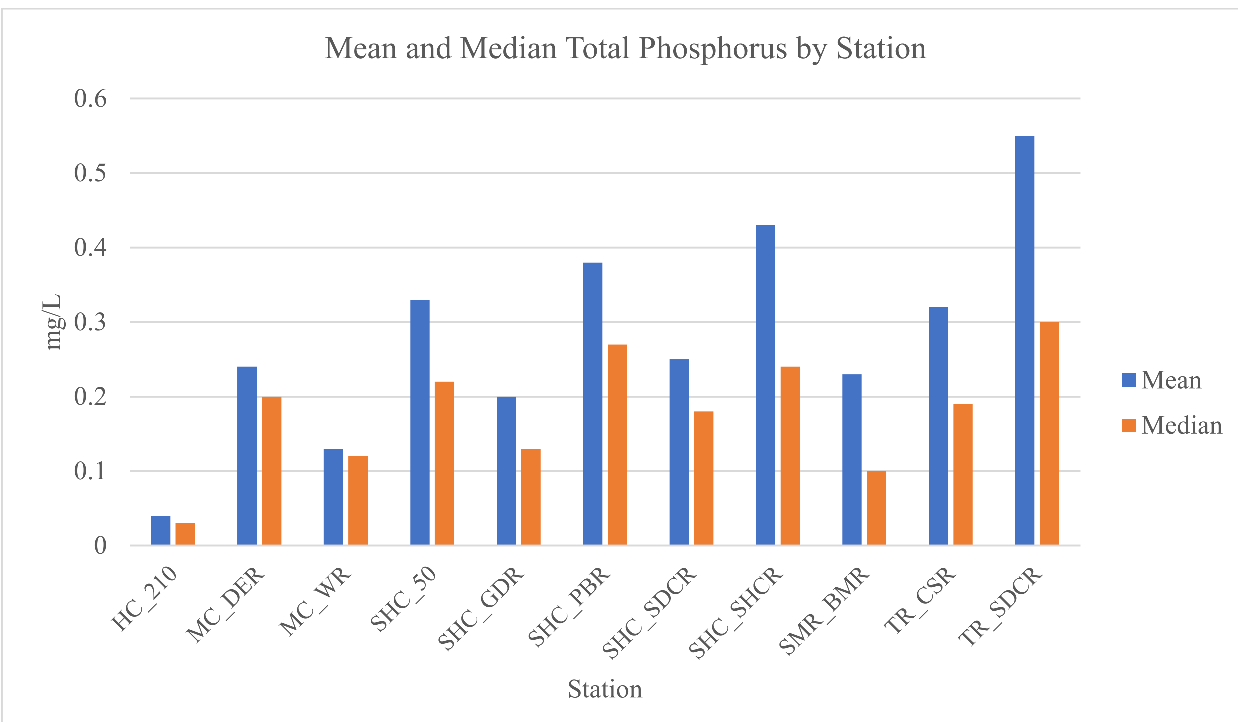


Figure 6. Mean and median concentrations for Total Phosphorus (April 2018-October 2019).

As shown in almost all cases in the tables above, the reference station (HC_210) had much lower mean and median concentrations by parameter than the test stations. To evaluate whether the parameter concentrations are significantly different in the test stations as compared to the reference stations, Kruskal-Wallis tests (2-sided) were conducted to compare the medians. The null hypothesis was that all medians are the same or not significantly different. Low p-values in the tables (highlighted in green) below indicate significant differences between the respective stations in the matrix.

	HC_210	MC_DER	MC_WR	SHC_50	SHC_GDR	SHC_PBR	SHC_SDCR	SHC_SHCR	SMR_BMR	TR_CSR
MC_DER	0.2309	-	-	-	-	-	-	-	-	-
MC_WR	0.847	0.3176	-	-	-	-	-	-	-	-
SHC_50	0.0255	0.2309	0.0683	-	-	-	-	-	-	-
SHC_GDR	0.019	0.3176	0.0728	0.9862	-	-	-	-	-	-
SHC_PBR	0.1575	0.6962	0.2735	0.5992	0.5992	-	-	-	-	-
SHC_SDCR	0.0121	0.0683	0.019	0.5992	0.6401	0.2811	-	-	-	-
SHC_SHCR	0.0172	0.1934	0.034	0.8029	0.7903	0.4845	0.7796	-	-	-
SMR_BMR	0.2087	0.8946	0.3464	0.3352	0.3352	0.7374	0.1266	0.2054	-	-
TR_CSR	0.0071	0.019	0.0121	0.3176	0.3176	0.1305	0.5554	0.5274	0.019	-
TR_SDCR	0.0222	0.2309	0.0641	0.5554	0.5992	0.3773	0.847	0.7903	0.166	0.8946

Table 3. Kruskal-Wallis tests results for Fecal coliform (p-value ≤ 0.05 indicates significant difference).

	HC_210	MC_DER	MC_WR	SHC_50	SHC_GDR	SHC_PBR	SHC_SDCR	SHC_SHCR	SMR_BMR	TR_CSR
MC_DER	0.32517	-	-	-	-	-	-	-	-	-
MC_WR	0.97248	0.69489	-	-	-	-	-	-	-	-
SHC_50	0.00023	0.00019	0.00679	-	-	-	-	-	-	-
SHC_GDR	0.00227	0.00103	0.0256	0.22747	-	-	-	-	-	-
SHC_PBR	0.17512	0.00438	0.1066	0.01689	0.1066	-	-	-	-	-
SHC_SDCR	0.00265	0.00045	0.01382	0.2438	0.97248	0.11461	-	-	-	-
SHC_SHCR	0.00018	0.00018	0.00454	0.45943	0.07447	0.00727	0.06768	-	-	-
SMR_BMR	0.1964	0.00587	0.0966	0.00587	0.05041	0.97248	0.06676	0.00435	-	-
TR_CSR	0.000057	0.000057	0.00046	0.00727	0.00142	0.00019	0.00436	0.04441	0.00088	-
TR_SDCR	0.00045	0.00034	0.00438	0.21572	0.0437	0.00587	0.06267	0.48405	0.00727	0.34634

Table 4. Kruskal-Wallis tests results for Ammonia as N (p-value ≤ 0.05 indicates significant difference).

	HC_210	MC_DER	MC_WR	SHC_50	SHC_GDR	SHC_PBR	SHC_SDCR	SHC_SHCR	SMR_BMR	TR_CSR
MC_DER	0.0000255	-	-	-	-	-	-	-	-	-
MC_WR	0.03362	0.92381	-	-	-	-	-	-	-	-
SHC_50	0.0000051	0.0000057	0.00038	-	-	-	-	-	-	-
SHC_GDR	0.0000051	0.0000051	0.0000091	0.00781	-	-	-	-	-	-
SHC_PBR	0.0000051	0.99049	1	0.0000051	0.0000051	-	-	-	-	-
SHC_SDCR	0.0000051	0.0000093	0.00025	0.60106	0.00123	0.0000083	-	-	-	-
SHC_SHCR	0.0000051	0.0000051	0.00014	0.69204	0.01154	0.0000051	0.26084	-	-	-
SMR_BMR	0.0000051	0.0000051	0.0000051	0.00022	0.05127	0.0000051	0.0000337	0.0004	-	-
TR_CSR	0.0000051	0.0000051	0.0000127	0.05437	0.71768	0.0000051	0.00401	0.12249	0.01816	-
TR_SDCR	0.0000051	0.03457	0.49163	0.00025	0.0000222	0.02251	0.00035	0.00021	0.0000091	0.000071

Table 5. Kruskal-Wallis tests results for Nitrate+Nitrite (p-value ≤ 0.05 indicates significant difference).

	HC_210	MC_DER	MC_WR	SHC_50	SHC_GDR	SHC_PBR	SHC_SDCR	SHC_SHCR	SMR_BMR	TR_CSR
MC_DER	0.38359	-	-	-	-	-	-	-	-	-
MC_WR	0.00036	0.00313	-	-	-	-	-	-	-	-
SHC_50	0.00017	0.00022	0.03036	-	-	-	-	-	-	-
SHC_GDR	0.00019	0.00036	0.08316	0.60385	-	-	-	-	-	-
SHC_PBR	0.00337	0.06051	0.28497	0.00407	0.01125	-	-	-	-	-
SHC_SDCR	0.0002	0.00039	0.17128	0.28027	0.87142	0.01776	-	-	-	-
SHC_SHCR	0.00017	0.00023	0.01737	0.96982	0.52844	0.00337	0.30209	-	-	-
SMR_BMR	0.00019	0.00039	0.08316	0.96982	0.77717	0.01466	0.60385	0.91308	-	-
TR_CSR	0.00017	0.0002	0.01116	0.72909	0.38359	0.00337	0.15221	0.79256	0.72909	-
TR_SDCR	0.00036	0.00138	0.08146	0.82468	0.60385	0.03363	0.39746	0.88527	0.88527	0.89921

Table 6. Kruskal-Wallis tests results for Total Kjeldahl Nitrogen (p-value ≤0.05 indicates significant difference).

	HC_210	MC_DER	MC_WR	SHC_50	SHC_GDR	SHC_PBR	SHC_SDCR	SHC_SHCR	SMR_BMR	TR_CSR
MC_DER	0.000053	-	-	-	-	-	-	-	-	-
MC_WR	0.000092	0.02403	-	-	-	-	-	-	-	-
SHC_50	0.000053	0.27671	0.00017	-	-	-	-	-	-	-
SHC_GDR	0.000053	0.31549	0.40585	0.0434	-	-	-	-	-	-
SHC_PBR	0.000053	0.13777	0.0003	0.59956	0.02403	-	-	-	-	-
SHC_SDCR	0.000053	0.98624	0.00933	0.15584	0.20016	0.15584	-	-	-	-
SHC_SHCR	0.000053	0.27671	0.00048	0.96254	0.03767	0.73428	0.2158	-	-	-
SMR_BMR	0.00044	0.14304	0.84618	0.02327	0.4406	0.02327	0.11305	0.02403	-	-
TR_CSR	0.000053	0.94835	0.04743	0.26193	0.34641	0.15584	0.84824	0.2862	0.15224	-
TR_SDCR	0.000053	0.23572	0.01076	0.61807	0.09569	0.90644	0.23572	0.73428	0.0555	0.2862

Table 7. Kruskal-Wallis tests results for Total Phosphorus (p-value ≤0.05 indicates significant difference).

For all parameters, a majority of the test stations (six or more) were significantly different from the reference station. For NO_3+NO_2, TKN, and TP, nine of more of the test stations were significantly different than the reference station.

Discussion

Based on the results of this study to date, it appears that nutrient and pathogen concentrations are higher for the test stations in the concentrated AFO areas as compared to the reference station with no AFOs in the drainage area. The next step for this study is to determine the source of the nutrients and pathogens. Source identification for ubiquitous parameters such as nutrients and pathogens can be difficult. However, technological advances have made the identification of organic nitrogen and specific pathogens sources much more reliable. Two specific enhanced analytical techniques will be used to assist in source identification, excitation-emission matrix (EEM) fluorescence analysis and quantitative polymerase chain reaction (qPCR) analysis

EEM

NCDEQ has pursued enhanced analysis of surface water samples for the purpose of nutrient source identification. Analytical techniques such as stable isotope and EEM fluorescence analyses were used to identify organic nitrogen sources in ambient waters. Organic nitrogen sources such as wastewater effluent, fertilizers, and animal wastes have different ranges of isotope ratios. These ratio ranges can act as fingerprints for sources of the nitrogen. Excitation-emission matrix fluorescence analysis can also be used to identify the fluorescent properties of dissolved organic nitrogen (Osburn et al. 2016). Organic nitrogen exhibits different fluorescence signatures depending on the source of the nitrogen. These signatures are modeled in a parallel factor analysis to identify sources such as wastewater, animal waste, and septage and the relative nitrogen contributions of the sources (Ibid.).

NCDEQ contracted with North Carolina State University's Osburn Lab to conduct stable isotope and EEM fluorescence analyses (FLUORMOD) for samples collected as part of this study. Dr. Osburn's FLUORMOD analysis was designed to analyze a variety of organic nitrogen sources such as septage, poultry, swine, and wetlands/soil. Swine was of particular interest to this study. However, results from the preliminary analyses detected only minimal organic nitrogen from swine sources. Tables 8-11 show the results from the analyses. Samples collected in June were lost in transport, so only four sets of samples (March, April, May and July) were analyzed. Further discussion with Dr. Osburn revealed that the FLUORMOD analysis used at that time could have been misidentifying the organic nitrogen sources. FLUORMOD was developed using swine waste sampled directly from a lagoon. FLUORMOD would be likely to detect this fluorescence signature in ambient waters only in the event of a direct discharge from a swine lagoon to surface waters. Current regulatory requirements for inspection and management make direct discharges from lagoons to surface waters unlikely except in catastrophic events. The more likely path of a discharge to surface waters is during spray irrigation of waste onto sprayfields due to overspray, ponding and runoff, or infiltration into groundwater or underground drain tile. Chemical changes that occur in waste during spray irrigation, infiltration into soil, and residence time in surface waters can significantly change the fluorescence signature (Osburn personal communication). Therefore, it is unlikely that FLUORMOD as designed would detect contributions of organic nitrogen from swine through these pathways.

Dr. Osburn is currently revising his analytical model to detect the fluorescence signatures from swine waste applied to sprayfields. Once the revisions are completed, NCDEQ will begin collecting samples concurrently with nutrient and pathogen samples from all eleven stations for analysis using FLUORMOD in an attempt to identify organic nitrogen sources in these surface waters.

March 2019	Reference	Poultry	Swine	Septic	Soil
HC_210	66%	1%	0%	0%	32%
MC_DER	73%	3%	0%	1%	24%
MC_WR	64%	2%	0%	0%	34%
SHC_50	76%	4%	0%	1%	20%
SHC_GDR	77%	4%	0%	1%	18%
SHC_PBR	72%	2%	0%	1%	25%
SHC_SDCR	73%	3%	0%	1%	23%
SHC_SHCR	76%	4%	0%	1%	20%
SMR_BMR	66%	2%	0%	0%	31%
TR_CSR	74%	5%	0%	1%	20%
TR_SDCR	73%	3%	0%	1%	24%

Table 8. FLUORMOD results for March 2019.

April 2019	Reference	Poultry	Swine	Septic	Soil
HC_210	71%	2%	0%	0%	27%
MC_DER	71%	2%	0%	1%	26%
MC_WR	78%	4%	0%	1%	17%
SHC_50	77%	4%	0%	1%	18%
SHC_GDR	73%	5%	0%	1%	21%
SHC_PBR	73%	2%	0%	1%	25%
SHC_SDCR	83%	4%	0%	5%	8%
SHC_SHCR	75%	4%	0%	1%	20%
SMR_BMR	61%	1%	0%	0%	38%
TR_CSR	63%	2%	0%	0%	35%
TR_SDCR	74%	3%	0%	1%	22%

Table 9. FLUORMOD results for April 2019.

May 2019	Reference	Poultry	Swine	Septic	Soil
HC_210	77%	3%	0%	1%	19%
MC_DER	76%	4%	0%	1%	19%
MC_WR	75%	5%	0%	1%	19%
SHC_50	76%	5%	0%	1%	18%
SHC_GDR	69%	4%	0%	1%	25%
SHC_PBR	70%	3%	0%	1%	27%
SHC_SDCR	74%	3%	0%	1%	22%
SHC_SHCR	74%	6%	0%	2%	18%
SMR_BMR	63%	2%	0%	0%	35%
TR_CSR	73%	13%	0%	4%	10%
TR_SDCR	41%	4%	2%	19%	34%

Table 10. FLUORMOD results for May 2019.

July 2019	Reference	Poultry	Swine	Septic	Soil
HC_210	75%	2%	0%	1%	22%
MC_DER	79%	4%	0%	4%	13%
MC_WR	70%	10%	0%	2%	18%
SHC_50	73%	7%	0%	2%	18%
SHC_GDR	65%	7%	0%	2%	26%
SHC_PBR	78%	4%	0%	1%	18%
SHC_SDCR	71%	6%	0%	2%	21%
SHC_SHCR	74%	7%	0%	2%	17%
SMR_BMR	67%	3%	0%	1%	29%
TR_CSR	55%	2%	0%	0%	42%
TR_SDCR	60%	8%	0%	12%	19%

Table 11. FLUORMOD results for July 2019.

Quantitative Polymerase Chain Reaction (qPCR)

qPCR is a genetic identification analysis often used to identify bacterial markers (Kralik and Ricchi, 2017). This highly sensitive analysis can identify down to specific genus and species of bacteria. This is useful in pathogen source identification in surface waters where bacteria found only in specific animals (e.g., swine, poultry, cattle, humans) can be selected for analysis as identifying markers (Ibid). NCDEQ is establishing collaboration with researchers at North Carolina universities who conduct this analysis to participate in the CFRAFOMS. Samples for this analysis will be collected concurrently with nutrient and pathogen samples.

Once the FLUORMOD model has been revised and the qPCR collaborator has been identified, sample collection will begin again. The explicit purpose of this sampling will be to attempt source identification using the target parameters organic nitrogen and pathogens. It is anticipated that the source identification monitoring will provide insight to NCDEQ on nutrient and pathogen sources in the Cape Fear River basin in areas populated with high concentrations of animal waste facilities.

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Appendix:1

The Department of Environmental Quality inspected 23 swine facilities as part of the Stocking Head Creek Watershed Study which could potentially impact surface water quality. During the inspections, NCDEQ looked for any unpermitted discharges coming from waste storage structures.

Notes from inspections conducted on the AFOs in the surrounding areas of water quality monitoring sites with corresponding subsurface drain tiles and average values for NH4-N and Fecal coliform.

Permit No.	Farm Name	Inspection Date	Notes	Nearest SHC Location	Average NH4-N (mg/L)	Average Fecal Coliform (CFU/100ml)	Subsurface Drains (Y/N)
AWS310466	Sands Farm	3/11/2019	DWR inspectors visited the farm on 3/11/2019. We rode and inspected the lagoons, fields and drainages. There are no subsurface drains on this farm and we did not see any areas of concern. This farm has no hogs or hog houses, and the representative on-site indicated that it has been approximately 2 years since they've land-applied wastewater. The facility is in the process of being converted to a truck wash.	MC_WR	0.19	436.46	No
AWS310445	Terry Miller Farm sites 1&2	3/14/2019	DWR inspected facility on 3/14/2019. Rode lagoon and viewed fields. No bad eroded areas. (Note: Farm was overtopped/inundated during Hurricane Florence)	SHC_PBR	0.16	1139.09	No
AWS310692	Liberty Farm	3/20/2019	DWR inspected facility on 3/20/2019. Lagoon and field was walked and some rode. No erosion or runoff issues. Subsurface drains are in field, currently waiting for a better map. Supposedly	SHC_SDCR	0.41	1727.27	Yes

			there are 2 laterals that go through the pivot pumping field and are run to the ditch on North side of the property. Farm didn't use correct Wa on some of the IRR2's and was missing calibration.				
AWS310386	**William Edward Brock Farm**	**3/20/2019**	**DWR inspected facility on 3/20/2019. Walked lagoons and fields on creekside of farm. No drain tiles in field that owner is aware. FB is noncompliant currently but POA is submitted. Discussed options and cost share "Pump and Haul". Instructed owner to communicate with DWR on FB. (Note: farm was inundated during hurricane Florence).**	**SHC_PBR**	**0.16**	**1139.09**	**No**
AWS310086	**ABS Family Farms, Inc.**	**3/26/2019**	**DWR on site 3/26/2019 to inspect farm for SHC study. Fields and records okay, fixing foundation cracks soon. Walked and rode fields looking for drains, found 1 in the ditch that leads to the pond, owner said there should be 2 more but could not find them.**	**SHC_SHCR**	**0.76**	**1979.09**	**Yes**
AWS310455	**Randy & Anna Harrell**	**3/26/2019**	**DWR inspected the facility on 3/26/2019 with a consultant. Numerous issues were discovered, including but not limited to, a lack of irrigation visible equipment (permitted for a solid set system), suspected equipment/pipe failure issues, suspected runoff from irrigation field, leaking from house and/or flush tank, multiple high and unreported FB events, no irrigation**	**SHC_50**	**0.49**	**1373.64**	**No**

			records for review, flooded irrigation field that is overgrown with trees/shrubs (has not been used), lack of suitable crop on the irrigation field that is used, etc. The consultant did not know if there were subsurface drains on the fields, and we did not locate any during the inspection. DWR will conduct a follow-up inspection with the farm owner or his son.				
AWS310035	**Waters Farm 1-5 M&M Rivenbark**	**3/11/2019**	**DWR inspectors visited the site on 3/11/2019. We rode and inspected the lagoons, fields, drainages and subsurface drains at the farms. We did not see any areas of concern other than one area in the center pivot field on the Waters 3,4,5 farm. There was a low spot where wastewater had the potential to pond and possibly runoff in the event of an over application event. DWR suggested that additional dirt be brought to address this area.**	**MC_WR**	**0.19**	**436.46**	**Yes**

AWS310160	**Carter and Sons Hog Farm 1&2**	**3/7/2019**	**David Powell inspected the farm on 3/7/2019. A pile of mulch was placed near a UT of SHC. Eroded areas from some fields had straw/hay around and in them to reduce erosion. Lagoons out of compliance currently with POA submitted. Amended POA coming to account for new lagoon levels. Summary of Findings: 1) Discharge to UT of SHC of <1000 gals leaving back of houses/piping, then running between lagoons and across small field to UT. Onsite observations show green grass in area of runoff and a drainage "swath", from stormwater mixed with nutrients from around lagoon and houses, have been doing this for a while 2) Mulch/hay bales in eroded areas. This can add nutrients into water of UT. Please replace soil in eroded areas and remove hay/mulch. Crop should be removed from fields irrigated on and disposed of properly. 3) Lagoon levels not in compliance; POA submitted and notification received 4) Fix leak at back of houses and eroded areas around farm. Replace soil, grass and reduce erosion. Replace also on dike walls and have markers reshot. Keep documentation. 5) Crop needs improvement. The fields are wet and are grazed. DWR**	**SHC_SDCR**	**0.41**	**1727.27**	**No**

			suggested having additional acreage for wet winters and additional pumping needs. Samples/pics taken. Sample 1 at 11:50 am; Sample 2 at 11:55 am; Sample 3 at 12:00; Sample 4 at 12:20 pm; Sample 5 at 1:40 pm; Sample at Envirochem at 3:05 pm. Suggest fixing stormwater runoff areas around lagoon/houses.				
AWS310321	**James E. King Farm**	**3/7/2019**	**DWR Michael Meilinger and Robb Marris visited farm on 3/7/2019 in response to SHC study, rode lagoons and walked fields, drain tiles in two fields that lead to SHC, marked on map. Some erosion from storm, fix spots and re-plant or re-seed field that was flooded from storm. Farm has cows and cow paths leading from the corrals and fields lead towards the ditches, told farmer to get grass cover and improve grass cover on dike wall. While riding the lagoon's saw signs of wild hogs rooting around the toe of the dike and in the edge of the woods next to the creek.**	**SHC_SHCR**		**1979.09**	**Yes**
AWS310451	**Otis Brown Farm**	**3/4/2019**	**DWR visited the farm as part of the SHC study. Rode roads along ditches and along field edges. Subsurface drains are known to be on the farm and marked on**	**SHC_SHCR**	**0.76**	**1979.09**	**Yes**

			the map. Water flowing out of the pipe into the ditch beside the road was a little dirty from all of the rain water. Farm and records look good, owner lost wheat crop after/during Hurricane in fields 1-4 has cover crop on fields now.				
AWS310254	**Bobby Brown Farm**	**3/4/2019**	**DWR visited farm in response to SHC study, farm had severe erosion after storm, fields have been fixed, farm has drain tiles and marked on map. Re-planting fescue where fields where flooded from hurricane Florence. Drain tiles that we say were flowing clear water and the ditches/creek was clear and flowing.**	**SHC_SHCR**	**0.76**	**1979.09**	**Yes**
AWS310371	**James P. Brown Farm**	**3/4/2019**	**DWR inspected the farm, farm looks well maintained, no drain tiles were found in fields, farm and records look good. Improve grass cover on dike wall.**	**SHC_SHCR**	**0.76**	**1979.09**	**No**
AWS310239	**Melvin Bostic Farm**	**3/7/2019**	**DWR Michael Meilinger visited the farm on 3/7/2019 in response to SHC study, rode lagoon's and farm fields, looked at the outfall of drain tiles, drain tiles marked on the map, farm and records look good**	**SHC_SDCR**	**0.41**	**1727.27**	**Yes**
AWS310017	**DM Farms Sec 2 Sites 1-4**	**2/28/2019**	**DWR inspectors visited farm on 2/28/2019, rode farm fields and looked at ditches and field edges that border Murpheys Creek. All water in the ditches and creek appeared to be clean. No drain tiles were found on the farm, and farm**	**MC_WR**	**0.19**	**436.46**	**No**

			looks well maintained.				
AWS310476	Greg Brown 1&2	2/11/2019	Soil analysis due 2019. Noticed few subsurface drains, water that we saw coming out of the drains and in the ditches was clear. DWR road farm, fix erosion spots in fields, fill in holes. When you resume pumping monitor to make sure nothing is running off from eroded areas. Farm looks well maintained.	SHC_SHCR	0.76	1979.09	Yes
AWS310077	Circle K I and II	2/12/2019	DWR onsite for compliance inspection and to survey streams, ditches, fields in support of SHC study. Numerous subsurface drains were documented, the drains observed were flowing clean/clear water. DWR requested the permittee to provide a map of drains located in the irrigation fields. Severe erosion along creek on south side of pivot 4 field. Permittee continues working on erosion from hurricane Florence.	SHC_GDR	0.36	1654.55	Yes
AWI310082	Vestal 1 and 2	2/12/2019	Visited for SHC study. DWR looked for erosion in fields, around lagoons and houses. Hurricane Florence has caused some areas to erode which are still being or need to be fixed when fields allow. Farm looks properly maintained. DWR requested the permittee to provide a map of drains located in the irrigation fields. Numerous subsurface	SHC_GDR	0.36	1654.55	Yes

			drains were documented, the drains observed were flowing clean/clear water.				
AWI310015	**Magnolia III, DM Section 4 Sites 1-4, Section 3 Sites 4-5**	**2/21/2019**	**DWR inspectors visited farm on 2/21/2019, rode farm fields and looked at ditches and field edges that border Sikes Mill Run. All water in the ditches and creek appeared to be clean. No drain tiles were found on the farm, and farm looks well maintained.**	**SMR_BMR**	**0.66**	**569.46**	**No**
AWS310048	**Stocking Head Creek Farm**	**2/11/2019**	**DWR walked the fields, looked for erosion and few subsurface drains were found coming from the fields, documented on the overview farm map that is in Laserfiche, water coming out of the drains was clear. Farm looks well maintained. Have been working on fixing eroded areas from Hurricane Florence. Soil analysis due 2019.**	**SHC_GDR**	**0.36**	**1654.55**	**Yes**
AWS310407	**JBJ Kilpatrick Farms Inc**	**2/6/2019**	**DWR rode farm and completed annual inspection, farm looks well maintained. DWR walked fields and creek ditches. Found few subsurface drain leading to ditch from irrigation field. Ditches/drain looked clean. Farm looks well maintained.**	**SHC_SHCR**	**0.76**	**1979.09**	**Yes**
AWS310725	**Kilpatrick Farms Inc**	**2/6/2019**	**DWR inspected the farm. Ditches and fields were walked and evaluated to find any eroding or bad areas. Farm has numerous subsurface drains. Need to fix eroded areas along irrigation fields. Farm looks good cover exposed PV**	**SHC_SHCR**	**0.76**	**1979.09**	**Yes**

			pipe and suggest pilings placed around it to ensure it doesn't get hit by a tractor.				
AWS310812	**Bowles and Sons Farm #3**	**2/7/2019**	**DWR visited the farm as part of the SHC study. Walked ditches along field edges. No subsurface drains are known to be on the farm, none seen. Permittee was recommended to (1) work on any areas in the fields/field edges that have eroded from Hurricane Florence; (2) Owner has removed cows from the farm on 2/18/2019. Check backs of houses for any possible leaks (grass is very green).**	**TR_SDCR**	**2.03**	**2783.09**	**No**
AWS310152	**Bowles & Sons Farm Inc Farm 2**	**2/7/2019**	**DWR walked fields, looking for erosion issues, 4 areas of severe erosion located on the left field that flow to SHC. Few subsurface drains were found in the application field and water coming out of the drain was clear. Permittee is waiting on approval for removing cows & fixing drainage issues. Storm water drain tiles on other side of ditch coming from neighbors fields flow into the UT beside the farm which flows to SHC. Check back of houses for leaks.**	**SHC_SDCR**	**0.41**	**1727.27**	**Yes**

ATTACHMENT 25

LIVESTOCK MANURE NUTRIENT ASSESSMENT IN NORTH CAROLINA

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LIVESTOCK MANURE NUTRIENT ASSESSMENT IN NORTH CAROLINA

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Abstract

North Carolina is one of the leading US states in livestock and poultry production. In 1993, the on-farm inventory was approximately 4.4 million animal units, as defined by the U.S. Environmental Protection Agency. Current trends toward farm consolidation, specialization and intensification are sound from an economic and management perspective, but may fail to adequately address important environmental impacts.

A nutrient assessment was initiated to determine where clusters of livestock and poultry are located, to assess manure generation by county, to determine the recoverable manure nutrients which can be made available for plant growth, to determine the quantity of nutrients required for agronomic crops and forages in each county and to determine the balance in each county between plant available manure nutrients and agronomic crop needs.

Approximatelv 27 million tons of fresh manure containing 205,000 tons of nitrogen, 138,000 tons of phosphorus (P205), and 133,000 tons of potassium (K2O) were generated in 1993. About 57% of the total manure was considered collectable. After storage, treatment and field losses, about 19% of the fresh manure nitrogen, 37% of the phosphorus, and 29% of the potassium were considered plant available as fertilizer nutrients. Statewide, about 20% of the nitrogen and 66% of the phosphorus could be met with animal manure. Three counties had enough manure to exceed their nitrogen requirements while 18 counties had more than enough phosphorus.

Kevwords: Livestock, Manure, Nutrient, Distribution, Balance

Introduction and Objectives

North Carolina is one of the leading US states in livestock and poultry production. In 1993, approximately 4.4 million animal units, as defined by the U.S. Environmental Protection Agency, were counted as on-farm inventories. Current trends are toward production farm consolidation, specialization and intensification. These efforts, while being sound from an economic and management perspective, must also consider the full environmental impact which can ensue from the increased generation of unevenly distributed animal manure. Processing by-products and animal mortalities also cont.ibute to the nutrient load to be managed, however, this project only deals with production farm generated manure nutrients.

All animal by-products contain organics and nutrients. Manure organics can provide a fuel source, while the nutrients, if managed properly, can be used as a feed supplement or as a plant fertilizer. "Best Management Practices" (BMPs) for manure used as a feed or fertilizer include manure collection, treatment, storage, and nutrient/feed analysis. Additional BMPs are required when manure is used as a fertilizer. These practices include proper timing, rate, and application uniformity in relation to the nutrient needs of the growing plants. In addition, soil conservation practices to reduce the soil, manure, and nutrient movement off site are needed.

The nutrient assessment project was initiated to: 1) geographically depict where the livestock are located and identify "clustering effects", i.e., high densities of livestock production around support facilities such as feed mills, hatcheries, processing plants, etc.; 2) assess current generation of manure by county; 3) determine the amount of nutrients from manure which can be recovered and made available to agronomic crops; 4) determine the quantity of nutrients required for non-legume agronomic crops and forages in each county; and 5) calculate the percent of agronomic crop and forage nutrients which can be supplied by animal manure.

ObViously, to proceed with a nutrient assessment, many assumptions must be made regarding production methods, manure handling systems, application techniques, crops, and nutrient needs. Most assumptions have been made on a statewide basis, although it is recognized that they will change somewhat county by county. The information is presented to provide a methodology for an animal manure nutrient assessment and to get a first glance as to the carrying capacities of localized areas within the state.

Methods and Assumptions

Manure Characterization

Data on manure, litter, wastewater, and sludge quantities and characteristics are necessary, to assist in the planning, design, and operation of manure collection, storage, pretreatment, and utilization systems for livestock and poultry enterprises. Databases have been developed over the past 12 years containing summaries from a wide base of published and unpublished information on livestock manure production and characterization (Barker et al., 1995). These summaries represent typical domestic food animal and poultry species as well as different farm manure management systems within species. Fresh manure values represent "as voided" feces and urine. Farm manure management systems include the following:

1. paved lot manure scraped within two days either directly into a manure spreader for field spreading or into a short-term storage;
2. annual accumulations of poultry manure with wood shavings or peanut hulls as a litter base;
3. liquid slurry accumulating for 6-12 months from manure, excess water usage, and storage surface rainfall surplus;
4. surface liquid from an anaerob~c treatment lagoon; and
5. bottom sludge from an anaerobic treatment lagoon.

Actual values vary due to differences in animal diet, age, usage, productivity, management and location. Whenever site-specific data are available or actual sample analyses can be performed, such information should be considered in lieu of the mean summarized values.

Collectable Manure

All animals are not raised in confinement where their manure can be easily collected for reuse. Cattle and sheep spend most of their time on pasture. Some hog enterprises consist of dirt or pasture lots. A small percentage of turkeys are still finished on open range. While these unconfined animals still contribute manure nutrients to the pasture system, these nutrients are not addressed in this assessment.

Nutrients Remaining After Storage and Treatment Losses

During the time between manure voiding by the animal and transport to the field for spreading, much of the nutrients can be lost through drying or dilution, surface runoff, volatilization, or microbial digestion. Since different manure management systems either conserve or sacrifice varying amounts of nutrients, an estimate must be made of the percentage of farms using specific systems. Applying these percentages to the manure characteristics appropriate to the specific method gives the remaining nutrients after storage and treatment losses.

Plant Available Nutrients

Estimates of nutrient availability coefficients for various manure management systems and application methods are summarized in Table 1. The plant-available portion of the manure nitrogen was determined by combining a percentage of the ammonia (ammonium) -nitrogen using the appropriate ammonia volatilization value based on the application method with one-half of the organic nitrozen assumed to be plant available during the same year of application. Availability of all other nutrients were based on the application method.

Table 1. Manure Nutrient Plant Availability Coefficients

Crop Nutrient Requirements

Crop nutrient requirements were determined for all agronomic crops (barley, oats, wheat, corn (grain), corn silage, sorghum (grain), cotton, soybeans and burley tobacco; and forages (hay and pasture). Horticultural and silvicultural crops, most legumes and flue- cured tobacco were not included. Nitrogen recommendations for all crops were calculated by using suggested nitrogen fertilizer rates based on realistic yield expectations (Zublena et al., 1994). Crop and hay yields were based on averaging the highest 2 years of the last 5 years according to the 1993 North Carolina Agricultural Statistics (NCDA, 1989-94a) Grass pasture data was obtained from the 1992 Census of Agnculture (USDC, 1994). No nitrogen credit was given to soybeans since they are legumes. No nitrogen credit was given to flue-cured tobacco since its N needs are relatively low and it is very sensitive to N. Burley tobacco has a high nitrogen requirement and was included.

When calculating application rates for phosphorus, specific Soil Test Index values of 10, 40, 70 and 110 from "Crop Fertilization Bases on N.C. Soil Tests" (NCDA, 1987), were used in the computations. These values coincide with the Soil Test Index ranges of 0-25, 26-50, 51-100, and 100+, respectively.

Copper and zinc nutrient recommendations are based on a mineral soil classification. When representative soil samples fall within the range of 0-25 on the Soil Test Index, copper and zinc application rates of 2.2 kg/ha and 6.7 kg/ha were suggested, respectively (NCDA, 1987).

County Nutrient Balances

All county crop acreages were based on "Acres Harvested" from the North Carolina Agricultural Statistics (NCDA, 1994a). County soil test data was obtained from the North Carolina Soil Test Summary (NCDA, 1994b). Organic and inorganic sources of nutrients other than animal manure were not included in this assessment because individual county sources of information for those nutrients were not considered reliable. It would be important in doina a complete nutrient balance to consider all sources of nutrients as well as all crop needs (agronomic, horticultural, silvicultural) in a given watershed.

RESULTS AND DISCUSSION

Manure Characterization

Table 2 gives the mean values of livestock and poultry manure, litter, wastewater and sludge amounts and nutrient concentrations for several animal species and manure management methods. Fresh manure values are mostly taken from published literature sources and with few exceptions tend to be from the 1960s and 70s. Very little new research has been done recently on quantifying and characterizing fresh livestock manures. The remaining databases are more recent and represent measured parameter concentrations. Values were obtained primarily from state analytical laboratories, university research labs, and published literature sources. Primary nutrients have substantial numbers of samples averaged ranging to 1400 for some parameters. Secondary and micronutrients have smaller numbers of values. The databases for manure volumes are rather sparse and are highly dependent on production systems, manure storage and treatment, and climatic factors. Extreme values outside the range of three standard deviations from the mean of the raw data set were not included in the averages in Table 2. The complete databases provide the numbers of samples for each parameter plus the range of values, mean, and standard deviation.

Livestock and poultry on-farm inventories as of December 1993 were obtained from the *North Carolina Agricultural Statistics* (NCDA, 1994). Animal production in North Carolina produced approximately 27 million tons of fresh manure in 1993 (Table 3). These manures contained 205,000 tons of nitrogen, 138,000 tons of phosphorus (P205), 133,000 tons of potassium (K20), 1,700 tons of zinc, and 290 tons of copper.

Collectable Manure

In North Carolina most ruminants are on pasture, while most of the hogs and poultry are fed in confinement where the manure is collected. Table 4 lists the estimated percent of time that animals of various ages and usages are in confinement. From Table 3 approximately 57% of the total fresh livestock and poultry manure is considered collectable.

Nutrients Remaining After Storage and Treatment Losses

Table 4 gives estimates on the percent of farms using either a lot scrape manure collection method, a liquid manure slurry handling and storage method, or an anaerobic treatment lagoon. Using the appropriate manure characterization databases, approximately 35% of the nitrogen, 50% of the phosphorus, 40% of the potassium, 47% of the zinc, and 71% of the copper were available for land application after handling, storage and treatment losses (Table 3).

Plant Available Nutrients

All lot scraped manure, liquid manure slurry, and lagoon sludge were assumed to be soil incorporated within 48 hours of application. Dry poultry litter was assumed to be equally divided between surface broadcast and soil incorporation. All lagoon liquid was assumed to be irrigated without soil incorporation. From Table 4, a 50% mineralization rate for organic nitrogen was used for all manures. The appropriate plant nutrient availability coefficient corresponding to the application method was chosen from Table 1. Approximately 19% of the fresh manure nitrogen, 37% of the phosphorus, 29% of the potassium, 34% of the zinc, and 60% of the copper were determined to be plant available. This illustrates the transient nature of manure nutrients and how they can be affected by collection, handling, storage, treatment and application method.

Crop Nutrient Requirements

When considering on a state-wide basis the nutrient requirements of all non-legume agronomic crops and forages and the plant available nutrients from livestock and poultry manure, about 20% of the nitrogen, 66% of the phosphorus, 85% of the zinc, and 42% of the copper needs can be met with animal manures.

County Nutrient Balances

Figures 1 and 2 present geographically the nutrient distribution and balances by county for nitrogen and phosphorus, respectively. From Table 5, at the county level in 1993, 3 counties (3% of state) had enough animal manure to exceed the nitrogen requirements of all non-legume agronomic crops and forages, while 55 counties (55% of state) could only supply less than 10% of their nitrogen needs with animal manure. However, 18 counties (18% of state) could exceed their phosphorus needs with animal and poultry manure, while 29 counties could supply less than 10% of their needs.

Figure 1. Percent of non-legume agronomic crop and forage nitrogen needs supplied by recoverable plant available manure nutrients.

Figure 2. Percent of non-legume agronomic crop and forage phosphorus needs supplied by recoverable plant available manure nutrients.

Table 5. Summary of North Carolina County Livestock Manure Plant Available Nutrient Balances with Agronomic and Forage Crop Needs.

SUMMARY

Information derived from this livestock manure nutrient assessment project can be geographically depicted to serve as a tool for:

1. determining where the livestock are located and identifying "clustering effects";
2. evaluating the quantities of manure nutrients available for plant gro~vth and how they may supplement the inorganic nutrient sources in a given area;
3. assessing the potential for nutrient impairment of water resources; and

4. providing a decision-making guide for future county or area-wide growth and development.

Nutrient assessments can be used for layers in a geographic information system (GIS). Obviously, large volumes of data must be manipulated and computerization is a must.

This assessment is being used by the North Carolina Cooperative Extension Service to focus and network educational efforts on animal manure management where there is the greatest need. Discussions have been initiated with livestock and poultry integrators on the need to consider dispersing livestock operations to prevent nutrient "saturation" in localized areas. This excess nutrient load might be sources of water impairments if they exceed the crop nutrient needs of the area. The information is shared with county commissioners, planners, and commodity advisory boards.

Meetings have also been conducted with representatives of the inorganic fertilizer industry to explore opportunities for incorporating organic sources into existing fertilizer operations.

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LIVESTOCK MANURE NUTRIENT ASSESSMENT IN NORTH CAROLINA

INCLUDED

- Beef
- Dairy
- Swine
- Horse
- Layer
- Broiler
- Turkey
- Animal time in confinement
- Scraped lot semi-solid manure
- Urine
- Litter
- Liquid manure slurry
- Wastewater
- Anaerobic lagoon liquid
- Anaerobic lagoon sludge
- Plant available nutrients
- Barley, grain
- Oats, grain
- Wheat, grain
- Corn, grain
- Corn, silage
- Sorghum, grain
- Soybeans
- Tobacco, burley
- Hay
- Pasture

NOT INCLUDED

- Aquaculture
- Exotics
- Sheep
- Goat
- Veal
- Animal time on pasture/drylot
- Animal mortalities
- Feedlot runoff
- Processing by-products
- Nutrients biodegraded, scavenged, immobilized, volatilized, lost
- Horticultural crops
- Legumes
- Peanuts
- Soybeans, (N)
- Silvicultural crops
- Tobacco, flue-cured

ATTACHMENT 26

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Characterization of atmospheric ammonia emissions from swine waste storage and treatment lagoons

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Abstract. Fluxes of atmospheric ammonia-nitrogen (NH_3-N, where NH_3-N = (14/17)NH_3) from an anaerobic ~2.5 ha (1 ha = 10,000 m^2) commercial hog waste storage lagoon were measured during the summer of 1997 through the spring of 1998 in order to study the seasonal variability in emissions of NH_3-N and its relationship to lagoon physicochemical properties. Ammonia-nitrogen fluxes were measured during each season (summer, fall, winter, and spring) using a dynamic flow through chamber system. Measured lagoon physicochemical parameters included surface lagoon temperature (T_ℓ°C, ~15 cm below surface), lagoon pH, and Total Kjeldahl Nitrogen (TKN). The pH and TKN of the surface lagoon water ranged from 7 to 8 pH units, and 500 to 750 mg N L^{-1}, respectively. The largest fluxes were observed during the summer (August 1997) (mean NH_3-N flux = 4017 ± 987 μg N m^{-2} min^{-1}). Fluxes decreased through the fall (December 1997) months (844 ± 401 μg N m^{-2} min^{-1}) to a minimum flux during the winter (February 1998) months (305 ± 154 μg N m^{-2} min^{-1}). Emission rates increased during spring (May 1998) (1706 ± 552 μg N m^{-2} min^{-1}), but did not reach the magnitude of fluxes observed during the summer. Lagoon emissions in eastern North Carolina were estimated to constitute ~33% of total NH_3-N emissions from commercial hog operations in North Carolina based on current inventories for NH_3-N emissions published by the North Carolina Division of Air Quality, North Carolina Department of Environment and Natural Resources. The ammonia flux may be predicted by an observational model $\log_{10}$ (NH_3-N flux) = 0.048 T_ℓ + 2.1.

1. Introduction

1.1. Background

Atmospheric ammonia (NH_3) emissions have garnered increased interest in the past few years, due in part to the detrimental effects of excess nitrogen deposition to nutrient-sensitive ecosystems [*Aneja et al.*, 1998a; *Asman et al.*, 1998; *Nihlgard*, 1985; *van Breemen et al.*, 1982]. Moreover, NH_3 is the primary gaseous base found in the atmosphere, and it is therefore fundamental in determining the overall acidity of precipitation [*Warneck*, 1988], cloudwater [*Li and Aneja*, 1992], and atmospheric aerosols [*Lefer et al.*, 1999]. Ammonia emissions contribute substantially to atmospheric nitrogen loading, and may contribute about the same order of magnitude as emissions of NO in some parts of the world [*Asman et al.*, 1998; *Steingröver and Boxman*, 1996]. The impact of atmospheric NH_3 deposition may be substantial as reduced nitrogen species are thought to be more biologically active than oxidized nitrogen species in coastal and estuarine ecosystems [*Paerl*, 1997]. In the atmosphere, NH_3 can react with acidic species to form ammonium sulfate, ammonium nitrate, or ammonium chloride, or it may be deposited to the Earth's surface. The spatial scale of a particular NH_3 source's contribution to atmospheric nitrogen deposition will be governed in part by the gas-to-particle conversion rate of NH_3 to NH_4^+. Because of the short lifetime of NH_3 in the atmosphere (τ = 1–5 days or less) [*Warneck*, 1988], low source height, and relatively high dry deposition velocity [*Asman and van Jaarsveld*, 1992] it may deposit near its source. However, ammonium (NH_4^+) aerosols with atmospheric lifetimes of the order of τ = 1–15 days [*Aneja et al.*, 1998b] will deposit at larger distances downwind of sources.

There are several environmental consequences associated with atmospheric NH_3 and its deposition; including particulate matter formation, soil acidification, aquatic eutrophication, and, near strong sources, odor emanation. In Great Britain and the Netherlands, which have dense spatial distributions of animal operations, soil acidification is a major environmental problem [*Aben and Dekkers*, 1996; *van Breemen et al.*, 1982]. *van Breemen et al.* [1982] identified deposition of ammonium sulfate as the major cause of soil acidification in the Netherlands because the oxidation of NH_4^+ via nitrifying bacteria releases 2 H^+ ions into soil. *Nihlgard* [1985] implicates NH_4^+ in Europe's forest decline, as nitrogen "oversaturated" trees succumb to wind, drought, and parasitic damage. Coastal Plain river systems in North Carolina have been under the influence of nutrient loading for several years [*Aneja et al.*, 1998c; *Paerl*, 1997, 1995]. Estimates suggest that atmospheric deposition may contribute 35–60% of total nitrogen loading to North Carolina coastal waters [*Paerl*, 1995]. The increase in nutrient loading over the past several years is related to agricultural management, human population growth, and increasing animal production. In July 1995 the appearance of the dinoflagellate Pfiesteria and its association with several large fish kills

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Table 1. Sources and Estimates of Nitrogen Emissions for North Carolina

Source[a]	Nitrogen Species Emitted[b]	Estimated Tons of Nitrogen Emitted per Year[c]	Percent of Total N[d]
Highway mobile (1990)	NO_x	78,509	23.7
Point sources (1994)	NO_x	77,798	23.6
Area and nonroad mobile	NO_x	24,452	7.4
Biogenic NO_x (1995)	NO_x	9,926	3.0
Swine (1995)	NH_3	68,540	20.6
Cattle (1995)	NH_3	24,952	7.5
Broilers (1995)	NH_3	13,669	4.1
Turkeys (1995)	NH_3	16,486	5.0
Fertilizer application (1999)	NH_3	8,270	2.5
"Other" chickens (1995)	NH_3	6,476	2.0
NH_3 point sources (1995)	NH_3	1,665	0.5
Total		330,743	100.0

Adapted from *Aneja et al.* [1998b].

[a]Nitrogen calculated from NO_x emissions assumes 100% NO_2 though the actual proportion is closer to 95%. Thus $NO_x - N$ (tons) × (14/46) and $NH_3 - N$ (tons) = NH_3 (tons) × 14/17).

[b]NO_x emission taken from Division of Air Quality (DAQ) inventories developed for modeling purposes; NH_3 emissions based upon factors presented by *Battye et al.* [1994]; and animal production statistics from the NCDA.

[c]Two minor ammonia sources, together totaling <6000 t statewide, have been omitted. Factors for these, emissions from sewage treatment plants and emission associated with human breathing, are based upon vary limited data and are currently being reevaluated.

[d]Relative proportions of NO_x = 58%, NH_x = 42%.

have resulted in efforts to reduce nitrogen loading into the Neuse River Basin [*Burkholder and Glasgow*, 1997]. A successful reduction strategy requires an accurate nitrogen budget for affected ecosystems and reliable source apportionment of nitrogen inputs to such systems. From an atmospheric standpoint, accurate emission factors for NH_3 sources, as well as measurement-based estimates of wet atmospheric deposition and dry deposition to various surface types, are essential. This study will address NH_3 emissions from swine waste lagoons, as this source is believed to contribute a substantial fraction of total NH_3 emissions in North Carolina [*Aneja et al.*, 1998b].

1.2. Ammonia Emissions

Ammonia is an important contributor to the atmospheric nitrogen budget; however, its sources and their emission strengths have received scant attention in the United States. The major global sources of ammonia include the decay of domestic livestock waste, volatilization losses from fertilizers, emissions from soils, and biomass burning. However, the largest contributor of ammonia to the global budget is domestic animal waste [*Bouwman et al.*, 1997; *Dentener and Crutzen*, 1994; *Schlesinger and Hartley*, 1992; *Warneck*, 1988; *Buijsman et al.*, 1987]. A preliminary nitrogen emission inventory for North Carolina (Table 1) suggests that ammonia emissions are primarily associated with livestock farming. Table 1 also reveals that swine operations contribute ~20% toward North Carolina's nitrogen emissions inventory and comprise ~47% of total NH_3 emissions in the state.

North Carolina has witnessed intense growth in its hog industry over the last decade (Figure 1). More than 90% of the states hog population resides in the Coastal Plain region [*Walker et al.*, 2000; *Walker*, 1998] where there is greater potential to directly impact coastal estuaries (Figure 2). The six most highly populated counties in this region have an average hog population density of ~528 hogs km^{-2} [*North Carolina Department of Agriculture* (*NCDA*), 1998] (Table 2), whereas the average hog population density for the remainder of the Coastal Plain region is ~65 hogs km^{-2} [*Walker et al.*, 2000]. Ammonia emissions from these six Coastal Plain counties account for approximately 36% of total statewide NH_3 emissions, with emissions from swine operations accounting for 77% of total NH_3 emissions within this six county region [*Chauhan*, 1999]. Using source-receptor modeling, *Walker et al.* [2000] have shown that under certain meteorological conditions NH_3 emissions from this six county area enhance wet deposition of NH_4^+/NH_3 at National Atmospheric Deposition Program/National Trend Network (NADP/NTN) sites up to 80 km away. The precipitation samples were collected by wet-only collectors (AeroChem Metrics Model 301 Wet/Dry Collector [*NADP/NTN*, 1998]). This distance is sufficient to allow for deposition to nitrogen-sensitive coastal and estuarine ecosystems. This illustrates the importance of quantifying NH_3 emissions from swine operations and properly relating these emission estimates to those factors which control emission rates. This information is necessary as inputs to atmospheric transport and deposition models such as the Regional Acid Deposition Model (RADM) currently being modified to accurately address the atmospheric transport and deposition of NH_x (=NH_3 + NH_4^+). Such models will help policy makers assess the impact and magnitude of atmospheric nitrogen deposition on local and regional scales.

Emission factors used thus far for generating preliminary North Carolina ammonia emission inventories for various animal husbandry operations are largely based on European work [*Battye et al.*, 1994]. The European factors must be verified or refined for conditions in North Carolina. An initial step in this process is the development of swine factors since total emissions from swine are greater than other animal categories in North Carolina. There are four principal sources of NH_3 emissions on a current commercial hog operation: hog production houses, waste storage and treatment lagoons, land application of lagoon slurry to adjacent cropland, and subsequent reemission of NH_3 from the soil. Lagoon NH_3 emissions arise

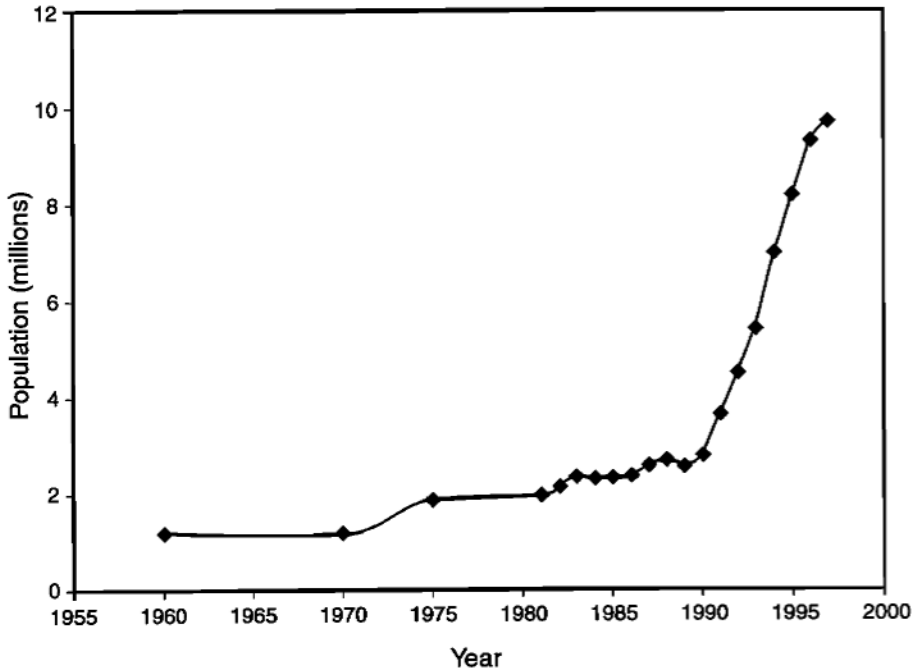

Figure 1. Trend in swine population in North Carolina, 1960–1997 (source of data: *NCDA* [1998]).

as urine and feces are flushed with water from the hog houses and discharged into the lagoon. Gas and liquid phase mass transfer processes are thought to be equally important in determining the overall desorption/absorption rate of ammonia [*Leuning et al.*, 1984]. As the slurry (~98% water [*Bicudo et al.*, 1999]) is stored in the lagoon, urea is hydrolyzed by the enzyme urease, present in feces, to produce NH_3 and carbon dioxide [*Aarnink et al.*, 1995; *Miller and Gardiner*, 1998].

The primary objective of this study was to measure the atmospheric flux of NH_3 from a swine waste storage and treatment lagoon during the four seasons at an intensively managed industrial hog operation in order to develop emission factors. The secondary objective was to parameterize the NH_3 flux process with respect to changes in lagoon physicochemical parameters (lagoon temperature, pH, and Total Kjeldahl Nitrogen (TKN)) for use in air quality models.

2. Methods and Materials

2.1. Sampling Site

Flux measurements were made at a farrow to finish commercial hog operation in Sampson County, North Carolina. The farm consisted of 13 hog production houses housing approximately 10,000 animals: 1212 sows and boars (average weight ~181 kg each), 7480 finishers (~61 kg each), and ~1410 suckling pigs (~11 kg each) (R. B. McCulloch, Division of Air Quality, North Carolina Department of Environment and Natural Resources, personal communication, 1998). The

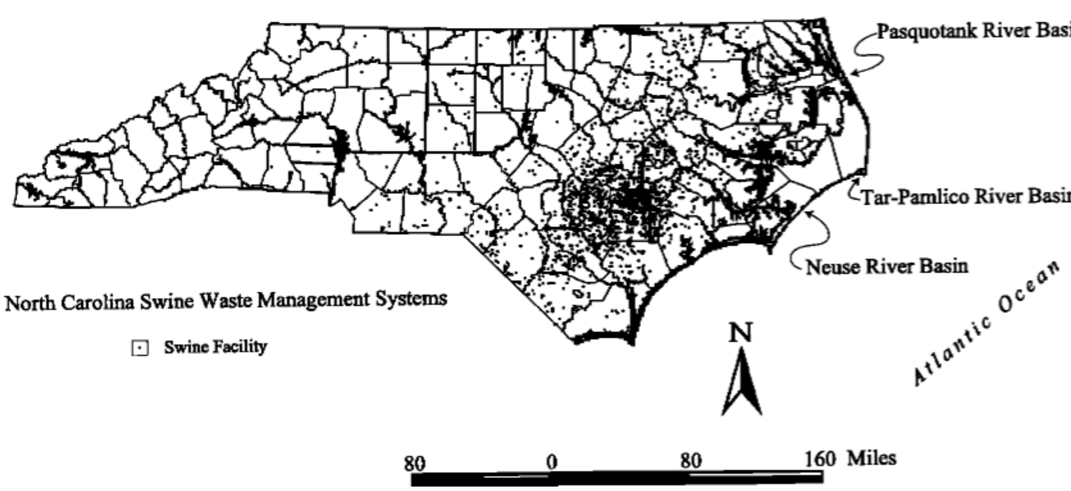

Figure 2. North Carolina swine waste management systems. Confined feedlots registered with NCDENR as required by 15A NCAC2H.0217 rule for waste not discharged to surface waters.

Table 2. Top Six North Carolina County Hog Population Densities

County	Hogs, km^{-2}
Duplin	1040
Sampson	752
Greene	560
Wayne	396
Bladen	309
Lenoir	304

Swine population statistics provided by the *North Carolina Department of Agriculture (NCDA)* [1998] for animal populations as of 1997.

waste from the hog sheds (urine and feces) was flushed out with recycled lagoon water and discharged into the lagoon from the top ("toploading"). The lagoon itself was an aboveground anaerobic system with sloping sides that reached a maximum depth of ~4 m at the center. The surface area of the lagoon was ~25,000 m^2 (~100 m × ~250 m).

2.2. Slurry Composition and Analysis

To determine the concentration of total nitrogen (aqueous ammonia, ammonium, and nitrates) in the slurry, lagoon water samples were taken once every day during the flux sampling periods and submitted to the Department of Soil Science, North Carolina State University for analysis. Samples were analyzed for Total Kjeldahl Nitrogen (TKN) using a digestion procedure, which converts all N in the lagoon sample to NH_4^+. The NH_4^+ concentration in the sample was determined by colorimetry.

2.3. Flux Measurements

Ammonia flux was measured using a dynamic chamber system [*Aneja et al.*, 1996a, b]. Summer measurements were made from August 1–15, 1997. The fall (December 1997) and winter (February 1998) intensives lasted intermittently for 6 and 10 days, respectively. Flux measurements were discontinued during precipitation events. In spring, ammonia flux measurements were made from May 16–27, 1998. Lagoon water temperature was monitored continuously with a temperature probe (Fascinating Electronics, Deer Island, Oregon) immersed ~15 cm below the lagoon water surface ≈48 cm from the chamber, and the lagoon pH was monitored continuously using a double junction submersible electrode (Cole Parmer, Vernon Hills, Illinois), also placed ≈48 cm from the chamber and adjacent to the temperature probe. Some measurements of temperature and pH were also recorded manually.

2.4. Chamber Design and Operation

The dynamic chamber used in this study is a fluorinated ethylene propylene (FEP) Teflon-lined (5 mil thick) open bottom cylinder (diameter ~27 cm, height ~42 cm, and volume ~25 L) inserted into a 1.22 m × 1.22 m floatable platform. When the platform and chamber were placed on the lagoon, the chamber penetrated the lagoon surface to a depth of ~4 cm forming a seal between the lagoon surface and the air within the chamber. The placement of the chamber on the lagoon surface was performed in a statistically random manner. Figure 3 shows a schematic of the floating dynamic chamber system. Compressed zero-grade air (National Welders, Raleigh, North Carolina) was pumped through the chamber at a constant flow rate of ~4.73, 4.14, 2.69, and 2.36 ℓ min^{-1} for the summer, fall, winter, and spring field measurement periods,

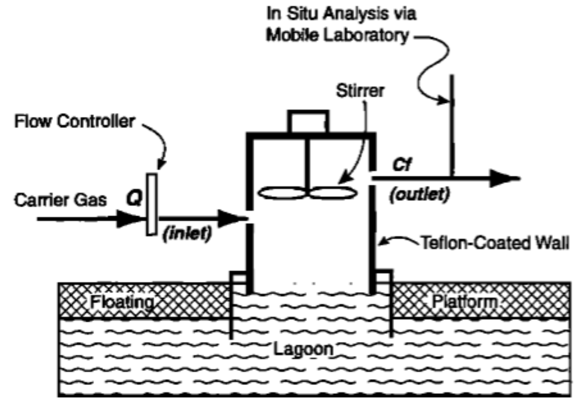

Figure 3. Schematic of a dynamic flow through chamber. The chamber fits on a floating platform which is placed on the surface of the waste treatment lagoon ~24 hours prior to start of measurements to minimize disturbances.

respectively. The air in the chamber was well mixed continuously by a motor-driven Teflon impeller (~20 cm diameter at ~50 rpm). The length of the Teflon tubing (0.64 cm OD) connecting the chamber and the ammonia analyzer was less than 10 m.

2.5. Flux Calculation

The mass balance for NH_3 in the chamber is given by

$$\frac{dC}{dt} = \left[\frac{Q[C]_o}{V} + \frac{JA_L}{V}\right] - \left[\frac{LA_c[C]}{V} + \frac{Q[C]_f}{V}\right] - R \quad (1)$$

where A_L is the lagoon surface area covered by the chamber, A_c is the inner surface area of the chamber, V is the volume of the chamber, Q is the flow rate of carrier gas through the chamber, J is the emission flux, $[C]$ is the NH_3 concentration in the chamber, $[C]_f$ is the NH_3 concentration at the outlet of the chamber, L is the loss term by the chamber wall per unit area assumed first order in $[C]$, and R is the chemical production rate in the chamber.

Because compressed zero air was used as the carrier gas, there is no inlet concentration of ammonia, $[C]_o = 0$, and for a well mixed chamber $[C]_f$ may be assumed to be equal to the NH_3 concentration everywhere in the chamber, $[C]$. Finally, at steady state the above equation reduces to

$$J = [C]_f\left[\frac{LA_c}{V} + \frac{Q}{V}\right]h \quad (2)$$

where h is the height of the chamber measured from the lagoon surface. The value of the total loss term L was obtained (0.079 m min^{-1} and 0.059 m min^{-1} during summer and winter seasons, respectively) by conducting the surface loss experiment as proposed by *Kaplan et al.* [1988] and *Aneja et al.* [1996b].

Once the chamber reached steady state conditions (~30 min of operation), the outgoing air was conducted through Teflon tubes to a Measurement Technologies 1000N stainless steel NH_3 converter which transformed the N_T (=NH_3 + R-NH_2 + NO + NO_2) constituents of the sampled air into nitric oxide (NO) at ~825°C [*Aneja et al.*, 1978]. The sample flow from the NH_3 converter was routed to an Advanced Pollution Instruments (API) Model 200 chemiluminescence-based NO monitor where the transformed N_T concentration in ppbv was de-

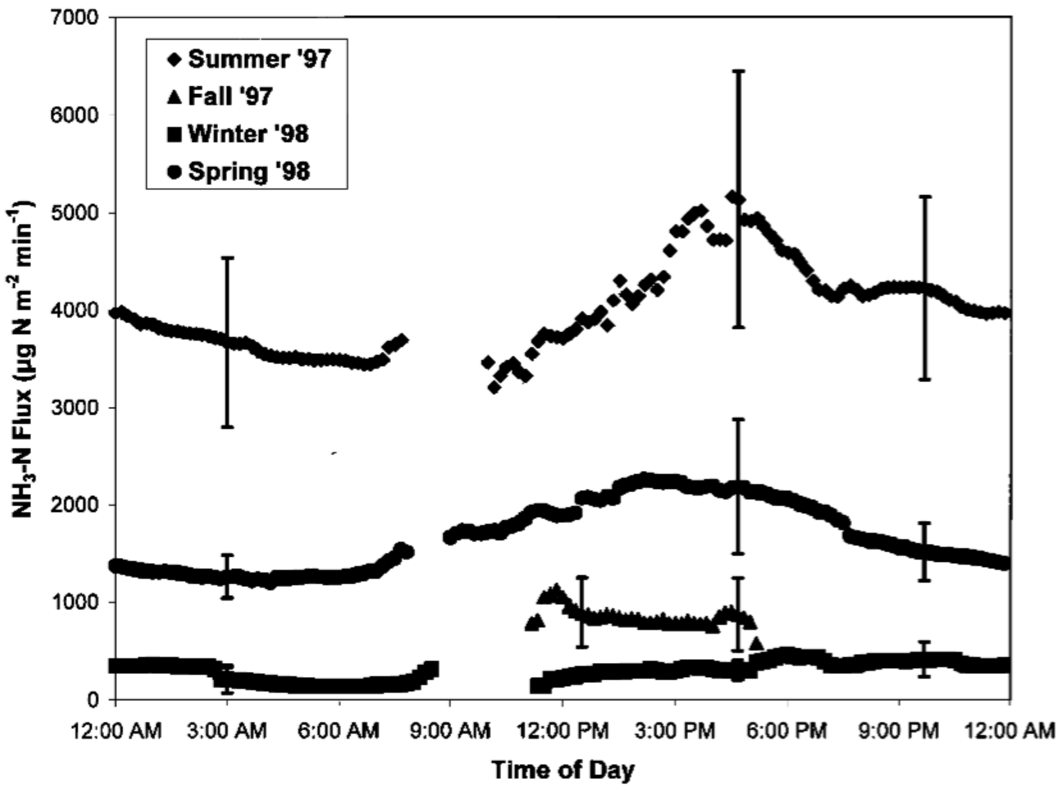

Figure 4. Daily trends of ammonia-N flux from the lagoon during the four seasons. $N = 12$ in summer, $n = 12$ in spring, $n = 6$ in fall, and $n = 4$ in winter except between 1100 and 1700 LT where $n = 8$. Vertical lines represent one standard deviation.

termined. Part of the flow from the chamber was bypassed directly to the API, which transformed to NO only the NO + NO_2 (=NO_x) portion of N_T via a molybdenum converter heated to ~350°C [*Aneja et al.*, 1996a, b; *Fehsenfeld et al.*, 1987]. The API then determined the NH_3 concentration in surface lagoon air by subtracting the NO_x signal from the N_T signal ($NH_3 = N_T - NO_x$). The API was calibrated following written protocols using a Thermo Environment Instruments Inc. Model 146 gas dilution/titration instrument with a calibration gas mixture of NO in N_2 (734 ppmv) and compressed zero-grade air. A multipoint calibration on the API analyzer was performed before each intensive, and the instrument was zeroed and spanned daily during each measurement intensive. The early morning data gaps (Figure 4) represent the time period during which the daily zero/span procedure was performed on the ammonia analyzer. The efficiency of the ammonia converter was checked regularly using a known ammonia concentration. Laboratory tests using a known concentration of NH_3 have shown no measurable conversion of NH_3 to NO at 350°C within a molybdenum converter. Instrumentation was housed in a temperature-controlled mobile laboratory (modified Ford Aerostar van).

3. Results and Discussion

The ammonia-water system has been studied in the past because of its industrial importance and as a means for studying the absorption/desorption mechanism [*Whitman and Davis*, 1924; *Godfrey*, 1973; *Levenspiel and Godfrey*, 1974; *Ibusuki and Aneja*, 1984; *Leuning et al.*, 1984]. All these previous studies indicate that generally both the gas and liquid phase resistance are equally important in determining the overall desorption/absorption rate.

The measurements described for determining ammonia flux at the lagoon-atmosphere interface are made with the dynamic chamber system with continuous impeller stirring (the carrier gas flow rate through the chamber and stirrer speed may be changed). Utilizing the power law profile which is frequently used in air pollution applications [*Arya*, 1999], we are able to estimate what wind velocities are at a height of 0.1 m (the height of the impeller above water-air interface) when 10 m wind heights are known. The power law profile is given by

$$\frac{V}{V_r} = \left(\frac{Z}{Z_r}\right)^m \tag{3}$$

where V_r is the wind velocity at a reference height Z_r and m is taken to be 0.1 for water surfaces [*Arya*, 1988].

Throughout the measurement period during this study, mean wind velocities were between 1 and 4 m/s at a height of 10 m. Through the power law profile above, this equates to wind speeds between 0.6 and 2.4 m/s at a height of 10 cm similar to wind speeds inside the chamber (measured with a hot wire anemometer between ~1 and 2.5 m/s) for our design configuration.

The dynamic chamber system with continuous impeller stirring meets the necessary criteria for performance as a Continuously Stirred Tank Reactor (CSTR). For performance as a CSTR, the chamber needs to be "ideally" mixed [*Aneja*, 1976]. In ideal mixing, the composition of any elemental volume within the chamber is the same as that of any other volume.

Table 3. Summary Table of Average Daily NH_3-N Fluxes by Season

	Mean[a]	Maximum	Minimum
Summer 1997	4017 (987)[b]	8526	2358
Fall 1997[c]	844 (401)	1913	369
Winter 1998[d]	305 (154)	695	90
Spring 1998	1706 (552)	3594	851

NH_3-N flux = (14/17) NH_3 flux.
[a]Units of daily flux are $\mu g\ N\ m^{-2}\ min^{-1}$.
[b]Numbers in parentheses are one standard deviation.
[c]All values were measured from 1100 to 1700 LT.
[d]Measurements were made on February 1, 2, 8, 15, 18, 20, 21, 22, 23, 25, and 26, 1998 (1100–1700 LT), with February 21, 22, and 25 being diurnal variations.

Tracer experiments (Residence Time Distribution) were used to test the flow and mixing characteristics of the system. The results of these mixing studies showed that the dynamic chamber behaved as a "perfect" mixer with negligible stagnancy or channeling.

3.1. Seasonal Fluxes

Table 3 summarizes the average fluxes for each season. Using seasonal averages, the percent of total yearly flux attributable to summer months is ~60%. The change in the daily flux pattern for each season can be seen in Figure 4 together with one standard deviation (±1 s.d.). Each data point in this figure represents an average of the flux measured at a particular time over the entire measurement period. In general, $n = 12$ (where n is the number of flux values that made up the average) for the summer and spring seasons, $n = 4$ for the winter season except between 1100 and 1700 LT where $n = 8$ and $n = 6$ for the fall season. The morning data gaps (Figure 4) represent the time period during which the daily zero/span procedure was performed on the ammonia analyzer. The analyzer was also multipoint-calibrated regularly during the same time period. The reasons for the slight flux increase during morning hours, prior to the daily zero/span procedure, is not known. However, ammonia desorption from the inner surfaces caused by morning temperature increase may be a possibility [*Williams et al.*, 1992; *Adema et al.*, 1990].

3.2. Lagoon Temperature and Ammonia Flux

The pronounced summer maximum flux suggests that temperature is an important factor regulating the loss of NH_3 from the waste lagoon to the atmosphere. In this study the relationship between lagoon temperature and NH_3 flux is examined over a relatively wide range of temperatures (~4°C to ~40°C). Table 4 lists average lagoon surface temperatures measured during each season. We observed an exponential ($r^2 = 0.78$) relationship between NH_3 flux and lagoon water temperature measured over the year as illustrated in Figure 5. Each point in this figure represents an hourly averaged NH_3 flux ($n = 6$, where n is the number of flux values in an hourly average) plotted against the corresponding hourly temperature measurement. Reasons for the exponential relationship are that the liquid phase mass transfer coefficients of NH_3 in water are exponential functions of temperature in the range 5°C to 30°C [*Ibusuki and Aneja*, 1984], and the dependence of Henry's law on temperature [*Dasgupta and Dong*, 1986; *Bates and Pinching*, 1950]. Thus the transfer of NH_3 across the liquid-gas interface follows an exponential model; and the flux increases exponentially with surface lagoon temperature. The ammonia flux from the waste storage and treatment lagoon in North Carolina may be predicted by the observational model

$$\log_{10}(NH_3 - N\ \text{flux}) = 0.048\ T_\ell + 2.1$$

where NH_3 − N flux ($\mu g\ N\ m^{-2}\ min^{-1}$); T_ℓ lagoon surface temperature (°C).

The reason for the high NH_3 flux during summer is a combination of chemical and physical processes occurring within the lagoon. First, the decomposing waste sludge at the bottom of the lagoon acts as a source of NH_3, and the rate of decomposition increases with temperature. As NH_3 from the surface of the lagoon is volatilized, NH_3 formed from the decomposition of sludge at the bottom of lagoon diffuses upward and replenishes the volatized NH_3 in the upper layers of the lagoon. Since this lagoon is not physically mixed by forced means, ammonia's principle mode of transport is through diffusion and mass transfer processes [*Muck and Steenhuis*, 1982]. As illustrated by *Ibusuki and Aneja* [1984], higher temperatures increase the transfer rate of NH_3 across the liquid-gas interface. Thus summer temperatures coupled with a readily avail-

Table 4. Sampling Periods for NH_3-N Flux Measurements and the Mean, Standard Deviation, and Range of Lagoon Surface (i.e., 15 cm Depth) Parameters Measured During the N-NH_3 Flux Experiments

Season	Sample Dates	Lagoon Temperature[a]	Lagoon pH	TKN[b]
Summer	Aug. 1–15, 1997	30 (3.3)[c]	7.5 (0.18)	648.1 (27.7)
		25.3–39.1	7.1–7.8	587–695
Fall	Dec. 1–17, 1997[d]	11.6 (2.2)[e]	8.0 (0.06)	663.3 (33.7)
		8.4–15.3	7.9–8.1	599–715
Winter	Feb. 1–26, 1998[f]	12.1 (2.1)[e]	7.8 (0.13)	641.7 (39.0)
		8.8–15.1	7.66–8.02	580–727
Spring	May 16–27, 1998	24.7 (3.2)	7.7 (0.06)	603.3 (48.2)
		20.4–35.9	7.64–7.81	540–720

[a]Units of lagoon temperature are degrees Celsius.
[b]Units of TKN nitrogen are mg N L^{-1}.
[c]Numbers in parentheses are one standard deviation.
[d]Flux measurements were made on December 1, 3, 5, 7, 15, and 17, 1997 (from 1100 to 1700 LT).
[e]All values of lagoon temperature and pH were measured from 1100 to 1700 LT manually.
[f]Measurements were made on February 1, 2, 8, 15, 18, 20, 21, 22, 23, 25, and 26, 1998 (from 1100 to 1700 LT), with February 21, 22, and 25 representing diurnal variations.

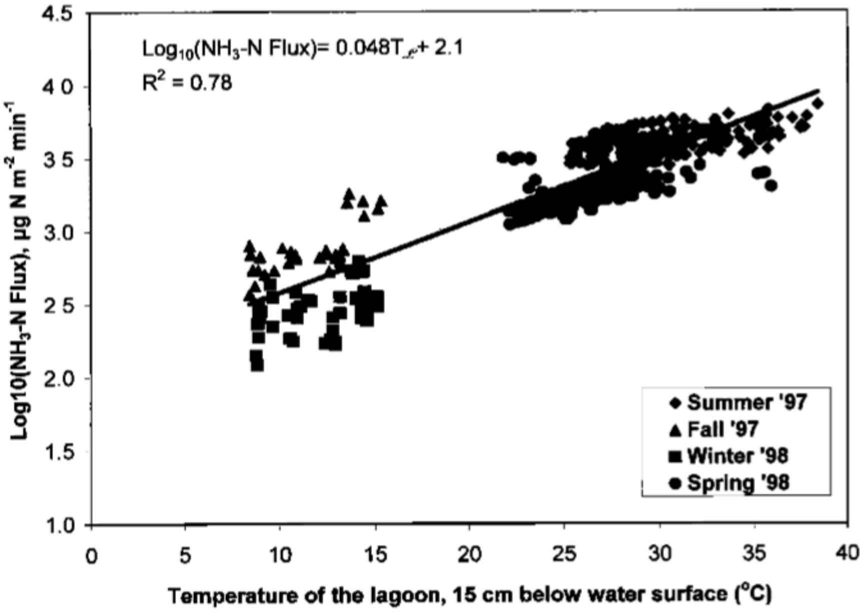

Figure 5. Log of hourly averaged ammonia-nitrogen flux from the lagoon surface plotted against lagoon aqueous phase surface temperature.

able source of NH_3 results in summer fluxes (4017 ± 987 μg N m^{-2} min^{-1}) which are about an order of magnitude greater than those observed during the winter (305 ± 154 μg N m^{-2} min^{-1}) season.

Temperature (lagoon surface water and/or ambient) is a controller of NH_3 emissions in the boundary layer, and therefore NH_3 mixing ratios in the lower troposphere. *Langford et al.* [1992] have suggested that the "typical" seasonal and diel cycles of boundary layer NH_3 levels are, in general, a function of air temperature, with higher NH_3 mixing ratios associated with warmer temperatures. These seasonal NH_3 concentrations should therefore be manifest in ammonium ion concentration [NH_4^+] in precipitation. Multiple regression analysis of monthly volume-weighted ammonium concentration in precipitation at the National Atmospheric Deposition Program (NADP) site located in close proximity to the flux study, NC35 (located in Sampson County, latitude 35°01′33″ N and longitude 78°16′21″W [*NADP/NTN*, 1998]), during the period 1982–1996 reveals a statistically significant ($p < 0.01$, $r^2 = 0.29$) positive correlation between mean monthly surface temperature and log of ammonium concentration (Figure 6). The present study has shown that volatilization of NH_3 from swine waste lagoons has a positively correlated exponential relationship with lagoon surface water temperature. At sites such as NC35, which are likely influenced by multiple nearby swine facilities, this may also contribute to the temperature dependence of ambient NH_3 concentrations and subsequent NH_4^+ concentrations in precipitation. *Walker* [1998] has shown a statistically significant seasonal cycle for [NH_4^+] in precipitation, which maximizes during summer months, at NADP sites across North Carolina.

3.3. Lagoon pH and Ammonia Flux

The highest pH values were observed in the fall and winter seasons and ranged from 7.7 to 8.1 (Table 4). *Koelliker and Minor* [1973] have also observed pH values up to 8 in the fall and winter at a Missouri hog lagoon. The relative stability of lagoon pH throughout the year is due to the high buffer capacity of the slurry [*Olesen and Sommer*, 1993]. The pH of the lagoon is maintained by the bicarbonate ion, formed as a product of the hydrolysis of urea and microbial conversion of organic matter (equation (4)), which neutralizes the H^+ ion released into solution by NH_4^+(aq) as NH_3(aq) volatilizes (equation (5)) [*Sommer et al.*, 1991; *Fordham and Schwerdman*, 1997]:

$$CO_2(aq) + H_2O(aq) \leftrightarrow CO_3^-(aq) + 2H^+(aq) \quad (4)$$

$$NH_4^+(aq) \leftrightarrow H^+(aq) + NH_3(aq). \quad (5)$$

Several published modeling studies [*Dewes*, 1996; *Muck and Steenhuis*, 1982; *Vlek and Stumpe*, 1978; *Sommer et al.*, 1991] report a positive relationship between lagoon NH_3 flux and pH. In lagoon slurry, NH_3 will be in solution with NH_4^+ according to the following equilibrium [*Warneck*, 1988]:

$$NH_3(aq) + H_2O(aq) \leftrightarrow NH_4^+(aq) + OH^-(aq). \quad (6)$$

The direction of the equilibrium in (6) depends on the pH. As the pH increases ([OH^-] increases), the equilibrium shifts toward the left increasing the concentration of NH_3(aq) and hence the potential for volatilization. Thus the proportion of the total NH_3 concentration that is ionized at any time is a function of lagoon pH. Equation (7) provides the theoretical relationship between the aqueous ammonia fraction, $F = NH_3/(NH_3 + NH_4^+)$, and pH of the lagoon [*Loehr*, 1984]:

$$F = \frac{1}{1 + 10^{(pK_a - \mathrm{pH})}} \quad (7)$$

where pK_a is the negative logarithm of the ionization constant for (6). The relationship between flux and the pH over the range observed in this study (7.1 to 8.1) follows the theoretical prediction given by (7).

3.4. TKN and Ammonia Flux

A plot of the average daily flux over the entire year against TKN nitrogen for lagoon samples collected on the same day as

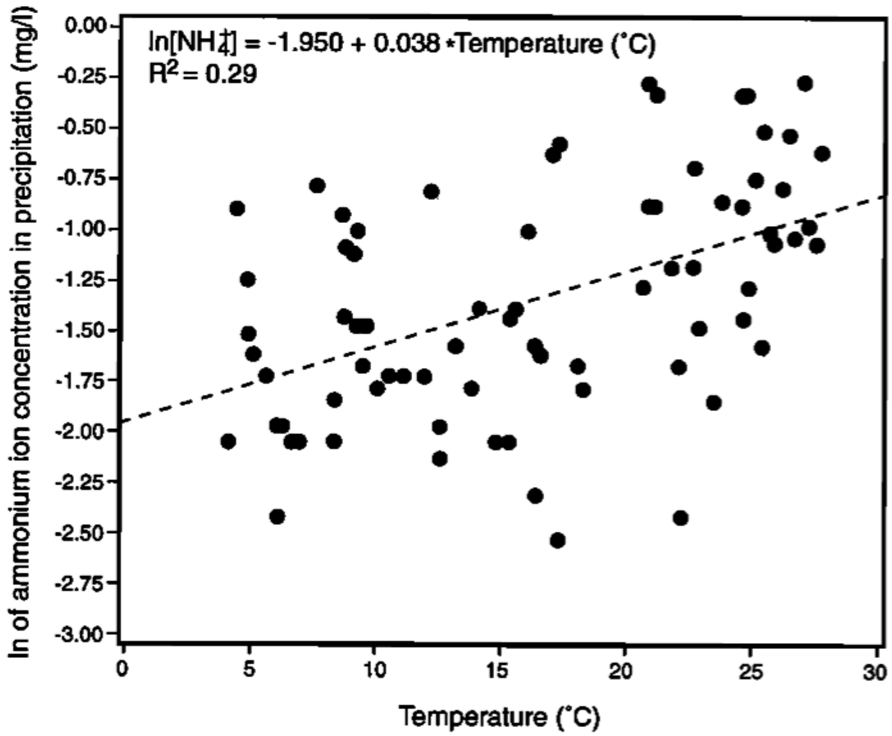

Figure 6. Natural log of monthly volume-weighted ammonium (NH_4^+) ion concentration in precipitation versus mean monthly atmospheric ambient temperature for the NADP/NTN site NC35, located in Sampson County during the period 1983–1996 (solid circles) and the corresponding regression line (dashed line).

the flux measurement is shown in Figure 7. We observed that the TKN levels remain relatively constant, varying between ~500 and ~750 mg N L^{-1} (Table 4). This stability is because the lagoon on which our measurements were taken is part of a steady state commercial operation at which animal weight and feed distributions are about the same throughout the year. Thus the fresh waste input into the lagoon has a relatively constant nitrogen content which keeps lagoon TKN levels steady throughout the year. Across different seasons the lagoon experiences various rates of evaporation and receives varying amounts of precipitation; however, these factors are likely to cause relatively minor fluctuations in the nitrogen concentration of the lagoon.

3.5. Scaling Ammonia Emissions Utilizing Remotely Sensed Data

Using the Geographic Information System (GIS) spot satellite image of North Carolina for the period 1995–1996, a statistically random sample of 50 lagoons was obtained for lagoon surface area in eastern North Carolina (http://www.lib.ncsu.edu/stacks/gis). The 50 lagoons sampled for surface area, measured with the help of ARCVIEW 3.0, was 54 ha (i.e., ~1 ha/lagoon). The number of lagoons in eastern North Carolina is ~2500 lagoons [*NCDA*, 1998]. The seasonal average lagoon ammonia emission from this study is ~1718 ± 523 μg N m^{-2} min^{-1}; and the total ammonia emissions from swine operations is ~68,540 t N yr^{-1} [*Aneja et al.*, 1998b, c] (Table 1). From the emissions obtained in this study, it is therefore estimated that lagoon ammonia emissions in eastern North Carolina comprise approximately 33% of total swine NH_3 emissions in North Carolina.

4. Conclusions

The data presented in this paper add to the growing knowledge of NH_3 flux from animal agricultural practice. The summer intensive conducted during August 1–15, 1997, presented us with typical summer temperatures observed in North Carolina and gave the highest flux estimates of the year. Using the dynamic chamber system, the average flux in the summer of 1997 was 4017 ± 987 μg N m^{-2} min^{-1}. The averages during the spring, fall, and winter seasons were 1706 ± 552, 844 ± 401, and 305 ± 154 μg N m^{-2} min^{-1}, respectively. We found that the NH_3 flux displays a diurnal variation which is highly correlated with lagoon surface water temperature and reaches a maximum between 1500 and 1800 LT. Ammonia flux is found to vary exponentially with lagoon surface water temperature. The pH and TKN levels in the lagoon remained relatively constant at an average value of ~7.5 pH, and ~650 mg N L^{-1}, respectively.

The emission factors obtained by this research and other similar measurements on swine agriculture are summarized in Table 5. In general, there is reasonable agreement between researchers making NH_3 flux measurements from swine agriculture. The emissions factors listed represent emissions values from varying locations around the globe. The scatter in the emission factors may be explained by differences in meteorology, management practices, animal feeds, and experimental error. Using remotely sensed lagoon surface area data, we have also estimated that lagoon ammonia emissions in eastern North Carolina comprise approximately 33% of total swine NH_3 emissions in North Carolina.

It is reasonable to assume that animal agriculture will be a

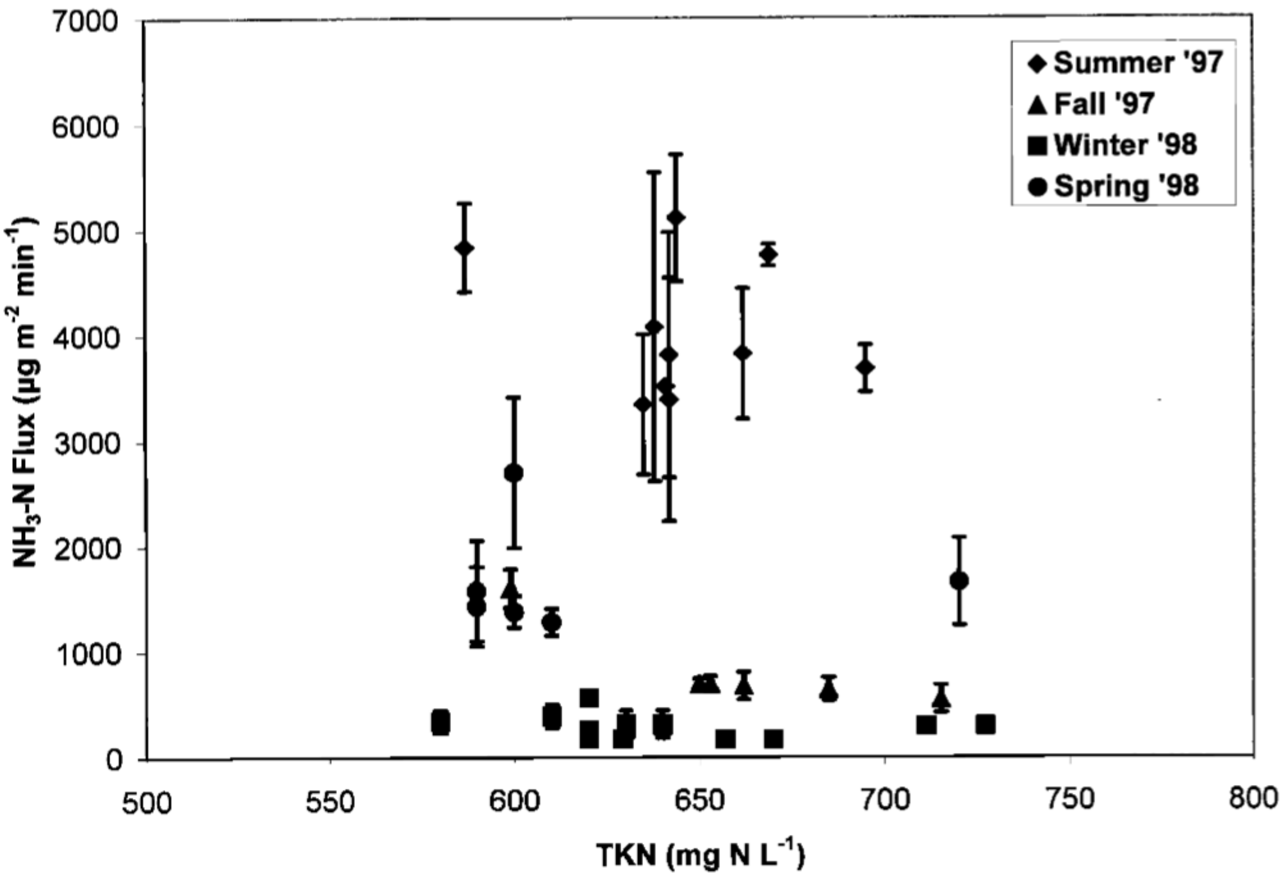

Figure 7. Average daily ammonia-N flux from the lagoon surface versus Total Kjeldahl Nitrogen (TKN) sampled from lagoon collected on the same day as the flux measurements. TKN measurement includes ammonia, ammonium, and nitrates in the aqueous phase.

continued source of significant atmospheric NH_3 emissions in eastern North Carolina. Moreover, *Walker et al.* [2000] have demonstrated that increasing trends in ammonium concentration in precipitation in eastern North Carolina are directly correlated to the ever-expanding hog population in this region. Atmospheric deposition of NH_x will undoubtedly continue to impact nearby ecosystems with the potential of enhancing eutrophication and soil acidification. Furthermore, enhanced NH_3 emissions will enhance particulate matter formation in the region, which reduces visibility [*Barthelmie and Pryor*, 1998] and also causes health problems for workers in livestock agriculture [*Reynolds and Wolf*, 1988; *Michaels*, 1999]. Another concern associated with NH_3 emission is its potentially harmful odor. To address these concerns and outline possible control strategies, further research is required in modeling the fate of NH_x with regional deposition models.

Table 5. Ammonia-Nitrogen Emissions Estimates and Comparison of Results

Author	Emission Factors, kg N animal^{-1} yr^{-1}
Asman and van Jaarsveld [1992][a]	4.41
Battye et al. [1994][a]	7.58
van der Hoek [1998]	0.7–1.79
ECETOC [1994][b]	1.0
Dragosits et al. [1998][a]	3.18
McCulloch et al. [1998][a,c]	4.88–9.52
This study (1998)[b]	
Summer	5.2
Spring	2.2
Winter	0.4
Fall	1.1
Average (this study)	2.2

[a]Includes emissions from waste lagoons, animal houses, and surrounding crops.
[b]Includes emissions from waste lagoons only.
[c]Derived from summer measurements only.

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Characterizing ammonia emissions from swine farms in eastern North Carolina: Reduction of emissions from water-holding structures at two candidate superior technologies for waste treatment

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Abstract

Program OPEN (Odor, Pathogens, and Emissions of Nitrogen) was an integrated study of the emissions of ammonia (NH_3), odor and odorants, and pathogens from potential environmentally superior technologies (ESTs) for swine facilities in eastern North Carolina. This paper, as part of program OPEN, focuses on quantifying emissions of NH_3 from water-holding structures at two of the best ESTs and compares them with the projected emissions from two conventional lagoon and spray technologies (LSTs). The evaluated ESTs are: (1) Super Soils at Goshen Ridge; and (2) Environmental Technologies at Red Hill. The water-holding structures for these two ESTs contained no conventional anaerobic lagoon. A dynamic flow-through chamber was used to measure NH_3 fluxes from the water-holding structures at both the ESTs and at the conventional LST farms. In order to compare the emissions from the water-holding structures at the ESTs with those from the lagoons at the conventional sites under similar conditions, a statistical-observational model for lagoon NH_3 emissions was used. A mass-balance approach was used to quantify the emissions. All emissions were normalized by nitrogen-excretion rates. The percentage reductions relative to the conventional lagoons were calculated for the two ESTs. Results showed substantial reductions in NH_3 emissions at both ESTs. Super Soils had reductions of 94.7% for the warm season and 99.0% for the cool season. Environmental Technologies had slightly larger reductions of 99.4% and 99.98% for the cool and warm season, respectively. As a result of such large reductions in ammonia emissions, both technologies meet the criteria to be classified as ESTs for ammonia emissions.

Keywords: Ammonia emissions; Ammonia flux; Water-holding structures; Environmentally superior technologies (ESTs); Lagoon and spray technologies (LSTs)

1. Introduction

Atmospheric ammonia (NH_3) is a very important alkaline constituent, and has a significant influence

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on a variety of environmental processes (Aneja et al., 2006a, b). Ammonia reacts with a variety of acidic atmospheric species, such as sulfuric acid (H_2SO_4), nitric acid (HNO_3), and hydrochloric acid (HCl), to form ammonium aerosols, namely, ammonium bisulfate (NH_4HSO_4), ammonium sulfate ($(NH_4)_2SO_4$), ammonium nitrate (NH_4NO_3), and ammonium chloride (NH_4Cl).

Ammonia and ammonium are removed from the atmosphere through both wet and dry deposition processes. Wet deposition occurs by either below cloud scavenging (washout) or by rainout (in-cloud processes). Atmospheric NH_3 and its deposition lead to a variety of environmental consequences such as fine particulate matter formation, soil acidification and aquatic eutrophication.

Ammonia is emitted by a large variety of sources such as soils and agricultural crops, synthetic fertilizers, animal waste, biomass burning, fossil-fuel combustion, and human excreta (Oliver et al., 1996; Bouwman et al., 1997). Domestic animal waste is the leading source of global atmospheric ammonia. Studies suggest that it contributes between 20 and 35 Tg of nitrogen per year (Bouwman et al., 1997; Warneck, 2000). In North Carolina, swine waste is the dominant source accounting for 47% of all ammonia emissions, and it is estimated that about 75 000 tons of nitrogen per year are released by hog waste (Aneja et al., 1998). These emissions are related to a rapid increase in hog population, from approximately 3 million in 1992 to 10 million in 1997, when a moratorium was placed. The increase in hog population has been concentrated in the coastal plain region of North Carolina, which contains about 85% of the current pig population (Aneja et al., 2000). The lagoon and spray technology (LST) is the system currently employed to manage hog waste in North Carolina. It consists of an anaerobic lagoon to store and biologically treat the hog waste, which is then sprayed on nearby crops as a source of nutrients.

Due, in part, to the environmental problems associated with ammonia/ammonium emissions from LST farms, a moratorium in 1997 was placed on the construction of swine facilities and the expansion of existing swine facilities until September 2007.

In order to develop sustainable solutions to this problem, an agreement between the North Carolina Attorney General and several commercial hog farming companies was reached to develop potential environmentally superior technologies (ESTs) for hog facilities (Williams, 2001). Program OPEN (Odor, Pathogens, and Emissions of Nitrogen) was an integrated study of the emissions of ammonia, odor and odorants, and pathogens from potential ESTs for hog facilities. Its objectives were to evaluate 16 potential ESTs at swine facilities to determine if they would be able to substantially reduce atmospheric emissions of NH_3, pathogens, and odor from their observed or estimated emissions from the conventional LST used at selected conventional farms in different (warm and cool) seasons or observation periods. Previous papers present the results for the conventional LST farms (Aneja et al., 2007a), and the evaluation of six potential ESTs, that would need improvements/modifications to qualify as ESTs (Aneja et al., 2007b). This paper focuses on characterizing and quantifying emissions of NH_3 from water-holding structures at two ESTs that met the specified performance standards (Williams, 2004) for ammonia emissions reduction, and therefore qualified as ESTs. This evaluation was achieved by comparing them with projected emissions from two conventional (also called, baseline) LST farms. The evaluated ESTs are: (1) Super Soils at Goshen Ridge; and, (2) Environmental Technologies at Red Hill. The water-holding structures for both of these ESTs contain no conventional anaerobic lagoon. Therefore, these might be considered to be most effective for reducing ammonia.

2. Methodology

2.1. Approach to evaluate ammonia emissions at EST farms

Ammonia flux measurements were conducted during 2-week periods representing different seasons (characterized here as warm and cool) at two EST sites in eastern North Carolina and also at two conventional farms (Stokes Farm and Moore Farm), which are also referred to as "baseline" sites for comparison with EST sites (for locations see Fig. 1). Measurements at the different sites were made at different times of the year. Therefore, to compare the EST and LST sites, the different environmental conditions at each site need to be taken into account. This is achieved by the development of a statistical-observational model (Aneja et al., 2007a).

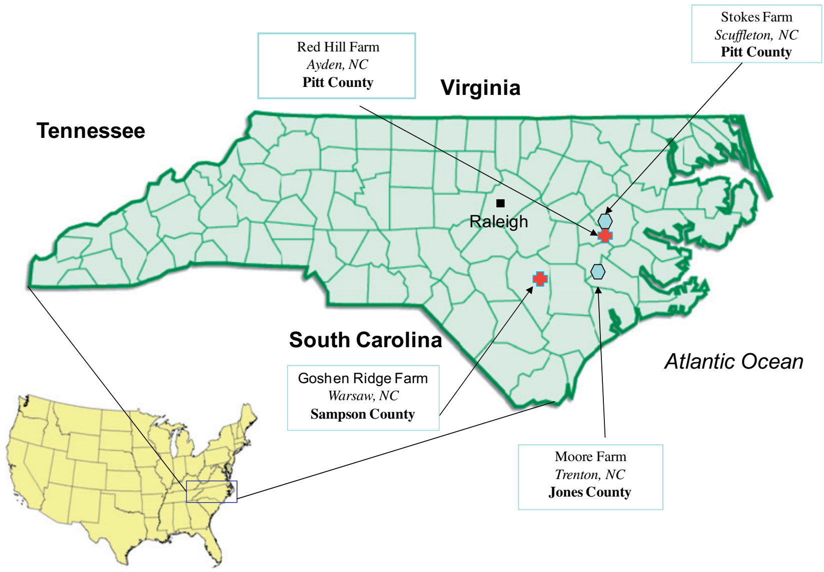

Fig. 1. Map of North Carolina indicating the location of the environmentally superior technologies (ESTs), and lagoon and spray technologies (LSTs). Blue hexagon indicates an LST farm, red cross an EST farm.

2.1.1. Statistical-observational model for lagoon NH_3 flux based on conventional farm measurements

Aneja et al. (2007a) developed a model based on flux measurement data from two conventional farms. For more information on the development of the model, the reader is referred to Aneja et al. (2007a). In this paper the model development is only briefly summarized.

The relationships between NH_3 flux, lagoon temperature, pH and a range of environmental parameters were examined over a relatively wide range of lagoon temperatures (~2–~35 °C) and lagoon–air temperature differences. These were observed during the warm and cool seasons at both conventional farms. The statistical–observational model was developed using multiple regression analysis on flux measurement data from two conventional farms. It is given as

$$\mathrm{Log}_{10}\, F = 3.8655 + 0.04491(T_1) - 0.05946(D). \quad (1)$$

Here, F denotes the average NH_3-N emission from the conventional lagoon in $\mu g\, min^{-1}(1000\, kg\, lw)^{-1}$, T_1 is the lagoon temperature in °C, and D is a hot-air variable that is equal to zero if lagoon is warmer than air, but is equal to $\Delta T = T_a - T_1$ when $T_a > T_1$, and T_a is air temperature in °C at 2 m height. This statistical–observational model was used to estimate the projected NH_3-N flux from lagoons at the LST baseline farms to compare with the measured NH_3-N flux from water-holding structures at an EST site, for the average values of T_1 and D observed at the latter.

2.1.2. Estimation of % reduction in ammonia emissions at EST sites

Both the measured EST emissions and the model estimated LST emissions were normalized by the nitrogen excretion rate (E) for the farm, and are called $\%E$, where $\%E$ represents the loss of ammonia from a source, as a percentage of N-excretion rate. Nitrogen excretion was based on a mass balance approach. Nitrogen excretion rate (E) in unit of $kg\, N\, week^{-1}\, (1000\, kg\, lw)^{-1}$ was determined using the following equation:

$$E = \frac{F_c \times N_f \times (1 - e_r)}{\bar{w}} \times 1000, \quad (2)$$

where F_c is the feed consumed ($kg\, pig^{-1}\, week^{-1}$), N_f is the fraction of nitrogen content in feed, e_r is the feed efficiency rate (ratio of average gain of nitrogen-to-nitrogen intake) (PigCHAMP, 1999), and $\bar{w}$ is the average live animal weight ($kg\, pig^{-1}$). Nitrogen excretion at each farm was calculated in term of the same units as NH_3-N emissions estimated from the water-holding structure of the EST farm and are shown in Table 1.

A potential EST was evaluated by comparison of $\%E$ value from the EST ($\%E_{EST}$) farm to $\%E$ value from a baseline conventional farm ($\%E_{CONV}$), and percent reduction of NH_3-N can

Table 1
The summary of production data at environmentally superior technologies (ESTs) farms

Farm information	No. of pigs	Average pig weight (kg pig^{-1})	Total pig weight (kg)	Feed consumed (kg pig^{-1} week^{-1})	N-content in feed (%)	*N*-excretion (kgN week^{-1} (1000 kg-lw)$^{-1}$)
Goshen Ridge						
April 2003	3519	93.4	328 675	17.03	2.67	3.41
February–March 2004	3138	99.8	313 172	16.18	2.49	2.83
Red Hill						
March–April 2005	2390	69.0	164 910	15.88	3.17	5.09
July–August 2005	3113	66.5	207 015	15.88	2.81	4.69

be estimated as

$$\%\text{reduction} = \frac{(\%E_{\text{CONV}} - \%E_{\text{EST}})}{\%E_{\text{CONV}}} \times 100. \qquad (3)$$

An algorithmic flow diagram is shown in Fig. 2, which summarizes the evaluation of NH_3 emissions from water-holding structures at EST farms.

2.2. Sampling sites

2.2.1. LST sites

Stokes Farm (35.43°N, 77.48°W, 17 m MSL) is located in Pitt County, North Carolina. Measurement campaigns were conducted during 9–20 September 2002 and 6–17 January 2003, respectively. Four naturally ventilated finishing barns housed 4392 animals with an average weight of 104 kg in the fall season and 3727 animals with an average weight of 88 kg in the winter season. The waste (urine and feces) from the hog houses was flushed periodically (4 times a day) with recycled lagoon water and discharged into a storage lagoon from a single effluent pipe. The storage and treatment lagoon was an anaerobic system with 15 170 m^2 of lagoon surface area.

Sampling at Moore Brothers Farm (35.14°N, 77.47°W, 13 m MSL) located near Kinston in Jones County, NC, was conducted during 30 September–11 October 2002 and 27 January–7 February 2003. The farm has eight fully slatted finishing houses (pit recharge) with tunnel ventilation system. The eight finishing barns housed 7611 animals with an average weight of 52 kg in the fall season and 5784 animals with an average weight of 67 kg in the winter season. Pit recharge houses are typically flushed once a week. Waste from all the hog barns was flushed out with recycled lagoon water and discharged into a storage and treatment lagoon from eight effluent pipes, one for each hog barn. The lagoon was an anaerobic system with 17 150 m^2 of surface area.

2.2.2. EST sites

The two EST sites were Goshen Ridge Farm and Red Hill Farm. A brief description of each of the potential ESTs that have been evaluated is provided here. Williams (2006) provides comprehensive detailed information including site plans, design schematics, economics, and projected operational characteristics associated with the technology.

2.2.2.1. Goshen Ridge Farm (solids separation/nitrification–denitrification/soluble phosphorus removal/solids processing system (Super Soils)). Goshen Ridge Farm is located near Beautancus, NC in Duplin County. The NH_3 measurements were conducted during 21 April–2 May 2003 for the warm season and 23 February–1 March 2004 for the cool season. A schematic layout of the EST at Goshen Ridge Farm, including the various sampling points, is given in Fig. 3.

The treatment system employed at Goshen Ridge Farm, known as Super Soils, treats the liquid portion of the waste. The liquid treatment begins with separation of the solid and liquid portions of the waste stream. Solids separation is accomplished using polyacryalmide, a flocculating agent.

The liquid portion of the waste stream flows between tanks in a circulating loop undergoing denitrification as a result of anaerobic activity in one tank, and nitrification through the use of concentrated nitrifying bacteria in the second tank under aerobic conditions. Nitrogen is removed from the waste stream during this stage of the process. The liquid then flows to a settling tank, where phosphorus is removed through the addition of

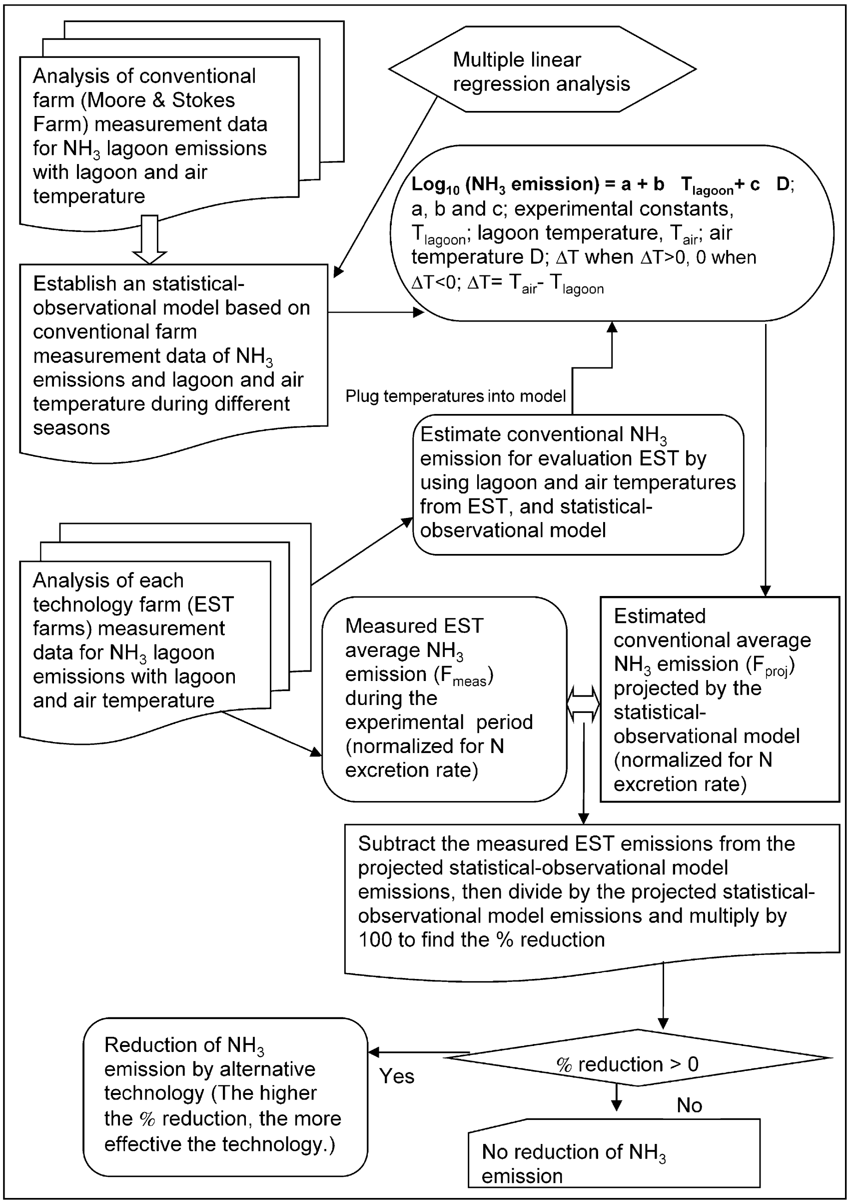

Fig. 2. Algorithm flow chart for evaluation of EST ammonia emissions from water-holding structures.

calcium hydroxide and a dewatering bag system. Calcium phosphate, which has value as a fertilizer, precipitates out during this process, providing a value-added product. During phosphorus removal, the pH of the liquid is raised to 10.5 using lime, which precipitates the soluble P and disinfects the effluent. Roughly 80% of the liquid is recycled through the hog houses, while 20% is used to irrigate crop fields.

At Goshen Ridge Farm, six naturally ventilated houses were treated by the potential EST. For the warm season evaluation period there were 3519 pigs with an average weight of 93.4 kg. For the cool season evaluations, there were 3138 pigs with an average weight of 99.8 kg for the February to March sampling period. For the warm evaluation period, NH_3 fluxes and emissions were measured from the homogenization tank, the 1st denitrification tank, the nitrification tank, the 2nd denitrification tank, and the storage tank. For the cool period evaluation, measurements were repeated in all the water-holding structures except the 2nd denitrification tank.

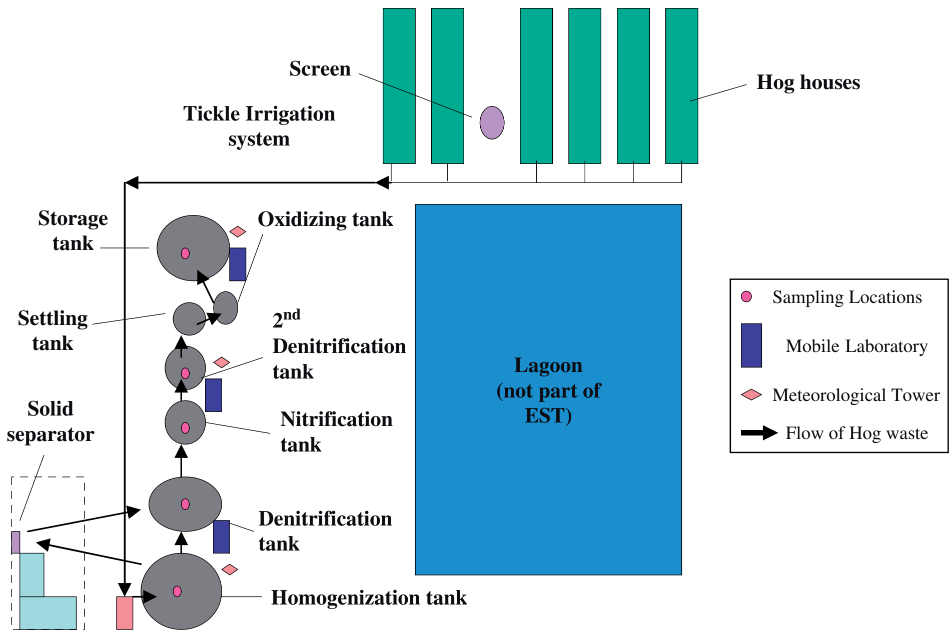

Fig. 3. A schematic layout of the potential EST at Goshen Ridge farm.

2.2.2.2. Red Hill Farm ('closed loop' swine waste treatment system). Red Hill Farm is located near Ayden, NC, in Pitt County. Field campaigns were conducted from 21 March–8 April 2005 for the cool season, and 18 July–5 August 2005 for the warm season. A schematic layout of the EST at Red Hill Farm is given in Fig. 4.

The EST at Red Hill farm was provided by 'Environmental Technologies'. This EST is described as a "closed-loop" system, and its primary objective is to treat the liquid fraction of the waste in such a way that it can be used both for flushing the hog barns and for hog drinking water. This could eliminate the need for the traditional hog waste lagoon. A flush system is used for removing the manure from the barns, which, prior to installation of the treatment system, flushed the waste into a lagoon. The first step in the closed loop process is collection of the waste in an "equalization" or buffering tank. The waste in the tank is continuously pumped to an inclined separator, where the solids are collected and further treated. The liquid collected from the separator is injected with a polymer flocculant and sanitizer/disinfectant and pumped into a settling tank, where flocculated solids collect at the bottom over a period of approximately four hours.

Most of the liquid fraction from the settling tank is returned to the hog barns for re-use as flush water. When the flush tanks are full, however, excess water is pumped to a tertiary treatment system. This system provides filtration and aeration and is housed in a septic tank. The treated water is blended with well water to achieve a dissolved-solids content that is consistent with human drinking water standards for use as hog drinking water. Solids from the settling tanks are combined with the solids from the inclined separator for further treatment.

At this EST farm there are three naturally ventilated hog houses in total. During the cool season evaluation period there were 2390 pigs with an average weight of 69.0 kg. For the warm season evaluation there were 3113 pigs with an average weight of 66.5 kg. During both experimental periods, measurements were conducted at the water tank and at both settling tanks.

2.3. Sampling technique and instrumentation

A dynamic flow-through chamber system was used to measure ammonia fluxes from water-holding structures at the potential ESTs and conventional farms. Various environmental measurements were also made simultaneously. Aneja et al. (2007a) gives a detailed description of the sampling techniques/scheme as well as the

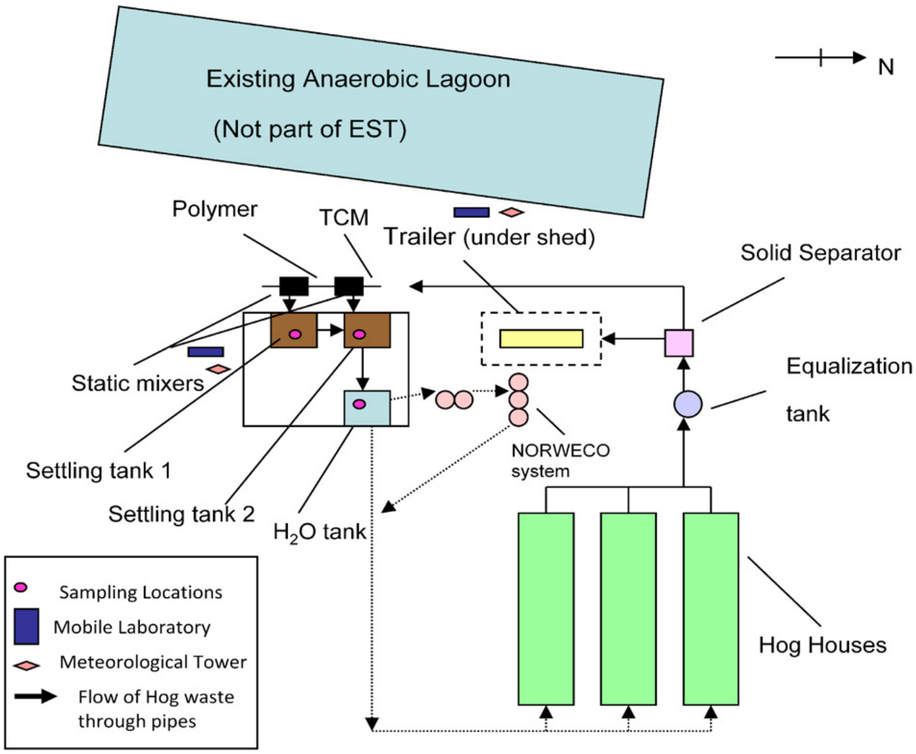

Fig. 4. A schematic layout of the potential EST at Red Hill farm.

instrumentation and environmental measurements used at each experimental site.

3. Results and discussion

3.1. NH_3 fluxes and emissions from water-holding structures

Water-holding structure emissions from two EST farms (Goshen Ridge and Red Hill) were calculated from measurements of NH_3 flux from EST farms, water-holding structure surface areas, and farm production data (number of pigs, feed consumed, and average pig weight) during experimental periods representing both cool and warm seasons. Emissions at the EST farms were normalized to steady-state live animal weight (lw) in the units of kg N week^{-1}(1000 kg lw)$^{-1}$. Average fluxes and total estimated emissions for the water-holding structures are given in Table 2.

At Goshen Ridge farm, ammonia flux measurements for the 1st experimental period were conducted from the homogenization tank, the denitrification tank, the nitrification tank, the 2nd denitrification tank and the storage tank. Their water-holding structure surface areas were 91.6, 67.9, 28.3, 28.3, and 91.6 m^2, respectively.

For the 2nd experimental period, measurements were conducted for all of the same water-holding structures, except the 2nd denitrification tank.

During the 1st sampling period, the highest flux was measured at the denitrification tank, with a 15 min average flux of 5838.1 μg NH_3-N m^{-2} min^{-1}, and a maximum hourly average flux of 6242.1 μg NH_3-N m^{-2} min^{-1}. It should be noted, though, that the concentrations in the flux chamber were beyond the upper limit of detection of the ammonia analyzer; therefore flux values from this tank are highly uncertain. The homogenization tank had the 2nd highest flux with an average 15 min flux of 3092.3 μg NH_3-N m^{-2} min^{-1}, with a maximum hourly flux of 6885.6 μg NH_3-N m^{-2} min^{-1}. Although the homogenization tank has the larger surface area, the higher flux from the 1st denitrification tank results in this tank having the larger emissions.

The other water-holding structures were found to have much lower fluxes and emissions relative to the denitrification and homogenization tanks. The nitrification tank, 2nd denitrification tank and the

Table 2
Average fluxes and total estimated emissions for the water-holding structures

Farm name and sampling period	Water holding structure	Average 15 min flux ($\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$)	Water holding structure surface area (m^2)	Weekly NH_3 emissions ($kg\ N\ week^{-1}$)	Total emissions from water holding structures ($kg\ N\ week^{-1}$)	Total emission/pig ($kg\ N\ pig^{-1}\ week^{-1}$)	Total emission/1000 kg-lw^{-1} ($kg\ N\ week^{-1}$ $(1000\ kg\text{-}lw)^{-1}$)[a]
Goshen Ridge							0.02
April–May 2003	Homogenization tank	3092.3	91.6	2.86	7.11	0.002	
	Denitrification tank	5838.1	67.9	4.01			
	Nitrification tank	213.7	28.3	0.06			
	2nd Denitrification tank	543.1	28.3	0.15			
	Storage tank	33.6	91.6	0.03			
Goshen Ridge							0.004
February–March 2004	Homogenization tank	881.1	91.6	1.36	1.41	0.0004	
	Denitrification tank	33.5	67.9	0.02			
	Nitrification tank	32.4	28.3	0.01			
	Storage tank	13.7	91.6	0.02			
Red Hill							
March–April 2005	Settling tank 1	2073.9	5.8	0.12	0.44	0.0002	0.003
	Settling tank 2	5492.8	5.8	0.32			
	Water tank	80.4	5.8	0.00			
Red Hill							0.0006
July–August 2005	Settling tank 1	996.9	5.8	0.06	0.13	0.00004	
	Settling tank 2	1223.3	5.8	0.07			
	Water tank	43.3	5.8	0.00			

[a]The pig weight used to estimate per pig emissions was based on average pig weight in Table 1.

storage tank had 15 min average fluxes of 213.7, 543.1, and 33.6 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$, respectively. Correspondingly, the emissions followed the same pattern as the fluxes.

For all of the water-holding structures, the flux and emissions were lower in the 2nd evaluation period. For this evaluation, the average 15 min flux for the homogenization tank was the highest at 881.1 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$, with a maximum hourly average flux of 2059.5 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$. The most significant decrease in flux and emissions was from the denitrification tank, with an average 15 min flux of 33.5 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$. The storage tank had the lowest average flux, 13.7 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$. The emissions though were higher than the emissions for the nitrification tank due to the storage tanks' larger surface area.

At Red Hill Farm, fluxes were measured from three water-holding structures, the two settling tanks and the treated water storage tank all of which had an area of 5.8 m^2. Average fluxes during the March–April sampling period for settling tanks 1 and 2 were 2073.9 and 5492.8 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$, respectively. The treated water storage tank had the lowest flux value, with a flux of 80.4 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$. For the July–August 2005 measurement period, the pattern of flux values was repeated. The fluxes were 996.9, 1223.3, and 43.3 $\mu g\ NH_3$-$N\ m^{-2}\ min^{-1}$, for settling tanks 1, 2, and treated water storage tank, respectively. The

Table 3
Summary of total NH_3 emissions from the EST farms and % reduction during the experimental periods

EST farms	Sampling periods	Measured emission (F_{meas}) (kg N week^{-1} (1000 kg lw)$^{-1}$)	% E_{EST}	EST avg. lagoon temp (°C)	EST avg. D (°C)	Conventional lagoon emission (model/projected (F_{proj})) (kg N week^{-1} (1000 kg lw)$^{-1}$)	% E_{CONV}	% reduction
Goshen Ridge	April–May 2003	0.02	0.6	17.2	0.7	0.40	11.3	94.7
	February–March 2004	0.004	0.1	14.2	0.3	0.31	9.7	99.0
Red Hill	March–April 2005	0.003	0.06	14.9	0.5	0.32	10.0	99.4
	July–August 2005	0.0006	0.01	31.6	0.0	1.95	54.9	99.98

emissions from individual water-holding structures follow the same pattern as the fluxes.

3.2. Evaluation of ammonia emissions from water-holding structures

In order to evaluate the percentage reduction of NH_3 emissions for the water-holding structures, measured or estimated EST emissions were compared with projected emissions at the conventional LST farms. The estimated emissions from the LST farms were adjusted to the environmental conditions, i.e. air and lagoon temperature, which have been determined to be statistically correlated with ammonia emissions. For Environmental Technologies, lagoon temperature measurements were made at a lagoon on the farm that was not part of the EST. At Super Soils, no measurements were made at a lagoon. Therefore, in order to make a fair and logical comparison, lagoon data was used from an earlier reported EST with similar air temperatures. For the 1st evaluation at Super Soils (April–May 2003), the Barham farm (April 2002) lagoon (waste water-holding pond component) temperatures were used (Aneja et al., 2007b). For the 2nd evaluation (February–March 2004) at Super Soils, Barham farm (November 2002) lagoon temperatures were used.

Table 3 shows the summary of the water-holding structure NH_3 emissions measured from EST farms, projected emissions from the water-holding structures at the conventional (LST) farms, and % reduction values for their evaluation of potential N reduction.

For both farms there is substantial reduction in NH_3 emissions from water-holding structures. The Super Soils technology employed at Goshen Ridge farm had reductions of 94.7% and 99.0% for the warm and cool season, respectively. The Environmental Technologies closed loop system had slightly larger reductions, with a reduction of 99.4% in the cool season, and 99.98% in the warm season.

4. Conclusions

Two potential ESTs with no conventional anaerobic lagoon component were evaluated to determine if they would substantially reduce atmospheric emissions of ammonia at the hog facilities and meet the performance standards as compared with estimated or projected emissions from the conventional LST used at two selected hog farms in two different (warm and cool) measurement periods. Both farms showed substantial reductions in NH_3 emissions from their water-holding structures. The Environmental Technologies closed loop system had the largest reductions, with reduction of 99.4% and 99.98% for the cool and warm season, respectively. Super Soils technology had a reduction of 94.7% in the cool season, and 99.0% in the warm season. This study did not address the potential reductions in odor and pathogens that were evaluated by other scientists in the OPEN project (Williams, 2006).

Under the conditions reported herein these two potential ESTs meet the criteria established for ammonia emissions as described for ESTs (Williams, 2004).

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ATTACHMENT 28

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Potential geographic distribution of atmospheric nitrogen deposition from intensive livestock production in North Carolina, USA

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ABSTRACT

To examine the consequences of increased spatial aggregation of livestock production facilities, we estimated the annual production of nitrogen in livestock waste in North Carolina, USA, and analyzed the potential distribution of atmospheric nitrogen deposition from confined animal feeding operations ("CAFO") lagoons. North Carolina is a national center for industrial livestock production. Livestock is increasingly being raised in CAFOs, where waste is frequently held, essentially untreated, in open-air lagoons. Reduced nitrogen in lagoons is volatilized as ammonia (NH_3), transported atmospherically, and deposited to other ecosystems. The Albemarle-Pamlico Sound, NC, is representative of nitrogen-sensitive coastal waters, and is a major component of the second largest estuarine complex in the U.S. We used GIS to model the area of water in the Sound within deposition range of CAFOs. We also evaluated the number of lagoons within deposition range of each 1 km^2 grid cell of the state. We considered multiple scenarios of atmospheric transport by varying distance and directionality.

Modeled nitrogen deposition rates were particularly elevated for the Coastal Plain. This pattern matches empirical data, suggesting that observed regional patterns of reduced nitrogen deposition can be largely explained by two factors: limited atmospheric transport distance, and spatial aggregation of CAFOs. Under our medium-distance scenario, a small portion (roughly 22%) of livestock production facilities contributes disproportionately to atmospheric deposition of nitrogen to the Albemarle-Pamlico Sound. Furthermore, we estimated that between 14–37% of the state receives 50% of the state's atmospheric nitrogen deposition from CAFO lagoons. The estimated total emission from livestock is 134,000 t NH_3 yr^{-1}, 73% of which originates from the Coastal Plain. Stronger waste management and emission standards for CAFOs, particularly those on the Coastal Plain nearest to sensitive water bodies, may help mitigate negative impacts on aquatic ecosystems.

1. Introduction

Agricultural waste is an inevitable byproduct of raising livestock. Historically, farmers have managed this problem by using nutrient-rich livestock wastes as a crop fertilizer. Yet, as small independent farms are increasingly converted to large-scale, industrialized, confined animal feeding operations ("CAFOs"), livestock waste management and disposal becomes more challenging. Here, we address the regional consequences of industrialized livestock production in North Carolina, where

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the number of animals produced has greatly increased while the number of farms is declining (Furuseth, 1997).

North Carolina is one of the leading states in livestock production. Here, we use the term "livestock" to include swine, cattle, and poultry. The state ranks second in the nation in both swine and turkey production, accounting for approximately one-sixth of the U.S. total (North Carolina Department of Agriculture and Consumer Services [NCDA & CS], 2007). Currently, the retention of liquid waste in large anaerobic storage reservoirs ("lagoons") is the most widespread and least costly treatment method for livestock waste (Barker, 1996). In North Carolina there are over 2500 CAFO lagoons, 85% of which are located in the Coastal Plain (NC Division of Environment and Natural Resources, Division of Water Quality [NCDENR DWQ], 2002a; also see Fig. 1). Unlike the municipal treatment of human waste, these waste management facilities employ few wastewater treatment processes. Recent studies report that CAFOs in the Coastal Plain of North Carolina are major sources of nitrogen to the region's nutrient-sensitive estuarine and coastal water (Cahoon et al., 1999; Walker et al., 2000; Whitall and Paerl, 2001; Mallin and Cahoon, 2003).

Nitrogen limitation is widespread worldwide, and occurs commonly in terrestrial, freshwater, and marine systems (Elser et al., 2007). Excessive nutrient loading to nitrogen limited waters can create eutrophic conditions, which are linked to noxious phytoplankton blooms and hypoxia (Paerl and Whitall, 1999; Paerl, 2002), changes in fish and benthic macroinvertebrate communities (Burkholder and Glasgow 1997, Glasgow and Burkholder, 2000; Alderman et al., 2005), and outbreaks of harmful aquatic organisms (e.g. Burkholder et al., 1992).

Atmospheric transport is a major pathway by which nitrogen is delivered to riverine, estuarine and coastal ecosystems (Walker et al., 2000; Whitall et al., 2004). Ammonia (NH_3) volatilizes from lagoons and terrestrial systems to which the waste is applied (Barker et al., 2006; Shaffer and Walls, 2005). In the atmosphere, it is transformed to other forms of reduced nitrogen, transported via air movement and atmospheric patterns, and deposited to other ecosystems through both wet and dry deposition. Approximately 80% of NH_3 emissions in the U.S. are generated by livestock waste (Battye et al., 1994).

Atmospheric deposition of nitrogen emissions now accounts for up to 40% of new nitrogen inputs to coastal ecosystems (Paerl et al., 2002). In North Carolina, direct deposition of reduced N is likely to have a large impact on the large estuarine complex including the Albemarle and Pamlico Sounds (Fig. 1), collectively called the Albemarle-Pamlico Sound. Together with its

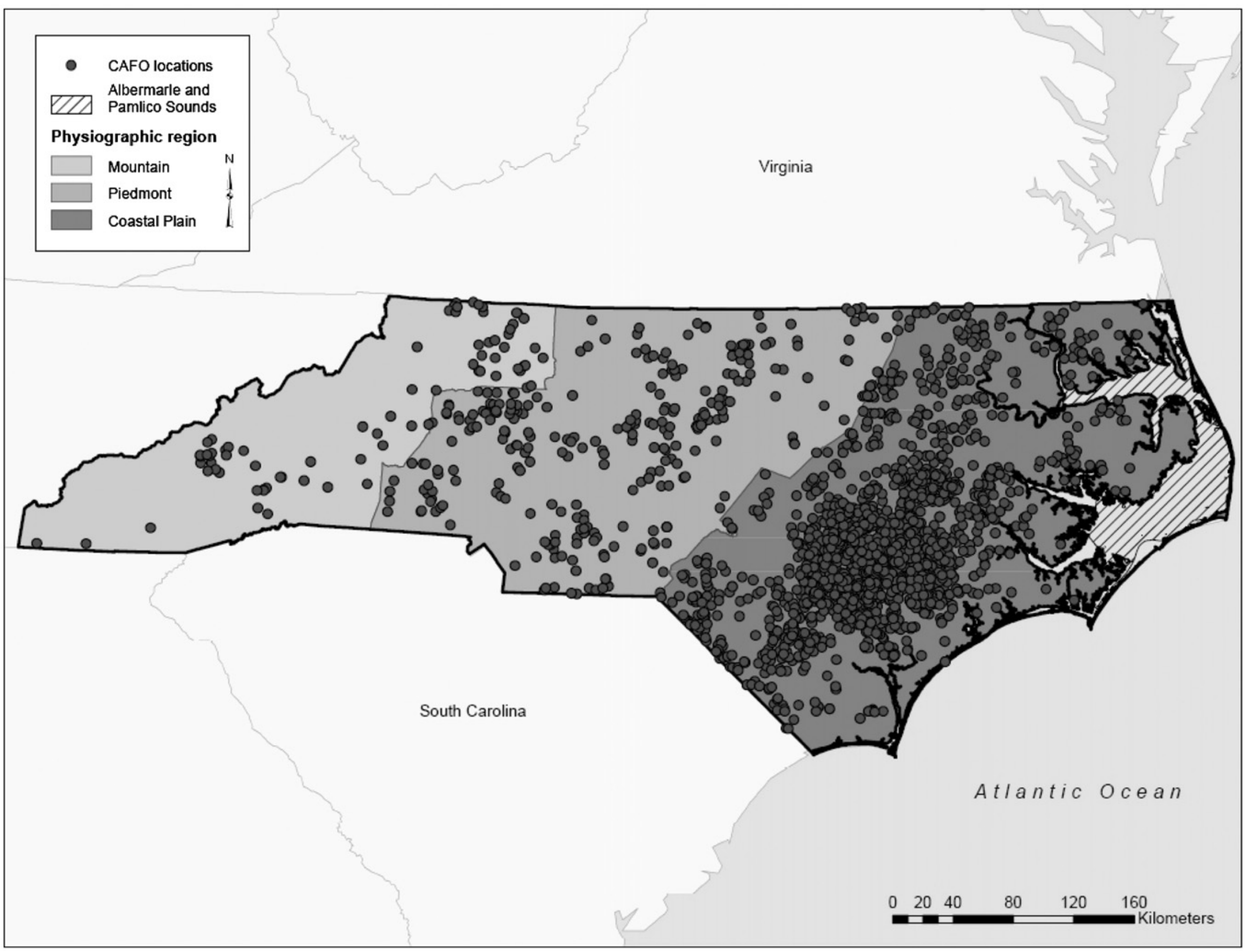

Fig. 1 – Map showing the statewide distribution and aggregation of CAFOs in North Carolina, the physiographic regions of the state, and the Albemarle-Pamlico Sound.

subestuaries, the Sound is the second largest estuarine system in the U.S. It is chronically N-limited, and direct deposition to open water can bypass the mesohaline zone of the estuary, where biological uptake removes much of the N from riverine inputs (Paerl et al., 2002).

National Atmospheric Deposition Program (NADP) data indicate that wet deposition of NH_4^+ has increased in the past decade in southeastern North Carolina, and rates are particularly high in portions of the state's Piedmont and Coastal Plain (NADP, 2006; see Fig. 2). The spatial aggregation of CAFOs in the state's Coastal Plain suggests that nitrogen loading from these operations may be greatest in this region. While CAFO lagoons are considered to be the largest contributors to NH_3 emissions (EPA, 2003), the observed NADP rates (Whitall and Paerl, 2001; Walker et al., 2004) have not been directly linked to CAFO nitrogen emissions.

To better understand the sources of nitrogen deposition, we address two questions. First, how much nitrogen do North Carolina livestock produce in manure, and how much is emitted as NH_3? Second, what is the potential spatial distribution of atmospheric nitrogen deposition from CAFO lagoons? Although management decisions are often made on a statewide basis, few studies have made such assessments at spatial scales larger than single watersheds (but see Paerl et al., 2002; Mallin and Cahoon, 2003; Walker et al., 2004). For this reason, and because North Carolina makes a significant contribution to national livestock production (NCDA & CS, 2007), we seek to address these two questions at the state level.

2. Methods

2.1. *Nitrogen production and ammonia emissions from livestock manure*

We calculated total annual nitrogen production and ammonia (NH_3) emission rates (in units of metric tons (t) NH_3 yr^{-1}) for three categories of livestock: swine (breeders and growers), poultry (broilers, other chickens and turkeys), and cattle (beef and dairy). We estimated total statewide annual nitrogen

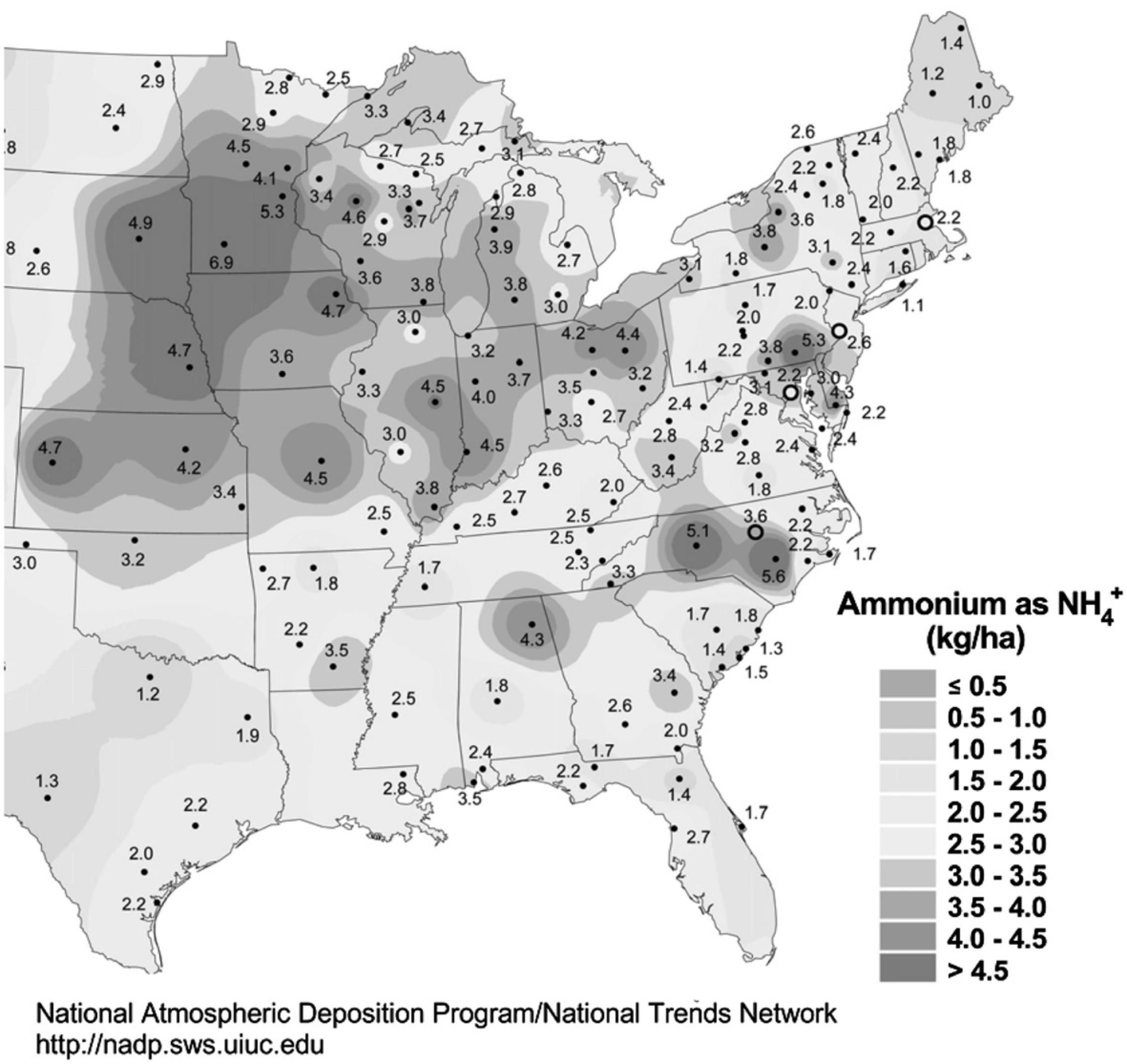

Fig. 2 – Map showing 2005 ammonium atmospheric deposition rate (kg NH_4^+ ha^{-1} yr^{-1}), reproduced from the National Atmospheric Deposition Program (2006).

production for the Mountains, Piedmont, and Coastal Plain physiographic provinces of North Carolina (Fig. 1) following Kellogg et al. (2006). In order to determine nutrient content of manure, we first calculated total manure production. Using 2006 county census of agriculture data, we divided census head counts by the number of animals per animal unit (AU), which represents 1000 lb of live animal weight and serves as a common unit for aggregating over different types of livestock (Kellogg et al., 2006). These standard animal unit conversions are based on the average live weight associated with each livestock category. We then multiplied this value by the mass of "as excreted" manure produced annually per AU to provide an estimate of the total mass of manure produced by all livestock categories. From this value, we determined total nitrogen production in t N yr^{-1} by multiplying metric tons of manure produced by published values for the mass of elemental nitrogen per metric ton of manure (Kellogg et al., 2006).

NH_3 emission rates for all livestock categories were estimated using the Carnegie Mellon University Ammonia Model Version 3.6 (Strader et al., 2004). Source activity data used in the model are from the 2002 Census of Agriculture and emission factors for diary cattle, beef cattle, swine, and poultry are from the EPA National Emission Inventory (2004). The emission factors used in the model are an improvement upon other existing emission inventories (Asman, 1992; Battye et al., 1994; Bouwman and Van Der Hoek, 1997; Doorn et al., 2002), which do not account for seasonal changes in emissions or type of animal production practices used. In addition, most other emission inventories, excluding Doorn et al. (2002), are from European studies and therefore may not be applicable to the U.S. (EPA, 2004).

2.2. Atmospheric transport of nitrogen from livestock waste lagoons

To examine the potential spatial distribution of atmospheric nitrogen in the form of reduced N inputs to North Carolina's aquatic ecosystems from CAFO lagoons, we used a geographic information systems (GIS)-based approach. All spatial analyses were performed using ArcGIS Desktop 9.2 software (Environmental Systems Research Institute, 1999–2006). We analyzed short-, medium-, and long-distance nitrogen transport scenarios by assuming the transport radius from each lagoon to be 80 km, 160 km, or 400 km for the three scenarios, respectively. These distances were derived from studies quantifying nitrogen atmospheric transport distances. We modeled an 80 km radius for our short-distance scenario based on Walker et al. (2000), who showed that wet deposition of reduced N can occur at sites up to 80 km away from NH_3 emissions sources in the North Carolina Coastal Plain. We modeled a 400 km radius for our long-distance scenario based on airshed modeling studies (Dennis and Mather, 2001; Paerl et al., 2002) that have estimated the normalized range of influence of ammonia sources, in the prevailing direction of transport, to be 300–450 km. For our intermediate-distance scenario, we modeled a 160 km transport radius based on the airshed modeling studies (Dennis and Mather, 2001; Paerl et al., 2002), but adjusted the distance to account for more restricted transport in directions other than the prevailing one, and for the fact that deposition rates are higher closer to the source.

For each of the three distance scenarios, we used GIS to perform two analyses based on the locations of CAFOs statewide and the number of lagoons used by each CAFO (NCDENR DWQ, 2002a). Our first analysis examined variation in potential atmospheric nitrogen deposition inputs from NC CAFO lagoons to the state as a whole, and to the Albemarle-Pamlico Sound specifically. To calculate potential deposition, we converted a polygon shapefile of North Carolina to a raster with 1 km×1 km grid cells. For each of the distance scenarios described above, we assumed that some of the emitted ammonia would subsequently be deposited in all directions within the given radius from the livestock operation (isotropic transport and deposition). Therefore, for each 1 km^2 grid cell in the state, we summed the number of CAFO lagoons within the given transport radius (i.e. 80, 160, or 400 km) in any direction.

However, because the prevailing direction of air mass transport in North Carolina is generally from the southwest for 10 months of the year (Brook et al., 1995; State Climate Office of North Carolina, 2007), we assumed that transport and subsequent deposition of reduced N from a given CAFO lagoon would be higher to the northeast of the lagoon. We therefore performed a separate calculation to sum only the number of CAFO lagoons to the southwest of a given grid cell (within the 90-degree arc between south and west): the lagoons that would contribute to deposition of reduced N assuming air mass transport was strictly to the northeast. For each distance scenario, we assumed that 50% of transport and deposition is isotropic; that is, it occurs evenly in all directions. The other 50% is deposited in the 90-degree arc between north and east from each CAFO. Combining these two assumptions in our model gave 62.5% of transport toward the northeast, and the other 37.5% of transport in all other directions combined. We then accounted for the area over which deposition of reduced N may occur under each scenario by dividing the weighted count by the deposition area (area=π r^2, where r is 80, 160, or 400 km under the three scenarios, respectively). We report this predicted weighted count of lagoons within transport range as relative deposition intensity.

We performed the same analysis for 1 km^2 grid cells in the Albemarle-Pamlico Sound. We used digital data showing salinity zones for coastal water (NOAA, 1999) and only included polygons corresponding to the portion of the Albemarle-Pamlico Sound that falls in the mixing zone (5–25 parts per thousand salinity). We converted these polygons to a 1 km^2 grid and performed the same analysis as above. While the area analyzed includes the mesohaline zone (5–18 ppt salinity) where much riverine N is biologically filtered, the majority of the area analyzed is open sound, which is predicted be highly sensitive to direct atmospheric deposition of nitrogen (Paerl et al., 2002).

To test how sensitive our results were to the assumption that 50% of transport and deposition is anisotropic, we constructed a scenario in which 70% of transport and deposition occurs only between north and east, and the other 30% is isotropic from each CAFO. Under this assumption, 77.5% of transport and deposition occurs toward the northeast, and 22.5% occurs in all other directions combined.

Additionally, we examined variation among CAFOs in their potential direct deposition of reduced N to the Albemarle-Pamlico Sound. We calculated the surface area of the Sound that fell within each assumed deposition radius (80-, 160-, and 400-km) of each CAFO in the state. We did this analysis twice.

First, for each distance scenario, we summed the area of the Albemarle-Pamlico Sound that fell within the given transport radius, assuming transport and deposition of reduced N from each CAFO is isotropic. Then, we summed the area of the Albemarle-Pamlico Sound that falls within the transport radius only to the northeast of each CAFO (within the 90-degree arc between north and east).

3. Results

3.1. *Nitrogen production and ammonia emissions from livestock waste*

We calculated total nitrogen production and NH_3 emissions for each physiographic region across the state of North Carolina for cattle, swine, and poultry (Table 1). An examination of 2006 county livestock census data for North Carolina shows that there were 860,000 cattle, 9,500,000 swine and 805,701,000 poultry, including broiler chickens, other chickens, and turkeys (NCDA & CS, 2007). This census data highlights the disparity among population sizes of livestock — they are orders of magnitude apart. In addition, the nitrogen content of manure for all categories of poultry (12.17, 12.22, and 13.77 kg N t^{-1} manure for broiler chickens, other chickens, and turkeys, respectively) was at least two times higher than that of swine and cattle manure (breeder pig manure had the next highest nitrogen content, with 6.01 kg N t^{-1} manure). The estimated total nitrogen production from livestock in North Carolina was 494,000 t, approximately 75.0% of which was produced by poultry, followed by swine, then cattle.

The estimated total NH_3 emission from livestock in North Carolina was 134,000 t NH_3 yr^{-1} (Table 2). Swine contribute 78,000 t NH_3 yr^{-1} or 58% of emissions, followed by poultry (50,700 t NH_3 yr^{-1}; 38% of total) and cattle (5000 t NH_3 yr^{-1}; 4% of total), respectively. With respect to total statewide emissions, approximately 72% were generated in the Coastal Plain region for all livestock categories; over half (55%) of total emissions were generated by swine alone. Poultry accounted for 38% of emissions, with nearly equal amounts generated in the Piedmont and Coastal Plain regions.

Table 2 – Estimated regional annual NH_3 emissions (metric tons yr^{-1}) by animal category

Category	Mountains	Piedmont	Coastal plain
Cattle	1400	2900	710
Swine	995	3570	73,500
Poultry	6820	21,000	22,900
Total	9215	27,470	97,110

3.2. *Atmospheric transport of nitrogen from livestock waste lagoons*

Livestock operations in North Carolina are, on average, located 155 km from the nearest portion of the Albemarle-Pamlico Sound. Therefore, in our 160 km and 400 km dispersal scenarios, a majority of CAFOs are within transport distance to the Sound. To examine variation among CAFOs in their potential direct deposition of reduced N to the Albemarle-Pamlico Sound, we ranked each CAFO based on the area of the Sound within the modeled transport distance. In most scenarios, a relatively small percentage of CAFOs contributed disproportionately large atmospheric reduced N inputs to the Sound. Under the 160 km scenario, 70% of CAFOs were within transport distance to the Albemarle-Pamlico Sound, regardless of whether transport was assumed to be isotropic (Fig. 3a) or strictly toward the northeast (Fig. 3b). Furthermore, 22% of CAFOs were in deposition range of one-third or more of the Albemarle-Pamlico Sound when isotropic deposition was assumed (Fig. 3a), and 10% were in deposition range of one-third or more of the Sound when deposition toward the northeast only was assumed (Fig. 3b). In the 80 km scenario,

Table 1 – Estimated annual nitrogen production (metric tons yr^{-1}) by animal category and physiographic province for North Carolina

Category	N content (kg N t^{-1} manure)	Pop. size			Manure amt (metric tons)			Total N (metric tons)		
		M	P	CP	M	P	CP	M	P	CP
Swine										
Breeder	6.01	4317	31,863	940,275	9900	72,700	2,150,000	60	437	12,900
Grower/finisher	5.13	37,683	278,137	8,207,725	61,000	450,000	13,300,000	310	2300	68,200
Poultry										
Broiler chickens	12.17	131,000,000	358,500,000	259,500,000	4,310,000	11,810,000	8,546,000	52,450	143,700	104,000
Other chickens	12.22	4,352,000	9,921,000	4,928,000	199,000	455,000	226,000	2430	5560	2760
Turkeys	13.77	0	5,000,000	32,500,000	0	600,000	3,980,000	0	8000	54,810
Cattle										
Dairy milk	4.85	14,856	29,523	9989	300,000	600,000	200,000	1000	3000	1000
Heifers	2.75	32,414	64,414	21,793	214,000	427,000	140,000	590	1170	380
Beef										
Steer/bulls/calves	4.98	79,684	158,351	53,575	548,000	1,090,000	369,000	2730	5430	1840
Cows	4.97	108,046	214,713	72,644	1,000,000	2,000,000	800,000	5000	10,000	4000

*M = Mountain; P = Piedmont; CP = Coastal Plain. Nitrogen content values are from Kellogg et al., 2006.

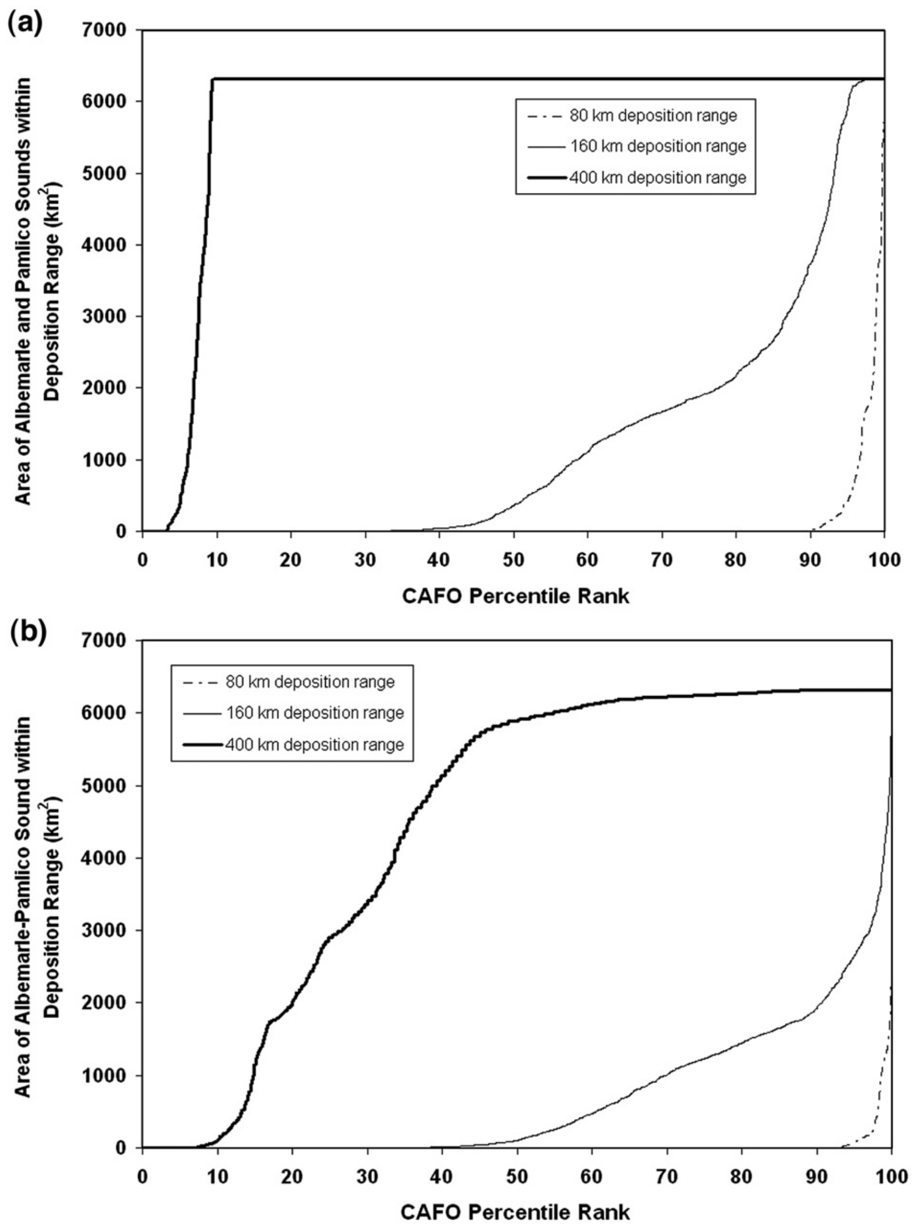

Fig. 3 – Area of the Albemarle-Pamlico Sound within atmospheric nitrogen deposition range of North Carolina CAFOs, for modeled transport distances of 80, 160, and 400 km, assuming (a) isotropic (omnidirectional) transport and deposition from each CAFO, and (b) anisotropic transport strictly toward the northeast of each CAFO.

only the 12% or 9% of CAFOs were within dispersal distance of the Albemarle-Pamlico Sound when isotropic or northeast transport and deposition are assumed, respectively (Fig. 3). Under the assumptions of 400 km transport distance and transport only toward the northeast, 55% of CAFOs were within deposition range of 93% of the Sound (Fig. 3b). In contrast to these five scenarios, under the assumptions of 400 km transport distance and isotropic transport, 90% of CAFOs were within deposition range of the entire Albemarle-Pamlico Sound (Fig. 3a).

We analyzed the spatial distribution of modeled relative deposition intensity both across the state of North Carolina, and within the Albemarle-Pamlico Sound specifically. For the 160 km transport scenario, portions of the Piedmont and Coastal Plain were contained within deposition range of up to 3356 lagoons (Fig. 4a). In addition, portions of the Albemarle-Pamlico Sound were within transport distance of 2645 lagoons. The 80 km scenario produced a qualitatively similar pattern with high relative deposition intensity on the Coastal Plain of North Carolina. Under this scenario (Fig. 4c), portions of the state were within the dispersal distance of up to 2494 lagoons. However, only 320 lagoons are within deposition range of the Albemarle-Pamlico Sound under this scenario (Fig. 4b). Under the short- and medium-distance scenarios, the areas of the state that received the bulk of the loading fall in several major watersheds, including the Tar-Pamlico, Neuse, and Cape Fear watersheds. Under our long-distance scenario, portions of North Carolina were contained within the airshed

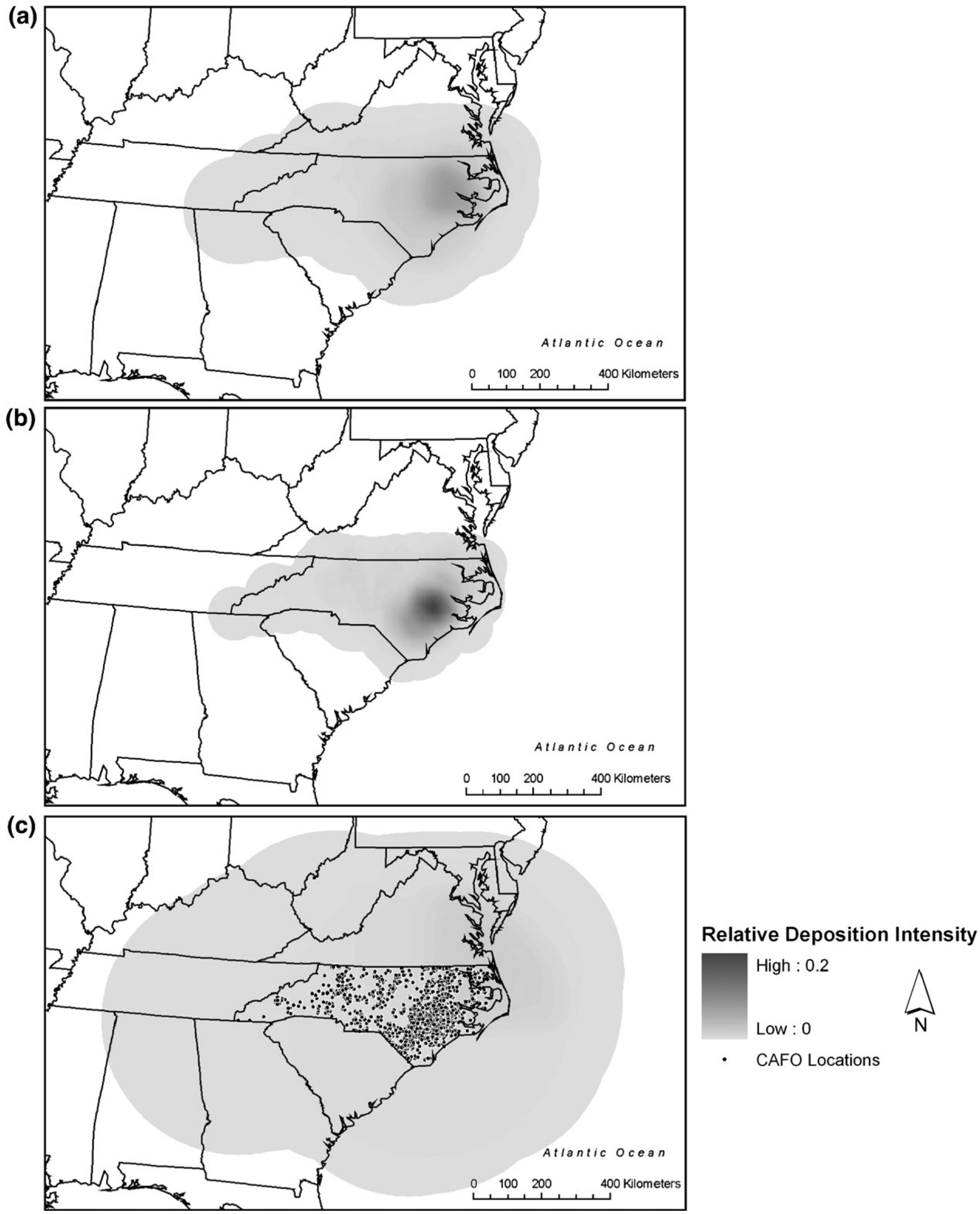

Fig. 4 – Relative deposition intensity for modeled transport distances of: (a) 160 km, (b) 80 km, (c) 400 km. Locations of CAFOs are shown in (c), but are not displayed for (a) and (b) because they would obscure the relative deposition intensity pattern shown on these maps. Atmospheric transport was assumed to be 50% isotropic, and 50% strictly to the northeast of CAFO lagoons.

of up to 3763 lagoons. Finally, we note that the maximum relative deposition intensity varied by an order of magnitude between the short- and long-distance transport scenarios. This is because the same quantity of ammonia emissions from the source is diluted over a larger deposition area as the deposition range is increased.

To further quantify the spatial concentration of reduced N deposition, we ranked each square kilometer of the state based on the number of CAFO lagoons within the modeled transport distance. Overall, we estimated that between 14% and 37% of the state potentially received 50% of atmospheric nitrogen deposition from CAFO lagoons (Fig. 5a). Therefore, a

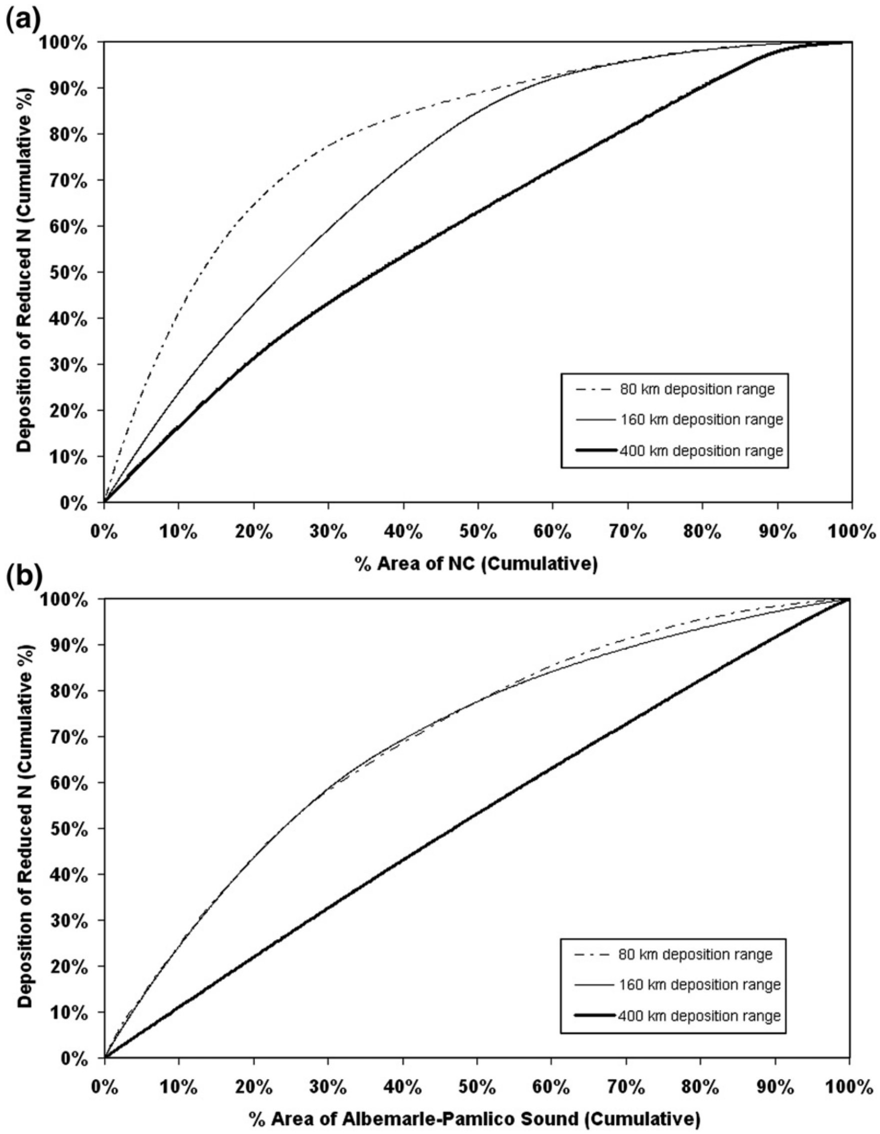

Fig. 5 – The cumulative spatial distribution of atmospheric reduced nitrogen deposition from CAFO lagoons, for modeled transport distances of 80, 160, and 400 km: (a) to North Carolina, and (b) to the Albemarle-Pamlico Sound from CAFO lagoons. Atmospheric transport was assumed to be 50% isotropic, and 50% strictly to the northeast of CAFO lagoons. Percentages are based on the total received by the state and the Sound, respectively.

small portion of the state potentially receives the bulk of the deposition. Under the 160 km scenario, 31% of reduced N may travel outside state boundaries, while in the 80 km scenario, only 10% of reduced N may travel outside state boundaries. However, assuming a 400 km transport distance, up to 77% of the total reduced N originating from CAFO lagoons may travel outside state boundaries. We also ranked each square kilometer of the Albemarle-Pamlico Sound based on the number of CAFO lagoons within transport distance. Between 24% and 47% of the Sound receives 50% of the atmospheric nitrogen deposition from CAFO lagoons (Fig. 5b).

When we modified our assumption of 50% anisotropic and 50% isotropic transport and deposition to 70/30% anisotropic/isotropic, we obtained similar results. Between 12% and 28% of the state potentially receives 50% of atmospheric N deposition from CAFO lagoons, and between 21% and 45% of the Albemarle-Pamlico Sound receives 50% of the deposition. Furthermore, between 10% and 78% of reduced N may travel outside state boundaries under the 80 km and 400 km scenarios, respectively, which is close to the range of values under the 50/50% assumption. Therefore, our results are not very sensitive to the assumption of 50/50% anisotropic/isotropic deposition and transport from CAFO lagoons.

Overall, the eastern portion of North Carolina had the highest number of waste lagoons within atmospheric transport distance regardless of the distance scenario assumed. This is not surprising considering the concentration of CAFOs in this region of the state. However, our results highlight the large number of lagoons within the deposition range of sensitive coastal systems, the concentration of the bulk of nitrogen inputs in a small fraction of the state, and the qualitative robustness of these patterns to variation in atmospheric nitrogen transport distance.

4. Discussion

Industrialized farming and CAFOs have replaced traditional farming practices by increasing livestock populations per farm, which also increases the amount and concentration of animal waste. Our results emphasize (a) the amount of nitrogen produced and emitted by livestock manure in industrialized farms in North Carolina, and (b) the significance of the spatial aggregation of CAFOs in the eastern portion of the state for patterns of atmospheric nitrogen deposition.

Much of the livestock feed originates from out-of-state sources (Cahoon et al., 1999). Thus, most of the nitrogen deposition from CAFOs originates from outside the state and represents a net input to the state's nutrient budget. Poultry contributes the largest percentage of total nitrogen production and is second only to swine in NH_3 emissions across the state of North Carolina. This resulted from both the size of the poultry population in North Carolina and the high nitrogen content in poultry manure. Poultry have significantly lower NH_3 emission rates than cattle and swine (Battye et al., 1994; Doorn et al., 2002), emphasizing the importance of livestock population size on NH_3 emissions. We examined the roles of livestock type and population size; however, NH_3 emissions are also influenced by farm facilities, waste management practices, and meteorological conditions (Battye et al., 1994).

Our results vary from other published figures. Mallin and Cahoon (2003) showed that swine were the top nitrogen producers on the Coastal Plain. However, our analysis also includes the state's Piedmont and Mountain physiographic regions (Table 2). Although swine may contribute the bulk of nitrogen when only CAFOs in the NC Coastal Plain are considered, our analysis includes the state's Piedmont, in which the bulk of poultry are produced. With the exception of poultry, our estimates are comparable with Mallin and Cahoon (2003) for the Coastal Plain region. The disparity for poultry appeared to result from our use of updated census data and animal waste nitrogen production rates.

Our analysis suggests that the aggregation of CAFOs on the Coastal Plain of North Carolina causes the majority of the reduced N deposition to be received by a small portion of the state's area, where aquatic systems are most abundant. In particular, the Albemarle-Pamlico Sound, which is vulnerable to direct deposition of reduced N, is located within transport range of the majority of CAFOs under our medium- and long-distance scenarios. These patterns highlight the relative magnitude of impacts from the aggregation of CAFOs, and underscore the importance of understanding transport distance and fate of NH_3 emitted from CAFOs.

We analyzed multiple scenarios regarding the atmospheric transport of reduced nitrogen originating from CAFOs, including short, medium, and long-distance transport, and varying degrees of isotropic vs. directional transport. Which of these scenarios is most realistic? Like all models, they are all caricatures of reality. Regional-scale atmospheric transport of nitrogen is complex (Walker et al., 2000; Dennis and Mather, 2001; Paerl et al., 2002) beyond what could be simultaneously applied to thousands of individually modeled sources. Deposition rates are higher closer to the source, declining with distance. Transport occurs via air mass movement, and thus is directional at a given time, but when averaged over a year occurs in all directions to some degree. Nonetheless, examining the consequences of different model assumptions begins to provide insight into these more complex processes. For example, while we assumed a constant deposition rate within the modeled transport distance, varying that distance allows some assessment of how deposition varies with distance from a source CAFO. Specifically, areas within the 80 km transport distance can be assumed to receive higher rates of deposition than those within the 160 km transport distance, which would receive higher rates than for the 400 km distance.

The analyses presented here are not intended to represent a detailed model of nitrogen deposition from CAFOs in North Carolina; rather, they are meant to explore the potential effects of a broad range of nitrogen transport scenarios from CAFOs. Our model examines general spatial patterns of potential transport of reduced N for all CAFOs throughout North Carolina, based on a few simple assumptions. First, we assume that all CAFOs and lagoons are equivalent, except in location. For discussion of variation among farms and lagoons, see Battye et al. (1994) and Aneja et al. (2001). Second, we assume that atmospheric reduced N occurs evenly over all landscapes and water types. In fact, the rate of deposition depends on water salinity and landcover type (Paerl et al., 2002). Airshed modeling (Dennis and Mather, 2001) and source-receptor modeling (Walker et al., 2000) have been able to examine in greater detail the transport distance of reduced N to and from a limited number of sites.

Nonetheless, the results of our GIS model agree qualitatively with NH_4^+ deposition maps developed by the NADP. NADP isopleth maps show a high amount of NH_4^+ wet deposition in southeastern North Carolina (NADP, 2006; see Fig. 2), corresponding to the area that is within transport distance of a high number of CAFO lagoons under our 80 km and 160 km transport scenarios (Fig. 4a,b). Our simplified model omitted many relevant factors. Thus, its ability to capture observed patterns of deposition suggests that observed regional patterns of reduced N deposition can be largely explained by the factors that it did include: limited atmospheric transport distance, and spatial aggregation of CAFOs.

We focused on one particular transport pathway in our analysis: atmospheric deposition of reduced N from North Carolina CAFO lagoons to the Albemarle-Pamlico Sound. This coastal system provides ecological goods and services to humans. In particular, it and its upstream estuaries provide a nursery for 80% of the fisheries on the mid-Atlantic U.S. coast (Copeland and Gray 1991). We did not directly consider other pathways such as surface runoff or groundwater transport from CAFO lagoons, riverine transport of nitrogen deposited to terrestrial ecosystems (Paerl et al., 2002), or other sources such as industrial wastewater effluent or agricultural and residential applications of synthetic fertilizers. However, due to the Sound's location downstream of the biological nitrogen sink in the mesohaline zone, direct deposition is likely to have a high impact on ecosystems there (Paerl et al., 2002).

Our analysis of the potential spatial distribution of CAFO reduced N deposition throughout North Carolina indicates that the highest deposition rates are in the eastern part of the state in the Neuse and Tar-Pamlico River watersheds, which are part of the Albemarle-Pamlico Estuarine System. These rivers are considered nutrient-sensitive waters by the State of

North Carolina (NCDENR DWQ, 2002b, 2004). In particular, the Neuse River is subject to the Neuse Basin Nutrient Sensitive Waters Management Strategy, which calls for a 30% reduction in total nitrogen entering the Neuse River Basin (NCDENR, 1997). Furthermore, the Neuse Estuary showed a 500% increase in NH^{+} concentration over the period 1993–2004 (Burkholder et al., 2006). Over the same period, the estuary showed a significant decrease in bottom water dissolved oxygen (Burkholder et al., 2006), a phenomenon that can result from nutrient loading to streams (Mallin et al., 2004).

Although we have focused on the Albemarle-Pamlico Sound, our results also have implications for freshwater systems. According to our analysis, heavy deposition of reduced N occurs in and near the Northeast Cape Fear River. This is a blackwater river that has shown increases in NH_4^+ at monitoring stations (Mallin and Cahoon 2003; Burkholder et al., 2006). Experiments in blackwater rivers indicate that because algal production in these systems is N-limited, N additions can increase biochemical oxygen demand, leading to lower dissolved oxygen in these freshwater systems as well (Mallin et al., 2004).

We suggest that these previous findings in both estuarine and freshwater systems are a direct result of the spatial distribution of transport and deposition of reduced N from CAFOs we modeled here. Thus, the increase in direct atmospheric deposition to freshwater, estuarine, and coastal systems can result in significant ecosystem impacts. Overall, our analysis emphasizes the magnitude and spatial distribution of NH_3 production and deposition from CAFOs, a significant source of atmospheric reduced N in North Carolina.

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ATTACHMENT 29

Agricultural ammonia emissions and ammonium concentrations associated with aerosols and precipitation in the southeast United States

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[1] Temporal and spatial variations in ammonia (NH_3) emissions and ammonium (NH_4^+) concentrations associated with aerosols and volume-weighted NH_4^+ concentration in precipitation are investigated over the period 1990–1998 in the southeast United States (Alabama, Florida, Georgia, Kentucky, North Carolina, South Carolina, Mississippi, and Tennessee). These variations were analyzed using an NH_3 emissions inventory developed for the southeast United States and ambient NH_4^+ data from the various Clean Air Status and Trends Network (CASTNet) and the National Atmospheric Deposition Program/National Trends Network (NADP/NTN). Results show that natural log-transformed annual NH_4^+ concentration associated with aerosols increases with natural log-transformed annual NH_3 emission density within the same county ($R^2 = 0.86$, $p < 0.0001$, $N = 12$). Natural log-transformed annual volume-weighted average NH_4^+ concentration in precipitation shows only a very weak positive correlation with natural log-transformed annual NH_3 emission densities within the corresponding county ($R^2 = 0.12$, $p = 0.04$, $N = 29$). Analysis of NH_4^+ concentration associated with aerosols at CASTNet sites revealed that temperature, precipitation amount, and relative humidity are the most statistically significant ($p < 0.05$) parameters in predicting the weekly concentrations of NH_4^+ during the period 1990–1998. Wind speed and wind direction were also statistically significant ($p < 0.05$) at several CASTNet sites, but the results were less consistent. Investigation into wet NH_4^+ concentration in precipitation consistently yielded temperature as a statistically significant ($p < 0.05$) parameter at individual sites. Trends over the period 1990–1998 revealed a slight decrease in NH_4^+ concentration at CASTNet site SPD, Claiborne County, Tennessee (2.14–1.88 $\mu g\ m^{-3}$), while positive trends in NH_4^+ concentration in precipitation were evident at NADP sites NC35, Sampson County, North Carolina (0.2–0.48 mg L^{-1}) and KY35, Rowan County, Kentucky (0.2–0.35 mg L^{-1}) over the period 1990–1998. *INDEX TERMS:* 0315 Atmospheric Composition and Structure: Biosphere/atmosphere interactions; 0322 Atmospheric Composition and Structure: Constituent sources and sinks; 0330 Atmospheric Composition and Structure: Geochemical cycles; 0365 Atmospheric Composition and Structure: Troposphere—composition and chemistry; *KEYWORDS:* ammonia, ammonium, aerosols, agriculture, southeast United States, statistical model

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1. Introduction

[2] Ammonia (NH_3) plays an important role in the atmosphere, neutralizing acids formed by the oxidation of such compounds as sulfur dioxide (SO_2) and nitrogen oxides ($NO_x = NO + NO_2$) [*Aneja et al.*, 2000; *RIVM*, 1995; *Nihlgard*, 1985; *Asman et al.*, 1982]. These reactions result in ammonium (NH_4^+)-containing aerosols, which may be of concern in particulate matter (PM_{fine}) nonattainment areas. At the Earth's surface, NH_x (= $NH_3 + NH_4^+$) at low concentrations can be a valuable source of nutrient input; however, high concentrations can lead to acidification of soils, forest decline, and eutrophication of waterways [*Asman*, 1994; *Aneja et al.*, 1998]. Therefore, the spatial distribution of NH_3 emissions and parameters which control

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the fate of this specie are important in determining areas of excess nitrogen input, and will aid in the development of models to predict the transport and deposition of atmospheric NH_x.

[3] Globally, approximately 54 (22–83) Tg N-NH_3 (1 Tg $=10^{12}$ g) are emitted each year [*Bouwman et al.*, 1997; *Schlesinger and Hartley*, 1992; *Warneck*, 1988]. The largest fraction (~41%) is from domestic animal excreta, at approximately 22 Tg N-NH_3 yr^{-1}. In the United States, domestic animal waste is also the largest contributor to atmospheric NH_3 emissions, responsible for approximately 80% nationwide [*Battye et al.*, 1994]. Combined with fertilizer application to farmland, animal husbandry and agricultural practices account for ~90% of the total NH_3 emitted in the United States each year [*Battye et al.*, 1994]. Approximately 32% of the southeast United States (Alabama, Florida, Georgia, Kentucky, North Carolina, South Carolina, Mississippi, and Tennessee) is used for farming practices and agriculture and while the southeast accounts for only 12% of the total area of the continental United States, it holds 18% of the total farmland [*USDA*, 1999].

[4] Analysis of the fate of NH_3 emissions in the United States is complicated by a lack of data on gaseous NH_3 in the ambient atmosphere. NH_3 monitoring data for the United States are rare, and data on long-term ambient trends are generally not available for gaseous NH_3. However, acid deposition monitoring networks provide considerable data on NH_4^+ ion concentrations in particulate matter. The Clean Air Status and Trends Network (CASTNet) measures concentrations of particulate NH_4^+, sulfate (SO_4^{2-}), and nitrate (NO_3^-). These data can provide some insights into the fate of gaseous NH_3 emissions. Therefore the objective of this study is to investigate concentrations of NH_4^+ associated with aerosols and in precipitation and NH_3 emissions in the southeast United States over the period 1990–1998, with the intent of defining relationships between NH_4^+ concentrations, local meteorology and NH_3 emissions.

2. Methods

2.1. Data Selection

[5] To estimate NH_3 emissions, agricultural data was obtained from the 1997 Census of Agriculture [*USDA*, 1999]. The census provided a complete data set for cattle and horses (i.e., an exact number was provided for each county in each state). However, the data for poultry, swine, and sheep was incomplete. In this case, the total number of animals was based on the average farm inventory. When the statewide population was less than 5000 for a particular animal, the estimate for that animal was considered negligible and therefore excluded from the final estimates. In the case of fertilizer, data was obtained from the Association of American Plant Food Control Officials, Inc. and is based on sales made by fertilizer registrants in each state.

[6] Data on NH_4^+ concentration associated with aerosols was obtained from the Clean Air Status and Trends Monitoring Network [*CASTNet*, 1998]. CASTNet was designed to be a rural monitoring network collecting data to establish site-specific measurements of total deposition and is considered the nation's primary source for estimates of dry acidic deposition and rural ozone (O_3) concentrations [*CASTNet*, 1998]. The network consists of 51 monitoring sites located across the United States, most of which have been operational since 1987. Continuous measurements of meteorological data including temperature, relative humidity, solar insolation, precipitation, wind speed, and wind direction are taken at each site and atmospheric concentrations of NH_4^+ are obtained from weekly filter pack measurements [*Holland et al.*, 1999].

[7] The CASTNet monitoring network [*USEPA*, 1998] measures weekly average ambient concentrations of particulate NH_4^+ [*Lawrence et al.*, 2000; *Sickles et al.*, 1999; and *Clarke et al.*, 1997]. The network also measures concentrations of particulate SO_4^{2-}, NO_3^-, base cations, O_3, SO_2, and nitric acid (HNO_3). NH_4^+ in the particulate, which is collected on a Teflon filter, is measured on a weekly basis by colorimetry. The CASTNet network was not designed to measure NH_3 gas. Ten CASTNet sites chosen for this study were selected based on location and availability of data. All of the sites, excluding CKT (located in Morgan County, KY), have more than 290 weekly NH_4^+ measurements from 1990 to 1998 making them suitable for long-term trend analysis. For more information regarding CASTNet data analysis and validation procedures, see the studies of *Clarke et al.* [1997] and of *USEPA* [1998].

[8] Data on NH_4^+ concentration associated with precipitation were obtained from the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) [*NADP/NTN*, 1999]. NADP/NTN began monitoring in 1978 and currently operates over 200 sites nationwide. The purpose of the network is to collect data on the chemistry and amount of precipitation for monitoring spatial and temporal long-term trends. The precipitation at each station is collected weekly from AeroChem Metrics wet–dry deposition samplers according to strict clean-handling procedures. The precipitation sample is then sent to the Central Analytical Laboratory in Illinois, where it is analyzed for hydrogen (acidity as pH), SO_4^{2-}, NO_3^-, NH_4^+, chloride, and base cations (such as calcium, magnesium, potassium, and sodium) (NADP/NTN). Data validation procedures used at the laboratory are described by *Bowersox* [1984].

[9] It should be noted that both positive and negative biases in NH_4^+ concentration in precipitation have been measured. A positive bias may result from the deposition of NH_3 gas to the open precipitation collector [*Jensen and Asman*, 1995]. This bias will, on average, be greatest in agricultural areas and will likely be positively correlated with ambient temperature. A negative bias, as large as 40% in some cases, has been shown to result from postcollection incorporation of NH_4^+ into microbial biomass [*Ramundo and Seastedt*, 1990; *Lamb and Comrie*, 1993]. This bias is also expected to be larger under warm temperatures. Unfortunately, it is not possible to quantify the net result of these biases. While this source of error may reduce the amplitude of season cycles, due to a net reduction of summer values at most sites, it should not greatly effect the magnitude of long-term trends. At agricultural sites, the competing biases may result in a relatively small net error.

[10] In order to assess the temporal variability in NH_4^+ concentrations associated with aerosols and precipitation across the southeast United States over the period 1990–1998, 10 CASTNet sites and 10 corresponding NADP sites were chosen based on location (distance between CASTNet and NADP sites) and availability of weekly data. For this

Table 1. CASTNet and NADP Sites Used in Weekly and Monthly Analyses of NH_4^+ in Ambient Air and Precipitation[a]

CASTNet Sites					Corresponding NADP Sites					Distance Between Sites (km)
State	County	Identifier	Lat./Long. (°)	Elevation (m)	State	County	Identifier	Lat./Long. (°)	Elevation (m)	
KY	Morgan	CKT	37.92/83.07	455	KY	Rowan	KY35	38.12/83.55	204	47.3
NC	Montgomery	CND	35.26/79.84	198	NC	Rowan	NC34	35.70/80.62	219	86.0
NC	Macon	COW	35.06/83.43	686	NC	Macon	NC25	35.06/83.43	686	0
MS	Yalobusha	CVL	34.00/89.80	134	MS	Yalobusha	MS30	34.00/89.8	134	0.1
GA	Pike	GAS	33.18/84.40	270	GA	Pike	GA41	33.18/84.41	270	0.2
KY	Washington	MCK	37.70/85.05	353	KY	Washington	KY03	37.70/85.05	293	0.1
NC	Avery	PNF	36.10/82.04	1219	NC	Yancey	NC45	35.73/82.12	1987	46.6
AL	Dekalb	SND	34.29/85.97	352	AL	DeKalb	AL99	34.29/85.97	349	0.2
TN	Claiborne	SPD	36.47/83.83	361	TN	Anderson	TN00	35.96/84.29	341	70.0
FL	Liberty	SUM	30.11/84.99	14	FL	Gadsden	FL14	30.55/84.60	60	62.0
					NC	Sampson	NC35	35.02/78.27	41	

[a]KY = Kentucky, NC = North Carolina, MS = Mississippi, GA = Georgia, AL = Alabama, TN = Tennessee, FL = Florida.

time series analysis, an additional NADP site was included (NC35, Sampson County, North Carolina) based on its location within an agricultural region, though this site does not have a corresponding CASTNet site. Table 1 summarizes the CASTNet and NADP sites used in the weekly and monthly analyses of NH_4^+ concentrations associated with aerosols and precipitation. To assess the influence of NH_3 emissions on wet and dry NH_4^+ concentrations, the remaining NADP and CASTNet sites located in the southeast United States which were active during 1997 were added to the analysis. Table 2 lists the CASTNet and NADP sites used in the analysis of NH_3 emissions and NH_4^+ concentrations in this study.

2.2. Data Analysis

[11] In this study, regression analysis, specifically the method of ordinary least squares, was used to identify relationships between dependent and independent variables. This method assumes that the regression errors have constant variance, are uncorrelated with each other in time, and have a normal distribution. Analyses were performed using SAS statistical analysis software.

2.3. Estimating Annual NH_3 Emissions

[12] The first goal of this study is to resolve the spatial variability of NH_3 emissions from agricultural sources in the southeast United States. To develop this regional emissions inventory, emission estimates from all major agricultural sources in the southeast United States were calculated using data from 1997. The sources considered in this inventory include dairy and beef cattle, poultry, swine, horses, and sheep, as well as fertilizer application. County totals are obtained for each source, and NH_3 emission estimates are performed at the county level. For the purpose of this study, NH_3 emissions are assumed to be uniform across the county. This provides a general spatial distribution of estimated NH_3 emissions across the eight-state region. County-level NH_3 emission estimates for each source type are based on the following equation:

$$\begin{aligned} &\mathit{Emission\ rate}\left(kg\ NH_3\ yr^{-1}\right)\\ &\quad = \mathit{Activity\ Data}(\mathit{animal\ population})\\ &\quad \times \mathit{Emission\ Factor}\left(kg\ NH_3\ animal^{-1}\ yr^{-1}\right) \end{aligned}$$

The activity data is simply the number of animals present in each county, and is obtained from the 1997 U.S. Census of Agriculture [*USDA*, 1999]. The emission factors are based on experimental measurements of average emissions per animal (kg NH_3 animal^{-1} yr^{-1}) and mass balance calculations. Most of the experimental emission factors are obtained from Europe, where animal practices may vary significantly from the United States. Furthermore, the NH_3 emissions are sensitive to changes in animal diet, atmo-

Table 2. CASTNet and NADP Sites Used in the NH_3 Emissions Analysis

	State	County	Identifier	Lat./Long. (°)	Elevation (m)
CASTNet	AL	Dekalb	SND	34.29/85.97	352
	FL	Liberty	SUM	30.11/84.99	14
	GA	Pike	GAS	33.18/84.40	270
	KY	Morgan	CKT	37.92/83.07	455
	KY	Washington	MCK	37.70/85.05	353
	MS	Yalobusha	CVL	34.00/89.80	134
	NC	Montgomery	CND	35.26/79.84	198
	NC	Macon	COW	35.06/83.43	686
	NC	Avery	PNF	36.10/82.04	1219
	NC	Carteret	BFT	34.88/76.62	2
	TN	Claiborne	SPD	36.47/83.83	361
	TN	Dekalb	ESP	36.04/85.73	302
NADP	AL	Dallas	AL10	32.46/87.24	58
	AL	Dekalb	AL99	34.29/85.97	349
	FL	Bradford	FL03	29.97/82.20	44
	FL	Citrus	FL05	28.75/82.55	3
	FL	Dade	FL11	25.39/80.68	2
	FL	Gadsen	FL14	30.55/84.60	60
	FL	Sarasota	FL41	27.38/82.28	25
	FL	Brevard	FL99	28.54/80.64	2
	GA	Charlton	GA09	30.74/82.13	47
	GA	Evans	GA20	32.14/81.97	62
	GA	Pike	GA41	33.18/84.41	270
	GA	Tift	GA99	31.52/83.55	107
	KY	Washington	KY03	37.70/85.05	293
	KY	Letcher	KY22	37.08/82.99	335
	KY	Rowan	KY35	38.12/83.55	204
	MS	Hinds	MS10	32.31/90.32	86
	MS	Newton	MS19	32.33/89.17	115
	MS	Yalobusha	MS30	34.00/89.90	134
	NC	Bertie	NC03	36.13/77.17	22
	NC	Macon	NC25	35.06/83.43	686
	NC	Rowan	NC34	35.70/80.62	219
	NC	Sampson	NC35	35.03/78.28	41
	NC	Scotland	NC36	34.97/79.53	132
	NC	Wake	NC41	35.73/78.68	120
	NC	Yancey	NC45	35.74/82.29	1987
	SC	Clarendon	SC06	33.54/80.44	24
	TN	Anderson	TN00	35.96/84.29	341
	TN	Sevier	TN11	35.66/83.59	640
	TN	Haywood	TN14	35.47/89.16	107

Table 3. Published Emission Factors for Livestock (kg NH_3 animal^{-1} yr^{-1})

Source	ECETOC (1994)[a]	EMEP (1996)[b]	*Misselbrook et al.* [2000]	*Bouwman et al.* [1997]	Buijsman (1987)[a]	*Asman* [1992]	UNECE [c]
Dairy Cow	39.5	29.1	26.52	24.8	–	39.7	28.5
Beef Cow	27.8	14.6	6.8	9.5	13.7	23.1	14.3
Pigs	4.25	–	–	4.9	2.8	5.34	–
Sow	–	16.6	5.2	–	–	–	16.43
Finishing pig	–	6.46	4.8	–	–	–	6.39
Poultry	0.19	–	–	0.24	0.26	.24	.37
Laying hen	–	0.38	.45	–	–	–	–
Broiler	–	0.27	.23	–	–	–	.28
Sheep	1.8	1.46	.73	0.77	3.16	1.7	1.34
Horses	11.9	–	–	9.2	9.35	12.1	8.0

[a]Data from *Sutton et al.* [1994].
[b]Data from *Misselbrook et al.* [2000].
[c]Data from *Van Der Hoek* [1998].

spheric temperature and humidity, waste-handling practices, and many other parameters [*Asman*, 1992]. Because of the many uncertainties, it may be difficult to obtain an accurate NH_3 emission estimate. Table 3 is a summary of documented emission factors that were considered in developing the emission inventory for this study. The large variation in estimates illustrates the difficulty in developing precise estimates.

[13] Emission factors were selected for each livestock group including beef and dairy cattle, hogs and pigs, chickens, broilers, turkeys, horses, and sheep. An earlier study and literature review by *Battye et al.* [1994] refined European emission factors based on United States agricultural practices. Their results have been used as a guide to obtain the emission factors employed in this study. The U.S. Census of Agriculture has provided estimates for both beef and dairy cattle; therefore, a unique emission factor was determined for each. *Battye et al.* [1994] recommend 15.19 kg NH_3 animal^{-1} yr^{-1} for beef cattle or "young cattle for fattening." This estimate includes total emissions resulting from animal housing, grazing, manure storage, and land spreading. The recommendation by *Battye et al.* [1994] is based on research and literature reviewed by *Asman* [1992]. Considering these estimates to be somewhat out of date, an average of more recent estimates by *Bouwman et al.* [1997], *Misselbrook et al.* [2000], and *Van Der Hoek* [1998] is used here resulting in an emission factor of 10.2 kg NH_3 animal^{-1} yr^{-1}. A similar approach is used for dairy cattle, taking the average of emission factors given by *Misselbrook et al.* [2000] and *Van Der Hoek* [1998] to obtain 28.04 kg NH_3 animal^{-1} yr^{-1}.

[14] Hogs and pigs are not divided into weight or class categories in the 1997 Census of Agriculture; however, *Van Der Hoek* [1998] suggests that 3 classes can be determined based on the total population of hogs. One can assume that approximately 50% are fattening hogs, 10% are sows, and the remaining 40% are young sows and piglets. Two unique emission factors, 6.39 and 16.43 kg NH_3 animal^{-1} yr^{-1}, are derived for fattening hogs and sows respectively. The factor 16.43 for sows includes a correction for young sows and piglets that account for 40% of the population. Therefore, to estimate total NH_3 emissions from a general hog population, 50% of the population was multiplied by 6.39 and 10% of the population by 16.43. This equates to an average emission factor of 4.84 kg NH_3 per hog, which has proved to be a satisfactory estimate based on recent studies at a commercial hog farm by *McCulloch* [1999]. His study estimated total NH_3 emissions from hog facilities to be in the range 3.4–6.9 kg NH_3 animal^{-1} yr^{-1} [*McCulloch*, 1999]. *Battye et al.* [1994] proposed emission factors for sheep, broilers, and laying hens older than 20 weeks based on the study of *Asman* [1992]. These estimates have been refined based on new experimental data, and the updated values are employed in this study [*Van Der Hoek*, 1998].

[15] For the remaining animal groups (pullets 13–20 weeks, pullets less than 30 weeks, and turkeys) and fertilizer application, the emission factors proposed by *Battye et al.* [1994] are used. Table 4 lists the estimated emission factors for various nitrogen fertilizers, based on total U.S. consumption in 1993. A summary of emission factors and corresponding 1997 emissions estimates for all domestic livestock and fertilizer application in the southeast are given in Table 5. Based on the emission factors and agricultural census data, both the relative

Table 4. Emission Factors, U.S. Consumption (1993), and Nitrogen Content of Selected Fertilizers [*Battye et al.*, 1994]

Fertilizer	U.S. Consumption (mg[a]) (1993)	Nitrogen Content (%)	Emission Factor (kg NH_3/mg N)
N-P-K	8,191,414	11.2	48
Nitrogen Solutions	7,162,419	33.9	30
Ammonium Phosphates	5,813,042	15.5	48
Anhydrous NH_3	3,593,380	82.0	12
Urea	3,247,631	45.9	182
Ammonium Nitrate	1,582,039	33.9	25
Other Straight Nitrogen	944,803	20.0	30
Ammonium Sulfate	718,400	21.0	97
Aqua NH_3	271,288	20.4	12
Ammonium Thiosulfate	156,047	12.0	30

[a]1 Mg = 10^3 kg.

Table 5. Emission Factors and Total Emission Estimates for the Southeast United States

Source	Emission Factor (kg NH_3 animal^{-1})	Total Emissions in Southeast (kt)
Beef Cattle	10.2	150.1
Dairy Cattle	28.04	20.5
Horses	8	3.6
Hogs and Pigs	–	62.8
Sows	16.43	–
Fattening Pigs	6.39	–
Sheep	1.34	0.1
Broilers	0.28	174.4
Chickens	–	32.7
Laying Hens	0.37	–
Pullets 13–20	0.269	–
Pullets <13	0.17	–
Turkeys	0.858	21.8
Fertilizer	listed in Table 2	62.9

contribution from each source category (Figure 1) and the spatial distribution of NH_3 emissions were determined (Figure 2).

2.4. Influence of NH_3 Emissions on Atmospheric NH_4^+ Concentrations

[16] Exploratory regression analysis is used to determine relationships between county-scale NH_3 emissions and within-county observed annual average concentrations of NH_4^+ associated with aerosols and annual volume-weighted average concentration of NH_4^+ in precipitation (Table 2). This analysis is performed for the year 1997 to correspond with the NH_3 emission inventory described above.

2.5. Weekly NH_4^+ Concentration Analysis

[17] A statistical analysis is performed to investigate and model trends in NH_4^+ concentration associated with aerosols and precipitation based on correlation with meteorological parameters such as surface temperature, relative humidity, wind speed, and wind direction. Measurements of NH_4^+ concentration in aerosols from 10 CASTNet sites across the southeast United States and NH_4^+ concentration in precipitation from 10 neighboring NADP sites were analyzed for the period January 1990 to December 1998. Due to the proximity of each CASTNet and NADP site (Table 1), the same meteorological data were used for both analyses.

[18] To investigate the relationship between ambient and wet NH_4^+ concentration and meteorology, a multiple linear regression model of the following type was employed at all sites [*Walker et al.*, 2000a; *Holland et al.*, 1999; *Buishand et al.*, 1988; *Dana and Easter*, 1987]:

$$\log(C_t) = a_0 + [a\cos(2\pi t/52) + b\sin(2\pi t/52)] + ct + d_n x_n + e_i \quad (1)$$

where $\log(C_t)$ refers to the natural log-transformed weekly concentration of ambient NH_4^+ (μg m^{-3}) or wet NH_4^+ in precipitation (mg L^{-1}) at time t weeks from 1 January 1990. Raw data were transformed to help achieve the condition of normality in regression residuals. The second term in model (1) contains sine and cosine functions, which are commonly used to model seasonal cycles in data [*Lynch et al.*, 1995; *Holland et al.*, 1999]. In model (1), the amplitude (A) of the cycle is determined as:

$$A = \sqrt{a^2 + b^2} \quad (2)$$

and the phase angle (ϕ) is determined as:

$$\hat{\phi} = \arctan(b/a) \quad \text{if } a \geq 0$$
$$\hat{\phi} = \arctan(b/a) + \pi \quad \text{if } a < 0. \quad (3)$$

The regression routine calculates p-values for coefficients a and b under the null hypothesis that no cycle is present at frequency $2\pi t/52$ (a = 0 and b = 0). If the p-value for either of the regression coefficients is less than the specified alpha level, the null hypothesis may be rejected. Meteorological parameters (temperature, precipitation amount, relative humidity, wind speed, and wind direction) included in the model are represented by $x_n = x_1, x_2, x_3, x_4$, and x_5. Finally, a_o represents the intercept of the regression line, while the residual (e_i) represents the error in the point prediction of $\log(C_t)$. Only parameters with regression coefficient p-values < 0.1 were considered statistically significant.

2.6. Monthly Mean NH_4^+ Concentration Analysis

[19] Monthly averaged concentrations of NH_4^+ associated with aerosols and monthly volume-weighted average concentrations of NH_4^+ associated with precipitation were also investigated at each CASTNet and NADP site for the period January 1990 to December 1998 (Table 1). For this exercise, two models were applied to all sites in an attempt to better understand the interactions between overall trend, seasonality, and temperature dependence [*Holland et al.*, 1999]. Other meteorological parameters were excluded from this model because monthly averages would perform poorly as predictive parameters. The selected models are:

$$\log(C_t) = a_0 + a\cos(2\pi t/12) + b\sin(2\pi t/12) + ct + e_i \quad (4)$$

$$\log(C_t) = a_0 + bT_t + ct + e_i \quad (5)$$

Model (4) accounts for seasonal variability of C_t at each site, while model (5) accounts for the dependence of C_t on air temperature. Parameters in model (5) are defined as in model (1). In model (5), T represents monthly average temperature while the remaining terms are defined as in model (1). The use of both temperature and seasonality in

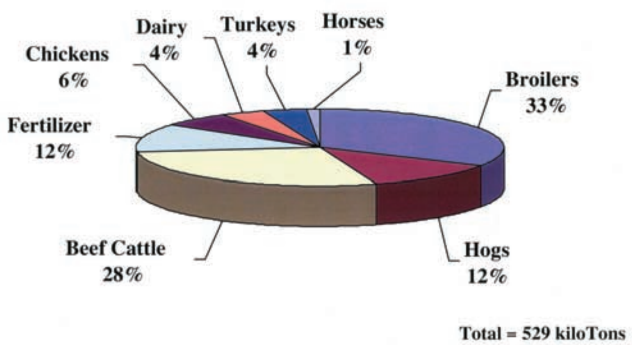

Figure 1. Relative contribution of NH_3 emissions in the southeast United States Source: Nelson, 2000.

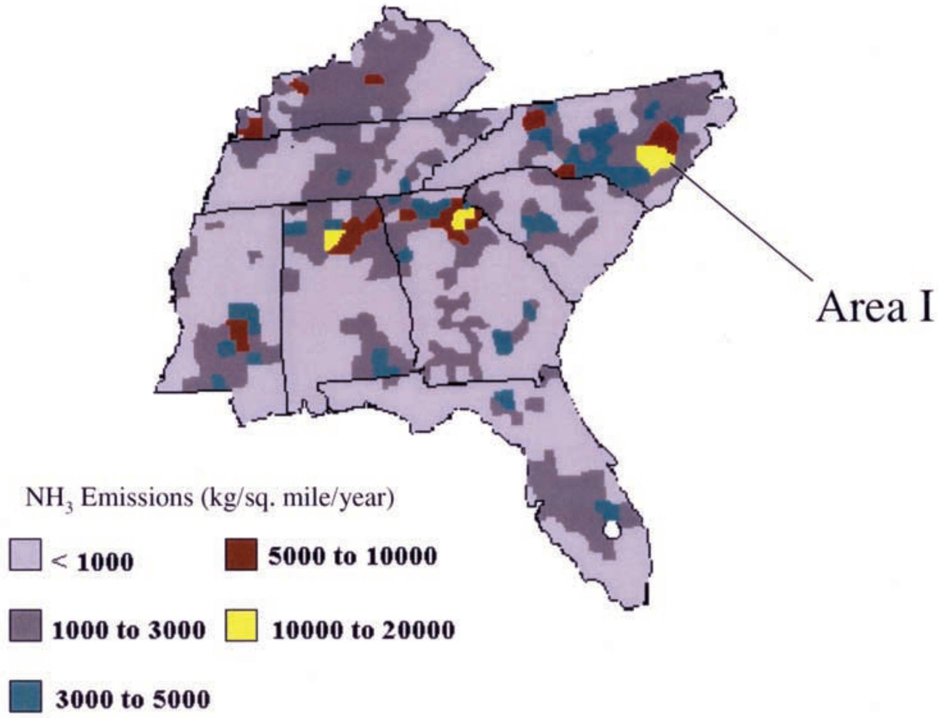

Figure 2. Spatial distribution of NH_3 emissions.

the same model can actually degrade the quality of the model because the two parameters are so highly correlated. Where a trend in NH_4^+ concentration was detected at a particular site, regression models were employed to test for trends in temperature and precipitation volume. This was necessary to determine if the trend in NH_4^+ concentration may have been caused by temporal changes in precipitation volume or temperature.

3. Results

3.1. Spatial Distribution of NH_4^+ Concentrations

[20] Linear regression analysis was performed to investigate the relationship between annual county-scale agricultural NH_3 emission density (Figure 2) and observed annual average NH_4^+ concentrations in aerosols and precipitation within that county during 1997. To improve the normality of regression residuals, NH_4^+ concentrations in aerosols and precipitation, along with emissions, were first natural log-transformed. Regression analyses were performed on transformed variables. Results show that natural log-transformed annual NH_4^+ concentrations associated with aerosols increase with natural log-transformed county annual NH_3 emission density ($R^2 = 0.86$, $p < 0.0001$, $N = 12$) (Figure 3a). The untransformed data show a clear logarithmic increase in NH_4^+ concentrations in aerosols with increasing emissions. This relationship suggests that local NH_3 emissions strongly influence ambient NH_4^+ concentrations, but that there exists a level above which NH_3 emission is no longer the primary source of variability in ambient NH_4^+ concentration. This can be explained by the fact that NH_4^+ aerosol formation is limited by the availability of acid gases in the presence of excess NH_3. Thus, counties with high NH_3 emission densities likely represent areas within which NH_4^+ aerosol formation is no longer NH_3 limited. A similar regression analysis shows that natural log-transformed annual volume-weighted average NH_4^+ concentration in precipitation shows only a very weak positive correlation with natural log-transformed annual NH_3 emission densities within the corresponding county ($R^2 = 0.12$, $p = 0.04$, $N = 28$) (Figure 3b). Perhaps the primary reason for this much weaker relationship is that the incorporation of NH_4^+ into rainfall takes place on a spatial scale greater than the area of individual counties. In most cases, the majority of NH_4^+ observed in rainfall at a particular location originates from relatively distant sources, and the local signal may result from the relatively inefficient process of below-cloud scavenging of NH_3 and NH_4^+ [*Shimshock and De Pena*, 1989]. In general, agricultural NH_3 sources are shown to influence local concentrations of NH_4^+ in aerosols and precipitation both in the United States and Europe [*Marquardt et al.*, 1996; *Erisman et al.*, 1998; *Asman et al.*, 1998; *Aneja et al.*, 2000; *Walker et al.*, 2000a; *Sutton et al.*, 2001a, 2001b].

3.2. Analysis of Weekly NH_4^+ Concentration Associated With Aerosols

[21] Table 6 summarizes the results from model (1) applied to the weekly ambient concentration data at each CASTNet site (Table 1). The R^2 values range from 0.18 at SUM (Liberty County, Florida) to 0.73 at PNF (Avery County, North Carolina). The poor performance of the model at site SUM may be attributed to its location, a Florida site with very little seasonal variability and strong coastal influence.

[22] Precipitation amount was the most significant parameter at 7 out of 10 sites, having a negative regression coefficient ranging from -0.0045 to -0.005. This inverse relationship has also been reported in other studies [*Prado-*

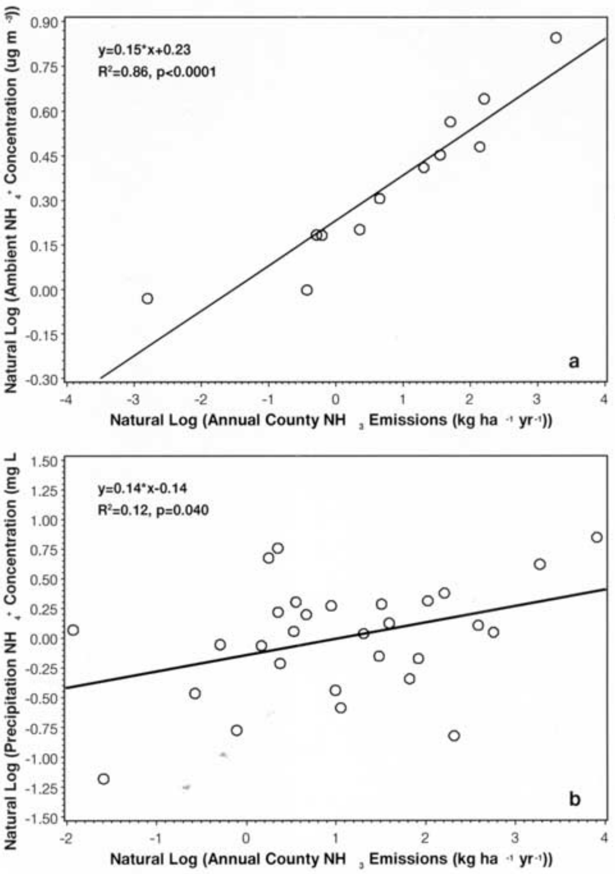

Figure 3. Natural log-transformed annual average ambient NH_4^+ concentrations ($\mu g\ m^{-3}$) versus natural log-transformed annual county NH_3 emission density (kg NH_3 ha^{-1} yr^{-1}) (a) and natural log-transformed annual volume-weighted average NH_4^+ concentration in precipitation (mg L^{-1}) versus natural log-transformed annual county NH_3 emission density (kg NH_3 ha^{-1} yr^{-1}) (b).

Fiedler, 1990; *Walker et al.*, 2000a]. Relative humidity is found to be a significant ($p < 0.05$) parameter at 6 sites. The positive regression coefficient (0.001–0.005) suggests that higher relative humidity leads to increased concentrations of NH_4^+ associated with aerosols. Therefore, increased water vapor in the atmosphere near sources of NH_3, lead to higher concentrations of NH_4^+ [*Andersen et al.*, 1999; *Asman*, 1994; *Warneck*, 1988; *McMurry et al.*, 1983]. Moreover, at high relative humidity (>62%) ammonium nitrate is less likely to dissociate into HNO_3 and NH_3 [*Stelson and Seinfeld*, 1982]. Wind speed is a significant parameter in the model at 6 sites and is negatively correlated to NH_4^+ concentrations. Low wind speeds often coincide with stable conditions and limited dispersion whereas lower concentrations are often characterized by higher wind speeds and increased mixing throughout the boundary layer [*Arya*, 1999]. Temperature is also a significant model predictor at half of the sites. Finally, wind direction is significant at 6 CASTNet sites. From the regression coefficients, however, it is difficult to interpret the effect wind direction has at any particular site and is better illustrated by showing NH_4^+ concentration associated with aerosols relative to wind direction for CASTNet site CND located in Montgomery County, North Carolina (Figure 4) [*Yamamoto et al.*, 1995]. From this plot, it is clear that higher concentrations of NH_4^+ associated with aerosols occur when the wind is from the E and SE. Indeed, Area I, previously defined as an area of elevated NH_3 emissions, is located to the E/SE of site CND (see Figure 2).

[23] The model with the best fit (R^2 = 0.73) was for CASTNet site PNF located in Avery County, North Carolina. A plot of measured and predicted concentrations at site PNF, for the period January 1990 to December 1998, shows that the model follows the general seasonal trends of the ambient NH_4^+ concentration but fails to predict the peaks (Figure 5). Temperature was found to be the most statistically significant parameter in this model ($p < 0.05$), and an analysis of temperature and NH_4^+ over the 9-year period

Table 6. Results of Regression Model (1) for CASTNet Weekly NH_4^+ Ambient Concentrations at Each Site

CASTNet Site	N[a]	Parameters[b,c]	MSE	R^2
CKT	224	T, WS, P	0.0115	0.68
CND	369	P, WS, WD, RH	0.0146	0.58
COW	449	P, T, WD	0.0218	0.65
CVL	380	P, RH, WD, WS	0.0250	0.43
GAS	398	P, T	0.0176	0.51
MCK	368	P, RH, T	0.0183	0.54
PNF	413	P, T, RH	0.0204	0.73
SND	262	WD, WS, RH, P	0.0232	0.41
SPD	394	P, WD, WS	0.0155	0.63
SUM	426	RH, WS, WD	0.0347	0.18

[a]Number of observations.
[b]Statistically significant parameters ($p < 0.05$).
[c]T = temperature, WS = wind speed, P = precipitation amount, WD = wind direction, RH = relative humidity.

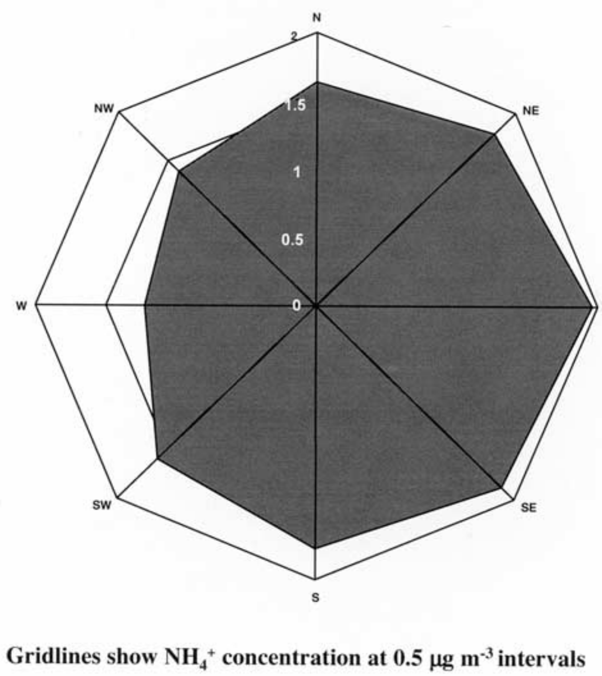

Figure 4. Ambient NH_4^+ concentration versus wind direction at CASTNet site CND located in Montgomery County, North Carolina.

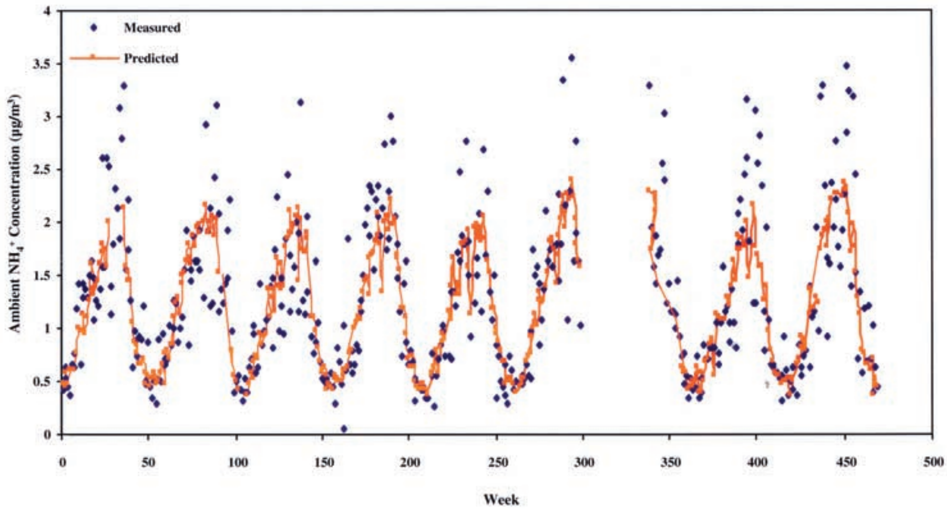

Figure 5. Measured versus predicted (model (1)) weekly NH_4^+ concentration in air at CASTNet site PNF located in Avery County, North Carolina. Week 0 corresponds to the first week in January 1990.

revealed that the concentration of NH_4^+ peaks during the summer when temperatures are warm. This relationship is to be expected based on the fact that atmospheric NH_4^+ is primarily a product of NH_3 reacting with acids formed in the atmosphere, such as H_2SO_4, HNO_3, and HCl and the formation of these acids depends on the availability of hydroxyl radical (OH) and O_3 in the atmosphere, which peak during the summer months [*Seinfeld*, 1986]. Furthermore, biogenic NH_3 emissions from soils and animal waste storage and treatment lagoons [*Aneja et al.*, 2000] are in part driven by temperature, where a 10°C increase in temperature approximately doubles the rate of ammonification [*Addiscott*, 1983].

3.3. Analysis of Monthly Mean NH_4^+ Concentration Associated With Aerosols

[24] Monthly mean concentrations of NH_4^+ associated with aerosols were modeled to test for the general trend and seasonal trends over the period 1990–1998. Models (4) and (5) were applied to each selected CASTNet site and the highest R^2 value [$R^2 = 0.86$, model (4)] was again found at site PNF. The results for all CASTNet sites (Table 1) selected are summarized in Table 7.

[25] The models of monthly NH_4^+ concentrations perform much better than those of weekly concentrations with R^2 values ranging from 0.14 to 0.86 for seasonality model (4) and 0.07–0.81 for temperature model (5). If we remove the Florida site SUM, the average R^2 for models (4) and (5) are 0.69 and 0.63, respectively. This means that 69% of the variability in NH_4^+ concentration in aerosols is explained by seasonality factors in model (4), while 63% of the variability is explained by temperature dependence in model (5). These results suggest that most of the variation in NH_4^+ concentrations in aerosols can be explained by temperature or seasonal effects. In general, Table 7 shows that R^2 values are consistently higher for the seasonality model (4), which is to be expected based on the strong interaction between temperature and seasonality inherent in this model. However, to account fully for spatial variations in ambient NH_4^+ concentrations and deposition, one must also consider such variables as ambient concentrations of NH_4^+ precursors, surface roughness, and vegetation properties, which have large spatial and temporal variability [*Asman*, 1994].

[26] Statistically significant trends were evident at 2 of the CASTNet sites over the period 1990–1998. A positive trend was present at site SUM in Sumatra County, Florida ($p = 0.05$). However, this site performed poorly in the above analyses, so this result may be inaccurate. A negative trend was observed at site SPD, located in Claiborne County, Tennessee. Mean concentrations of NH_4^+ associated with aerosols fell from approximately 2.14 in 1990 to 1.88 in 1998 ($p = 0.06$). Trends in mean surface temperature were investigated at site SPD (Claiborne County, Tennessee). However, no statistically significant trend was present over the period 1990–1998 suggesting that temperature is likely not responsible for the decreasing trend in NH_4^+ concentration at these sites.

3.4. Analysis of Weekly NH_4^+ Concentration Associated With Precipitation

[27] Model (1) was run for all NADP sites (Table 1) to select the best parameter fit. The results from this analysis were low with R^2 ranging from 0.13 to 0.31. Due to the

Table 7. Results of Regression Models for CASTNet Monthly Mean NH_4^+ Concentration Associated With Aerosols

CASTNet Site	Temperature R^2 [a]	Seasonal R^2 [b]	Trend p-value[b]
CKT	0.73	0.69	0.86
CND	0.62	0.67	0.74
COW	0.81	0.83	0.47
CVL	0.35	0.45	0.57
GAS	0.66	0.67	0.89
MCK	0.61	0.73	0.09 (−)[c]
PNF	0.81	0.86	0.76
SND	0.38	0.54	0.22
SPD	0.69	0.77	0.06 (−)
SUM	0.07	0.14	0.05 (+)[d]

[a]Model (5).
[b]Model (4).
[c]Indicates negative trend.
[d]Indicates positive trend.

Table 8. Results of Regression Models for NADP Monthly Mean NH_4^+ Concentration in Precipitation[a]

NADP Site	Temperature R^{2b}	Seasonal R^{2c}	Trend p-value[c]
KY35	0.34	0.56	0.004 (+)[d]
NC34	0.15	0.36	0.12
NC25	0.30	0.54	0.52
MS30	0.12	0.30	0.07 (+)
GA41	0.11	0.25	0.56
KY03	0.32	0.62	0.97
NC45	0.35	0.48	0.78
AL99	0.45	0.49	0.26
TN00	0.30	0.50	0.48
FL14	0.06	0.31	0.23
NC35	0.46	0.50	<0.0001 (+)

[a]KY = Kentucky, NC = North Carolina, MS = Mississippi, GA = Georgia, AL = Alabama, TN = Tennessee, FL = Florida.
[b]Model (5).
[c]Model (4).
[d]Indicates positive trend.

poor performance of this model, no conclusions could be made regarding the relationship between weekly NH_4^+ concentration in precipitation and local meteorological parameters. However, an investigation of monthly mean volume-weighted NH_4^+ concentration in precipitation proved to be more insightful.

3.5. Analysis of Monthly Mean NH_4^+ Concentration Associated With Precipitation

[28] Models (4) and (5) were applied to all NADP sites (Table 1) using monthly mean volume-weighted NH_4^+ concentration in precipitation. The results for each NADP site are summarized in Table 8. The seasonality dependence in model (4) resulted in R^2 values ranging from 0.25 to 0.62, while the temperature dependence in model (5) had an R^2 range of 0.06–0.46. In general, models performed better when applied to monthly rather than weekly values. Volume-weighted averaging of weekly values to generate monthly averages tends to smooth the noise present in weekly values, resulting in higher R^2 values associated with monthly models. The site FL14, located in Gadsden County, Florida, had the worst overall performance. This site is located near CASTNet site SUM, which also performed poorly in the NH_4^+ associated with aerosols analysis. The exact reason for the poor performance of these sites is not known, although their proximity to the coast and possible

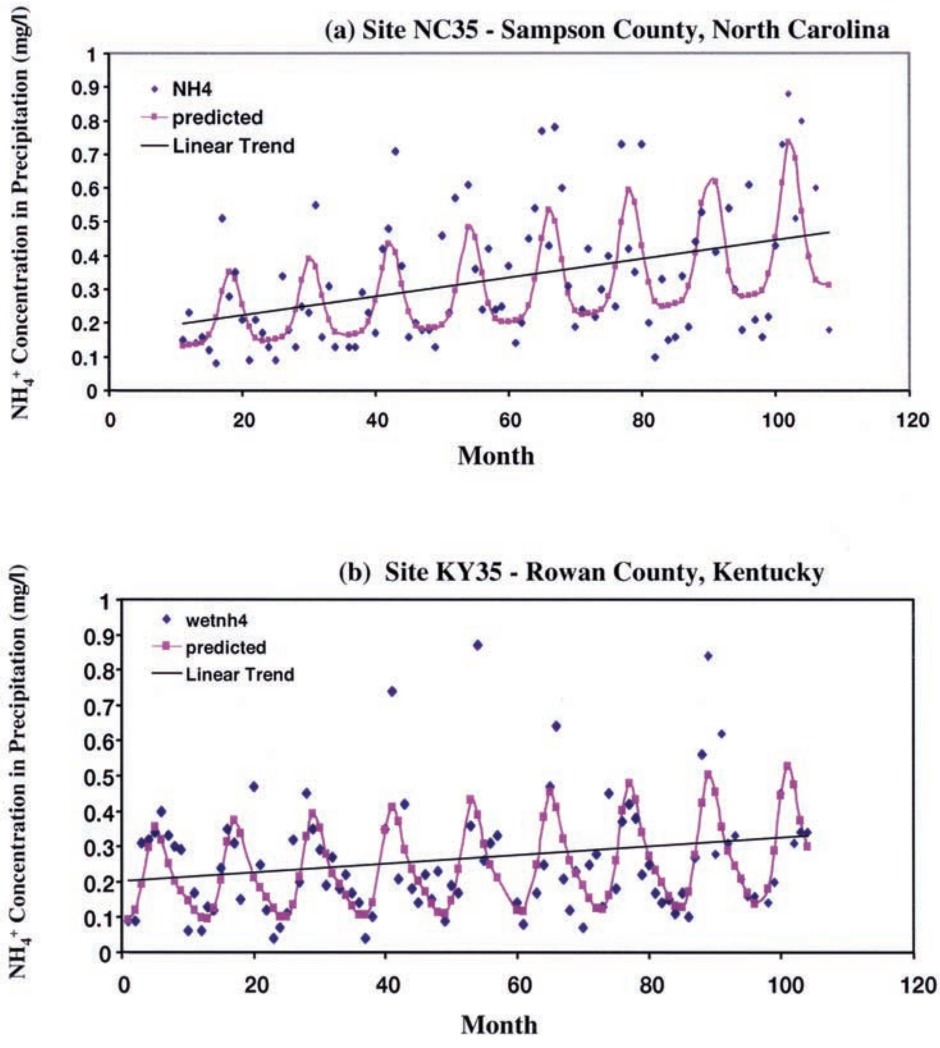

Figure 6. Trends at NADP sites NC35, Sampson County, North Carolina and KY35, Rowan County, Kentucky over the period 1990–1998 where month 0 corresponds to January 1990.

overriding influences of sea and land breezes are considered to be contributing factors [*Arya*, 1999]. Two of the NADP sites showed a statistically significant positive ($p < 0.05$) trend for the period 1990–1998; KY35, located in Rowan County, Kentucky, and NC35, located in Sampson County, North Carolina. The linear trends for these two sites are shown in Figure 6.

[29] At the Sampson County, North Carolina site (NC35), located in Area I (see Figure 2), monthly mean volume-weighted NH_4^+ concentration in precipitation rose from approximately 0.2 mg L^{-1} in 1990 to 0.48 mg L^{-1} ($p < 0.0001$) in 1998. The dramatic increase in NH_4^+ wet deposition is also documented by *Aneja et al* [1998] and *Walker et al.* [2000a, 2000b]. Their findings reveal that no significant increasing trends in temperature or precipitation are present for the period 1990–1996; therefore, meteorology is likely not responsible for the increasing trend in NH_4^+. *Walker et al.* [2000b] go on to relate the increasing trend in NH_4^+ in precipitation to an increase in local NH_3 emissions caused by swine facilities. In fact, the hog population in North Carolina rose from approximately 2 million to 10 million hogs between 1990 and 1998, with 50% of the statewide population, and hence the emissions, located in the concentrated area surrounding Sampson County. The natural summertime peaks of NH_4^+ concentration in this area are further enhanced by the presence of waste from hogs.

[30] At site KY35, the average NH_4^+ concentration in precipitation rose from approximately 0.2 mg L^{-1} in 1990 to 0.35 mg L^{-1} ($p = 0.004$) in 1998. Analyses of precipitation and temperature revealed no statistically significant trend in either variable over the 8-year span; therefore, temperature and precipitation amount do not appear to be responsible for the increasing trend in NH_4^+ concentration in precipitation found at NADP site KY35. The reason behind the increasing NH_4^+ trend at KY35 is less obvious than at NC35, because Rowan County, Kentucky, has an average NH_3 flux of only 131 kg NH_3 km^{-2} yr^{-1}. Based on CASTNet meteorological data, the prevailing wind at site KY35 is from the SW; however, concentrations of NH_4^+ in aerosols, and presumably NH_4^+ in precipitation, are slightly higher when the wind is out of the N. The trend observed at this site likely results from increasing upwind NH_3 source strengths over the period.

4. Conclusions

[31] This study provides insight into the coupling between NH_3 emissions and NH_4^+ concentrations associated with both aerosols and precipitation and how environmental parameters affect these relationships. Regression modeling shows that counties with relatively higher agricultural NH_3 emissions exhibit higher annual average concentrations of NH_4^+ associated with aerosols though the influence on NH_4^+ concentration in precipitation is much less clear. Analysis of NH_4^+ concentration in aerosols at the various CASTNet sites revealed that temperature, precipitation amount, and relative humidity are the most statistically significant ($p < 0.05$) parameters in predicting the weekly concentrations of NH_4^+. Wind speed and direction were also statistically significant ($p < 0.05$) at several CASTNet sites, but the results were less consistent. Investigation into NH_4^+ concentration in precipitation yielded temperature as a statistically significant ($p < 0.05$) parameter. Trends over the period 1990–1998 revealed a slight decrease in ambient NH_4^+ concentration at CASTNet site SPD, Claiborne County, Tennessee (2.14–1.88 μg m^{-3}, $p = 0.06$), while positive trends in NH_4^+ concentration in precipitation were evident at NADP sites NC35, Sampson County, NC (0.2–0.48 mg L^{-1}, $p < 0.0001$) and KY35, Rowan County, Kentucky (0.2–0.35 mg L^{-1}, $p = 0.004$)) over the period 1990–1998. Analyses of NH_4^+ emissions and deposition in the United States are complicated by a lack of data on ambient levels of NH_3 gas and the complex interrelations among NH_3 gas, HNO_3 gas, and SO_4^{2-} and NO_3^- particulate components.

[32] Results from this study provide additional evidence that agricultural NH_3 source strengths are seasonally dependent. Modeling exercises which use annual emissions estimates derived from factors such as those presented in this study should attempt to account for this effect. Furthermore, seasonality in NH_3 emissions may translate to seasonality in ammonium nitrate and ammonium sulfate aerosol concentrations in some areas. This relationship, however, is confounded by seasonality in nitric and sulfuric acid concentrations. Further research (both measurement and modeling) is warranted to investigate such dynamic NH_3/aerosol relationships and the influence of NH_3 on total $PM_{2.5}$. The general form of the parametric models presented here may be useful in examining the temporal variability in NH_3, SO_4^{2-}, NO_3^-, HNO_3, and sulfuric acid to better characterize the seasonal nature of NH_3/NH_4 partitioning.

[33] **Acknowledgments.** This research was funded in part by the Division of Air Quality, North Carolina Department of Environmental and Natural Resources (NC DENR) (contract No. EA01001. We acknowledge assistance of the Air Quality Research Program at North Carolina State University for technical discussions and review of the manuscript; North Carolina Department of Agriculture and Consumer Services for providing animal populations and fertilizer data, and Ms. P. Aneja, for assisting in manuscript preparation. Financial support does not constitute an endorsement by NC DENR of the views expressed in the article, nor does mention of trade names of commercial or noncommercial products constitute endorsement or recommendation for use.

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ATTACHMENT 30

Memo to: NC Division of Water Resources, and NC Environmental Management Commission

From: Dr. Michael A. Mallin, Research Professor, Center for Marine Sciences, University of North Carolina Wilmington, Wilmington, NC, 28409

Date: February 9, 2015

Subject: Comment on the proposed reclassification of the lower Cape Fear River and Estuary to Class Sc-Swamp (Sw) classification.

1) I am very supportive of the statement in the reclassification proposal that states that any further municipal point sources will require the highest level of treatment in North Carolina. I would ask for more specifics regarding industrial discharges – at the least setting some limits on biochemical oxygen demanding agents such as biochemical oxygen demand (BOD), ammonia, total nitrogen (TN) and total phosphorus (TP).

2) An important statement that needs to be clarified is found in the narrative standards where it states that DO should not be less than 5.0 mg/L except that "swamp waters, poorly flushed tidally influenced streams or embayments, or estuarine bottom waters may have lower values if caused by natural conditions" . The issue that requires clarification is who decides, and by what criteria, if such a deviation is caused by "natural" conditions.

3) The proposed CFR reclassification does not adequately address non-point contributions of BOD or nutrients (which lead to BOD increases). If focus on non-point sources potentially contributing to oxygen depletion is continued to be addressed by on-going water quality programs; based on the summer blue-green algal blooms that occurred annually from 2009-2012, this approach has been inadequate and will continue to be inadequate.

4) In the lower Cape Fear River and Estuary, peer-reviewed research published in *Limnology and Oceanography* has demonstrated that BOD is driven by a number of biological and chemical factors (Mallin et al. 2004; Tables 4, 5 and 6) see the following:

- Chlorophyll *a* (the principal measure of algal bloom strength) has been positively correlated with BOD5 in the mainstem river at Lock and Dam #1 ($r = 0.55$, $p = 0.0001$), Browns Creek ($r = 0.45$, $p = 0.007$), Hammond Creek ($r = 0.45$, $p = 0.004$), Great Coharie Creek ($r = 0.51$, $p = 0.001$), Colly Creek ($r = 0.64$, $p = 0.0001$), Barnards Creek ($r = 0.37$, $p = 0.040$), Motts Creek ($r = 0.42$, $p = 0.020$), and Smith Creek ($r = 0.57$, $p = 0.0009$). I note that Browns, Hammond, Barnards and Smith Creeks drain directly into the mainstem river or estuary, while Colly and Great Coharie creeks drain into the lower Black River, a major 5th order tributary of the 6th order Cape Fear River.
- TN has been positively correlated with either BOD5 or BOD20 or both in the 5th-order Northeast Cape Fear River ($r = 0.30$, $p = 0.02$), the Black River ($r = 0.45$, $p = 0.0003$), Hammond Creek ($r = 0.47$, $p = 0.0003$), Six Runs Creek ($r = 0.54$, $p = 0.0005$), Great

Coharie Creek (r = 0.44, p = 0.006), Little Coharie Creek (r = 0.52, p = p = 0.0008), and Colly Creek (r = 0.54, p = 0.0005).

- TP has been positively correlated with either BOD5, BOD20 or both in the Northeast Cape Fear River (r = 0.34, p = 0.008) the Black River (r = 0.33, p = 0.010), Browns Creek (r = 0.40, p = 0.012), Hammond Creek (r = 0.42, p = 0.009), Six Runs Creek (r = 0.49, p = 0.002), Great Coharie Creek (r = 0.66, p = 0.0001), and Colly Creek (r = 0.39, p = 0.015).
- Chlorophyll *a* represents algal blooms, which upon death and decomposition become highly labile sources of BOD. Nutrients drive BOD in two ways: directly and indirectly. A peer-reviewed article in *Ecological Applications* by Mallin et al. (2004) showed that for streams in the Black and Northeast Cape Fear River basins, inputs of dissolved phosphorus directly stimulate BOD5 and BOD20, as well as natural bacteria abundance (the direct driver of BOD). The data also showed that inputs of dissolved nitrogen (nitrate ammonium, and urea) significantly stimulate algal growth, which in turn significantly stimulates BOD. Thus, the correlation between nutrient loading and BOD is not surprising.

5) The proposed reclassification is based on the Bowen (2009) model predicting DO concentrations in the lower Cape Fear River Estuary

- The Bowen model concludes that further reduction of current point sources would have little effect on DO concentrations – I will accept the model's conclusions on that matter.
- But, Bowen's model shows that reducing nutrient, carbon and BOD loads from the incoming rivers, creeks and wetlands by 30% and 70% would <u>increase</u> median DO from 5.6 mg/L to 5.85 and 6.2 mg/L, respectively – and this assumes sediment oxygen demand (SOD) stays the same regardless of reductions! See Bowen (2009) pages 6-4, 6-8, and 6-22 in particular for more on this topic.
- Assuming that such BOD load reduction would similarly reduce SOD, than the model says summer DO violations would decrease from 45% to 22% violations (30% reduction case), down to 7% (with 50% reduction) and down to only 1% violations (70% reduction case).
- I further note that SOD cannot simply be considered "natural" only. A year-long study of several tidal creeks in New Hanover County was published in the peer-reviewed journal *Hydrobiologia* (MacPherson et al. 2007). Results demonstrated that chlorophyll *a* concentrations were positively correlated with SOD (r = 0.35, p < 0.05), as well as BOD5 (r = 0.50, p < 0.05).

6) I note that Bowen does not discuss non-point source pollution sources specifically.

7) Yet, non-point runoff plays a major role in the middle to lower basin of the mainstem Cape Fear River, from crop agriculture, urban runoff and some livestock production. In the lower Cape Fear system I note that livestock waste pollution and crop agriculture are the predominant non-point nutrient and BOD sources in the Black and Northeast Cape Fear River basins.

8) Livestock manures as waste inputs were *not even mentioned* in Bowen's model! However, 2012 livestock counts for Brunswick, Pender, Duplin, Sampson, Cumberland

and parts of Bladen and Onslow Counties (Cape Fear lower watershed) are as follows (information for counties that are partially within the basin, Bladen and Onslow, are estimates):

- Hogs: approximately 5,000,000
- Turkeys: approximately 21,500,000
- Broiler chickens: > 122,000,000
- Other chickens: > 870,000
- Cattle: approximately 72,000

(from NCDA website September 2014)

Livestock wastes are clearly the largest source of BOD-forcing pollutants in the Cape Fear Basin – and remain virtually unregulated (i.e. no required streamside buffers, no required control of ammonia off gassing, etc.).

9) Industrialized swine farms (CAFOs) are a source of large-scale chronic nitrogen and phosphorus loading to nearby soils and receiving water bodies, nutrients which have been directly correlated to BOD in the blackwater streams and rivers of the Cape Fear Basin (Mallin et al. 2006). An peer-reviewed analysis by Cahoon et al. (1999) published in *Environmental Science and Technology* found that vast quantities of nitrogen and phosphorus feed are imported into the watershed annually to feed swine, poultry, and cattle in production facilities (CAFOs), which in turn annually load large quantities of nutrients as waste into the watershed. This analysis found that for the Cape Fear River basin alone, CAFOs produce 82,700 tons of nitrogen and 25,950 tons of phosphorus annually into this watershed. Thus, N and P enter the state as animal feed from elsewhere, but much of it leaves the livestock as manure (or carcasses) and enters soils or waters of the Coastal Plain.

10) Finally, swine waste lagoons, as well as lagoons servicing egg-laying poultry CAFOs, produce copious amounts of ammonia to the atmosphere; NC Division of Air Quality estimates a swine ammonia emission factor of 9.21 kg/hog-year. 9.21 x 5,000,000 head of swine = 46,050,000 kg or 46,050 metric tons of ammonia released to the airshed of the Cape Fear River basin (and coastal ocean) <u>per year</u>, much of which comes to earth within 60 miles of the source (Walker et al. 2000; Costanza et al. 2008). Ammonia is well-known in the environmental engineering literature to exert an oxygen demand (nitrogenous BOD) on waters – that is why it is regulated in wastewater discharges (Clark et al. 1977). Efforts need to be made to control this major source of oxygen-demanding wastes to the Cape Fear system as well.

11) Clearly, non-point sources of BOD, nitrogen, and phosphorus entering the waters of the lower Cape Fear River system are very large and lead to reduced dissolved oxygen levels.

I conclude that the proposed reclassification, as it stands, will be inadequate to produce or maintain proper dissolved oxygen concentrations in the lower Cape Fear River and Estuary due to the lack of attention to non-point sources of nutrients and BOD. The source of much of this pollution is industrial livestock production, along with unknown inputs from traditional agriculture, and some urban runoff in the Fayetteville and Wilmington areas. **Any**

proposed reclassification of the lower Cape Fear River and Estuary must include strong language specifically aimed at reducing such non-point sources of pollution.

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ATTACHMENT 31

PERGAMON

Atmospheric Environment 35 (2001) 1903–1911

ATMOSPHERIC ENVIRONMENT

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Atmospheric nitrogen compounds II: emissions, transport, transformation, deposition and assessment

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Abstract

The Atmospheric Nitrogen Compounds II: Emissions, Transport, Transformation, Deposition and Assessment workshop was held in Chapel Hill, NC from 7 to 9 June 1999. This international conference, which served as a follow-up to the workshop held in March 1997, was sponsored by: North Carolina Department of Environment and Natural Resources; North Carolina Department of Health and Human Services, North Carolina Office of the State Health Director; Mid-Atlantic Regional Air Management Association; North Carolina Water Resources Research Institute; Air and Waste Management Association, RTP Chapter; the US Environmental Protection Agency and the North Carolina State University (College of Physical and Mathematical Sciences, and North Carolina Agricultural Research Service). The workshop was structured as an open forum at which scientists, policy makers, industry representatives and others could freely share current knowledge and ideas, and included international perspectives. The workshop commenced with international perspectives from the United States, Canada, United Kingdom, the Netherlands, and Denmark. This article summarizes the findings of the workshop and articulates future research needs and ways to address nitrogen/ammonia from intensively managed animal agriculture. The need for developing sustainable solutions for managing the animal waste problem is vital for shaping the future of North Carolina. As part of that process, all aspects of environmental issues (air, water, soil) must be addressed as part of a comprehensive and long-term strategy. There is an urgent need for North Carolina policy makers to create a new, independent organization that will build consensus and mobilize resources to find technologically and economically feasible solutions to this aspect of the animal waste problem.

Keywords: Ammonia; Nitrogen compounds; Emissions; Effects; Transport; Transformation; Swine operations and abatement

1. Background

Nitrogen is perhaps the most important nutrient governing the growth and reproduction of living organisms. Nitrogen compound emissions also have a profound effect on air quality. Two major needs that drive the contemporary perturbations of the nitrogen cycle are the seemingly insatiable human appetite for energy, leading to the emission of nitrogen oxides into the atmosphere, and the need for food to sustain growing numbers of people all over the world, leading to the agricultural emission of ammonia. Once released into the atmosphere by either man-made (anthropogenic) or natural sources,

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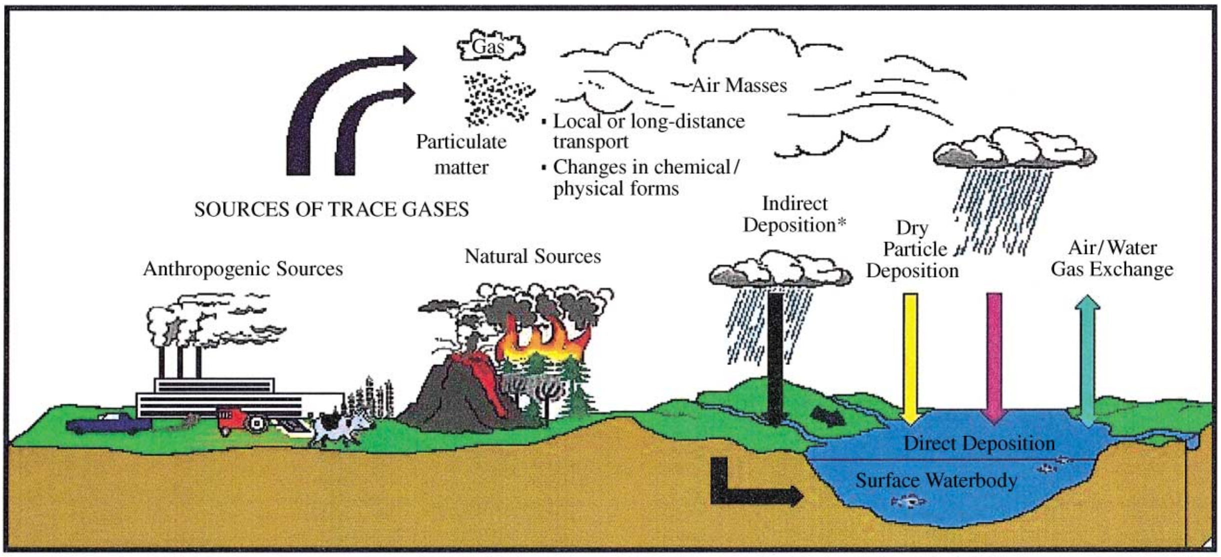

Fig. 1. Atmospheric emissions, transport, transformation and deposition of trace gases.

these nitrogen compounds can undergo several different processes such as transformation due to atmospheric reactions (e.g. gas-to-particle conversion), transport associated with wind, and finally wet and dry deposition (Fig. 1). All of these processes can perturb the environment with a host of beneficial and detrimental effects, such as increased crop yields from nitrogen loading or decreased visibility from increased aerosol production. Table 1 represents the current global estimates for sources and sinks of several key nitrogen species (oxidized nitrogen compounds, nitrous oxide, and ammonia). Scientists have focused recently on the oxidized species of nitrogen ($NO_x = NO + NO_2$) and their role as precursors to ozone (O_3) formation, and the reduced species (NH_x = ammonia + ammonium + amines) and their role in nitrogen enrichment and eutrophication of aquatic ecosystems. Nitrous oxide (N_2O), while contributing to ozone destruction in the stratosphere, is relatively inert in the troposphere and therefore has negligible consequences in tropospheric photochemistry, but does contribute to climate change as a greenhouse gas (Warneck, 1988).

2. Emissions

Fossil fuel combustion has increased to meet growing energy demands. The global amount of fossil fuel use per person (Fig. 2) has increased by more than a factor of 6 over the last 75 years. At the same time, scientists have synthesized nitrogen-based fertilizers to enhance crop development and to maximize production on limited land space. Before the mass production of fertilizers, it can be assumed that there was an approximate balance between the relatively unreactive molecular nitrogen (N_2 comprises approximately 80% of air) in the atmosphere, which was naturally converted to forms used by plants and animals, and the amount of nitrogen returned to the atmosphere via natural processes (Delwiche, 1970). Currently, however, the global production of fertilizer is approximately 100 million metric tons of nitrogen yr^{-1}, compared to approximately one million metric tons only 40 years ago (The Fertilizer Institute, 2000). The results of increased fertilizer and power production have reached a point where the scientific community has major concerns about the fate of the nitrogen produced.

Estimates of NH_3 emissions and the contribution from different source categories given in Figs. 3 and 4 show that hog operations are responsible for a larger percentage of the nitrogen budget in North Carolina than they are in the US as a whole. The relatively large NH_3 contribution from hog operations in North Carolina as compared to the US as a whole can be explained by Fig. 5 which shows the growth of the hog industry during the last two decades. Data presented at the workshop (Fig. 6) revealed that NH_3 emissions in a six-county (Bladen, Duplin, Greene, Lenoir, Sampson, Wayne) area of North Carolina that maintains the state's densest and largest population of hogs (Fig. 7) increased significantly during the same time period that the hog operations increased (Walker et al., 2000). Mean NH_3 emissions from hog operations increased 316% between 1982–1989 and 1990–1997; 84% of the growth from all sources (i.e., hogs, fertilizer, cattle, turkeys, broilers, chickens) can be

Table 1
Global atmospheric budgets of NO_x, N_2O, and NH_3

Source or sink	NO_x[a]	N_2O[b]	NH_3[c]
	(Tg N yr^{-1})[d]		
Fossil fuel combustion	21	0.5	2
Biomass burning	8.0	0.4	5
Sea surface	< 1.0	5.7	13
Domestic animal waste	—[e]	1.6	32
Human excrement	—	—	4
Lightning	8	—	—
NH_3 oxidation by OH	1	0.6	—
Stratospheric input	0.5	—	—
Soil emissions	20.2	10.7	19
Other[f]		6.3	
Total sources[g]	59	26	75
Wet deposition	12–42	—	46
Dry deposition	12–22	—	10
Stratospheric sink	—	19.3	—
NH_3 oxidation by OH	—	—	1
Atmospheric accumulation	—	3.5	—
Total sinks	59	19.3	57

[a]Source: Levine (1991).
[b]Source: Bouwman et al. (1995); stratospheric sink from Houghton et al. (1995).
[c]Source: Schlesinger and Hartley (1992).
[d](1 Tg = 10^{12} g).
[e](—) indicates insignificant or unavailable terms.
[f]Includes adipic and nitric acid production, nitrogen fertilizer, land use change and other small sources.
[g]It is accepted that wet and dry NO_x deposition should total the sum of NO_x sources and that the apparent difference between total NH_3 sources and sinks represents uncertainties in identified budget terms, not atmospheric accumulation.

attributed to the increase in number of hogs (Walker et al., 2000). Fig. 6 also shows that the ammonium ion concentration $[NH_4^+]$ in precipitation collected at a deposition sampling site in Sampson County also increased throughout this period.

3. Atmospheric behavior

Atmospheric ammonia (NH_3) emissions have garnered increased interest in the past few years, due in part to the detrimental effects of excess nitrogen deposition to nutrient sensitive ecosystems (Aneja et al., 1998; Nihlgard, 1985; van Breemen, 1982). Moreover, NH_3 is the most prevalent gaseous base found in the atmosphere, and is, therefore, fundamental in determining the overall acidity of precipitation (Warneck, 1988), cloudwater (Li and Aneja, 1992), and atmospheric aerosols (Lefer et al., 1999). New gaseous ammonia instruments for monitoring and research are currently in advanced stages of development (Erisman et al., 1999). The ecological impact of atmospheric NH_3 deposition may be substantial as reduced nitrogen species are thought to be the most biologically available of nitrogen species in N-limited coastal and estuarine ecosystems (Paerl, 1997). In the atmosphere, NH_3 reacts primarily with acidic species to form ammonium sulfate, ammonium nitrate or ammonium chloride, or it may be deposited to the earth's surface by either dry or wet deposition processes.

The spatial scale of a particular NH_3 source's contribution to atmospheric nitrogen deposition is governed in part by the gas-to-particle conversion rate of NH_3 to NH_4^+. Because of the short lifetime of NH_3 in the atmosphere ($\tau = 1$–5 days or less) (Warneck, 1988), low source height, and relatively high dry deposition velocity (Asman and van Jaarsveld, 1992), a substantial fraction (20–40%) will likely deposit near its source. However, ammonium (NH_4^+) aerosols, with atmospheric lifetimes on the order of $\tau = 1$–15 days (Aneja and Murray, 1998; Aneja et al., 2000) will tend to deposit at larger distances downwind of sources. Ammonia emissions from animal operations contribute substantially to atmospheric nitrogen loading and may contribute the same order of magnitude as emissions of NO in some parts of the world (Steingröver and Boxman, 1996); highlighting the need for new sustainable technologies for intensively managed animal production.

4. Effects

Although nitrogen is a critical nutrient for the survival of micro-organisms, plants, humans and animals, it can cause detrimental effects when concentrations reach excessive levels (Paerl, 1997; Erisman et al., 1998). Fig. 8 (Gundersen, 1992) illustrates this point by showing how an ecosystem responds to increased N loadings. The horizontal line is a crop which receives no atmospheric N deposition, and as indicated by the vertical axis, has a stable index of productivity. However, as N is initially added to the system, the index of productivity steadily increases to the point of diminishing returns, where any additional N loading actually reduces productivity (Schlesinger, 1997). In addition to the productivity concerns of aquatic and terrestrial ecosystems, oxidized and reduced N compounds each play a specialized role in degrading human health and its welfare. Some of the consequences associated with elevated concentrations and depositions of both oxidized and reduced N species are:

1. Respiratory disease caused by exposures to high concentrations of:
   1.1. Tropospheric ozone.

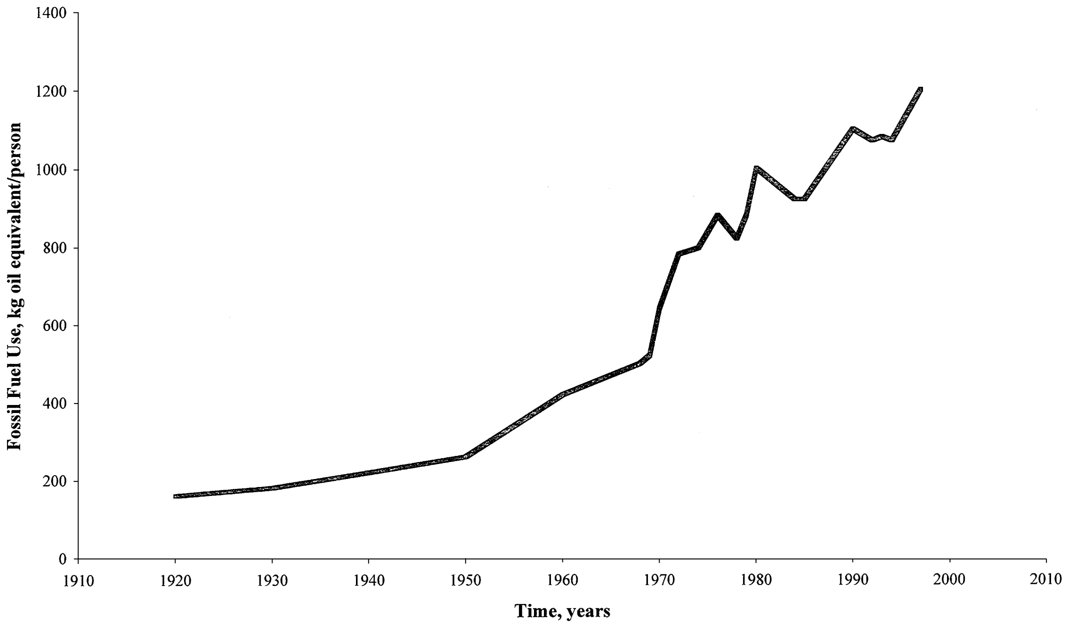

Fig. 2. Trends in global fossil fuel use per person (source: Galloway, 1988).

  - Other photochemical oxidants.
  - Fine particulate aerosol (e.g., PM 2.5).
  - Direct toxicity of NO_2 (on rare occasions).
2. Nitrate contamination of drinking water.
3. Eutrophication, harmful algal blooms and decreased surface water quality.
4. Climatic changes associated with increases in nitrous oxide (greenhouse gas).
5. Nitrogen saturation of forest soils (Erisman et al., 1998).

5. Abatement

Air quality issues associated with intensively managed animal agriculture are now being addressed in Europe and Canada under several initiatives and in consultation and partnership with stakeholders. Emission inventories for several different pollutants including atmospheric nitrogen compounds are maintained by federal governments. For example, the new Air Pollution Protocol for Europe has set reduction targets to be achieved by 2010

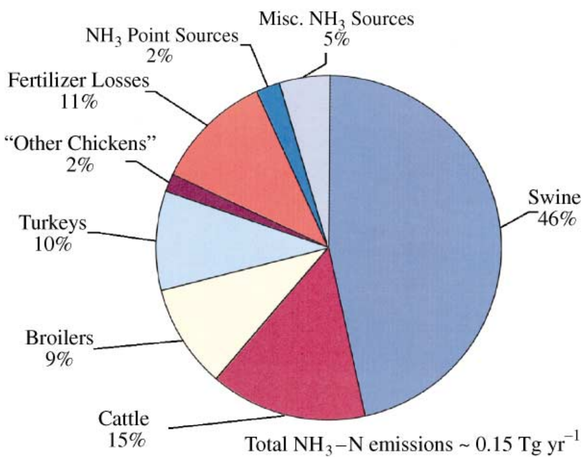

Fig. 3. Percent of ammonia–nitrogen from various sources in North Carolina for 1996 (source: Aneja et al., 1998a).

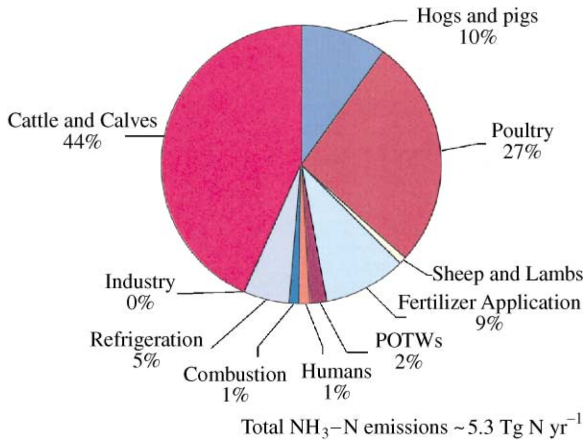

Fig. 4. Relative contribution of ammonia–nitrogen emissions in the US from different source categories (source: Battye et al., 1994).

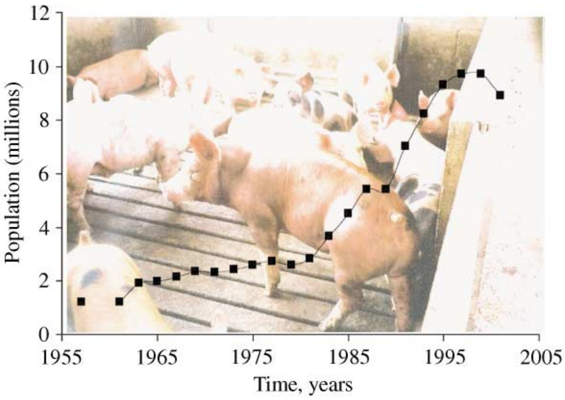

Fig. 5. Hog population in North Carolina (source: North Carolina Agricultural Statistics, 2000).

as compared to 1980 emission levels, for the following: $SO_2 = 63\%$, $NO_x = 41\%$, $VOC = 40\%$, $NH_3 = 17\%$ (http:www.unece.org/press/99env11e.htm). Moreover, in Europe the focus of environmental effects related research is primarily on acidification, eutrophication, biodiversity and groundwater pollution, and the use of critical loads to the ecosystem which accounts for atmospheric deposition pathways.

A "Livestock Environmental Initiative" was launched in Canada during December 1999 in which the livestock industry is working in partnership with the federal government to address environmental concerns through research and development of technology, and for acceleration of technology assessment and transfer. Air quality, including greenhouse gases, is a priority area of concern. Experts from a broad cross-section of government, business and industry, the academic community, environmental groups and non-government organizations have evaluated available information. These options will be reviewed and analyzed to determine the actions needed to reduce emissions.

Improvements in air quality from implementation of the Clean Air Act Amendments (CAAA) of 1990 or other efforts (e.g., Southern Appalachian Mountain Initiative, SAMI) are likely to receive widespread attention only if a target pollutant in question is regulated under the CAAA. At this time, emissions of atmospheric ammonia, ammonium, and organic nitrogen compounds (N_{org}) are not federally regulated, thus minimizing the benefits that might result from the CAAA. In North Carolina, under state law, ammonia is regulated as a toxic air pollutant (15 NCAC 2D.1104(a)(4)).

6. Research needs

The workshop highlighted areas which require further research in North Carolina and elsewhere, such as the further refinement of emission estimates, the role of ammonia and factors that contribute to gas to particle conversion processes (PM_{fine}) in rural/urban and regional areas, computer models to quantify and simulate impacts of deposition, and the establishment of a full scale and continuing monitoring program. Results of the RADM (Regional Acid Deposition Model) and progress that has been made with adapting this model to ammonia deposition were presented at the conference. However, if the main processes and characteristics, specifically

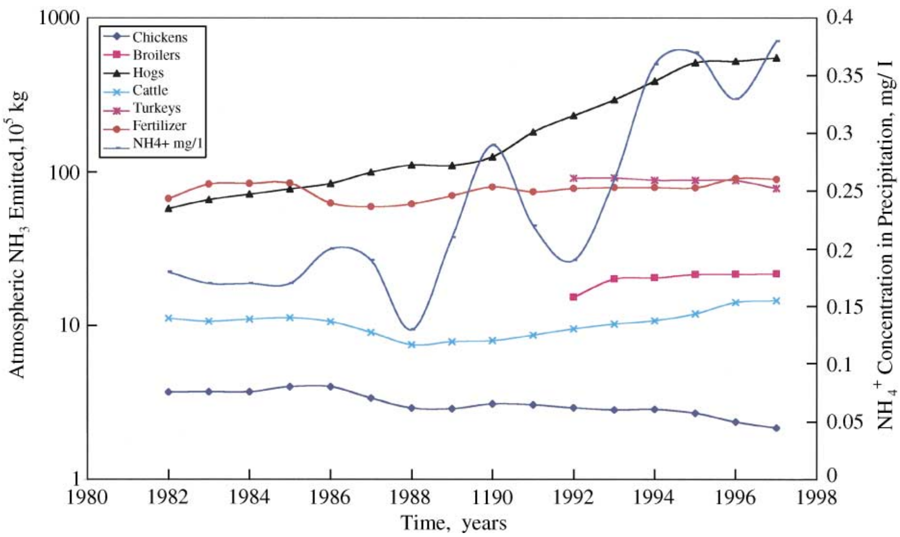

Fig. 6. NH_3 emission estimates by source type in the six North Carolina counties (Bladen, Duplin, Greene, Lenoir, Sampson, Wayne), and annual volume-weighted NH_4^+ concentration in precipitation at Sampson County, North Carolina (source: Walker et al., 2000).

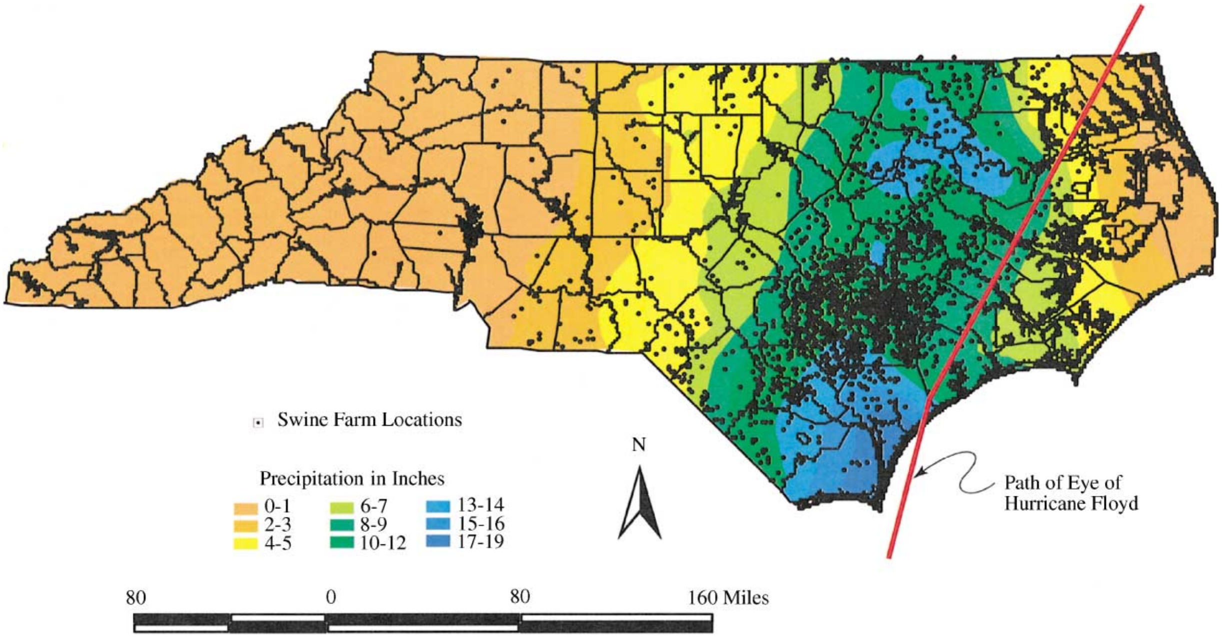

Fig. 7. Map of North Carolina indicating hog sites, rainfall totals associated with Hurricane Floyd (14–16 September 1999), and track of storm.

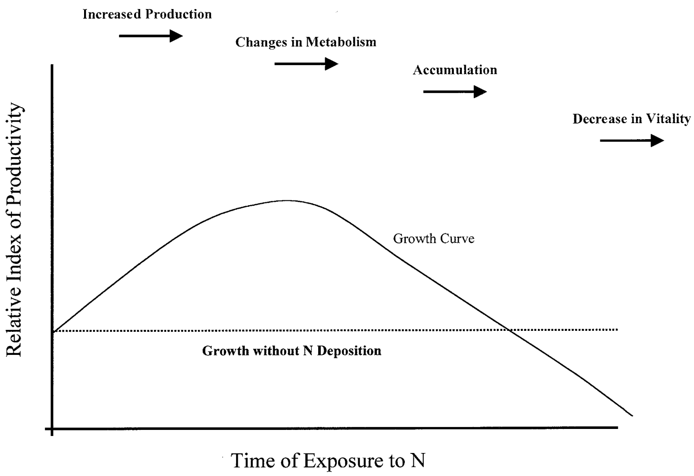

Fig. 8. Hypothetical growth curve for an ecosystem, given different lengths of exposure to nitrogen (source: Gundersen, 1992).

concerning dry deposition, are to be described, then the current grid scaling (20 × 20 km) of this model is still too coarse. Further, a targeted monitoring program in North Carolina needs to be established which includes emissions and both wet and dry deposition at several different land use types. The data collected during this program can then be used to support the modeling effort and assess its performance. A monitoring program will also assist in evaluating any future regulatory policy, which still remains one of the most complicated issues facing North Carolina and the Nation today. Additional research needs are:

1. A detailed understanding of the cycling of atmospheric reduced and oxidized nitrogen compounds, their linkage with emissions of sulphur dioxide (SO_2) and volatile organic compounds (VOCs), and subsequent oxidation products, their spatial and temporal distributions, and their contribution to the chemical composition of aerosols.

Table 2
A comparison of physical and chemical characteristics among the US, North Carolina, and the Netherlands[a]

Parameter	United States	North Carolina	Netherlands
Total land and water area	9,629,000 km^2	136,000 km^2	42,000 km^2
Land area	9,159,000 km^2	126,000 km^2	34,000 km^2
NC coastal plain land area (where majority of hog operations are located)		45,333 km^2	
Inland water area	470,000 km^2	10,000 km^2	6,000 km^2
People	270,312,000 30 km^{-2}	7,651,000 61 km^{-2}	15,731,000 463 km^{-2}
Swine (1996)	56,124,000	9,300,000	14,400,000
Total cattle (1996)	101,656,000	1,100,000	4,412,000
Income from animal agriculture	92.4 billion dollars yr^{-1}	5.7 billion dollars yr^{-1}	4.4 billion dollars yr^{-1}
1995 NO_x emissions	21,600,000,000 NO_x yr^{-1} 6,560,000,000 kgN yr^{-1}	570,000,000 kgNO_x yr^{-1} 173,000,000 kgN yr^{-1}	518,000,000 kgNO_x yr^{-1} 158,000,000 kgN yr^{-1}
1995 NH_3 emissions	2,730,000,000 kgNH_3 yr^{-1} 2,250,000,000 kgN yr^{-1}	155,000,000 kgNH_3 yr^{-1} 127,000,000 kgN yr^{-1}	152,000,000 kgNH_3 yr^{-1} 125,000,000 kgN yr^{-1}

[a]Source: http://www.cia.gov. http://www.minlnv.nl/international/stat/factagricult1.htm. http://www.usda.gov/news/pubs/fbook98/ch3g.htm. http://www.epa.gov/ttn/chief/trends97/browse.html.

2. Need to know the contribution of atmospheric deposition of ammonia/ammonium to estuarine and coastal N loading.
3. Need to better understand the ecological effects of ammonia/ammonium as a new N source causing eutrophication of N-sensitive waters.

7. Summary and conclusions

Although North Carolina faces many challenges regarding nitrogen issues, the problem is not limited to North Carolina. Many of the issues which face politicians, farmers, citizens, and international researchers are similar. Therefore, much can be gained through collaboration and exchange of ideas. For example, a comparison between various factors which influence the emission and deposition of total fixed nitrogen (Table 2) in the Netherlands and the US reveals striking similarities. Although land size and human and animal populations differ, the estimates for NO_x and NH_3 emissions, income from agriculture, and inland water areas (adjusted for coastal districts in NC) are all very similar (Table 2). Due to the many similarities and the fact that North Carolina's rapid growth in animal husbandry started almost 2 decades later than the Netherlands, North Carolina can significantly benefit from their experiences.

The current technology used in North Carolina to manage the hog waste is known as the Lagoon and Spray System, which consists of an exposed waste lagoon to store the waste (~98% liquid) and mechanisms through which the waste is periodically sprayed onto the crops as a nutrient source. The technology can be subdivided into four distinct processes (Fig. 9), all of which release NH_3 to the atmosphere: Production houses; Waste Storage and Treatment Systems (Aneja et al., 2000) (Fig. 10); Land application i.e., spraying; and Biogenic Emissions

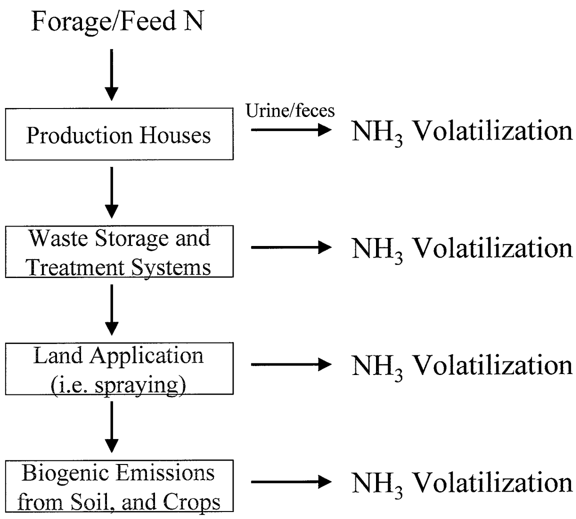

Fig. 9. Major routes for NH_3 emissions from intensively managed animal operations in North Carolina, USA.

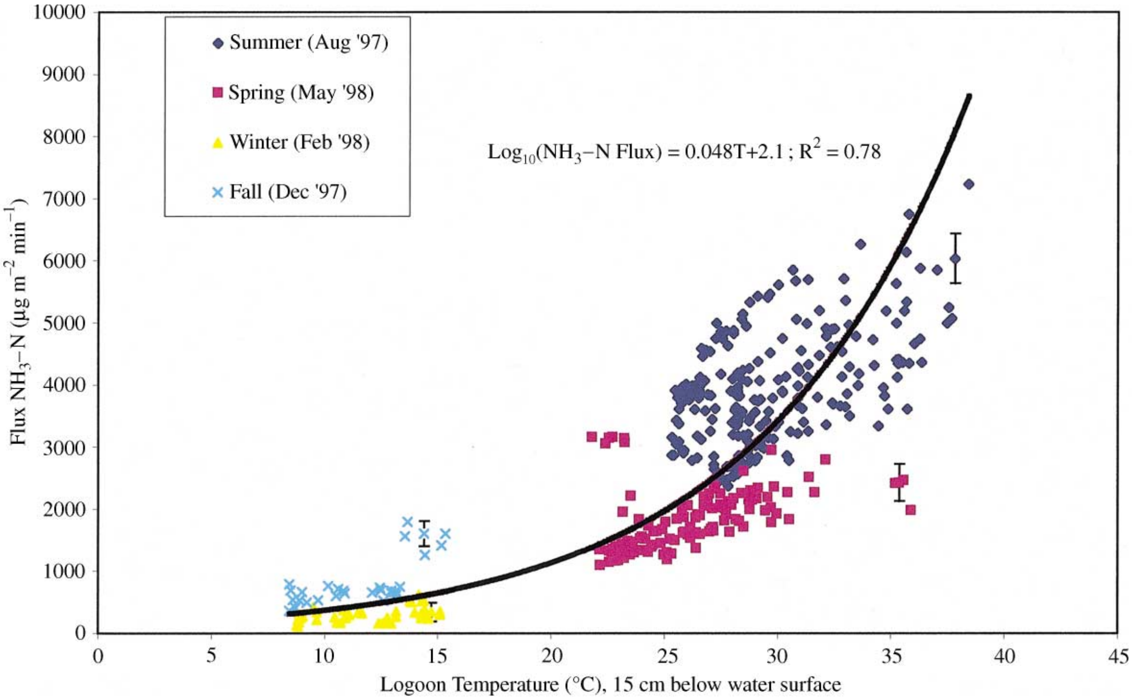

Fig. 10. North Carolina lagoon ammonia–N flux versus lagoon surface water temperature. pH of lagoon: 7–8 pH units, total Kjeldahl nitrogen (TKN) in lagoon: 500–750 mg-N^{-1}. Vertical bars represent one standard deviation (source: Aneja et al., 2000).

from Soil and Crops. Current estimates of NH_3 emissions in North Carolina from hogs alone, utilizing an emission factor (20.3 lb of NH_3 hog^{-1} yr^{-1}) determined by Battye et al. (1994) are: 1994, ~ 195 t of NH_3 d^{-1}; 1996, ~ 258 t of NH_3 d^{-1}; 1999, ~ 264 t of NH_3 d^{-1} (where t = metric tons, and d = day). The lagoon and spray system requires continuous attention due to its susceptibility to flooding, the potential for release of waste to nearby water sources, and also due to odor issues.

The lagoon system recently gained renewed national attention in the aftermath of Hurricane Floyd (15–16 September 1999). The eye of the storm passed over the most intensively managed animal husbandry sites in North Carolina (Fig. 7). The storm resulted in the death of approximately 3 million chickens and turkeys, 880 cattle and 30,000 hogs with many of the carcasses floating in the flood waters; 50 animal operations with waste lagoons were flooded, allowing millions of gallons of animal waste to be spilled into flood waters; and 24 municipal wastewater treatment plants were flooded (WRRI News, 1999). The environmental consequences of this disaster, not yet fully known, include nitrogen release from lagoons and wastewater treatment plants.

Sustainable solutions must be found for managing the animal waste problem in North Carolina. As part of that process, all aspects of environmental issues (air, water, soil) must be addressed as part of a comprehensive and long-term strategy. There is an urgent need for North Carolina policy makers to create a new, independent organization that will build consensus and mobilize resources to find technologically and economically feasible solutions to this aspect of the animal waste problem.

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ATTACHMENT 32

CESSPOOLS OF SHAME

How Factory Farm Lagoons and Sprayfields Threaten Environmental and Public Health

Author
Robbin Marks

Natural Resources Defense Council and the Clean Water Network
July 2001

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TABLE OF CONTENTS

EXECUTIVE SUMMARY

Animal waste from large factory farms is threatening our health, the water we drink and swim in, and the future of our nation's rivers, lakes, and streams. This report documents the public health and environmental risks associated with the use of the lagoon and sprayfield system, which is commonly used by many types of factory farms to dispose of animal waste. The problems with lagoons and sprayfields described in this report are documented through scientific studies, records of pollution events, and victims' accounts of their experiences.

Lagoons and Sprayfields of the Largest Companies Pollute the Environment

Multi-million dollar corporations control many factory farms. The factory farms owned or controlled by these corporations are plagued with pollution problems. Lagoons at many of these operations have broken, failed, or overflowed, leading to major fish kills and other pollution incidents. Operators have sprayed waste in windy and wet weather, on frozen ground, or on land already saturated with manure. More and more, local communities and environmental groups are looking to the courts to remedy environmental violations.

Lagoons and Sprayfields Threaten Public Health

People living near factory farms are placed at risk. Hundreds of gases are emitted by lagoons and the irrigation pivots associated with sprayfields, including ammonia (a toxic form of nitrogen), hydrogen sulfide, and methane. The accumulation of gases formed in the process of breaking down animal waste is toxic, oxygen consuming, and potentially explosive, and farm workers' exposure to lagoon gases has even caused deaths. People living close to hog operations have reported headaches, runny noses, sore throats, excessive coughing, respiratory problems, nausea, diarrhea, dizziness, burning eyes, depression, and fatigue.

The pathogenic microbes in animal waste can also infect people. Water contaminated by animal manure contributes to human diseases such as acute gastroenteritis, fever, kidney failure, and even death. Nitrates seeping from lagoons and sprayfields have contaminated groundwater used for human drinking water. Nitrate levels above 10 mg/l in drinking water increase the risk of methemoglominemia, or blue baby syndrome, which can cause deaths in infants, and contamination from manure has also been linked to spontaneous abortions. Moreover, the practice of feeding huge quantities of antibiotics to animals in subthereapeutic doses to promote growth has contributed to the rise of bacteria resistant to antibiotics, making it more difficult to treat human diseases. Scientists recently found bacteria with antibiotic resistant genes in groundwater downstream from hog operations.

The Lagoons and Sprayfields Harm Water Quality

Lagoons and sprayfields pose a grave danger to the water we use for drinking and swimming. Lagoons filled with manure have spilled and burst, dumping thousands and often millions of gallons of waste into rivers, lakes, streams, and estuaries. In addition, the impact of runoff from sprayfields can be severe over time since manure is often over-applied or

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misapplied to cropland and pastures. There are also often cumulative effects from sprayfield runoff within local watersheds because multiple large-scale feedlots cluster around slaughterhouses. Watersheds as far as 300 hundred miles away are also affected by the atmospheric deposition of ammonia that is emitted from lagoons and sprayfields.

Lagoons and sprayfields are often located in close proximity to waterways and floodplains, which increases the likelihood of ecological damage. Lagoon spills and leaks and runoff from sprayfields have killed fish, depleted oxygen in water, contaminated drinking water, and threatened aquatic life. In many cases, lagoons leak because they are not lined, but leakage may even occur with the use of clay liners, with seepage rates as high as millions of gallons per year. How much a lagoon or sprayfield seeps depends, in part, upon where it is sited. In many places, lagoons and sprayfields have been permitted for places where groundwater can be threatened, such as over alluvial aquifers and in locations with shallow groundwater tables. The lagoon system also depletes groundwater supplies by using large quantities of water to flush the manure into the lagoon and spray it onto fields.

Alternative Approaches to the Lagoon and Sprayfield System Exist but Are Rarely Used by Factory Farms

A wide range of alternatives to the lagoon and sprayfield system currently exist, which illustrates that it is not the lack of other options that is driving factory farms to rely almost exclusively on the lagoon and sprayfield system. Instead, factory farms continue to use this polluting system because they have been allowed to use farmland, rural waterways, and air as disposal sites for untreated wastes. Alternative approaches include sustainable agriculture practices that prevent pollution, such as management intensive rotational grazing, hoop houses, and composting. Alternative technologies that treat the wastewater, including anaerobic digestion, wetlands treatment, and sequencing batch reactors also mitigate some of the risks to surface water, groundwater, air, and public health.

Recommendations

Despite the growing body of evidence that the lagoon and sprayfield system pollutes the environment in numerous ways, the Environmental Protection Agency's (EPA) proposed technology rules under the Clean Water Act would allow the riskiest lagoons to continue to operate and also allow new lagoons to be built. Instead, EPA should ban new lagoons and sprayfields from being built, and phase-out existing systems. The agency should encourage new concentrated animal feeding operations to use sustainable animal production systems. In addition, EPA's final regulations should include controls that address all air, surface water, and groundwater pollution that can contaminate our lakes, streams, and coastal waters, including ammonia, bacteria, viruses, heavy metals, salt, antibiotics, and other toxins.

CHAPTER 1

FACTORY FARM POLLUTION IS A GROWING PROBLEM

American livestock production has changed dramatically over the past sixty years. Like other agricultural enterprises, raising animals has been influenced by new technologies and scientific advancements. But the forces that have had the greatest impact on the business are intensive confinement and the conglomeration of small farms into large corporations. These two changes have had grave impacts on the ecosystems and human communities that surround livestock farms.

Intensive livestock operations first appeared in the 1940s with poultry production.[1] In the egg production segment of the industry, the shift went from chicken houses with bedding to bird confinement in cages. In the swine industry, farmers made a shift from pasture-based and open-lot or production systems to totally controlled confinement. The dairy industry replaced stanchions with free-stall barns. All of these moves to greater confinement decreased or eliminated the need for bedding. While this reduction in bedding materials reduced production costs, it created new problems in disposing manure. To address this problem, producers began adding water to manure and handling it like it was a liquid or pumpable slurry.

It is now common for intensive livestock operations to raise thousands–and sometimes even hundreds of thousands–of animals that produce enormous quantities of manure. A single hog produces two to four times the amount of waste as a human produces, while a dairy cow produces 23 times the waste of a human. In total, these animals generate 220 billion gallons of waste each year. (See Table 1-1.) In our country, 130 times more animal waste is produced than human waste.[2]

Currently, most swine and many dairy and egg-laying poultry concentrated animal feeding operations (CAFOs) in the United States collect the waste produced by their animals with scrapers, flushing systems, or gravity flow gutters, and then store the wet livestock manure in vast open-air pits. Producers use a variety of lagoon systems for liquid manure, including anaerobic lagoons, aerobic lagoons, and temporary storage bins.[3] These lagoons have a size as great as six to seven-and-a-half acres and can contain as much as 20 to 45 million gallons of wastewater.[4] In North Carolina, a facility of 2,500 swine may generate 26 million gallons of lagoon liquid, close to one million gallons of lagoon sludge, and 21 million gallons of slurry. (See Table 1-2.) The operation of the lagoon system may differ depending upon the type of animal waste placed in the lagoon. For example, dairy waste may contain fibrous bedding and grit that is separated before the waste and flush water are placed in the lagoon.[5] For swine manure, such separation is not customary.

TABLE 1-1
Animal Waste Summary in the United States

Animal Type	Number of Head	Waste Amount Tons/Year	Waste Volume Gallons/Year	Amount of Pounds/Year	Nitrogen Lost to Atmosphere Pounds/Year
Hogs	57.5 million	110 million	27 billion	1.3 billion	960 million
Cattle	99.3 million	750 million	180 billion	8.2 billion	4.1 billion
Poultry	1.3 billion	50 million	12 billion	1.3 billion	530 million
Sheep	7.6 million	3 million	730 million	64 million	46 million
TOTAL	1.5 billion	910 million	220 billion	11 billion	5.7 billion

National totals are compiled using 1997 Census of Agriculture data.
Source: Environmental Defense, Animal Waste Summary, http://www.hogwatch.org/maps/index_wherethehogsare.html; Pollution Locator, Animal Waste, http://www.scorecard.org/env-releases/aw/us.tcl#summary (visited May 15, 2001).

Liquid manure stored in a lagoon is typically sprayed untreated on cropland or pastures through a large sprinkler system. This land-application practice is known as the sprayfield system. While the nutrients in manure can help build and maintain soil fertility when applied at agronomic rates, CAFOs often overapply animal waste. Excess manure can harm crop growth, contaminate soils, cause surface and groundwater pollution, and waste valuable nutrients.

The trend towards confinement as an animal production method has been coupled with domination of the nation's animal production system by large, corporate entities. About 50 large pork producers are responsible for about 45 percent of the industry's product.[6] Industry officials predict that their market domination will rise to 75 percent within the next few years.[7] The trend in industrialized animal production has meant that more animals are being raised in fewer operations. In the pork industry, for example, the number of hog farms has fallen from 600,000 to less than 100,000 over the past fifteen years, while the number of hogs produced has stayed about the same.[8] In Iowa, the nation's number one hog producing state, from 1998 to 1999, 17 percent of the hog producers exited from the business.[9]

EPA NOW HAS THE OPPORTUNITY TO MOVE BEYOND THE LAGOON AND SPRAYFIELD SYSTEM

For more than twenty years, the Clean Water Act has specified that CAFOs are industrial point sources of pollution that must obtain Clean Water Act permits before discharging into lakes, rivers, and streams. However, EPA has failed to implement and enforce these statutory requirements. For example, according to EPA, approximately 13,000 operations should be permitted under existing EPA regulations, yet only an estimated 2,520 CAFOs (19 percent) are actually covered under either a general or an individual permit.[10]

The Clean Water Act also specifies that dischargers must meet technology standards, known as effluent guidelines, for their discharges. Effluent guidelines specify wastewater treatment technology, monitoring, and discharge requirements for specific industrial sources.

TABLE 1-2

Average Swine Waste Generated Annually by Different Types of North Carolina Facilities with 2,500 Swine[i]

Production Unit[ii]	Animal Unit	Animal Unit Equivalent Live Weight (pounds)	Lagoon Liquid[i] (gallons per animal unit/year)	Lagoon Sludge[iv] (gallons per animal unit/year)	Slurry[v] (gallons per animal unit/year)
Feeder-to finish	Per head capacity	135	2,317,500	82,500	1,877,500
Farrow-to weanling	Per active sow	433	8,007,500	195,000	6,595,000
Farrow-to feeder	Per active sow	522	9,652,500	235,000	7,950,000
Farrow-to finish	Per active sow	1,417	26,202,500	955,000	21,585,000

i. Swine feeding operations with 2,500 swine weighing over 55 pounds each are considered to be CAFOs (if they also meet the other operational requirements of the Clean Water Act). Many CAFOs are significantly larger.
ii. Assumes 400-pound sow or boar on limited feed, 3-week old weanling, 50 pound feeder pig, 220 pound market hog, and 20 pigs/sow/year.
iii. Estimated total lagoon liquid included total liquid waste plus average annual rainfall surplus falling on lagoon.
iv. Net solids removal prior to lagoon input.
v. Six month accumulation of waste, urine excess water usage; does not include fresh water for flushing or lot runoff.
Figures were derived from Table 3-1, *Average Swine Waste Generation Values for Different Production Units*, North Carolina Cooperative Extension Service, College of Agriculture and Life Sciences, North Carolina State University, Certification Training for Operators of Animal Waste Management Systems, AG-538 (April 1996), and were based upon a model by Nicolette Hahn, Water Keeper Alliance.
Note: Citing these figures does not imply endorsement. Instead they are being used for illustrative purposes.

The current effluent guidelines applicable to CAFOs are 25 years old and severely outdated. However, pursuant to a consent decree in settlement of litigation with the Natural Resources Defense Council, EPA has agreed to issue new technology standards by the end of 2002. EPA has authority to consider non-water quality impacts (such as air pollution) as well as water quality concerns in setting technology standards. The NRDC-EPA agreement requires that EPA evaluate a range of non-lagoon systems for CAFOs and study the effects on surface water, groundwater, air quality, and public health of any technology the agency recommends. It also requires the new rules to cover not only manure storage, but also the land application of manure.[11]

These rules provide the best opportunity on the national level in over twenty years to ensure that factory farms protect the environment and public health. In its new effluent guidelines, EPA has the opportunity to require that CAFOs adopt a technology different than the lagoon and sprayfield system. In fact, EPA has a legal obligation to require the best technology economically achievable. The question remains whether EPA will fully account for the environmental and economic harm caused by the present manure storage and application system and recommend an approach that better protects the environment.

The Lagoons and Sprayfields of the Largest Corporations

In 1999, almost 20 percent of the hogs sold in the United States were produced by four corporations: Smithfield Foods, Inc., Contigroup (Continental Grain and Premium Standard Farms), Seaboard Corporation, and Prestage Farms.[1] These companies operate feedlots themselves or under contracts with producers. The contracts state that the corporation owns the animals, but under most state permit programs, the contractor owns the waste. Thus, the contract system allows corporate owners to avoid responsibility for the waste. These corporations have no responsibility to contribute financially to clean-up or pollution control. Many of the largest corporations have taken advantage of the contract system; facilities under their control have had numerous pollution problems, but the companies have often been able to evade responsibility.

SMITHFIELD FOODS, INC.

Smithfield Foods, Inc. is the nation's largest pork producer, with sales of $3.8 billion in 1999, ownership of about 700,000 sows (four times more than the company's biggest competitor), and the capacity to produce 12 million hogs per year.[2] The company processes pork, contracts with smaller producers to raise hogs, and also produces hogs on its own factory farms and those of the company's subsidiaries: Brown's of Carolina, Inc., Carroll's Foods, Inc., Carroll's Foods of Virginia, Inc., Quarter M Farms, and Murphy Family Farms, Inc.[3] There have been numerous pollution problems from slaughterhouses the company operates in North Carolina and Virginia. The company was fined $12.6 million for dumping pollutants from a Virginia slaughterhouse into the Pagan River–a fine that, as of 1997, was the largest ever imposed under the Clean Water Act.[4] A Smithfield slaughterhouse in North Carolina has polluted the Cape Fear River nearly 40 times.[5] However, the company, its subsidiaries, and contractors have also been responsible for many pollution problems attributed to lagoon spills, polluted runoff from sprayfields, and general mishandling of liquid hog waste. Even when actual discharges have not occurred, practices such as overfilling lagoons ("inadequate freeboard") present a substantial risk that a discharge could occur and also indicate problems with operator error and the generation of too much liquid waste. Some of the incidents are listed below.

August 1995: Two million gallons of liquid hog waste from a Brown's Inc. lagoon in New Hanover County spilled into a tributary of the Cape Fear River in North Carolina.[6]

1996: One million gallons of liquid hog waste from a Smithfield hog factory spilled into the Trent River in Jones County, North Carolina.[7] Also, at a Brown's of Carolina facility in Bladen County, North Carolina, inspectors found ponding of waste on fields, indicating that wastewater had been overapplied, and that waste had been applied when the ground was wet or frozen.[8]

1997: Smith Farms, a finishing operation for Brown's of Carolina, flooded over an acre of wetlands bordering the New River with hog waste from its Onslow County, North Carolina operation.[9]

July 1997: North Carolina inspectors found that a lagoon or lagoons had been overfilled at a Brown's of Carolina facility in Bladen County ("inadequate freeboard"). Also in July 1997, state inspectors found that areas around the dikes at the facility might erode if improvements were not made.[10]

September 1998: State inspectors from North Carolina noted that the waste management system of a Brown's of Carolina facility in Jones County was in disrepair.[11]

March 1998: Inspectors from the state of North Carolina found trash floating in the lagoons and inadequate lagoon storage capacity at a Brown's of Carolina facility in Jones County.[12]

March and May 1998: At a facility in Duplin County under contract with Smithfield Foods, North Carolina inspectors found numerous deficiencies including ponding of animal waste in sprayfields that indicated over-application, lagoon seepage, and the pointing of a spray mechanism towards a ditch that led to a waterway.[13]

April 1999: A hole in a lagoon at a Murphy Family Farms factory farm in Duplin County, North Carolina spilled 1.5 million gallons of manure and urine into a swamp adjoining the Persimmon Branch, a tributary of the Northeast Cape Fear River.[14]

Operators of a hog factory farm dumped waste into this stream in Duplin County, North Carolina. This stream leads to the Cape Fear River. This was discovered and reported by a local citizen activist.

Rick Dove

June 1999: At two Brown's of Carolina facilities in Bladen County, North Carolina, inspectors found that the discharge pipes that carried waste from the confinement buildings to the storage pond or lagoons were not functioning properly.[15]

September 1999: A state inspection of a Brown's of Carolina facility in Jones County, North Carolina found that the lagoon storage capacity was inadequate.[16]

October 1999: At a facility under contract with Smithfield Foods in Duplin County, North Carolina, inspectors found that animal waste had been discharged to navigable waters during flooding. In July of that year, the overfilling of lagoons had been identified as a problem by inspectors.[17]

November 1999: A Duplin County, North Carolina factory farm owned and operated by Murphy Family Farms,[18] spilled 5,000 gallons of hog waste into wetlands, and then into a tributary of Persimmon Branch. Whether the problem originated from waste applied to saturated fields or a lagoon leak was unclear. This same facility spilled 1.5 million gallons of manure in April of the same year.[19]

December 1999: A Carroll's Foods lagoon in Sampson County, North Carolina spilled nearly 200,000 gallons of hog waste into Turkey Creek and a nearby swamp. The waste spill was caused by a pump that was left running overnight between a lagoon and a field where waste was applied. By the time the pump was shut down, the four-acre lagoon had dropped by two inches.[20]

1999-2000: At a Brown's of Carolina facility in Bladen County, North Carolina, inspectors found insufficient storage levels at lagoons three times.[21]

February 2000: Inspectors from the state of North Carolina found trash floating in the lagoons, evidence of over-application of waste, ponding, and inadequate lagoon storage capacity at a Brown's of Carolina facility in Jones County.[22]

March and July 2000: Neighbors saw waste coming from pipes at a Brown's of Carolina facility and entering into White's Creek, a tributary of the Cape Fear River. At other times, neighbors had reported the spraying of waste in windy and wet weather.[23]

March 2000: At a facility under contract with Smithfield in Duplin County, the Riverkeepers documented discharges of waste from the waste management system into a waterway.[24] During that same month, North Carolina inspectors noted that waste had been sprayed into woods and near the edge of a forest close to a waterway.[25]

April, May, September 2000: At four separate times, North Carolina inspectors found inadequate lagoon storage capacity at a Brown's of Carolina facility in Bladen County.[26]

May 2000: The Riverkeepers documented that swine waste coming from a facility under contract with Smithfield Foods in Duplin County was flowing into woods next to a waterway.[27] Later that month, North Carolina inspectors found on several occasions that wastewater from sprayfields was flowing into woods and waterways. In August of the same year, inspectors observed excessive ponding in fields and woods near the facility.[28]

August 2000: Smithfield agreed to convert its existing open-air lagoon systems on company-owned farms in North Carolina to "environmentally superior technologies" within five years.[29] Several non-lagoon technologies have already been identified for installation under the terms of the agreement, including constructed wetlands treatment and a sequencing batch reactor.[30] (See Chapter Five.) However, this agreement does not apply to the hundreds of thousands of sows raised at hog farms in South Carolina, Virginia, Utah, Colorado, Texas, Oklahoma, South Dakota, Missouri, Illinois, Mexico, and Brazil.[31]

December 2000 and February 2001: Water Keeper Alliance, supported by environmental, family farm, and animal welfare organizations, launched a legal campaign against the hog industry, sending notice-of-intent-to-sue letters to Smithfield producers in North Carolina and Missouri, filing two lawsuits in North Carolina federal court based on company violations of the federal Clean Water Act and Resource Conservation and Recovery Act, a lawsuit in North Carolina state court based on public nuisance and the public trust doctrine, and a lawsuit in Florida federal court based on violations of the Racketeer Influenced and Corrupt Organizations law (RICO).[32]

SEABOARD CORPORATION

Seaboard Corporation is currently the third largest pork producer in the United States,[33] despite the fact that, as of 1989, the company did not own a single hog.[34] According to *Time Magazine*, the growth of the company has been achieved, in part, through $150 million in economic incentives the company received from federal, state, and local governments from 1990 to 1997, including various types of financial assistance to its poultry and hog processing plants in the U.S.[35] Despite all that support, Seaboard continues to violate environmental laws.

February 1999: A Seaboard factory farm in Texas County, Oklahoma sprayed effluent in the air during a windy day. The facility was later fined $5,000 by the state.[36]

October 1999: Seaboard Farms' Dorman Sow Farm had three manure spills in one month. In one spill, over-application of manure onto already over-saturated fields resulted in one pool of manure that was 250 feet by 100 feet and over six feet deep. In another incident, a hole was punched in the irrigation line creating a pool 150 feet long by six feet wide and a foot deep. The final incident that month occurred when an underground pipeline failed and produced a waste stream that ran almost a mile and ended within 300 feet of the Beaver River Wildlife Sanctuary in Oklahoma.[37]

February 2000: The Sierra Club filed a notice of intent to sue Seaboard for 12 separate violations of the Clean Water Act at the Dorman Sow Farm. Violations cited include pouring waste directly into streams, over-applying waste to the land, and being unable to properly store manure.[38]

November 2000: The Sierra Club filed another notice of intent to sue Seaboard's Dorman Sow Farm under the Superfund law for its failure to report ammonia emissions. The

Superfund law Comprehensive, Environmental Response, Compensation, and Liability Act) provides that any site that releases more than 100 pounds of ammonia per day is required to report its releases.[39]

The facility's record of violations is of a particular concern because of its close proximity to the 23 square-mile Beaver River Wildlife Management Area, home to pheasants, quail, turkeys, rabbits, deer, and other animal life.[40] The decision by the state of Oklahoma to grant an operating license to Seaboard is the subject of a lawsuit by adjacent landowners and the Oklahoma Department of Wildlife Conservation (operator of the Beaver River Wildlife Management Area). The suit charges that a license to operate the facility should never have been issued in the first place, because of the risk to the Beaver River and wildlife.[41]

PRESTAGE FARMS

Prestage Farms is the nation's fourth largest hog corporation.[42] The company's facilities in Mississippi have raised concerns among neighbors. One facility in particular in Oktibbeha County, a 7,040-head operation under contract with Prestage, has been the focus of controversy because of its location adjacent to the Noxubee Wildlife Refuge, home to many endangered species and a local tourist attraction. Neighbors believe that polluted runoff from a manure lagoon or from fields sprayed with liquid hog manure could foul adjacent Browning Creek and ultimately the Noxubee River, which runs through the wildlife refuge. Despite community objections, the state granted the facility a permit and the facility has been in full operation since October 1997.[43] Odor and contaminants from the facility were the subject of a $10 million lawsuit brought by a neighbor–a father who claimed that the pollutants aggravated his teenager's asthma.[44] In 1997, a chancery court judge ruled that because of the particulate air pollution emitted from the facility, the state was obligated to issue an air quality permit to the contract farm. However, the Mississippi Department of Environmental Quality has not enforced the decision.[45] This ruling prompted the state's legislature to exempt hog farms from air quality standards, but allowed local governments to establish local controls.[46] Since the establishment of the original facility two years ago, more facilities have requested permits to establish operations near the refuge.[47] Fifty-two counties established rules for CAFOs, but Prestage Farms sued the six counties in which the company had facilities. Concerned about the expenses of litigation, five of the counties eliminated their ordinances. One county, Monroe County, defended its ordinance and won the legal challenge. Then a state moratorium on the building of new CAFOs was established until January 2000. After the Health Department decided to take a cautionary approach regarding possible health impacts of these facilities, the moratorium was extended. The factory farms in Mississippi controlled by Prestage produce 300,000 hogs per year. If the moratorium is lifted, the corporation plans to establish 33 more facilities.[48]

November 1994: In Lowndes County, Mississippi, several discharges from over-application of manure onto the land from Prestage Farms into James Creek resulted in a $15,000 state fine that was then reduced to $6,375.[49]

January 2000: Over 500 neighbors of Prestage Farms hog factory farms, processing plants, and meat packers in Chicksaw and Clay counties in Mississippi filed a $75 million class action lawsuit claiming that air pollution from the facilities has led to unusually high levels of asthma, migraines, and other illnesses. The families that have brought the lawsuit have the backing of the Sierra Club.[50]

CONTIGROUP COMPANIES/PREMIUM STANDARDS FARMS

Contigroup, formerly Continental Grain, became a top hog producer and processor when it acquired a majority interest in Premium Standard Farms in 1998.[51] As of 1999, Contigroup was the second largest pork producer in the United States.[52] Prior to its acquisition, Premium Standard Farms was itself ranked in the top five pork producers,[53] and the company has facilities in Missouri, North Carolina, and Texas.[54]

1995: Six spills at Premium Standard Farms facilities killed more than 268,000 fish. One spill resulted in the loss of all aquatic life in an 11-mile stretch of Mussel Fork Creek in Missouri.[55]

March 1997-July 1998: Premium Standard Farms facilities in Missouri were responsible for 20 spills of liquified feces and urine totaling over a quarter-million gallons.[56]

July 1999: A valve was left open on spraying equipment at a Premium Standard Farms facility in Missouri resulting in about 2,000 gallons of manure spilling into Raccoon Creek.[57]

Throughout 1999: Twenty-five animal waste spills from Premium Standards Farms facilities in Missouri caused the discharge of over 224,000 gallons of manure, wastewater, and liquids from dead animals.[58]

Ongoing Legal Action Against Premium Standard Farms: In 1997 and 1998, the Citizens Legal Environmental Action Network (CLEAN) filed legal action against Premium Standard Farms and Continental Grain. CLEAN's lawsuit alleged Premium Standard Farms violated the Clean Water Act by discharging animal wastes into waterways, violated the Clean Air Act by failing to obtain permits, and failed to report releases of hazardous substances in violation of the Comprehensive Enviornmental Response, Compensation, and Liability Act.[59] EPA intervened in the suit in 1999, taking the citizens' position against Premium Standard Farms. Also in 1999, the Missouri attorney general filed a lawsuit against Premium Standard Farms for manure spills and other environmental violations; the settlement agreement on the case resulted in a $1 million fine and a commitment by the company to invest $25 million in new technology to reduce pollution.[60] In late April 2000, EPA issued a Notice of Violation under the Clean Air Act against seven Premium Standard Farms facilities in Missouri stating that the facilities "emit, in addition to odors, regulated pollutants such as particulate matter and hydrogen sulfide."[61]

DECOSTER EGG FARMS

With the production of 12 to 14 million eggs a week from 3.5 million chickens and with $40 million in sales, DeCoster Egg Farms of Turner, Maine, is one of the largest egg producers in the United States. The firm also has facilities in Iowa, Ohio, and Minnesota, some of which produce pork.[62]

DeCoster has been the subject of numerous federal fines and legal actions for workplace and civil rights violations from 1988-2000. In 1988, the company was fined $46,250 for 184 workplace violations. Eight years later, a fine of $3.6 million was levied for bad working and housing conditions. This second fine prompted Labor Secretary Robert Reich to say that conditions at DeCoster are "as dangerous and oppressive as any sweatshop we have ever seen." In 1997, the company agreed to pay $2 million in fines for a range of health and safety violations. The company was set back $24,000 for failing to pay workers on time and failing to compensate workers for overtime hours in 1999. In 2000 DeCoster agreed to pay $850,000 in worker overtime claims dating from 1991 through 1997.[63] The company's civil rights violations have included restrictions on visitors to the company-owned trailer park that housed Hispanic migrant workers (1992) and the use of force by supervisors against worker-tenants living on DeCoster's property (1995).[64]

The company, which has used lagoons to store manure, has also had environmental violations.

September 1996: State inspectors found evidence of faulty construction at 19 DeCoster hog farms in north-central Iowa. Lagoons at many of the sites were below the water table, lagoon walls had eroded, and many of the operations were built in sandy soil that is unsuitable for manure lagoons. The Iowa Department of Natural Resoures threatened to shut down a 16,000 hog nursery in north-central Iowa after a state inspection found part of the lagoon sat more than 20 feet below the groundwater level.[65]

July 1997: The Maine Board of Environmental Protection levied a $75,000 fine against DeCoster for installing a wastewater disposal system at a 77-acre Leeds site without first obtaining the required state approvals.

November 1997: The Maine Board of Environmental Protection approved a consent agreement drafted by the Department of Environmental Protection (DEP) and signed by the Attorney General's Office and DeCoster president Austin "Jack" DeCoster. The 13-page consent agreement, which carries a $68,500 fine, acknowledges wrongdoing by the company at its egg-processing plants in Turner, Maine. According to the consent agreement, the company had built a large uncovered manure pit without approval, stored septage material in unapproved tanks in unapproved locations, and installed a mobile home and septic system without approval.[66]

Related Action, June, 2000: In a settlement agreement with the state of Iowa over DeCoster's operation of hog CAFOs in Iowa, DeCoster agreed to pay a $150,000 fine and build additional manure storage facilities to settle two pending environmental cases. The fine is the largest ever assessed against a livestock producer for violating environmental laws. DeCoster had previously been fined $69,000 for three previous violations and accrued enough environmental violations to be designated as a “habitual offender” under Iowa law– the first such designation in Iowa history. It prohibits DeCoster from expanding or building a new hog confinement operation for five years and puts the company’s current operations under increased regulatory scrutiny.[67]

DAIRY OPERATIONS

Presently most dairy operations are not controlled by large corporations through contract arrangements. Instead, many operate within cooperatives, like Land O Lakes. Ten cooperatives produced half of the nation’s milk in 1998.[68] However, when pollution incidents occur, the operator, rather than the cooperative to which the facility is a part, is held responsible. Thus, it is difficult to identify pollution incidents that happen in multiple states attributed to a single cooperative. Moreover, despite the increasing power of the cooperatives, many independent dairies continue to operate in many states. There are numerous examples of pollution problems from dairies attributed to the lagoon system, the application of liquid manure to the land, and liquid manure systems generally. A few examples are noted below.

March 1998: In the first imposition of jail time for dairy water pollution, Pete Hetinga of the 3H Dairy Farm in Oakdale, California was sentenced to 90 days of jail time, 90 days of home confinement, and four years probation for Clean Water Act violations. Hetinga also had to pay a $100,000 fine and make $101,000 of improvements on his farm. The defendant admitted that over a four-year period, he discharged wastewater polluted with cow urine, feces, and wash water into streams that flow into the Tuolumne River and the Sacramento Delta.[69]

Spring 1998: In the state of Washington, two dairy feedlot operators reported catastrophic lagoon failures. Each of the spills dumped the contents of an entire lagoon. One spill dumped 1.3 million gallons of waste, while the other dumped 700,000 gallons. Within one week of each other, both spills polluted the Yakima River. Meager fines of $2,000 and $3,000 respectively, were levied by the state.[70]

March and April 1999: Inspection by Washington state inspectors found spills at a dairy near Little Rock. The fecal coliform bacteria in the water in one of the facility’s drainage swales was 23,000 colonies per 100 ml; another one registered at 130,000 colonies per 100 ml. (The state water quality standard is 100 colonies per 100 ml of water.) Liquid dairy waste was also observed covering a road, pooled in ditches surrounding the property, and seeping on to a neighbor’s property.[71]

March 2000: A Washington state inspection of a dairy operation in Orting identified five streams of manure and wastewater entering Horse Haven Creek from the dairy. Manure was running off the confinement areas and over the top of a manure storage lagoon. Also, a pipe from field drainage tiles surrounding the manure storage lagoon was discharging wastewater directly into the creek.[72]

Karen Hudson

In February 2001, Inwood Dairy, LLC, near Elmwood, Illinois, pumped and dumped waste via long hoses across fields into a ditch from a lagoon.

February 2001: Inwood Dairy, near Elmwood, Illinois, pumped two million gallons of cattle waste into a nearby ravine.[73] Earthen dams along the ravine failed to retain the waste, which killed fish in a nearby pond and drained into Kickapoo Creek. At the time, the dairy was under a court-ordered injunction to keep its 8.3-acre, 40-million-gallon lagoon from overflowing. The Illinois attorney general sought the injunction after state inspectors found dairy employees sandbagging the berm of the lagoon and applying wastewater onto saturated fields to stop the lagoon from overflowing.[74]

March 2001: NRDC filed a lawsuit in Florida state court on behalf of three environmental groups and a Florida activist against the Florida Department of Environmental Protection (DEP). The civil action alleges that DEP has failed to require large-scale dairies to obtain permits mandated by the Clean Water Act and Florida law.[75] The groups are concerned that attempts at voluntary compliance by the unpermitted dairies are not preventing water pollution. Under Florida law, citizens have the power to sue the state to compel the agency to enforce its laws and regulations designed to protect surface and groundwater.[76] Florida's failure to regulate large-scale dairies is one example of a regulatory shortfall prevalent in many states across the country.

The litany of violations by these large hog, egg, and dairy producers provides an indication of the environmental harm caused by industrial livestock productions through the use of the lagoon and sprayfield system. Every year, millions of gallons of waste spills from lagoons into rivers and wetlands, is sprayed into waterways, and kill countless fish. Yet, despite the ample financial resources of the largest corporations, lagoons continue to be constructed poorly and waste management systems are not adequately maintained.

Health Effects of Lagoons and Sprayfields

Lagoons are a common feature of the growing number of factory-sized animal operations located in more than 30 states across the United States. Hundreds of gases from these lagoons can pollute the air around the operation. Researchers indicate that feedlot odor may contain 170 separate chemical substances.[1] A report released by the Minnesota Pollution Control Agency (MPCA)[2] indicates that lagoon emissions contain toxic constituents and greenhouse gases, including hydrogen sulfide, ammonia, and methane.[3] Large scale feedlots also emit particulate matter from confinement buildings. In late April 2000, EPA issued a Notice of Violation under the Clean Air Act against seven Premium Standard Farm factory farms in Missouri stating that the facilities "emit, in addition to odors, regulated pollutants such as particulate matter and hydrogen sulfide."[4] Pathogens, including bacteria, viruses, and parasites from ruptured, overflowing, and leaching lagoons, are a major concern when they flow into streams, rivers, and bays, and poison drinking water supplies.

Another threat associated with lagoons comes from sprayfields. Once manure is stored in open-air lagoons, it is periodically pumped out to be sprayed on fields surrounding the factory farm, ostensibly to be used as fertilizer. The spray emits the same gases as lagoons. Spraying the wastes increases evaporation and volatilization of pollutants into the air.

Manure applied to crops is helpful as a fertilizer; however, factory farms often produce too much manure for the amount of land available to use it. Manure is often over-applied and misapplied to land, which causes it to run off the fields, polluting our rivers and streams with pathogens and leaching into groundwater and poisoning our drinking water supplies. This poses a problem even if the manure is applied in dry form; however, the likelihood of runoff increases when the manure is in liquid form as it is with the sprayfield system.

AIR EMISSIONS

Studies consistently show that lagoons emit toxic airborne chemicals that can result in human health problems through inflammatory, immunologic, irritant, neurochemical, and psychophysiologic mechanisms.[5] The emissions are the result of the decomposition of liquid manure by anaerobic bacteria during storage and treatment. This process releases 400 volatile organic compounds,[6] including hydrogen sulfide, ammonia, dusts, endotoxins,[7] and methane.

Ammonia. Up to 80 percent of a lagoon's nitrogen may change from a liquid into a gas in the process known as ammonia volatilization.[8] This process also causes a sprayfield's nitrogen to be lost to the atmosphere.[9] In contrast, dry manure systems lose 15 to 40 percent of their nitrogen to the atmosphere.[10] Once the ammonia is volatilized, it may be redeposited onto land and water as far away as 300 miles.[11]

Although ammonia can cause eye irritation or even death at high levels, ammonia emissions associated with the lagoon system may not be at levels as toxic as other gases emitted. Nonetheless, European studies have found that reducing ammonia levels reduces the levels of odors across-the-board.[12] Ammonia may adsorb dust particles that may be then carried into the lungs.[13]

Hydrogen Sulfide. Among the many feedlot emissions, hydrogen sulfide is one of the most threatening.[14] Hydrogen sulfide is a gas that can cause eye, nose, and throat irritation, diarrhea, hoarseness, sore throat, cough, chest tightness, nasal congestion, heart palpitations, shortness of breath, stress, mood alterations, sudden fatigue, headaches, nausea, sudden loss of consciousness, comas, seizures, and even death.[15] Even when exposure is at a low level, health impacts can be irreversible.[16] A recent study by the Minnesota Pollution Control Agency (MPCA) revealed that manure storage methods appear to affect the amount of hydrogen sulfide emitted into the air. Earthen lagoons had the greatest hydrogen sulfide emissions, with averages greater than 30 parts per billion, while stockpiling manure had a rating of 20 parts per billion.[17] Another study by the same agency evaluated hydrogen sulfide emissions from 42 animal feedlots that used lagoons and cement pits in a nine-township area targeted for hog-farm expansions. The study found that concentrations of hydrogen sulfide, estimated by using a standard EPA approach to model emissions, exceeded the state standard significantly, even as far away as 4.9 miles.[18]

Air quality monitoring by the Minnesota Department of Health affirmed that toxic gas emanating from the manure lagoon of ValAdCo in Renville County, one of the state's largest operations for finishing hogs for market, posed a potential threat to human health. After two years of testing the swine facility, the state found hydrogen sulfide levels far exceeding the state standard (50 parts per billion), 53 times in 1998, and 271 times in 1999 and 2000. The violations in 2000 occurred despite a 1999 settlement between the company and state pollution officials designed to reduce odor and prevent health problems. The latest violations have prompted a new agreement between the company and state officials, which includes a penalty of $125,000, new technology to cover fourteen lagoons, additional air quality monitoring, and more expeditious resolutions of odor problems. The facility is already required to install covers over some of its lagoons under the 1999 settlement agreement.[19] For most of the violations that have occurred over the last two years, the hydrogen sulfide reading was 90 parts per billion. In 2000, Kathy Norlien of the Health Department's Health Risk Assessment stated that "without delay, actions should be taken to reduce the emissions for the protection and well-being of human health."[20] The monitoring was done under a 1997 law by the Minnesota legislature that required the MPCA to monitor hydrogen sulfide emissions from feedlots.

MINNESOTA HOG FARM SICKENS CHILDREN

When the poison control center official spoke to Julie Jansen, his words were shocking: "Ma'am, the only symptoms of hydrogen sulfide poisoning you're not experiencing are seizures, convulsions, and death. Leave the area immediately." Panic-stricken, Jansen grabbed her six children and her friends' two children and drove away from her home.

Jansen first thought the 11-year-old, home-based day care center she owned in Olivia, Minnesota had been hit by a flu bug. In the spring of 1995, 17 children ranging in age from newborn to 13 shared a long list of symptoms–diarrhea, nausea, headaches, vomiting, teary eyes, and stuffy noses. She soon noticed that it only happened when the wind blew from the south. Two factory-scale hog farms had recently located not more than a mile and a half away. It turned out the hog operations were poisoning the air with toxic wastes.

As a result, Jansen fought with state politicians and officials, helped pass a law to ensure that the state's air quality standards for hydrogen sulfide were applied to factory farms, and forced the Minnesota Pollution Control Agency to monitor the air quality of neighboring hog factories. In 1998, a massive hog operation in Renville was cited for 46 violations of air quality standards and was ordered to improve the way it stored hog manure.[21] In 1999, an agreement with state pollution authorities for violations required the company to install covers over many of its lagoons, and demanded the payment of a $32,000 fine. However, the violations have continued, resulting in a new agreement with state officials in June 2001.[22]

Carbon Dioxide. Organic matter in livestock manure is converted to carbon dioxide and methane during the anaerobic decomposition process that occurs in lagoons. The most abundant gas produced during this process is carbon dioxide,[24] although oceans, plants and soils are constantly absorbing it from the atmosphere.[24] Carbon dioxide is not highly toxic itself, but contributes to oxygen deficiency, or asphyxiation.[25] Health problems associated with elevated levels of carbon dioxide include respiratory problems, eye irritation, and headaches.[26] Carbon dioxide is also a greenhouse gas.[27]

Methane. Methane generated during anaerobic decomposition is released from lagoons into the air. Methane is toxic at high levels, levels that typically are not found surrounding open-air lagoons, but which may be found at the top of unventilated areas such as closed manure pits. Moreover, persons exposed to toxic amounts may be unaware of the danger because methane is colorless, odorless, and tasteless.[28] In August 2000, three farm workers in Canada died after they climbed into a liquid manure tank used to spread manure on a farm field; police believe that the cause of the deaths was inhalation of the methane gas.[29] In high temperatures, the methane in the air can be highly combustible and thus extremely dangerous.[30] The level of methane concentration along a waste lagoon's berm is greater than that at a surface coal mine.[31] Methane is also a potent greenhouse gas implicated in global climate change. EPA estimated that nearly 13 percent of the total U.S. methane emissions was from livestock manure in 1998.[32] Methane emissions from manure management activities increased 53 percent from 1990 to 1998 and EPA attributes the increase in methane emissions to the growing number of large hog and dairy operations and their use of liquid manure systems.[33] EPA claims that liquid manure systems produce conditions that result in large quantities of methane emissions.[34]

STUDIES FIND PHYSICAL AND MENTAL HEALTH OF NEIGHBORS AFFECTED BY LARGE-SCALE LIVESTOCK OPERATIONS

Steven Wing and Susanne Wolf of the University of North Carolina at Chapel Hill's School of Public Health conducted a detailed survey of residents in three eastern North Carolina communities. One community was located close to a 6,000-head hog factory farm with a lagoon, another community lived close to two large dairy operations with two lagoons, and the third community served as the control group because there were no intensive livestock operations nearby. More than half of the respondents living within two miles of the intensive swine operation with an open lagoon reported not being able to open windows or go outside even in nice weather because of the noxious smell. Also, the people living close to the hog operation reported headaches, runny noses, sore throats, excessive coughing, diarrhea, and burning eyes significantly more often than the other groups in the study.[35]

Another study, led by Kendall Thu, former associate director of the University of Iowa's Center for Agricultural Safety and Health, evaluated the health of 18 neighbors living within a two-mile radius of a 4,000-head hog confinement facility. Their physical and mental health was compared to a random sample of comparable rural residents who did not live near livestock facilities. Neighbors reported respiratory problems similar to those of workers on factory farms. Of greatest frequency among the neighbors surrounding the hog facility were symptoms that indicated bronchitis and hyperactive airways, including coughing, shortness of breath, wheezing, and chest tightness. Other common symptoms among this group of residents were nausea, weakness, dizziness, and fainting. Symptoms that were less statistically significant but still mentioned among the neighbors were headaches, burning eyes, runny noses, and scratchy throats.[36]

Finally, a study by Dr. Susan Schiffman from the Duke University Department of Psychiatry study found significantly higher levels of tension, depression, anger, and fatigue among North Carolina residents who lived near large swine factory farms as compared to rural residents located away from these facilities.[37] This study used a standardized scale to quantify objectively the moods of people exposed to odors near large-scale hog operations. According to the study, investigating mood in persons exposed to odors is an important health issue because a negative mood can affect immunity and can influence susceptibility to disease.[38]

HOW PATHOGENS FROM MANURE CAUSE HUMAN DISEASES

As the number of CAFOs increase around the country, scientists and policymakers are becoming more concerned about the presence of pathogens–microorganisms which are a potential source of infection for animals and humans–in livestock waste and wastewater. One of the reasons that pathogens pose a significant concern is that within the next decade, one in five persons will fit into a category considered vulnerable to the impacts of pathogens and chemical pollutants–infants, the elderly, or persons with compromised immune systems.[39] Home drinking water wells near animal waste application sites may become contaminated by pathogens or other pollutants.[40]

Until recently, the microbial quality of feedlot wastewater was not a regulatory issue or research focus. Vincent R. Hill and Mark D. Sobsey of the University of North Carolina at Chapel Hill, one of the few health research teams studying CAFOs, found that a number of the pathogenic microbes in swine and poultry wastes can infect people. These researchers also found that the "bacterial indicator levels in swine lagoon effluents are much higher than allowed for municipal wastewater effluents discharged to land or water." Thus, the land application of swine-lagoon effluent could pose a risk to communities that rely on groundwater for drinking water and could also degrade the microbial quality of nearby surface waters.[41] According to EPA, "bacteria and viruses such as *E. Coli*, salmonella, and giardia found in dairy waste can contaminate drinking water, cause acute gastroenteritis and fever, kidney failure, and even death."[42] Scientist Jeffrey C. Burnham also concludes that water contaminated by animal manure significantly contributes to the occurrence of human disease, especially from water-borne infections.[43] Burnham contends that infectious diseases from manure can result from direct contamination of water, a change in the levels of nutrients found in the environment, or the transfer of drug-resistant pathogens infecting the human population (resistance to antibiotics). Burnham reports that the following contaminants cause dangerous human health problems.

Water contaminated by animal manure significantly contributes to the occurrence of human disease, especially from water-borne infections.

1. *E. Coli*, which is found in the intestines and feces of both animal and humans, is extremely virulent.[44] A recent case, in May 2000, occurred in Walkerton, Ontario, where 1,300 cases of gastroenteritis occurred and six people died. The Ontario Ministry of Health and Long-Term Care determined that the likely source was cattle manure runoff from a farm adjacent to a drinking water supply well.[45]
2. *Cryptosporidium* poses a real problem to manure and wastewater processing because it is resistant to most treatment protocols. In healthy individuals, *cryptosporidiosis* lasts for a few days, causing diarrhea, vomiting, stomach cramps, and fever.
3. *Pfiesteria piscicida* results from an increase in nutrients in water sources. The toxic dinoflagellate causes lesions in fish and neurological damage in infected humans.

Cryptosporidium can cause death in persons with compromised and weak immune systems. It is estimated that five to ten percent of all AIDS patients may have *cryptosporidial* infections each year. In the United States, there have been six outbreaks of *cryptosporidiosis* from drinking water. One of these occurred in Milwaukee in 1993, left 400,000 persons ill,[46] and resulted in $37 million in lost wages and productivity.[47] In

LIFE NEXT DOOR TO A HOG FARM

Neil Julian Savage, Bladen County, North Carolina

"I am a farmer and I have lived with my wife Charlotte on our farm here in Bladen County since 1952. Originally,… this land belonged to my father. Things began to change for my family and me when a large corporate hog farm opened up operations on the property adjoining mine. This took place about 1991…The hog farm, which is owned by Brown's of Carolina (Brown's 91-Smithfield), has a lagoon, ten barns, and multiple sprayfields which are directly connected to my property in many places. Often times, even when they are not spraying hog waste, the smell from the barns and lagoon gets so bad I can smell it in my house with all the doors and windows shut. During these times, it is impossible to stay outside for even short periods. When Brown's is spraying hog waste on the fields, especially when it is near my property, living here is almost impossible.

"The overall situation has been so bad that I have not been able to farm my land for some time and I am forced to live on what little savings I have put aside over the years. My wife and I have been made sick by the rancid odors that are forced upon us. If Brown's is spraying near my house I cannot stand to be outside for more than a few minutes. It makes me so sick that I have fallen to the ground and had to crawl back to the house on several occasions. The same thing has happened to my wife. Sometimes it is so bad that my wife and I feel like giving up. We are getting old and this situation is very difficult to handle….

"Often times, when Brown's is spraying hog waste, they spray within just a few feet of my front yard. Some of the sprayers are so close that the hog waste is sprayed on my property. This happens when the wind blows it over in my direction. Sometimes on windy days, when the hog waste being sprayed, a mist of hog waste gets all over my house, vehicles, equipment, and land. During those times, my family and I are forced to breathe that hog waste into our lungs."[48]

Milwaukee, the likely sources of the infection were cattle manure upstream of the city, slaughterhouses, or human sewage.[49] A Canadian study of the presence of *Cryptosporidium* at ten swine farms found the parasite in liquid swine manure storage structures, surface drain water, and subsurface tile drainage water. Thirty-seven percent of the samples taken of swine liquid manure structures contained the parasite, demonstrating that "conditions in a typical swine liquid manure storage are not such that there is a complete die-off of *Cryptosporidium* oocysts."[50] Forty-four percent of the 32 water samples tested positive for *Cryptosporidium.*[51]

Michael Mallin and JoAnn Burkholder studied the effects of lagoon spills on the surface waters of North Carolina and found that there are high counts of fecal coliform–indicating the presence of bacteria–even 61 days after a spill. Natural or man-made disturbance of contaminated water re-suspend potentially dangerous amounts of bacteria and other microbes back into the water column for weeks after a spill.[52]

Waste can enter surface and groundwater supplies even in dry weather through spills or leaks from lagoons and from over-spraying manure onto croplands. The U.S. Fish and

Wildlife Service reported that elevated levels of fecal coliforms and fecal streptocolli were found on fields on which animal manure was applied.[53] The problem is exacerbated when there is extensive rain and hurricanes resulting in flooding–a problem common to North Carolina and states along the Mississippi River. For example, when Hurricane Floyd hit North Carolina in 1999, at least five manure lagoons burst and approximately 47 lagoons were completely inundated–allowing manure to flow out with the flood waters.[54]

Rick Dove

Brown's 5 & 6, a wholly-owned Smithfield hog factory, lost the contents of one of its cesspools to floodwaters caused by Hurricane Floyd. This factory is located along the Trent River above New Bern, North Carolina.

A study completed by the Centers for Disease Control of nine large Iowa confinement sites found chemical pollutants and pathogens, metals, bacteria, nitrates, and parasites in lagoons and other sites including agricultural drainage wells, tile line inlets, tile line outlets, lagoon monitoring wells, underground water, and a river.[55] Samples from the earthen lagoons contained the highest levels of chemical pollutants and pathogens. Their findings suggest that both chemical pollutants and microbial pathogens may move through soil from the site of the lagoon and flow over the land away from where the manure was applied.[56] The study called for additional research to accurately determine the potential level of risk to human health, possible pathways of exposure, and critical control points to avoid any potential exposure.

NITRATES IN WATER SUPPLIES CAN HARM HUMAN HEALTH

Nitrates above 10 mg/l in drinking water can cause human health risks, especially to children younger than five years old, the elderly, and people with suppressed immune systems. For example, infants who drink nitrate-contaminated water may be at risk of methemoglobin-emia, or baby-blue syndrome, which can cause developmental deficiencies or even death. In 1996, the Centers for Disease Control linked the high nitrate levels in Indiana well water near feedlots to spontaneous abortions in humans.[57] Increased levels of nitrates may be the result of lagoon seepage, lagoon spills or leaks, or the over- or misapplication of manure onto the land. Manure contains nitrogen which changes into nitrates in the soil. After that step, the nitrates may move through the soil and accumulate in water supplies.[58]

Infants who drink nitrate-contaminated water may be at risk of methemoglobin-emia, or baby-blue syndrome, which can cause developmental deficiencies or even death.

A survey of domestic drinking-water wells in nine Midwestern states by the Centers for Disease Control found that 32 percent of the 5,500 samples taken were above the 3 mg/l level for nitrates (assumed to be the background level); 13 percent were above the drinking water maximum contaminant standard of 10 mg/l. The study compared the contamination rates of samples from wells that, in the past five years, had manure applied within 100 feet of the wellhead to the rates of samples where no applications had occurred. This analysis found that the use of manure doubled the likelihood of an elevated nitrate level.[59]

EPA ISSUES EMERGENCY ORDER TO HOG FARMS CONTAMINATING DRINKING WATER

On June 7, 2001, EPA Region 6 exercised rarely used emergency powers under the federal Safe Drinking Water Act to compel five hog operations in Kingfisher and Major Counties in Oklahoma to provide area residents with safe drinking water.

In March and May 2001, EPA sampled drinking-water supply wells and found nitrate concentrations as high as 15.7 mg/l, where the acceptable level is 10 mg/l. Nitrates from the hog operations contaminated the surficial aquifer, which serves as an underground source of drinking water for four nearby households. EPA warned one area resident, a pregnant woman, to drink only bottled water, when an investigation team visited her home on May 30, 2001.

The hog facilities continue to contaminate the surficial aquifer by spraying waste and/or leaking waste lagoons. EPA's order clearly states, "[n]itrate contamination in the soil and ground water at the facility and in the vicinity will continue to threaten human health and the environment until the source of the contamination is removed and the site is remediated."

EPA issued the Emergency Order to Seaboard Farms, Inc., Shawnee Funding Limited Partnership, and PIC International Group, Inc. The order requires the companies to deliver an emergency supply of water for human consumption to area residents and hog farm employees. In addition, the companies must sample and test wells to determine the presence of nitrate, ammonia, bacteria, and other contaminants.[60]

ANTIBIOTICS IN LAGOONS MAY CONTRIBUTE TO ANTIBIOTIC RESISTANCE

Antibiotic resistance poses a major public health concern. Many antibiotics can no longer effectively fight infectious diseases because bacteria resistant to them. Exposure of the elderly, children, and immune-compromised individuals to antibiotic-resistant bacteria can be deadly. Even for healthy adult-human populations, antibiotic resistance makes it take longer to treat an infection and increases the cost of treatment.

Low doses of antibiotics are routinely added to livestock feed and water to promote growth and prevent disease in crowded conditions. The Union of Concerned Scientists estimated that livestock producers in the United States use 24.6 million pounds of antimicrobials each year for nontherapeutic purposes, compared to just 3 million pounds used by humans to treat diseases.[61] Recognizing the increasingly serious public health threat caused by bacterial resistance, the American Medical Association (AMA) passed a resolution in June 2001 opposing nontherapeutic uses of antibiotics in agriculture. The resolution states, the "AMA is opposed to the use of antimicrobials at nontherapeutic levels in agriculture or as pesticides or growth promoters and urges that non-therapeutic use in animals of antimicrobials (that are also used in humans) should be terminated or phased out."[62]

“THE SMELL ALMOST KNOCKS YOU OVER”

Rolf Christen, family farmer who raises cattle, chickens, and various crops in Green City, Missouri

“I moved from Switzerland in 1983 to my part of Missouri because of the wide-open spaces and clean environment. When Premium Standard Farms purchased 4,000 acres next to my farm in 1993, I knew nothing about the company. But when I heard about the type of hog facility that they were intending to build there, I immediately opposed them. The facility next to my property has a 80,000 hog head capacity, which is replaced 2.8 times a year. This facility and others nearby generate so much waste that they have turned our land into waste handling facilities, which is an immoral and unethical way to use the land.

“I can’t describe how terrible the odor from the lagoons, sprayfields, and barns often is. We can’t keep our windows open, and sometimes you can even smell the odor through the shut windows. You open the door and the smell almost knocks you over. One of the worst parts of it is that the odor hits at unpredictable times so it is a constant threat. Breathing in such a terrible stench makes you feel desperate. One time I was planting soybeans on a field and I got so sick to my stomach that I had to stop planting. My wife has allergies which are aggravated by the odor. She is in the health field, and she believes that many of her patients are also suffering from worse allergies from the hog odor. A year or so ago, I went on vacation to a beautiful national park; when I entered my house upon my return and smelled the terrible odor, I broke down and cried.”[63]

According to EPA’s National Research Exposure Laboratory, “in some cases as much as 80 percent of antibiotics administered orally pass through the animal unchanged into bacteria-rich waste lagoons and is then spread on croplands as fertilizer leaving antibiotics available for entry into groundwater and runoff into surface waters carrying both the drugs and resistant bacteria or genetic material to other bacteria in soils and waterways.”[64]

The Centers for Disease Control study of nine large Iowa confinement sites found antibiotics in the lagoons and other sites including agricultural drainage ditches, agricultural drainage wells, tile line inlets, tile line outlets, lagoon monitoring wells, underground water, and a river.[65] Examples of the antibiotics identified included tetracyclines, beta lactams, and macrolides. Researchers from the University of Illinois have found bacteria resistant to tetracycline in soil and groundwater near two hog facilities that use antibiotics as growth promoters.[66] The research team concluded that, “the presence of the tetracycline resistance genes is due to seepage and movement of groundwater underlying the lagoon,” and cautioned, “the occurrence of antibiotic resistance genes in drinking water provides a possible way for antibiotic resistance to enter the animal and food chain.”[67]

According to Environmental Defense, effluents from German sewage treatment works and groundwater/surface waters were found to contain antibiotics in the microgram per liter range.[68] Given that CAFO effluents undergo even less treatment than human waste, these effluents also are likely to contain antibiotics. Even if the antibiotics have short half-lives,

MANURE LAGOONS KILL

In December 1999, a man drowned in a six-acre, 25 foot-deep, Murphy farm manure lagoon in Ellis County, Oklahoma. The man was transferring pig effluent from a malfunctioning lift station near the hog facility when he and his truck went over the bank of the lagoon and fell in. It took 18 days for the body to be recovered. A jury in a federal civil trial found Murphy Farms to be negligent and awarded the man's widow close to $2 million.[69]

In another incident, a farm worker entered a lagoon to make a repair. When he attempted to climb out, he was overcome and fell to the bottom. His 15-year-old nephew went into the lagoon to try to rescue him and collapsed. The boy's father, his cousin, and his grandfather, the owner of the operation, entered the lagoon one by one and, tragically, all five family members died. In August 1992, two separate instances took the lives of four men. Two men were overcome, fell into the lagoon and died of hydrogen sulfide poisoning. Another two men died from asphyxiation. In both cases, one man fell in the lagoon while making repairs or removing obstructions and the second man died in rescue attempts.[70]

the supply is replenished and thus they may continue to contaminate the environment, kill susceptible bacteria, and allow resistant bacteria to multiply, potentially exposing people who boat, swim, or drink water.

WORKER DEATH AND INJURY

Nineteen people have died due to hydrogen sulfide emissions from manure pits.

According to the National Institute for Occupational Safety and Health (NIOSH) of the Centers for Disease Control, manure pit and lagoon systems always hold the potential to kill farm workers, particularly in the hot summer months. In 1998, the National Institute of Health notes that some 19 people died due to hydrogen sulfide emissions from manure pits.[71] NIOSH warns that the accumulation of gases formed in the process of breaking down the waste are toxic, oxygen-consuming, and explosive.[72] Though NIOSH has issued warnings to workers, no Occupational Safety and Health Administration standard exists for work around manure pits.[73]

There are four dangerous gases emitted from lagoons of concern to farm workers:

Methane: Usually found at the top of the pit, this gas is highly flammable and explosive at concentrations of five to 15 percent. At high concentrations, methane can displace enough oxygen to suffocate a worker.[74] In addition to concerns about methane from lagoons, methane-related deaths have been attributed to entering liquid manure tanks used to spread manure on farm fields.[75]

Hydrogen Sulfide: This highly toxic gas usually settles at the bottom of a pit and, at low concentrations, can cause severe eye irrititation, dizziness, headache, nausea, and irritation of the respiratory tract. At high concentrations, the gas can result in unconsciousness, respiratory failure, and death within minutes. The gas is also explosive at concentrations ranging from 4.3 to 46 percent by volume.[76]

Carbon Dioxide: Carbon dioxide, an odorless gas, can settle at the bottom of the pit. A worker exposed to low concentrations of this gas can have headaches, labored breathing, and drowsiness. At high levels of concentration, this gas will displace oxygen and suffocate a worker.[77]

Ammonia: This gas can result in severely irritated eyes, nose, throat, and lungs and, in high concentrations, can be fatal.[78]

Because of the hazards posed to workers, NIOSH recommends that a number of precautionary steps be taken, including alerting workers to the possible dangers.

WATER QUALITY IMPACTS OF THE LAGOON AND SPRAYFIELD SYSTEM

There are multiple ways that the lagoon and sprayfield system causes water pollution, kills fish, degrades aquatic habitats, and threatens drinking water supplies. Lagoons can break, spill, or fail, sending wastewater into streams, lakes, rivers, or estuaries. Liquid waste can be over-applied or inappropriately applied to farm fields through irrigation pivots with resulting runoff into lakes, rivers, and streams or seepage into groundwater. Lagoon linings can allow liquefied manure to seep into groundwater, and cracks in the linings can make the problem worse. Pipes and hoses connecting to lagoons or sprayfields may fail or leak. Finally, ammonia emissions from lagoons and sprayfields may result in atmospheric deposition, which sends a toxic form of nitrogen through the air miles away, where it is then deposited in waterways.

Siting and cumulative effects are particular concerns with the lagoon and sprayfield system. States and the federal government have allowed lagoons and sprayfields to be located in places where environmental harm is likely, such as in floodplains and wetlands, near water bodies, and on sandy soils, agricultural drainage wells, and karst topography that provide direct access to groundwater sources. Additionally, when multiple sprayfields cluster around slaughterhouses, as is common, their runoff causes cumulative effects within local watersheds.

The pollution from animal waste can harm waterways, human health, and aquatic life. The primary pollutants of concern for water quality purposes are the nutrients nitrogen and phosphorus. At a 2,500 hog operation in North Carolina, the slurry, which is the liquid from the lagoon that will be land applied, can contain between 58,000 and 700,000 pounds of nitrogen and 41,000 and 475,000 pounds of phosphorus depending upon the live weight of the pigs raised at the facility.[1]

While nutients are essential for plant, animal, and human life, excessive amounts can be harmful. Impacts include drinking water contamination; toxic and nontoxic algal blooms that impair recreational waters and kill fish; climatic changes from greenhouse gas increases (due to the process in which nitrates are converted to nitrous oxide); changes to coastal marine fisheries; acidification of soils and terrestrial and aquatic ecosystems; and increases in ozone and particulate matter that may harm human health and the productivity of crops and forests.[2] Nutrient pollution fosters the growth of a type of algae known as *Pfiesteria piscicida*, which has been implicated in the death of more than one billion fish in coastal waters in North Carolina.[3] The nation's primary expert on *Pfiesteria*, Dr. JoAnn Burkholder from North Carolina State University, has stated that one of the sources of pollution respon-

A CREEK THE COLOR OF HOG WASTE

Neil Julian Savage, Bladen County, North Carolina

"There is a little creek called Whites Creek that runs alongside of the Brown's hog farm, then downstream to my farm. It eventually goes into Hammonds Creek and the Cape Fear River. The creek has a wide spot next to my property. It is a special place, a place that my family and I used to be able to enjoy. Before the hog farm opened next to us, my family would go there to sit and recreate in many different ways. It was a wonderful place for the family to get together, but not any more. Since the hog farm opened it has become nearly impossible to enjoy this family treasure…

"What has happened to Whites Creek is one of our major concerns. Since the hog farm began operating, I have seen changes in this creek which are not good. Now the creek often clogs up with vegetative growth that was not seen prior to when the farm was there. The look and smell of the water has changed for the worst. There is nothing else I can think of that would account for the changes in the condition of the creek besides the Brown's hog farm. Much of the sprayfield areas at the Brown's hog farm are on a downward slope towards the creek. It is a very bad situation so far as the creek is concerned. Since I live downstream on this creek, I have been directly and adversely impacted by the degradation of Whites Creek.

"On Sunday, March 19, 2000…as I walked back to the Whites Creek and up that creek to where the hog farm was located, I could see that the creek had turned the color of hog waste. There was a strong smell coming from the creek. It was the smell of hog waste. As I continued to walk along the creek, I saw many places where Brown's had put pipes at the end of their sprayfields. These pipes were carrying hog waste from the fields, through an earthen berm, and discharging that waste directly across the land to Whites Creek. At the end of the pipes, which were connected to the sprayfields, I could see hog waste puddled and ponded as deep as 5 or 6 inches. There was a lot of hog waste ponded up everywhere…

"In some places, there was so much hog waste on the ground that it had washed out of the earthen bern and was running across the ground and into Whites Creek… On Sunday night, they sprayed again. I know this because I could hear the pumps running all night. On Monday morning, I walked around and the whole back field next to Whites Creek was heavily ponded with hog waste. It was ponded much worse than the day before. As I looked around, I found several places where it was running into the creek.

"On July 20 and 21, 2000, I heard the pumps once again running all night. On the morning of the 21st, the wind was blowing right, and I took a chance by taking a look at what was taking place. I saw two pipes connected to the lagoon. The pipes ran into the woods. I saw where the hog waste had run through the pipes and into the creek.

"On several occasions, Brown's has over-sprayed so much hog waste next to my property that, during heavy rains, the rainwater and the waste that was carried in it actually ran, in large amounts, right onto my land. A number of times this waste has ponded in my front yard. The waste has come up to my front porch, surrounded my drinking well, and run past my house on to what used to be my farm fields. I have complained to Brown's, but they have done little to correct the problems. We are afraid to drink the water from our well."[4]

sible for the outbreaks is hog waste in coastal areas.[5] Nitrogen also causes the eutrophication of lakes and estuaries, which in turn harms fish and is likely to result in species changes, since plants–and the animals and microorganisms that depend upon them–that are tolerant of low nitrogen conditions diminish while nitrophilous plant species increase.[6]

According to EPA's 1998 National Water Quality Inventory, 30 percent of surveyed rivers, 44 percent of surveyed lakes, and 23 percent of surveyed estuaries suffer from nutrient pollution. The impairment is sufficient to make it unsafe to use the waterbody for the purpose for which it is designated–fishing, swimming, and other activities.[7] Nutrients caused severe pollution problems in 44 of the coastal areas examined by the National Academy of Sciences' National Research Council, including Washington, California, Louisiana, Texas, Florida, North Carolina, Maryland, New York, and Massachusetts.[8] The U.S. Fish and Wildlife Service estimates that in 1995, manure contributed 37 percent of all nitrogen and 65 percent of all phosphorus inputs to watersheds in the central United States.[9] Another example of impairment is the Dead Zone located in the Gulf of Mexico, up to 7,000 square miles of oxygen-deprived water where no aquatic life exists, that is the result of agricultural and municipal wastewater runoff.[10] According to the Committee of Environment and Natural Resources of the White House Office of Science and Technology Policy, animal manure alone contributes 15 percent of the nitrogen to the Gulf of Mexico, while industrial and municipal point sources together contribute only 11 percent.[11]


Rick Dove

Pfiesteria-like sores appearing on fish taken from the Neuse River in North Carolina. In the fall of 2000, nearly 100 percent of the menhaden swimming in a 40-square mile area of the Neuse River below New Bern had these sores. A very high percentage died.

SALT AND HEAVY METALS

Manure may contain trace elements of arsenic, copper, selenium, zinc, cadmium, molybdenum, nickel, lead, iron, manganese, aluminum, and boron. Some of these elements are added to animal feed as growth stimulants, others are present in pesticides applied to livestock to rid the animals of insects. Salts may also be in the manure, passed through the animals in undigested feed.[12] Heavy metals and salts are transported to the environment via wastewater. Additionally, heavy metals accumulate in the solid sludge in the bottom of lagoons, reaching toxic levels until they are emptied out every five to fifteen years, or abandoned after ten or twenty years.[13]

Trace elements of metals and salts from animal manure present risks to human health and ecosystems. Excessive salt can impact ecosystems, making drinking water undrinkable, making irrigation water unusable, and increasing the blood pressure of salt-sensitive individuals. In California's Chino Basin, once the number-one milk-producing area in California and home to 300,000 cows in 50 square miles, groundwater contaminated with high levels of total dissolved salts and nitrates flows into the Santa Ana River, which then is used as a recharge source for the Orange County drinking water aquifer.[14] The application of dairy manure and dairy wastewater is considered the major threat to groundwater. A 1990 study found that dairy operations were responsible for 88 percent of the agricultural salt load within the dairy area.[15]

SPRAYING 348 DAYS A YEAR

Rolf Christen, family farmer who raises cattle, chickens, and various crops in Green City, Missouri

"Premium Standard Farms, my neighbor, assured us that they were building a safe hog facility next to our property and that there would be no problems. But then we discovered a waste spill into Spring Creek and found dead fish and destroyed aquatic life. A neighbor of mine grew up near where I live and used to bathe and drink out of Spring Creek. He keeps on saying that the creek has changed. Now there is a black sediment in the creek that no one can explain. After the first spill into Spring Creek, the Premium Standard Farms facilities in the state had 20 to 30 spills. Within a three-month period, I believe that more fish were killed in the state than had died in the last ten years. I feel an indescribable pain when I see fish killed in creeks and rivers that were once picturesque.

"Until recently, the company sprayed the waste onto the land through irrigation guns. The company used to send me a notice every year that I was being given four holidays on which no spraying would occur, July 4, Memorial Day, Labor Day, and Thanksgiving. I have seen pipes burst and irrigation pivots stuck in ditches. Now the company "injects" the manure, but that does not mean that the manure is sent deep into the soil and knifed in. Instead, tiny shallow holes are pressed into the soil and hoses fill the holes with liquid waste. The excess waste runs off.

"As farmers and human inhabitants of this planet, we are but caretakers of what has been given to us. The tree in front of my house will be here long after I am gone. Every tree I plant will be for my grandchildren to enjoy. Everything I destroy will be gone forever. So will it be with the land."[16]

Manure runoff contaminated with trace elements can end up in waterbodies where the metals become more concentrated as they make their way up the food chain. When heavy metals accumulate in sediments, aquatic biota, and plant and animal tissue, the reproduction and immune systems of many aquatic and avian species may be harmed and waterways may become impaired.[17] For example, feed additives such as zinc can contaminate plants, while arsenic, copper, and selenium in feed additives can poison aquatic and terrestrial life, such as bottom feeding birds.[18]

According to the U.S. Fish and Wildlife Service, concentrations of selenium in lagoons or waste storage pits may be ten times the level that is safe for aquatic life.[19] A 1998 study by that agency found that wetlands in Nebraska that received wastewater from a swine production operation had concentrations of copper and zinc that exceeded the current protective criterion of 121 ug/l; the level of copper even exceeded a proposed aquatic life criterion of 43 ug/l.[20]

Humans can also become impacted by heavy metal contamination. Human illnesses associated with high levels of trace elements include skin and internal-organ cancer and vascular complications from arsenic, liver dysfunction, and hair and nail loss from selenium, and upper deficiency anemia from zinc. While the concentration in animal manure of arsenic, cadmium, chromium, copper, lead, mercury, molybdenum, nickel, selenium, and zinc may be comparable to the levels found in some municipal sludges. EPA regulations restricting levels of heavy metals in human sludge do not apply to animal waste.[21]

Another risk associated with heavy metals is the deterioration of soil quality. In 1995, 10 percent of the soil samples in North Carolina's largest swine-producing counties had zinc levels ten times greater than the levels crops need for their growth. The number of soil samples from these counties that exceeded this level doubled since 1985. Already this level of zinc makes it hard to grow peanuts, and other crops will begin to suffer in future decades as the metals reach higher concentrations.[22]

GROUNDWATER POLLUTION

Federal and most state regulations concerning CAFOs focus on the nutrient content of the wastewater. However, the wastewater generated by lagoon systems contains pollutants other than nutrients, such as metals and salts, that can also adversely affect surface and groundwater quality. Additionally, the pollution strength of raw manure is as much as 110 times greater than that of raw municipal sewage.[23] (See Table 4-1.)

As of 1998, close to 1,600 wells located near factory farms in North Carolina were tested for nitrate contamination. Thirty-four percent of the wells showed nitrate contamination; ten percent of the wells had a nitrate level that exceeded the drinking water standard. The state's Department of Health and Human Services stated that the cause of contamination was leaking hog lagoons and hog wastewater sprayfields.[24]

In many cases, lagoons leak because they are not lined. Only in the last few years have some states required that lagoons be lined. Lagoon linings include clay, concrete, and plastic, with clay linings the most commonly required. But many types of linings still allow seepage to occur. Seepage from lagoons occurs in two ways: vertical seepage along the bottom of the lagoons, and vertical and horizontal seepage at the berms.[25] Moreover, cracks may occur in the lagoon lining. Visual inspections are insufficient to detect when linings have been compromised, because lagoons may be structurally sound in certain places and cracked in others. Groundwater testing wells can detect problems, but groundwater monitoring is rarely required by regulatory agencies.[26]

Studies from across the nation compiled by the Minnesota Pollution Control Agency found numerous incidents of groundwater contamination near unlined, earthen manure-storage basins. Nitrate levels were above the drinking water standard of 10 mg/l in about half of the 42 basins that were monitored; the other half of the facilities


Grasses choke off a part of North Carolina's Trent River due to over-nutrification. There are more than 450 hogs-per-square mile in this watershed.

Rick Dove

TABLE 4-1

BOD_5 Concentrations for Manures and Domestic Sewage

Waste	BOD_5 (mg/l)*
Swine Manure	
Untreated	27,000 to 33,000
Anaerobic lagoon influent	13,000
Anaerobic lagoon effluent	300 to 3,600
Poultry manure	
Untreated (chicken)	24,000
Anaerobic lagoon influent (poultry)	9,800
Anaerobic lagoon effluent (poultry)	600 to 3,800
Dairy cattle manure	
Untreated	26,000
Anaerobic lagoon influent	6,000
Anaerobic lagoon effluent	200 to 1,200
Beef cattle manure	
Untreated	28,000
Anaerobic lagoon influent	6,700
Anaerobic lagoon effluent	200 to 2,500
Domestic sewage	
Untreated	100 to 300
After secondary treatment	20

*The pollution strength of the organic matter in manure or wastewater is expressed as the biochemical oxygen demand (BOD_5).
Source: U.S. Environmental Protection Agency, Office of Water, *Environmental Assessment of Proposed Revisions to the National Pollutant Discharge Elimination System Regulation and the Effluent Guidelines for Concentrated Animal Feeding Operations*, EPA-821-B-01-001 (January 2001).

were found to have no or only slight levels of groundwater contamination.[27] Of particular concern were lagoons with earthen liners that were constructed on sandy soil, from which researchers found "significant" leakage of nitrogen and phosphorus, and leakage that increased greatly over time.[28] This study highlights the importance of soil type. When lagoons are located on coarse soils, seepage is more likely, but if the soil is firm, runoff from sprayfields may result.[29]

A study for the Iowa legislature by Iowa State University found that over 50 percent of the earthen waste-storage structures (slurry pits and lagoons) studied had seepage losses that exceeded current standards.[30] The study also looked at soil cores to measure the migration of waste. The soil cores were tested for a number of contaminants, and most of the sites had at least one of the contaminants present at a high concentration level. The pollution at some of the sites might have been attributed to waste spills or previous use of the areas for livestock production, but the high levels of ammonium nitrogen at five sites was attributed to seepage. The seepage was measured outside of the berm at a distance of 30 to 50 feet from inside the earthen waste-storage structure.[31] Another study in Iowa, by the state's Department of Natural Resources, found that two of three earthen manure lagoons constructed around 1994 on clay soils seeped into the water table. Levels of chloride, organic carbon, and organic-N were increasing over time, while sulfate and nitrate concentrations were decreasing. The three-year study concluded that seepage was continuing and that none of the basins had sealed.[32]

Clay linings, which are made of compacted soil, offer greater protection, but may still seep. For example, Kansas State University researchers studied four swine lagoons and found that all of them leaked between .05 and .08 inches a day, which translates to between .99 million and 4.35 million gallons per year, or between 19.8 and 87.1 million gallons over the twenty-year life of the lagoons. The one cattle lagoon studied seeped at a rate of .094 inches a day, 6.88 million gallons per year, or 137.7 million gallons over the life of the lagoon.[33] Just for nitrogen, seepage losses from a swine waste lagoon could add up to more than 2,600 pounds/acre per year, or 250,000 pounds over the twenty-year life of the lagoon. Nitrogen losses for cattle lagoons were less than those for swine, because the effluent contains a lower concentration of nitrogen.[34]

Also, the seal that exists below and on the sides of the lagoon may weaken over time. Several studies have found that within two to four years, chlorides and ammonium begin to leak through a clay lining.[35] A study of dairy lagoons by the New Mexico Department of Health and Environment found elevated levels of nitrogen in unlined and clay-lined lagoons that were more then ten years old.[36]

Numerous studies have found seepage from and cracking of earthen and clay liners due to wet/dry cycles, the removal of manure from the lagoon, worms, roots, rodents, freeze/thaw cycles, erosion of lagoon berms, agitation during pumping, and liner collapse due to external pressure and groundwater intrusion.[37] A study for the Iowa legislature found that 27 percent of the 33 earthen waste-storage structures studied had compacted clay liners or berms that had eroded, while 6 percent had tree growth on berms.[38] Additionally, clay liners may crack if the dredging of the sludge off the bottom of the lagoon is not done properly. This dredging is typically done every five to fifteen years.[39]

Lagoon siting is critical. The study for the Iowa legislature found that 18 percent of the 34 earthen waste-storage structures (lagoons and slurry pits) studied were located over alluvial aquifers where there was a risk of contaminating private and municipal water supplies. Moreover, the study showed that 65 percent of the site areas were located on soils with seasonal water tables of less than 5 feet. Since earthen waste-storage structures are often deeper than 10 feet, the lagoon bottoms sat below the water table.[40] The study speculated that "a large percentage of earthen waste-storage structures in this study and in the state are probably below the water table or at least in contact with the water table." Moreover, "locating an earthen waste-storage structure and applying manure on permeable soils poses a substantial risk for contaminants to reach the water table."[41]

Concrete liners can offer greater protection, but concrete can crack if builders do not follow specifications related to soil suitability and structural reinforcement. Despite the fact that these specifications contribute to liner stability, there are no requirements that compel builders to follow them.[42] Plastic liners are also not fail-safe. In 1998, two of the plastic lagoon liners used at the huge Circle Four factory farm in Beaver County, Utah developed bubbles and the liners floated to the surfaces of the lagoons.[43]

SURFACE WATER POLLUTION OF LAKES, RIVERS, AND STREAMS

While many recently-built lagoons have been designed to meet the capacity of a 24-hour, 25-year storm event, several days of rain can compromise the system, because the steady rainfall weakens the berms and prevents the excess wastewater from being sprayed on already saturated fields.[44] Bursting and overflowing manure lagoons have spawned environmental disasters around the country, sending animal waste gushing into rivers, groundwater, and coastal wetlands.

▶ Between 1990 and 1994, 63 percent of Missouri's factory farms suffered spills according to Missouri's Department of Natural Resources.[45]
▶ In 1995, an eight-acre animal waste lagoon in North Carolina burst, spilling 25 million gallons of animal waste into the New River. The spill killed 10 million fish and closed 364,000 acres of coastal wetlands to shellfishing.[46]
▶ In 1997, animal feedlots were responsible for 2,391 spills of manure in Indiana.[47]
▶ In 1998, a 100,000-gallon spill into Minnesota's Beaver Creek killed close to 700,000 fish.[48]
▶ In 1996, 40 spills killed close to 700,000 fish in Iowa, Minnesota, and Missouri.[49]
▶ From 1995 to 1998, there were at least 1,000 spills or other pollution incidents at livestock feedlots in ten states, and 200 manure-related fish kills that resulted in the death of 13 million fish.[50]

The spills listed above can be attributed to a host of problems, but they point to the inherent risks of the lagoon and sprayfield system. Some problems are due to poor management. A 1999 study for the Iowa legislature found that of the 33 earthen waste-storage structures (lagoons and slurry pits) studied, over half had minor spills when manure was being unloaded, 12 percent had flow inlet pipes that were plugged or frozen, and 6 percent had inadequate freeboard.[51]

Poor siting of lagoons poses another concern. If lagoons are located near waterbodies, wetlands, floodplains, or other ecologically sensitive areas, spills are more likely to cause harm. The study for the Iowa legislature found that 18 percent of the earthen waste-storage structures studied were located in floodplains, 21 percent of the structures were within 500 of feet of ephemeral streams, and 12 percent of the structures were within 500 feet of perennial streams. A Kansas State University study speculated that in areas with high rainfall, nitrates which are highly mobile and accumulate in significant amounts in the soil beneath lagoons, could seep deeper into the soil and closer to groundwater. This happens particularly when lagoons are closed or abandoned and their bottoms dry out.[52]

A study by the U.S. Geological Survey found that in 88 percent of the 2,056 watershed outlets, manure contributed more to in-stream total nitrogen than traditional point sources; in 113 watersheds, manure was the single largest contributor.

While spills can cause catastrophic damage, the more common problem is over- or misapplication of waste onto cropland, which sends polluted runoff into waterways and leaches pollutants into groundwater. A 1992 study by the U.S. Department of Agriculture found that "nutrients from confined animals exceed the uptake potential of non-legume harvested cropland and hayland…[R]ecovereable manure nitrogen exceeds crop system needs in 266 of 3,141 counties, and that recoverable manure phosphorus exceeds crop system needs in 485 counties."[53] A study by the U.S. Geological Survey found that in 88 percent of the 2,056 watershed outlets, manure contributed more to in-stream total nitrogen than traditional point sources; in 113 watersheds, manure was the single largest contributor. This study also concluded that manure is a major contributor to in-stream, total phosphorus concentrations, even more so than commercial fertilizers.[54] The Texas Institute for Applied Environmental

Research identified the dairy industry as the primary contributor to nutrient loading in the Upper North Bosque River. The institute also measured elevated phosphorus levels at fields where animal manure had been applied.[55]

A study by the University of Northern Iowa looked at hog CAFOs with lagoons and earthen storage basins, and their field application of manure onto corn and soybean crops. The study asserted that if farm workers had applied manure at the rate at which the crops could have absorbed phosphorus, the CAFOs would need more than nine times the field area used for manure application by these CAFOs.[56] Since clearly the land area for spreading the manure was so much less than that needed for proper application, one can assume that the excess phosphorus was running off. This same study found other practices in manure management plans submitted by most of the CAFOs that would likely lead to over-application of manure, including over-estimating crop yields and underestimating the nutrient content of manure. The study also questioned the common practice of applying manure to soybeans–a crop that traditionally has not been fertilized.[57]

Inactive lagoons pose an additional threat to groundwater. North Carolina's Department of Environment and Natural Resources inventoried 1,142 inactive lagoons. The study determined that only 43 structures were low risk, while 39 lagoons were judged to be high risk because they were either overflowing or had a high likelihood of overflowing. Over 90 percent of the lagoons were determined to present a risk for groundwater contamination.[58]

ATMOSPHERIC DEPOSITION

Studies in the United States and Europe show that livestock farms in general have the potential to contribute large amounts of nitrogen to the atmosphere as ammonia.[59] Although the amount varies based on weather, sprayfield application method, type of livestock species, and manure storage method, the impact can be significant. In fact, up to 80 percent of a swine lagoon's nitrogen may change from a liquid into a gas in the process known as ammonia volatilization.[60] In contrast, dry manure systems lose 15 to 40 percent of their nitrogen to the atmosphere.[61] For beef manure, the fewer solids and more liquid that is present in the slurry, the greater the volatilization.[62] Once the ammonia is volatilized, it can be deposited onto land and water 300 miles away.[63] Sprayfields also result in ammonia losses. Several studies have found that if manure is not incorporated into the soil, more than half of the manure is lost, presumably to volatilization.[64] One study "found that soil-incorporated manure may release as little as one-tenth the ammonia emitted from surface-spread manure, other factors being equal."[65]

Up to 80 percent of a swine lagoon's nitrogen may change from a liquid into a gas in the process known as ammonia volatilization.

In North Carolina, swine operations contribute nearly half of the total atmospheric ammonia in the state coming from all other industrial and livestock sources combined; lagoons in eastern North Carolina are responsible for a third of the total swine ammonia emissions.[66] In the six-county area of North Carolina that has the most concentrated hog production, a comparison of ammonia emission levels from hog operations in two seven-year periods, 1982-1989 and 1990-1997, showed an increase of 316 percent.[67] The increase tracks the period of rapid growth in North Carolina's hog industry which occurred starting in 1989.[68] North Carolina Environmental Defense estimated that the nitrogen from atmospheric emissions from sprayfields into rivers alone ranged from eight to 38 percent in the Neuse River basin and 16 to 38 percent in the Cape Fear River basin.[69]

In addition to threatening groundwater with pollutants, the lagoon system also causes its depletion. The lagoon system relies upon a steady supply of water. It needs water to clean the barns, cool the animals, and provide drinking water for the animals. Most significant, the system requires sufficient water to make the manure wet enough to flush into the lagoon and spray onto fields.[70] Missouri activists estimate that a swine operation that finishes 80,000 animals per day consumes over 200,000 gallons of water per day, or 73 million gallons per year.[71] In many areas in which factory farms are located, the water that is utilized is groundwater–which is provided to the factory farm for free.

ALTERNATIVE APPROACHES TO THE LAGOON AND SPRAYFIELD SYSTEM

The lagoon and sprayfield system presents numerous risks to surface and groundwater quality, air quality, and public health. A number of alternative approaches are being used by sustainable operations. Researchers are studying many alternative technologies, at least on a pilot level. The list of approaches described below illustrates one important point–it is not the lack of alternatives that is driving factory farms to rely almost exclusively on the lagoon and sprayfield system. Factory farms continue to use this polluting system because they have been allowed to use our farmland, rural waterways, and air as disposal sites for untreated wastes. With its effluent guidelines, EPA has a historic opportunity to move feedlots beyond the lagoon and sprayfield system, but it has not yet chosen to do so. Instead, the agency may allow factory farms to continue to pass the cost of waste disposal on to the public. The wealth of approaches increasingly available across the country shows that pollution from these facilities can be eliminated or reduced, but only if we require facilities to use them.

Unfortunately, many of the systems that are being evaluated presume that it is necessary to liquify and then treat the liquid manure. There is another approach–one based upon the principle of pollution prevention and proper manure management. The sustainable agriculture approach dries the manure and often adds other dry material to keep waste from running off, or seeping into water supplies. Sustainable agriculture practices that embody these approaches benefit the environment, the producer, the animals themselves, and the communities that surround them.

It is important to note that some of the alternatives to the lagoon and sprayfield system are still in the development stage and have not necessarily been evaluated for all pollution risks. Moreover, while some of the technologies have been in use for the treatment of human waste for years, they have not been widely used for animal waste. Thus, studies on pollution reductions may only be based upon a limited sample, pilot projects, or a limited history of use. Also, many university studies evaluate certain chemical parameters, but not others, so risks could be high for pollutants not measured, such as heavy metals, which, even at low concentrations, can be toxic to plants and animals. Finally, many impacts on the environment have not been measured, for example, the likelihood of groundwater contamination. For the purposes of simplicity, "public health impacts" are identified as pathogen reduction, even though reductions in surface water, groundwater, and air contamination can all benefit public health.

SUSTAINABLE AGRICULTURE APPROACHES

Management Intensive Rotational Grazing

The use of an intensive rotational grazing system provides environmental benefits, such as enhanced soil quality, minimal soil erosion, improved stream bank quality, and enhanced wildlife habitat.

Intensive rotational grazing represents a return to the practice of using managed pasture to supply at least a portion of the nutritional needs of animals with growing grasses and legumes. Animals (dairy cows, beef cattle, egg layers, broilers, hogs and others) graze on sections of pasture that are divided into paddocks. In the pasture sections that do not contain animals, forage is allowed to grow. Using fences, animals move into different sections of the pasture where they graze, but are moved before the pasture section is over-grazed.[1] The use of an intensive rotational grazing system provides environmental benefits, such as enhanced soil quality, minimal soil erosion,[2] improved stream bank quality,[3] and enhanced wildlife habitat.[4]

Assessment of Pollution Risks:

▶ To surface water quality: If the animal ratio is appropriate for the acreage and stream banks are fenced, the system reduces agricultural runoff.[5] The potential for accumulation of soil phosphorus is also reduced if the animals' diet is not supplemented with purchased feed or mineral supplements–both of which can be significant sources of phosphorus inputs to individual operations.[6]

▶ To groundwater quality: The potential for groundwater contamination remains low as long as the site is not located in an area with karst geology or other preferential flow paths to groundwater.[7]

▶ To air quality: Ammonia volatilization is minimal.[8]

▶ To public health: Pathogen transport is minimal due to the reduced volume of surface runoff, which is further reduced if combined with filter strips.[9]

Hoop Houses

Hoop houses are structures that house hogs indoors, but allow them freedom of movement. Hoop houses are built on arched metal frames covered with a tarp. Sidewalls reach four to six feet above ground level on the sides, while end walls are typically tarps or plywood doors that can be opened for ventilation.[10] The houses are bedded deeply with straw to absorb urine and moisture in feces. The straw binds with manure and urine, keeping the waste in a more solid form than liquid-flushing systems used by industrial confinement facilities. Typically, pigs are not confined in pens in hoop houses.[11]

In a system invented by Virginia farmer Joel Salatin, egg layers are allowed to graze through the use of a mobile "eggmobile."

Joel Salatin

Assessment of Pollution Risks:

▶ To surface water quality: Hoop houses eliminate the risk of accidental discharges from manure storage ponds and lagoons during precipitation events because the manure/straw mixture is in a more solid form and the tarp provides rain protection.[12]

However, once the bedding is cleared out of the house (two or three times a year), it is either composted or, having been composted in the house, it is directly spread on fields. Proper land application of the manure is needed to prevent polluted runoff.[13]

▶ To groundwater quality: In order to prevent seepage of nitrate to groundwater, an impermeable barrier, such as a concrete floor, should be constructed between the bedding/ manure mixture and the underlying soil. The potential for the downward movement of this accumulated nitrate nitrogen to groundwater can be reduced by proper siting.[14]

▶ To air quality: If the accumulating bedding/manure mixture is not overly compacted, ammonia nitrogen mineralized from organic nitrogen compounds may oxidize to nitrate nitrogen. If the potential for nitrification is high, the potential for the formation and release of hydrogen sulfide will be low. The likelihood of the release of both ammonia nitrogen and hydrogen sulfide to the atmosphere depends on the ability to maintain conditions in the bedding/manure mixture that are conducive to aerobic microbiological activity.[15]

▶ To public health: Some producers have found that the use of hoop houses eliminates the need for the routine incorporation of antibiotics in feed, because this method of production produces healthy hogs.[16] Additionally, weaning the pigs on deep straw may prevent them from being exposed to pathogens that exist on bare, urine- or manure-covered floors.[17] Pathogens generally are poor competitors in environments with diverse microbial populations and especially in aerobic environments because pathogens usually are anaerobes. Thus, significant pathogen reduction in the bedding/ manure mixture in hoop houses is a reasonable expectation if the mixture is removed infrequently and especially if there is an elevation in temperature due to microbial heat production.[18]


Weaning pigs on deep straw may prevent them from being exposed to pathogens that exist on bare, urine- or manure-covered floors.

Diane Halverson

Composting

Composting is a biological process in which aerobic bacteria convert organic material into a soil-like material called compost that reduces erosion and enhances organic matter, soil quality, and nutrients.[19] In a simple composting system, material is laced in long rows called windrows and turned occasionally to ensure that the material is mixed well. In a complex system, odoriferous materials can be processed in drums, trenches, or tunnels for initial processing, and then cured in a covered facility.[20] Several universities are studying the use of complex systems for swine and dairy waste that involve some dewatering of the waste and adding dry materials or earthworms.[21] Most composting is aerobic, but some producers are turning to anaerobic composting systems in which methane gas is produced for electricity.[22] As with other treatment systems, proper siting of the composting away from waterways and wells is essential to prevent pollution.[23]

Assessment of Pollution Risks:

▶ To surface water quality: Composting reduces the nitrogen content of manure.[24] A roof over the compost site and a system to capture and dispose of leachate and runoff can protect adjacent surface waters.[25] Threats to water quality can be prevented by off-site disposal of composted waste when land resources for on-site utilization of manure are limited.[26]

▶ To groundwater quality: Paving and covering sites used for composting help prevent groundwater contamination.[27]

▶ To air quality: With proper management, release of objectionable odors can be minimized. However, significant odor problems can occur during the start-up of the composting process. Due to the low carbon/nitrogen ratio of animal manure, some volatilization of ammonia nitrogen is unavoidable.[28] To minimize ammonia nitrogen volatilization, readily biodegradable organic carbon, e.g., paper, leaves, or sawdust, must be added to increase the carbon/nitrogen ratio to about 30:1. Otherwise, nitrogen levels in both wet and dry atmospheric deposition will increase with subsequent adverse surface water quality impacts.[29]

▶ To public health: When thermophilic temperatures are achieved, pathogen densities are reduced substantially.[30]

TECHNOLOGY-BASED APPROACHES

Anaerobic Digestion

Anaerobic digestion uses microbes to convert the carbon fraction of livestock and poultry manures to methane and carbon dioxide, commonly referred to as biogas.[31] Digesters create an effluent that has a different chemical composition than raw manure.[32] In all digesters, liquid effluent remains after the process that must be stored until it can be applied to crops.[33] Anaerobic digestion has been used for more than 50 years to stabilize wastewater treatment sludges, and at the same time, reduce pathogen densities.[34]

Releases of noxious odors, ammonia nitrogen, and methane from anaerobic lagoons can be controlled using a flexible, gas-tight cover resulting in a covered lagoon digester.[35] Other types of digesters used successfully with animal wastes include completely mixed and plug-flow digesters. A complete mixed digester treats slurry manure in above- or below-ground tanks.[36] A plug-flow digester typically is an in-ground rectangular trench lined with an impermeable material and covered with a gas-tight flexible membrane cover.[37] These types of digesters are usually insulated and heated using a fraction of the biogas recovered as fuel. Thus, stabilization and the total yield of biogas are constant throughout the year. In contrast, the degree of stabilization and biogas yield from covered lagoon digesters varies with time of year, particularly in colder climates.

Assessment of Pollution Risks:

▶ To surface water quality: Anaerobic digestion can significantly reduce the concentration of oxygen demanding carbon compounds in animal manure. However, the concentration of these compounds will still exceed concentrations in untreated municipal wastewaters.[38] The process converts organic forms of nitrogen to ammonia nitrogen, but does not reduce phosphorus.[39] Reports of phosphorus reduction associated with anaerobic digestion are the result of phosphorus in settled solids accumulated in digesters.[40] The use of anaerobic digesters requires storage tanks or ponds. Therefore, the potential for accidental discharges due to mismanagement or during extreme precipitation events is not eliminated.

▶ To groundwater quality: The risk associated with lined covered-lagoon digesters varies and depends on soil type, geology, depth to seasonally high groundwater, and the method, quality, and management of the liner.[41]

▶ To air quality: Anaerobic digestion with biogas utilization or flaring substantially reduces methane and noxious odor emissions.[42] The system also results in lower ammonia emissions than open-air lagoons, although there still may be some volatilization when the digested manure is stored in a pond prior to land application or when it is land applied without proper injection.[43]

▶ To public health: Risk can be variable, depending upon whether the digesters are heated. Pathogenic bacteria can be significantly reduced or even essentially eliminated if the digesters are heated.[44]

Wetlands Treatment

Constructed wetlands have been used successfully for the tertiary treatment of municipal and industrial wastewaters to further reduce concentrations of nitrogen, phosphorus, biochemical oxygen demand, and suspended solids before discharge to surface waters.[45] These reductions are the result of a combination of microbial activity, plant uptake of nitrogen and phosphorus, and physical/chemical processes such as ammonia volatilization and phosphorus adsorption.[46] In this system, a wetland is constructed that filters wastewater using plants, soil, and water. After treatment, the effluent is sprayed onto crops. Seasonal weather conditions, such as cold and drought, may make the system less reliable.[47] For relatively dilute waste streams, such as swine and dairy flush waters after removal of manure solids, constructed wetlands may be a viable management option prior to storage.[48] For other animal waste, solids must be removed before the wastewater is put in the wetland and pretreatment is necessary to ensure that high ammonia concentrations do not kill off plant life.[49] These steps can be accomplished through the use of an anaerobic or aerobic lagoon, a sequencing batch reactor, a digester, or another system. Thus, constructed wetlands should be viewed as a component of a larger system to treat waste.[50]

Assessment of Pollution Risks:

▶ To surface water quality: Under normal weather conditions, any polluted runoff that results will be far less toxic than that of an anaerobic lagoon. Studies vary on pollutant reduction results, with some studies reporting reductions of half or more in nitrate, ammonia, and oxygen-depleting substances and suspended solids for dairy waste. For swine waste, some studies have found that the system removes most of the nitrogen, phosphorus, and suspended solids,[51] while other studies have found less phosphorus reduction.[52] However, excess rainwater could flood the system, so a holding pond should be constructed near the wetland to accept stormwater and act as a settling basin for removing solids.[53]

▶ To groundwater quality: The risk wetlands treatment poses to groundwater is unknown. However, like lagoon systems, siting may be a concern. If a wetland is located on porous soils, karst geography, or areas near agricultural drainage wells, groundwater contamination may be a high risk.

▶ To air quality: Wetlands may emit some ammonia, which could be reduced if the wastewater is nitrified prior to wetland treatment.[54]

▶ To public health: The risk is unknown.

Sequencing Batch Reactor

The aerated sequencing batch reactor (SBR) process is a variant of the activated sludge process, which has been used for over 75 years to treat municipal and industrial wastewater. The SBR maintains an adapted microbial population to convert the fraction of organic compounds in solution into microbial biomass that subsequently can be removed by settling or filtration. With the conventional activated sludge process, a portion of solids separated by settling after aeration is returned to the aeration basin to maintain the desired microbial population and to minimize wastewater retention time and tank size.[55]

Wastewater treatment using the conventional activated sludge process uses a continuous flow mode of operation. In contrast, SBRs are operated as batch reactors with the following sequence of operations. First, the reactor is filled with untreated wastewater, which is mixed with some fraction of the settled solids from the previous batch. Second, there is a period of aeration that also provides mixing. Third, aeration is terminated to allow suspended solids (particulate matter) to settle and separate. Finally, the clarified effluent is discharged, a fraction of the settled solids is removed, and the process sequence is repeated. The principal advantage of this batch mode method of operation, especially in municipal and industrial wastewater treatment, is that it offers more precise process control. With operating conditions conducive for nitrification, a substantial degree of nitrogen reduction through nitrification-denitrification can be achieved.[56]

Assessment of Pollution Risks:

▶ To surface water quality: The water pollution potential of clarified effluent from a SBR treating animal wastes is substantially lower than untreated waste. A university pilot program for swine waste found that the system removed chemical oxygen demand and total nitrogen by more than 90 percent, and volatile solids and phosphorus by more than 70 percent.[57] Another pilot project showed that over 60 percent of the nitrogen was removed using this system.[58] Since wastewater is in a tank, there is little likelihood of breaches.

▶ To groundwater quality: Risk is lower than using a lagoon since the manure is in a tank.

▶ To air quality: With nitrification, ammonia nitrogen emissions will be reduced. However, nitrous oxide emissions resulting from denitrification and noxious odors from stored solids may occur.[59]

▶ To public health: Significant reductions in pathogen densities in the clarified effluent are obtainable. However, there will be a concurrent concentration of these organisms in the settled solids due to sorption onto particles.[60]

Lagoon Covers

Placing a cover over a lagoon offers an approach that can reduce emissions. A number of different covers have been studied, each of which have benefits and drawbacks. The least costly option entails the use of biofilters, readily available materials that are blown onto the surface of the lagoon, such as peat, moss and straw. Other materials that may be used include plastic mats, polystyrene foam, air filled clay balls, geo-textile membranes, such as high-density polyethylene or reinforced polypropylene materials, and pumice or construction matting.[61] The problem with biofilters is that they need to be replaced often to prevent the material from sinking to the bottom of the lagoon.[62] Other cover options include rigid concrete or wood lids or roofs made of fiberglass.[63]

Assessment of Pollution Risks:

▶ To surface water quality: The risk is unknown, but it is unlikely that biocovers could prevent a major lagoon breach or spill. Impervious covers might sink during a storm event. Rigid covers and impervious covers would divert rainwater.

▶ To groundwater quality: Covers are not designed to address groundwater quality.

▶ To air quality: One study showed that straw covers reduce ammonia emissions, while peat moss absorbs ammonia and reduces nitrogen losses by 40 to 60 percent.[64] Another study found that a straw cover with a thickness of 12 inches reduced ammonia and hydrogen sulfide emissions by 80 percent, while putting straw on top of a thin geo-textile cover also resulted in significant reductions in the emissions of these gases.[65] However, in Renville County, Minnesota, ValAdCo, a large hog farming cooperative, continued to have air quality violations depsite the use of straw and cloth covers, which may be the result of the failure of the cover systems or mismanagement of them.[66] Rigid covers can reduce ammonia emissions between 80 and 95 percent.[67]

▶ To public health: The risk is unknown.

Lagoon Liners

Many earthen lagoons and storage ponds used for livestock and poultry manure have been constructed without any attempt to prevent seepage through the soil profile to groundwater except by compaction of the soil. Neither clay nor concrete liners are risk free with respect to seepage. Clay liners can crack if allowed to dry after a lagoon or storage pond is emptied. Concrete liners can also crack due to thermal expansion and contraction and settling. Also, reliable sealing of necessary expansion joints can be problematic.

Assessment of Pollution Risks:

▶ To surface water quality: Lagoon liners will not protect surface water quality directly but will prevent the base flow discharge of contaminated groundwater to adjacent surface waters.

▶ To groundwater quality: Numerous studies have found seepage through clay liners due to: 1) cracking during wet/dry and/or freeze/thaw cycles, 2) penetration by worms, roots, or rodents, 3) physical damage due to erosion of lagoon berms and agitation during pumping, and 4) liner collapse due to external pressure and groundwater intrusion.[68] Geo-textile materials are less permeable than clays but proper installation is essential. Geo-textile lagoon liners also are subject to the possibility of physical damage during agitation and pumping and due to burrowing animals.[69] Concrete liners can offer greater protection, but concrete can crack if specifications related to soil suitability and structural reinforcement are not followed; however, there are presently no such requirements.[70] Plastic liners composed of impervious materials, like those presently required by EPA for solid waste lagoons, but not for animal waste lagoons, are another option.[71] However, plastic liners also must be constructed properly to prevent problems.

▶ To air quality: Liners are not designed to address air quality.

▶ To public health: Where seepage from lined or unlined lagoons can enter groundwater without filtration through unsaturated soil, discharge of bacterial pathogens to groundwater is likely. Even where seepage is filtered through unsaturated soil, viral pathogens may be transported to groundwater.[72] (See Chapter Four.)

Many of the alternatives to the lagoon and sprayfield system for managing livestock and poultry manure described in this chapter are viable, and provide substantially greater protection to public health and the environment than the lagoon and sprayfield system. The options are available, but factory farms will not begin to use them as long as they are able to continue to externalize their pollution costs by dumping manure into the environment. However, for any method, surface water, groundwater, and public health will only be protected if manure is applied properly to the land.

CHAPTER 6

POLICY RECOMMENDATIONS

The numerous studies summarized in this report make it clear that the reliance on lagoon and sprayfield systems to store and treat animal wastes harms the health of nearby communities and pollutes the environment in numerous ways. The promulgation of EPA's new technology rules ("effluent guidelines") under the Clean Water Act presents a major opportunity to phase out open-air lagoons and sprayfields on large-scale animal operations, and to promote a more sustainable animal production system. Sustainable facilities limit the number of animals that they confine to generate only the amount of waste that can be used as fertilizer. For that reason, sustainable systems generate much less pollution and are at much less risk of catastrophic failure than lagoon and sprayfield operations.

PHASE OUT THE USE OF LAGOONS

The concentration and industrialization of animal production in this country has resulted in single facilities that produce thousands of animals and generate millions of gallons of manure. Industry trends indicate the likelihood of further concentration, resulting in more and larger lagoons if regulations are not tightened.[1]

EPA's existing technology standard allows lagoons that pollute surface water in a variety of ways indirectly to claim that they do not discharge and, therefore, that they do not need a Clean Water Act discharge permit. This has been a major factor in widespread lagoon adoption by concentrated animal feeding operations (CAFOs). (Another factor has been the fact that lagoons have been considered the least costly manure storage option.) When the existing technology standard was promulgated over twenty years ago, animal operations were smaller and lagoons were built on a much smaller scale. Today with the enormous quantities of manure that is generated and stored in lagoons, there are multiple ways for discharges to occur through the air, surface water, and groundwater. However, despite the growing body of evidence that these huge lagoons pollute the environment in numerous ways, EPA's proposed regulations would allow the riskiest lagoons to continue to operate and new lagoons to be built. Moreover, EPA envisions little or no water quality monitoring to determine if pollution problems are occurring.

Ban New Lagoons To protect the environment and public health, EPA should use all regulatory avenues possible to ensure that no new lagoons will be built. It is not enough to simply require that new lagoons to be lined and covered. While these measures may be appropriate as an interim step for existing operations to mitigate some potential problems, these approaches fail to address problems associated with lagoon breaches and overflows. For example, cracks can develop even in lagoons lined with concrete, and seepage can be a problem for lagoons lined with clay, and neither liners nor covers will prevent overflows in significant rain events. (See Chapter Four.) The storage of vast quantities of liquid manure in lagoons presents an unacceptable risk to public health and the environment. Weather events, human error, and system failures have resulted in numerous problems. EPA should not allow any CAFOs to build new lagoons.[2]

Encourage the Use of Sustainable Animal Production Systems There are sustainable animal production systems that do not impose high risk of pollution and other harms and that minimize cumulative impacts from animal production. A key component of a sustainable animal production system is that the scale of the system does not exceed the capacity of the local region to use the waste from the system beneficially. In other words, the total number of animals in a watershed should not exceed the nutrient requirements of available crop and/or timberland in that watershed. Generally, sustainable animal production systems are integrated with crop, forage, or pasture production and on-farm, best-management practices to prevent stormwater runoff from farm fields and barns. These sustainable systems are true pollution prevention systems.

While the U.S. Department of Agriculture and EPA Unified Strategy for Animal Feeding Operations[3] included an overarching principle that sustainable livestock systems be supported, this concept has not been incorporated into EPA's proposed regulations for new CAFOs. The new technology rules should encourage all new CAFOs to adopt sustainable livestock systems that protect air, surface, and water resources.

Phase Out Existing Lagoons Existing lagoons should be phased out over a five-year period. The corporations that own the animals and reap most of the profit can and should be held responsible for paying the costs of installing technologies that do not rely on lagoons and sprayfields. Alternatives to lagoons are available and now in use at farm operations that turn a profit. During this period, existing operations should be required to monitor their surface and groundwater quality to ensure that no discharges occur, and be required to line and cover their lagoons to prevent further contamination. Berms should be required to be built surrounding existing lagoons and be large enough to hold the entire contents of lagoons should they burst. Existing operations should be prohibited from expanding their lagoon systems in the 100-year flood plain or in any area in which there is a potential for seepage into groundwater that may be hydrologically connected to surface water.

The idea of a phaseout has already gained acceptance in North Carolina, a state that ranks second among hog producing states.[4] In 1999, former Governor Hunt proposed a widespread conversion of swine waste lagoons and sprayfields to new technologies.[5] Although North Carolina's Governor Easley has not publically endorsed former Governor Hunt's phaseout plan since he took office, it is worth mentioning that Governor Easley himself initiated the agreement with Smithfield, requiring the company to convert to "environmentally superior technologies," when he was the state's attorney general.[6]

BAN SPRAYFIELDS AND LIMIT LAND APPLICATION

Sprayfields constitute an integral feature of many lagoon systems, in which the waste is sprayed onto crops or pastureland. The sprayfield system poses significant risks to the environment. Since the waste is sprayed in massive quantities, polluted runoff into surface and groundwater presents a recurring problem, and the spray pollutes the air. EPA's proposed rules attempt to address the land application of manure from CAFOs. However, to do so adequately, the regulations must ensure that manure from CAFOs is applied in a manner that protects the environment and public health. The multitude of environmental risks associated with sprayfields should be comprehensively evaluated.

Rather than allowing manure to be sprayed onto fields, EPA should require that manure that is land applied be injected or incorporated into the soil. Manure should be applied at the rate at which it can be absorbed, and the rate should be based on the most limiting nutrient–whether that is nitrogen or phosphorus–found in recent soil tests. Manure should not exceed safe levels of pathogens, metals, salts, or antibiotics. Manure application should be prohibited in sensitive areas, including floodplains, wetlands, areas that drain into groundwater and drinking water sources, areas close to waterbodies, and highly erodible lands. Finally, manure should not be applied to saturated or frozen ground.

All of the requirements listed above should be explicitly mandated in Clean Water Act NPDES permits, not just in a self-drafted, largely unenforceable nutrient management plan. EPA should also require feedlots meeting the current animal unit threshold to obtain a Clean Water Act permit. All the loopholes that have allowed the majority of large feedlots to evade permitting requirements for almost three decades should be eliminated.

CONTROL ALL CAFO POLLUTION

Control All Surface Water Pollution EPA's primary focus on direct discharges into surface water ignores the harm from the lagoon and sprayfield system on air quality, even when air pollutants degrade surface water quality. Up to 80 percent of a lagoon's nitrogen escapes into the air either from the lagoon or during land application as ammonia, which is then deposited into streams or coastal waters and causes fish kills and algal blooms.[7]

The agency also directs little regulatory attention to groundwater quality, though seepage from lagoons and sprayfields is a major public health concern. In many places, groundwater connects with surface water or is used for drinking water. Numerous studies have found groundwater impacts from the use of lagoons due to events such as wet/dry cycles, worms, roots, and freeze/thaw cycles.[8] In 1996, the Centers for Disease Control linked the high nitrate levels in Indiana well water near feedlots to spontaneous abortions in humans.[9] EPA should require monitoring and controls to prevent groundwater contamination of surface water.

EPA Should Address All Clean Water Act Pollutants The regulatory action focuses almost exclusively on nutrients, ignoring resistant bacteria, antibiotic residues, and other pollutants in manure than can cause gastroenteritis and other illnesses. Though swine manure contains 100 to 10,000 times the number of pathogens in crop-applied hog waste than are allowed in treated human waste, lagoons are not designed to reduce concentrations of these pathogens.[10] Yet EPA's proposed standards for land application do not even address pathogens. EPA should set limits for pathogens and heavy metals in land-applied waste, as it does now for sewage sludge, and also require reductions in the levels of other pollutants. Discharges from land application areas should be considered violations of a Clean Water Act permit.

In addition to ensuring that the technologies that the agency recommends do not harm surface water, directly or indirectly, the agency should also provide an incentive to facilities that take proactive steps to minimize other impacts on the environment. Under this approach, multi-media permit holders that agree to monitor and control all discharges into air, water, and groundwater in advance of federal requirements would obtain benefits like a longer compliance period or a longer permit term.

Finally, EPA should join other federal agencies in a comprehensive examination of the multiple problems generated by concentrated, industrialized animal production systems. This review should include the costs and risks imposed on society and the federal subsidies expended in promoting these systems. Where EPA lacks legal authority under the Clean Water Act to address CAFO pollution, EPA should evaluate other legal avenues, including use of the Safe Drinking Water Act and the Clean Air Act.

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ATTACHMENT 33

Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations

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A consensus of the Workgroup on Community and Socioeconomic Issues was that improving and sustaining healthy rural communities depends on integrating socioeconomic development and environmental protection. The workgroup agreed that the World Health Organization's definition of health, "a state of complete physical, mental and social well-being and not merely the absence of disease or infirmity," applies to rural communities. These principles are embodied in the following main points agreed upon by this workgroup. Healthy rural communities ensure *a*) the physical and mental health of individuals, *b*) financial security for individuals and the greater community, *c*) social well-being, *d*) social and environmental justice, and *e*) political equity and access. This workgroup evaluated impacts of the proliferation of concentrated animal feeding operations (CAFOs) on sustaining the health of rural communities. Recommended policy changes include a more stringent process for issuing permits for CAFOs, considering bonding for manure storage basins, limiting animal density per watershed, enhancing local control, and mandating environmental impact statements. *Key words:* animal confinements, environmental impact, livestock, mental health, odor, poultry, right-to-farm legislation, swine. *Environ Health Perspect* 115:317–320 (2007). doi:10.1289/ehp.8836 available via *http://dx.doi.org/* [Online 14 November 2006]

Background and Recent Developments

The agricultural community in areas of large-scale livestock production. The rural and agricultural community has changed dramatically over the past half century. The trends include an overall reduction in the number of farms, an increase in size of the farms, and economic concentration in the industries that supply inputs and purchase commodities from farms. The structure of the pork industry has also changed dramatically during the past three decades. The number of hog producers in the United States was more than 1 million in the 1960s but fell to about 67,000 by 2005 [U.S. Department of Agriculture (USDA) 2005]. Although the total inventory of hogs has changed little over the years, the structural shift toward concentration has been dramatic with the 110 largest hog operations in the country, each of which has over 50,000 hogs, now constituting 55% of the total national inventory (USDA 2005). The swine industry includes the following types of producers: small independent "niche" operators who often market organic pork to local markets, traditional independent operators, and large family or unaffiliated corporations. Former independent operators are increasingly raising livestock on contract for larger corporations. According to the U.S. Government Accountability Office, in 1999 contract production constituted more than 60% of total hog output and 35% of the cattle market (U.S. Government Accountability Office 2005), while poultry is produced almost entirely via contracts. Corporate producers or incorporated family-based operations employ from a few individuals to several hundred. Most often upper management and many of the workers in such operations do not come from or live in the vicinity of concentrated animal feeding operations (CAFOs).

The community of people living in the region of large-scale livestock production consists of residents of small family farms (that may or may not produce pork), workers at the production facilities, rural nonfarm residents, and the residents of neighboring towns. The challenges CAFOs place on neighbors were extensively reviewed in 1996 (Thu 1996) and again in a 2002 report accompanied by a number of consensus recommendations for the future of the hog industry in Iowa (Iowa State University and University of Iowa 2002). A number of additional scientific reviews and symposia summaries have been issued (Centers for Disease Control and Prevention 1998; Cole et al. 2000; Donham 2000; National Academy of Sciences 2002; Schiffman et al. 2000; Thu 2002).

Economic health. Economic concentration of agricultural operations tends to remove a higher percentage of money from rural communities than when the industry is dominated by smaller farm operations, which tend to circulate money within the community. Goldschmidt (1978) documented this as early as 1946 in California, one of the first states where industrialized agriculture developed. Specifically, he compared two agricultural communities, one dominated by larger industrialized farms with absentee ownership and a high percentage of hired farm labor, and the other community was dominated by smaller owner-operated farms. The latter community was found to have a richer civic and social fabric with more retail purchases made locally and with income more equitably distributed. A similar study by MacCannell (1988) of comparable types of communities found that the concentration and industrialization of agriculture were associated with economic and community decline locally and regionally. Studies in Illinois (Gomez and Zhang 2000), Iowa (Durrenberger and Thu 1996), Michigan (Abeles-Allison and Conner 1990), and Wisconsin (Foltz et al. 2002) demonstrated decreased tax receipts and declining local purchases with larger operations. A Minnesota study (Chism and Levins 1994) found that the local spending decline was related to enlargement in scale of individual livestock operations rather than crop production. These findings consistently show that the social and economic well-being of local rural communities benefits from increasing the number of farmers, not simply increasing the volume of commodity produced (Osterberg and Wallinga 2004).

Physical health. There have been more than 70 papers published on the adverse health effects of the confinement environment on swine producers by authors in the United States, Canada, most European countries, and Australia (Cormier et al. 1997; Donham 2000; Donham et al. 1977, 1982, 1986, 1990, 2002; Kirkhorn and Schenker 2002; Kline et al. 2004; Preller et al. 1995; Reynolds et al. 1996; Rylander et al. 1989; Schiffman

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et al. 1995; Schwartz et al. 1992; Thu et al. 1997; Wing and Wolf 2000). It is clear that at least 25% of confinement workers suffer from respiratory diseases including bronchitis, mucus membrane irritation, asthmalike syndrome, and acute respiratory distress syndrome. Recent findings substantiate anecdotal observations that a small proportion of workers experience acute respiratory symptoms early in their work history that may be sufficiently severe to cause immediate withdrawal from the work place (Dosman et. al. 2004). An additional acute respiratory condition, organic dust toxic syndrome, related to high concentrations of bioaerosols in livestock buildings occurs episodically in more than 30% of swine workers.

Environmental assessments of air quality inside livestock buildings reveal unhealthful concentrations of hydrogen sulfide, ammonia, inhalable particulate matter, and endotoxin (Iowa State University and University of Iowa 2002; Schenker et al. 1998). While there is less information on adverse effects among residents living in the vicinity of swine operations, that body of literature has been growing in recent years (Avery et al. 2004; Bullers 2005; Centers for Disease Control and Prevention 1998; Kilburn 1997; Merchant et al. 2005; Mirabelli et al. 2006a; Reynolds et al. 1997; Schiffman et al. 1995, 2000; Thu 2002; Thu et al. 1997; Wing and Wolf 2000).

Thu et al. (1997) documented excessive respiratory symptoms in neighbors of large-scale CAFOs, relative to comparison populations in low-density livestock-producing areas. The pattern of these symptoms was similar to those experienced by CAFO workers. Wing and Wolf (2000) and Bullers (2005) found similar differences in North Carolina. A case report associated with hydrogen sulfide exposure from a livestock processing facility in South Sioux City, Nebraska, revealed excessive diagnoses of respiratory and digestive disturbances in people living nearby (Campagna et al. 2004). Schiffman and colleagues reported that neighbors of confinement facilities experienced increased levels of mood disorders including anxiety, depression, and sleep disturbances attributable to exposures to malodorous compounds (Schiffman et al. 1995, 2000). Avery et al. (2004) found lower concentration and secretion of salivary immunoglobulin A among swine CAFO neighbors during times of moderate to high odor compared with times of low or no odor, suggesting a stress-mediated physiologic response to malodor (Shusterman 1992).

Community environmental air quality assessments have shown concentrations of hydrogen sulfide and ammonia that exceed U.S. Environmental Protection Agency (U.S. EPA) and Agency for Toxic Substances and Disease Registry recommendations (Reynolds et al. 1997). A recent study revealed that children living on farms raising swine have an increased risk for asthma, with increasing prevalence of asthma outcomes associated with the increased size of the swine operation (Merchant et. al. 2005). Children in North Carolina attending middle schools within 3 miles of one or more swine CAFOs and children attending schools where school staff report CAFO odors in school buildings were found to have a higher prevalence of wheezing compared with other middle school children (Mirabelli et al. 2006a, 2006b). It should be noted that these studies (although controlled) lack contemporaneous exposure assessment and health outcomes ascertainment. Additional research to include environmental exposure data related to biomarkers of response is needed.

Mental health. Living in proximity to large-scale CAFOs has been linked to symptoms of impaired mental health, as assessed by epidemiologic measures. Greater self-reported depression and anxiety were found among North Carolina residents living near CAFOs (Bullers 2005; Schiffman et al. 1995). This finding was not corroborated in a small study by Thu et al. (1997) of depression among people living near to or far from CAFOs. However, it should be noted that the study of Thu et al. differed in that residents were not asked to report on their mental state during an actual odor episode as was the case in the study by Schiffman et al. (1995).

Greater CAFO-related posttraumatic stress disorder (PTSD) cognitions have been reported among Iowans living in an area of CAFO concentration compared with Iowans living in an area of a low concentration of livestock production (Hodne CJ, unpublished data). PTSD cognitions were consistent with interviewees' multiple concerns about the decline in the quality of life and socioeconomic vitality caused by CAFOs, in areas of CAFO concentration with declining traditional family farm production.

Social health. One of the most significant social impacts of CAFOs is the disruption of quality of life for neighboring residents. More than an unpleasant odor, the smell can have dramatic consequences for rural communities where lives are rooted in enjoying the outdoors (Thu 2002). The encroachment of a large-scale livestock facility near homes is significantly disruptive of rural living. The highly cherished values of freedom and independence associated with life oriented toward the outdoors gives way to feelings of violation and infringement. Social gatherings when family and friends come together are affected either in practice or through disruption of routines that normally provide a sense of belonging and identity—backyard barbecues and visits by friends and family. Homes are no longer an extension of or a means for enjoying the outdoors. Rather, homes become a barrier against the outdoors that must be escaped.

Studies evaluating the impacts of CAFOs on communities suggest that CAFOs generally attract controversy and often threaten community social capital (Kleiner AM, Rikoon JS, Seipel M, unpublished data; 2000; Ryan VD, Terry Al, Besser TL, unpublished data; Thu 1996). The rifts that develop among community members can be deep and long-standing (DeLind 1998). Wright et al. (2001), in an in-depth six-county study in southern Minnesota, identified three patterns that reflect the decline of social capital that resulted from the siting of CAFOs in all six rural communities they studied: *a*) widening gaps between CAFO and non-CAFO producers; *b*) harassment of vocal opponents of CAFOs; and *c*) perceptions by both CAFO supporters and CAFO opponents of hostility, neglect, or inattention by public institutions that resulted in perpetuation of an adversarial and inequitable community climate. Threats to CAFO neighbors have also been reported in North Carolina (Wing 2002). Clearly, community conflict often follows the siting of a CAFO in a community. What is not known is if community conflict resulting from the siting or presence of CAFOs has an impact on the ability of communities to act on other issues.

Environmental injustice. Disproportionate location of CAFOs in areas populated by people of color or people with low incomes is a form of environmental injustice that can have negative impacts on community health (Wing et al. 2000). Several studies have shown that a disproportionate number of swine CAFOs are located in low-income and nonwhite areas (Ladd and Edwards 2002; Wilson et al. 2002; Wing et al. 2000) and near low-income and nonwhite schools (Mirabelli et al. 2006a, 2006b). These facilities and the hazardous agents associated with them are generally unwanted in local communities and are often thrust upon those sectors with the lowest levels of political influence. CAFOs are locally unwanted because of their emissions of malodor, nutrients, and toxicants that negatively affect community health and quality of life. Low-income communities and populations that experience institutional discrimination based on race have higher susceptibilities to CAFO impacts due to poor housing, low income, poor health status, and lack of access to medical care.

Failure of the political process. In 2005 the U.S. Government Accountability Office issued a report on the effectiveness of U.S. EPA efforts in meeting its obligations to regulate concentrated animal feeding operations (U.S. Government Accountability Office 2005). The report identified two major flaws:

a) allowing an estimated 60% of animal feeding operations in the United States to go unregulated, and *b*) lack of federal oversight of state governments to ensure they are adequately implementing required federal regulations for CAFOs. Additionally, many states have not taken a proactive stance to comply with the U.S. EPA regulations. Therefore, the concentration of livestock production, most noted by CAFO-style production, has continued to expand in most states. This has resulted in many rural communities and individuals taking action on their own, through local ordinances or litigation, as they have not been able to find access through usual governmental channels.

Several studies have found that property values decrease when CAFOs move into a community (Abeles-Allison and Conner 1990; Hamed et. al. 1999; Herriges et al. 2003; Palmquist et al. 1997). Neighbors of CAFOs are interested in preventing loss of property value, loss of their homes and land, forced changes in their life style, adverse changes in their communities, and threats to their health (Thu and Durrenberger 1998). The democratic process offers citizens access to lawmakers, to the courts, and to direct action to redress their grievances. However, the legislative process in many states has often been unresponsive to citizen wishes concerning CAFOs (Cantrell et al. 1996). For example, 13 states have enacted laws that inhibit citizens from speaking freely about agriculture if it is disparaging. A representative example can be seen in a South Dakota law that defines disparagement as

> dissemination in any manner to the public of any information that the disseminator knows to be false and that states or implies that an agricultural food product is not safe for consumption by the public or that generally accepted agricultural and management practices make agricultural food products unsafe for consumption by the public. (South Dakota Codified Laws 2006)

All 50 states have some form of right-to-farm legislation. This legislation serves to protect farming operations from zoning laws or lawsuits that would overly restrict the ability of farmers to do business (Chapin et al. 1998; Hamilton 1998). Right-to-farm legislation varies from state to state but may include laws that prevent zoning from limiting farm practices that have substantial detrimental effects on neighbors, such as CAFO production. Right-to-farm laws may also include preemption of other actions of local government that normally could limit what businesses are allowed to do, known as home rule. For example, the Iowa Supreme Court has ruled that county governments cannot use home rule powers or protection of public health to promulgate laws that are more restrictive than state laws currently in force (Worth County Friends of Agriculture v. Worth County, Iowa, 2004). Although local governmental action has been limited by the bias toward agricultural producers, individual actions have not. Courts in several states have ruled that right-to-farm laws give only limited protection from nuisance action. The Iowa Supreme Court in June 2004 found that CAFO immunity provisions written in Iowa statutes were unconstitutional (Gacke v. Pork XTRA 2004). A district court in Illinois granted a temporary injunction stopping the construction of a nearby CAFO based on an anticipatory nuisance premise (Nickels et al. vs. Burnett 2002) that such a facility would constitute reasonable interference with neighbors' quality of life.

Most states have enacted some forms of environmental laws aimed at protecting the environment from agricultural discharges or emissions. One form of these laws requires establishment of manure management plans. Typically, these laws call for certain sizes of operations to apply for permits. These permits may include the filing of a manure management plan, which calls for a plan for CAFO operators to manage their manure in a manner to prevent water and soil pollution. However, there is little if any performance inspection or enforcement of these plans (Jackson et al. 2000). Nonenforcement is primarily due to the lack of personnel and technical resources at state environmental agencies. For example, some states may have 2,000 or more such operations but not enough staff to efficiently process permit applications, much less get out into the field to inspect performance of these operations.

Workshop Recommendations

Priority research needs. **Community health studies**. Although sufficient research supports actions to protect rural residents from the negative impacts of CAFOs on community health, additional research could be conducted to further delineate mechanisms of effects and impacts on susceptible subgroups. These areas include psychophysiologic impacts of malodor; impacts of malodor on mental health and quality of life; and respiratory impacts of bioaerosol mixtures, especially among asthmatics, children, and the elderly. Wider and more effective application of community-based participatory research will be important to advance research in these areas.

Sustainability of livestock production. Federal funding for agricultural research should be reoriented to promote innovation in sustainable livestock production.

Translation of science to policy. Requirements for issuing permits for CAFOs should include increased protections for health and the environment including the following:

- CAFOs should be sited and issued permits on the basis of total animal density allowed in a given watershed as determined by the carrying capacity.
- Environmental impact statements should be mandated for all new CAFOs. These should include environmental health, social justice, and socioeconomic issues.
- Decisions to issue permits for CAFOs should be considered in public meetings and decided at the local level.
- CAFOs should be regulated using standards applied to general industry based on the level of emissions and type of waste handling.
- Permits for manure storage basins should require bonding for performance and remediation.
- The current state of knowledge of community impacts of CAFOs warrants support for the American Public Health Association recommendation for a moratorium on all new CAFO construction.

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ATTACHMENT 34

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The Effect of Environmental Odors Emanating From Commercial Swine Operations on the Mood of Nearby Residents

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ABSTRACT: The effect of environmental odors emanating from large-scale hog operations on the mood of nearby residents was determined using the POMS (Profile of Mood States). The scores for six POMS factors and the TMD (total mood disturbance score) for 44 experimental subjects were compared to those of 44 control subjects who were matched according to gender, race, age, and years of education. The results indicated a significant difference between control and experimental subjects for all six POMS factors and the TMD. Persons living near the intensive swine operations who experienced the odors reported significantly more tension, more depression, more anger, less vigor, more fatigue, and more confusion than control subjects as measured by the POMS. Persons exposed to the odors also had more total mood disturbance than controls as determined by their ratings on the POMS. Both innate physiological responses and learned responses may play a role in the impairment of mood found here.

KEY WORDS: Odors, Mood, Pollution, Swine, Psychological effects, Brain-immune connections.

INTRODUCTION

Odors have always been associated with livestock and poultry production [24,55,72,78,79,86,88]. However, odors have recently become a major challenge for the livestock industry due to the present trend toward intensive livestock operations in which large numbers of animals are confined on small areas of land [8,19,51,69,120,122–124,127]. Environmental odors can have a considerable impact upon a population's general well-being, affecting both physiological and psychological status [93,103,128]. Miner [70] concluded that unpleasant odors can affect well-being by "eliciting unpleasant sensations, triggering possible harmful reflexes, modifying olfactory function and other physiological reactions." He also reported that annoyance and depression can result from exposure to unpleasant odors along with nausea, vomiting, headache, shallow breathing, coughing, sleep disturbances, and loss of appetite. Odorous compounds associated with livestock production that are at low concentrations but above odor thresholds are still likely to generate complaints [18,52].

Neutra et al. [77] studied people living near hazardous waste sites, and found that those complaining of odors had a higher number of symptoms than those who did not complain, regardless of proximity to the site. Shusterman [103] reviewed several studies [e.g., 4,37,47,95–97] in which there was a direct relationship between nontoxicological odors and symptomatology. In a variety of settings (municipal, agricultural, and industrial) where airborne toxicants were negligible and odors had been complained about, there was a strong relationship between reported symptoms and odor exposure.

The sources of the odors from swine operations include ventilation air released from swine buildings, waste storage and handling systems including lagoons, and land application of manure to fertilize fields [15]. The odors are produced by a mixture of fresh and decomposing feces, urine, and spilled feed. The more objectionable odors appear to result from anaerobic microbial decomposition of the feces [90]. A broad range of compounds has been identified in livestock manure including volatile organic acids, alcohols, aldehydes, amines, fixed gases, carbonyls, esters, sulfides, disulfides, mercaptans, and nitrogen heterocycles [30,70,71,73,104]. It is likely that the mixture of compounds rather than a single component contributes to the mood changes measured here.

A variety of techniques for reducing odor have been evaluated, but overall the results have been disappointing [123]. Aerobic treatment has been found to be the most effective method to date for deodorizing pig slurry [2,9,11,54,105–107,127]. Odorous compounds can be carried in a plume, and the concentration of these compounds in the plume may not be significantly reduced at distances of 750–1500 feet or more downwind from a source [36]. Dispersion models have been developed to predict the peak and mean concentrations of odors and environmental air pollutants at various distances from the source [20,36,46,80], and complaint patterns at a variety of distances from an odor source have been studied [21].

The purpose of the present study was to use a well-standardized scale to quantify objectively the moods of people living near large-scale hog operations who are exposed to odors. The Profile

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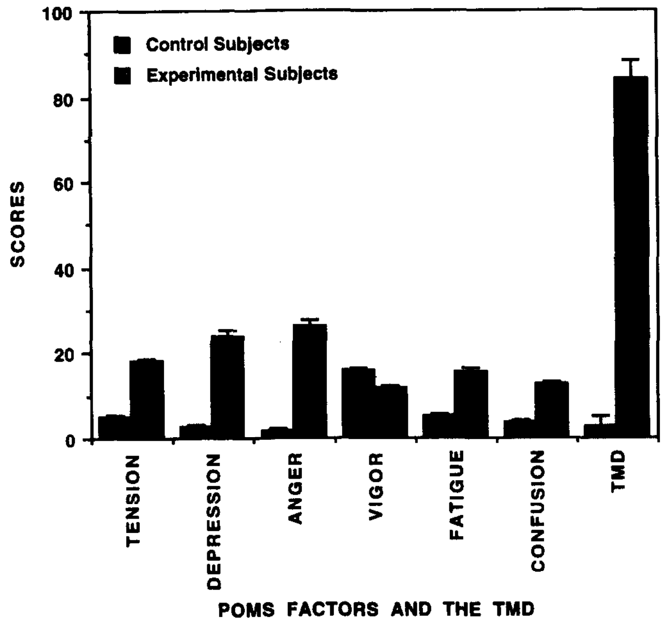

FIG. 1. Mean POMS scores of each factor and the total mood disturbance score (TMD) for experimental and control subjects.

of Mood States questionnaire [65,66] was used to assess mood in persons living near swine operations and in control subjects. This scale has been used extensively in many situations including previous studies that evaluated the effect of pleasant odors on mood [98,99]. The study of mood in persons exposed to odors is important because negative mood has been found to play a role in immunity [16,81,111,125] and can potentially affect subsequent disease.

METHOD

Subjects

Forty-four experimental (persons living near hog operations) and 44 control subjects participated in the study; all of the subjects were residents of North Carolina. The subjects in the two groups (control and experimental) were matched according to gender, race, age, and years of education. Twenty-six subjects in each group were female, and 18 subjects were male. The mean age of the experimental group was 52.0 ± 13.4 years, and the mean age of the control group was 51.7 ± 8.3 years. The experimental group had an average of 12.8 ± 3.3 years of education, and the control group had an average of 13.0 ± 3.1 years of education. The majority of subjects in both groups were employed as skilled laborers. The groups were also matched for the number of chronic illnesses that they had experienced; 14 subjects in each group suffered from allergies. The experimental group lived an average of 5.3 ± 6.5 years near hog operations, with a maximum of 27 years and a minimum of 8 months.

Materials

Subjects in both groups signed a consent form and filled out a general information questionnaire that asked demographic, medical, and dietary information. Mood ratings were obtained from all subjects by filling out Profile of Mood States questionnaires (POMS). The POMS was chosen to measure the impact of the hog odors on mood because it has been shown to be sensitive to transient mood shifts [65,66]. There are 65 adjectives/feelings on the POMS, most of which may be grouped into one of six factors: tension/anxiety, depression/dejection, anger/hostility, vigor/activity, fatigue/inertia, and confusion/bewilderment. Each feeling is rated on a scale from 0 (not at all) to 4 (extremely). The feelings for each factor were added together, according to the POMS manual, to get a total score for that factor. The totals for each factor were then added together, with the vigor/activity factor weighted negatively, to derive a total mood disturbance score (TMD).

Procedure

At the beginning of the study, all subjects filled out the consent form as well as the general information questionnaire. Experimental subjects were asked to complete one POMS questionnaire per day on 4 days when the hog odor could be smelled. The 4 days did not have to be consecutive, and subjects had as long as needed to complete all four POMS questionnaires. Control subjects were asked to complete one POMS per day for 2 days. All subjects were asked to complete the POMS based upon how they recently had been feeling, including at that particular time.

RESULTS

Figure 1 shows the means and standard errors for the experimental group vs. the control group for all POMS factors and the TMD. An analysis of variance was performed to determine if there were any main effects or interactions between group (control or experimental) and gender for each POMS factor and the TMD. Subjects were nested within group and gender. Table 1 gives the results of the analysis. There was a significant difference (at $p < 0.0001$ level) between the control group and the experimental group for all of the POMS factors as well as the TMD. The experimental group had significantly worse scores than the control group for every factor and the TMD. There was a significant main effect of gender for the anger factor, $p < 0.01$, and a significant gender × group interaction for the confusion factor, $p < 0.005$. Males had significantly higher (worse) anger scores than the females. For the confusion factor, scores for experimental males were significantly higher than those for experimental females and control males and females; scores for ex-

TABLE 1

RESULTS OF THE ANALYSIS OF VARIANCE

Effect	Tension	Depression	Anger	Vigor	Fatigue	Confusion	Total Mood Disturbance Score
Group	*	*	*	*	*	*	*
Gender			*				
Group × gender						*	
Subject (group, gender)	*	*	*	*	*	*	*

* Significant at $\alpha = 0.05$ level.

perimental females were significantly higher than those of control males and females. Only scores for control males and control females were not significantly different from each other.

DISCUSSION

The main finding of this study is that persons living near the swine operations who experienced the odors had significantly more tension, more depression, more anger, less vigor, more fatigue, and more confusion than control subjects as measured by the Profile of Mood States (POMS). In addition, persons exposed to the odors also had more total mood disturbance than controls as determined by their ratings on the POMS. These findings are consistent with previous studies in which odors of varying hedonic properties have been found to affect mood [7,32,93,98,99,103,128]. In other settings, odors have also been reported to affect cognitive performance [57,62] and physiological responses including heart rate and electroencephalographic patterns [56,58–61,64].

Possible Causes of Altered Mood

A variety of factors may play a role in the altered mood of residents who are exposed to odors from nearby swine operations. These factors include: a) the unpleasantness of the sensory quality of the odor; b) the intermittent nature of the stimulus; c) learned aversions to the odor; d) potential neural stimulation of immune responses via direct neural connections between odor centers in the brain and lymphoid tissue; e) direct physical effects from molecules in the plume including nasal and respiratory irritation; f) possible chemosensory disorders; and g) unpleasant thoughts associated with the odor.

At moderate to high odor intensities, most persons rate the quality of the odor from the swine operations as unpleasant. The odor is not only perceived while breathing outdoor air but can also be perceived within the homes of nearby residents due to air circulation through open windows and air conditioning systems. The odorant molecules can be absorbed by clothing, curtains, and building materials which act as a sink; the molecules are then released slowly over a period of time from textiles and other materials after the plume has passed the house increasing the temporal exposure to the odor. The intermittent nature of the odors may also be a factor in the mood of persons living near swine operations. Studies of noise have shown that intermittent stimuli produce more arousal and are more likely to affect performance negatively than constant noise [22]. This is due in part to feelings of lack of control over the timing of unwanted transient stimuli. Differences in responses to irregular noise and predictable noise are not only found in humans but in animals as well [27].

Learning (via conditioning) may also play a role in the psychological and physical effects from odors. Conditioned aversions to odors are well-documented in the scientific literature [31,38,44,67,75,119]. Aversive conditioning can occur if environmental odors are associated with an irritant or other toxic chemicals such as pesticides [103]. In addition, conditioned alterations in immune responses using chemosensory (smell and taste) stimuli provide strong evidence for functional relationships between chemosensory centers in the brain and the immune system [1]. Both conditioned immunosuppression and immunoenhancement have been reported using chemosensory stimuli as the conditioned stimulus [1,31,42,43,109,110].

There is a potential for unpleasant odors to influence physical health without involvement of learning or conditioning due to the direct anatomical connections between the olfactory system and the immune system. Brain structures broadly involved in smell [12,35,39,49,82–85,101,112,114–116] can modulate immune responses, especially via the integrated circuitry of the limbic cortex, limbic forebrain, hypothalamus, and brain stem [13,25,26,48,50,76,92,118]. These studies provide an anatomical basis for the possibility that sensory stimulation of the limbic forebrain, hypothalamus, and other odor projection areas of the brain can directly alter immune status. The links between the brain and the immune system are bidirectional [108] so that immune responses can also affect odor centers in the brain [10,94].

Components in the odorous plume may also have direct physical effects on the body. Some of the odorant molecules implicated in malodor from hog farms can cause nasal and respiratory irritation [15,23,29,70,103]. Nasal irritation has been shown to elevate adrenalin [3] which may contribute to feelings of anger and tension. The volatile organic compounds (VOCs) responsible for odors may also be absorbed directly by the body (into the bloodstream and fat stores) via gas exchange in the lungs. Many VOCs that are inhaled into the lungs are known to reach blood and adipose tissue [4,6,53,63,126]. Persons who have absorbed odorants through the lungs can sometimes smell the odor for hours after exposure due to slow release of the odorants from the bloodstream into expired air activating the olfactory receptors. Volatile organic compounds are well known to be eliminated in breath after exposure [89,121], and methods for measuring VOCs in breath have been described [87,89,117]. It is also theoretically possible for some compounds in the plume to be transmitted to the brain via olfactory neurons because a range of agents have been found to reach the brain through the nasal route [28,33,45,74,91,102]. Endotoxin, a component of bacteria, found in the swine house air environment [29], may also be present in the plume. Persons with olfactory dysfunction caused by factors unrelated to swine odor such as concurrent medical conditions, drugs they are taking, or pesticide exposure [100], may find the odor even more objectionable due to their abnormal smell functioning.

Finally, odors may alter mood because they are associated with unpleasant thoughts. Some persons consider the smell from hog farms a taboo odor, which they should not have to endure. For other persons, the odors generate environmental concerns, fear of loss of use and value of property, or a conviction that odors interfere with their enjoyment of life and property. Livestock odors may also be considered inappropriate in certain environments. Odor complaints have been reported to be most frequent among new, large, or recently expanded facilities that are located near existing residences or shopping areas [70,113]. Part of the motivation for odor complaints may be the increased awareness of other environmental agents, such as tobacco smoke, which is malodorous and is considered dangerous to one's health.

Lack of Legislation to Monitor Odor Levels

Odors are not regulated by the Clean Air Act because they are generally regarded as nontoxic [15]. In addition, nonfederal legislation for controlling odors from swine operations is imprecise or lacking in many states. For example, North Carolina Administrative Code Title 15A-02D.0522(c) specifies that "a person shall not cause, allow, or permit any plant to be operated without employing suitable measures for the control of odorous emissions including wet scrubbers, incinerators, or such other devices as approved by the Commission." This regulation is subjective because it gives no provision for either emission standards or ambient air standards. Under this regulation, it appears that as long as a plant has suitable control devices, it is lawful for them to emit offensive odors. In addition, it is unclear what type of operation is to be considered a plant. In contrast, Connecticut's laws on odor emissions set specific standards, as shown in Table

TABLE 2

ACCEPTANCE LIMITS FOR ODORS (FROM 17)

Chemical	ppm by Volume
Acetaldehyde	0.21
Acetic acid	1.0
Acetone	100.0
Acrolein	0.21*
Acrylonitrile	21.4*
Allyl chloride	0.47
Amine, dimethyl	0.047
Amine, monomethyl	0.021
Amine, trimethyl	0.00021
Ammonia	46.8*
Aniline	1.0
Benzene	4.68
Benzyl chloride	0.047
Benzyl sulfide	0.0021
Bromine	0.047
Butyric acid	0.001
Carbon disulfide	0.21
Carbon tetrachloride (chlorination of CS_2)	21.4*
Carbon tetrachloride (chlorination of CH_4)	100.0*
Chloral	0.047
Chlorine	0.314
Dimethylacetamide	46.8*
Dimethylformamide	100.0*
Dimethyl sulfide	0.001
Diphenyl ether	0.1
Diphenyl sulfide	0.0047
Ethanol (synthetic)	10.0
Ethyl acrylate	0.00047
Ethyl mercaptan	0.001
Formaldehyde	1.0
Hydrochloric acid gas	10.0*
Hydrogen sulfide gas	0.00047
Methanol	100.0
Methyl chloride	(above 10 ppm)
Methylene chloride	214.0*
Methyl ethyl ketone	10.0
Methyl isobutyl ketone	0.47
Methyl mercaptan	0.0021
Methyl methacrylate	0.21
Monochlorobenzene	0.21
Monomethylamine	0.021
Nitrobenzene	0.0047
Paracresol	0.001
Paraxylene	0.47
Perchloroethylene	4.68
Phenol	0.047
Phosgene	1.0*
Phosphine	0.021
Pyridine	0.021
Styrene (inhibited)	0.1
Styrene (uninhibited)	0.047
Sulfur dichloride	0.001
Sulfur dioxide	0.47
Toluene (from coke)	4.68
Toluene (from petroleum)	2.14
Toluene diisocyanate	2.14*
Trichloroethylene	21.4

* Exceeds the Threshold Limit Value adopted by the American conference of Industrial Hygienists for 1971.

2 [17]. Similarly, in the Netherlands, regulations are based on accurate records of manure production and bookkeeping, and violations are considered a criminal offense [14].

Regulations need to be established in all 50 states because animal wastes contain high levels of volatile organic compounds that can produce strong odors. The annual production of animal manure in the US in 1987 was estimated at 1.5 billion tons per year, which is enough to apply one ton per acre on each of the 1.9 billion acres of the continental US [14].

Persons exposed to high levels of odor from agricultural sources generally use nuisance laws to protect their rights. However, there are many caveats in nuisance laws that consider a) which party was there first; b) the character of the neighborhood; c) the reasonableness of the use of the land; and d) the nature and degree of the interference [40]. In addition, most states have right-to-farm statutes that supersede nuisance laws in some circumstances [40]. Strong support against nuisance suits involving agriculture is not specific to the United States but is found in the laws of many countries [5]. Suits against agricultural activities based on odor nuisance are harder to prove than those based on water pollution [68]. In addition, nuisance claims fall under state laws, while suits on water pollution are most frequently filed in federal courts.

Conclusion

Odors from swine operations have a significant negative impact on mood of nearby residents. Methods must be found to lower the concentrations of compounds responsible for the odors so that swine operations do not affect the emotional lives of residents in the local vicinities. This may involve legislation that sets standards for odor. In addition, technological solutions must be found to reduce the concentrations of the offending compounds.

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ATTACHMENT 35

United States
Environmental Protection
Agency

Detecting and Mitigating the Environmental Impact of Fecal Pathogens Originating from Confined Animal Feeding Operations: Review

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Detecting and Mitigating the Environmental Impact of Fecal Pathogens Originating from Confined Animal Feeding Operations: Review

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Notice

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Foreword

The U.S. Environmental Protection Agency (EPA) is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how pollutants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory (NRMRL) is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threaten human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

This publication has been produced as part of the Laboratory's strategic long-term research plan. It is published and made available by EPA's Office of Research and Development to assist the user community and to link researchers with their clients.

Sally Gutierrez, Director
National Risk Management Research Laboratory

Table of Contents

List of Figures

List of Tables

1. Introduction and Overview

The trend in animal production has shown a dramatic shift in the last 50-60 years from small family farms and grazing operations towards large commercial confinement operations. Since 1982, animal production at these facilities has nearly doubled while at the same time they have become more spatially concentrated (U.S. Department of Agriculture, National Resources Conservation Services (USDA-NRCS), 2000). Recently, the U.S. Department of Agriculture (USDA) reported that more than 80% of all livestock revenues are generated in confinement facilities that account for a scant 18% of all livestock operations (USDA-NRCS, 2002). In fact, more than 43% of all beef cattle, dairy cattle, swine, and poultry are raised in the largest two percent of operations (Goellehon *et al.*, 2001). The concentration of animals into confinement facilities poses many environmental challenges, among which pathogenic microorganisms of fecal origin are of concern.

The U.S. Environmental Protection Agency (USEPA) defines a concentrated animal feeding operation (CAFO) as an animal feeding facility that houses more than 1,000 animal units (AU), has 300 to 1000 AU but meets certain conditions, or is designated a CAFO by the state (USEPA, 2001). The number of animal units are based on an equivalent number of beef cattle. Therefore, 1,000 AU equals 1,000 beef cattle, 700 mature dairy cattle, 2,500 swine, 5,000 ducks, 10,000 sheep, 55,000 turkeys, or between 30,000 and 100,000 laying hens or broilers depending on the animal waste management system employed. According to National Resources Conservation Service (NRCS) estimates, 11,398 CAFOs (>1000 AU) were in operation in the U.S. in 1997, and comprised five percent of all livestock facilities (USDA-NRCS, 2002). These CAFOs were largely commercial operations (94%) with large revenues. Total agricultural sales for 97.9% of CAFO owners exceeded $500,000 per year. In comparison, non-commercial livestock facilities (intermediate or rural-residence farms) earned 22.3% of all livestock revenue, generating on average $18,500 per farm. Figure 1 shows the distribution of confined poultry, swine, dairy cattle, and feed cattle operations in the U.S. in 1997.

Animal agriculture results in the production of copious amounts of manure, much of which is ultimately used as fertilizer for crops or spread onto land. On a per weight basis, livestock animals produce between 13 and 25 times more manure than humans. Comparing the most recent U.S. census data and USDA livestock reports, it can be estimated that animals produce somewhere between 3 and 20 times more manure than people in the U.S. each year, as much as 1.2 – 1.37 billion tons (wet weight) (American Society for Microbiology (ASM), 1998; USEPA, 2003; USEPA, 2004). This is enough to cover a land mass the size of Rhode Island with more than twelve inches of manure. Even moderate livestock operations can produce as much manure as a small sized city. For example, a 2,500-head dairy cattle operation can produce a waste load similar to a city of 61,000 people. Two important differences are that livestock CAFO animal wastes can be as much as 100 times more concentrated than human wastes, and the treatment of human wastes is required by law prior to discharge into the environment (USEPA, 2001).

Animal wastes contain zoonotic pathogens, which are viruses, bacteria, and parasites of animal origin that cause disease in humans. Diseases that can be caused by zoonotic pathogens include Salmonellosis, Tuberculosis, Leptospirosis, infantile diarrheal disease, Q-Fever, Trichinosis, Cryptosporidiosis, and Giardiasis to name a few. These diseases typically present as mild

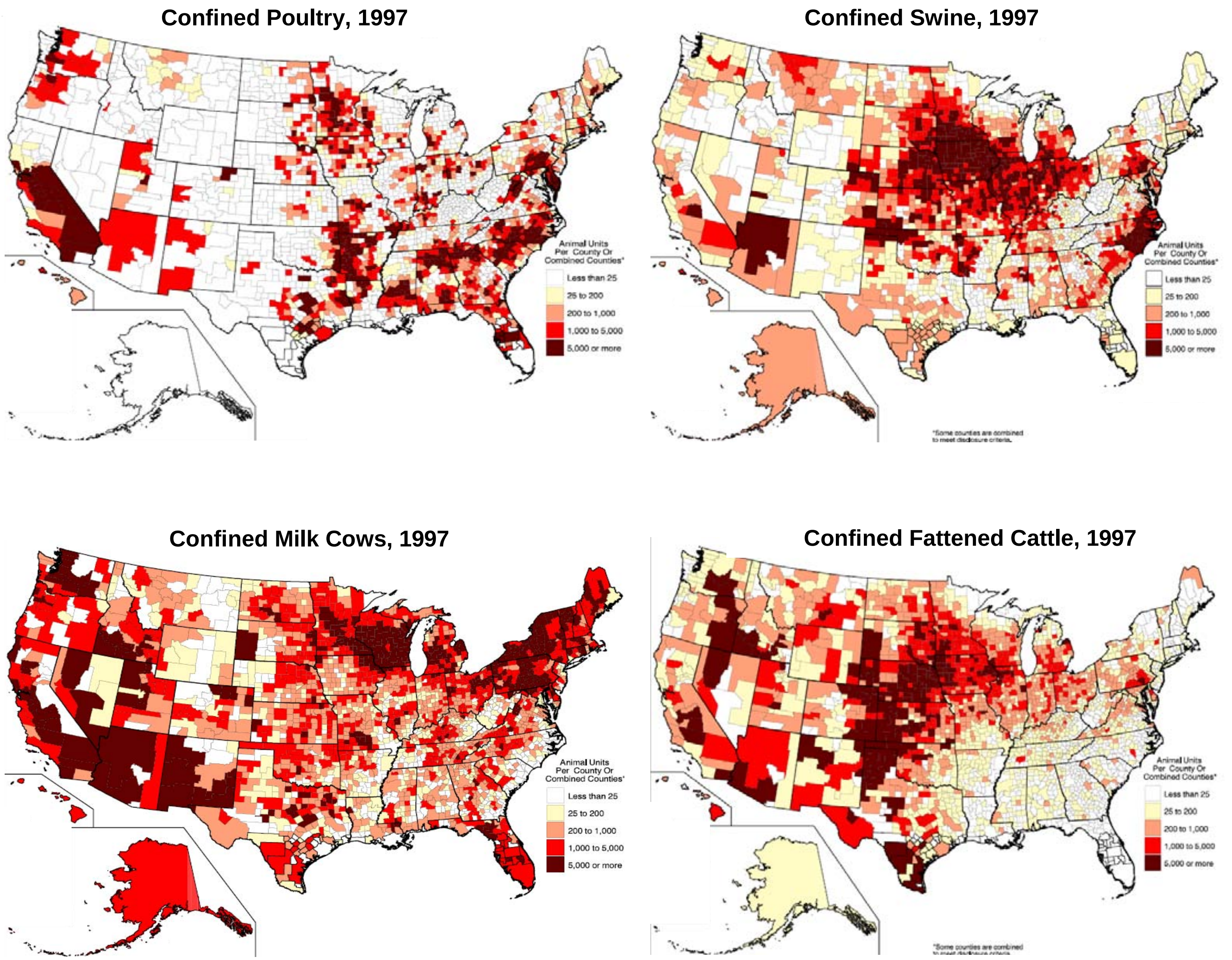

Figure 1. Confined swine, poultry, dairy cattle, and feed cattle per county in 1997 (adapted from USDA-NRCS, 2002).

diarrhea, fever, headaches, vomiting, and muscle cramps. In more severe cases, however, these diseases may cause meningitis, hepatitis, reactive arthritis, mental retardation, miscarriages, and even death, particularly in the immunocompromised. The dosing of livestock animals with copious amounts of antimicrobial agents for growth promotion and prophylaxis may promote antimicrobial resistance in pathogens, increasing the severity of disease and limiting treatment options for sickened individuals (Lee *et al.*, 1994; Marano *et al.*, 2000).

Zoonotic diseases from livestock animals, transmitted through air, water, and food, cause significant human suffering and economic losses in the U.S. every year (Schlech *et al.*, 1983; Besser *et al.*, 1993; MacKenzie *et al.*, 1994; Solo-Gabriele and Neumeister, 1996; Hoxie *et al.*, 1997; Mead *et al.*, 1999; Valcour *et al.*, 2002; Clark *et al.*, 2003). Increasing the concentration of animals in confinement facilities amplifies the potential for localized runoff and contamination, increasing the probability for accidental exposure of susceptible individuals. In fact, living near CAFO operations has been associated with significant deterioration in human health including increased gastrointestinal illness, headaches, sore throats, sinusitis, and childhood asthma (Wing and Wolf, 2000; Merchant *et al.*, 2005). There is increasing evidence that impoverished and nonwhite communities may be burdened with a disproportionate share of not only these negative health outcomes, but also pollution and offensive odors emanating from CAFO facilities (Wing *et al.*, 2000; Wilson *et al.*, 2002; Wing *et al.*, 2002). Based on studies in North Carolina, operations run by cooperate investors may be more likely to be concentrated in poor and nonwhite areas than operations run by independent growers (Wing *et al.*, 2000).

The USEPA recognizes the need to improve manure management practices at confined animal feeding operations (USEPA, 2003). Several other U.S. governmental entities, including the U.S. Department of Agriculture (USDA), U.S. Geological Survey (USGS), and Centers for Disease Control and Prevention (CDC), have also recognized the need for control of pathogens at CAFOs and have robust research and surveillance activities to improve the outcomes for public health and welfare in the U.S. Several recent advances in the fields of medicine, molecular microbiology, engineering, agronomy, and epidemiology are addressing issues pertinent to the control of pathogens from CAFOs at a rapid pace. However, reported literature and research activities can in some cases be divergent between some disciplines and repetitive between others. There is a lack of integration of both knowledge and skills necessary to drive the research in an appropriate direction. As stated by Landry and Wolfe (1999):

> *The range of disciplines conducting fecal bacteria research and the diverse nature of the literature are obstacles to application and synthesis of existing knowledge by animal waste managers and scientists.*

In this report, we synthesize the current state of knowledge regarding pathogen research as it relates to livestock CAFOs, including a summary of research ongoing at USDA and other federal agencies. Pathways for the release of zoonotic agents and antimicrobial-resistant bacteria endemic in animals raised in confinement and their potential to persist in different *milieus* are reviewed. We discuss the impact to the environment and public health and welfare posed by the release of these agents from CAFOs, as well as manure management practices that are employed to mitigate their release into the environment. The objectives of this review are to summarize pathogen issues with regard to livestock CAFOs and identify and discuss gaps in the research that need to be addressed to improve public health.

2. Pathogens

Livestock animals can harbor and shed viruses, bacteria, protozoan parasites, and helminthes that are pathogenic for humans, other domestic animals, and wildlife. Pathogens present in animal carcasses or shed in animal wastes may include rotaviruses, hepatitis E virus, *Salmonella* spp., *E. coli* O157:H7, *Yersinia enterocolitica, Campylobacter* spp., *Cryptosporidium parvum,* and *Giardia lamblia* to name a few (Sobsey *et al.*, 2002). These zoonotic pathogens can exceed millions to billions per gram of feces, and may infect humans through various routes such as contaminated air, contact with livestock animals or their waste products, swimming in water impacted by animal feces, exposure to potential vectors (such as flies, mosquitoes, water fowl, and rodents), or consumption of food or water contaminated by animal wastes (Schlech *et al.*, 1983; Bezanson *et al.*, 1983; Hawker *et al.*, 1998; Valcour *et al.*, 2002; Armand-LeFevre *et al.*, 2005). The consequences of infection by pathogens originating from animal wastes can range from temporary morbidity to mortality, especially in high-risk individuals. Antimicrobial use in animal agriculture may exacerbate the problem by increasing the resistance of these pathogens to therapeutic drugs used to treat human disease.

It has been estimated that 61% of all human pathogens and 75% of emerging human pathogens are zoonotic (Mahy and Murphy, 1998; Murphy, 1998; Taylor *et al.*, 2001; Woolhouse *et al.*, 2002). The overwhelming majority of these pathogenic zoonoses that commonly infect humans are related to animal husbandry practices. Table 1 lists some of the zoonotic pathogens that may be of concern in animal agriculture. Many of these pathogens are endemic in livestock and difficult to eradicate from the animals or their production facilities (Sobsey *et al.*, 2002). For instance, a study of healthy swine on eight farms in Iowa and North Carolina revealed greater than 90% incidence of *Campylobacter coli* in all three growth stages (nursery, grower, and finisher) (Wesley *et al.*, 1998). Similarly, the prevalence of *Salmonella* spp. and *Campylobacter jejuni* have been reported to be as high as 100% in poultry operations, *Yersinia enterocolitica* as high as 18% in swine operations, and *Giardia lamblia* and *Cryptosporidium* spp. as high as 100% in cattle operations (Olson, 2003). The primary reservoir for *E. coli* O157:H7 was determined to be healthy cattle in one study in Canada, although this bacterium is also endemic to swine and sheep (Jackson *et al.*, 1998). In the U.S., *E. coli* O157:H7 infection was widely distributed across all 13 states at an average rate of 1.61% of all cattle when tested in 1994 (Dargatz, 1996). At slaughter, the prevalence of *E. coli* O157:H7 in Scottish cattle may be greater than 13% (Low *et al.*, 2005). Fratamico *et al.*, (2004) tested 687 swine fecal samples from swine operations in 13 of the top 17 swine-producing states and determined that 70% of the samples were positive for shiga-toxin (*stx 1* and *stx 2*) genes. Due to the endemic nature of zoonotic pathogens in livestock, there is a clear need for appropriate management practices at livestock facilities that are protective of human health and the environment and firmly grounded in risk analyses.

The risk of contracting disease following exposure to livestock wastes is dependent on the properties of the infectious agent, the exposed individual, the route of exposure, and the dose. There are a wide range of infective doses for different pathogens as shown (Table 1). For instance, severe gastrointestinal illness may require the ingestion of millions of *Yersinia enterocolitica* bacteria, or as little as 5 to 10 *E. coli* O157:H7 cells (PHAC, 2005). The infectious doses listed in Table 1 were established based on infectivity studies in healthy individuals, and therefore, may not be particularly useful for establishing safe exposure limits for human health. Particularly susceptible individuals such as children, the elderly, or the immunocompromised, which represent nearly 25% of the U.S. population may succumb to infection at much lower doses than the general population (Naumova *et al.*, 2003). For instance, approximately 70% of the diarrhea-associated deaths in the U.S. each year occur among individuals 55 or older.

Regulatory limits on the concentrations of pathogens in the environment protective of human health have not been established. As such, pathogenic organisms are rarely monitored in waste streams from animal feeding operations. Difficulty in quantifying pathogens at relevant concentrations in environmental matrices, the large number of analytical tests that would be required to measure all of the zoonotic pathogens shed in livestock feces, and a lack of epidemiological data to establish appropriate and safe levels of pathogens in the environment have all led to this deficiency. Due to the difficulties in quantifying pathogens, indicators of fecal pollution, including coliform bacteria, fecal coliforms, *E. coli*, and/or Enterococci have been monitored in lieu of overt pathogens for more than 100 years (Smith, 1893; Allen *et al.*, 1952; Kirschner *et al.*, 2004; Byamukama *et al.*, 2005). Epidemiological evidence supports the relationship between the fecal indicator bacteria *E. coli* and enterococci and incidence of gastrointestinal illness following recreational water exposure, and provides the basis for local, state, and federal water quality regulations (USEPA, 1986). However, the works of several researchers has shown that these indicators are not reliable surrogates for many pathogens, including bacteria and most viruses and parasites (Seligmann and Reitler, 1965; Boring *et al.*, 1971; Wetzler *et al.*, 1979; Carter *et al.*, 1987; Geldreich, 1996; Ashbolt *et al.*, 2001; Grabow, 2001; Leclerc *et al.*, 2001; Tillett *et al.*, 2001; Hörman *et al.*, 2004; Harwood *et al.*, 2005). New approaches for detecting pathogens are needed to improve monitoring systems. There also remains a need for epidemiological data to enable the identification of appropriate and safe limits of pathogens in the air, drinking water, recreational water, and in food. Based on surveillance of water and foodborne outbreaks in the U.S., priority for standard methods and recreational and drinking water guidelines should be given to *Salmonella* spp., *Campylobacter jejuni*, *E. coli* O157:H7, *Cryptosporidium*, *Giardia*, and selected viral agents indicative of viral contamination. Priority should be established on the incidence of a particular illness due to a pathogen or the severity of the illness or possibly both.

Table 1. Selected zoonotic pathogens zoonoses that may be of concern for water quality near CAFOs †

Infectious Agent	Infectious Dose	Incubation Period	Disease *Symptoms*	Host Range	Reservoir
Bacterial					
Bacillus anthracis	8000-50000 *(by inhalation)*	2-5 days	**Anthrax, Wool sorter's disease** *Cutaneous – skin lesions, death (5-20%)* *Inhalation – respiratory distress, fever, shock, death* *Intestinal – abdominal distress, fever, septicemia, death (rare)*	Humans, cattle, swine, goats, sheep, horses	Spores remain viable in soil contaminated by animal wastes for years
Brucella spp.	Unknown	Highly Variable 5-60 days	**Brucellosis, Undulant Fever, Bang's Disease, Malta Fever, Mediterranean Fever** *Intermittent fever, headache, weakness, profuse sweating, chills, arthralgia*	Humans, cattle, swine, goats, sheep, deer, caribou, elk, dogs, coyotes	Cattle most common
Campylobacter jejuni	≤500 *(by ingestion)*	1-10 days	**Campylobacter enteritis, Vibrionic enteritis, Traveler's Diarrhea** *Diarrhea, abdominal pain, malaise, fever, nausea, vomiting, septicemia, meningitis, Guillain-Barré syndrome, death (rare)*	Humans, cattle, swine, goats, sheep, poultry, rodents, birds, household pets,	Cattle, swine, sheep, poultry household pets, rodents, birds
Clostridium tetani	Toxin is extremely potent	3-21 days	**Lockjaw, Tetanus** *Painful muscular contractions, abdominal rigidity, spasm, death (30-90%)*	Humans, animals	Intestine of animals and humans, soil contaminated with animal feces
Coxiella burnetii	10 *(by inhalation)*	2-3 weeks	**Q fever, Query Fever, Rickettsia** *Acute febrile disease – chills, headache, weakness, malaise, severe sweats, pneumonitis, pericarditis, hepatitis* *generalized infections – endocarditis*	Humans, cattle, sheep, goats	Sheep, cattle, goats, especially at parturition

† Hazen and Toranaos, 1990; WHO, 1993; DuPont *et al.*, 1995; Morris and Levin, 1995; Geldrich, 1996; ASM, 1998; Haines *et al.*, 2004; PHAC, 2005

Table 1. Selected pathogenic zoonoses that may be of concern for water quality near CAFOs (*Continued*)

Infectious Agent	Infectious Dose	Incubation Period	Disease *Symptoms*	Host Range	Reservoir
Bacterial (*Cont.*)					
Enterohemorrhagic *Escherichia coli* (*E. coli* O157:H7 and others)	5-10	2-8 days	**EHEC, Verotoxin-produding *E. coli*, VTEC, Shiga toxin-producing *E. coli*, STEC** *Hemorrhagic colitis, abdominal pain, bloody diarrhea, fever, hemolytic uremic syndrome, thrombocytopenic purpura, death (in children)*	Humans, cattle, swine, goats, sheep, poultry	Humans and livestock animals
Enteropathogenic *Escherichia coli*	10^8-10^{10} in adults, Unknown in infants	0.5-3 days	**Attaching and effacing *E. coli*, enteroadherant *E. coli*, infantile diarrheal disease** *Watery diarrhea, fever, cramps, vomiting, bloody stool in some cases, serious disease in infants*	Humans (esp. infants), cattle, swine, goats, sheep, poultry	Humans and livestock animals
***Leptospira* spp.**	Unknown, but may be as low as 3	4-19 days	**Leptospirosis, Weil's Disease, Canicola fever, Hemorrhagic jaundice, Mud fever, Swineherd's disease** *Fever, headache, chills, muscle aches, vomiting, meningitis, rash, jaundice death (rare)*	Humans, cattle, swine, horses, dogs, rats, wild animals	Farm and pet animals, rats and rodents (urine and abortion products)
Listeria monocytogenes	Unknown, but likely less than 10^3	3-70 days *(mean = 21)*	**Listeriosis, Listerella** *Fever, muscle aches, nausea, diarrhea, headache, stiff neck, confusion, loss of balance, convulsions miscarriage or stillbirth, premature delivery, death in about 20% of all cases*	Mammals, birds, fish, crustaceans, and insects	Domestic and wild mammals, fowl, and humans (aborted fetuses of livestock animals)
Mycobacterium bovis M. tuberculosis	10 *(by inhalation)*	4-12 weeks	**Tuberculosis, TB** *Fatigue, fever, cough, chest pain, hemoptysis fibrosis, irreversible damage to lungs*	Humans, cattle, swine, other animals	Humans, diseased cattle, swine, and other mammals
***Salmonella* spp.** (non-*typhi* or *paratyphi*)	100-1000 *(by ingestion)*	0.25-3 days	**Salmonellosis, Acute Gatroenteritis** *Abdominal pain, diarrhea, nausea, vomiting, dehydration, septicemia, reactive arthritis*	Humans, cattle, swine, poultry, horses, rodents, household pets	Humans, cattle, swine, poultry, horses, rodents, domestic pets
Yersinia enterocolitica	10^6	3-7 days	**Yersiniosis, enterocolitis, pseudotuberculosis** *Diarrhea, acute mesenteric lymphadenitis mimicking appendicitis, fever, headache, anorexia, vomiting, pharyngitis, reactive arthritis*	Humans, swine, household pets	Primarily swine

Table 1. Selected pathogenic zoonoses that may be of concern for water quality near CAFOs (*Continued*)

Infectious Agent	Infectious dose	Incubation Period	Disease *Symptoms*	Host Range	Reservoir
Protozoans					
Balantidium coli	Unknown, may be as low as 10-100	4-5	**Balantidiasis, Balantidiosis, Balantidial dysentery** *Diarrhea, dysentery, abdominal colic, tenesmus, nausea, vomiting, bloody and mucoid stools*	Humans, swine	Primarily swine, also rodents
Cryptosporidium parvum	132	1-12	**Cryptosporidiosis** *Diarrhea, cramping, abdominal pain, weight loss, nausea, vomiting, fever* *Prolonged symptoms and in some instances death in immunocompromised host*	Humans, small and large mammals, poultry, fish, reptiles	Humans, cattle, and other domestic animals
Giardia lamblia	1-10 *(by ingestion)*	3-25	**Giardiasis, Lambliasis, "Beaver Fever"** *Diarrhea, abdominal cramps, bloating, fatigue, weight loss, severe hypothyroidism, lactose intolerance, chronic joint pain*	Humans, wild and domestic animals, household pets	Humans, wild and domestic animals
Toxoplasma gondii	Unknown	10-23 *(by ingestion)*	**Toxoplasmosis** *Mild cases – diarrhea, localized lymphadenopathy, fever, sore throat, and rash* *Severe cases – stillbirths, abortion, newborn syndrome, hearing and visual loss, mental retardation, dementia and/or seizures*	Humans, felines, most warm blooded animals and birds	Cats, cattle, swine, chicken, sheep, goats, rodents, and birds

Table 1. Selected pathogenic zoonoses that may be of concern for water quality near CAFOs (*Continued*)

Infectious Agent	Infectious dose	Incubation Period	Disease *Symptoms*	Host Range	Reservoir
Helminthes					
***Schistosoma* spp.**	Unknown	14-42	**Schistosomiasis, Bilharziasis, Snail Fever, Swimmer's Itch** *S. mansoni and S. japonicum –diarrhea, abdominal pain, and hepatosplenomegaly* *S. haematobium – urinary manifestation including dysuria and hematuria* *Chronic infections may lead to liver fibrosis, portal hypertension, or colorectal malignancy*	Humans, cattle, swine, water buffalo, horses, rodents, and household pets	Humans, cattle, swine, water buffalo, horses, rodents, household pets
Trichinella spiralis	Unknown	1-2 days for gastrointestinal symptoms 2-4 weeks for systemic symptoms	**Trichinelosis, Trichinosis, Trichiniasis** *Malaise, nausea, diarrhea, abdominal cramping, muscular soreness, edema of upper eyelids, eosinophila, ocular pain, photophobi, pneumonitis, remittent fever, cardiac and neurologic complications or death*	Humans, swine, household pets, rodents, wild mammals, and marine mammals	Swine, household pets, rodents, wild animals

Table 1. Selected pathogenic zoonoses that may be of concern for water quality near CAFOs (*Continued*)

Infectious Agent	Infectious dose	Incubation Period *days*	Disease *Symptoms*	Host Range	Reservoir
Viruses					
Hepatitis E Virus	Unknown	14-63	**HEV** *Jaundice, anorexia, hepatomegaly, abdominal pain, nausea, vomiting, fever, Liver Failure; most severe hepatitis during pregnancy of all hepatitis viruses*	Humans, swine, rodents, chicken	Unknown – possibly in swine
Influenza A virus	2-790	1-4	**Flu** *Acute fever, chills, headache, myalgia, weakness, runny nose, sore throat, cough*	Humans, swine, horses, domestic and wild avian species	Humans, animal reservoirs (particularly swine) are suspected as sources of new human subtypes
Lymphocytic choriomeningitis virus	Unknown	8-21	**LCM, Lymphocytic meningitis** *Mild influenza-like illness or maningeal or meningoencephalomyelitic symptoms, Guillain-Barré type syndrome, orchitis or parotitis.* *In more severe cases, temporary or permanent neurological damage, abortion, congenital hydrocephalus, and mental retardation*	Humans, swine, household pets, rodents	Rodents, swine, household pets
SARS Coronavirus	Unknown	6.4 (mean)	**SARS** *High fever, dry cough, dyspnoea, myalgia, diarrhea, vomiting, death (13.2% for infected individuals under 60, 43.3% for those over 60)*	Humans, swine chickens, ferrets, cats, macaques	Unknown – but animal reservoir is suspected
West Nile Virus	Unknown	3-14	**West Nile Encephalitis, Viral Encephalitis** *Sudden onset of flu-like illness, malaise, anorexia, nausea, vomiting, rash, and lymphadenopathy.* *More severe infections can result in aseptic meningitis or encephalitis, mental status changes, seizures, coma, severe neurologic disease, and death (4-11%)*	Mammal, reptilian, and avian hosts. Mammals generally considered dead-end hosts	Birds are the amplifying host

3. Antimicrobial Resistance

Antimicrobial agents include all types of natural or synthetic substances capable of killing or inhibiting the growth of microorganisms. Antimicrobials include antibiotics, antivirals, antifungals, probiotics, disinfectants, sanitizers, food preservatives, antimicrobial pesticides/biocides, and wood preservatives among others (Health Canada, 2002). The proper use of antimicrobial agents is an integral component of good animal agriculture practices. However, their use may be exacerbated in large confinement facilities where animals are raised in close quarters and infection in one animal can rapidly spread through hundreds or even thousands of animals. Many times, infection of one animal leads to the treatment of many animals within the facility prophylactically (Shea, 2004). Additionally, antimicrobial agents have long been administered in sub-therapeutic (non-lethal) doses to livestock animals in their feed, with the ultimate goal of increased animal growth rates. Table 2 lists the antimicrobials used therapeutically and non-therapeutically (prophylaxis and growth promotion) in livestock animals in the United States.

The use of antimicrobial compounds in animal feed has increased more than 10-fold since the 1950s, as total U.S. production of antimicrobials increased from approximately 1 million pounds in 1950 to as much as 44 million pounds in 1986 (Levy, 1992; U.S. Congress, OTA, 1995; McEwen and Fedorka-Cray, 2002). The rise in agricultural use of antimicrobial agents is certainly related to changes in their production and availability, improvements in animal health practices, increasing need for therapeutic use as animals are confined into smaller and more densely packed housing units, a perceived need for prophylactic use due to close confinement and increased risk of the spread of disease, and realization of the financial benefits of shortening the time to reach market weight. According to Dewet *et al.*, (1997) farmers with large operations are much more likely than those with small farms to use antibiotics in feed supplements for growth promotion and prophylaxis. Of the large confinement operations, those working with veterinary consultants were twice as likely to use such feed additives. In a recent survey of antimicrobial treatment practices, approximately 83% of feedlots administered at least one antimicrobial to cattle in feed or water for prophylaxis or growth promotion (Animal and Plant Health Inspection Service, 1999). Precise figures on the use of antibiotics in animal agriculture are not available, but Table 3 shows some recent estimates by various sources. Although estimates shown in Table 3 vary, three facts remain: the use of antimicrobials in animal agriculture has increased substantially since the 1950s, copious amounts of antibiotics are used every year in livestock animals, and most of the antimicrobials used are for growth promotion and prophylaxis, not for the treatment of sickened animals.

3.1 Mechanisms of bacterial resistance

Each class of antimicrobial compound operates at a specific site within the bacterial cell. Bacitracin, cephalosporins, penicillins, ionophores, and polymyxins attack cell walls and membranes. Aminoglycosides, chloramphenicols, and tetracyclines act on cellular components responsible for protein synthesis. Rifamycins, nalidixic acid, and quinolones act upon nucleic acids, and methotrezate and sulfonamides interrupt important biochemical pathways within the cell (Khachatourians, 1998). To combat the action of antimicrobial compounds, bacterial cells have adapted three primary mechanisms including reducing the accumulation of antimicrobial

Table 2. Selected Antimicrobial Agents Approved for Use in Animal Agriculture*

Antimicrobial Class and Drug	Use in Animal Agriculture			Analogs Used for Human Therapy ‡
	Animal Species	Therapeutic	Non-therapeutic†	
<u>Aminoglycosides</u>				
Gentamicin	Cattle (Beef and Dairy), Horses, Swine, Poultry[§]		X	Amikacin, Gentamicin,
Neomycin	Cattle (Beef and Dairy), Sheep, Swine, Poultry	X	X	Neomycin, Streptomycin
Spectinomycin	Beef Cattle, Swine, Poultry	X	X	
Streptomycin	Cattle (Beef and Dairy), Swine, Poultry X	X	X	
<u>Aminopenicillins</u>				
Ampicillin	Cattle (Beef and Dairy), Horses, Swine	X		Amoxicillin, Ampicillin,
Amoxicillin	Cattle (Beef and Dairy), Swine	X		Amoxicillin-clavulanic acid, Pivampicillin
<u>Cephalosporins (*3rd generation*)</u>				
Ceftiofur	Cattle (Beef and Dairy), Horses, Swine, Poultry, Sheep	X	X	Ceftriaxone, Cefixime, Cefotaxime, Ceftazidime, Ceftizoxime
<u>Fluoroquinilones</u>				
Enrofloxacin	Beef Cattle, Poultry	X		Ciprofloxacin, Difloxacin, Gatifloxacin, Levofloxacin, Moxifloxacin, Norfloxacin, Ofloxacin, Trovafloacin-Nalidixic acid
<u>Lincosamides</u>				
Lincomycin hydrochloride	Swine, Poultry	X	X	Clindamycin, Lincomycin hydrochloride
<u>Macrolides</u>				
Erythromycin	Cattle (Beef and Dairy), Swine, Poultry, Layers	X	X	Erythromycin, Azithromycin
Tylosin	Cattle (Beef and Dairy), Swine, Poultry	X	X	
Tilmicosin	Cattle (Beef and Dairy), Sheep, Swine	X	X	

* US Congress OTA, 1995; Khachatourians, 1998; US GAO, 1999; NRC, 1999; Mellon *et al.*, 2001; Shea, 2003; Sayah *et al.*, 2005; USFDA, 2005

† Non-therapeutic uses include prophylaxis and/or growth promotion.

‡ Antimicrobials used in human medicine that are similar to or the same as antimicrobials used in animal agriculture.

§ Poultry = Broilers and/or turkeys; Fowl = Quail, pheasant, duck, and/or geese; Sheep = sheep and/or goats.

Table 2 (cont.) Selected Antimicrobial Agents Approved for Animal Agriculture*

Antimicrobial Class and Drug	Use in Animal Agriculture: Animal Species	Therapeutic	Non-therapeutic[†]	Used for Human Therapy [‡]
Penicillins				
Cloxacillin sodium	Dairy Cattle	X		Ampicillin sublactam,
Penicillin G procaine	Cattle (Beef and Dairy), Horses, Sheep, Swine, Poultry, Fowl	X	X	Cloxacillin sodium, Penicillin G benzathine, Penicillin G
Penicillin G benzathine	Beef Cattle, Horses	X		potassium, Piperacillin Ticarcillin
Peptides				
Bacitracin	Cattle (Beef and Dairy), Sheep, Swine, Poultry, Layers	X	X	Bacitracin
Sulfonamides				
Sulfadiazine	Horse	X		sulfamethoxazole
Sulfadimethoxine	Cattle (Beef and Dairy), Horse, Poultry, Fowl, Fish	X	X	
Sulfamethazine	Cattle (Beef and Dairy), Swine, Poultry	X	X	
Sulfanitran	Poultry		X	
Sulfaquinoxaline	Cattle (Beef and Dairy), Poultry	X	X	
Sulfathiozole	Swine		X	
Streptogramins				
Virginiamycin	Beef Cattle, Swine, Poultry		X	Quinipristin, Dalfopristin
Tetracyclines				
Chlortetracycline	Cattle (Beef and Dairy), Sheep, Swine, Poultry	X	X	Tetracycline hydrochloride,
Oxytetracycline	Cattle (Beef and Dairy), Sheep, Swine, Poultry, Fish, Honey bees	X	X	Doxycycline
Tetracycline hydrochloride	Cattle (Beef and Dairy), Horses, Sheep, Swine, Poultry	X	X	

* US Congress OTA, 1995; Khachatourians, 1998; US GAO, 1999; NRC, 1999; Mellon *et al.*, 2001; Shea, 2003; Sayah *et al.*, 2005; USFDA, 2005

† Non-therapeutic uses include prophylaxis and/or growth promotion.

‡ Antimicrobials used in human medicine that are similar to or the same as antimicrobials used in animal agriculture.

§ Poultry = Broilers and/or turkeys; Fowl = Quail, pheasant, duck, and/or geese; Sheep = sheep and/or goats.

Table 3. Estimates of the use of antimicrobial agents in livestock animal production

Total Mass Used	Specific Use	Source
20 million pounds used annually	20% for treating disease 80% for growth promotion and prophylaxis	Swartz, 1989
18 million pounds used in 1985	12.2% for treating disease 63.2% for prophylaxis 24.6% for growth promotion	U.S. Congress, OTA, 1995
17.8 million pounds used in 1998	83% for prophylaxis and treating disease 17% for growth promotion	Animal Health Institute, 2000
29.5 million pounds used annually	7% for treating disease 93% for growth promotion and prophylaxis	Mellon *et al.*, 2001
14.4 million pounds used in 1997	Not Reported	Silbergeld, 2004

agents within the cell, attacking and inactivating the antimicrobial compounds enzymatically, or altering, protecting, or replacing target cellular structures. Bacteria may gain these resistance mechanisms in three ways: (1) acquire resistance genes from the DNA of antibiotic producers and modify them such that they are optimized for resistance to the antimicrobial agent (2) mutate genes whose products play a role in physiological cell metabolism such that they attack or inactivate the antimicrobial agent, and/or (3) mutate genes whose products are the target structures of the antimicrobial compounds such that the target structures become resistant to the inhibitory effects of the respective antimicrobials (Schwartz and Chaslus-Dancla, 2001).

The initial development of antimicrobial resistance may be relatively slow as single point mutations that give rise to resistance genes are rare events (10^{-9} to 10^{-8} per cell per generation) (Kelly et al, 1986; Freifelder, 1987; Smith et al, 1999). Once acquired, antimicrobial resistance traits can be rapidly transferred vertically through division of the host cell, and/or horizontally between different bacteria (both commensal and pathogenic) via transduction (a bacteriophage-mediated process), conjugation/mobilization (requiring contact between donor and recipient cell), or transformation (transfer of free DNA into competent recipient cells). In the mixed bacterial populations of animal and human skin and mucosa, conjugation and mobilization are considered to be of primary importance for the spread of resistance genes (Schwartz and Chaslus-Dancla, 2001) and may occur on the order of 10^{-5} to 10^{-4} per cell per generation (Summers, 2002). Transduction only occurs between bacteria of very similar species and genera as it is limited by host-specificity of bacteriophages, and therefore plays a lesser role in the spread of resistance traits in these milieus. Spread of resistance traits via transformation is considered to be very limited (Bennett, 1995).

The primary genetic elements involved in horizontal gene transfer include plasmids, transposons, and integrons/gene cassettes. Aside from the antimicrobial-resistance traits, plasmids and transposons may also carry genes (such as the *tra* gene complex) which allow them to move from one bacterial cell to another via conjugation or mobilization. Plasmids may serve as vectors for transposons and integrons/gene cassettes facilitating their horizontal transfer to competent cells. Transposons and integrons/gene cassettes can be transferred via transduction when resistance genes are co-located with prophage genes that are not excised precisely from chromosomal DNA prior to packing into phage heads. Small plasmids may also be transferred via transduction if they are packed into bacteriophage heads instead of phage DNA during phage assembly (pseudophages), however this process is limited compared to conjugation and mobilization (Schwartz and Chaslus-Dancla, 2001). Once established, resistance genes may persist in commensal bacteria serving as a reservoir for rapid acquisition of antimicrobial resistance for any new pathogen that may inhabit the intestinal tract (Barza, 2002). Of particular interest are enterococci and *E. coli* that can play a major role in the transmission of mobile resistance genes (Salyers, 1995).

Antimicrobial-resistance in bacteria may be conferred by tandem arrays of genetically linked resistance genes borne by integrons or other transposons that can reside in the chromosome and on conjugative or mobilizable plasmids (O'Brien et al, 1985; Zhao *et al.*, 2001; Roe *et al.*, 2003). Adaptation of a bacterial cell to any given antimicrobial via gene transfer can thus result in selection for resistance to not only that specific agent, but also, by genetic linkage of resistance genes, to other antimicrobials (Summers, 2002). Antimicrobial resistance determinants are also often co-located with virulence determinants on mobile genetic elements. Treatment with antimicrobials for which resistance is conferred may result in the enrichment of more virulent bacterial strains in the selective environment. Epidemiological evidence from reported *Salmonella* and *Campylobacter* infections suggest that resistant strains are somewhat more virulent than susceptible strains, exhibiting prolonged or more severe illness (Travers and Barza, 2002). In a study of 67 individuals not treated with antimicrobials, diarrhea lasted longer when the isolates were ciprofloxacin-resistant (12 days) than when they were ciprofloxacin susceptible (6 days) ($P=0.02$) (Marano *et al.*, 2000). The likelihood of hospitalization and average length of hospital stay are significantly higher in those infected with antimicrobial-resistant organisms than those with susceptible stains (Lee *et al.*, 1994).

Resistance to one antimicrobial compound may also confer resistance to other antimicrobial compounds through similarity of the antimicrobial agents (Khachatourians, 1998). Cases of multi-drug resistance in bacterial zoonoses caused by structural similarity of human-use antimicrobials to those used in animal agriculture have been documented. Virginiamycin-resistant bacterial isolates from turkeys were found to be resistant to the structurally similar and clinically important human-use drugs quinipristin and dalfopristin (Feinman, 1998; Chadwick and Goode, 1997). Tylosin-resistant streptococci and staphylococci-resistant animal isolates were determined to be resistant to the structurally similar and clinically important human-use drug erythromycin, and were found not only in the livestock animals, but in their caretakers as well (Feinman, 1998; Chadwick and Goode, 1997). Virginiamycin and Tylosin are both used prophylactically and/or for growth promotion in beef and dairy cattle, swine, broilers, and turkeys. Table 2 lists human-use drugs that are structurally similar to several antimicrobial compounds used in animal agriculture.

3.2 Antimicrobial resistance in livestock animals

The occurrence of antimicrobial-resistant bacteria tends to be rapid following introduction of antimicrobial agents into clinical or agricultural use. For instance, occurrence of tetracycline resistant bacteria was reported in 1956, four years following its introduction to clinical use and only eight years following its initial discovery. The time lag between introduction to clinical use and occurrence of antimicrobial resistant bacteria was 15 years for vancomycin, 4 years for nalidixic acid, 3 years for gentamicin, 3 years for fluoroquinolones, one year for erythromycin, and less than one year for streptomycin (Schwartz and Chaslus-Dancla, 2001). Although the latent period between the introduction of an antimicrobial and the emergence of resistance may vary, once the prevalence of resistance in a population reaches a certain level, reversal of the problem may be extremely difficult (Swartz, 2002). For example, fluoroquinolone-resistant Campylobacter were detected in 43-96% of market chickens from two producers more than one year after fluoroquinolones were no longer used in their poultry production (Price *et al.*, 2005).

Repeated exposure of bacteria to antimicrobial agents and access of bacteria to increasingly large pools of antimicrobial resistance genes in mixed bacterial populations are the primary driving forces for emerging antimicrobial resistance (Schwartz and Chaslus-Dancla, 2001). Resistance of both commensal and pathogenic bacteria in livestock animals to antimicrobials of clinical importance is now commonplace and is related to their increased use for growth promotion and prophylaxis over the last 50 years (Shere et al, 1998; Maynard *et al.*, 2003). Hayes et al (2004) surveyed 541 *Enterococcus faecium* isolates from 82 farms within a poultry production region in the eastern United States. Sixty-three percent were resistant to quinipristin-dalfopristin and 52.7% were resistant to four or more antimicrobials. In a study of several swine farms in the United States, Jackson *et al.*, (2004) determined that Tylosin use for growth promotion resulted in erythromycin-resistance in 59% of enterococci isolates, compared to 28% at a farm where Tylosin was used for treatment of disease only, and 2% at a farm that did not use Tylosin. National surveillance of Salmonella in swine in the U.S. has revealed resistance to several important antimicrobials including tetracycline (50%), ampicillin (12%), sulfamethoxazole (23%), and streptomycin (23%) (NARMS, 1998). Hoyle *et al.*, (2004) studied ampicillin-resistant *E. coli* in calves in the United Kingdom and determined that ampicillin resistance peaked over 80% within 4 months, steadily declining to less than 10% as the calves aged to 8 months. Schroeder *et al.*, (2002) tested 752 *E. coli* isolates from humans and animals for resistance to several antimicrobials of clinical importance. Approximately half of the isolates displayed resistance to one or more antimicrobials including penicillins, sulfonamides, cephalosporins, tetracyclines, and Aminoglycosides, with the highest frequencies of antimicrobial resistance in humans and turkeys and the lowest in non-food animals. Sayah *et al.*, (2005) studied antimicrobial resistance patterns in livestock, companion animals, human septage, wildlife, farm environments (manure storage facilities, lagoons, and livestock holding areas) and surface water in the Red Cedar Watershed in Michigan. *E. coli* isolates from livestock showed resistance to the largest number of antimicrobials and multidrug resistance was most common in swine fecal samples. Resistance was demonstrated most frequently to tetracycline, cephalothin, sulfisoxazole, and streptomycin. Similarities in patterns of resistance in *E. coli* were observed in livestock animals and environmental samples taken from their respective farms. These authors suggest that farm environment samples may best describe potential contamination of nearby waters with antimicrobial-resistant bacteria.

3.3 Risk to public health

Much concern surrounds the elevated use of antimicrobial agents in confinement facilities and, in particular, the use of antimicrobial agents at non-therapeutic doses in animal agriculture (American Academy of Microbiology, 1999, Mellon *et al.*, 2001). The use of antimicrobial agents inevitably selects for resistance of both commensal and pathogenic microorganisms exposed to the agents (Linton et al, 1975; Dawson *et al.*, 1984; Levy et al, 1976; Dunlop et al, 1998; Endtz *et al.*, 1991; Jacob-Rietsma *et al.*, 1994; Bager *et al.*, 1997; Low *et al.*, 1997; Tauxe, 1997; Gynn *et al.*, 1998; McEwen and Fadorka-Cray, 2002; Vasil' *et al.*, 2002). The conditions of widespread, prolonged exposure to antimicrobial compounds at sublethal doses with little dose control in CAFOs may exacerbate their development. Once established, the movement of antimicrobial-resistant microorganisms from animal to animal or animal to animal care worker may be facilitated by the crowding of animals into confinements, often with suboptimal hygiene. The co-colonization of animal gastrointestinal tracts by antimicrobial-resistant commensal bacteria and bacterial pathogens may lead to further development of antimicrobial-resistant bacterial zoonoses (Kruse *et al.*, 1999). As much as 75-80% of an antibiotic may pass undigested through an animal, thus its waste may not only harbor high concentrations of antimicrobial-resistant bacteria, but also their resistance genes and raw (undigested) antimicrobial compounds (Campagnolo and Rubin, 1998). This waste is often stored in open air lagoons and/or spread on fields where these compounds, resistant organisms, and antimicrobial-resistance gene reservoirs may move into the environment via aerosolization, infiltration into the groundwater, or runoff into surface water resources.

Antimicrobial resistance in zoonotic pathogens is a serious threat to human health (Ghidán *et al.*, 2000; Cheng *et al.*, 2002; Travers and Barza, 2002). Many of the drugs are used in animal agriculture and human medicine are the same or very similar including, but not limited to, beta-lactams (penicillin, ampicillin, cloxacillin), tetracyclines, sulfonamides and potentiated sulfonamides, cephalosporins, and fluoroquinolones (McEwen and Fadorka-Cray, 2002). Exposure to zoonotic pathogens harboring resistance to antimicrobials of clinical importance may lead to diseases with few or no treatment options in humans. In cases where pathogens are resistant to administered antimicrobial compounds, vulnerability to infection can increase up to three-fold, primarily resulting from a transient decrease in an individual's resistance to colonization by the pathogen (Barza and Travers, 2002). Antimicrobial-resistant pathogens tend to be more virulent than their susceptible counterparts, causing more prolonged or severe illnesses (Marano *et al.*, 2000; Travers and Barza, 2002; Swartz *et al.*, 2002). There is circumstantial evidence that increased prevalence of antimicrobial resistance in human isolates may be linked to the use of antimicrobial agents in animal agriculture (Levy *et al.*, 1976; Jensen *et al.*, 1998; Swartz *et al.*, 2002; Silbergeld, 2004). Many cases of severe human disease caused by acquisition of antimicrobial-resistant zoonotic pathogens from animal agriculture have been documented (Levy *et al.*, 1978; Schlech *et al.*, 1983; Holmberg *et al.*, 1984; Morgan *et al.*, 1988; Besser *et al.*, 1993; Cieslak et al, 1993; Isaacson *et al.*, 1993; Lee *et al.*, 1994; MacKenzie *et al.*, 1994; Millard et al, 1994; Tschape *et al.*, 1995; Centers for Disease Control and Prevention, 1998; Jackson *et al.*, 1998; Crampin *et al.*, 1999; Huovinen, 1999; Wegner *et al.*, 1999; Franklin, 1999; Kruse, 1999; Health Canada, 2000; License *et al.*, 2001; Clark *et al.*, 2003).

4. Survival of Pathogens in the Environment

Pathogens at concentrated animal feeding operations may be present in animal wastes, water used for maintenance of livestock and animal housing units, soils where animal manures and wastewaters are spread, on crops grown in soils where manures were applied or where contaminated irrigation waters are used, and groundwater and surface waters contaminated by manure runoff. The survival of pathogenic organisms in the environment varies widely depending on the pathogen, environmental conditions, and the chemical, physical, and biological composition of milieu of interest. Enteric bacterial, viral, and protozoan pathogen inactivation in soil, water, crops, or manure may be affected by predation, competition, water stress/osmotic potential, temperature, UV radiation, pH, inorganic ammonia, and organic nutrients (Geldreich *et al.*, 1968; Davenport *et al.*, 1976; Crane and Moore,1986; Hurst *et al.*, 1989; Davies and Evison, 1991; Olson *et al.*, 1999; Sattar *et al.*, 1999; Burkhardt *et al.*, 2000; Davies-Colley *et al.*, 2000; Wait and Sobsey, 2001; Jamieson *et al.*, 2002; Ferguson *et al.*, 2003). The importance of each factor is strongly related to the milieu of interest. In general, the survival of pathogens is inversely related to predation, competition, temperature, UV radiation, water stress, and inorganic ammonia, except for *Cryptosporidium* oocysts and *Giardia* cysts, which have low survival at sub-zero (<-20ºC) temperatures (Van Donsel *et al.*, 1967; Zibilske and Weaver, 1978; Reddy *et al.*, 1981; Jamieson *et al.*, 2002; Ferguson *et al.*, 2003). The relationship of pathogen survival to pH and organic nutrients may be more complex. Under the right conditions, pathogens are capable of surviving in the environment for days to more than a year.

4.1 Manure and manure slurries

Table 4 summarizes the survival of bacterial and parasitic pathogens noted in literature in manures and manure slurries. These nutrient rich environments may offer protection from environmental insults such as solar UV radiation, desiccation, and temperature fluctuations, promoting survival or even regrowth of pathogenic zoonoses. For instance, Muirhead *et al.*, (2005) determined that within cowpats, *E. coli* grew for 6 to 14 days instead of following a traditional logarithmic die off curve and Olson (2003) noted that the eggs of *Ascaris suum*, a common parasite in swine, are highly resistant to inactivation in feces, potentially remaining infectious for years. However, these environments may also me hostile, as they may harbor predators and competitors, or produce toxic components that may reduce pathogen viability. For instance, inorganic ammonia, naturally produced by hydrolysis of urea and in decomposing manure, can be biocidal at high concentrations, and has been exhibited to be directly proportional to *Cryptosporidium* oocyst inactivation (Jenkins *et al.*, 1998; Jenkins et al, 1999). As seen in Table 4, animal manures and manure slurries may remain significant reservoirs for environmental contamination by zoonotic pathogens for many months.

Bacterial pathogens may persist for long periods in animal manures under typical environmental conditions. This may be exacerbated when the temperatures are low, moisture remains optimal, and aeration is not used. For instance, *Salmonella* and *E. coli* O157:H7 have been noted to survive for 4-6 months in animal manures and manure slurries kept at 1-9ºC, up to 49 times longer than at 40-60ºC. Nicholson *et al.*, (2002) studied the survival of *E. coli* O157:H7,

Table 4. Survival of pathogenic zoonoses in livestock manures and manure slurries

Environment	Temperature (ºC)	Survival[†] (days)						
		Bacterial Pathogens[‡]					Parasites[§]	
		Salmonella sp.	*Campylobacter* sp.	*Yersinia enterocolitica*	*E. coli* O157:H7	*Listeria* sp.	*Giardia*	*Cryptosporidium*
Manure								
Broiler Litter	40-60	4	4		4	8		
Cattle, beef or dairy	-20 to -4	>180	56	>365	>100		<1	>365
	1-9	196	21	100	130[*]		7	56
	10-19				45			
	20-29	65[*]	3		90		7	28
	30-39	48	7	30	49		7	28
	40-60	4	4		8	4		
	On farm (<23)				47			
Swine	40-60	16	2		32	4		
Sheep	1-10				>100			
	10-19				>100			
	20-29				40			
	On farm (<23)				630			
Manure slurries								
Cattle, beef or dairy	-20 to -4				21			
	1-9	115[*]			150[*]			
	10-19				40			
	20-29	89[*]	3		103[*]			
	30-39	19			22[*]			
	40-60				<2			
	On farm (5-20)	93	32		93	185		
Swine	1-9	14						
	20-29	8	2					
	30-39	<8						

[†] Longest survival time reported

[‡] Bolton *et al.*, (1999); Kudva *et al.*, (1998); Wang *et al.*, (1996); Himathongkham *et al.*, (1999); Mitscherlich and Marth (1984); Guan and Holley (2003); Olson (2003); Tauxe (1997); Plym-Forshell (1993); Nicholson *et al.*, (2002)

[§] Cole *et al.*, (1999); Robertson *et al.*, (1992); Fayer *et al.*, (1998); Olson (2003); Olson *et al.*, (1999)

[*] Calculated as 7 times the reported decimal reduction time (time required for 1-log reduction in pathogen concentration) assuming logarithmic die-off and based on a reported initial inocula of 10^6-10^8 organisms per gram manure or milliliter slurry.

Salmonella, *Listeria,* and *Campylobacter* in dairy cattle, swine, and poultry manures stored at 40-60ºC and determined that aeration of the solid manures decreased survival times for *E. coli* O157:H7 and *Salmonella* by as much as 88%. These researchers noted a decrease in the survival of *E. coli* O157:H7 and *Salmonella* sp. when a higher dry matter content was maintained in the slurry. Kudva *et al.*, (1998) noted similar changes in *E. coli* O157:H7 in sheep manure, which survived for 630 days at temperatures below 23ºC when not aerated versus 120 days when aerated, the difference likely due to drying of the aerated manure.

Parasitic protozoan survival in animal manures may also be related to temperature, but the trends are not as strong as those reported for bacterial pathogens. This is likely due to their ability to form cysts and oocysts for protection from environmental pressures under the range of temperatures reported in Table 4. These parasites have been shown to be susceptible to temperature extremes, with reported survival of *Cryptosporidium* oocysts ranging from 1 hour at -70ºC, 1 day at -20ºC, one or more years at 4ºC, 3-4 months at 25ºC, 1-2 weeks at 35ºC, and just minutes at 64ºC (Fayer and Nerad, 1996; Finstein, 2004). *Cryptosporidium* oocysts in manures may also be susceptible to desiccation and bacterial degradation whereby warmer temperatures may accelerate the degradation process. A similar pattern exists for *Giardia* cysts, but they are inactivated more rapidly than *Cryptosporidium* oocysts and are less resistant to temperature extremes.

Information regarding the survival of zoonotic viruses in animal wastes is sparse. Although not shown in Table 4, zoonotic viruses in animal manures and manure slurries may exhibit long inactivation times that extend for weeks to months. Karetnyi *et al.*, (1999) determined that swine hepatitis E was detectable in positive stool samples for more than 2 weeks, regardless of whether the samples were maintained at -85ºC, 4ºC, or room temperature. Pesaro et al (1995) studied the survival of several viruses including picnoraviruses, rotaviruses, parvoviruses, adenoviruses, and herpes viruses as well as the coliphage f2 in nonaerated liquid and semisolid animal wastes. Ninety percent reduction in virus titer ranged from less than 1 week for herpes virus to more than 6 months for rotavirus, suggesting that a 4-$\log_{10}$ reduction in viruses may require storage for as much as two years for some pathogens. Although little information exists regarding the survival of viral pathogens in fecal environments, these studies show that under non-aerated conditions viruses may exhibit prolonged persistence in manure and manure slurries, suggesting a strong potential for viral pathogen contamination when manure is spread on land.

In general, pathogen survival in animal manures is dictated by the effects of aeration and temperature, whereby increased aeration and higher temperatures lead to more rapid die-off. Of the pathogens listed in Table 4, *E. coli* O157:H7, *Listeria* sp., and *Salmonella* sp. were the most persistent in manure and manure slurries regardless of the temperature. However, considering the work of Pesaro *et al.*, (1995), viral pathogens may persist much longer than the bacterial pathogens, and should be given more consideration in future studies.

Much of the work to date has concentrated on the survival of pathogens in cattle manures and manure slurries. However, based on the summary presented in Table 4, there seems to be dissimilarities in the survival of pathogens in different animal feces. This may be due to differences in the physical, chemical, or biological properties of the various animal manures, but could also be a result of the low numbers of studies on swine and poultry manures versus those of cattle. The lack of studies on pathogen survival in swine and poultry manures impedes the development of safe management practices.

The survival of pathogens in animal manures and manure slurries is typically studied under controlled laboratory conditions. Kudva *et al.*, (1998) noted that survival of pathogens in laboratory studies were generally lower than those observed in field studies. For instance, these researchers determined that *E. coli* O157:H7 survived in sheep manure for 100d under a controlled (4-10ºC) laboratory setting versus 630 days when exposed to environmental (ambient) conditions (<23ºC). Based on their observations, laboratory experiments may not provide a reasonable estimate of pathogen survival in on-farm conditions. Future efforts should concentrate on measuring the survival of pathogens *in-situ*.

4.2 Natural Waters

Manure runoff and wastewaters from concentrated animal feeding operations may contain pathogenic zoonoses and antimicrobial-resistant bacteria that can survive and proliferate in nearby natural waters. Runoff and wastewater discharges may also contribute both organic and inorganic nutrients that may encourage the growth and proliferation of indigenous or introduced pathogens (Grimes *et al.*, 1986). Table 5 summarizes the survival of bacterial and parasitic pathogens in dirty waters from livestock operations, natural waters, and drinking water as reported in literature. In these milieus, UV radiation, disinfectants, temperature, predators, and toxin producers generally challenge the survival of pathogenic zoonoses and antimicrobial-resistant bacteria (Chao *et al.*, 1988; Johnson *et al.*, 1997).

The survival of bacteria in natural waters may be longer than exhibited in manures or manure slurries. *Yersinia enterocolitica* exhibited the greatest survival among all bacterial pathogens considered in Table 5 whereas *Campylobacter* was the least. However, in a viable but not cultivable state, *Campylobacter* may survive for as much as 120 days at 4ºC. *E. coli* O157:H7 has also been noted to enter a viable but not cultivable state in water increasing the survival time over that reported in Table 5 (Wang and Doyle, 1998). Pathogens may also settle into streambed sediments, decreasing exposure to UV radiation and predators and increasing survival times over those reported in Table 5. For instance, Anderson *et al.*, (2005) determined that a 90% reduction in fecal coliforms in fresh waters required 4.2 days, whereas 50 days was required to achieve the same reduction in the underlying sediments. Although differences in the survival of *Enterococcus* spp. was observed, the trend was the same (1.4 days in water and 4.5 days in the underlying sediments), and held true for salt water environments.

Cryptosporidium oocysts may also be especially resistant in environmental waters, surviving for more than a year in optimal (low temperature) conditions. For instance, Robertson *et al.*, (1992) studied oocyst infectivity during incubation in cold river water and reported up to 66% viability at 33 days and 11% viability at 176 days. In another study, Medema *et al.*, (1997) determined that the time required for one-log reduction in *Cryptosporidium* oocyst infectivity in river water at 15ºC was 40-160d, whereas at 5ºC it was 100d. Even more extreme are viruses, which can persist for several years in the subsurface. Azadpour-Keeley *et al.*, (2003) reviewed the movement and longevity of viruses in the subsurface and suggested that soil environments may actually enhance viral survival. They report a wide variation in inactivation rates in different

Table 5. Survival of pathogenic zoonoses in soils, contaminated water-irrigated soils, and manure-amended soils

Environment	Temperature (°C)	Survival[†] (days)						
		Bacterial Pathogens[‡]					Parasites[§]	
		Salmonella sp.	*Campylobacter* sp.	*Yersinia enterocolitica*	*E. coli* O157:H7	*Listeria* sp.	*Giardia*	*Cryptosporidium*
Soil								
	-20 to -4	>84	56	>365	>300		<7	>365
	1-9	196	20	>365	100		49	56
	20-29	>45	10	10	>56		14	28
Dirty Water-Irrigated Soils [*]								
	0-22	120	120		34	128		30
Farm-yard manure-amended soil								
Cattle								
Beef	0-22	63	120		64	120		30
Dairy	0-22	120	64		34	120		30
Poultry litter								
Broilers	0-22	32	16		32	>32		
Broilers & layers	0-22	63	64		32	56		30
Sheep	0-22	120	34		63	128		30
Swine	0-22	120	34		32	120		30
Manure slurry-amended soil								
Cattle								
Beef	0-22	120	64		32	120		30
Dairy	0-22	120	63		64	120		30
Swine	0-22	299	36		32	120		63

† Longest survival time reported

‡ Mubiru *et al.*, (2000); Mitscherlich and Marth (1984); Zibilske and Weaver (1978); Guo *et al.*, (2002); Chao *et al.*, (1988); Guan and Holley (2003); Olson (2003); Ciesak *et al.*, (1993); Nicholson *et al.*, (2002); Hutchinson *et al.*, (2004); Hutchinson *et al.*, (2005)

§ Cole *et al.*, (1999); Robertson *et al.*, (1992); Fayer *et al.*, (1998); Olson (2003); Olson *et al.*, (1999)

* Dirty water from livestock operations.

soils at near-neutral pH suggesting it may take as little as 0.8 days to as many as 11 years to achieve 99.99% (4-$\log_{10}$) die off of some viruses in aquifers. Keswick *et al.*, (1982) report that survival of enteric bacteria and viruses were longer in groundwater than surface water, presumably due to lower temperatures and protection from sunlight and microbial antagonism.

Maintenance of antimicrobial-resistance in natural waters has not been studied extensively. In untreated seawater suspensions, Guardabassi and Dalsgaard (2002) noted that multiple antibiotic resistant *E. coli* and *Citrobacter freundii* survived and maintained their multiple resistance properties for more than 30 days, whereas a multi-antibiotic resistant *Acinetobacter johnsonii* survived and maintained its multiple resistance properties for 14 days. In untreated pond water suspensions, these authors noted survival times of 21 days (*E. coli* and *A. johnsonii*) and 28 days (*C. freundii*), while maintaining multiple resistance properties. This suggests that antimicrobial resistant microorganisms may survive for long periods upon discharge to aquatic environments and that stress and nutrient depletion may not affect the stability of their resistance phenotypes. The effect of low concentrations of antimicrobial compounds discharging to surface or ground waters via manure runoff, lagoon leakage, or wastewater discharge on the maintenance of antimicrobial-resistant phenotypes or genotypes has not been studied.

4.3 Manure-amended soil

Table 6 summarizes the survival of bacterial and protozoan pathogens in soils. The survival times reported in Table 6 are more similar to those of manures and manure slurries and less than those exhibited in water. In general, it has been reported that survival of pathogens in soil increases when manures are incorporated into soils rather than unincorporated. For instance, Hutchison et al (2004) studied the die off of *Salmonella*, *Listeria*, *Campylobacter*, and *E. coli* O157 following application of manure to soil and incorporation of the manure upon application, one week following application, or no incorporation. The authors noted that die-off was similar in summer and winter months, but more rapid when the manure was not incorporated into the soil. The increased survival of pathogens incorporated into soils may be related to decreased exposure to UV radiation, temperature extremes, and desiccation and increased availability of nutrients. However, soils may harbor competitor organisms and predators that can reduce pathogen survival. Survival of pathogenic bacteria in soils may also be limited by low soil pH (Jamieson *et al.*, 2002) or freeze-thaw cycling. Jenkins *et al.*, (1999) determined that *Cryptosporidium* oocyst infectivity decreased from greater than 50% to less than 1% when exposed to freeze-thaw cycles in a soil environment. Walker et al (2001) noted that inactivation of *Cryptosporidium* oocysts during freeze-thaw cycling or heating was enhanced by increased osmotic stress (decreased water potential).

The most important factor affecting the survival of enteric pathogens in soils systems may be the moisture status, which is influenced not only by precipitation, but also by moisture retaining properties such as particle size distribution and organic matter content (Gerba *et al.*, 1975; Tate *et al.*, 1978; Kibbey *et al.*, 1978; Chandler and Craven, 1980; Crane *et al.*, 1981; Reddy *et al.*, 1981; Faust, 1982; Mubiru *et al.*, 2000, Entry *et al.*, 2000b; Jamieson *et al.*, 2002). For instance, Nicholson *et al.*, (2002) studied the survival of bacterial pathogens following land spreading and determined that there are some indications that pathogen survival is longer in clay loam grassland soil than in sandy arable soil. Burton et al (1983) determined that *Salmonella newport*

Table 6. Survival of pathogenic zoonoses in drinking water, livestock rinse waters, surface fresh waters, surface salt waters, surface water sediments, soils irrigated with livestock rinse waters, and ground waters

Environment	**Temperature** (ºC)	**Survival**[†] (days)						
		Bacterial Pathogens[‡]					**Parasites**[§]	
		Salmonella sp.	*Campylobacter* sp.	*Yersinia enterocolitica*	*E. coli* O157:H7	*Listeria* sp.	*Giardia*	*Cryptosporidium*
Water								
Drinking	1-9	90	12[*]	90	90		25	
	10-19		12					
	20-29		2					
	30-39		1.5					
Ground or Spring	-20 to -4							
	1-9			448				
	10-19	152						
	20-29							
Surface	-20 to -4	>180	56	>365	>300		<7	>365
	1-9	>180	12[**]	>365	>300		77	>365
	10-19			14				
	20-29	>180	4				14	70
	30-39			10	84			
Dirty water[††]	5-20	32	16		16	93		

[†] Longest survival time reported

[‡] Wang and Doyle (1998); Bolton *et al.*, (1999); Santo Domingo *et al.*, (2000); Mitscherlich and Marth (1984); Karapinar and Gonul (1991); Chao *et al.*, (1988); Buswell *et al.*, (1998); Rollins and Colwell (1986); Blaser *et al.*, (1980); Guan and Holley (2003); Olson (2003); Fayer *et al.*, (1998); Kenneth *et al.*, (1998); Ford, 1999; Nicholson *et al.*, (2002)

[§] Cole *et al.*, (1999); Olson (2003); Robertson *et al.*, (1992); Fayer *et al.*, (1998); Olson *et al.*, (1999)

[*] In the presence of a biofilm, survival was as much as 29 days at 4ºC and 11 days at 30ºC

[**] Survival may be more than 120 days in a viable but not cultivable (VBNC) state

[††] Dirty water from livestock operations

survived longer in soils with higher clay content, potentially owing to a higher concentration of organic matter and nutrients. Mubiro *et al.*, (2000) suggested that survival of *E. coli* O157:H7 may also be enhanced in soils of higher matric potential not only due to enhanced water holding capabilities, but also because these soils better retained nutrients. The addition of manure to the soils may enhance survival of pathogens such as *Campylobacter* spp. or *E. coli* O157:H7, possibly due to increased organic and inorganic nutrient availability (Gagliardi and Karns, 2000).

The effects of water/osmotic potential on microbial stress in soil environments may be exacerbated by specific properties of the pathogen of interest. Bacteria and viruses with a hydrophobic envelope tend to accumulate at the air water interface leading to increased inactivation (Johnson and Gregory, 1993; Thompson *et al.*, 1998; Thompson and Yates, 1999). The lack of a hydrophobic envelope may reduce attraction to the air-water interface, and thus may afford some protection from viral inactivation due to osmotic stress (Ferguson *et al.*, 2003). In soils, osmotic stress typically increases near the soil surface, and may lead to reduced pathogen survival (Gerba, 1999).

Even when not incorporated into soils, the survival of pathogens following application of manures to land may be lengthy. Hutchinson *et al.*, (2005) determined decimal reduction times (the time required for 1-$\log_{10}$ reduction) for *E. coli* O157:H7, *Listeria monocytogenes*, *Salmonella* spp., and *C. jejuni* of 1.31 – 3.20 days (mean) and *Cryptosporidium parvum* oocysts of 8-31 days following the application of livestock waste onto fescue plots (no incorporation). Most zoonotic agents declined below detectible levels by 64 days, except for *L. monocytogenes*, which persisted for up to 128 days in some plots. Potential mechanisms for pathogen reduction may have included, among others, desiccation, UV radiation, and runoff from the grasslands to nearby receiving waters.

Where food crops are grown in manured soils or using contaminated irrigation waters, pathogens can contaminate produce surfaces. The level and persistence of contamination may be related to the irrigation method (spray irrigation or surface irrigation) and time of contact of produce with contaminated soils. For instance, Ingham *et al.*, (2004) identified *E. coli* contamination on carrots, lettuce, and radishes up to 120 days following application of non-composted bovine manure as a fertilizer in fields in Wisconsin. Following growth in *E. coli* O157:H7 contaminated manure-fertilized soil; Johannessen *et al.*, (2005) detected *E. coli* O157:H7 on the stems, but not on the edible parts of lettuce. Solomon *et al.*, (2002) noted that spray irrigation following a single exposure to *E. coli* O157:H7 resulted in 90% of the lettuce being contaminated with *E. coli* O157:H7 and the contamination persisted for more than 20 days in 82% of the plants. Where surface irrigation was used under the same circumstances, only 19% of the lettuce was contaminated. Immersion of harvested lettuce heads in 200ppm chlorine solution for 1 minute did not eliminate all *E. coli* O157:H7 cells from infected lettuce, regardless of irrigation method. Guo *et al.*, (2002) investigated water and soil as reservoirs of *Salmonella* for contaminating mature green tomatoes, and determined that the population of *Salmonella* on tomatoes in contact with contaminated soil increased over 4 days by 2.5 $\log_{10}$ CFU per tomato during storage at 20ºC, and remained constant for an additional 10 days. In contrast, where tomatoes were not in contact with soil, but *Salmonella* were inoculated onto the fruit surface, the number of cells declined over 14 days by 4 $\log_{10}$ CFU per tomato when held at 20ºC. At day one, *Salmonella* was associated with the skin surface. As time of storage increased, more *Salmonella* cells were associated with less accessible stem scar and subsurface areas of the tomatoes, which may render the fruits more resistant to disinfection with sanitizing agents.

4.4 Discussion

Relatively few studies are available describing the survival of pathogenic zoonoses in environmental milieu, especially considering the broad range of properties of soils, manures, and waters that may potentially be contaminated. Much of the emphasis has been placed on cattle manures, manure-amended soils, and surface waters, with less emphasis on ground waters and manures from other livestock animals such as swine and poultry. In general, pathogenic zoonoses tend to survive longer cooler rather than warmer temperatures and in water rather than in manures or soils. This may be problematic as manures and soils are stationary whereas water is a significant transport medium for pathogens. Further, very few studies have been reported on the survival of viruses which is troubling because the relatively few studies that are available suggest that viruses are more persistent than bacteria and parasitic protozoa and can travel vast distances in both surface and ground waters. A significant limitation is the lack of information regarding the survival of antimicrobial-resistant bacteria in various milieus including the persistence of phenotypic and genotypic antimicrobial-resistance traits. Most studies reported in Tables 4-6 were carried out in the laboratory instead of *in-situ*, and only a few examined more than one environmental stressor simultaneously. The combined effects of multiple stressors in the natural environment or presence of additional growth and maintenance factors may limit or enhance pathogen survival in reference to lab-scale studies of single stressors (Crane and Moore, 1986; Robertson et al, 1992; Kudva *et al.*, 1998; Jenkins *et al.*, 1999; Friere-Santos *et al.*, 2000; Walker *et al.*, 2001).

Current methods for detecting pathogens in environmental systems may limit the ability to determine accurate survival times in difficult milieu. Specific soil or manure properties or survival strategies of the various pathogens may limit their detection with cultivation techniques. For instance, upon being stressed, bacteria may die or adapt using a number of mechanisms including formation of spores, formation of ultramicrobacteria, or entering viable but not cultivable (VBNC) states. Many of the bacterial pathogens can survive for much longer periods of time than indicated in Tables 4-6 in VBNC states (Wang and Doyle, 1998; Santo Domingo *et al.*, 2000; Rollins and Colwell, 1986). Better and more sensitive methods for pathogen detection in different media need to be developed to determine more accurately pathogen survival. Accurate information regarding the survival of pathogenic zoonoses and antimicrobial resistant bacteria is necessary for modeling their fate and transport from confined animal feeding operations. Based on available information, ensuring the safety of food crops and water resources may require management practices that eliminate pathogens in manures and other CAFO wastes prior to land application or discharge to natural waters.

5. Pathogen Movement – An Ecological Perspective

Figure 2 provides a partial picture of the potential routes of transmission of zoonotic pathogens from confined livestock animals to humans and the environment. The movement of pathogens onto, within, and off farms is a complex ecological issue owing to the continuous exchange of microbes between human and animal hosts and environmental reservoirs (Sobsey *et al.*, 2002; Summers, 2002). For instance, Herriott *et al.*, (1996) tested twelve herds and their feeds and water troughs as well as co-located (non-bovine) livestock, companion animals, wild birds, rodents and flies at dairies and feedlots in Idaho, Oregon, and Washington for the presence of *E. coli* O157:H7. *E. coli* O157:H7-positive cattle were identified in all 12 herds with a prevalence of 1.1-4.4% in dairies and 1.5-6.1% in feedlots. It was also detected in 1.3% of trough water samples, 2.0% of trough water biofilm samples, in a nearby horse, two dogs, pooled bird droppings, and composite fly samples. Considering antimicrobial resistance, the issue becomes more complicated as mobile genetic elements conferring resistance provide a distinct selective advantage in stressed environments such as the colonic tract of humans and animals being treated with antimicrobials. In these environments, proliferation of resistance traits among bacteria can be rapid and have lasting effects (O'Brien, 2002; Summers, 2002). Addressing the movement of pathogens between intensive livestock operations and the environment will require understanding of the ecological principle that everything is connected to everything else. The following is a discussion of some of the potential pathways for movement of zoonotic pathogens from livestock animals raised in confinement to humans and the environment.

5.1 CAFOs and Abattoirs

The presence of zoonotic pathogens in CAFO environments may begin with the stocking of infected animals or with the use of selected feed products on the farm. Animal feeds and drinking water containing antimicrobial compounds may lead to the development and persistence of resistant bacterial zoonoses in livestock animals which may proliferate through the farm environment. Animal feeds can also be a direct source of zoonotic pathogens and antimicrobial-resistant bacteria for livestock animals (Curtain, 1984; Durand *et al.*, 1990; Izat and Waldroup, 1990; Gabis, 1991; Veldman *et al.*, 1995; Davies and Wray, 1997; Primm, 1998; Shirota *et al.*, 2001a,b). For instance, of ten feed ingredient piles at 12 commodity dairy feeding farms, Kidd *et al.*, (1999) identified two feeds contaminated with *Salmonella enteritidis*. Sixty two percent of Enterobacteriaceae isolates from the ten piles were ampicillin-resistant and 10% were tetracycline-resistant. Although feed can be contaminated on-farm, it may also arrive contaminated, as shown in a recent survey of 629 feed samples from 3 feed mills where 8.8% of feed mash samples and 4.2% of pelleted feed samples were positive for Salmonella (Jones and Richardson, 2004). Antimicrobial-resistant bacteria and other pathogens can also be present in trough waters (Marshall *et al.*, 1990; Herriott *et al.*, 2002; Kemp *et al.*, 2005) potentially resulting from either stocking troughs with contaminated water or through deposition of contaminated material into the water from an animal harboring the disease (via the saliva, mucosa, or feces). Antimicrobial compounds in the water and the presence of biofilms in which bacteria are in close contact may lead to proliferation of antimicrobial resistance within these microbial communities. The confinement of animals into dense units where trough waters are shared and where animals have increased contact with each other and their fecal matter may exacerbate the spread of pathogens from animal to animal.

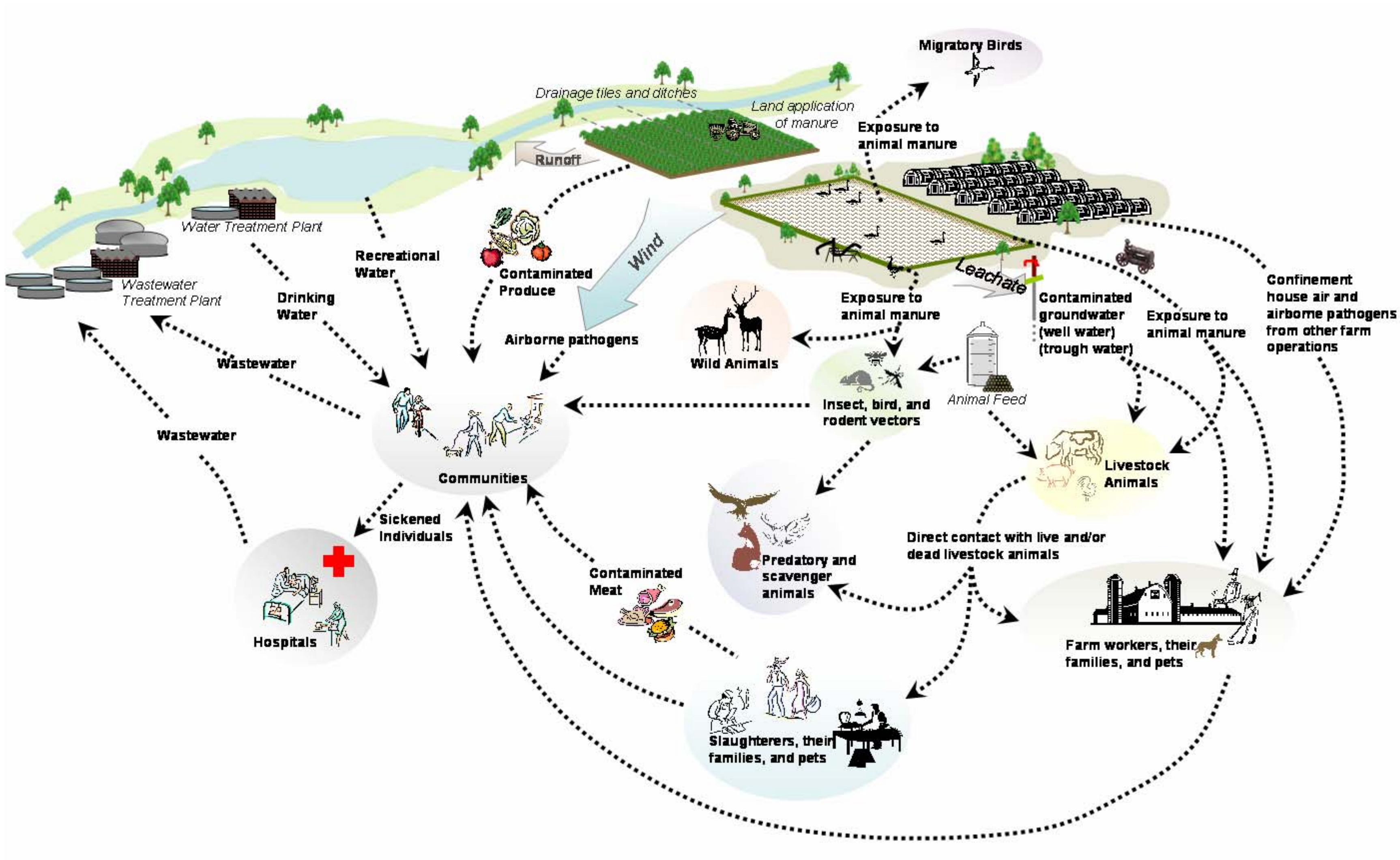

Figure 2. Movement of pathogens - an ecological perspective

Standing trough waters and animal feeds laden with antimicrobial compounds and pathogenic zoonoses as well as unsanitary conditions and poor manure management practices pose other problems for controlling the spread of disease. Animal and insect vectors may be attracted to feed piles, trough waters, fecal matter, manure treatment lagoons, treatment wetlands, or the dense animal populations present in CAFOs resulting in movement of pathogens on and between farms as well as off of farms and into human populations. Several studies have supported the movement of pathogens and antimicrobial resistant bacteria through animal vectoring. In a longitudinal study on Wisconsin farms, Shere *et al.*, (1998) reported that the use of antimicrobials subtherapeutically in animal feeds or trough waters and therapeutically for treatment of diarrhea correlated well to the emergence of antimicrobial-resistant *E. coli* O157:H7, which may have been transmitted through birds eating the animal feed and drinking contaminated trough waters. Nielsen et al (2004) screened 446 fecal samples at eight Danish cattle and swine farms and detected stx1 and stx2 genes in production animals, wild birds, and rodents suggesting transmission between the livestock animals and vectors. Halos *et al.*, (2004) detected the *Bartonella* citrate synthase gene in Hippoboscidae flies on wild roe deer, cattle, horses, and sheep in France suggesting that these flies may act as a vector for transmission of *Bartonella* between wild and domestic ruminants. Waldenström et al (2005) observed antimicrobial resistant *Campylobacter jejuni* in wild thrushes, shorebirds, and raptors in Sweden suggesting the spread of antimicrobial-resistant pathogens to wild birds. Raptors had the highest prevalence of antimicrobial-resistant strains, potentially from predation on infected animal vectors. On 12 diary and beef feedlots in Idaho, Oregon, and Washington, Herriott *et al.*, (2002) identified *E. coli* O157:H7 in composite fly samples and pooled bird droppings. Marshall *et al.*, (1990) inoculated pigs with an antimicrobial-resistant strain of *E. coli* and within a four month period was able to isolate the same strain from trough water, bedding materials, mice, flies, and a human caretaker.

Other studies may point to broader ecological implications of animal vectoring in the environment. Cole et al (2005) compared free-living Canadian geese in Craven county, Georgia that were using swine waste lagoons and surface waters adjacent to farm fields to Canadian geese in Griffin, Georgia, where there were crop fields, but no nearby animal production facilities. The proportion of *E. coli* isolates resistant to antimicrobial agents was significantly greater ($p=0.0004$) among Craven county geese (72%), where interaction with swine waste lagoons was observed, than in Griffin geese (19%). These researchers proposed that Canadian Geese may be acting as vectors for antimicrobial-resistance and resistance genes in agricultural animal-production environments. Their findings suggest the spread of pathogens and antimicrobial resistant bacteria from livestock operations may be vast considering potential migration of Canadian geese over hundreds of kilometers.

Other factors unique to concentrated animal feeding operations may encourage the spread of disease on farms. Several researchers have detected high levels of airborne bacteria ($2x10^3$ – $8x10^5$ CFU/m^3) in confinement house air including antimicrobial-resistant bacteria and other zoonotic pathogens such as *Enterococcus*, *Staphylococcus*, *Pseudomonas*, *Bacillus*, *Listeria*, *Salmonella*, *Campylobacter*, and *E. coli* (Cormier *et al.*, 1990; Cazwala *et al.*, 1990; Crook *et al.*, 1991; Heederick *et al.*, 1991; Predicala *et al.*, 2002). In houses of experimentally infected broiler chickens, Gast et al (2004) were able to detect *Salmonella* spp. in the air for four weeks post-infection, even when the litter was cleaned from the floors weekly. In a study of three mechanically ventilated swine CAFOs, Zahn *et al.*, (2001) detected tylosin resistance in 80% of cultivable airborne bacteria. Chapin *et al.*, (2005) isolated 137 *Enterococcus* and staphylococci from the air within a concentrated swine feeding operation and screened them for resistance to erythromycin, clindamycin, virginiamycin, tetracycline, and vancomycin. 88% of the isolates expressed high-level resistance to at least two antibiotics and 84% to at least three antibiotics commonly used in swine production, but none were resistant to vancomycin, an antibiotic that has never been approved for use in livestock in the United States. Thirty seven percent of the isolates were resistant to virginiamycin, an analog to quinipristin-dalfopristin which is a drug of last resort for multidrug-resistant gram-positive infections characterized by glycopeptide-resistant *E. facium* and coagulase-negative staphylococci.

These findings have significant implications for the health of livestock animals, animal care workers, their families, and casual farm visitors, and to a lesser extent to nearby communities that may be susceptible to secondary infections via exposure to sickened animal care workers and their families. Chapin *et al.*, (2005) proposed a scenario by which airborne pathogenic zoonoses resistant to clinically important antimicrobials may spread from confined livestock animals to the public through exposure to sickened animal care workers and their families.

> *"Bacteria resistant to virginiamycin are often cross-resistant to quinipristin-dalfopristin, and a previous study has shown that transfer of streptogramin-resistant Enterococcus can occur between animals and humans in the livestock environment (Jensen et al., 1998)… Inhalation of air contaminated with multidrug resistant Enterococcus or streptococci could lead to colonization of both the nasal passages (Aubry-Damon, 2004) and the lungs of swine CAFO workers, potentially making the workers themselves reservoirs of antibiotic-resistant organisms. Co-exposures to other aerosols and gases in the swine environment such as organic dusts, molds, and ammonia have been shown to induce symptoms associated with chronic bronchitis, including a persistent cough characterized by expectoration (Mackiewicz, 1998). The presence of this type of cough can increase the potential for secondary spread of antibiotic-resistant organisms into the community, where additional individuals could serve as reservoirs of multidrug-resistant bacteria… Thus, the inhalation of virginiamycin-resistant gram-positive bacteria in the swine environment could contribute to the appearance of quinipristin-dalfopristin-resistant gram-positive infections in humans, leaving few or no treatment options for the affected individual(s)"* – Chapin *et al.,* (2005)

The more recent work of Armand-LeFevre *et al.*, (2005), who determined that a number of *Staphylococcus aureus* strains that caused infections in swine populations (including four methicillin-resistant strains) were also present in healthy swine farmer nasal cavities, but not in the nasal cavities of healthy non-farmer controls, further supports their hypothesis.

Many other pathways for infection of animal care workers with pathogenic zoonoses and antimicrobial-resistant bacteria exist including, but not limited to direct contact with infected animals, increased exposure to insect and wild animal vectors on the farm, exposure to animal excreta, handling animal carcasses, exposure to contaminated air from manure spreading, and drinking water from fecally-contaminated wells (Skilbeck and Miller, 1986; Everard *et al.*, 1989; Seuri and Granfors, 1992;Thomas *et al.*, 1994; Hogue *et al.*, 1997; Cole *et al.*, 2000; Barkocy-

Gallagher et al, 2001; Chomel, 2004). These exposures have manifested in increased illness in animal care workers, their families and pets, and casual farm visitors (Levy *et al.*, 1976; CDC, 2000), and have potentially spread into nearby communities based on empirical evidence reported in literature. For instance, McDonald *et al.*, (1997) genotyped vancomycin-resistant fecal bacterial isolates from swine, poultry, farm workers and their pets in Denmark and concluded that transmission had occurred between livestock animals, humans, and household pets. Hummel *et al.*, (1986) detected nourseothricin-resistance traits in 33% of fecal isolates of swine exhibiting diarrhea, 18% of fecal isolates from swine farmers and their families, and 16% fecal isolates from outpatients exhibiting diarrhea in communities adjacent to the swine farms. Nourseothricin was not used for treatment of human disease in the region, but was used for two years for promoting growth of swine on the farms.

> The presence of antimicrobial-resistant bacteria may occur rapidly following the introduction of antimicrobials as growth promoters in feed animals. Levy et al (1976) determined that tetracycline-resistance in fecal isolates from chickens increased rapidly from 10% of animals excreting less than 0.1% of organisms resistant to tetracycline (baseline) to 90% of animals excreting 100% of organisms resistant to tetracycline within 2 weeks of introducing tetracycline to chicken feed, whereas no increase was observed in the control group. Further, multidrug resistance developed, even though only tetracycline was being supplemented in the feed. By 12 weeks, more than 60% of the animals from the experimental group excreted bacteria resistant to tetracycline plus one or more other antimicrobial compounds. More then 25% were resistant to 4 or more antimicrobials. After 4 months, antimicrobial resistance had spread from the experimental group to the control group, where a third of the chickens excreted bacteria of which more than 50% of isolates were tetracycline resistant. Within 6 months, antimicrobial resistance had also spread to the farm workers and their immediate families. More than 30% of fecal samples form farm workers and their families contained more than 80% tetracycline-resistant organisms, versus 6.8% from their neighbors. A 4-drug resistance pattern similar to that observed in the experimental chickens was observed in the farm workers and their families. Stopping the feed additives eventually reduced the incidence of tetracycline-resistant bacteria in the farm dwellers.

Livestock animals can also be a source of antimicrobial-resistant bacteria and pathogenic zoonoses such as *E. coli* O157:H7, *Salmonella* sp., and *Staphylococcus aureus* in abattoirs, which may slowly die off or in some instances regrow in the waste products (Hepburn *et al.*, 2002). When improperly handled, these wastes may potentially contaminate adjacent land and nearby watercourses or infect slaughters, and through secondary infections, their families and pets (Crawford *et al.*, 1969; Nesbakken, 1988; Molin *et al.*, 1989; Reboli and Farrar, 1989; Merilahti *et al.*, 1991; Seuri and Granfors, 1992; Huys *et al.*, 2005). In fact, as with animal care workers, epidemiological evidence supports transmission of these pathogens from livestock animals to humans in the abattoir environment, but suggests the infection rate is lower than that observed in farmers and their families. For instance, van den Bogaard *et al.*, (1997) phenotyped fecal *Enterococcus* spp. isolates from turkeys, turkey farmers, turkey slaughterers, and nearby residents of the turkey farms in Europe. Vancomycin-resistant *Enterococcus* (VRE) was detected in half of turkey samples, 39% of turkey farmers, 20% of turkey slaughterers, and 14%

In a recent study, van den Bogaard *et al.*, (2001) surveyed three poultry operations (broilers, turkeys, and laying hens) and five human populations (turkey farmers, broiler farmers, laying-hen farmers, broilers slaughterers, and turkey slaughterers) in the Netherlands for antimicrobial-resistant fecal *E. coli*. These researchers determined that 35% of isolates from laying hens were antimicrobial-resistant, as compared to 84% of the isolates from turkeys and 80% of the isolates from broilers. Similarly, 66% of *E. coli* isolates from turkey farmers, 60% from broiler farmers, 67% from turkey slaughterers, and 59% from poultry slaughterers were antimicrobial-resistant whereas only 45% of isolates from laying hen farmers were antimicrobial-resistant. Antimicrobial resistance patterns of the isolates were similar between turkeys, turkey farmers and turkey slaughterers, and in broilers, broiler farmers, and broiler slaughterers. Pulsed-field gel electrophoresis (PFGE) "fingerprinting" patterns of an *E. coli* isolate from a turkey was identical to one from a turkey farmer. Similarly, one isolate from a broiler chicken was identical to an isolate found in a broiler chicken farmer. Their results strongly indicate transmission of antimicrobial resistant bacteria between humans and poultry commonly occurs.

of area residents. VRE is one of the leading causes of nosocomial infections in the hospital environment. Nijsten *et al.*, (1994) determined that the resistance of fecal isolates to antimicrobial compounds was more prevalent in swine farmers than slaughterhouse workers and suburban residents in the same geographic region. In Japan, antimicrobial resistance of fecal microbes was also noted to be highest in swine farmers and elevated in slaughterhouse workers when compared to urban control cohorts (Saida *et al.*, 1981). Others have realized similar trends supporting the movement of antimicrobial-resistant bacteria from farm animal to farmer or slaughterer (Ozanne *et al.*, 1987; Levy, 1978; Marshall *et al.*, 1990).

5.2 Food

It is well established that pathogenic zoonoses can cause human disease via consumption of contaminated meat products (Corpet, 1993; U.S. Congress, Office of Technology and Assessment, 1995; Milleman *et al.*, 2000). The amplified use of antimicrobial compounds in confinement animals for growth promotion and prophylaxis may exacerbate disease by reducing treatment options and potentially increasing the virulence of bacterial pathogens in meats. For instance, in 1995 fluoroquinolone antibiotics were approved for use in poultry for growth promotion and prophylaxis. In 1997, Smith *et al.*, (1999) screened chicken obtained from Minnesota shopping markets that originated from 15 abattoirs in nine states for *Campylobacter jejuni* and resistance to ciprofloxacin, an important human-use fluoroquinolone antibiotic of choice for presumptively treating severe bacterial food poisoning. Fourteen percent of the samples were contaminated with ciprofloxacin-resistant *C. jejuni*. During a similar period, statewide-surveillance indicated that fluoroquinolone-resistance increased from 1.3% of all human *C. jejuni* infections in 1992 to 10.2% in 1998 (Smith et al, 1999). In a more recent study, Wallinga *et al.*, (2002) surveyed 200 fresh whole market chickens and 200 packages of ground turkey from stores in Iowa and Minnesota and determined that 95% of whole chickens were contaminated with *Campylobacter* and 18% with *Salmonella*. Two percent of ground turkey samples and were contaminated with *Campylobacter* and 45% with *Salmonella*. Six percent of the *Salmonella* isolates were resistant to 4 or more antimicrobials, while 62% of the *Campylobacter* isolates were resistant to 1 or more antimicrobial compound including an 8%

prevalence of resistance to ciprofloxacin. Greater than ninety percent of enterococci isolated from the chicken or turkey were resistant to quinipristin-dalfopristin, an important antibiotic for the control of VRE infections in hospitals. In a similar study, Hayes *et al.*, (2003) screened 981 samples of raw retail meats including chicken, turkey, pork, and beef from 263 grocery stores in Iowa and found high levels of resistance to several antimicrobials in *Enterococcus* isolates. Their results indicate that antimicrobial-resistant *Enterococcus* spp. commonly contaminate retail meat products and that the antimicrobial resistance pattern of isolates from each meat product (poultry, pork, and beef) reflected well the use of approved agents in each food animal production class (broilers, swine, and beef cattle).

Dairy products may also be contaminated with pathogenic zoonoses and antimicrobial-resistant bacteria following direct contact of dairy cattle to contaminated sources in the farm environment and subsequent excretion from the udders of infected animals (Oliver *et al.*, 2005). For instance, Van Kessel *et al.*, (2004) surveyed 861 bulk tank milks on farms in 21 states and detected *Listeria monocytogenes* (6.5%) and several *Salmonella* serotypes (2.6%) including Montevideo, Newport, Muentster, Meleagris, Cerro, Dublin, and Anatum. Kim *et al.*, (2005) tested 316 bulk milk tank samples across the U.S. between January 2001 and December 2003 for *Coxiella burnetii*, the causative agent for Q-fever. These researchers detected *C. burnetii* in greater than 94% of bulk tank milk. Jayarao and Henning (2001) surveyed bulk tank milks from 131 dairy herds in South Dakota and Minnesota and detected *Campylobacter jejuni* (9.2%), shiga-toxin producing *Escherichia coli* (3.8%), *Listeria monocytogenes* (4.6%), *Salmonella* spp. (6.1%), and *Yersinia enterocolitica* (6.1%), with one or more species of pathogenic bacteria in 26.7% of the samples. Although pasteurization may reduce the incidence of disease in humans attributable to contaminated milk, Oliver *et al.*, (2005) argue that outbreaks of disease have been traced back to both unpasteurized and pasteurized milk, and that unpasteurized milk is often consumed directly by dairy producers, farm employees, and their families, as well as by their neighbors and raw milk advocates. In their bulk tank-milk study, Jayarao and Henning (2001) observed that 60% of the dairy producers drank unpasteurized milk, 27% of which contained one or more types of pathogenic bacteria. According to the model presented in Figure 1, disease contracted via this route may be spread to nearby communities via contact with infected individuals. It has also been noted that an even larger segment of the population may be directly exposed to contaminated dairy products via consumption of cheeses made from unpasteurized milk (Oliver *et al.*, 2005).

CAFOs produce massive quantities of manure, much of which is spread onto agricultural fields as fertilizer. Fecally-contaminted water, potentially resulting from runoff from manure-treated fields or discharge of wastes from agricultural operations, may be used to irrigate crops in arid regions of the United States. Direct contact with soils on which manure was applied and/or irrigation with fecally-contaminated water may result in contamination of produce such as lettuce, radishes, apples, and sprouts with pathogenic zoonoses including antimicrobial-resistant bacteria, especially where the edible parts are exposed to the soil or water (Besser *et al.*, 1993; Tschäpe *et al.*, 1995; Nelson, 1997; Taormina *et al.*, 1999). In a recent study, Ingham *et al.*, (2004) identified *E. coli* contamination on carrots, lettuce, and radishes up to 120 days following application of non-composted bovine manure as a fertilizer in fields in Wisconsin. In contrast, Johannessen *et al.*, (2005) did not detect *E. coli* O157:H7 on the edible parts of lettuce after growth in *E. coli* O157:H7-contaminated manure fertilized soil. Solomon *et al.*, (2002) noted that spray irrigation following a single exposure to *E. coli* O157:H7 resulted in 90% of the lettuce being contaminated, persisting for more than 20 days in 82% of the plants. Where surface irrigation was used under the same circumstances, only 19% contamination was observed on the lettuce. Immersion of harvested lettuce heads in 200ppm chlorine solution for 1 minute did not eliminate all *E. coli* O157:H7 cells from infected lettuce, regardless of irrigation method. It has been suggested that some produce may absorb pathogens into their internal tissues through the root system, protecting them from cleaning procedures such as washing or irradiation. In a survey of fresh domestic produce conducted in the spring of 2000, the US Food and Drug Administration detected Salmonella on 2.6% of cantaloupe, 1.6% of cilantro, and 1.8% of lettuce originating from U.S. farms (US FDA, 2001).

5.3 Air

Vast quantities of manure produced at CAFOs containing high levels of pathogenic microorganisms and antimicrobial-resistant bacteria are applied to agricultural lands each year. Viable bacteria and viruses become airborne from agricultural sprayers, pasturelands, and farm fields treated with manure, ultimately decreasing the quality of air near CAFOs. The upward flux of viable bacteria may be strongly related to plant cover and soil moisture condition. For instance, upward flux of viable bacteria from bare soil and various crops has been reported to increase an order of magnitude in dry soil over young corn in wet soil, another order of magnitude in a closed wheat canopy over dry soil, and four orders of magnitude between bare soil and an alfalfa field (Lindemann *et al.*, 1982). Although plants have been found to be a stronger source of bacteria than soil (Lindemann and Upper, 1985), specific agricultural practices that increase particle emissions may significantly impact bacterial loading to an airshed. Strong vertical temperature gradients, low relative humidity and low soil moisture may lead to increased emission of PM10 from agricultural fields during tilling (Holmen *et al.*, 2000; Clausnitzer and Singer, 2000). As dust may harbor viable bacteria, these factors may increase pathogen loading to an airshed. If pathogens survive in soils until harvest, it is possible that significant airborne spread may occur. It has been estimated that during harvest, up to 42% of bacterial loading in an airshed can be attributed to harvesting activities (Lighthart, 1984; Tong and Lighthart, 2000).

Upward flux of viable bacteria may also be related to temperature, exposure to solar radiation, protection from associated soil particles, and wind speed. Lindeman and Upper (1985) report that upward flux of bacteria over bean plants in Wisconsin occurred during the warmest parts of the day with a maximum around noon, especially on windy days, and was observed to cease when wind speeds were less than 1m/s. Tong and Lighthart (1999) suggest that peak concentrations of viable bacteria over agricultural lands in Oregon may occur in late afternoon, presumably due to less exposure to solar radiation or association with larger particles protective from the effects of the sun. Upward flux of viable bacteria over a high desert chaparral have been observed to peak late in the evening, with minimum viable bacterial concentrations at 13:20 hours and a maximum at 22:00 hours, presumably due to the strong effects of solar radiation (Lighthart and Schaffer, 1994). The effects of temperature on upward flux of bacteria may overcome the effects of solar radiation. Summer months have been associated with higher incidence of airborne viable bacteria, even though increased solar radiation may negatively influence bacterial viability due to UV damage or desiccation of bacterial cells (Tong and Lighthart, 1997; Tong and Lighthart, 2000).

Several studies have documented increased airborne pathogens directly attributable to the spread of human or animal manure on agricultural lands through spray irrigation with contaminated waters or deposition of animal placental and fecal wastes and subsequent distribution to downwind animal or human receptors (Boutin *et al.*, 1988; Hughes, 2003; Donnison *et al.*, 2005; Brooks *et al.,* 2005). Boutin *et al.*, (1988) identified bacterial counts as high as 2000 viable particles per cubic meter at the edge of applied areas following land spreading of cattle and pig slurry. Donnison *et al.*, (2005) studied the survival of *Bacillus subtilis* and *Serratia entomophila* in irrigation aerosols in spring and summer in New Zealand. Viable *B. subtilis* and *S. entomophila* corresponding to the respirable fraction of inhaled air were recovered at 100 m from a low pressure sprayer and 200 m from a high pressure sprayer. Brooks et al (2005) studied the aerosolization of *E. coli* and coliphage MS-2 from liquid biosolids applied from a spray tanker under hot (22-37.5C) and arid (5-15% relative humidity) conditions. At wind speeds between 0.7-6 m/s, these researchers could not detect aerosolized *E. coli* at distances as low as 2 m, but detected coliphage MS-2 at distances as far as 60 m. Paez-Rubio et al (2005) identified aerosolization as a potential mechanism for the dissemination of wastewater bacteria and other microorganisms at flood irrigation wastewater reuse sites. These researchers identified more than 1 billion enteric bacteria per cubic meter in downwind air samples. Airborne *Coxiella burnetii* associated with sheep operations and Picnoravirus from swine operations have been estimated to travel several kilometers in the air in concentrations sufficient to cause infectious disease in humans and animals (Henderson, 1969; Hugh-Jones and Wright, 1970; Smith *et al.*, 1993; Hawker *et al.*, 1998; Lyytikainen *et al.*, 1998).

5.4 Recreational and Drinking Water

The USEPA's 1998 National Water Quality Inventory indicates that agricultural operations, including animal feeding operations, are the most common polluters of rivers and streams, contributing to the impairment of 59% of those surveyed. Agricultural operations also have significant impacts on lakes, ponds, reservoirs, and estuaries, contributing to the impairment of more than 3,590,000 acres of these valuable water resources (USEPA-NACAC, 2005). Figure 3 illustrates the relationship between confined livestock animals in the U.S. in 1997 and impairment of surface waters indicated in the 1998 National Water Quality Inventory. In 1998, pathogens (microbial indicators, not overt pathogens) were the most common water pollutant contributing to 7,742 impairments (14.24% of surveyed waters). Sources of these microorganisms may have included wastewater and storm water outflows, the spreading of biosolids and animal manures on agricultural lands, and wild animals. However, the sheer quantities of animal wastes generated and spread onto land compared to those of other sources suggests animal agriculture may be the dominant contributor. Pathogenic microorganisms continue to pose a major challenge to the quality of U.S. waters, contributing to a total of 7,894 impairments (13.16% of surveyed waters) on the USEPA's 2002 National Water Quality Inventory.

A survey of literature regarding overt pathogens in agricultural waters and drinking water sources suggest that the National Section 303(d) listings may underscore the actual extent of microbial contamination in agricultural watersheds resulting from livestock activities. For instance, two surveys of source waters for surface water treatment plants in 29 states resulted in detection of *Cryptosporidium* oocysts in 55% and 87% of the waters tested. Similarly, *Giardia*

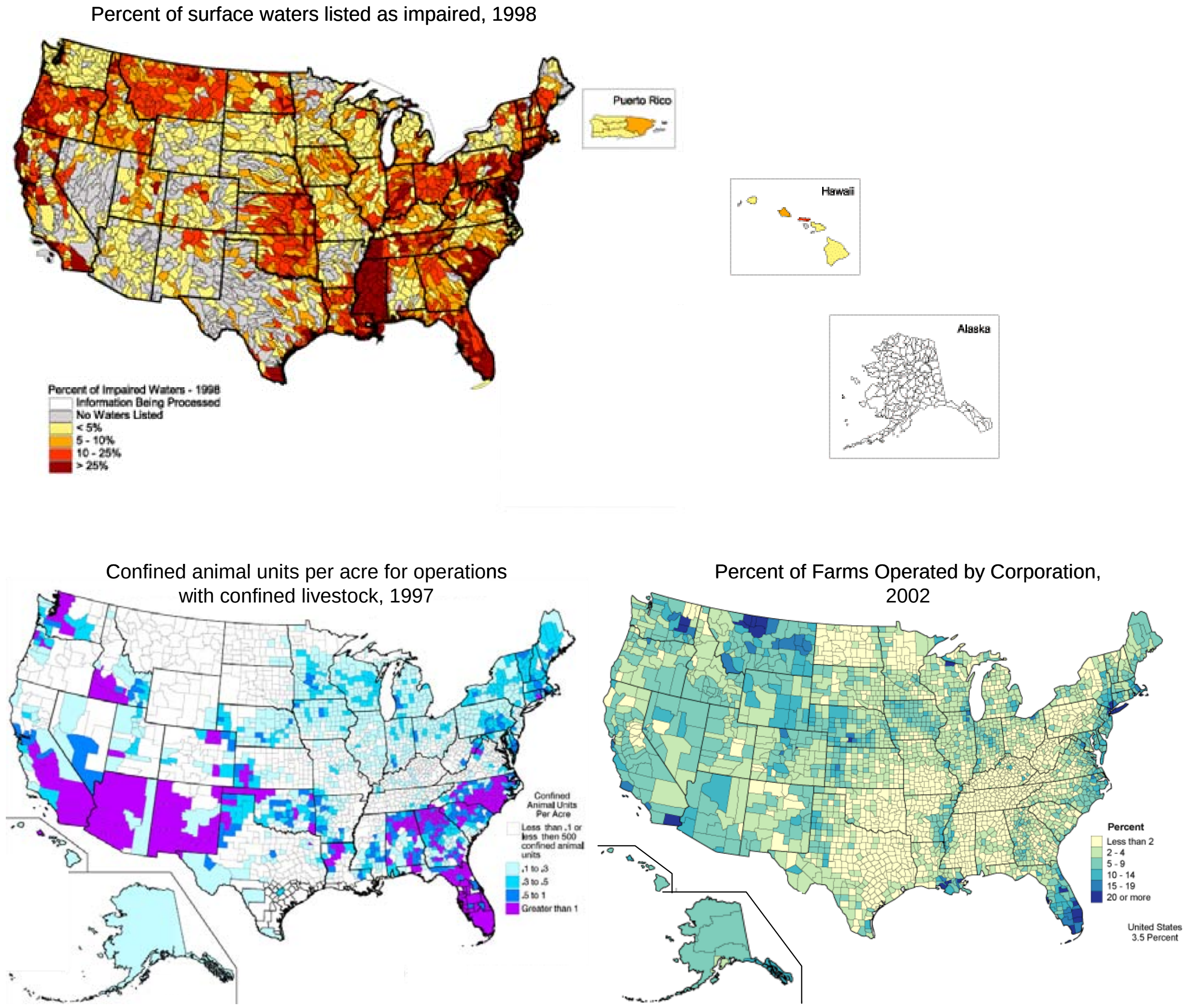

Figure 3. The impact of confined animal feeding operations on agricultural watersheds (adapted from USDA-NRCS, 2002; USEPA, 1998)

cysts were detected in 16% and 81% of the waters tested (LeChevallier *et al.*, 1991; Rose *et al.*,1991). *Mycobacterium avium*, potentially from cattle, swine, and broiler operations, have been detected in several marine waters, rivers, lakes, streams, ponds, and spring waters (AWWARF, 1997; Ichiyama *et al.*, 1988; Falkinham *et al.*, 2001; LeChevallier, 1999). A high prevalence of *Campylobacter* spp. in environmental water samples in a dairy farming area in the United Kingdom including 56.7 % of running waters (streams and ditches) and 45.9% of standing waters (ponds) was recently noted (Kemp *et al.*, 2005). Swine farming activities have been implicated in the contamination of at least one major Canadian river by enteroviruses (Payment, 1989), and have also been correlated to the presence of *Cryptosporidium parvum* oocysts, *Yersinia enterocolitica*, and *Salmonella* spp. in nearby drainage canals, groundwater wells, and surface waters where lagoon and spray systems are used (CDC, 1998). Groundwater surveys in Ontario Canada indicated that wells located near manure application areas were at higher risk for fecal bacterial contamination, and the level of contamination was inversely correlated to the distance the wells were from animal feedlots or exercise yards (Conboy and Goss, 2002).

The delivery of zoonotic pathogens to environmental waters following manure application is dependent on several factors including, but not limited to, the initial and persisting pathogen load, properties of the pathogen of interest, the soil and vegetation type, travel distance/time to the receiving water, pathogen inactivation by various environmental stressors, and potential engineered or natural barriers to pathogen transport. When manure is spread onto land, overland flow of pathogens to water bodies may occur via attachment to applied waste products, attachment to soil particles, or movement in the free form (Tyrrel and Quinton, 2003, Muirhead et al, 2005). Leachate from manure-amended fields and poorly designed manure holding lagoons may inundate natural soil barriers resulting in contamination of underlying groundwater (Jones, 1980; Kowel, 1982; Natsch *et al.*, 1996; Jogbloed and Lenis, 1998). Within the soil profile, movement of pathogenic zoonoses may be limited by soil moisture and solid-phase interactions (adsorption-desorption) or facilitated by the presence of macropores from burrowing animals, fractured media, or plant roots (Ferguson *et al.*, 2003). Contaminated groundwater may be captured by drainage tiles that discharge to surface waters, bypassing overland treatment in natural or engineered vegetative or riparian buffers. Alternatively, pathogens may enter groundwater where they may potentially migrate towards wells or natural springs that may be used for drinking water. The delivery of an infective dose to a susceptible individual will depend not only on the transport properties of pathogens, but also on the time required for pathogen inactivation due to environmental stressors or predation in surface and groundwater resources.

The concentration of pathogens reaching subsurface tile drains that discharge to nearby streams often exceeds drinking water supply and recreational use standards (Warnemuende and Kanwar 2000). Tile drainage has been noted to be a significant pathway for pathogens to enter surface waters from manure-treated fields, especially during periods of wet weather (Dean and Foran, 1992; Joy *et al.*, 1998; Geohring *et al.*, 1999; Hunter *et al.*, 2000; Monaghan and Smith, 2005). Evans and Owens (1972) noted that approximately 0.05% if *E. coli* applied with swine manure applications to a sandy clay loam pasture could be recovered in the tile drainage water. In a study of swine operations in Iowa and Missouri, Karetnyi *et al.*, (1999) identified swine hepatitis E in a tile outlet draining a field to which manure had been applied. Evans and Owens (1972) determined that fecal bacteria present in swine waste slurries could be detected in the tiles draining the pasture to which the waste was applied within a few hours of application.

Where pathogens bypass tile drainage systems or tile drainage is non-existent, significant contamination of groundwater resources may occur. The transport of pathogens that have infiltrated the soil profile depends strongly on adsorption-desorption interactions. Key characteristics of pathogenic zoonoses related to adsorption-desorption phenomena include size, surface electrostatic properties, cell wall hydrophobicity, and the presence of flagella (Gerba, 1984; Dowd *et al.*, 1998; Heise and Gust, 1999). For viruses, attachment to soil particles is rapid, and may be increased by low pH or high ionic strength groundwater or by high soil organic carbon content (Gerba, 1981; Gerba *et al.*, 1978; Goyal and Gerba, 1979; Taylor *et al.*, 1980; Moore *et al.*, 1981; Taylor et al, 1981; Moore, 1982; Singh *et al.*, 1986; Bales *et al.*, 1991; Bales *et al.*, 1993; Sakoda *et al.*, 1997). At the neutral pH of most groundwater, organic carbon content of the soil may dominate retardation of viral particles. Retardation of bacterial particles in saturated porous media may also be dominated by organic-carbon partitioning, but can also be a product of straining or simple filtration (Heise and Gust, 1999). Straining and filtration may be

In a collaborative study performed at 9 Swine CAFOs in Iowa employing lagoon and spray systems, the CDC tested the swine waste lagoons and several selected points near the agricultural facilities for pathogenic zoonoses. They identified elevated concentrations of *E. coli* (≤380,000 per 100mL), *Enterococcus* sp. (≤1,900,000 per 100 mL), *Salmonella* sp. (≤9,300 per 100 mL), and *Cryptosporidium parvum* oocysts (≤2250 per liter) in the swine waste lagoons. *C. parvum* oocysts were detected in monitoring wells nears the swine waste lagoons of three CAFOs (9-15 oocysts per L) and in the river adjacent one CAFO (6 oocysts per L). A single *Yersinia* sp. was detected in an agricultural drainage ditch draining the spray field at one facility. Elevated *E. coli* were detected in the agricultural drainage wells (300-740/100mL), drainage ditches (520-3,700/100mL), monitoring wells (10-390/100mL), and drainage tile inlet/outlets (10-2,900/100mL). Similarly, *Enterococcus* sp. were detected in the agricultural drainage wells (4,500/100mL), drainage ditches (610-13,000/100mL), monitoring wells (80-910/100mL), and drainage tile inlet/outlets (30-2,400/100mL). *Campylobacter* sp. were not detected at any of the sampling points. Of the 18 *E. coli*, 3 *Salmonella* sp., and 20 *Enterococcus* sp. isolates tested for antimicrobial resistance, 16 *E. coli*, and all 3 *Salmonella* and 20 *Enterococcus* sp. were resistant to one or more antimicrobials commonly used in swine management practice as feed supplements and therapeutics (16 total including fluorfenicol, tetracycline, sulfamethoxazole, ampicillin, streptomycin, apramycin, bacitracin, lincomycin, penicillin, synercid, kanamycin, cephalothin, amoxicillin-clavulanic acid, ceftiofur, chloramphenicol, and gentamicin). Eight *E. coli* and all 3 *Salmonella* sp. and 20 *Enterococcus* sp. were multi-drug resistant (2-11 antimicrobials) (CDC, 1998).

even more significant for the larger protozoan parasites such as Cryptosporidium oocysts and Giardia cysts. Once contaminated, restoration of water quality in contaminated aquifers is very slow (Olson, 2003).

In packed sand columns, it has been demonstrated that Cryptosporidium oocysts, although initially filtered, may exhibit time-dependent detachment leading to a constant low-level elution from porous media (Harter *et al.*, 2000). Free oocysts have been observed to move in pore water without retardation suggesting potential for considerable transport in aquifers considering their long survival times (Brush *et al.*, 1999; Harter *et al.*, 2000). Similarly, viruses become attached to sediments near the source of contamination and leach slowly into the groundwater. Therefore, even single contamination events may provide a lingering source of viral contamination to groundwater (de Borde *et al.*, 1998b). Viruses have been shown to be able to travel considerable distances through the subsurface depending on their size, adsorption characteristics, and degree of inactivation (Keswick and Gerba, 1980; Dowd *et al.*, 1998). For instance, enteric viruses, some of which may remain infective for more than 9 months, have been observed to move up to 1000-1600 m per year in channelized limestones and several hundred meters per year in glacial silt-sand aquifers with travel times similar to bromide tracers (Skilton and Wheeler, 1988; Bales et al, 1995; Bosch, 1998; de Borde et al; 1998a; de Borde et al; 1999). Bacterial pathogens may similarly move considerable distances as indicated in a study by Withers *et al.*, (1997), who identified groundwater contamination by *E. coli* from an unlined cattle waste lagoon 76 m below ground surface and 80 m downstream the lagoon in the United Kingdom.

A significant limitation of the National 303(d) listings is the lack of monitoring for antimicrobial-resistant bacteria. Antimicrobial-resistant bacteria are generally shed in animal feces, but may also be present in the mucosa of livestock animals. The massive use of antibiotics in animal agriculture pose a great risk as antimicrobial-resistant bacteria shed in animal wastes and stored in lagoons or spread onto land may eventually find their way to the aquatic environment (CDC, 1998; Levy, 1998; Chee-sanford *et al.*, 2001). For instance, Chee-Sanford *et al.*, (2001) detected all eight classes of tetracycline-resistance genes in two swine waste lagoons and the underlying groundwater up to 250 meters down-gradient the lagoons. Tetracycline resistant bacterial isolates from groundwater harbored a *tet*(M) gene identical to that detected in the swine waste lagoons. Resistance genes from antimicrobial-resistant bacteria in contaminated discharge waters can be transferred to otherwise susceptible bacteria living in unpolluted aquatic habitats, encouraging the spread of antimicrobial resistance in environmental waters (Guardabassi and Dalsgaard, 2002; Gurdin *et al.*, 2002). The extent of proliferation may be limited by the distance from the discharge point.

Antimicrobials in livestock animals are primarily removed in the urine and bile, either unchanged or in metabolite form, and therefore can directly contaminate environmental waters. Once in the environment, antimicrobial compounds and their metabolites may degrade rapidly (tetracyclines, penicillins, and fluoroquinolones) or persist (macrolides and sulfonamides), resulting in long-term contamination near animal confinement operations. For instance, Campagnolo *et al.*, (2002) detected several antimicrobials used in animal agriculture in animal waste lagoons (2.5-1000 µg/L) and in monitoring wells, field drainage tiles, springs, streams, and rivers (0.06-7.6 µg/L) proximal to confined animal feeding operations in Iowa and Ohio. The use of antimicrobials in animal agriculture most certainly contributed to the frequent detection of antimicrobials in a recent U.S.G.S. survey of rivers and streams of the United States (Kolpin *et al.*, 2002). Although the presence of a pharmaceutical residues and their metabolites in potable water sources present their own ecological challenges (Goni-Urriza *et al.*, 2000; Zuccato *et al.*, 2000; Hirsch *et al.*, 1999; Halling-Sorensen *et al.*, 1998; Daughton *et al.*, 1999), their typical concentrations in environmental waters are usually far below (approximately 1000-fold) those that would selectively enrich for resistant bacteria. Resistant bacteria found in surface waters are likely to have originated from wastewater or manure runoff from antimicrobial-rich settings such as animal feeding operations or wastewater treatment plants or subsequently contaminated animal vectors (Levetin, 1997; Stetzenbach, 1997; Summers, 2002).

Although low environmental concentrations of antimicrobials may not be adequate to enrich for resistant strains of bacteria, their role in the proliferation and maintenance of antimicrobial-resistance genes in these complex milieus is uncertain. For example, Gurdin et al (2002) screened isolates of *E. coli* and enterococci from swine farm wastes, and environmental isolates of *E. coli*, enterococci, *Kleibsiella*, and *Aeromonas* in surface waters upstream and downstream of study farms for antimicrobial resistance. These researchers observed that the diverse resistance patterns exhibited by rural background surface water isolates likely reflected human and animal impacts. In contrast, bacteria isolated downstream from swine farms exhibited increased antimicrobial-resistance that reflected the swine waste isolates. Sixty seven percent of Aeromonas and 12% of enterococci isolates upstream the study farms were resistant to erythromycin, whereas 91% of Aeromonas and 30% of enterococci isolates down-stream the study farms were resistant. Antimicrobial residues were also more likely to be detected downstream rather than upstream swine farms. However, antimicrobial resistance did not always correlate to detection of residues. Swine farms were shown to be capable of contributing resistant enteric bacteria that act as reservoirs for the spread of resistance traits to susceptible bacteria, and antimicrobial residues which may encourage the maintenance and spread of the resistance traits. More work is needed to clearly identify threshold concentrations of antimicrobial residues in environmental waters that encourage the spread of antimicrobial resistance.

5.5 Hydrologic events

Once in natural water bodies, viral particles, bacteria, and protozoan cysts and oocysts may attach to larger particles such as organic matter or soils and settle into the sediments of streams or reservoirs. Due to their size, settling of free particles may be limited. Their association with sediments may offer some protection from environmental stressors such as solar and UV radiation, pH extremes, desiccation, antibiotics, and predators leading to increased survival (Gerba and McLeod, 1976; Smith *et al.*, 1978; Roper and Marshall, 1979; Bitton and Marshall, 1980; LaBelle and Gerba, 1980; Schaiberger *et al.*, 1982; Metcalf *et al.*, 1984; Rao *et al.*, 1984; Long and Davies, 1993). .As such, the sediments of natural water bodies may act as reservoirs for pathogenic zoonoses and antimicrobial-resistant bacteria discharged from CAFOs (Hendricks, 1971; Grimes, 1975; Gerba *et al.*, 1977; Davies *et al.*, 1995). For instance, in estuary waters, Metcalf *et al.*, (1984) detected enteroviruses and rotaviruses in 14 and 50% of two water samples but 72 and 78% of their respective sediments contained these viruses. In 20-70% of surface waters, it has been observed that viruses occur as solid-associated particles, and may be present in high concentrations in bed sediments when compared to overlying water even at vast distances from the original source of contamination (Ferguson *et al.*, 2003).

The movement of pathogens from CAFO operations can be exacerbated by rainfall, which may stimulate the release of pathogens from otherwise stable manure-treated fields or fecal pats leading to increased overland transport, discharge to surface waters by drainage tiles, or infiltration into groundwater resources (Kress and Gifford, 1984; Mawdsley *et al.*, 1996a,b; Hunter *et al.*, 2000; Ogden *et al.*, 2001; Davies *et al.*, 2004; Monaghan and Smith, 2005). Often, stream flow increases significantly during hydrologic events, stirring up bedded sediments and further increasing pathogen concentrations, especially in shallow surface waters (Ferguson *et al.*, 2003). For example, Ferguson (1994) determined that an increase of 1-cm in rainfall increased *Cryptosporidium* oocysts in the Georges River by 24%. Atherholt *et al.*, (1998) demonstrated a positive correlation between parasitic protozoan concentrations in the Delaware River Watershed and precipitation events. Kistemann *et al.*, (2002) measured *E. coli*, fecal streptococci, *Clostridium perfringens*, *Cryptosporidium*, and *Giardia* in the tributaries of 3 drinking water reservoirs during normal and wet weather events and noted a 1-2 $\log_{10}$ increase in bacterial and parasitic microbial concentrations during runoff compared to normal conditions. Crowther *et al.*, (2002) observed highly significant positive correlations between concentrations of coliforms, *E. coli*, and enterococci in two watersheds in the United Kingdom during hydrologic (high flow) events and land use/management variables associated with intensive livestock farming. High flow conditions were associated with a greater than 10-fold increase in geometric mean fecal indicator concentrations (coliforms, *E. coli*, and enterococci) potentially due to storage and resuspension of viable organisms in channel bed sediments. Kunkle (1972) noted a marked dependence of bacterial concentrations on stream flow in the Sleepers River Basin near St. Johnsbury, Vermont, and emphasized the importance of stream surveillance that accounts for the hydrology involved. Joy *et al.*, (1998) reported bacterial contamination of surface water due to the application of liquid manure by accepted practices over a two year period. Drainage tiles were determined to deliver significant amounts of bacteria to surface waters, which was exacerbated by rainfall shortly following manure application.

Extreme precipitation may pose more significant problems for CAFO operators as lagoons and other engineered manure management systems such as vegetative buffers, infiltration basins, and constructed wetlands may be challenged by the level of flooding associated with these events. Passive manure management systems are typically designed for 20-50 year flood events, and may be overtopped during more rigorous flooding. Flood waters may engulf vegetative buffers allowing direct contact with animal wastes applied to fields. Flooding may also engulf animal confinement houses drowning animals and transporting raw fecal material and animal carcasses downstream. Waste management systems that do not fail will experience elevated discharge, reducing their efficacy as a barrier to pathogens. Because of the potential liability associated with overtopping or failing waste lagoons during flooding, many CAFO operators opt to spray down their lagoons during heavy rainfall in lieu of violating freeboard limits (Wing *et al.*, 2000). Significant environmental damage associated with intentional and accidental release of manures and other potentially infectious materials from CAFO operations during flooding events has been documented and the danger still persists (Taylor, 1999; Mallin, 2000; Scmidt, 2000; Wing *et al.*, 2002). In 1999, Hurricane Floyd flooded several CAFOs and caused extensive environmental damage to river and coastal waters in North Carolina. During this event, it was estimated that dozens of animal waste lagoons were breached and more than 100,000 hogs, 2.4 million chickens, and 500,000 turkeys drown in the flood waters. Wing *et al.*, (2002) estimate that greater than 240 CAFOs still operate within the region flooded by this category 3 hurricane.

6. Public Health Outcomes

Pathogens may enter and proliferate in a farm environment through the stocking of new animals, exposure to airborne pathogens from an upwind source, contaminated trough water or feed, insect or rodent vectors, human-to-animal and animal-to-animal transmission, to name a few. Concentrating animals in confinement with suboptimal hygiene may encourage the spread of disease within farms. As discussed eariler (Section 4: Survival of Pathogens in the Environment), the survival of zoonotic pathogens in animal manures and the environment can range from days to years depending on the pathogen, the medium, and environmental conditions. Where animal wastes are improperly managed, there exists potential for the movement of pathogens off farms and into nearby water, land, and air. Uncontrolled releases of pathogens may occur via runoff, aerosolization, or infiltration into soils and groundwater, especially when manure is spread onto land. Stored animal feeds and manure can attract animal vectors that can spread disease within a farm, to nearby farms or communities, or, in the case of migratory birds, over large distances spanning hundreds of kilometers. Animal care workers are exposed to elevated levels of pathogens in confinement house air and through direct contact with livestock and animal manures, leading to an increased incidence of illness and spread of disease to their families and communities. A similar trend is seen in abattoir workers and their families due to the proliferation of pathogens within slaughterhouse environments. Contamination of produce or meat products with zoonotic pathogens may further spread disease within human populations. Even where extensive management practices are in place, exposures can and do occur. The outcomes of these exposures are animal and human disease, sometimes with serious consequences.

6.1 Waterborne and Foodborne Outbreaks

The impacts animal feeding operations may have on public health are evident in surveillance of waterborne and foodborne outbreaks in the U.S. reported by the Centers for Disease Control and Prevention (CDC). Table 7 summarizes the CDC outbreak data between 1991 and 1997. During this period, there were more than 3,900 reported outbreaks infecting more than 500,000 individuals. Based on reported data, foodborne outbreaks were 8.3 times more likely to be reported than waterborne outbreaks. However, waterborne outbreaks tend to affect larger numbers of individuals per incident, most likely because communities share drinking water resources and recreational waters. Between 1991 and 1997, the number of infected individuals per waterborne outbreak was 35 times larger than for foodborne outbreaks (2-3 times larger discounting the *Cryptosporidium* outbreak in Milwaukee in 1993 that infected more than 400,000 individuals). Of the outbreaks of known etiology reported from 1991-1997, slightly less than half (48%) of the recreational water outbreaks and nearly two thirds (66%) of the outbreaks associated with untreated drinking water were caused by zoonotic pathogens. During the same period, 82% of the foodborne outbreaks of known etiology were caused by zoonotic pathogens. The pathogens most often

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associated with outbreaks include *Giardia*, *Cryptosporidium*, *Campylobacter*, *Salmonella*, and toxigenic *E. coli* (including *E. coli* O157:H7, *E. coli* O126:NM, and *E. coli* O121:H19). As noted above, all of these microbial agents are endemic in cattle, swine, and poultry flocks, and all are characterized by a low infectious dose.

Although the number of outbreaks and cases of illness reported to the CDC due to recreational and drinking water exposure, as well as foodborne sources, are massive, they greatly underscore the true incidence of disease caused by these sources. A complex chain of events must occur in order for a foodborne or waterborne disease outbreak to be reported to the CDC's foodborne and waterborne outbreak surveillance systems. A break at any point in the chain results in an unreported incident. Significant limitations to the reporting system begin at infection, as there is a continuum of disease from asymptomatic infection and mild illness to death. Illness can be sporadic in the population following exposure, and most sickened individuals seek medical attention only in severe cases. Outbreaks that are most likely to be brought to the attention of public health authorities include those that are large, such as interstate or restaurant-associated outbreaks, or those that can cause serious illness, hospitalization, or death. The identification of the source of infection in many cases is difficult and may be compounded by the long incubation periods of some agents, as noted in Table 1 (Section 2: Pathogens). For instance, the illness following exposure to *Brucella* spp. may manifest in as little as five days or as much as 60 days, a time in which the number of potential vehicles of transmission may be massive. Even where cases may be simple, reporting may be limited. Reporting of outbreak data is at the discretion of the states, many of which do not have adequate monitoring and reporting systems in place, primarily due to lack of financial resources to implement such systems. Outbreaks reported in the foodborne and waterborne outbreak surveillance summaries are a small and variable fraction of all outbreaks and cases that occur in the U.S. every year. They do not include those caused by secondary infections, animal contact infections, airborne infections, or many of the other pathways discussed above. As a result, the true incidence of illness that may be caused by zoonotic pathogens remains largely unknown. Table 8 shows the estimated total yearly incidence of disease caused by selected pathogens in the U.S. (Mead *et al.*, 1999). Based on these estimates, zoonotic pathogens may be responsible for as much as 90% of bacterial and parasitic infections of known etiology.

The actual incidence of waterborne and foodborne disease is certainly much higher than that reported in annual surveillance activities. For instance, Mead *et al.*, (1999) estimated that foodborne disease causes 76 million illnesses, 325,000 hospitalizations, and 5,000 deaths in the U.S. each year. The American Society for Microbiology (1998) reported that 900,000 illnesses and 900 deaths each year may be caused by waterborne microbial infections following recreational water contact. Morris and Levin (1996) estimated that disease-causing microbes in drinking water alone may cause 7.66 million illnesses and 1,200 deaths each year. Based on these estimates, acquiring infection by a foodborne organism may be 10-84 times more likely than for waterborne infections (either through recreation or drinking contaminated water). However, actual studies suggest that the risks associated with drinking water that meets federal standards are understated. The reasons for this are unclear, but may be related to the perception that water is "clean". There may be a tendency of individuals and medical practitioners to identify food as a source of contamination when the vehicle of transmission is unclear, especially when the etiological agent is not identified. In any case, evidence from studies of several water

Table 7. Water and foodborne outbreaks in the U.S. reported by the CDC (1991-1997).

Etiologic Agent	**Waterborne** Outbreaks (*Cases*)						**Foodborne** Outbreaks (*Cases*)	**Total** Outbreaks (*Cases*)
	Drinking Water			Recreational Water		Total Waterborne		
	Untreated	Treated	Unknown	Natural	Man-Made			
Bacteria								
Bacillus spp.					1 (*20*)	1 (*20*)	22 (*969*)	23 (*989*)
Brucella spp.							1 (*19*)	1 (19)
Campylobacter spp.	5 (*253*)	1 (*32*)	2 (*274*)		1 (*6*)	9 (*565*)	38 (*773*)	47 (1338)
Clostridium spp.							107 (*4991*)	107 (4991)
E. coli (toxigenic) [†]	4 (*747*)	5 (*90*)	2 (*9*)	32 (*476*)		43 (*1322*)	90 (*3312*)	133 (4634)
Legionella spp.		6 (*80*)			1 (*149*)	7 (*229*)		7 (229)
Leptospira spp.				3 (*402*)		3 (*402*)		3 (402)
Listeria monocytogenes							3 (*100*)	3 (100)
Mycobacteria spp.					1 (*5*)	1 (*5*)		1 (5)
Plesiomonas shigelloides		1 (*60*)				1 (*60*)		1 (60)
Pseudomonas spp.				1 (*50*)	63 (*1090*)	64 (*1140*)		64 (1140)
Salmonella spp.		2 (*749*)	1 (*84*)		1 (*3*)	4 (*836*)	560 (*35861*)	564 (36697)
Shigella spp.	4 (*496*)	4 (*109*)		13 (*1256*)	5 (*120*)	26 (*1981*)	48 (*1671*)	74 (3652)
Staphylococcus spp.					1 (*3*)	1 (*3*)	57 (*1950*)	58 (1953)
Streptococcus spp.							3 (*228*)	3 (228)
Vibrio spp.			2 (*114*)			2 (*114*)	9 (*50*)	11 (164)
Yersinia enterocolitica	1 (*2*)					1 (*2*)	2 (*27*)	3 (29)
Other bacterial							6 (*609*)	6 (609)
Protozoa								
Cryptosporidia spp.	2 (*141*)	8 (*407701*)	3 (*162*)	6 (*654*)	45 (*12494*)	64 (*421152*)		64 (421152)
Giardia spp.	8 (*61*)	16 (*2218*)	1 (*4*)	6 (*85*)	5 (*187*)	36 (*2555*)	7 (*79*)	43 (2634)
Niagleria fowleri	1 (*2*)			29 (*29*)		30 (*31*)		30 (31)
Helminthes								
Schistosoma spp.				11 (*234*)		11 (*234*)		11 (234)
Trichinella spiralis							3 (*60*)	3 (60)
Virus								
Adenovirus 3				1 (*595*)		1 (*595*)		1 (595)
Hepatitis A	2 (*56*)					2 (*56*)	38 (*1262*)	40 (1318)
Norovirus	6 (*882*)	4 (*1804*)	2 (*665*)	8 (*391*)	3 (*60*)	23 (*3802*)	10 (*1483*)	33 (5285)
Uncharacterized		1 (*70*)				1 (*70*)	24 (*2104*)	25 (2174)
AGI and Other Unknown[‡]	21 (*2731*)	34 (*11997*)	6 (*634*)	16 (*1176*)	10 (*268*)	87 (*16806*)	2461 (*51731*)	2548 (68537)

† Includes *E. coli* O157:H7, *E. coli* O121:H19, and *E. coli* O26:NM

‡ AGI= Acute gastrointestinal illness of unknown etiology; also includes other illnesses of unknown etiology

Table 8. Estimated number of total cases, hospitalizations, and fatalities that may occur annually in the U.S. by selected etiological agent as reported by Mead *et al.*, (1999).

Etiologic Agent	Total Cases	Hospitalizations	Fatalities
Bacteria			
Brucella spp.	1,554	122	11
Campylobacter spp.	2,453,926	13,174	124
Escherichia coli O157:H7	73,480	2,168	61
Enterohemorrhagic *Escherichia coli* (non-O157:H7 STEC)	36,740	1,084	30
Listeria monocytogenes	2,518	2,322	504
Salmonella spp.	1,412,498	16,430	582
Yersinia enterocolitica	96,368	1,228	3
Protozoans and Helminthes			
Cryptosporidium parvum	300,000	1,989	66
Giardia lamblia	2,000,000	5,000	10
Toxoplasma gondii	225,000	5,000	750
Trichinella spiralis	52	4	0

systems meeting federal drinking water standards suggests that as much as 6-40% of gastrointestinal illness in the U.S. may be drinking water related (Payment *et al.*, 1991; Golstein *et al.*, 1996; Cottle *et al.*, 1999; Morris *et al.*, 1996; Schwartz *et al.*, 1997; Schwartz *et al.*, 2000; Levin *et al.*, 2002). For instance, in a study conducted in Contra Costa County, California, reverse osmosis drinking water treatment systems (half sham and half real) were installed on the taps of more than 400 participants (50% sham and 50% real). Participants with true systems had 20.4% less gastrointestinal illness episodes than those who used tap water meeting all federal and state drinking water treatment standards (Colford *et al.*, 2002). These results are similar to those of earlier Canadian studies (Payment *et al.*, 1991a,b; Payment, 1994; Payment *et al.*, 1994; Payment *et al.*, 1997), and indicate that infections acquired through contaminated drinking water may approach those acquired through consumption of tainted food.

6.2 Specific Cases

Although the scale of infections caused by zoonotic pathogens remains unclear, the transmission of pathogenic zoonoses from livestock animals to humans and other negative public health outcomes resulting from living in proximity to confinement animals has been clearly documented in reported literature. Both epidemiological studies (See sidelights.) and specific incidences reported in the U.S. and other high income countries, such as the United Kingdom (UK), Canada, The Netherlands, and Japan, have implicated livestock animals and their wastes as the source of illness and other health outcomes. Animal manures in particular have been implicated as the source of pathogens in several waterborne outbreaks (Jackson *et al.*, 1998; Crampin *et al.*, 1999; License *et al.*, 2001; Health Canada; 2001). When manure has been implicated as the source of outbreak, the consequences have been severe. For instance, manure runoff contaminating groundwater near a municipal well in Walkerton, Ontario, Canada resulted in an outbreak of *E. coli* O157:H7 and *Campylobacter* spp. in May, 2000 that caused 2,300 illnesses and 6 deaths (Valcour *et al.*, 2002; Clark *et al.*, 2003; Federal-Provincial-Territorial

Committee on Drinking Water, 2005). Solo-Gabriele and Neumeister (1996) describe a *Cryptosporidium* outbreak in Corrollton, GA in 1989 in which manure runoff was suspected to have been the cause of over 13,000 illnesses. Richardson *et al.*, (1991) and Atherton *et al.*, (1995) describe *Cryptosporidium* outbreaks in Swindon, Oxfordshire, and Bradford UK in 1989 and 1994, respectively, in which storm runoff from farm fields was suspected to have been the cause of 641 illnesses. MacKenzie *et al.*, (1994) describe a *Cryptosporidium* outbreak in Milwaukee, Wisconsin in 1994 in which 87 deaths and over 400,000 illnesses were attributed to animal manure and/or human excrement contaminating the water supply.

Merchant et al. (2005) studied the association between farm living and the prevalence of asthma outcomes. Children living on swine farms were more likely to have asthma outcomes, and the prevalence was more dramatic where antibiotics were added to feed. Nearly 43% of children on farms with less than 500 pigs had asthma or asthma indicators. This number climbed to 46% on farms with more than 500 pigs. However, 55.8% of children living on hog farms where antibiotics were added to feed experienced asthma or asthma indicators . This compared to 26.2% prevalence in children on farms that did not raise hogs. The study indicated that 33.6% of children not living on a farm and not around swine had at least one indicator of asthma. Although farms that use antibiotics tended to be larger, the research team concluded that antibiotic exposure may also have played a role in the development of childhood asthma.

Animal manure has also been implicated as the source of many foodborne outbreaks, mostly resulting from contaminated produce (Schlech *et al.*, 1983; Morgan *et al.*, 1988; Besser *et al.*, 1993; Cieslak et al, 1993; Millard et al, 1994; Tschape *et al.*, 1995). Manure-contaminated produce (fruit and vegetables, including juices and salads) tends to result in more illnesses per outbreak than those associated with contaminated meat. This is because fertilizing fields with manure or irrigating with fecally-contaminated water results in larger numbers of potentially infectious products that are eaten raw in most cases. For instance, Fukushima *et al.*, (1995) describe an outbreak of *E. coli* O157:H7 in Sakai City, Japan in which animal manure-contaminated alfalfa sprouts were suspected of causing 12,680 illnesses, 425 hospitalizations, and 3 deaths. Outbreaks of *E. coli* O157:H7 have also been associated with the consumption of manure-contaminated apple cider (Bresser *et al.*, 1993) and potatoes (Levy *et al.*, 1978). In contrast, contaminated meats, which may result from infected animals or contamination at the abattoir, are generally cooked, destroying pathogens and leading to more sporadic incidence of illness per outbreak. However, the number of outbreaks and total number of cases

Wing and Wolf (2000) surveyed residents of three rural communities, one in the vicinity of a 6000-head hog operation, one in the vicinity of two intensive cattle operations, and a third without livestock operations using liquid waste management systems. Residents in the vicinity of the hog operation were 7.6 times more likely to report occurrences of headaches, 5.2 times more likely to experience runny noses, 3.6 times more likely to have sore throats, 4.7 times more likely to excessively cough, 3.0 times more likely to have bouts of diarrhea, and 5.6 times more likely to have burning eyes than residents of the community without intensive livestock operations. All results were adjusted for sex, age, smoking, and work outside the home.

associated with contaminated meat are higher than produce. According to surveillance of outbreaks in the U.S. between 1990 and 1998, contaminated produce accounted for about 24% of the outbreaks and 41% of the cases (Griffiths, 2000).

Although proper cooking can eliminate most pathogens from meat products, contaminated meat remains a significant link between humans and pathogenic zoonoses. Outbreaks of enterohemmhorhagic *E. coli* in the U.S. between 1982 and 2002 can be attributed primarily to contaminated meat (41%), followed by produce (21%), person to person contact during illness (14%), contaminated drinking or recreational water (9%), and directly contacting infected animals or their wastes (3%) (Rangel *et al.*, 2005). The emergence of many antimicrobial-resistant zoonotic pathogens in human populations has been linked to the consumption of food animals and dairy products. For instance, Holmberg *et al.*, (1984) attributed a 6-state outbreak of multi-drug resistant *Salmonella newport* to consumption of beef from a feedlot that was using subtherapeutic doses of chlorotetracycline as a growth promoter. The emergence of multidrug-resistant *Salmonella typhimurium* DT 104 in 1988 in cattle in the UK was rapidly followed by its detection in meat (Threlfall *et al.*, 1997) and later in humans, presumably via the consumption of contaminated beef, pork sausages, and chickens. Between 1990 and 1995, human illnesses in UK attributed to *S. typhimurium* DT 104 increased from 259 to 3837 (Lee *et al.*, 1994). The emergence of fluoroquinolone-resistant pathogens in the Netherlands rapidly followed its introduction as veterinary drug in chickens and humans. Enrofloxacin-resistant *Campylobacter* in poultry increased from 0-14% between 1982 and 1989, while resistant *Campylobacter* causing human infections rose from 0-11% (Endtz, 1991). As poultry are a primary reservoir for *Campylobacter* spp., the use of fluoroquinolones in the poultry industry was implicated as the vehicle for human-acquired enrofloxacin-resistant *Campylobacter*.

Airborne zoonotic pathogens from animal feeding operations may also infect humans and other livestock animals. As noted above, pathogens and antimicrobial-resistant bacteria have been detected at elevated concentrations in confinement house air (Cormier *et al.*, 1990; Cazwala *et al.*, 1990; Crook *et al.*, 1991; Heederick *et al.*, 1991; Zahn *et al.*, 2001; Predicala *et al.*, 2002; Gast et al 2004; Chapin *et al.*, 2005). Farm workers exposed to confinement house air are much more likely than the general population to acquire infections of the lungs and sinuses (Mackiewicz, 1998; Aubry-Damon, 2004; Armand-LeFevre *et al.*, 2005), and the potential for secondary infection of nearby populations is high. Airborne zoonotic pathogens may also travel over vast distances downwind of an infected livestock source. Henderson (1969) and

Smith et al. (1993) and Hawker et al. (1998) describe an outbreak in the West Midlands, UK in which airborne transmission of *Coxiella burnetii* was identified as the causative agent of 147 illnesses. Outdoor lambing and calving was performed on farms south of the urban area. Strong gales blew towards the urban area on a single day approximately three weeks prior to the onset of illness. *Coxiella burnetii* is known to multiply to very high concentrations in the placenta of sheep which, following deposition on the ground during outdoor birthing, can dry out allowing bacterial release with airborne particulates (Welsh et al., 1958; Jones and Harrison, 2004). The mean incubation period for *Coxiella burnetii* in humans has been shown to be 20 days, consistent with the period of time between the day of strong gales and the peak onset of symptoms in the outbreak (Aitken et al., 1987).

Hughes and Wright (1970) describe a series of airborne picnoravirus outbreaks (foot and mouth disease) in pigs, cattle, and sheep in Worcester, UK in 1967, in which infected animals at three pig farms were suspected as the cause. Casal *et al.*, (1997) estimated that the airborne dispersion could have transported an infectious dose of this virus from the three source swine farms to cattle as far as 7 km away. However, secondary infection from cattle or sheep was unlikely to affect cattle or sheep more than 200 m away (Donaldson *et al.*, 2002). It has been estimated that picnoravirus can be transported in the air over distances as great as 60 km overland and 300 km over seas (Gloster *et al.*, 1982; OIE, 2005). Lyytikainen *et al.*, (1998) describe an outbreak of Q-fever in a small rural community in Germany in which airborne transmission of *Coxiella burnetii* from a nearby infected flock of 1,000-2,000 sheep may have caused 45 illnesses over a four month period. Outdoor calving was performed on the farm, and the wind blew from the farm towards the town 57% of the time during the course of the outbreak. Both picnoraviruses and *Coxiella burnetii* may be shed in animal feces suggesting that these organisms, among others, could be dispersed over vast distances following spray irrigation of animal manures onto croplands.

6.3 Antimicrobial Resistance

Antimicrobial-resistant bacteria and other zoonotic pathogens from CAFOs often infect humans, many times with serious consequences. Evidence in the reported literature overwhelmingly supports this conclusion and includes direct epidemiological studies, temporal evidence of the emergence of resistance in livestock animal populations prior to the emergence in human populations, trends in resistance among human isolates that mimic the use of antimicrobials in livestock animals, and studies that show farmers, slaughterers, and their family members are much more likely than the general population to acquire antimicrobial zoonoses. Antimicrobial resistance can limit treatment options in sickened individuals and increase the number, severity, and duration of infections (FAAIR Scientific Advisory Council, 2002). Varma *et al.*, (2005) evaluated *Salmonella* outbreaks in the U.S., and determined that among 32 reported outbreaks between 1984 and 2002, 22% of 13,286 people in ten *Salmonella*-resistant outbreaks were hospitalized compared with 8% of 2,194 people in 22 outbreaks caused by pansusceptible strains. These differences are not only the consequence of limited options for antimicrobials, but are also related to the increased virulence often associated with antimicrobial-resistant organisms. For instance, Lee *et al.*, (1994) determined that individuals infected with resistant organisms were ill 25% longer and were significantly more likely to be hospitalized than those infected with pansusceptible strains. Those

> Bezanson et al (1983) describe the infection of a newborn child with a multidrug-resistant strain of *Salmonella* ser. typhimurium resulting in septicemia and meningitis. The source of infection was the child's asymptomatic mother, who acquired the bacterium through ingestion of unpasteurized milk and passed it to her child during delivery in the hospital. Illness in the newborn child manifested within 24 hours, and within 72-96 hours had spread to several other infants in the hospital nursery. In another case described by Lyon et al. (1980), a multidrug-resistant strain of *Salmonella heidelberg* was spread from an asymptomatic mother to newborn child during delivery via cesarean section. The mother was a farmer who shortly before delivery had been working with calves from an infected herd. Three infants in the hospital nursery were infected with the organism and developed bloody diarrhea.

infected with resistant strains were hospitalized on average ten days versus eight days for those infected with susceptible strains, even though most subjects in both cases were treated with an antimicrobial to which the infectious agent was susceptible. The difference in hospitalization rates likely reflects the higher virulence of the resistant infectious organism, and, to a much lesser extent, an inappropriate first choice of antimicrobial for treatment. Resistance to antimicrobial agents, resulting from their extensive use in animal agriculture, may result in tens of thousands of additional infections by zoonotic pathogens compared to what would be experienced with pansusceptible strains. This may result in more than ten thousand additional days of hospitalization, and hundreds of thousands of excess days of diarrhea in the U.S. each year (Barza and Travers, 2002; Travers and Barza, 2002)

6.4 Hydrologic Events

Hydrologic events ranging from mild rainfall to flooding can increase the movement of pathogens from CAFOs or manure-amended fields to waters that are likely to come into contact with people. Serious public health consequences of the increased pathogen load, especially during flooding events, are common (Isaacson *et al.*, 1993; MacKenzie *et al.*, 1994; Health Canada, 2000; CDC, 1998). Several studies in low income countries have reported increases in morbidity and/or mortality following flood events due to cholera, cryptosporidiosis, nonspecific diarrhea, poliomyelitis, rotavirus, and typhoid and paratyphoid (Fun *et al.*, 1991; van Middelkoop *et al.*, 1992; Katsumata *et al.*, 1998; Biswas *et al.*, 1999; Sur *et al.*, 2000; Mondal *et al.*, 2001; Kunji *et al.*, 2002; Kondo *et al.*, 2002; Heller *et al.*, 2003; Vollard *et al.*, 2004). The increased relative risk (RR) or odds ratio (OR) of contracting disease during flooding in these cases ranged from 1.39 to 4.52. Significant increases in vector- and rodent-borne diseases were also observed (Trevejo *et al.*, 1998; Han *et al.*, 1999; Sanders *et al.*, 1999; Sarkar *et al.*, 2002; Leal-Castellanos *et al.*, 2003).

The increased risk of contracting disease post-flood in high income countries such as the U.S., UK, and Australia exist, but are less pronounced (Ahern *et al.*, 2005). Bennet *et al.*, (1976) observed hospital visits by the flooded to more than double in the year following an event in Bristol, UK, in 1968. These researchers also observed a 50% increase in mortality among the flooded, mostly in the elderly. Reacher *et al.*, (2004) interviewed 467 households following a flood in Lewes, UK, and observed a slight increase in gastrointestinal illness in those whose homes were flooded. In Brisbane, Australia (1974), a flood led to increased morbidity, but not mortality, in the flooded group (Price, 1978; Abrahams *et al.*, 1976). However, Handmer and Smith (1983) noticed no flood-related increase in hospital admissions during flooding in Lismore, Australia the same year. In a cohort study of 1,110 people in a U.S. Midwestern community, Wade *et al.*, (2004) reported an increase in the incidence of gastrointestinal illness during a flood event in April and May of 2001. The increase in gastrointestinal illness was pronounced in persons with potential sensitivity to infectious gastrointestinal agents and those who came into contact with the flood water, especially children. Heather *et al.*, (2004) noted the significance of heavy rainfall in the Walkerton, Canada outbreak of *E. coli* O157:H7 and *Campylobacter*. The rainfall was equivalent to a 60-year event, and it was suggested that this extreme precipitation may have mobilized animal wastes and led to the outbreak. Curriero *et al.*, (2001) studied the link between reported waterborne disease outbreaks in the U.S. between 1948 and 1994 and extreme precipitation events. These authors found a strong correlation between rainfall and disease. Disease due to surface water contamination primarily occurred during the month of the precipitation event, whereas disease associated with groundwater contamination occurred two months following extreme precipitation events.

Serious health outcomes from flooding events can and do occur in the U.S. and may be unfairly weighted against the underprivileged. The recent flooding of New Orleans, Louisiana following hurricane Katrina, a category 4 event, resulted in the exposure of tens of thousands of people to floodwaters laden not only with chemical wastes, but also decomposing bodies, animal carcasses, sewage, and animal wastes. *E. coli* concentrations in these waters reached as high as 42,000 per 100 mL, hundreds of times higher than levels associated with gastrointestinal illnesses that result from “recreational contact”. Those unable to escape the city prior to the hurricane were primarily the underprivileged, and illness was exacerbated by the lack of availability of medical provisions and personnel. The eye of the hurricane traveled through Mississippi, the fourth largest poultry-producing state in the U.S., with the highest rainfall amounts (ranging between 12.5-22.5 centimeters, falling at a rate of 1-2 cm per hour) tracking over the central and northeastern portions of the state. As can be seen in Figure 4, these regions are associated with the bulk of large concentrated swine feeding operations in the state of Mississippi. The pollution from these operations has been previously reported to disproportionately affect impoverished and African-American peoples (Wilson *et al.*, 2002). The full breadth of public health outcomes from this hurricane, as well as potential environmental injustice resulting from the flooding, have yet to be fully understood. These events, however, signify the need for water quality officials to seriously consider precipitation events during planning.

6.5 Economic considerations

Infection by zoonotic pathogens results not only in extensive human suffering, but also significant economic loss. For instance, the Milwaukee outbreak of Cryptosporidiosis in 1993 cost the community as much as 96.2 million; 31.7 million in medical costs and 64.6 million in lost productivity (ASM, 1998; Corso *et al.*, 2003; Water Health Connection, 2005). The Walkerton, Ontario outbreak of *E. coli* O157:H7 and Campylobacteriosis in 2002, with 2300 cases and 6 deaths, cost the community an estimated 40 million in lost productivity, medicine, and hospitalization costs. The American Society for Microbiology estimates that even a mild case of diarrhea may cost $330 in lost work productivity and over-the-counter medicines (adjusted to 2005 dollars). More severe cases were estimated to cost up to $9,500 per person for medical diagnosis and treatment. Considering the number of illnesses that may be experienced in the U.S. each year, foodborne illnesses may amount to three billion dollars per year due to hospitalization and more than 20 billion in lost work productivity and over-the-counter medicines, a significant portion of which may be due to transmission of disease from livestock animals. Similarly, waterborne illnesses may result in a total of two to twenty billion dollars in costs annually (Garthright *et al.*, 1988; Hardy *et al.*, 1994; Gerba, 1996; Liddle *et al.*, 1997; Fleisher *et al.*, 1998; Scott *et al.*, 2000; Dwight *et al.*, 2001; Fruhwirth *et al.*, 2001; Corso *et al.*, 2003). Considering the annual healthcare costs of managing antimicrobial resistance, which may be in the range of 4-30 billion dollars (Khachatourians, 1998; American Academy of Microbiology (AAM), 1999; Montague, 2000), the annual costs associated with illness caused by

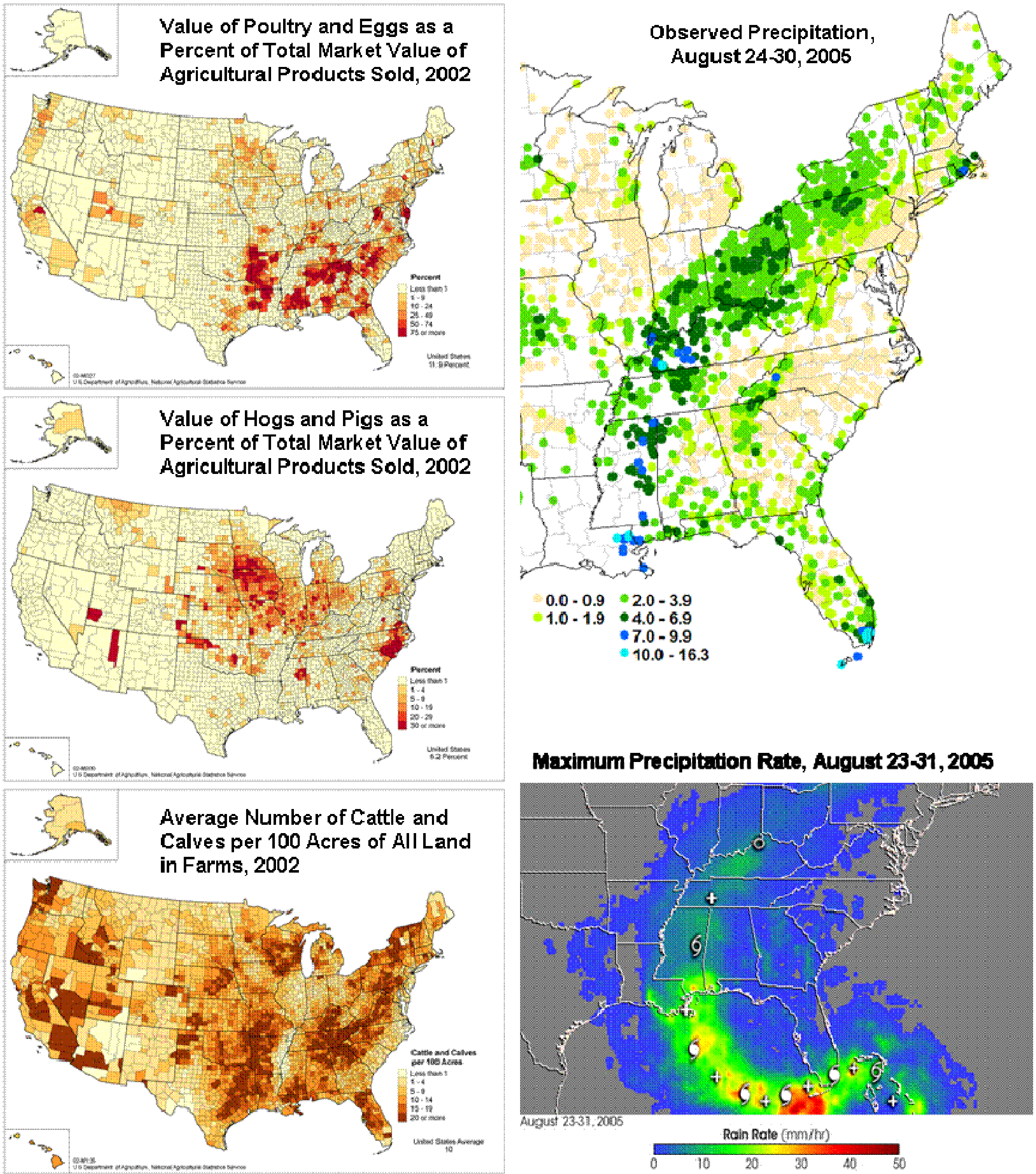

Figure 4. Distribution of livestock animals in regions impacted by Hurricane Katrina, August, 2005 (adapted from USDA, 2002; National Oceanic and Atmospheric Administration, 2005; National Aeronautics and Space Administration, 2005).

zoonotic pathogens and antimicrobial resistant bacteria from livestock operations may be staggering.

Significant economic losses may also be incurred by the closing of beaches when waters cannot meet USEPA recreational water guidelines. Of the thousands of beach closings every year, more than 80 percent are due to excessive levels of bacteria (ASM, 1998). A beach closing due to bacteria indicates that levels were excessive the day prior to the closing, during which time thousands of individuals may have been exposed to contaminated water. Dwight *et al.*, (2001) estimated the economic burden from illness associated with recreational coastal water pollution at Newport and Huntington Beaches, Orange County, California alone to be 3.3 million per year. Considering the thousands of beaches closed every year, economic losses may be in the billions of dollars.

6.6 Discussion

Both waterborne outbreaks and those associated with fresh produce have been on the rise in recent decades and will likely continue to increase as surveillance is improved. Although the source of contamination in many of these outbreaks remain unreported, poor manure management in livestock operations most assuredly plays a significant role as alternative sources of contamination are limited in scale compared to manure applications and typically much less infectious in character. The annual costs of infectious zoonotic diseases in the U.S. may reach into the tens of billions of dollars considering both food and waterborne illnesses. These estimates exclude such costs as death, pain and suffering, lost leisure time, financial losses to food establishments, legal expenses, and long-term health outcomes due to infections that may result in degenerative diseases or cancer. The economic burden of pathogenic zoonoses has been shifted from corporate farms who fail to use appropriate manure management at the source of disease to sickened individuals and businesses that experience decreased revenues due to beach closures and lost productivity. Economic burdens of CAFO pollution may be especially shouldered unfairly by minority groups and the poor, as evidenced by recent works describing environmental injustice surrounding the swine industries of North Carolina and Mississippi (Wing *et al.*, 2000; Wilson *et al.*, 2002). It is at present unclear how new molecular microbiological technologies such as microbial source tracking will affect litigation and potential liability of CAFO operators in future disease outbreaks.

7. Emerging Technologies: Monitoring Pathogens in the Environment

Concentrated animal feeding operations may release pathogens into the environment through a variety of mechanisms that may result in extensive human suffering and economic loss. Current surveillance activities may be inadequate to identify fully the scope of problems surrounding the release of overt pathogens from CAFOs to the environment. For instance, surface water quality surveillance in the U.S. relies on the quantitative detection of bacterial indicators of fecal pollution including *E. coli* and enterococci rather than direct identification of selected etiologic agents associated with disease in humans. Although related to gastrointestinal illness following recreational water contact, these indicators may not be reliable surrogates for all bacterial pathogens and most parasites and viruses. Human illness can occur even when the concentrations of *E. coli* and enterococci indicate that bathing waters are safe. The use of microbial indicators as surrogates for pathogens continues because infectious concentrations of pathogens in waters may be low and difficult to detect, and standard methods for analysis do not exist for many pathogens.

The February 28, 2005 ruling by the 2nd U.S. Circuit Court of Appeals required the USEPA to identify and characterize the performance of animal waste management practices and barrier technologies that specifically address contamination of the nation's waters by pathogens emanating from CAFOs. Transport properties and the virulence of various pathogens vary to a wide degree, and may be poorly represented by the bacterial indicators *E. coli* and enterococci (Ferguson *et al.*, 2003). Thus, this requirement signals the need for new and improved pathogen detection technologies. New approaches to water quality monitoring and emerging technologies enabling the identification of overtly pathogenic agents in natural waters and their source will greatly improve human health and welfare and increase the biosecurity of our natural resources.

Considering the many potential exposure routes following release of pathogenic zoonoses from CAFO facilities, identification of the risks associated with pathogens emanating from concentrated animal feeding operations may require technologies that enable the measurement of overt pathogens in air, drinking and recreational water, meat and produce, soil and sediments, and feces of various animals among others. For foods, standard methods for analysis for a small number of zoonotic pathogens already exist (FDA, 2005). For other matrices, such as soil and sediments, drinking water, and natural waters, methods are lacking or have not been standardized. Methods reported in literature include classical cultivation approaches, and more recently, identification and quantification of agents via the detection of surface antigens or nucleic acids. In either case, the detection of zoonotic pathogens with infectious doses as low as a few ingested or respired particles presents specific challenges including concentration of large environmental samples, removal of inhibitory compounds from sample concentrates, detection of viability, detection of multiple agents in a limited sample, and long analysis periods, especially for cultivation-based approaches. Considerable advances in emerging nucleic acids and sensor technologies are reducing analysis times from weeks to hours in some instances, but present trade-offs in terms of the costs and technical expertise required to apply the technologies. The intent of this review is not to provide a complete description of all of the emerging detection technologies and report their application, but rather to identify their strengths and limitations and discuss the challenges posed by their use.

7.1 Sample Processing

Appropriate sample processing is critical to detecting pathogenic agents at concentrations relevant to their infectious dose in environmental matrices. For some media that may contain high numbers of pathogens but are extremely heterogeneous such as animal manure, obtaining an appropriate sample may hinge on careful compositing procedures. For instance, Pearce *et al.*, (2004) examined the distribution of *E. coli* O157 in bovine fecal pats and determined that the density of O157 in the pats was highly variable, differing by as much as 76,800 CFU/g between samples of the same fecal pat. These researchers determined that most positive samples bordered the detection limit, and that testing of only 1 g per pat (as is commonly performed) may result in as much as 20-50% false-negatives. For other media such as air, sample processing may need to be particularly careful regarding stressing organisms in the sampling device.

The low infectious dose associated with many etiological agents leads to the need to concentrate copious quantities of air, food, or water into smaller volumes amenable to detection with classical cultivation or the newer molecular microbial methods. Several mechanisms have been used to concentrate these agents to detectable numbers including filtration, immunocapture, and enrichment. However, some of these methods may concentrate inhibitory and/or interfering compounds with the agent of interest, while others require additional analysis time or alter the sample from its initial state. Thus, the use of any of these mechanisms may present tradeoffs in downstream analysis, potentially affecting analytical detection limits, analysis times, or the number of agents that can be detected from a single sample. At present, standard methods for the concentration of viruses from water samples rely on electrostatic capture from 100 or more liters of water onto positively-charged filters followed by elution, precipitation, and resuspension in a small volume of sodium phosphate buffer (USEPA, 1993). Concentration of the protozoan parasites *Giardia* and *Cryptosporidium* require filtration of ten or more liters of water through a depth filter followed by elution, centrifugation, and immunomagnetic separation (USEPA, 2001). Concentration of bacterial pathogens is usually performed by membrane filtration, although turbidity of water can severely inhibit the volume of water that can be passed through the filter.

None of the accepted concentration techniques listed above is applicable to all of the various groups of etiologic agents (viruses, bacteria, protozoans). The detection of several agents may require the collection of multiple large volume samples from a single location and concentration by a number of techniques. To overcome these limitations, newer methods for sample concentration applicable to all classes of etiologic agents have been proposed. Most notably, hollow-fiber ultrafiltration has been used to simultaneously concentrate viruses, bacteria, and protozoan parasites from water samples as large as 100 L to volumes as low as 250 mL with recovery efficiencies on the order of 20-92% (Juliano and Sobsey, 1997; Kuhn and Oshima, 2001; Olsezewski *et al.*, 2001; Evans-Strickfaden *et al.*, 1996; Simmons *et al.*, 2001; Morales-Morales *et al.*, 2003; Ferguson *et al.*, 2004). Subsequent analyses with small portions of a single eluent can lead to detection of several pathogens at environmentally-relevant concentrations (Olsezewski *et al.*, 2001; Morales-Morales *et al.*, 2003). Hollow-fiber ultrafiltration may also have the added benefit of allowing small or water soluble inhibitors of nucleic acids techniques to pass into the permeate, rather than co-concentrate with the pathogens in the retentate, prior to sample analysis (Wilson, 1997).

7.2 Conventional Cultivation and Nucleic Acids Approaches

Conventional cultivation methods for the detection of bacterial pathogens usually require several steps including (1) sampling and release of bacteria from the environmental matrix, (2) pre-enrichment in non-selective broth to allow small numbers of stressed bacterial pathogens to recover and grow prior to applying further environmental stress in selective broths, (3) transfer to selective broth to enrich low numbers of pathogens and reduce competitor bacteria, (4) inoculation of a selective solid medium to identify presumptive positive colonies, and (5) biochemical and/or serological confirmation of presumptive-positive colonies. Most probable number (MPN) techniques can be used to arrive at a quantitative result (USEPA, 2005). Depending on the number of steps required; confirmation of the presence of specific pathogens by conventional methods may take as few as two days to two weeks or more.

Nucleic acid technologies follow the same framework for detection as cultivation methods, but detection or quantitation can occur prior to or following pre-enrichment in nonselective media or further enrichment in selective broths. Nucleic acid technologies are also commonly used in lieu of biochemical and/or serological confirmation for presumptive colonies, or to acquire more detailed genomic information on bacterial isolates, such as possession of antimicrobial-resistance or virulence traits. Enrichment broths (both selective and nonselective) for nucleic acids techniques are used not only to increase the numbers of pathogens, thereby improving detection, but also to dilute potential inhibitors of the polymerase chain reaction (PCR). Immunomagnetic separations have also been used to separate etiological agents from large volumes and/or samples with inhibitory agents prior to or following enrichment steps.

The detection limit of nucleic acid assays is usually dependent on the amount of time available for analysis. In general, if the etiological agent of interest is in high concentration or the medium is relatively clean (such as drinking water), short analysis times of less than 6-8 hours can be realized, as inhibitors may be in low concentration relative to pathogens in the original sample retentate. In more turbid samples with low numbers of infectious agents, such as stream waters, analysis times can extend from a day to as much as four days. Extensive sample processing and selective enrichments may be required to achieve detection limits relevant to the infectious dose. Tables 9 and 10 list several studies that have used nucleic acids techniques for the detection of etiological agents.

As can be seen in Table 9, nucleic acids techniques may provide very sensitive detection of selected etiological agents in clean samples, such as air or drinking water, even without enrichment. For turbid environmental samples, such as surface water, feces, or soils, nucleic acid technologies may have detection limits several orders of magnitude higher than cultivation techniques (See Table 9). This is because PCR inhibitors, such as humic substances associated with many environmental samples, are co-extracted with the etiological agent prior to detection. Even with inhibitors present, nucleic acid technologies may occasionally yield more sensitive results than cultivation-based techniques. For instance, Inglis and Kalischuk (2004) used nested real-time SYBR Green-PCR to quantify *Campylobacter lanienae* in cattle feces without enrichment. These researchers were able to detect *C. lanienae* at concentrations as low as 250 CFU/g feces in less than 4 hours, a level more sensitive than could be achieved with cultivation-based methods that required 2 days for presumptive results. However, in most instances,

Table 9. Sample times and detection limits of several nucleic acids-based techniques for detecting pathogens in different matrices without enrichment.

Etiologic Agent	Matrix	Time *hours*	Detection Limit	Reference
No Enrichment				
Anthrax Spores	Air	1-2	1 spore/100 L	Makino and Chuen (2003)
E. coli	Drinking Water	5	1 CFU/L	Abd El-Haleem *et al.*, (2003)
Salmonella	Drinking Water	5	1 CFU/L	Abd El-Haleem *et al.*, (2003)
E. coli O157:H7	Ground Water	3	20,000 CFU/100 mL	Vaughn *et al.*, (2003)
Hepatitis A	Sewage effluent	<24	1,000,000 PFU/mL	Jean *et al.*, (2001)
Rotavirus	Sewage effluent	<24	3,000 PFU/mL	Jean *et al.*, (2002)
Hepatitis A	Produce	<24	1,000,000 PFU/surface	Jean *et al.*, (2001)
Campylobacter spp.	Meat †	<24	<100,000 CFU/10 g	Uyttendale *et al.*, (1995-1997)
Listeria monocytogenes	Meat	4	100 CFU/g	Rodríguez-Lázaro *et al.*, (2004)
E. coli O157:H7	Cattle Feces	4	26,000 CFU/g	Ibekwe and Grieve (2003)
Campylobacter jejuni	Cattle Feces	4	3,000 CFU/g	Inglis and Kalischuk (2004)
Campylobacter lanienae	Cattle Feces	4	250 CFU/g	Inglis and Kalischuk (2004)
Clostridium difficile	Human feces	1	50,000 CFU/g	Bélanger *et al.*, (2003)
Salmonella spp.	Biosolids	24	10^6 CFU/g	Burtscher and Wuertz (2003)
Staphylococcus aureus	Biosolids	28	10^6 CFU/g	Burtscher and Wuertz (2003)
E. coli O157:H7	Soil	4	26,000 CFU/g	Ibekwe and Grieve (2003)
E. coli O157:H7	Soil	4	35,000 CFU/g	Ibekwe *et al.*, (2002)
No Enrichment, Immunomagnetic Separation				
Cryptosporidium parvum	Clean Water	<24	50 Oocysts/100 L	Baeumner *et al.*, (2001)
Cryptosporidium parvum	Turbid Water	<24	50 Oocysts/100 L	Baeumner *et al.*, (2001)
Hepatitis A	Ground Water	6-12	20 PFU/20 mL	Abd El Galil *et al.*, (2004)
Enterohemorrhagic *E. coli*	Chicken Rinsate	<24	55 CFU/mL	Call *et al.*, (2001b)
Campylobacter jejuni	Chicken Feces	6	230 CFU/g	Rudi *et al.*, (2004)
Campylobacter jejuni	Chicken Ceces	6	2,000 CFU/g	Rudi *et al.*, (2004)

† Meat = beef, poultry, or pork

improved sensitivity over cultivation-based techniques will not be realized without further sample processing.

Immunomagnetic separation methods (IMS) have been used successfully in several studies to concentrate etiological agents prior to or following sample concentration. These methods use paramagnetic particles coated with antibodies specific to the pathogen of interest to bind the agent and remove it from the sample matrix in a concentrated form via magnetic attraction of the pathogen-paramagnetic particle complex (Campbell and Smith, 1997; Bukhari *et al.*, 1998; Rochelle *et al.*, 1999). The separation of the agent of interest from the environmental sample reduces the presence of inhibitory substances improving detection by PCR. For instance, Abd El Galil *et al.*, (2004) developed a protocol for using combined Immunomagnetic Separation-Molecular Beacon-Reverse Transcription-PCR to detect Hepatitis A virus in groundwater samples. These authors concentrated 100 liters of groundwater using an electropositive microporous (1-MDS) filter followed by elution in beef extract, centrifugation, and resuspension of the pellet in 20 mL sodium phosphate buffer (final pH=7.4). Immunomagnetic separation with two-hour incubation was used to recover the virus followed by extraction of viral RNA, reverse transcription, and real-time PCR with a molecular beacon probe. As few as 20 plaque-forming units (PFU) per 20 mL groundwater concentrate were recovered by these methods. Immunomagnetic separation methods may be attractive where selective enrichment cannot be used effectively.

Selected amplification facilitators or specific DNA treatments may also be used to reduce (but not eliminate) the effects of inhibitory compounds on PCR (Satoh *et al.*, 1998; Abu Al-Soud and Rådström, 2000; Böddinghaus *et al.*, 2001). Common facilitators may include bovine serum albumin (BSA), the single-stranded DNA-binding T4 gene 32 protein (gp32), betaine, and several proteinase inhibitors, all of which work to a varying degree depending on sample type (Abu Al-Soud and Rådström, 2000). Of the facilitors used, BSA is the most common, and tends to work with samples from a wide variety of origins, including blood, human and animal feces, surface and ground waters, soils and sediments, and meat. For instance, Rudi *et al.*, (2004) applied integrated cell concentration and DNA purification using immunomagnetic beads with real-time (TaqMan) PCR to detect and quantify *Campylobacter jejuni* in chicken fecal samples. These researchers could detect as few as 1,000-10,000 CFU/g feces in untreated samples, but with 0.4% BSA in the reaction mix, could reduce PCR inhibition caused by the fecal extract, so that they could reduce their detection limit to as low as 230-2,300 CFU/g feces. Since the mode of action of many facilitators is similar (removal of inhibitors), their benefits are not additive (Abu Al-Soud and Rådström, 2000).

To improve detection of bacterial pathogens in difficult matrices, especially for particularly infectious agents that may be of interest at very low concentrations, enrichments can be used to revive stressed bacteria and increase their numbers prior to detection. Enrichments have the added benefit of diluting PCR inhibitors prior to detection by nucleic acid techniques, resulting in lower detection limits than can be realized by direct sampling (See Table 10). The enrichment of bacterial pathogens presents a trade-off: selective media may enrich one pathogen at the expense of other agents of concern. Several enrichment broths may be required to detect several agents. However, in some cases, nonselective broths may improve the detection of many bacterial pathogens following a single enrichment. For instance, Nam *et al.*, (2004) evaluated the use of universal pre-enrichment broth (UPB) versus selective enrichment broths [lactose broth, modified trypticase soy broth (plus novobiocin), and *Listeria* enrichment broth] for detection of *Salmonella* spp., *E. coli* O157:H7, and *Listeria monocytogenes* from dairy fecal slurry, lagoon water, drinking water, silage/feed, trapped rats, bird droppings, calf fecal swabs, milking parlor floor swabs, bulk tank milk, and in-line milk filters. These researchers observed no differences between growth in UPB and selective media using pure cultures of the three pathogens, either individually or mixed. However, slightly better recovery of pathogens from environmental samples was observed when UPB was used for the initial enrichment step, and transfers were made from the single pre-enriched sample to selective media. UPB supported the growth of all three pathogens to levels detectable by culture techniques within 24 hours from the different environmental matrices.

Table 10. Sample times and detection limits of several nucleic acids-based techniques for detecting pathogens in different matrices following enrichment.

Etiologic Agent	Matrix	Time *hours*	Detection Limit	Reference
Nonselective Enrichment				
E. coli O157:H7	Drinking Water	<24	100 CFU/100 mL	Campbell *et al.*, (2001)
E. coli O157:H7	Surface water	<24	600 CFU/100 mL	Campbell *et al.*, (2001)
E. coli O157:H7	Apple Juice	15	100 CFU/100 mL	Fortin *et al.*, (2001)
Salmonella spp.	Produce	20	4 CFU/25 g	Liming and Bhagwat (2004)
E. coli O157:H7	Milk	10	100 CFU/100 mL	Fortin *et al.*, (2001)
Salmonella spp.	Milk	24	Equal to cultivation §	Malorny *et al.*, (2004)
Salmonella enteritidis	Egg	<48	<10 CFU/25 g	Cook *et al.*, (2002)
Salmonella typhimurium	Oysters	<24	100 CFU/g	Lee *et al.*, (2003)
Vibrio spp.	Oysters	<24	100 CFU/g	Lee *et al.*, (2003)
Salmonella spp.	Chicken Rinsate	24	Equal to cultivation	Malorny *et al.*, (2004)
E. coli O157:H7	Meat †	8-10	580 CFU/g	Sharma *et al.*, (1999)
Salmonella spp.	Meat	<24	1500 CFU/25 g	Cheung *et al.*, (2004)
Salmonella spp.	Meat	24	Equal to cultivation	Malorny *et al.*, (2004)
E. coli O157:H7	Cattle Feces	8-10	1200 CFU/g	Sharma *et al.*, (1999)
Salmonella spp.	Biosolids	24	10 CFU/g	Burtscher and Wuertz (2003)
Staphylococcus aureus	Biosolids	28	10 CFU/g	Burtscher and Wuertz (2003)
E. coli O157:H7	Soil	10	10000 CFU/g	Campbell *et al.*, (2001)
E. coli O157:H7	Soil	14	6 CFU/g	Campbell *et al.*, (2001)
E. coli O157:H7	Soil	24	2 CFU/g	Campbell *et al.*, (2001)
E. coli O157:H7	Soil	<24	<10 CFU/g	Ibekwe and Grieve (2003)
E. coli O157:H7	Soil	<24	<10 CFU/g	Ibekwe *et al.*, (2002)
Selective Enrichment				
Campylobacter jejuni	Surface Water	72	Equal to cultivation	Sails *et al.*, (2002)
Campylobacter coli	Surface Water	72	Equal to cultivation	Sails *et al.*, (2002)
E. coli O157:H7	Surface Water	<48	120 CFU/100 mL	Müller *et al.*, (2003)
Listeria monocytogenes	Produce	<48	<10 CFU/10 g	Uyttendale *et al.*, (1995-1997)
Listeria monocytogenes	Dairy Products	<48	<10 CFU/10 g	Uyttendale *et al.*, (1995-1997)
Listeria monocytogenes	Dairy Products	<72	<10 CFU/60 g	Blais *et al.*, (2001)
Salmonella spp.	Milk	24	Better than cultivation §	Kessel *et al.*, (2003)
Listeria monocytogenes	Eggs	<72	<10 CFU/60 g	Blais *et al.*, (2001)
Vibrio parahaemolyticus	Oyster	24	Better than cultivation	Blackstone *et al.*, (2003)
Campylobacter spp.	Meat	<48	10 CFU/10 g	Uyttendale *et al.*, (1995-1997)
Listeria monocytogenes	Meat	<48	<10 CFU/10 g	Uyttendale *et al.*, (1995-1997)
E. coli O157:H7	Sewage Sludge	<48	120 CFU/100 mL	Müller *et al.*, (2003)
Listeria monocytogenes	Biosolids	28	10 CFU/g	Burtscher and Wuertz (2003)
Listeria monocytogenes	Biosolids	28	10 CFU/g	Burtscher and Wuertz (2003)
Yersinia enterocolitica	Biosolids	28	10 CFU/g	Burtscher and Wuertz (2003)
Salmonella spp.	Biosolids	24	10 CFU/g	Burtscher and Wuertz (2003)

† Meat = beef, poultry, or pork

§ Nucleic acid techniques provided a result equivalent to or better than cultivation methods based on trials in actual samples

Several studies have documented improvements in detection of bacterial pathogens using nucleic acids techniques that can be realized using enrichments in both non-selective and selective broths. Ibekwe and Grieve (2003) used real-time TaqMan PCR to with a detection limit of 26,000 CFU *E. coli* O157:H7 per gram of soil without enrichment. Using a 16 hour pre-enrichment in modified Luria-Bertani broth (containing vancomycin, cefoxime, and cefsulodin); these researchers could reduce their detection limits to less than 10 CFU per gram soil. Burtscher and Wuertz (2003) evaluated the use of PCR for the detection of *Salmonella* spp. *Listeria monocytogenes*, *Yersinia enterocolitica*, and *Staphylococcus aureus* in biosolids from anaerobic digesters and aerobic composters following one- and two- step enrichment (24-48 hours) in selective broths. These researchers were able to detect less than 10 CFU per gram of waste for each organism following enrichment, versus 10^6-10^7 CFU per gram of waste without enrichment.

Nucleic acids techniques may also be useful as a surrogate for biochemical confirmation or for genotyping environmental isolates following detection with cultivation techniques, potentially saving days in analysis time. Müller *et al.*, (2003) tested sewage sludges and river waters for *E. coli* O157:H7 using combined cultivation and nucleic acids techniques. These researchers filtered 100 mL river water samples through 0.45 µm nitrocellulose membranes then enriched the retentate in peptone-saline water (PSW) supplemented with vancomycin-cefixime-cefsulodin solution for six hours. Similarly, 100 µL sewage sludge was enriched for six hours directly in the antibiotic-PSW solution. Immunomagnetic separation was used to isolate *E. coli* O157:H7 from a small portion of the enrichment media, and the paramagnetic bead-bacteria complexes were further enriched on selective media for 24 hours. Suspect colonies were investigated with PCR targeting genes associated with Shiga-like toxins 1 and 2, attachment and effacement, and enterohaemolysin. With these methods, the researchers could detect as few as 120 CFU/100 mL in sewage sludge and river samples in less than 48 hours.

Several researchers have also noted that PCR detection of antibiotic-resistance traits is more rapid and sensitive, and potentially more cost-effective, than culture or selective media. This is attractive for clinical diagnosis and surveillance (Lévesque *et al.*, 1995; Briggs *et al.*, 1999; Paule *et al.*, 2001; White *et al.*, 2001; Paule *et al.*, 2003; Blickwede and Schwartz, 2004; Sundsfjord *et al.*, 2004; Shamputa *et al.*, 2004; Jalava and Marttila, 2004). However, in some cases, resistance to antimicrobials can be phenotypically observed with the lack of detection of antimicrobial resistance determinants (Patel *et al.*, 1997). This may occur, as antimicrobial resistance may be conferred by several different genes, not all of which have been characterized. Because of the potential risks associated with misdiagnosis of disease, resistance screening by molecular methods in a clinical setting should be used as a compliment to classical phenotypic approaches.

Other problems may arise when using nucleic acids techniques if researchers are not careful with their methods. Of particular concern is establishing detection limits for nucleic acid techniques for pathogens in food, clinical diagnostic samples, and environmental matrices. Detection limits should be established with samples that closely mimic the matrix of interestenvironmental conditions. If detection limits are reported that over-estimate the efficacy of the method, their use for surveillance or diagnosis may put the public at risk. For instance, Lyon (2001) developed a real-time (TaqMan) PCR method to detect *Vibrio cholerae* O1 and O139 in raw oysters without enrichment. This researcher spiked 25 g oyster homogenates with a single inocula (approximately 6.2 x 10^6 *V. cholerae* O1 and 6.7 x 10^6 *V. cholerae* O139), then serially diluted with alkaline-peptone water to 6 $\log_{10}$ the original concentration. Because both organisms could be detected in the most dilute samples, a detection limit of 6-8 CFU/g oyster was reported. However, by diluting their samples up to six-fold in alkaline-peptone water instead of unspiked raw oyster homogenate, they may have diluted out a significant amount of PCR inhibitors for their most dilute samples. The true detection limits of this assay are unclear.

7.3 Pathogen Viability

For pathogenic zoonoses in different environmental matrices to pose a threat to human health, they must be in a viable state. The detection of viability by either cultivation-based techniques or nucleic acids approaches, however, may not be straight-forward. Cultivation techniques require the viability of microorganisms to yield a result. However, viable-but-nonculturable (VBNC) cells can remain undetected and may complicate interpretation of results. Aside from a clear definition of what constitutes a VBNC state (Barer and Harwood, 1999; Kell *et al.*, 1998; Keer and Birch, 2003; Besnard *et al.*, 2000; del Mar Lleó *et al.*, 2000; Grey and Steck, 2001b; Nilsson *et al.*, 1991; Turner *et al.*, 2000: Bogosian *et al.*, 2000), it still remains unclear as to whether cells in a VBNC state are pathogenic (Barer *et al.*, 2000; Grimes *et al.*, 1986; Steinert *et al.*, 1997; Grey and Steck, 2001a; Cappelier *et al.*, 1999). What is known, however, is that degradation of nucleic acids in VBNC cells may proceed at much slower rates than in killed cells. In fact, some studies have indicated that the pool of messenger RNA (mRNA) may stabilize within VBNC cells rather than continually degrade (Thorne Williams, 1997; Smuelders *et al.*, 1999). Indeed, VBNC bacterial pathogens may harbor genes encoding antimicrobial resistance and other virulence mechanisms for long periods of time after entering a VBNC state (Chaiyanan *et al.*, 2001), serving as a potential reservoir for virulence determinants in the environment. It is clear that molecular methods cannot differentiate between viable and VBNC pathogens when nucleic acids persist in the cells (Thorne and Williams, 1997; Smeulders *et al.*, 1999; Lázaro *et al.*, 1999; del Mar Lleó *et al.*, 2000; Weichart *et al.*, 1997).

Nucleic acid techniques may yield more rapid and sensitive results than cultivation-based techniques for detecting pathogens in different matrices, but there still remains a question as to what a positive PCR result means. Presence of DNA is not a reliable indicator of bacterial viability (McCarty and Atlas, 1993; Masters *et al.*, 1994; Deere *et al.*, 1996; Hellyer *et al.*, 1999). Ribosomal RNA has been shown to be a better indicator of bacterial viability due to a more rapid degradation than DNA upon cell death, but may not be reliable in all cases (McKillip *et al.*, 1999; Villarino *et al.*, 2000l McKillip *et al.*, 1998; Meijer *et al.*, 2000; Tolker-Nielsen *et al.*, 1997). Because of its extremely short half-life following cell death (seconds), mRNA may be the most reliable nucleic acid for indicating cell viability (Keer and Birch, 2003). However, it has been shown that mRNA may still persist. Therefore, care must be taken to design probes and primers that target regions of mRNA more susceptible to degradation (Cook, 2003).

Detecting mRNA requires a higher level of technical expertise than standard DNA-based methods, and does not lend to direct quantitation of pathogens in a sample as multiple and variable quantities of mRNA may be present in a single cell. Quantitative results may, however, be achieved using MPN techniques. Two methods commonly used for detecting ribonucleic acids are reverse-transcriptase PCR and nucleic acid sequence-based amplification (NASBA).

Enrichment of environmental or food samples prior to detection may enhance method sensitivity and aid in the detection of viable versus non-viable cells. Considering the potential of VBNC pathogens to act as environmental reservoirs for virulence determinants, as well as the ability of nucleic acids techniques to detect these cells, molecular methods may offer a distinct advantage over more classic cultivation-based assays for the protection of human health and the environment.

7.4 Emerging Surveillance Technologies

Considerable advances in nucleic acids and sensor technologies continue at a rapid pace (Walker, 2002; Dunbar *et al.*, 2003; Petrenko and Vodyanoy, 2003; Turnbough, 2003; Olsen *et al.*, 2003; Greene and Voordouw, 2003; Grow *et al.*, 2003; Unnevehr *et al.*, 2004; Panicker *et al.*, 2004; Raymond *et al.*, 2005). Of the emerging nucleic acids technologies, perhaps the most promising for surveillance and biosecurity are microarrays. Call *et al.*, (2003) and Ye *et al.*, (2001) describe the emerging use of microarrays and their potential for pathogen detection and genotyping. Microarrays are essentially a large set of very small southern blots, an array of many nucleic acid probes complimentary to discrete gene sequences, bound to a solid or semi-solid matrix, usually a modified glass surface. Because of the miniscule size of the blots (100-200 µm diameter "spots" separated from neighbors by typically 200-500 µm), thousands of sequences can be screened on a single array of less than four square centimeters. Target nucleic acids, which are typically, but not necessarily, PCR products, are challenged by the microarray under stringent hybridization conditions. Targets are usually prepared prior to hybridization with fluorescent labels or incorporating specific chemistries such as biotin-streptovidin that permit detection with a secondary fluorescent marker. Once post-hybridization steps are complete, arrays are catalogued using high resolution laser- or filter-based scanners and charged-coupled device (CCD) imaging. Based on hybridization patterns between the spotted arrays and the nucleic acids targets, the genotype of the original pathogen or the presence of specific pathogens in complex samples can be identified.

Microarrays are presently limited to endpoint detection, rather than quantification, of specific microbial targets. Although in some instances they can be used to detect nucleic acids isolated directly from complex matrices, the sensitivity of microarrays severely impedes their use for pathogen detection at the very low concentrations of interest in environmental samples without a specific nucleic acid amplification strategy (Call *et al.*, 2003). When coupled with nucleic acid amplification techniques, microarrays have been used successfully to detect enterohemorrhagic *E. coli* in chicken rinsate at concentrations as low as 55 CFU/mL. Perhaps more promising, microarrays can be used to rapidly genotype specific pathogens with greater sensitivity (Chizhikov *et al.*, 2001; Call *et al.*, 2001b; Johnson and Stell, 2000; Bekal *et al.*, 2003). Whole and partial genome microarrays for many pathogens are also commercially available, allowing for "fingerprinting" of microbial pathogens by establishing patterns unique to particular species that may further enable genotyping studies. By challenging these arrays with nucleic acids of a wide variety of sources, very small sets of unique markers for specific pathogens may be identifiable, better enabling environmental pathogen detection.

7.5 Discussion

Pathogen detection is performed routinely for meat, produce, seafood, milk, occasionally for biosolids from human municipal treatment facilities, but rarely for environmental matrices such as animal manure and their treated residuals, environmental waters, air, soils, and sediments. The recovery and detection of pathogens in environmental matrices are imperative to identify the extent to which these agents are removed, inactivated, or persist in livestock animal waste treatment processes and management systems at CAFOs. Conventional cultivation-based and newer nucleic acids-based approaches to detect or enumerate etiological agents in some environmental matrices are available. However, these methods are not amenable across different groups of pathogens or matrices and can complicate environmental sampling. Very limited standardization of pathogen detection methods exists, except for in the case of foods (Association of Analytical Communities (AOAC), 2005; FDA, 2005; USDA, 2005). The applicability of the standard methods for detecting pathogens in food to other matrices such as feces, water, and soil has not been established. Standard methods with the required sensitivity for detecting pathogens at relevant concentrations in environmental milieus are sorely lacking, especially for hyper endemic or emerging pathogens such as *E. coli* O157:H7, *Salmonella typhimurium*, *Yersinia enterocolitica*, *Campylobacter jejuni*, swine hepatitis E virus and the protozoan parasites *Giardia lamblia* and *Cryptosporidium parvum* (Sobsey *et al.*, 2002). The efficacy of animal waste management systems for removing zoonotic pathogens and antimicrobial-resistant bacteria from waste streams at CAFOs remains uncertain.

Aside from assessing the efficacy of livestock animal waste management systems, the recovery and detection of pathogens in water is imperative to protecting human health and the environment. The wise old adage of indicator organisms is becoming outmoded, as their reliability to predict all waterborne outbreaks is uncertain, their results come “a day late and a dollar short”, and newer technologies are becoming available that negate the need to rely solely on bacterial indicators of pathogenicity. Improving surveillance activities in recreational and drinking waters will require these new methods for detecting pathogens to be available in near real time. However, this may be hindered by specific physical or chemical properties of environmental waters that, combined with low concentrations of pathogens, may increase sample processing times. In general, the cleaner the sample, the more methods will be effective for rapid pathogen detection. For clean matrices such as drinking water, sample concentration methods alone may be sufficient to yield results directly usable by both cultivation and direct molecular detection. For more challenging matrices, such as ground and surface waters, significant sample clean-up may be required to remove inhibitors prior to processing with molecular methods.

As discussed above, the detection of etiologic agents in environmental waters is problematic, not only due to very low (but potentially significant) concentrations, but also due to a lack of methods with the required sensitivity and competing methods that are not amenable across different groups of pathogens. As such, there is a need to develop a unified and automated system for the detection of all waterborne pathogens (Straub and Chandler, 2003). It has been suggested that such technologies rely on nucleic acids analyses because they are amenable to automation and are at present the most promising for rapid and specific quantitation of viable microbial pathogens (Jothikumar *et al.*, 1998; Levin *et al.*, 2002; Straub and Chandler, 2003). Hollow-fiber ultrafiltration systems can concentrate all classes of pathogens in a single step and can be reused (Kuhn and Oshima, 2001; Olszewski *et al.*, 2001). Therefore, these filters may serve well as a basis for sample concentration in such a system. More recently, renewable surface technologies for automated sample processing coupled with microarray technologies have also shown promise as a basis of such a system (Chandler *et al.*, 2000a,b). Nucleic acids technologies, however, are still primarily in the hands of researchers and beyond the scope of all but the most highly trained staff and most affluent utility laboratories (Levin *et al.*, 2002). Significant investments need to be made in the development of simple and reliable technologies that are less technically-demanding.

Several pertinent issues need to be addressed before nucleic acids technologies are exclusively used as standard methods for surveillance activities, such as monitoring recreational and drinking water quality. First and foremost, there remains a need for regulatory establishment of acceptable concentrations in environmental matrices to determine the relevancy of detection limits established in these studies. Method development and standardization cannot proceed until target detection limits that reflect true risks of illness are established. These limits need to identify target concentrations based on epidemiological studies of health risk rather than indicate that pathogens should not be detected in a given sample volume. Regulatory guidelines should also clearly indicate standards regarding acceptable recoveries from environmental samples during concentration, analysis sensitivities, and standard errors. This is because sample concentration methods may present a wide variation in recovery of etiological agents from environmental waters, and regardless of the sample concentration methods used, PCR-based detection systems must confront a number of front-end challenges inherent to complex environmental waters that may reduce the sensitivity of the assay (Chandler, 1998; Loge *et al.*, 2002; Call *et al.*, 2003). As noted by others, a positive detection is relatively simple to interpret. However, knowledge of assay sensitivity, which varies from sample to sample, is critical to interpreting negative results (Loge *et al.*, 2002; Call *et al.*, 2003). It is possible that specific sample properties, such as the presence of PCR inhibitors, may affect the sensitivity of an assay thereby resulting in a false-negative result. The probability of false negative results will increase with lower numbers of pathogens per sample. Regulatory guidelines should consider these limitations in order to reduce the public health impacts of false-negative results. Good sampling designs need to consider how much sample is processed, the efficiency of pathogen isolation, the efficiency of nucleic acid extraction, and the effect of co-precipitating factors that inhibit PCR (Loge *et al.*, 2002; Call *et al.*, 2003).

Considering that acceptable regulatory concentrations of specific pathogens in environmental matrices will likely be very low, it is unlikely that current technology would be able to detect pathogens in real-time. Measurement of pathogens to satisfy what would be regulatory levels may still take 24-48 hours as they will require enrichment to increase pathogen numbers or reduce co-precipitating factors. This may also pose challenges as several different enrichment media may be necessary to detect several different pathogens. The only true method to alleviate the need for enrichment to detect low numbers of pathogens in environmental samples would be intensive sample concentration. Unless significant advances are made in sample concentration, nucleic acid extraction methods, and nucleic acid clean-up to remove inhibitory compounds, real-time pathogen detection will remain unrealistic. This will remain a critical issue for biodefense applications where near real-time identification of etiological agents may be imperative to protecting human health.

8. Microbial Source Tracking

Microbial source tracking (MST) is a set of methodologies by which the animal or human source of fecal pollution in a contaminated water body may be identified. MST technologies rely on phenotypic and genotypic differences in fecal microorganisms shed in the wastes of animals and humans that make them unique to a particular animal host (host-specific). These differences may arise due to variations in growth environments and the selective pressures of various animal guts such as differences in diet, antimicrobial treatments, temperatures, pH, and more. Upon release into the environment, it is assumed that these organisms remain unchanged and migrate with fecal pollution. Their detection in concert with indicators of fecal pollution or overt pathogens is thus assumed to be indicative of the animal source.

Knowledge of the source(s) of microbial contamination in water bodies and their relative load contribution helps to focus remedial efforts and resources in the right direction at an earlier time. Source identification may also enable investigation of best management practice (BMP) effectiveness leading to improvements in total maximum daily load (TMDL) development and implementation. There are many potential sources of bacteria extant in watersheds, and it is important to be able to sort out the source of observed contamination so that an evaluation of the effectiveness of control strategies can be made. MST may also improve enforcement activities when discharges exceed permitted levels.

In MST, the selection of the right indicator is important, since it is the single element which provides the measurable parameters to determine the origin of the pollution. Both phenotypic and genotypic technologies have been developed primarily using fecal coliforms, *E. coli*, fecal enterococci, fecal streptococci and viruses. Some of these technologies are library-dependent; they rely on comparing fingerprint databases, either phenotypic or genotypic, of microorganisms from known sources to the fingerprints of unknown samples. These may include antibiotic resistance analysis (ARA), carbon utilization profiles (CUP), repetitive extragenic palindromic PCR (rep-PCR) DNA fingerprinting, randomly amplified polymorphic DNA (RAPD) analysis, amplified fragment length polymorphism (AFLP) analysis, pulse field gel electrophoresis (PFGE), and ribotyping (Harwood, 2000; McClellan *et al.*, 2001; Hagedorn *et al.*, 2003; Carson *et al.*, 2003; Ting *et al.*, 2003; Leung *et al.*, 2004; Scott *et al.*, 2004; Webster *et al.*, 2004). Other MST technologies are library-independent; they rely on the conservation of unique genetic identifiers inherent to a specific fecal microorganism endemic to the members of a single animal species (the in-group) that are different from the genetic identifiers of the same or different fecal microorganisms in other animals or humans (the out-group). Examples of library-independent MST technologies include gene-specific PCR, 16S rRNA gene clone libraries, and target-specific PCR-based methods (Bernhard and Field, 2000; Khatib *et al.*, 2002; Khatib *et al.*, 2003; Field *et al.*, 2003; Simpson, *et al.*, 2003; Bonjoch *et al.*, 2004; Scott *et al.*, 2004; Simpson *et al.*, 2004; Suerinck *et al.*, 2005).

At present, MST studies have primarily employed library-dependent methods including ARA, AFLP, CUP, and ribotyping (EPA, 2005). Library-independent methods, especially gene-specific and target-specific PCR, have been the focus of recent literature as extensive libraries are not needed for their application, and they can be easily and rapidly applied to source identification studies. Potentially, phenotypic and genotypic methods could complement each other according to training, equipment, and funding available. This section highlights the more common methods of analysis that provide information about the source of microbial contamination and the methods used to analyze the data obtained by these methods. A full review of all available MST technologies is beyond the scope of this review. An excellent resource for further information regarding MST, its application, promises, and limitations is available in the USEPA Microbial Source Tracking Guide Document (2005).

8.1 Antibiotic Resistance Analysis (ARA)

The antibiotic-resistance analysis (ARA) method has gained popularity over the last decade because it is readily applicable and simple to use. However, the classification accuracy is usually lower than that of the molecular methods at the level of individual species. When animal species are grouped into larger animal categories like human, livestock, and wildlife, the accuracy improves notably to values of 95% or more. This method has been reported to provide sensible classifications of known and unknown fecal isolates and to resolve MST queries to satisfaction in various case studies (Parveen *et al.*, 1997; Hagedorn, 1999; Harwood *et al.*, 2000).

ARA is based on the following two premises (1) the use of antibiotics in humans and animals can result in antibiotic-resistant bacteria, and (2) differences in the selective pressures resulting from dosing with different types and concentrations of antimicrobials, as well as different growth environments in various animal intestines, result in unique patterns of antibiotic resistance specific to different animal types. The development and study of such specific patterns are the basis of this methodology, which uses fecal coliforms, *E. coli*, enterococci, or fecal streptococci as indicators. The laboratory procedure requires only conventional microbiology training and techniques. Therefore, it is cost-efficient and can be rapid when performed by an experienced research team. It begins with the recovery of the fecal bacteria from samples, mostly by membrane filtration and incubation in or on selective media. It continues with the inoculation of the isolated fecal bacteria onto agar or into broth medium containing a number of antibiotics at increasing concentrations. Lastly, the results are evaluated by comparing the antibiotic resistance profiles of the polluted water to the reference source library profiles.

The key to success of this method is having a representative source library with an acceptable *average of correct classification* (ARCC). Most important is to perform a cross-validation test before using the library to classify unknown isolates. The cross-validation test can be done with hold-out isolates or with new known isolates that are submitted as unknowns to the statistical software (Harwood *et al.*, 2000). The rate of correct classification from this test should not be significantly different from the original rates obtained when the library was initially classified. Similarly, it is important to use the antibiotics and the concentrations that provide the more accurate classification of the library isolates. For instance (as noted above), several antimicrobials used in animal agriculture have human-use analogs. Resistance that develops in livestock animals may therefore confer resistance to human antimicrobials and vice-versa. Therefore, preliminary tests are recommended prior to the analysis of the water isolates (Hagedorn *et al.*, 1999). While this method does not classify individual animal species very accurately, it has been used with satisfaction for human, poultry, livestock and wildlife categories (See examples below). In most cases, this is sufficient and supports the development of a restoration strategy.

Several examples of the use of ARA for MST exist. Parveen, *et al.*, (1997) used ARA methodology to differentiate point-source (PS) from nonpoint-source (NPS) *E. coli* from the Apalachicola Bay, Florida. They used 765 isolates and obtained average MAR indexes of 0.25 for PS and 0.13 for NPS. PS isolates showed higher resistance to single antibiotics and to combinations of antibiotics than NPS isolates. Sixty-five resistance patterns were observed for PS isolates, compared to only 32 patterns for NPS isolates, when cluster analysis was used. Wiggins (1996) developed a protocol to generate more extensive AR profiles by using various concentrations for each antibiotic tested. He also introduced the use of a reference source library and the multivariate analysis of variance discriminant function analysis to the studies of MST by ARA. He studied 193 water fecal streptococci, with a source library of 1,435 fecal streptococci isolates against a battery of five antibiotics at four concentrations each. An ARCC of 72% was obtained when the source categories were analyzed at the species level and 82% when some species were pooled into "poultry" and "beef" categories. In general, increasing the number of antibiotics used in an analysis increased the ARCC that could be achieved.

Others have further validated the ARA method by using additional statistical analysis. Hagedorn (1999) created separate source databases with 7,058 and 892 isolates with ARCCs of 87% and 88%, respectively, which increased to 97% and 95% after pooling species into poultry and beef. They used discriminant analysis to classify 4,615 water isolates from Page Brook River and obtained 82% beef, 7.3% deer, 5.6% waterfowl, and 0% human, and 5.3% unknown. Cluster analysis was also used, which generated very good separation between sources with high antibiotic resistance including chicken, dairy cattle and human clusters. Among beef cattle and deer, which had low levels of resistance, there was separation, but the deer tended to sub-cluster within the larger beef cow isolate cluster. Most of the unknown source isolates were grouped in beef cow, deer, and waterfowl clusters, whereas none grouped in the human cluster. Based on the results of this work, cattle access to the stream was reduced by fencing.

Harwood *et al.*, (2000) obtained ARCCs of 64% and 62 % with a source database of 6,144 fecal coliform and 4,619 fecal streptococci isolates. Similar to Wiggins (1996) and Hagedorn (1999), the ARCCs improved to 75% and 72%, respectively, after pooling the species into human and animal groups. Fecal coliforms from cattle were classified correctly at a higher rate than those of fecal streptococci. Conversely, fecal streptococci from humans were correctly classified at a higher rate than those from fecal coliform isolates. Overall, the fecal coliform database had a significantly greater ARCC than the fecal streptococci database. Spearman's ranked correlation using the percentage of correctly classified isolates versus the corresponding number of sampling events resulted in a significant negative-correlation between sampling events and the percentage of correctly classified isolates for the fecal coliform database, but not for the fecal streptococci database. They analyzed 91 fecal coliform isolates from surface water receiving effluent from faulty septic systems, 81 of which were classified into the human category. Similarly, 38 of 51 fecal streptococci from the same samples were categorized as human. After the septic systems were repaired, only 7.8% of fecal coliforms and 1.2% of fecal streptococci were classified as human.

Graves, *et al.*, (2002) constructed a library of 1,174 enterococci isolates, and with two categories (human and nonhuman) achieved an ARCC of 96%. By splitting nonhuman sources into livestock and wildlife, they were able to achieve an ARCC of 92%. They analyzed 2,012 enterococci isolates from a stream that drains a watershed with large populations of livestock and wildlife and that passes through a community of 82 homes served by individual septic systems. The yearly average classification was 10% human, 40% wildlife, and 50% livestock. Burnes (2003) analyzed 800 fecal coliform isolates from Big Creek, a mixed-use watershed, against a source library of 1,125 fecal coliform isolates (human and non-human categories, ARCC=94%). He found that human sources contributed greater than 50% of the base flow fecal coliforms in urbanized areas. Chicken and livestock appeared to be responsible for the base flow fecal coliforms found in rural reaches of the stream. Hydrologic events changed the contribution of each source to the stream such that fecal coliform pollution was 16% attributable to domestic sources, 21% attributable to wildlife, and up to 60% attributable to chickens and other livestock sources.

8.2 Ribotyping

Ribotyping is a DNA fingerprinting method that exploits small differences in 16S and 23S rRNA-coding regions of bacterial DNA to identify genetic relationships between unknown bacteria and a set of known index organisms (Grimont and Grimont, 1986; Stull *et al.*, 1988; Graves *et al.*, 1999). This method works on the premises that (1) multiple copies of the genes encoding 16S and 23S rRNA may appear within the bacterial genome with different flanking restriction site locations, (2) there is variability amongst 16S and 23S rRNA genes, and (3) there is variability in the intergenic spacer region between 16S and 23S rRNA genes. Ribotyping involves the culturing of a bacterium followed by DNA extraction and purification. Subsequently, the DNA is digested with one or more enzymes and the digestion products are separated by gel electrophoresis. DNA bands are typically transferred onto a nylon membrane and challenged by hybridization analysis with a chemically-labeled nucleic acid probe. The probes may be generated from an index bacterium, such as a particular strain of *E. coli*, by reverse transcribing the 16S and 23S rRNA and labeling the cDNA with a chemical labeling scheme. Because of small differences in the restriction sites of different bacteria, the resulting band patterns from the hybridization analysis will be distinct. This pattern is called a ribotype. By comparing ribotypes of unknown samples to a library of known samples challenged by probes from the same index organism, genetic and evolutionary relationships can be discerned. Ribotypes may be translated to a binary code facilitating a discriminant statistical analysis to aid in interpretation of results (Grimont and Grimont, 1986; Parveen, 1999; Carson *et al.*, 2001).

In early years, ribotyping was used in epidemiological studies to characterize bacteria such as *E. coli*, *Salmonella enterica* and *Vibrio cholerae* (Stull *et al.*, 1988; Olsen *et al.*, 1992; Popovic *et al.*, 1993). Gradually, ribotyping found application in multidisciplinary areas such as plant pathology, animal science, food technology, and MST (Nassar *et al.*, 1994; Nagai *et al.*, 1995; Kilic *et al.*, 2002; Scott *et al.*, 2004). By comparing libraries of ribotypes grouped by animal source, some researchers have noted that distinct bands unique to specific sources may emerge from non-distinct bands, thus facilitating the identification of the source of pollution in unknown samples. For instance, Parveen (1999) tested the applicability of this methodology to predict the source of *E. coli* pollution in the Apalachicola Bay, Florida. They analyzed a library of 238 *E. coli* isolates and found that discriminant analysis of the ribotype profiles showed an ARCC of 82%. A total of 97% of nonhuman and 67% of the human isolates were correctly classified. Carson *et al.*, (2001) extended the application of ribotyping to distinguish *E. coli* from humans and seven nonhuman hosts. When ribotypes from a library of 287 *E. coli* isolates obtained from humans, cattle, pigs, horses, chickens, turkeys, migratory geese, and dogs were used, the ARCC was 73%. By reducing the discrimination to human and nonhuman sources, these authors were able to achieve an ARCC of 97%. Using a library of 160 *E. coli* isolates, Scott *et al.*, (2004) was able to employ ribotyping to identify animals as the primary source of pollution in a water way near Charleston in South Carolina. Prior to this investigation, a significant human input was suspected. Kuntz *et al.*, (2003) successfully combined targeted sampling protocols with ribotyping to identify the source of fecal contamination of Sapelo River in Georgia. *E. faecalis* DNA fingerprints in the river were a 43% match to those in a nearby wastewater lagoon, suggesting that fecal contamination of the river originated from the wastewater treatment facility.

Other studies have used ribotyping successfully but have noted caveats in its application. For instance, Hartel (2002) studied 568 *E. coli* isolates from different locations in Georgia and Idaho to determine the geographic variability of *E. coli* from different animal species. They found that the percentage of ribotype sharing within an animal species increased with decreased distance between geographic locations for cattle and horses, but not for swine and chicken. The data suggested that the ability of libraries to classify unknown isolates is good provided both library and unknown isolates belong to the same geographic area. Wheeler *et al.*, (2002) explored the potential of *E. faecalis* as a human fecal indicator for MST using ribotyping. He analyzed fecal samples from humans and a variety of livestock, domestic, and wildlife and found that the host range of *E. faecalis* was limited to dogs, humans, and chickens and the ribotypes clearly distinguished between human and chicken hosts. The dog isolates were apparently eliminated when a protocol to quickly isolate *E. faecalis* was used. Hartel *et al.*, (2003) noted that ribotypes of *E. coli* isolates from wild deer was significantly affected by their diets. Wild deer exhibited 35 *E. coli* ribotypes, whereas penned deer generated only 11. Although issues of geographic stability, host range of target bacteria, and host stability of ribotypes may be of concern in some instances, ribotyping remains a preferred and well-accepted method for MST. From the results above, it appears to be a reliable tool to discriminate between pollution sources and provide valuable information for water management purposes.

8.3 Amplified Fragment Length Polymorphisms (AFLP)

AFLP is a DNA-fingerprinting method based on the detection of characteristic differences in the fingerprints of two genomes resulting from polymorphisms, insertions, and deletions that occur within or immediately adjacent to restriction sites. In this method, genomic DNA is extracted from the target cells and digested with a pair of restriction enzymes. Linkers specific to the restriction enzyme sites chosen are ligated to the DNA fragments providing the sequences for hybridization of PCR primers in the amplification steps. As large numbers of different fragments can be generated during digestion (more than 10^6 fragments per digestion of a genome of 10^9 base pairs (bp)), selective DNA amplification is used to limit the number of amplification products. Selective amplification can be achieved through a variety of mechanisms including 3’-extensions to one or both linkers, “pre-amplification” with only one primer complementary to one of the restriction enzyme sites, inclusion of 3’-extensions on the “pre-amplification” primer or one or both AFLP-PCR primers, and labeling only one of the AFLP-PCR primers. Amplified products are separated on a denaturing polyacrylamide gel in an automated DNA sequencer, and “fingerprints” are captured by specialized software which can scan the fingerprints for discriminatory bands. Evaluation of the results by various statistical analyses can be performed provided the DNA banding patterns are converted to binary form.

AFLP has proved to be highly discriminatory and reproducible when compared to other molecular typing techniques. During the last decade, it has become a reliable tool to classify bacteria to the strain level, as well as to perform genetic mapping of higher organisms (Janssen *et al.*, 1996; Desai *et al.*, 1998; Savelkoul *et al.*, 1999; Iyoda *et al.*, 1999; and Zhao *et al.*, 2000). Two reported evaluations of AFLP as a tool for MST stand out in literature. Both investigations compared AFLP to other fingerprinting techniques and utilized *E. coli* as the indicator organism. In the first of these studies, Guan *et al.*, (2002) studied a collection of 105 *E. coli* isolates from the feces of cattle, poultry, swine, deer, goose, moose, and human samples and compared AFLP to the ARA and the 16S rDNA methods. The results indicated that AFLP was significantly more effective than the other two methods. Ninety-four percent of the livestock isolates, 97% of the wildlife isolates, and 97% of the human isolates were correctly classified by AFLP. In comparison, 46% of livestock isolates, 95% of wildlife isolates, and 55% of human isolates were correctly classified by ARA, while 16S rDNA-based techniques resulted in 78%, 74%, and 80% correct classification, respectively. Although additional isolates in the source library may improve the ARCCs of the ARA and 16S rDNA methods, the resolution achieved by AFLP with a small library was impressive in this instance.

In another study, Leung *et al.*, (2004) used Shiga-toxin-producing *E. coli* (STEC), enterotoxigenic *E. coli* (ETEC), and non-pathogenic *E. coli* from cattle, swine, and human sources from very diverse geographic areas, including the U.S., Canada, Europe, and Australia, to construct source libraries. A multiple-response permutation procedure analysis of the data obtained with AFLP indicated that the seven groups defined by host-pathogenicity combinations (bovine STEC, bovine ETEC, bovine non-pathogenic *E. coli,* human STEC, human ETEC, human non-pathogenic *E. coli* and swine non-pathogenic *E. coli*) were significantly different. Subsequently, stepwise discriminant function analysis was used to select 39 discriminant DNA bands distinguishing the host specificity of the *E. coli* strains for the analysis. The overall cross-validation classification efficiency was 93.6% with 91.4% of human, 90.6% of bovine, and 97.7% of swine isolates being classified into their correct host types.

AFLP also distinguished the non-pathogenic *E. coli* from STEC and ETEC, and was able to classify the strains based on both host specificity and virulence (Leung *et al.*, 2004). Stepwise discriminant function analysis selected 41 DNA bands to classify the isolates based on pathogenicity with an overall cross-validation classification efficiency of 99.1% (100% of non-pathogenic *E. coli,* 100% of STEC, and 90.9% of ETEC correctly classified). Fifty DNA bands were selected by that same means to differentiate the seven host-pathogenicity combinations (bovine VTEC, bovine ETEC, bovine non-pathogenic *E. coli*, human VTEC, human ETEC, human non-pathogenic *E. coli* and swine non-pathogenic *E. coli*) with an average cross-validation classification efficiency of 86.4% (individual group classification efficiency ranging from 50 to 100%). Despite the wide geographic origin of the *E. coli* strains in this study, AFLP was capable of differentiating the *E. coli* strains with a high rate of correct classification from the various hosts.

Like most methods used in MST, AFLP needs the development of a reference source library of the indicator organism. Based on the reports above and the fact that AFLP screens the entire genome, this tool has great potential for MST. However, reports of case studies with successful application of AFLP are necessary to further MST using AFLP.

8.4 Host-specific molecular biomarkers

Host-specific molecular biomarkers for MST studies, target-specific PCR-based methods, are attractive because they offer rapid analysis (no source library or cultivation are needed) and greatly reduced cost. They are technically less demanding than most alternative MST techniques. Several bacterial targets have been proposed in the literature for MST applications including *Bacteroides*, *Bifidobacterium*, *Streptococcus* Lancefield Group D, and *Rhodococcus coprophilus* (Whitehead and Cotta, 2000; Vancanney *et al.*, 2002; Bernhard *et al.*, 2003; Bonjoch *et al.*, 2004; USEPA, 2005). However, since each marker is specific to a single animal host, a combination of markers may be required to fully identify potential sources of contamination. Because no cultivation is required, these methods are applicable to detection of several biomarkers in a single sample, which may strengthen the argument for source identification (USEPA, 2005). For instance, samples with positive results for several human-specific biomarkers, such as human-specific *Bacteroides* spp. (Bernhard *et al.*, 2003), both the human-specific primer pairs for *Bifidobacterium adolescentis* and *Bifidobacterium dentium* (Bonjoch *et al.*, 2004), and the human-specific *Enterococcus* spp. *esp* markers (Scott *et al.*, 2005), but negative for the cattle-specific *Bacteroides* (Bernhard *et al.*, 2003) or *E. coli* LTIIa toxin (Khatib *et al.*, 2002), would strongly implicate a human rather than cattle source. At present, *Bacteroides* markers are the most commonly used host-specific molecular biomarkers for MST (Bernhard *et al.*, 2003; Seurinck *et al.*, 2005; USEPA, 2005). However, the use of these markers for resolving watershed-scale microbial pollution is unknown.

The development of the human- and ruminant-specific *Bacteroides* biomarkers is described by Bernard and Field (2000a,b). These researchers created 16S rDNA clone libraries of members of the *Bacteroides-Prevotella* group from human and cow fecal samples. Individual and pooled clones were examined by length heterogeneity (LH)-PCR and terminal-restriction-fragment–length-polymorphism (T-RFLP) techniques. Sequencing and phylogenetic analysis demonstrated that the human-specific sequences clustered together and were closely related, but not identical, to sequences of *Bacteroides vulgatus*, which is commonly found in human feces. The cattle-specific sequences formed the new gene clusters CF123 and CF151. All human- and cattle-specific genetic markers were found in DNA extracted from river and estuary water contaminated with fecal pollution.

Primer sets were developed to amplify specific sequences within the *Bacteroides-Prevotella* host-specific gene clusters (Bernhard and Field, 2000a,b): one forward primer specific for the human-specific gene cluster HF8, two forward primers for the cattle-specific gene clusters CF123 and CF151, and a general *Bacteroides-Prevotella* reverse primer for use with all three forward primers. These primers were successfully used to amplify 16 human and 19 cow fecal 16S rDNAs (Bernard and Field, 2000b). In subsequent work, Bernhard *et al.*, (2003) used these primers to test 22 water samples from Tillamook Bay and the results were congruent with land use. For example, the human specific primer pair, HF183F/Bac708R, amplified only DNA from waters around urban areas and sewage treatment plants, demonstrating specificity for human *Bacteroides* DNA. The cattle specific primers, CF128F/Bac708R and CF193F/Bac708R, amplified DNA from waters primarily near rural areas. However, these primers amplified other ruminant DNA as well. Therefore, positive results with these primers should be scrutinized against land use and ruminant wild-life populations to prevent misinterpretation of results. Dick *et al.*, (2005) have also recently published host-specific primers for swine and horse.

Relatively few other host-specific molecular biomarkers have been reported in literature (Khatib *et al.*, 2002a,b; Nebra *et al.*, 2003; Bonjoch *et al.*, 2004; Scott *et al.*, 2005). Most of these markers target 16S rDNA, but some markers are emerging that detect alternative regions of the bacterial genome, such as virulence genes (Khatib *et al.*, 2002a,b; Scott *et al.*, 2005). The efficacy of these and other biomarkers have not been well established. Therefore, interpretation of the presence of these molecular markers in different *milieus* may be complicated. Because of the small number of environmental samples studied so far, host-specific molecular biomarker technology needs further exploration in case studies to show field-applicability.

8.5 Discussion

Several methodologies are available for MST studies. As shown by the publication evidence, ARA represents a good tool for MST validated by various applications in real-life case studies. However, specific considerations may limit its usefulness including human-livestock analogs, and transfer of resistance traits in different *milieus*. Ribotyping is probably the most field-tested method among the molecular methodologies used for MST. From the results above, it appears to be a reliable tool to discriminate between pollution sources and provides valuable information for water management purposes. Although ribotyping may generate high ARCCs, it has also been associated with a high cost and may be labor intensive, requiring long studies to achieve results (Hartel *et al.*, 2003). AFLP technologies are emerging as a promising tool for MST. These methods offer a high degree of specificity, as they can screen the entire genome instead of selected regions such as the 16S and 23S rDNA screened by ribotyping. However, considerable technical expertise and expense may be needed to fully utilize the technique. Other molecular methods, although sophisticated and capable of measuring parameters with high resolution, are in various stages of development. More research is needed for these methods to become accessible for a broader population of users. As of today, these techniques have mostly been used in feasibility studies with a small amount of isolates. Full-scale watershed studies are needed to assess the potential of these technologies for future use.

Several factors may complicate the use of MST technologies in contaminated watersheds. Poor survival of reference organisms in the environment may result in little or no detection, limiting the ability of the various methods to identify the source of fecal contamination (Simpson *et al.*, 2003; U.S.EPA, 2005). Even those reference organisms that are reasonably hearty may exhibit variable survival times for different phenotypes or genotypes dependent on the environmental milieu. This may lead to changes in genotypic and/or phenotypic signatures of the overall populations and divergence from fingerprints in source libraries established with the raw fecal material. For instance, Anderson *et al.*, (2005) studied decay rates for fecal bacterial indicator organisms (fecal coliforms and *Enterococcus* spp.) originating from dog feces, wastewaters, and soil in freshwater and sediment and saltwater and sediment. These researchers observed variable decay rates based on fecal bacterial source, environment, and even ribotype. These changes may complicate interpretation of environmental fingerprints and undermine source tracking efforts, potentially resulting in misinterpretation of environmental data.

Another serious complicating factor is that the transport properties of different bacteria may vary several-fold depending on the specific microbial agent and the *milieu* (Ferguson *et al.*, 2003). If transport and survival of the index organism(s) used to identify fecal pollution source(s) do not match that of pathogens emanating from potential sources, MST may yield questionable information. For instance, Simpson *et al.*, (2003) could not establish a relationship between the molecular fingerprints of 16S rDNA fecal *Bacteroides* clones in a large horse manure pile immediately adjacent to a receiving stream and downstream water as close as 5 m from the manure pile. In contrast, other researchers have noted significant overland and downstream transport of antibiotic-resistant bacteria and several bacterial, viral, and parasitic pathogens, such as *Cryptosporidium parvum*, *Salmonella* spp., swine hepatitis E, and *Yersinia enterocolitica*, for several hundred meters from concentrated animal feeding operations (CDC, 1998; Karetnyi *et al.*, 1999; Gurdin *et al.*, 2002). Further, if several different bacterial index organisms are used for source identification, as suggested by the USEPA (2005), the transport properties of the individual agents need to be clearly defined to interpret what differences in the level of detection for different animal sources mean. MST studies need to identify the distance at which selected biomarkers may be detectable from their pollution source(s) and whether or not this may be indicative of the fecal pathogens emanating from the same source.

9. Treatment Technologies and Management Practices

As animal agriculture has evolved to larger facilities with large numbers of animals in limited spaces, the problems associated with manure handling have grown. A single swine, beef, dairy, or poultry facility can produce waste equivalent to a small city. The waste is, for the most part, untreated and spread into the environment with little control on dissemination of microorganisms in the waste. All animal manures contain microorganisms, some of which are pathogenic to humans and other animals. Zhao *et al.*, 1995 surveyed dairy herds in 14 states to determine the prevalence of *E. coli* O157:H7. Their results indicated that *E. coli* O157:H7 was present in about 5% of each herd.

Prior sections of this report have enumerated the pathogenic organisms in manure and detailed the illnesses associated with them. Environmental problems originate when the manure containing pathogens is distributed into the open environment with no effort made to reduce the content of pathogens or limit their movement in the environment. Wind, surface flow, and subsurface flow can all carry enough pathogens to receiving waters to exceed water quality standards. In many cases, streams and lakes are used for recreation, and the people using them can be exposed to infection without knowing that they have been exposed. Knowledge of the survival and transport of potential pathogens in the environment is critical for implementing corrective actions on the landscape to limit people's exposure to pathogens.

Microorganisms can move in the environment in several ways. Organisms can move with any dust produced in animal housing, feedlots, or manure spreading. Other airborne transport can happen as liquid waste is spread by spraying as an irrigation process, spraying from an application vehicle, or agitation of lagoons prior to spraying. After manure has been applied to a field surface, microbes can move with water when rainfall exceeds the infiltration rate, thereby creating runoff. Rainfall impact dislodges the organisms from soil or manure particles, and flowing water transports them to receiving waters. Another path for movement of organisms is through subsurface drainage. Microorganisms can enter worm burrows or root channels and move downward in the soil profile as the water flows to groundwater or to drainage tile. If groundwater is shallow, it is possible for serious contamination to occur in wells situated too close to application sites or CAFO installations.

Tile drains can short circuit groundwater recharge by intercepting water and diverting it to streams before it can percolate through the soil. Water that infiltrates the soil down to tile depth is usually below the majority of the root zone and thus not available for plant uptake. Tile drains can accelerate the movement of nutrients and bacteria into receiving waters (Joy *et al.*, 1998, McLellan *et al.*, 1993). Hunter *et al.*, 2000 found that sheep grazing in pastures in England could adversely affect water quality even though the animal population was quite low, one animal per square kilometer basis. Tile drains and open ditches were conduits for microorganisms from pastures to streams. Janzen *et al.*, 1973 found that water quality in streams near dairy farms frequently exceeded coliform limits due to bacterial contamination. About 42 % of the farms were responsible for exceeding standards. Some research has shown that waste applied to the surface of a tiled field can enter the tile quickly after a rainfall. The width of the zone affected by tile at the soil surface is on the order of a meter. In fields that have tile drains, the spacing of tile lines is on the order of 25 or more yards depending on the soil type. Coarser grained soils will allow wider spacing of tile lines than fine grained soils. Each tile line will drain infiltrating water down to its level after a rain event. A portion of the infiltrating water will drain below the tile level and enter groundwater as recharge. There is a possibility that groundwater can be contaminated by manure spread on the surface.

As shown earlier in this report, beef, dairy, poultry and swine operations have become fewer in number and larger in animal populations. The result is to produce more manure in limited areas with little opportunity apply it to land at low enough levels to reduce pathogenic organisms to background levels. Microorganisms in manure produced in large CAFOs pose a serious risk to water quality for recreation, human health, and possibly to nearby farms by spreading disease. One of the most feared occurrences in the agricultural community is an outbreak of disease among farm animals. Recent outbreaks of hoof and mouth disease in the United Kingdom led to multiple billion dollar losses (Ferguson *et al.*, 2001). Reducing the presence of potential pathogens in wastes applied to land will go a long way to improving the safety of farms from disease. Reduction of the bacterial population in water can also improve downstream biosecurity of adjacent farm operations. Manure management practices and potential treatment technologies can be applied to reduce the number of microorganisms distributed into the open environment. There is a great need to implement microorganism reduction techniques to animal waste to prevent detrimental environmental effects.

9.1 Manure management: active and passive systems

Common practices used for managing manures in the U.S. include passive and active approaches. The passive systems include lagoons, storage prior to disposal, vegetated buffer strips, constructed wetlands, separation of different ages of animals, and land application. The passive systems do involve manipulation of manure to move it and eventually land apply the materials, but do not require more than minimal operator input. Active systems include composting, anaerobic digesters, aerobic digesters, and actively operated lagoons. Active systems require more operator attention, such as turning compost windrows, monitoring digesters, and mixing lagoons. In both cases, the key factor is operation with minimum input of labor and capital.

Lagoons are large excavations that may or may not be lined with plastic or clay that receive liquid wastes from animal confinement buildings. Lagoons can be single or multiple cell designs. A passive lagoon is effectively anaerobic due to the large load of organic matter flowing into the lagoon and limited aeration from diffusion and wind. Multiple cell designs can have anaerobic cells, followed by increasingly aerobic cells as the organic content of the waste stream decreases by settling and degradation by microorganisms. Lagoons can also be modified by adding covers to collect methane, or as a permeable cover to oxidize ammonia to nitrate that can be reduced to nitrogen gas in the microenvironment of the cover.

In some cases, usually dairy, beef, and poultry operations, manure is simply scraped into piles and held until a convenient time for disposal by land application. Separation of animals into age groups has also been shown to be effective in reducing specific pathogens, especially *C. parvum* (Atwill *et al.*, 1999). Young animals frequently shed large numbers of oocysts, and older animals do not. Manure from calves can be collected and treated separately from the larger quantities of manure from older animals (Hutchinson *et al.*, 2005). The costs of treatment are lower due to the much smaller mass to treat. In some cases, manure is periodically removed from the animal confinement buildings and directly land applied with no treatment. Simple holding of waste for greater than 90 days will achieve bacterial reductions of >90% (Thayer *et al.*, 1974).

Vegetated buffer strips (VBS) are placed along the downslope sides of fields where wastes are applied to intercept runoff water. As the water flows across the buffer strips contaminants are retained in the vegetated area. The vegetation slows water velocity, allowing settling of particles and infiltration of water into the soil. A well-designed VBS will reduce the quantity of microorganisms leaving a field during runoff events. Many studies have shown greater than 50% reduction of bacterial populations between water entering a VBS and water leaving a VBS. While this reduction may not fully achieve primary contact standards, it is a step in the right direction. Key factors in VBS success are the width of the VBS, slope of the soil, type of soil, and degree of vegetative cover. Good buffers are usually about ten meters wide, with slopes less than 8%, and have about 90% coverage with vegetation. Other management options can help lower the numbers of organisms being applied to the fields. Microorganisms can be retained in strips and wetlands to a significant degree; however, reduction of loads to meet water standards has proven elusive. Many VBS studies show reductions of organisms reaching streams by as much as 90+% (Coyne *et al.*, 1998). The difficulty arises in that reducing populations 90% (e.g., $1x10^6$ to $1x10^5$ per 100 mL) can still leave more organisms in the water than standards allow (Coyne *et al.*, 1995). Table 11 summarizes research done on the trapping of microorganisms in VBS systems and wetlands which primarily treat surface water flow and some shallow ground water.

Active manure management systems involve more operator participation to maintain functionality. Composting requires attention to carbon to nitrogen ratio, moisture, and periodic aeration. In composting, the degradable organic matter is consumed by microorganisms reducing the mass of material. During the process, the temperature of the compost pile will rise to over 50^o C. Under these conditions, pathogenic organisms cannot survive. Composting is very effective in reducing pathogenic organism content of wastes (Olson, 2003). Adequate disinfection requires that conditions of specified time at specified temperatures be met.

Anaerobic digesters can be either plug-flow or mixed reactors. In both cases, the easily degraded organic matter in the waste stream is consumed, reducing the oxygen content of the reactor to methanogenic conditions. Generation of methane can partially offset the costs of reactors and maintenance by providing electricity and / or hot water for the farm. Anaerobic digesters can be operated at ambient (20-30^{o}C), mesophilic (30-37^{o}C), or thermophilic (45-55^{o}C) temperatures. The efficiency of the reaction and reduction of pathogens is different under the different temperature conditions. The thermophilic reactors are more efficient in production of methane and in destruction of pathogenic organisms (Sobsey *et al.*, 2002). However, they are more susceptible to upsets. Aerobic reactors actively incorporate oxygen into the reactor fluid with the goal of maintaining aerobic metabolism by the microorganisms. Aerobic reactors can also operate at ambient, mesophilic, and thermophilic temperatures. The benefit of an aerobic reactor lies in odor reduction and greater carbon mass reduction. Pathogenic organisms are also reduced in aerobic reactors, with greater reductions occurring at higher temperatures (Hill, 2003). In an earlier study, Munch *et al.*, 1987 determined the decimation times for several pathogens in cattle and swine manure slurries from five herds in the temperature ranges 18-20^{o}C and 6-9^{o}C. The results are shown in Table 12. As can be seen in these results some organisms are relatively poor in survival. In general, colder environments favor survival, and aeration favors the reduction of microorganisms. Other organisms, especially fecal streptococci and *E. coli,* can have very long decimation times at cool temperatures. Management practices that address the resistant organisms should at the same time reduce the less-resistant organisms.

Table 11. Summary of microorganism retention in vegetated buffer strips and wetlands.

Type	Width (meters)	Slope (%)	*Protozoan Parasites* (*Cryptosporidium or Giarda*)	Viruses	Fecal Indicator Bacteria: Coliforms	Fecal Indicator Bacteria: Streptococci	Reference
Vegetative Buffer Strips							
Grass	1	5-20	35->99%				Atwill *et al.*,2002
Grass	1.1	5-20	90-99%				Tate *et al.*,2004
Grass	1.5	10, 20	>99%				Davies *et al.*, 2004
Grass	2	8-10			370-600[†]		Pote *et al.*, 2003
Grass	3	3.3			90%	>95%	McCaskey *et al.*, 1971
Grass	4.5				75%	68%	Coyne *et al.*, 1998
Grass	5	8			>95%[‡]		Collins *et al.*, 2005
Grass	6.1	3			6,000[†]		Busheé *et al.*, 1998
Grass	9	9			43-74%		Coyne *et al.*, 1995
Grass	9				91%	74%	Coyne *et al.*, 1998
Grass	22	10-30	>99%				Tate *et al.*, 2000
Grass	70				2,900-10,000[†]	4,800-17,000[†]	Heinonen-Tanski *et al.*, 2001
Grass	NR[§]	1.5-4.5	99.4- 98.3%				Trask *et al.*, 2004
Grass plus forest	30				>90%		Entry *et al.*, 2000
Corn and grass	41				69%	70%	Young *et al.*, 1980
Grass and barley stubble	600 m^2				>93%		Fenlon *et al.*, 2000
Wetlands							
Wetland	100			85%			Chendorian *et al.*, 1998
Wetland	NR		87% and 64%[§§]		99%		Ferguson *et al.*, 2003
Wetland	2 cell				96-97%[*]		Sobsey and Hill, 2002
Wetland	3 cell			85%			Chendorian *et al.*, 1998
Wetland	4 cell				99-99.9%		Behrends *et al.*, 1999

† CFU per 100 mL in runoff
‡ under low flow conditions
§ NR = not reported
* in each cell
§§ For 87% for *Cryptosporidium*, 64% for *Giardia*

Table 12. Bacterial decimation times in aerated and non-aerated manure slurries in weeks.

Organism	Aerated		Non-aerated	
	7 ºC	20 ºC	7 ºC	20 ºC
Fecal streptococci				
Cattle	6.3-18.5	2.5-3.9	12.1	4.1-6.9
Pig	19.2	5.1-6.7	21.9	5.5-7.0
Overall	**12.0**	**5.4**	**21.4**	**5.7**
Escherichia coli				
Cattle	1.4-1.8	0.7-2.2	3.4-6.9	1.6-4.5
Pig	1.7-2.7	0.7-1.7	3.4-17.2	1.3-1.9
Overall	**2.1**	**1.5**	**8.8**	**2.0**
Salmonella typhimurium				
Cattle	1.3	0.5	4.7	1.9
Pig	1.6	0.7	5.8	1.8
Overall	**1.6**	**0.6**	**5.9**	**2.0**
Staphylococcus aureus				
Cattle	NR †	NR	NR	NR
Pig	1.8-2.4	0.5-1.1	2.3-7.5	0.8-1.2
Overall	**2.6**	**0.7**	**7.1**	**0.9**
Yersinia enterocolitica				
Cattle	NR	NR	0.9	NR
Pig	0.6-0.7	0.3	1.0-1.5	0.5
Overall	**0.7**	**0.3**	**1.6**	**0.6**

† NR = Not Reported

The pathogen reduction effectiveness of different manure management practices is shown in Table 13. In most cases, potential pathogens are reduced in common practices by 2 $\log_{10}$ orders or 99%. While this is important, it is not enough to achieve acceptable water quality standards in receiving waters. Lagoon effluent, surface runoff water, tile drain water, or digester effluent may need to have 4 to 6 $\log_{10}$ orders of organism reduction to meet water quality standards.

Olsen and Larsen, 1987, identified bacterial decimation times of several pathogenic organisms in meso and thermophilic anaerobic digesters. Their results are shown in Table 14. The important factors were the species of bacteria and temperature, but not the source of manure, reactor process (batch or continuous), gas produced, ammonia content, or pH.

Combining management practices has been shown to accomplish greater reductions of pathogenic organisms than single practices. Among the practices tested are multicell lagoons with constructed wetlands (Ibekwe *et al.*, 2002), multicell lagoons followed by constructed wetlands (Sobsey and Hill, 2002), solid separation prior to wetlands (Hill *et al.*,1999), lagoons followed by constructed wetlands followed by infiltration basins (Lorimor *et al.*, 2003), solids separation followed by composting of solids and treatment of the water, digesters followed by constructed wetlands (Bicudo and Goyal, 2003), animal diet manipulation to reduce pathogen

Table 13. Microorganism inactivation by different management techniques.

Etiologic Agent	Composting	Anaerobic	Aerobic	Solar	Diet	Separation	Reference
Bacteria							
Campylobacter spp.	T[†]		T			X[§]	Olson, 2003; Hutchinson *et al.*, 2005
E. coli	T	T, M[*]	T	X	X	X	Olson, 2003; Davies-Colley *et al.*, 1999; Collins *et al.*, 2005; McCaskey *et al.*, 1998; Martin *et al.*, 2003; Shaw *et al.*, 2004; Hutchinson *et al.*, 2005; Schamberger *et al.*, 2004; Wright *et al.*, 2003
	T						
Listeria spp.	T	T	T			X	
Salmonella spp.	T	T	T		X	X	Losinger, 1995
Yersinia enterocolitica	T	T	T				
Protozoan Parasites							
Cryptosporidium	T	T	T	X		X	Olson, 2003; Mendez-Hermida, 2005; Whitman *et al.*, 2004
Giardia	T	T	T				Olson, 2003
Viruses	T	T, X					Monteith *et al.*, 1986

[†] T = Thermophilic process, yields virtually complete reduction of pathogens
[§] X = Approximately 90% reduction
[*] M= Mesophilic process, yielded approximately 99.9% reduction

Table 14. Bacterial decimation times in anaerobic digesters. †

Etiologic Agent	Decimation Time	
	Days at 35 ºC	Hours at 53 ºC
Erysipelothrix rhusiopathiae	1.8	1.2
Escherichia coli	1.8	0.4
Salmonella dublin	2.0	0.6
Salmonella typhimurium	2.4	0.7
Staphylococcus aureus	0.9	0.5
Streptococcus faecalis	2.0	1.0
Group D streptococci	7.1	
Fecal coliforms	3.2	
Total coliforms	3.1	
Clostridium perfringens (spores) §	Not inactivated	Not inactivated
Bacillus cereus (spores)	Not inactivated	Not Inactivated

† Adapted from Olsen and Larsen, 1987

§ C. perfringens was still present after 300 days at 35ºC and 180 days at 53ºC

loads, and separation of animals into different buildings at susceptible life stages (Shaw *et al.*, 2004). North Carolina State University has compared conventional lagoons followed by sprayfields with solids separation followed by a constructed wetland. The second treatment system reduced coliforms and *E. coli* by 3 to 4 $\log_{10}$, while the first reduced coliforms and *E. coli* by 1 to 2 $\log_{10}$. Multicell lagoons have been shown to reduce potential pathogens in wastes by about 99 % in each cell. With two or three cells in series, microbial populations can be reduced to acceptable water quality levels. New York City has published a multiple barrier approach to protecting source water watersheds (New York City and the Watershed Agricultural Council, 1996). The approach starts with good animal husbandry, including herd health, separation of age groups, sanitation improvements, and crop system changes. Szostakowska *et al.*, 2004 reported on the presence of *C. parvum* and *G. lamblia* in cattle barn flies and landfill flies. The flies from the barn had a greater load of infective cysts than the flies from the landfill. These studies show the importance of controlling the spread of pathogenic organisms in the environment by nonagricultural vectors. The second stage is improving barnyards, manure handling, application timing, soil management, and composting. The third stage improves stream corridors, adds vegetated buffer strips, adds stream crossings for pastured animals, fences animals away from streams, and adds watering stations remote from streams. Milne (1976) showed that livestock in proximity to a stream increased the nutrient and organism load in the stream. Fecal coliforms and fecal streptococci were found in the stream above water quality standards. Jellison *et al.*, 2002 found major sources of *Cryptosporidium* spp. in a watershed to be from wildlife and cattle. *C. parvum* was found in cattle and deer. One example of a method of runoff control is simply fencing livestock away from streams (Line *et al.*, 2000; Owens *et al.*, 1996). In both cases, fencing cattle away from streams led to significant reductions of nutrients and sediments entering the streams. Owens *et al.*, 1996 also showed that 1% of storm flow accounted for 27% of the sediment losses. Peak losses were in May and June.

9.2 Disussion

Conventional manure management techniques do reduce the populations of pathogenic microorganisms. The extent of the reduction for most techniques is on the order of 90-99%. The pathogenic organisms originate in the digestive tracts of warm blooded animals, so it is not surprising that conditions in the open environment are inimical to their survival. Important factors in organism survival are nutrient content of the waste, organic content of the waste, temperature, and the species of microorganism. Organic acids, ammonia, and pH changes can also act to reduce the survival of microbial populations. Competition and predation can also affect the population of pathogens in waste materials. Once the manures are spread on fields, microbes can move with air, surface and subsurface flow. VBSs will retain large fractions of microbial populations, but not enough to allow discharge into receiving streams. Solar radiation and timing of manure application with regard to rainfall have a significant effect on microorganism survival. Soil management techniques are also important in planning for reduction of pathogen reduction. Surface application of manure will take advantage of solar radiation as a disinfection technique, but ammonia losses to the atmosphere may be increased. Injection of liquid waste will retain ammonia as a fertilizer, but pathogen survival is enhanced when organisms are protected from drying and the effects of sunlight. The most effective methods for manure management that also control pathogenic organisms are composting and thermophilic digestion. In both cases, the temperature of the process is adequate to destroy many pathogenic organisms. Composting is probably the least costly process to use. It does require attention to solids content, moisture content, and C: N ratios. Properly done, composting can yield a value-added product that can be marketed to the public. This does require development of a marketing plan. High temperature digestion will also destroy pathogens. Under anaerobic conditions, methane can be recovered and used to offset the cost of building and running the digester. High temperature aerobic digesters destroy pathogens and reduce the carbon mass that must be handled, but operational costs are likely to be high and are hard to justify.

The best methods for reducing pathogen loads in manures will combine more than one management tool to achieve reduction of microbial populations to levels that will meet water quality standards. One example of a combined management process is separation of solids from the waste stream, composting of the solids, and digestion of the liquid portion followed by a constructed wetland. This process would reduce pathogen loads in the waste so that land application of either the solid or liquid phase would pose little or no risk to receiving waters. Another example of a combined treatment system is an anaerobic digester, followed by solids separation and constructed wetlands for treatment of the liquid phase of the waste. Multicell lagoons, followed by constructed wetlands, are another form of combined systems. Wastes should not be applied to land unless two or three management/treatment steps are used before land application. VBS should also be present at the edges of the fields used for application of the wastes. Waste management will include management of the animals in terms of diet and housing sanitation. Animal producers will need to work with agricultural researchers and planners to reduce the dissemination of pathogenic microorganisms from their facilities. Achieving these reductions need not require high cost technologies. In some cases, the reductions can be achieved by modifying existing facilities and perhaps adding additional processes, such as wetlands. Adding baffles to lagoons to increase the length of the flow path is one such modest cost option.

Changing the manure management process at CAFOs will require that each facility examine its manure handling process and look for ways to incorporate more steps that reduce pathogen loads. The USDA Natural Resources Conservation Service publishes conservation standard practices that can be applied to manure management problems (http://www.nrcs.usda.gov/technical/Standards/). Collaboration between producers, conservation officers, and environmental advisors can lead to great improvements in the handling of animal manures in the U.S.

10. Ongoing research at the EPA and Other Federal Agencies

The presence of CAFOs and the associated wastes are a topic of interest for several agencies of the U.S. government. Natural resources in the U.S. that are potentially affected by the presence of CAFOs and the attendant manure include, but are not limited to the air, water, and soils. In particular, the United States Department of Agriculture (USDA) has a significant interest and commitment to animal issues, both from a production and an environmental perspective. Within the USDA, two organizations have a dominant interest in manure-related research including the Agricultural Research Service (ARS) and the Cooperative State Research, Education, and Extension Service (CSREES). ARS has a formal research area known as National Program 206 (NP206) that is tasked with manure related research with a wide array of topics of interest to the EPA. CSREES sponsors research by funding of universities and other organizations. Research goals encompass atmospheric emissions, nutrient management, pathogens, and pharmaceutically active chemicals, and byproducts. All phases of animal production will enter into aspects of manure management from feed formulas to field application of manure. In addition, the Natural Resources Conservation Service (NRCS) funds implementation of conservation practices through a variety of programs. NRCS does not conduct research directly. In the conservation activities of NRCS, there are several different programs established to help livestock producers improve the environmental performance of their operations. These programs are either cost-sharing or outright grants to help producers mitigate environmental impacts of livestock operations. One of the larger programs is the Environmental Quality Incentive Program (EQIP) that provides low interest loans and cost sharing to producers that install conservation practices on their property. The EQIP funding level was about $1.5 billion in Fiscal Year 2005 (http://www.nrcs.usda.gov/programs/2005_allocations/2005_allocations.html).

Another agency that conducts research into microbial problems related to CAFOs is the United States Geological Survey (USGS). The USGS conducts water quality research across the U.S. and complements USDA research in several ways. USGS emphasizes assessment of water bodies and the impacts of animal waste in karst terrain and ground and surface waters. USGS tends to place its research in a watershed context while USDA tends to place its research into a location specific context. Both approaches are needed to fully address the impact of animal waste on water resources in the U.S. See Table 15 for a listing of some current microbiological research carried out by USGS.

The Centers for Disease Control (CDC) has conducted research on the public health effects of animal waste in the environment. The areas of interest to CDC include antibiotic resistance, bacterial populations, and nitrates in water. The research conducted by CDC has been severely limited due to budget cutbacks. There is an important need for epidemiological analysis of microorganisms originating from animal waste to determine if human health is at significant risk. Waterborne disease is believed to be greatly under-reported (Morris and Levin, 1996; American Society for Microbiology, 1998). Many cases of gastroenteric illness that could be attributed to contaminated water are likely to go unreported by people because they simply do not associate swimming in streams, lakes, and ponds with the onset of symptoms. It is more likely for people to assume that a gastroenteric illness was associated with foods consumed during recreational activities.

Table 15. Studies carried out or in progress in the United States Geological Survey

Location	Media [†]	Analytes	Waste Type	Observations
Missouri	GW	Nutrients, Fecal Bacteria	Poultry	Wells showed contamination with no history of manure in the area
Delaware	SW		Poultry	
Arkansas, California, Missouri, Colorado, Virginia	SW	Bacteria, Viruses, Protozoa, Nutrients	Various	Presence in streams, fate and transport, methods, isotopes
Arkansas	GW	Nitrate, Bacteria		Effects in karst terrain, spring resurgences, nitrate from septic systems, highest bacteria in initial flow
Not identified	Feed	*E. coli*, *Salmonella*	Cattle	Resistance development, cost analysis
Not identified	GW, SW	Viruses	Swine	Survey of waste from hog to stream, survival
Iowa	GW, SW	Nutrients, Bacteria	Swine	Effect of CAFO on GW,SW
Missouri	SW	Bacteria	Poultry	Coliforms in Shoal creek watershed
Not identified	GW, SW	Antibiotics		Methods, presence of antibiotics in water
Iowa	SW	Antibiotics		Survey of streams for antibiotics
Florida	GW	Nutrients, Bacteria	Dairy	Survey of wells for bacteria and nitrate, downstream had elevated levels of both.
Central Appalachia	GW	Bacteria	Dairy	Land use effects, seasonal variation, soil-water effects, BMP effects
New Mexico	GW	Nitrate	Dairy	94 dairies surveyed, high nitrate found at many
Missouri	GW	Viruses, Bacteria	Various	Survey of wells, few had contaminants, positives in areas of high agriculture
Michigan	GW, SW	Bacteria	Various	Models currently inadequate to describe transport, no well contamination
Five States, eastern US	GW	Nitrate, Phosphorus		Permeability versus runoff
Not identified	GW, SW	Nutrients, Bacteria	Various	
Michigan	SW	Bacteria	Wild Birds	Birds were dominant sources, antibiotic resistance patterns, rainfall increased counts in 48-72 hours depending on wind, collection time affected counts.
Nebraska	SW, wetlands	Bacteria, Nutrients	Swine	Decline of contaminants through wetlands before a wildlife refuge.

[†] GW = Groundwater; SW = Surface water

The various agencies with interests in microorganisms associated with animal waste have a broad range of topics that they are pursuing. A key topic for all agencies is development and validation of methods for the identification and enumeration of potential pathogens in the different media associated with manure in the environment. The methods for total and fecal coliforms and enterococci are mature, but largely limited to water. There are few methods suitable for a broad range of media such as soil, sediment, lagoons, manure, and water. New methods need to be developed especially to identify overt pathogens in different media. Assuming that a given method is credible, the survival and transport of microorganisms in the environment becomes the next major topic of research. Currently, there is limited research on the movement of microorganisms in the environment. Do they move with soil particulates? What affects movement of organisms in the environment? Are they independent of soil? How long do different organisms survive in the environment? What are safe limits of the different organisms in recreational waters? Beyond the questions raised here are larger questions of how to control the content of microorganisms in animal waste. What effect do animal diets have on microbial populations in the manure? What effect does manure storage have on pathogen populations? Is pathogen regrowth a significant problem?

In addition to development of methods for enumerating pathogenic organisms in environmental samples, there are several other common topics of interest to USDA, USGS, CDC, and EPA. These topics include survival and transport of organisms in the environment, source identification and tracking, and antibiotic resistance characteristics of the microorganisms. Do tile drain lines enable transport of bacteria into receiving waters? What effects do different soil types have on microorganism survival? What effect does timing of manure application to soil have on microbial populations? What is the effect of solar radiation on bacterial survival? What effect does rainfall have on transport of microorganisms from application sites to nearby streams? What effects do different Best Management Practices (BMPs) have on the movement of microorganisms in the environment? BMPs commonly include vegetated buffer strips, constructed wetlands, runoff retention basins, infiltration basins, terracing, injection of waste into the soil rather than broadcast application, and more. The development of models of microbial behavior in the environment is a topic of interest to the different Agencies because good models can help to conserve resources and assist in planning for Total Daily Maximum Load (TMDL) implementation and assessing plans for placement of new animal operations on the landscape.

Research carried out by the different agencies is performed on many different scales. Laboratory scale studies are performed to develop new detection and enumeration methods, measure movement of microorganisms through small soil columns under controlled conditions, develop source-tracking techniques, and evaluate small model digester performance. The work done to develop new methods for detecting and estimating microbial populations uses several approaches: 1) Culture-based methods are being refined to be more selective and to reduce the number of steps or time required to provide data for analysis; 2) antimicrobial compound resistance patterns are used as one approach to identify organisms originating either from humans or animals; 3) genetic analysis techniques are also being developed to discriminate human from nonhuman isolates to help identify sources of organisms in water. Another goal is to develop robust methods that can detect and estimate the population of specific organisms in the environment and track them to their source. The benefit of source identification is that corrective actions can be focused more effectively on specific problems rather than being applied to a broad area. Laboratory-scale work is also done to conduct analyses of samples from different environments using legally standard methods. Beyond the laboratory, field experiments are conducted at plot-scale, field-scale and watershed-scale. The plot and field-scale experiments usually evaluate the effects of application rate, application timing, rainfall effects on transport, and survival of microorganisms in the field. Also included at this scale are effects of vegetated buffers, wetlands, and other field management practices that can impact the transport and fate of microorganisms. Watershed-scale examination of effects of CAFOs on microbial populations in waters is perhaps the most difficult to carry out. Examination of waters for populations of total or fecal coliforms only reveals if the waters meet standards or not. The current methods simply do not allow for estimation of the contribution of human versus other animal inputs. Similarly, the inputs of wildlife cannot be separated from domestic animals or humans using total or fecal coliform methods. Consequently, installation of management practices at one CAFO may reduce that facility's input to the stream, but have little effect on the total load of microorganisms. Much work needs to be done to adequately model microorganism behavior in the environment and to identify critical control points. A means for separating the total microbial load into its important components needs to be developed and validated to enable estimation of the maximum load for individual water bodies. Associated with this work is a need to estimate or identify the health risk of fecal organisms in water.

The majority of the research conducted by USDA has dealt with the control and retention of plant nutrients in manures. The nitrogen and phosphorus content of manure represents a valuable resource for fertilization of agricultural land. Similarly, the organic matter content of manure is a valuable soil conditioner. In recent years, the ARS has added a significant amount of research on the microbial content of manures with regard to the presence of pathogenic microorganisms, the transport of organisms in the environment, and the survival of organisms in the environment (Table 16). When farms were smaller and had small numbers of livestock present, the manure produced was largely used as a soil amendment and fertilizer. In most cases, the small farm manure load would have been indistinguishable from the background of wildlife sources. The advent of large animal production units has altered the quantity of manure generated in small land areas. It is common now to have poultry houses with over 100,000 birds and swine houses with more than 1,000 animals in a building. Beef feedlots and dairy facilities can also have very large numbers of animals present. The amount of land available within economical transport distances for application of manure is also limited. The result is that too much manure is applied to too little land, leading to the possibility for serious runoff losses of nutrients and potential pathogens.

The importance of understanding microbial behavior in the environment cannot be overestimated. The health of humans and animals can be seriously affected by microorganisms that are commonly present in manures. If one farm has animals that are shedding pathogens in their manure, that manure can be a source of infection for other farms, recreational water users, and possibly municipal water supplies downstream of the farm. The true risks remain largely unknown because there is little information on the presence and survival of pathogens in animal waste after it enters the environment. USDA research on the microorganisms in manure is addressing this concern in studies from laboratory to entire watershed studies. The indicator organisms (coliforms and Enterococci) are useful for screening of waters for the presence of fecal contamination, but are limited in revealing the presence of pathogens.

Table 16. Studies carried out or in progress by the United States Department of Agriculture, National Program 206 †

Location	Media §	Analytes	Waste Type	Observations
Georgia	SW, Soil	Bacteria	Poultry	Survival and transport of pathogens
North Carolina	SW	Nutrients, Bacteria	Swine	Advanced waste treatment system evaluation
	SW	Bacteria	Cattle	Runoff content of bacteria, vegetative treatment system
Virginia	SW	Viruses	Dairy	
Maryland	SW	*Cryptosporidium parvum*		Modeling, runoff, transport
Chesapeake	SW	Bacteria, Protozoa		
Illinois	SW	Nutrients, Bacteria		Integrated waste systems
Wisconsin	SW	Bacteria	Dairy	
California	GW	Nitrate, Bacteria	Dairy	Dairy lagoon water site
Idaho	SW	Nutrients, Bacteria	Swine, Poultry, Fish	Gases, PM2.5, management effects, percolation
Texas	SW	Bacteria, Protozoa	Cattle	Best Management Practice effects, method recovery efficiency
Texas	SW	Nutrients, Bacteria	Cattle	Commercial additive effects
Texas	SW	Nutrients, Bacteria	Poultry, Swine	Transport in soil columns
Kentucky	SW	Bacteria	Swine	Survival of pathogens
Kentucky	GW, SW	Bacteria	Swine	Runoff, antibiotics, treatment methods
Idaho	SW	Bacteria		Transport, riparian buffer effects, at different scales
Maryland	SW	Bacteria		BMP effectiveness
Pennsylvania	SW	Nutrients, Bacteria	Various	BMP placement, stream processes
Iowa	SW	BMP Effectiveness	Various	BMP effectiveness, crop effects
Iowa	SW	Iowa		Tile water, soil effects, survival
Colorado	GW	Nutrients, Bacteria	Human	
	SW	Nutrients, Bacteria	Dairy	Pond effects on removal
Mississippi	SW, Wetlands	Nutrients, Bacteria		Pollutant removal at edge of field
Virginia	SW	Nutrients, Bacteria		Methods, source ID, *E. coli* O157 prevalence

† There are also several projects not associated with a specific state that are examining fate and transport of microorganisms in the environment. These projects also examine the factors affecting microorganisms and their movement. Pathogen identification, antibiotic resistance, modeling, composting, wetlands, management practices, and animal diet effects are among the research topics.

§ GW = Groundwater; SW = Surface water

Table 17. Studies carried out or in progress by USDA or cooperating Universities listed in the CRIS database †

Location	Media §	Analytes	Waste Type	Observations
Texas	SW, sediments	*E. coli*, *Salmonella*	Cattle	5 to 7 types of *E. coli* dominate each group.
North Carolina	SW, constructed wetlands	Nitrification	Swine	
Georgia	SW, riparian buffers	Bacteria	Swine, Poultry	Buffers can be effective in removing bacteria
Louisiana	SW	Bacteria	Dairy	*E. coli declines with time after application*.
California	Multiple	Bacteria	Dairy	*E. coli* can survive 45 days after application
California	SW	*E. coli* O157:H7		Pathogen transport
California	Air, SW	Protozoa, Bacteria	Dairy	Protozoans increased, bacteria decreased after application
California	Food surfaces	Bacteria		Method development to measure populations
Colorado	Manure piles, compost	Bacteria, Antibiotic resistance	Horse, cattle, poultry	
Georgia	SW, riparian buffers	Bacteria	Dairy, swine, alligator, poultry	Buffers are effective with swine waste, upland cropping was effective with poultry and dairy waste
Georgia	SW	Bacteria, nutrients	Dairy, Swine, Poultry	Buffers alone are not adequate, multiple cropping and forest help limit loads.
Georgia	SW	Bacteria	Poultry	Composting, UV, Chemical treatment effects on survival
Georgia	SW, GW, soil	Bacteria, Hormones, Protozoa	Poultry	E. coli not best source tracking organism, protozoa can penetrate soil to depth, small ponds can reduce organism load, tillage, temperature, texture were important
Georgia	SW, soil	Bacteria, Hormones	Poultry	Watershed, landscape scale, methods, filtering by plants
Georgia	Compost	Bacteria	Various	Compost has to be well managed to reduce pathogen levels
Hawaii	SW	Bacteria	Various	Multiple scales, bacterial reduction
Idaho	SW	Bacteria	Various	Use of flocculants as a treatment
Idaho	SW	Bacteria	Various	Diet modification effects
Idaho	SW	Bacteria	Various	Landuse and coliform levels
Illinois	Various	Bacteria	Swine	Feed and odor, antibiotic resistance

† Some of the entries may duplicate entries in Table 2.
§ GW = Groundwater; SW = Surface water

Table 17. Studies carried out or in progress by USDA or cooperating Universities listed in the CRIS database (continued) †

Location	Media §	Analytes	Waste Type	Observations
Indiana	Tile drain	Bacteria	Various	DOC and pathogen transport, Effect of manure on bacterial survival
Iowa	SW	Bacteria	Swine	Bacteria at different places in waste streams, diet effects
Iowa	Soil, manure	Bacteria	Swine	Control strategies, native community effects on manure bacteria
Kentucky	Multiple		Various	Waste management in karst areas
Louisiana	SW	Bacteria	Dairy	Differentiation of sources
Maryland	SW	Bacteria	Dairy, swine	Multiple research areas to reduce bacteria and recover value from manure
Maryland	Milk	Bacteria, Viruses	Dairy	3 to 8 % of milk tanks had contamination
Maryland	Water, air, manure	Bacteria	Dairy, beef	Land use and buffers affect organisms, methods, source ID
Maryland	SW	*E. coli* O157, *C. parvum*	Dairy, beef	O157 is more diverse than previously known, DOC enables percolation of pathogens, urban water has greater E. coli, oysters can be 90% contaminated
Maryland	SW	Bacteria, Nutrients	Dairy	Algal treatment of dairy waste retained nutrients
Maryland	Soils	Bacteria		Manure particles reduce attachment and enable percolation of bacteria
Minnesota, Wisconsin	Soils	Bacteria, Antibiotics	Beef, swine, turkey	Tillage, soil type had large effects on resistance, transport
Mississippi	Soil	Bacteria	Poultry, swine	Feeding study, methods, survival, phage control
Nebraska	SW	Bacteria	Beef	Runoff control, compost, vegetative treatment area
Nebraska	SW, sediment	Bacteria, protozoa, Phage	Beef	Survival, wetland, runoff, methods
New York	SW	*C. parvum*	Various	Transport models, vegetation, soil type, slope, management practices
North Carolina	SW	Bacteria	Swine, poultry	Diet, new waste systems, survival after treatment and application
North Carolina	Wetlands	Nutrients, Metals, Bacteria	Swine	Continuous marsh reduced nutrients better than other patterns, water depth was important, solid liquid separation

† Some of the entries may duplicate entries in Table 2.
§ GW = Groundwater; SW = Surface water

Table 17. Studies carried out or in progress by USDA or cooperating Universities listed in the CRIS database (continued) †

Location	Media §	Analytes	Waste Type	Observations
North Carolina	Various	Nutrients, Bacteria	Swine	Nitrification, denitrification, phosphorus recovery
Oklahoma	Soil	Bacteria, Metals		
Pennsylvania	SW, soil	Bacteria		Management practice effects, wetlands, hydrogen production
South Carolina	Wastewater	Nutrients		Nitrification denitrification
South Carolina	Wastewater, wetlands,	Nutrients, Bacteria	Swine	Waste treated with different materials and practices for the recovery of nutrients and reduction of pathogens.
Texas		Bacteria		Develop phage as a bacterial control technology for waste
Texas	Soil. Irrigation water	Bacteria, Protozoa,		Multiple aspect study examining many aspects of animal waste in the environment.

† Some of the entries may duplicate entries in Table 2.
§ GW = Groundwater; SW = Surface water

Considering that microorganisms originating in animal waste represent a significant risk to people and animals, methods to reduce the microbial load of waste are important. There is a great need to develop manure management procedures that will reduce the load of microorganisms before waste is allowed to enter the open environment. Anaerobic digestion is one technique with promise to be cost neutral or beneficial due to the use of generated methane as a fuel source. Aerobic digestion is a net cost, but reduces odors and microorganisms. Composting reduces odors and microorganisms and produces a potentially salable product. Composting may be practical if markets can be developed. There are other approaches that generate activated carbon, pelletized fertilizers and other products. Combinations of waste management methods may also be used to reduce microorganism loads before waste disposal. Storage of wastes for six months has shown reduction of bacterial populations. Storage in concert with another management practice may be able to reduce loads of organisms to the point where application to land would pose little fecal load runoff potential. The important factor is that any treatment approach has to be economically feasible in comparison to existing manure management practices.

A key task to be completed is integration of the various government agency research activities. The benefit of integration will be to maximize efficiency of planned research by expanding the scope of work, avoiding duplication of effort, and sharing of information across interest groups. EPA is establishing a scientist to scientist level series of workgroups with the goal of integrating work across agencies. Other goals include enabling scientists to participate in larger projects than any individual could manage alone and prepare documents that are useful to producers at the farm level for implementation of environmentally sound practices. Collaboration with the USDA and Extension services will facilitate these goals.

11. Summary and Outstanding Issues

Bacteria, viruses, and parasites that can cause disease in humans are endemic in livestock animals. The confinement of animals into densely-populated feeding operations exacerbates the spread of disease and encourages the use of antimicrobial agents for both prophylaxis and to increase animal growth rates, resulting in the emergence of antimicrobial-resistant bacteria. These zoonotic pathogens may proliferate in confinement houses and are shed in animal wastes that, in most cases, are stored and eventually spread onto land. Exposure to antimicrobial-resistant bacteria and other zoonotic pathogens may occur through direct contact with livestock animals, breathing confinement house air, contact with insect and animal vectors, recreational or drinking waters contaminated with manure runoff or leaking manure storage pits, eating produce from manure fertilized fields, and secondary infection from exposed individuals. Several mechanisms are in place to prevent the spread of disease from livestock animals to humans and may include animal stocking techniques, animal waste treatment practices to destroy pathogens (such as composting and thermophilic anaerobic digestion), storage of animal manure to reduce pathogen concentrations prior to spreading, barriers (such as wetlands and buffer strips) to control runoff from manured fields, and surveillance of our nation's food and waters for pathogenic organisms. However, from reported literature, it is clear that exposure to zoonotic pathogens cause significant human suffering and economic losses in the billions of dollars annually due to lost productivity, treatment of disease, and beach closures. Because of the continuing human disease caused by zoonoses contaminating food and water resources in the U.S., we believe that the current environmental regulations and conventional animal manure management practices are inadequate for protection of human health and the environment.

The USEPA and other governmental entities including the USDA, USGS, and the CDC are actively working towards resolving the threat to human health and welfare posed by antimicrobial-resistant bacteria and other zoonotic pathogens that may be released into the environment from CAFOs. As can be seen in this review, the outstanding issues regarding the fate and transport of zoonotic pathogens are vast; addressing these issues will require the expertise of all of these agencies and the many disciplines they represent. Of particular concern is the synthesis of information generated in these studies into a comprehensive and usable package, so that resources can be pooled to arrive at a more complete and usable plan.

Much work is still needed to fully address issues surrounding the contamination of our environment and with antimicrobial-resistant bacteria and zoonotic pathogens originating from livestock animals. Based on our review, we recommend that the pathway forward involve not only value-added research, but also policy changes that are consistent with current limitations on the use of human waste biosolids as fertilizers.

11.1 General recommendations

Animal agriculture produces copious amounts of manure, most of which is stored untreated and spread onto land. Based on available manure management technologies, ensuring the safety of food crops and water resources will require active treatment practices that greatly reduce or eliminate pathogens in manures and other CAFO wastes prior to land application or discharge to natural waters. At present, animal manures applied to land as a fertilizer are not regulated in terms of pathogen reduction. This lack of regulation is at odds with requirements for the application of biosolids originating from human septage (USEPA, 2003). Consider:

- Even moderately-sized concentrated animal feed operations, such as a 2,500 dairy cattle operation, may produce as much manure as a city of 61,000 people. Serious fines for environmental pollution and lawsuits would result if a city of that size spread all of its sewage onto land without treatment.
- Animal manures and other animal wastes may contain high concentrations of pathogens, hormones, antimicrobials and other pharmaceutically active compounds, metals, nutrients, and other chemicals, similar to human sewage.
- Animal manures can be as much as 100 times more concentrated than human sewage, as human wastes are diluted with other domestic wastewaters prior to treatment.
- Because of their concentrated form, animal manures have a higher demand for oxygen, higher nutrient content, and higher concentration of pathogens than human septage on a per weight basis.
- Every year animals raised in CAFOs produce three times as much manure as humans in the U.S.

Regulatory bodies should carefully weigh the full costs associated with zoonotic disease, which are estimated to reach into the billions annually, when considering difficult decisions regarding the regulation of livestock animal wastes. Several cost effective options for animal waste treatment can be implemented at CAFO facilities that would reduce pathogens to safe levels prior to application as a fertilizer. The most effective and cost-efficient methods for achieving these ends may be composting or thermophilic anaerobic digestion with recovery of methane that can be used as a fuel. However, circumstances specific to each animal confinement facility would need to be considered when choosing appropriate manure treatment systems. These active treatment systems should be used in concert with management practices to reduce pollution of water bodies by treated manure fertilizers, such as vegetative filter strips, terraced landscapes, and constructed wetlands.

Of great concern is the continued use of antimicrobials in animal agriculture for growth promotion and prophylaxis. Many of the drugs used to promote growth in animal agriculture are the same as or very similar to human medicines, and result in the shedding of high concentrations of antimicrobial resistant bacteria that may infect humans and other animals. Antimicrobial resistant zoonotic pathogens are a serious threat to human health (Ghidán *et al.*, 2000; Cheng *et al.*, 2002; Travers and Barza, 2002), and billions of dollars are spent in the U.S. every year treating diseases resistant to antimicrobials and managing the spread of resistance in hospital environments. The benefits of growth promotion in livestock animals are certain, and at present, difficult to offset completely with market alternatives (Harper, 2004; Gill, 2005). However, a combination of education of owner/operators, alternative feed additives, and improved and more sanitary animal husbandry practices are promising for achieving this end (Gill, 2005). Regulatory agencies should fully weigh the costs and benefits of continued use of antimicrobial compounds in animal agriculture for growth promotion and consider the phased removal of these feed additives from the market in favor of alternative technologies. Tighter regulation of the use of antimicrobial compounds for prophylaxis should also be considered.

11.2 Recommendations for Future Research

Significant progress has been made to date to address the release and movement of microorganisms from CAFOs and fields fertilized with their manure byproducts. Research has ranged from bench studies on pathogen survival to investigations of specific management practices for impeding the movement of fecal indicator bacteria to receiving waters and specific surveys of pathogens and antimicrobial-resistant bacteria near CAFO facilities. Current research is exploring new and innovative ways to detect and quantify pathogens in soils, manures, and natural waters that are enabling more specific characterization of animal waste management practices and technologies performance. These techniques have also opened the door for development of improved monitoring and surveillance systems that may revolutionize the way we look at water quality. Some of these new technologies are progressing rapidly towards the end of being able to identify with great accuracy the source of pathogenic agents in recreational and drinking water resources that may cause disease. Other advances are being made in the development of cost efficient and reliable livestock animal waste treatment technologies that may ultimately reduce the burden of zoonotic disease in the U.S.

Although advances are being made, significant amounts of work are still required to fully address the issues surrounding antimicrobial resistant bacteria and other zoonotic pathogens from CAFO facilities. There is a need for fundamental information on specific etiological agents pertinent to their movement and inactivation in manures, soils and sediments, and natural waters. There remain questions as to what levels of these agents are acceptable in natural systems such that the risk of contracting disease upon accidental exposure is low. New models that can predict with accuracy the fate and transport of pathogens in the environment following the application of manure fertilizers to land are needed to identify potential control points to locate new operations safely and in a sustainable manner. In addition, integrated systems that can monitor our nation's water resources in real-time for threats that may be posed by zoonoses and other biological agents are needed to improve biosecurity. All of these research needs are integral to improving human health and welfare in the U.S., especially in areas of intensive livestock farming. The following is a top ten list of research needs to address the pathogen issue and reach this goal.

1. *There is a need for standardized methods of analysis for zoonotic pathogens in animal manure, soil and sediments, wastewater, recreational water, and drinking water.*

    Standard methods with the required sensitivity for recovering and enumerating pathogens at environmentally relevant concentrations in animal manures, soils, wastewaters, recreational water, and drinking water are sorely lacking, especially for hyper-endemic or emerging pathogens. These methods are needed to (a) identify the extent to which these agents are removed, inactivated, or persist in animal waste treatment processes and management systems at livestock operations, (b) determine the survival of these agents in manures, soils, sediments, and natural waters to improve our ability to predict their fate and transport in the environment, and (c) improve surveillance and biosecurity of our nation's recreational and drinking water resources.

2. *There is a need for rapid methods of analysis for pathogens in recreational and drinking water to improve surveillance and biosecurity of our nation's water resources.*

   Microbiological water quality surveillance in the U.S. relies on the detection of bacterial indicators of fecal pollution. Although epidemiologically related to gastrointestinal illness, these indicators do not fully describe the risks associated with recreational or drinking waters contaminated with some bacteria and most viruses and parasites. Furthermore, since conventional cultivation methods take 18-24 hours to yield a presumptive-positive result, a positive result today means that everyone drinking the water or swimming in it yesterday was exposed to unacceptable levels of fecal pollution.

   As such, there is a need to develop rapid and reliable methods for the detection of fecal bacterial indicators and overt pathogens in recreational and drinking waters. A tiered approach to rapid monitoring methods may be the most reasonable, starting with indicators and then adding pathogens as methods become available. It has been suggested that such technologies rely on nucleic acids analysis because the tests lend themselves to automation and are at present the most promising for rapid and specific quantitation of both fecal indicator organisms and viable microbial pathogens (Jothikumar *et al.*, 1998; Levin *et al.*, 2002; Straub and Chandler, 2003).

   Current technologies for rapid detection of the fecal bacterial indicators are being field tested against proven cultivation methods to develop guidelines for improving recreational water quality monitoring by the USEPA and CDC (USEPA, 2005). However, the near real-time detection of overt pathogens with very low infective doses, such as *E. coli* O157:H7, *Campylobacter jejuni*, and *Cryptosporidium* will require significant advances in technologies to concentrate these agents from large volumes of potentially dirty water. In particular, there is a need for effective and reliable sample concentration technologies capable of co-concentrating viruses, bacteria, and parasites into clean samples amenable to detection with nucleic acids technologies. At present, hollow fiber ultrafiltration systems may be the most promising to this end. However, the retention of a variety of pathogens from different waters by these filters needs testing and validation.

   Ultimately, the development of a unified and automated system for the detection of all waterborne pathogens is needed (Straub and Chandler, 2003). Hollow fiber ultrafiltration devices, renewable surface technologies for automated sample processing, and microarray technologies have shown promise as a basis of such a system (Chandler *et al.*, 2000a,b). However, such a system should be constructed in a way that it is simple, reliable, and technically less demanding than current nucleic acids technologies. The need for rapid pathogen detection technologies will remain a critical issue for biodefense where real-time identification of etiological agents may be imperative to protecting human health.

3. *There is a need for epidemiological data to establish regulatory guidelines for pathogens in manure, wastewater, recreational water, and drinking water.*

   Regulatory guidelines on the concentrations of pathogens in the manure, wastewater, recreational water, and drinking water protective of human health do not exist because (a) there is a lack epidemiological data to ascertain the risks of illness associated with exposure, and (b) there remain questions as to what level of risk is acceptable. There remains a need for epidemiological data to enable the identification of appropriate and safe limits of pathogens in manure, drinking water, recreational water, and in food. Based on surveillance of water and foodborne outbreaks in the U.S., priority should be given to *Salmonella* spp., *Campylobacter jejuni*, *E. coli* O157:H7, *Cryptosporidium*, *Giardia*, and viral agents such as swine hepatitis E virus.

4. *There is a need to identify inactivation kinetics of zoonotic pathogens in manures, soils, and environmental waters.*

   Relatively few studies are available describing the survival of zoonotic pathogens in environmental matrices, especially considering the broad range of properties of soils, manures, and waters that may potentially be contaminated. A significant limitation is the lack of information regarding the survival of antimicrobial-resistant bacteria in various *milieus*, including the persistence of phenotypic and genotypic antimicrobial-resistance traits. Most studies on the survival of pathogens have been carried out in the laboratory instead of *in-situ*, and only a few examined more than one environmental stressor simultaneously. Accurate information regarding the survival of pathogenic zoonoses and antimicrobial resistant bacteria is necessary for modeling their fate and transport from CAFOs.

   Based on these limitations, the following needs have been identified:

   - Comprehensive studies that examine the combined effect of several stressors simultaneously on the survival of zoonotic pathogens and antimicrobial-resistant bacteria in manures, soils, and surface water sediments, and natural waters are needed. Stressors that should be considered include:
     - Biological factors, such as antagonism, competition, and predation;
     - Physical factors, such as temperature, soils and sediment properties, and solar radiation;
     - Growth factors, such as pH and availability of nutrients.
   - There is a need to identify the effect of the retention of some pathogens on soils and sediments on survival in various matrices.
   - There is a need for small-scale studies to determine the concentration of antimicrobial compounds needed for an organism to maintain antibiotic resistance and the number of growth cycles that lead to the loss of the resistance trait.

5. *There is a need for fundamental research to characterize the transport of zoonotic pathogens over land, through soils and ground water, and in surface water bodies.*

   The movement of antimicrobial-resistant bacteria and other zoonotic pathogens from animal wastes through the environment is a complex issue. Research is needed to address significant data gaps regarding the properties of etiological agents that may affect their retention or mobilization in soils and stream bed sediments. In order to better address the transport of pathogens in the environment, several needs must be met, including:

   - Characterization of the properties of zoonotic pathogens that may affect their fate and transport in the environment, which, if understood, would allow them to be incorporated into existing hydrologic and geogaphical information systems (GIS)-based transport models.
   - Identification of the particle sizes with which zoonotic pathogens may be transported in the environment.
   - Identification of the potential effects of soil and sediment retention of some pathogens on overland transport and resuspension in stream bed sediments.
   - Verification that batch and column studies performed in the laboratory to determine pathogen fate and transport properties accurately describe field observations.

6. *There is a need for research to characterize the movement of antimicrobial-resistant bacteria and other pathogenic zoonoses into the environment following land application of animal manures with particular attention paid to the effects of hydrologic (rainfall) events.*

   Information is lacking regarding the concentrations of antimicrobial-resistant bacteria and zoonotic pathogens in the environments proximal to CAFOs and fields where their manures are applied. However, on a larger scale, significant microbial contamination in agricultural watersheds has been observed by the USEPA. Rainfall has been noted to increase concentrations of fecal indicator bacteria in agricultural watersheds, and much of the outbreaks of waterborne disease in the U.S. and Canada have been linked to heavy rainfall events.

   Surveillance of pathogens and antimicrobial resistant bacteria near several CAFOs with different confinement animals and manure management practices is needed to ascertain the potential pathways for pathogen transport from manured fields. Monitoring plans should also consider sampling at the sub-watershed and watershed scales. Field studies are needed to identify the role of drainage tiles and overland transport of pathogens to receiving waters during rainfall events. Continuous or event-triggered sampling devices should be used so that events are not missed. Samples should be taken following manure application and 24 to 48 hours after a substantial rainfall. Future studies should also consider management records on the use of antimicrobials on each specific farm that may be helpful to correlate farm practices with findings obtained through the studies.

7. *There is a need for continued research into methods for tracking fecal pollution in natural waters to its source.*

   Much of our nation's water resources are impaired due to high concentrations of fecal microorganisms. In many instances, the source(s) of fecal contamination are unclear. MST is an emerging technology that identifies the animal origin of fecal bacterial pollution. However, many caveats to the use of MST still exist, and much work is needed to improve these technologies for more widespread application. Some of the data gaps for the various methods include:

   - Poor survival of MST reference organisms in the environment may result in little or no detection, limiting the ability to identify the source of fecal contamination. Reference organisms need to be chosen so that they are useful at a distance from the potential source.
   - Variability in the survival of different phenotypes or genotypes of MST reference organisms in environmental matrices that may lead to divergence from host-specific fingerprints in source libraries need to be clearly defined. Limitations to interpretation of MST results dependent on these findings need to be documented.
   - Variability in transport and survival of the index organism(s) used to identify fecal pollution source(s) and pathogens of the same source that may lead to misidentification of the source of disease. Studies are needed to ascertain whether or not reference organisms are reliable indicators of pathogen transport. It may be that several reference organisms are needed to describe the full suite of pathogens that may contaminate water bodies.
   - Variability in the transport properties of different index organisms needs to be clearly defined to enable interpretation of MST results.
   - Advances in MST technology need to be made to reduce the time of analysis, the level of expertise required, and the cost. Host-specific molecular biomarkers offer the most promise for achieving this end, but significant advances in their development must be achieved before they are off-the shelf ready.
   - MST techniques need more field-scale testing to prove their utility in varied circumstances. Full-scale watershed type studies are needed to assess the potential of these technologies for future use.
   - There is a need for different levels of analytical methods to address microorganism tracking from simple indicators to methods for exact pathogen and source identification.
   - Additional research is needed in the area of spatial and temporal variability for library-independent MST methods.

8. *Fundamental studies on the efficacy of various manure management practices including uncertainty in their performance are needed.*

   Many management practices have been proven effective for reducing the discharge of stressors such as nutrients and sediment runoff to surface waters. However, the efficacy of different management practices for impeding the movement of zoonotic pathogens and antimicrobial-resistant bacteria to receiving waters following land application of animal manures remains uncertain. Based on studies using fecal indicator organisms, these practices may reduce the discharge of pathogenic microorganisms. However, the reductions associated with most practices are only on the order of 90-99%, a scant number considering that animal manures may contain billions to trillions of bacteria, viruses, and parasites per gram. Therefore, although specific management practices such as vegetative buffer strips will retain large fractions of microbial populations, they will not retain them well enough to protect receiving streams from contamination.

   There is a need to identify the performance of common barrier technologies such as infiltration basins, wetlands, and buffer strips for the retention and inactivation of pathogenic organisms. Studies should address retention in the context of factors related to the design of the systems such as size, slope, solids or hydraulic residence time, vegetation, undercutting by tile drainage, etc. Studies are also needed to address the impacts of rainfall on management practice performance. Of particular interest is exploration of a multibarrier approach versus single barrier BMPs.

   Aside from barrier technologies, there is a need to verify and field test manure treatment technologies like anaerobic digestion, not only for pathogen reduction, but also to identify the potential for fuel recovery. These technologies should be compared and contrasted to conventional manure storage technologies in terms of stressor reduction and cost. Vector attraction reduction and pathogen regrowth in treated materials should also be explored.

   Many of these questions are being addressed in the areas of public wastewater treatment and biosolids from public treatment works. Analogies for manure treatment and runoff barrier technologies for pathogens, as well as vector attraction reduction, may be drawn from the extensive pool of research available within the biosolids community. However, livestock animal wastes tend to be more concentrated than human sewage; thus, treatment solutions for human wastes need to be feild-tested for application at CAFOs. The most readily applicable technologies may be those for pathogen reduction in biosolids, but liquid separation and treatment may need to be performed prior to application of solids treatment technologies.

9. *Models are needed to better predict site-specific optimal manure treatment technologies and runoff management practices for pathogen and other stressor reductions*

   Better models of microbial behavior in the environment are needed to assist in planning for TMDL implementations and assessing plans for placement of new animal operations on the landscape. Of particular interest would be lifecycle assessment models capable of analyzing the effects of different treatment technologies and management practices. Models should be capable of predicting potential outcomes regarding not only pathogens but other stressors such as nutrients and pharmaceutically active compounds. Best possible treatment and management practice combinations, as well as sustainable livestock populations based on environmental and human health outcomes, should be predicted considering uncertainty in the performance of the various treatment technologies and management practices. Models that integrate the fate and transport of antimicrobial-resistant bacteria and zoonotic pathogens may be different from present models in many ways. The issues of multi-drug resistance, microbial reservoirs, horizontal gene transfer of resistance determinants, and the ranges of infectious doses resulting from various host characteristics are not part of current models for chemical risk assessment. These factors need to be integrated into CAFO models. Further, particular attention should be given to the relationship between pathogens and organic matter, sediments, and nutrients, particularly in terms of survival and facilitated transport during hydrologic (rainfall) events. These models will ultimately need to be proven at the sub-watershed and watershed scales.

10. *There is a need to improve the coordination of research activities and dissemination of technical information, methodologies, and new technologies between research scientists of the various agencies and to a vast array of end users such as educators, regulators, and CAFO owners and operators.*

    There is a confounding level of technical literature relevant to pathogens and livestock animals dating back more than 100 years, and literature propagates at an astounding rate. Researchers in the fields of engineering, microbiology, agronomy, epidemiology and infectious diseases, as well as the geological sciences and others, are conducting a wide variety of studies on pathogens and/or fecal bacteria relevant to CAFO issues. Interpreting the literature is difficult not only due to the massive amounts of technical information available, but also due to the diverse nature of these disciplines. As such, there remains a need for better integration of the various government research activities. The benefit of integration would include (a) pooling of resources, (b) broadening of technical expertise, (c) maximizing efficiency, (d) expanding the scope of work that can be performed, (e) avoiding duplication of effort, and (f) sharing information across interest groups. Without significant interdisciplinary integration and cooperation, the assimilation of available information into a comprehensive and meaningful form for the waste managers, educators, and regulators is unlikely.

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ATTACHMENT 36

Industrial Hog Operations in North Carolina
Disproportionately Impact African-Americans, Hispanics and American Indians

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August 29, 2014

Summary

Background: In 2014, the North Carolina Department of Environment and Natural Resources (NC-DENR) issued a swine waste management general permit (the General Permit), which is expected to cover more than 2,000 industrial hog operations (IHOs). These facilities house animals in confinement, store their feces and urine in open pits, and apply the waste to surrounding fields. Air pollutants from the routine operation of confinement houses, cesspools, and waste sprayers affect nearby neighborhoods where they cause disruption of activities of daily living, stress, anxiety, mucous membrane irritation, respiratory conditions, reduced lung function, and acute blood pressure elevation. Prior studies showed that this industry disproportionately impacts people of color in NC, mostly African Americans.

Methods: We obtained records on the sizes and locations of permitted IHOs from NC-DENR and calculated the steady state live weight (SSLW) of hogs as an indicator of the amount of feces and urine produced at each IHO. We obtained block-level information on race and ethnicity from the 2010 census of the United States. We compared the proportions of people of color (POC), Blacks, Hispanics, and American Indians living within 3 miles of an IHO to the proportion of non-Hispanic Whites. We quantified relationships between race/ethnicity, presence of one or more IHOs, and the SSLW of IHOs, using Poisson regression and linear regression to adjust for rurality.

Results: Analyses based on a study area that excludes the state's five major cities and western counties that have no presence of this industry show that the proportion of POC living within 3 miles of an industrial hog operation is 1.52 times higher than the proportion of non-Hispanic Whites. The proportions of Blacks, Hispanics and American Indians living within 3 miles of an industrial hog operation are 1.54, 1.39 and 2.18 times higher, respectively, than the proportion of non-Hispanic Whites ($p<0.0001$). In census blocks with 80 or more percent people of color, the proportion of the population living within 3 miles of an industrial hog operation is 2.14 times higher than in blocks with no people of color. This excess increases to 3.30 times higher with adjustment for rurality. Adjusted for rurality, the SSLW of hogs within 3 miles of a census block increases, on average, 100,000, 64,000, 243,000, and 93,000 pounds for every 10 percent increase in POC, Black, Hispanic, and American Indian population ($p<0.0001$).

Conclusions: IHOs in NC disproportionately affect Black, Hispanic and American Indian residents. Although we did not examine poverty or wealth in this study, the results are consistent with previous research showing that NC's IHOs are relatively absent from low-poverty White communities. This spatial pattern is generally recognized as environmental racism.

Background

Swine production in North Carolina (NC) changed dramatically during the last decades of the 20th century. Between 1982 and 2006 the number of hog operations in the state declined precipitously while the hog population increased from approximately 2 to 10 million (Edwards and Driscoll 2009). Production became concentrated in eastern NC (Furuseth 1997).

Traditional NC producers raised small numbers of hogs, commonly fewer than 25, and hogs were one of several commercial crops on diversified farms (Edwards and Driscoll 2009). In contrast, industrial producers raise large numbers of hogs, often many thousands, in confinement houses that are designed to vent toxic gases and particles into the environment. Animal wastes are flushed into open cesspools and then sprayed on nearby fields. Pollutants emitted by IHOs include hydrogen sulfide, ammonia, a wide array of volatile organic compounds, and bioaerosols including endotoxins and other respiratory irritants (Cole et al. 2000) (Schiffman et al. 2001).

The negative impacts of particles and gases inside IHO confinements on worker health have been extensively described (Cole et al. 2000; Donham 1993; Donham et al. 1995; Donham et al. 2000; Donham 1990). Environmental pollutants from IHOs affect people who are more susceptible than workers due to young or old age, asthma or allergies, or other conditions. An extensive body of peer-reviewed scientific evidence shows that IHOs release contaminants into neighboring communities where they affect the health and quality of life of neighbors. Many of these studies have been conducted in NC. Hydrogen sulfide concentrations within 1.5 miles of IHOs in NC are associated with neighbors' ratings of hog odor and inability to engage in routine daily activities (Wing et al. 2008), increased stress and anxiety (Horton et al. 2009), irritation of the eyes, nose and throat, respiratory symptoms (Schinasi et al. 2011), and acute elevation of systolic blood pressure (Wing et al. 2013). A study of NC public middle school children who participated in an asthma survey, which was conducted by the NC Department of Health and Human Services, found that children attending schools within three miles of an IHO had more asthma-related symptoms, more doctor-diagnosed asthma, and more asthma-related medical visits than students who attended schools further away (Mirabelli et al. 2006). The same study reported a 23% higher prevalence of wheezing symptoms among children who attended schools where staff reported noticing livestock odor inside school buildings twice or more per month compared to children who attended schools where no livestock odor was reported (Mirabelli et al. 2006). Other studies in NC (Tajik et al. 2008) (Wing and Wolf 2000) (Bullers 2005) (Schiffman et al. 1995) and elsewhere (Donham et al. 2007) (Thu et al. 1997) (Radon et al. 2007) also document negative impacts of IHO air pollution on neighbors' health and quality of life.

Liquid contaminants from IHOs are released to the environment through leakage of animal waste storage pits, runoff from land application of liquid wastes, atmospheric deposition, and failure of the earthen walls of waste pits (Burkholder et al. 2007). Overflow of waste pits during heavy rain events results in massive spills of animal waste into neighboring communities and waterways. For example, in late September, 1999, 237 NC IHOs were located in flooded areas identified from satellite imagery provided by the NC Division of Emergency Management (Wing et al. 2002). Parasites, bacteria, viruses, nitrates, and other components of liquid IHO waste pose threats to human health (Burkholder et al. 2007; Cole et al. 2000).

Routine use of sub-therapeutic doses of antibiotics to promote weight gain of hogs promotes antibiotic resistance, making infections in humans more difficult to treat (Silbergeld et al. 2008). Airborne bacteria, including antibiotic resistant strains, have been isolated from IHO air emissions (Schulz et al. 2012) (Green et al. 2006) (Gibbs et al. 2006), and antibiotic resistant bacteria are associated with animal vectors near industrial animal operations, including flies (Graham et al. 2009), rodents (van de Giessen et al. 2009), and migratory geese that land on NC's IHO liquid waste pits (Cole et al. 2005). A recent medical records study from Pennsylvania shows that people living near IHO liquid waste application sites have elevated rates of infection with methicillin resistant *Staphylococcus aureus* (Casey et al. 2013). NC industrial livestock workers carry strains of *Staphylococcus aureus* that are associated with swine, including antibiotic resistant strains (Rinsky et al. 2013). These bacteria could be spread by liquid waste and airborne particles.

Using information from the United States Census of 1990 and locations of IHOs reported by the North Carolina Department of Environment and Natural Resources (NC-DENR) in 1998, we showed that the state's IHOs were disproportionately located in areas where more people of color (POC), primarily African Americans, live (Wing et al. 2000). We concluded that their disproportionate location in communities of color represented an environmental injustice. Since 1998 additional IHOs have obtained permission to operate and others are no longer in business. Additionally, between 1990 and 2010 the state's population size and spatial distribution changed due to births, deaths and migration. In this report we update our previous findings by evaluating whether IHOs operating under the general permit issued on March 7, 2014, will disproportionately impact POC, Blacks, Hispanics, and American Indians.

Materials and Methods

Lacking a list of the unique IHOs operating under the General Permit finalized in 2014, we used a list of all permitted industrial animal operations provided by NC-DENR on January 24, 2013 that we had prepared for prior research. First we excluded all non-swine operations from the list. Next we excluded swine operations with expired permits and permits with an allowable head count equal to zero. We also excluded permits that did not appear on a list of permitted animal operations published by DENR in January, 2014. We merged multiple permits issued for the same facilities to obtain a total head count for each operation. However the head count may be misleading as a measure of the pollution from each IHO because some facilities primarily house small pigs while others primarily house large hogs. We therefore calculated each facility's total steady state live weight (SSLW) using NC-DENR's formula based on the number and average weight of each growth stage of swine permitted at the facility. We interpret SSLW as a summary measure of the feces and urine produced by the swine of different growth stages at each facility.

Following the protocol provided in our previous study we excluded facilities operated by research institutions because they are subject to different location and management decisions than are commercial operations (Wing et al. 2000). Finally, we excluded facilities that do not hold a certificate of coverage to operate under the General Permit because they operate under individual permits or National Pollutant Discharge Elimination System general permits. The resulting facilities should closely approximate those expected to seek to continue operating under the renewed General Permit. The renewed General Permit takes effect on October 1, 2014, at which time we plan to update the list created for this research.

The vulnerability of people of any race/ethnicity to having polluting facilities nearby can be affected by the race and ethnicity of other people in their community. For example, African-Americans who live in areas primarily populated by non-Hispanic Whites have, generally, a lower susceptibility to being near polluting facilities than African-Americans who live in areas primarily populated by Hispanics or American Indians. We therefore conducted our primary analyses of disproportionate impact using the POC category. We also conducted analyses for specific racial/ethnic categories. We defined the following racial/ethnic categories: non-Hispanic White (non-Hispanics who identified as White and no other race), POC (all people not categorized as non-Hispanic white), Black (people who identified themselves as African-American or Black with or without any other race), Hispanic of any race, and American Indian (people who identified themselves as American Indian with or without any other race). We used block-level race/ethnicity-specific population counts from the US Census of 2010.

As large-scale agricultural facilities, IHOs are not located in major cities. Following the protocol adopted in our prior research, we defined a study area for our primary analyses that excluded census blocks in the five major metropolitan areas of NC (Charlotte, Winston Salem, Greensboro, Durham and Raleigh) as well as 19 western counties that neither have an IHO nor border a county that has an IHO. We conducted additional analyses for the entire state.

We considered residents of blocks to be affected by IHOs within three miles of the block centroid. Blocks were categorized as either having, or not having, an IHO within three miles. Additionally, we calculated the total permitted SSLW of hogs within three miles of the centroid of each block as a measure of the total potential influence of pollutants from nearby IHOs on the residents of the block.

As in our prior study, we also calculated the population density of each block, defined as the number of people per square mile. Population density is a measure of rurality, which is strongly related to the availability of land for agriculture and the price of land. Racial/ethnic groups in NC differ in their urban vs. rural residence, making them differentially susceptible to types of polluting facilities that locate in rural vs. urban locations. For example, a larger proportion of non-Hispanic Whites in NC live in remote rural areas than do Blacks, the racial comparison is affected not only by the susceptibility of Whites vs. Blacks to IHOs, but also by differences in whether they live in rural vs. urban areas. By adjusting for population density (or rurality), we compare racial vulnerability to IHOs for racial groups within each level of rurality. This adjustment is analogous to other statistical adjustments in epidemiology, as when the death rates of two countries are compared: even though death rates at every age may be higher in a poor than a rich country, the poor country may have a lower overall death rate simply because it has a younger age distribution. In that case, age-adjustment is used to compare mortality in the two countries just as we use density-adjustment to compare the proximity to IHOs in areas with different racial/ethnic make-up.

We used weighted Poisson regression to quantify relationships between race/ethnicity and the presence of one or more IHOs within three miles of a block. We used weighted linear regression to quantify relationships between race/ethnicity and the SSLW of hogs permitted within three miles of a block. We used census block populations as weights. In density-adjusted models we included variables for the natural log of population density raised to the first, second and third power. As in our prior analysis, this cubic model fit the data well and additional power terms added little to the model fit (Wing et al. 2000). For the two largest racial/ethnic groups other than non-Hispanic Whites, POC and Blacks, we categorized race/ethnicity in groups of blocks 20% in width compared to blocks with no POC using indicator variables. Due to smaller numbers in these categories we did not fit models with indicator variables for Hispanics and American Indians. We also considered the percent of population of each race/ethnicity as a continuous variable, estimating the added burden of IHOs for a 10% increase in the population.

This study involves neither random sampling nor randomization of exposure to IHOs, therefore statistical significance testing is inappropriate and confidence intervals do not correspond to the probability that the true values of measures of association are within the interval. However, the US-EPA considers statistical significance in its assessment of environmental racism. We therefore report p-values for differences in proportions of each racial/ethnic group within 3 miles of an IHO using t-tests. We report 95% confidence intervals (CIs) as measures of precision of the associations estimated from regression models. 95% CIs that exclude the null value (1.0 for ratios and 0.0 for differences) are commonly considered to be statistically significant at $p<0.05$.

Results

We estimate that 2,055 IHOs were operating under the General Permit in January 2014, and that they were permitted to house approximately 1.2 billion pounds of swine (Table 1). The 160 (7.7%) IHOs permitted to house between 20 and 100 thousand pounds accounted for only 1% of the total permitted SSLW. The 342 (17.2%) IHOs permitted to house between 1 and 10.2 million pounds accounted for 46.5% of the total.

Table 2 shows that there are over 6.5 million residents of the study area. Approximately 986,000 (15.1%) of these live in census blocks whose centroid is within 3 miles of an IHO that operates under the General Permit. This includes 602,380 non-Hispanic Whites and 383,522 POC. 13.1% of non-Hispanic Whites and 19.9% of POC in the study area live in blocks within 3 miles of an IHO.

Based on the study area population in Table 2, Table 3 shows ratios of percentage of POC living within 3 miles of an IHO compared to the percentage of non-Hispanic Whites living within 3 miles of an IHO. The percentage of POC living within 3 miles of an IHO is 1.52 times higher than the percentage of non-Hispanic Whites. The percentages of Blacks, Hispanics and American Indians living within 3 miles of an IHO are 1.54, 1.39 and 2.18 times higher, respectively, than non-Hispanic Whites. If residents of the study area had been randomized to live within 3 miles of an IHO, the probabilities of observing differences of these magnitudes or greater are less than 0.0001; the observed differences are considered to be highly statistically significant.

We calculated these same ratios based on the entire state population of 9,535,483. The percentages of POC, Blacks, Hispanics and American Indians living within 3 miles of an IHO are 1.38, 1.40, 1.26 and 2.39 times higher than the percentage of non-Hispanic Whites, respectively. These ratios are considered to be highly statistically significant.

Figure 2 shows the percent of people living within 3 miles of an IHO in relation to the percent of people of color in blocks. In areas with less than 20% POC, just over 10% of the population lives within 3 miles of an IHO. In areas with 60-80% POC, over 20% of the population lives so close to an IHO. In areas with more than 80% POC, more than a quarter of the population lives within 3 miles of an IHO.

Table 4 presents ratios of the percent of people living within 3 miles of an IHO in blocks with >0 to <20%, 20 to <40%, 40 to <60%, 60 to <80% and 80 to 100% POC compared to blocks with no POC. The total population in these categories ranges from 526,305 in blocks with 60 to <80% POC to 2,577,015 in blocks with >0 to <20% POC. Ratios are statistically significantly elevated for all areas with more than 40% POC with or without adjustment for rurality. Ratios on the right side of Table 4 are adjusted for rurality. These ratios increase with the percentage POC. The highest ratios occur in areas with more than 80% POC, where over three times as many people live near IHOs, adjusted for rurality, compared to areas with no POC. These excesses are considered to be highly statistically significant.

Table 5 shows the results of analyses for Blacks parallel results to in Table 4 for all POC. Although ratios are somewhat lower for Blacks than POC, the percent of people living within 3 miles of an IHO is statistically significantly elevated in all groups of blocks that are more than 40% Black, with or without adjustment for rurality. In areas that are 80% or more Black, twice as many people live within 3 miles of an IHO compared to areas with no Blacks, a disparity that increases to three times more with adjustment for rurality. These excesses are considered to be highly statistically significant.

Table 6 presents the increased percent of the population living within 3 miles of an IHO for each additional 10 percent of the population of POC, Blacks, Hispanics, and American Indians. This analysis is similar to the results in Tables 4 and 5, but rather than using categories, the relationship between race/ethnicity and proximity to IHOs is modelled as a linear function. For every ten percent increase in POC, the proportion of people residing within 3 miles of an IHO increases, on average, by 10.7%. These values are 9.4, 8.5, and 16.2 for Blacks, Hispanics, and American Indians, respectively. Adjusting for rurality, 14.8% more people reside within 3 miles of an IHO for each additional ten percent POC. Adjusted values are 13.0, 16.3 and 11.8 for Blacks, Hispanics and American Indians, respectively. These linear relationships between race/ethnicity and living near IHOs are considered to be highly statistically significant.

Table 7 shows the difference in SSLW of hogs within 3 miles of residents of blocks with >0 to <20%, 20 to <40%, 40 to <60%, 60 to <80% and 80 to 100% POC compared to blocks with no POC. Blocks in categories with more than 20% POC have, on average, between 177 and 510 thousand pounds more hogs within 3 miles than blocks with no POC. Adjusting for population density, blocks with more than 60 percent POC have, on average, more than three-quarters of a million pounds more hogs permitted within 3 miles than areas with no POC. These excesses are considered to be highly statistically significant.

Table 8 presents parallel results for percentage Black population. As for POC, areas with more than 20% Black residents have an excess SSLW of hogs compared to areas with no Black residents, and differences are greater with adjustment for rurality. Adjusted for population density, blocks with more than 40% Black residents have between 493,000 and 620,000 more pounds of hogs within 3 miles than areas with no Black residents. These excesses are considered to be highly statistically significant.

Table 9 provides the average additional SSLW of hogs permitted in areas with POC for each percent increase in specific racial/ethnic categories. Adjusted for population density, the permitted SSLW of hogs within 3 miles of blocks increases 100, 64, 242, and 92 thousand pounds for each ten percent increase in POC, Black, Hispanic, and American Indian population, respectively. These linear relationships between race/ethnicity and SSLW are considered to be highly statistically significant.

Figure 3 depicts the data analyzed above. Each dot represents an IHO that was operating under the General Permit in 2014. IHOs are concentrated in NC's Coastal Plain Region, between the Piedmont and Tidewater. The red areas of Figure 3 indicate that this region has more people of color than other parts of the study area.

Conclusion

IHOs operating under the NC-DENR General Permit in 2014 are disproportionately located near communities of color. The disparities are considered to be highly statistically significant for Blacks, Hispanics, American Indians, and all POC. IHOs pollute local ground and surface water. They routinely emit air pollutants that negatively impact the quality of life and health of nearby residents. In addition to their well-documented effects on physical, mental and social well-being, residents of areas with a high density of IHOs, and especially residents of color, have been subjected to intimidation including threats of legal action, violence, and job loss (Wing 2002). The industry's close ties with local and state government officials help it to avoid regulation that could protect neighbors, and creates barriers to democracy in rural communities of color (Thu 2001, 2003). These discriminatory impacts could be reduced by decreasing the density of production and use of technologies that prevent releases of pollutants.

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Figure 1
North Carolina study area, 2014

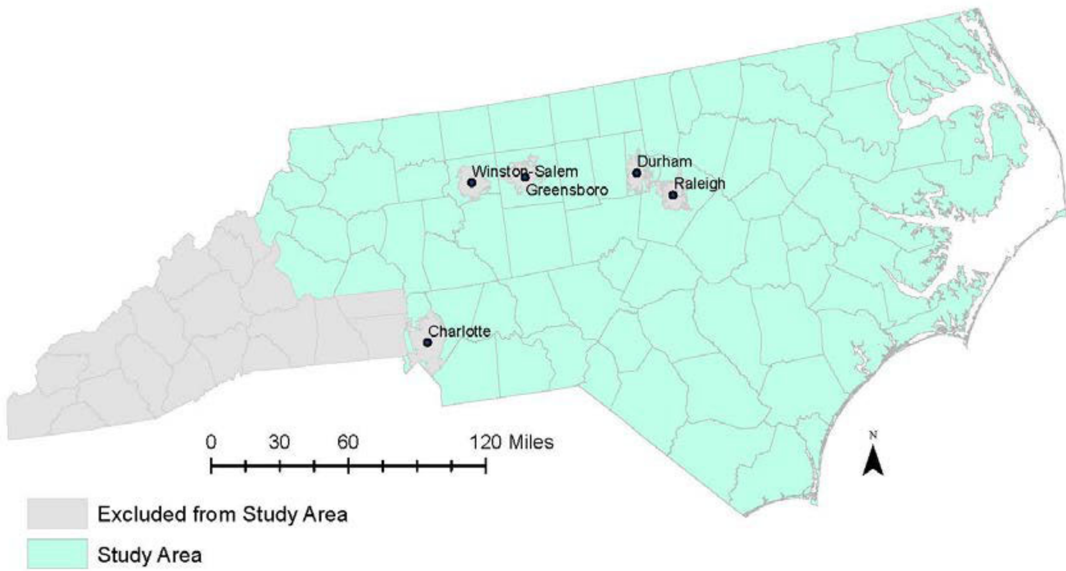

Figure 2
Percent of population living within 3 miles of an IHO
in relation to percent people of color, NC, 2014

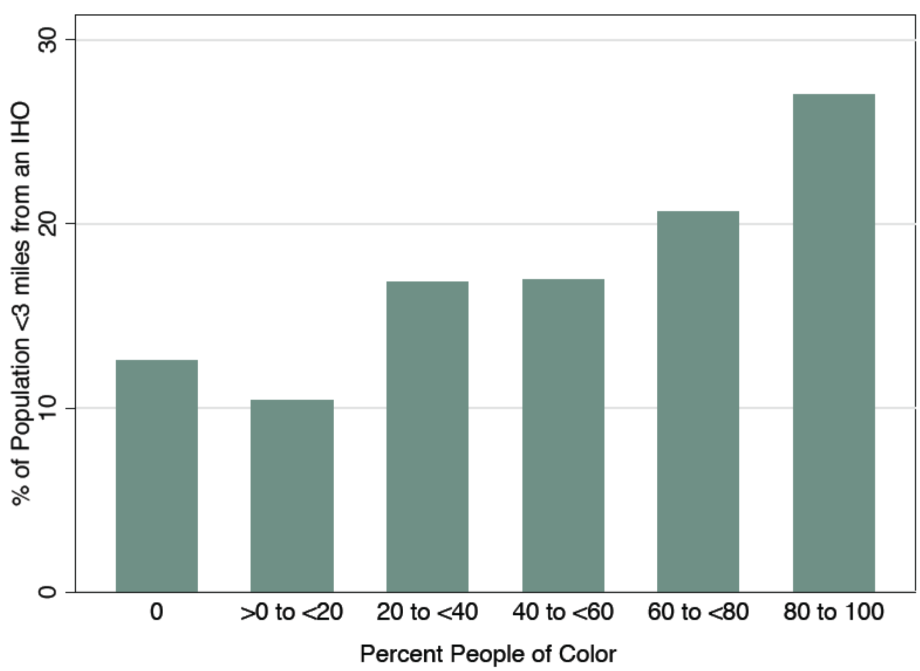

Figure 3
Racial and ethnic composition of census blocks and the locations of NC IHOs operating under the General Permit, 2014

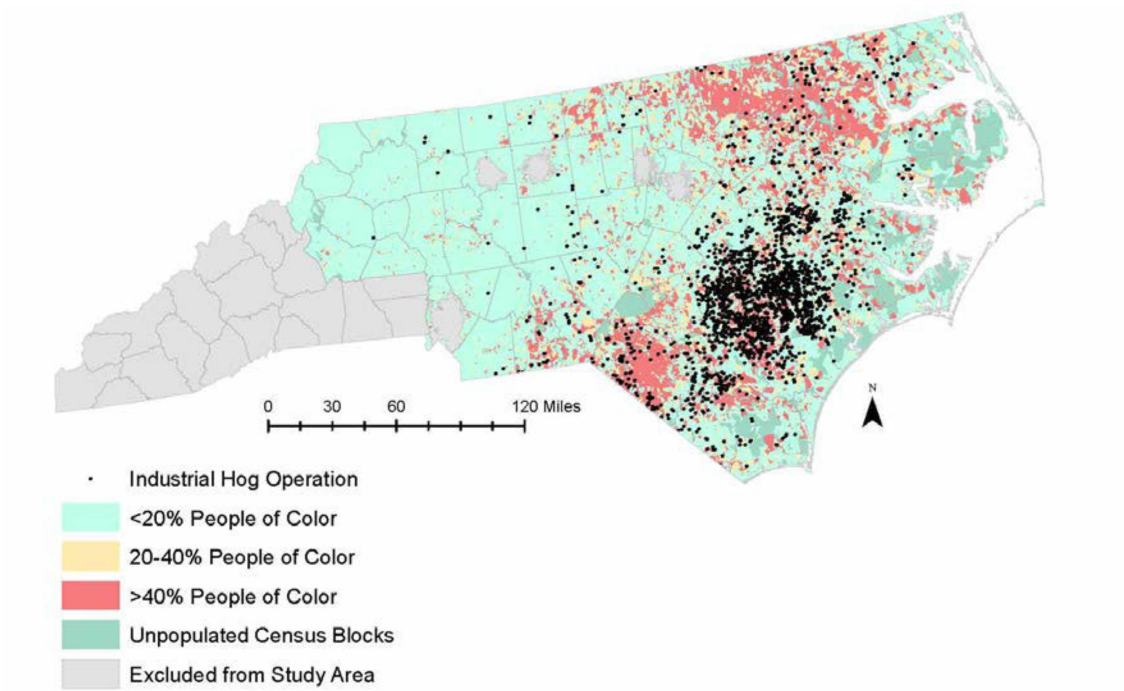

Table 1
Steady state live weight of IHOs
operating under the General Permit, NC, 2014

Permitted SSLW[1]	Number of IHOs	Percent of IHOs	Total SSLW[1]	Percent of total SSLW
20-	160	7.7	12,574	1.0
100-	447	21.6	76,626	5.9
250-	577	28.1	222,003	17.1
500-	529	25.4	383,918	29.6
1,000-10,200	342	17.2	603,354	46.5
Total	2055	100.0	1,298,474	100.0

[1]Thousands of pounds

Table 2
Racial and ethnic composition of NC census blocks within 3 miles
of an IHO and more than 3 Miles of an IHO, 2014

	<u>≤3 miles from an IHO</u>		<u>>3 miles from an IHO</u>		
Racial Category	Number	Percent	Number	Percent	Total[1]
Non-Hispanic white	602,380	13.1	4,003,455	86.9	4,605,835
POC[1]	383,522	19.9	1,548,276	80.1	1,931,798
Black	277,199	20.2	1,096,795	79.8	1,373,994
Hispanic	92,679	18.1	418,292	81.9	510,971
American Indian	40,621	28.5	101,872	71.5	142,493
Total[1]	985,902	15.1	5,551,731	84.9	6,537,633

[1]POC can be counted in more than one racial/ethnic category. The total population is equal to the number of non-Hispanic Whites plus the number of POC.

Table 3
Ratios of POC compared to non-Hispanic Whites living within 3 Miles
of an IHO operating under the General Permit, 2014

Racial/ethnic Category	Population	≤3 miles from an IHO Number	Percent	Ratio[2]	p-value[3]
Non-Hispanic white	4,605,835	602,380	13.1	1.00	--
POC[1]	1,931,798	383,522	19.9	1.52	<0.0001
Black	1,373,994	277,199	20.2	1.54	<0.0001
Hispanic	510,971	92,679	18.1	1.38	<0.0001
American Indian	142,493	40,621	28.5	2.18	<0.0001
Total[1]	6,537,633	985,902	15.1		

[1]People of color can be counted in more than one racial/ethnic category. The total population is equal to the number of non-Hispanic Whites plus the number of POC.
[2]Ratio of the percent of people of other racial/ethnic groups to percent of non-Hispanic Whites living within 3 miles of an IHO
[3]A difference in proportions of this magnitude or greater would be expected to occur less than one time in ten thousand if people of different racial/ethnic groups had been randomized to live within 3 miles of an IHO.

Table 4
Ratios comparing the percent of people residing within 3 miles of an IHO
in blocks with POC compared to blocks with no POC

Percent POC	Population	Unadjusted Prevalence Ratio	95% CI	Adjusted[1] Prevalence Ratio	95% CI
0	694,747	1.0	referent	1.00	referent
>0 to <20	2,577,015	0.83	0.82, 0.83	1.01	1.00,1.02
20 to <40	1,364,923	1.34	1.33, 1.45	1.95	1.93, 1.97
40 to <60	799,124	1.35	1.34, 1.36	2.15	2.13, 2.16
60 to <80	526,305	1.64	1.62, 1.65	2.53	2.50, 2.55
80 to 100	575,519	2.14	2.12, 2.16	3.30	3.27, 3.32

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density

Table 5
Ratios comparing the percent of people residing within 3 miles of an IHO in blocks with Black residents compared to blocks with no Black residents

Percent Black	Population	Unadjusted Prevalence Ratio	95% CI	Adjusted[1] Prevalence Ratio	95% CI
0	1,308,061	1.00	referent	1.00	referent
>0 to <20	2,941,746	0.93	0.92, 0.94	1.20	1.19,1.21
20 to <40	1,043,277	1.44	1.43, 1.45	2.07	2.05, 2.08
40 to <60	536,198	1.52	1.51, 1.53	2.18	2.17, 2.20
60 to <80	336,232	1.57	1.56, 1.59	2.19	2.17, 2.21
80 to 100	372,119	2.01	1.99, 2.02	3.06	3.04, 3.09

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density

Table 6
Percent difference in the percent of people residing within 3 miles of an IHO for a ten percent increase in the population of each racial/ethnic group

Racial/ethnic group	Unadjusted Percent	95% CI	Adjusted[1] Percent	95% CI
POC	10.7	10.6, 10.8	14.8	14.7, 14.9
Black	9.4	9.3, 9.4	13.0	12.9, 13.1
Hispanic	8.5	8.4, 8.6	16.3	16.1, 16.4
American Indian	16.2	16.0, 16.4	11.8	11.6, 12.0

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density

Table 7
Difference in SSLW of hogs within 3 miles of residents of blocks with POC compared to blocks with no POC

Percent POC	Unadjusted SSLW[2]	95% CI	Adjusted[1] SSLW	95% CI
0	Referent	-	Referent	-
>0 to <20	-35	-73, 3	190	154, 227
20 to <40	177	136, 219	535	495, 575
40 to <60	308	262, 353	717	672, 762
60 to <80	510	459, 561	896	846, 946
80 to 100	453	403, 503	837	788, 885

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density
[2]1,000s of pounds

Table 8
Difference in SSLW of hogs within 3 miles of residents of blocks with Black residents compared to blocks with no Black residents

Percent Black	Unadjusted SSLW[2]	95% CI	Adjusted[1] SSLW	95% CI
0	Referent	-	Referent	-
>0 to <20	-4	-33, 25	237	207, 265
20 to <40	190	153, 227	493	457, 530
40 to <60	327	281, 372	620	576, 665
60 to <80	275	221, 330	547	494, 599
80 to 100	165	113, 218	494	444, 545

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density
[2]1,000s of pounds

Table 9
Difference in SSLW of hogs within 3 miles of residents of blocks for a ten percent increase in population of each racial group

Racial/ethnic group	Unadjusted SSLW[2]	95% CI	Adjusted[1] SSLW	95% CI
POC	67	63, 71	100	96, 104
Black	38	34, 42	64	60, 68
Hispanic	183	174, 192	242	234, 251
American Indian	124	111, 137	92	80, 105

[1]Adjusted for rurality using a cubic polynomial of the natural log of population density
[2]1,000s of pound

ATTACHMENT 37

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Short Communication

Terra incognita: The unknown risks to environmental quality posed by the spatial distribution and abundance of concentrated animal feeding operations

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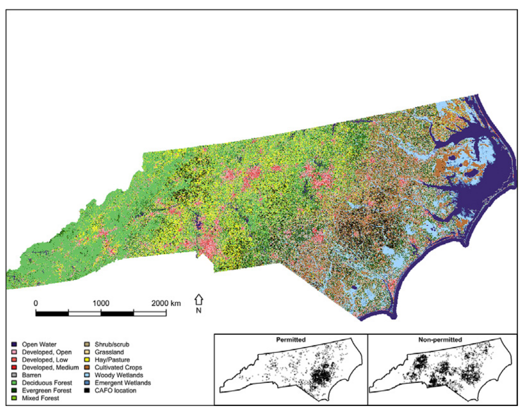
HIGHLIGHTS

- Environmental risk assessments of CAFOs are complicated by a lack of spatial data.
- North Carolina CAFOs are concentrated in the Coastal Plain, subject to large storms.
- 19% of CAFO points (1262) across the state are within 100 m of streams.
- Data gaps prohibit landscape modeling of impacts under changing conditions.

GRAPHICAL ABSTRACT



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ABSTRACT

Concentrated animal feeding operations (CAFOs) pose wide ranging environmental risks to many parts of the US and across the globe, but datasets for CAFO risk assessments are not readily available. Within the United States, some of the greatest concentrations of CAFOs occur in North Carolina. It is also one of the only states with publicly accessible location data for classes of CAFOs that are required to obtain water quality permits from the U.S. Environmental Protection Agency (EPA); however, there are no public data sources for the large number of CAFOs that do not require EPA water quality permits. We combined public records of CAFO locations with data collected in North Carolina by the Waterkeeper and Riverkeeper Alliances to examine the distribution of both permitted and non-permitted CAFOs across the state. Over half (55%) of the state's 6646 CAFOs are located in the Coastal Plain, a low-lying region vulnerable to flooding associated with regular cyclonic and convective storms. We identified 19% of CAFOs ≤ 100 m of the nearest stream, and some as close as 15 m to the nearest stream, a common riparian buffer width for water quality management. Future climate scenarios suggest large storm events are expected to become increasingly extreme, and dry interstorm periods could lengthen. Such extremes could exacerbate the environmental impacts of CAFOs. Understanding the potential impacts of CAFO agroecosystems will require remote sensing to identify CAFOs, fieldwork to determine the extent of environmental footprints, and modeling to identify thresholds that determine environmental risk under changing conditions.

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1. Introduction

Beginning in the mid twentieth century, there was a significant shift in US agriculture toward concentrated animal feeding operations, or CAFOs (Mallin, 2000). The transition from small, family farms to consolidated operations began in the poultry industry during the 1950s, and the model was adopted by swine farmers in the Midwest during the 1970s and 80s. The trend of increasing CAFOs reached the southeastern US in the late 1980s (Mallin, 2000). As a result, North Carolina experienced a nearly four-fold increase in swine inventory from 1975 to 2000 (Yang et al., 2016). Poultry production has increased in North Carolina during the same approximate time period, and the state has been one of the top poultry producers in the United States (Yang et al., 2016). The state Department of Environmental Quality estimated that from 1992 to 2014, poultry inventory increased where it is most concentrated (16% increase Yadkin-Pee Dee River basin, 9% increase Cape Fear River basin), and expanded rapidly in new areas of the state (393% increase Lumber River basin, 331% increase Broad River basin) (Patt, 2017). Although CAFOs provide a rapid and profitable way to provide food to a growing human population, they present significant risks to human health and environmental quality (Burkholder et al., 2007; Greger and Koneswaran, 2010; Mallin et al., 2015). Due to the high volumes of animal waste produced, CAFOs have high potential to contribute to soil, air, and water pollution, posing health risks to nearby communities (Burkholder et al., 2007; Donham et al., 2007; Greger and Koneswaran, 2010; Nicole, 2013). These operations tend to be spatially clustered in areas with environmental regulations and zoning requirements that favor industrial agriculture, particularly the southeastern US (Mallin, 2000) and in rural, impoverished, minority communities (Emanuel, 2018; Nicole, 2013; Wing et al., 2002).

Understanding the impacts of CAFOs and developing and implementing best management practices to mitigate impacts, requires fine-scale spatial data on CAFO locations. Existing research on the spatial distribution of CAFOs and potential impacts to environmental and human health have been conducted at relatively large spatial scales, such as counties (Yang et al., 2016) or watersheds (Harden, 2015). County level agricultural statistics such as the total number of animals housed are available from USDA (https://www.nass.usda.gov/). However, county-scale assessments and similar large-scale studies are not aligned with many ecological processes, and thus are limited in their ability to evaluate the potential impacts of CAFOs on nutrient cycling and water resources at scales that are most appropriate for improving management practices. Data are not publicly or readily available at finer spatial scales or scales more aligned with ecological processes, such as watersheds.

Recognizing the potential environmental and human health risks of CAFOS, some federally mandated best management practices have been developed and implemented. Large CAFOs that meet the EPA definition of >1000 animal units using a liquid waste disposal system are recognized as point sources of pollution and thus, a water quality permit is required (hereafter, permitted CAFOs). Liquid waste disposal is primarily used in swine, egg-producing poultry operations, and some cattle operations. The EPA considers an animal unit to be the equivalent of 1000 pounds of live weight, and large CAFOs are defined as having a minimum of 1000 head of beef cattle, 2500 swine, or 125,000 broiler chickens. The site must also house confined animals for at least 45 days a year and not sustain vegetation during the normal growing season over any portion of the lot to meet the regulatory CAFO definition. CAFO water quality permits regulate waste lagoons, from which liquid waste is generally transferred to a spray field, often of Bermuda grass (Mallin et al., 2015). EPA permitted CAFOs also require Comprehensive Nutrient Management Plans that detail feed, manure, and land management. States can add requirements to permits; for example, all CAFOS are inspected annually in North Carolina. As long as farms maintain a nutrient management plan, spray fields are not regulated by the water quality permit (Centner and Feitshans, 2006). Therefore, the locations or extents of spray fields associated with permitted CAFOs are generally unknown (Patt, 2017). The regulatory assumption is that nutrients and other contaminants from spray fields will remain on site, although this is not always the case (Wing et al., 2002). The environmental risk posed by spray fields is likely underestimated because impacts on agricultural runoff, groundwater recharge, or dispersal of airborne substances cannot be assessed without additional data. Further, public perceptions might not include farmland and spray fields as potential sources of CAFO impacts, resulting in an underestimate of the full risks to their communities posed by this form of industrial agriculture.

Farms with <1000 animal units and CAFOs without liquid waste disposal systems are not regulated in the same way as larger, permitted operations (hereafter, non-permitted CAFOs). Most poultry operations and some cattle operations generate dry litter waste and are thus not required to have water quality permits. In North Carolina, the state Department of Environmental Quality estimates that over 96% of poultry and over 88% of cattle operations use dry waste disposal (Patt, 2017). Waste from these operations is commonly spread on fields as fertilizer, often after transport far from the source farm, complicating the geography of the environmental impact (Patt, 2017).

Our goal was to identify the distribution of potential CAFO risk in a region with high CAFO concentrations as a first step toward improving the ability to evaluate and project the footprint of CAFO land use on environmental quality, including the export of nutrients, microbes, pathogens, and pollutants throughout surface water, ground water, the atmosphere and the terrestrial system. This assessment is also a first step toward assessing the effectiveness of mitigation practices. In some US states, locations of permitted CAFOs are publicly available. For example, an online search identified that Wisconsin, Michigan, Missouri, and North Carolina have publicly available, spatial datasets of permitted CAFOs; however, public records or datasets on the spatial locations are not available for non-permitted CAFOs. In some states, such as North Carolina, private nonprofits (e.g., Waterkeeper and Riverkeeper Alliances) have collected data on non-permitted CAFO locations. As location data are available for both permitted and non-permitted CAFOs, and because of the proliferation of CAFOs throughout the state, North Carolina provides an excellent case study to examine the spatial distribution of CAFOs.

We determined how CAFOs were distributed spatially among and within watersheds in North Carolina. We also evaluated the predominant NLCD land cover classifications surrounding CAFOs. In the United States, the National Land Cover Database (NLCD) is a publicly available dataset that aims to provide information necessary to assess ecosystem health and facilitate nutrient modeling, land use planning, and the development of best land management practices (BMPs) (Homer et al., 2015). The NLCD is scaled to at a 30-m resolution grid and updated every 5 years. Watershed models frequently use NLCD data to inform hydrologic simulations by assuming relationships between land cover and nutrient loading rates, infiltration capacities, or other factors that influence water availability and quality (Karcher et al., 2013). NLCD data layers are considered the most comprehensive, publicly available, datasets of land cover. Previous studies (Burkholder et al., 2007; Rothenberger et al., 2009) have identified the NLCD category “hay/pasture” as animal agriculture and thus, a proxy to identify CAFO locations; however, the EPA defines CAFOs as areas that do not produce crops, forage, or other vegetation. We tested whether CAFO locations are consistently categorized this way or whether they fall into other NLCD categories that are not typically associated with the water quality footprints of CAFOs.

2. Methods

We collected data on permitted CAFO locations from the North Carolina Department of Environmental Quality, which maintains a publicly available spatial dataset (https://deq.nc.gov/cafo-map). Spatial point data for non-permitted CAFOs were shared by the Waterkeeper and Riverkeeper Alliances who created the dataset by inspecting satellite imagery in Google Earth. In the imagery, large, rectangular barns used in poultry operations were identified; therefore, this dataset would not include dry waste cattle operations. We also assume that identification of the distinct geometry of a large, rectangular barn several times longer than its width, or sets of such barns, without manure lagoons is an accurate representation of a non-permitted CAFO. From the dataset, we independently verified 800 randomly selected sites (approx. 20%) using Google Earth Pro imagery set to December 2016, in accordance with the timing of the Riverkeeper assessment. We found that 710 (88.8%) of the points were within the footprint of the facility (on barns or within groupings of barns) and an additional 10.5% were and average of 24.3 m from the facility footprint, less than the width of one NLCD pixel. We found only two points (0.3%) not located adjacent to a farm and three points (0.4%) where barns had been removed but were present in imagery within the previous 2–5 years.

We used the National Watershed Boundary dataset (http://datagateway.nrcs.usda.gov/) to determine the distribution of both permitted and non-permitted CAFOs among major river basins. The National Hydrography Dataset (NHD) (U.S. Geological Survey, 2013) was used to determine the straight-line distance between CAFO points and the nearest stream channel. Then, using the most recent NLCD, 2011, we examined the distribution of land cover classification of CAFO sites.

We determined the land cover at CAFO points by taking the modal (most frequent) land cover from the 2011 NLCD layer within a 50-m buffer of each CAFO point. NLCD data for North Carolina were downloaded from the USGS National Map data platform (TNM Download V1.0: https://viewer.nationalmap.gov/basic/). The NCLD includes 16 different land cover categories derived from Landsat imagery (Homer et al., 2015). The two primary categories for agricultural land cover are cultivated crops, and hay/pasture. The cultivated crops category is defined as actively tilled land or land where annual or perennial crops represent at least 20% of the total vegetation (Homer et al., 2015). The hay/pasture land cover category is defined as areas with at least 20% coverage by grasses or legumes. Some (Burkholder et al., 2007; Rothenberger et al., 2009) categorize hay/pasture as animal agriculture, although the EPA defines CAFOs as areas that do not produce vegetation.

3. Results

3.1. Spatial distribution of CAFOs

North Carolina had a total of 6646 CAFOs as of 2015, including 2679 permitted CAFOs (40%) and 3967 non-permitted CAFOs (60%). Permitted CAFOs were primarily for swine (87%) with a few cattle operations (10%) and few egg producing poultry operations (1%) or other types of operations (2%). Permitted CAFOS are concentrated in the Coastal Plain physiographic region of southeastern North Carolina (2241/2679, 84%, Fig. 1). Non-permitted CAFOS are distributed primarily across the Piedmont (62%) and Coastal Plain (36%).

Half of the permitted CAFOs and 28% of the non-permitted CAFOS were located in the Cape Fear River basin, which is a large (23,735 km^2) basin that drains 18% of the state. Within the Cape Fear, CAFOs are concentrated heavily within the Black River and Northeast Cape Fear River sub-watersheds. Together, the two sub-watersheds drain only 6% of the state land area, but they contain 43% of the state's permitted CAFOs. The Upper Yadkin basin covers only 5% of the state

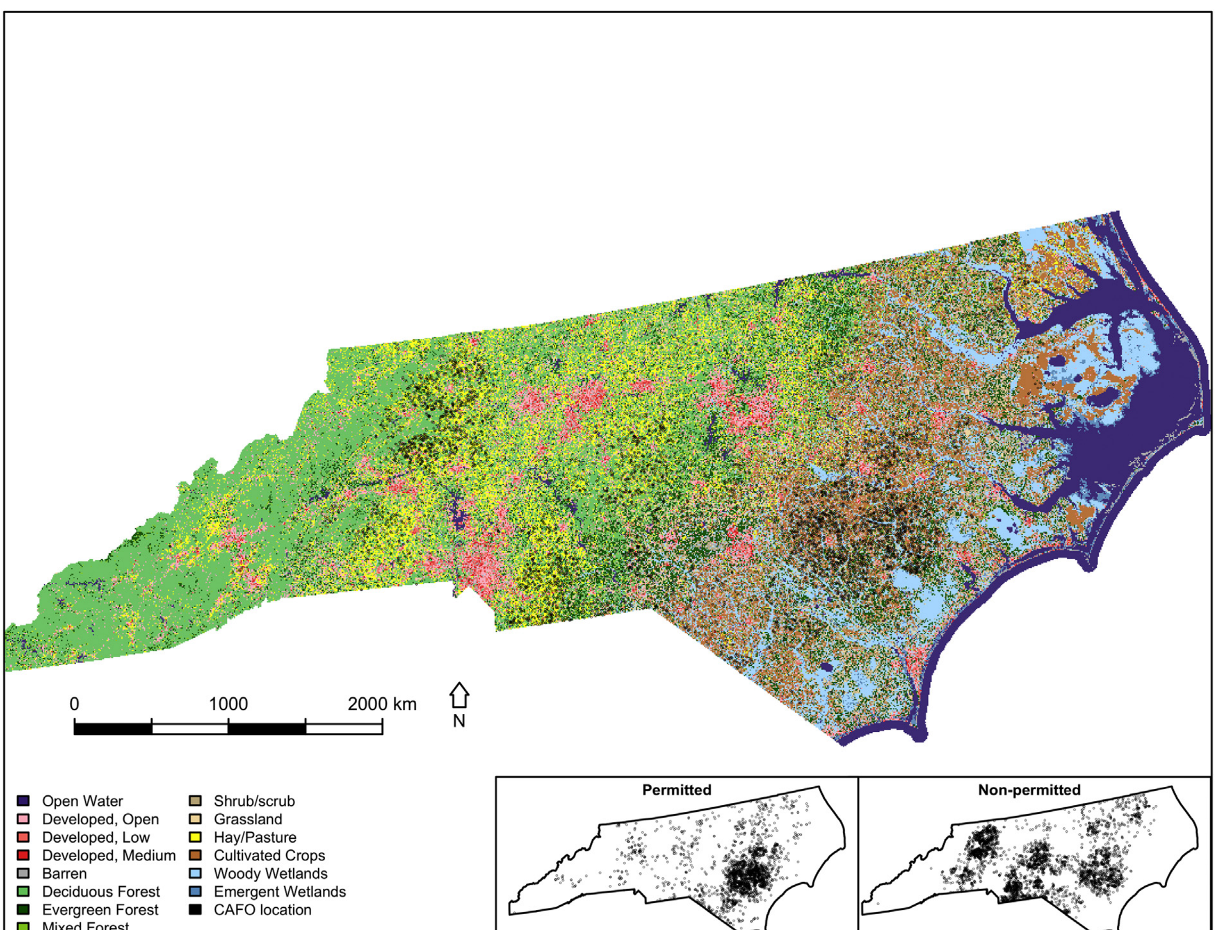

Fig. 1. Locations of CAFOs throughout the NLCD land cover classification of North Carolina.

land area but contains the highest concentration of non-permitted CAFOS, at 11% or 437 CAFOs. The concentration of CAFOs suggests that certain parts of the state, namely sub-watersheds of the Cape Fear and Yadkin River basins, are at higher risk of water quality degradation and other environmental impacts (Fig. 2).

Statewide, half of CAFOs are located within 203 m of a stream (205 m for permitted, 202 m for non-permitted, Fig. 3). We identified 1189 CAFOs (19%) within 100 m of the nearest stream. Of these operations, 67 permitted and 38 non-permitted CAFOs were less than ~15 m (50 ft) from a stream, which is the North Carolina State forestry recommended forest riparian buffer width for perennial water bodies (http://www.ncforestservice.gov/water_quality/bmp_manual.htm). (Aside from forestry, riparian buffer recommendations or regulations vary by watershed within the state). When analyzed by watershed rather than by CAFO type, there was some variability in the median distance to the stream (Fig. 4). Not only does the Northeast Cape Fear watershed contain a high CAFO concentration, but the median distance to a stream was 97 m, and 24 CAFOs were within the 15 m of streams.

3.2. NLCD land cover classification of CAFOs

Previous studies on the environmental impacts of CAFO land use have used the NCLD hay/pasture category is used as a proxy for animal agriculture and CAFO locations (Burkholder et al., 2007; Rothenberger et al., 2009). In our analysis, only 13% of permitted and 42% of non-permitted CAFOs were categorized as hay/pasture. We found that CAFO locations were frequently classified by the NLCD as cultivated crops, including 57% of permitted and 35% of non-permitted CAFOs. Considering both hay/pasture and cultivated cropland, 70% of permitted CAFOs and 77% of non-permitted CAFOs were characterized as an agricultural land cover type by the NLCD.

The remaining CAFOs were primarily classified as natural ecosystems. Thirteen percent of permitted CAFOs and 8% of non-permitted CAFOs were classified as natural terrestrial ecosystems (forest, scrub/shrub, or grassland). An additional 14% of permitted CAFOs and <1% of non-permitted CAFOs were classified as aquatic ecosystems (wetland or open water). Overall, 27% of permitted CAFOs and 8% of non-permitted CAFOs were classified as a natural land cover type by the NLCD. Permitted CAFOs were classified as developed land 3% of the time, but the rate for non-permitted CAFOs was much higher, at 12%. A small number of each CAFO type was classified as barren land by the NLCD (<1%permitted, and 1% non-permitted).

4. Discussion

Proliferation of CAFOs has significantly altered nutrient cycling in the United States (Robertson et al., 2013; Yang et al., 2016). From 1930 to 2012, Yang et al. (2016) identified manure loading increases of 46% for nitrogen and 92% for phosphorus. These increases were spatially clustered as CAFOs proliferated, concentrating manure nutrients in regions including the southeastern US and western Mississippi River basin. North Carolina produces the highest concentration of manure per acre of farmland in the country, (U.S. Enviornmental Protection Agency, 2013), and Yang et al. (2016) estimate manure N and P increased >70% across the state from 1930 to 2012. While this suggests nutrient loading is a spatially clustered environmental risk, the pathways these nutrients and associated manure pollutants (antibiotics, pathogens) take through the terrestrial, aquatic, and atmospheric systems are not well quantified. In a study of small (3.1–45.3 km^2) agricultural watersheds in eastern North Carolina, Harden (2015) found that the presence of CAFOs was often associated with degraded surface water quality. Such studies suggest the need for larger scale studies that

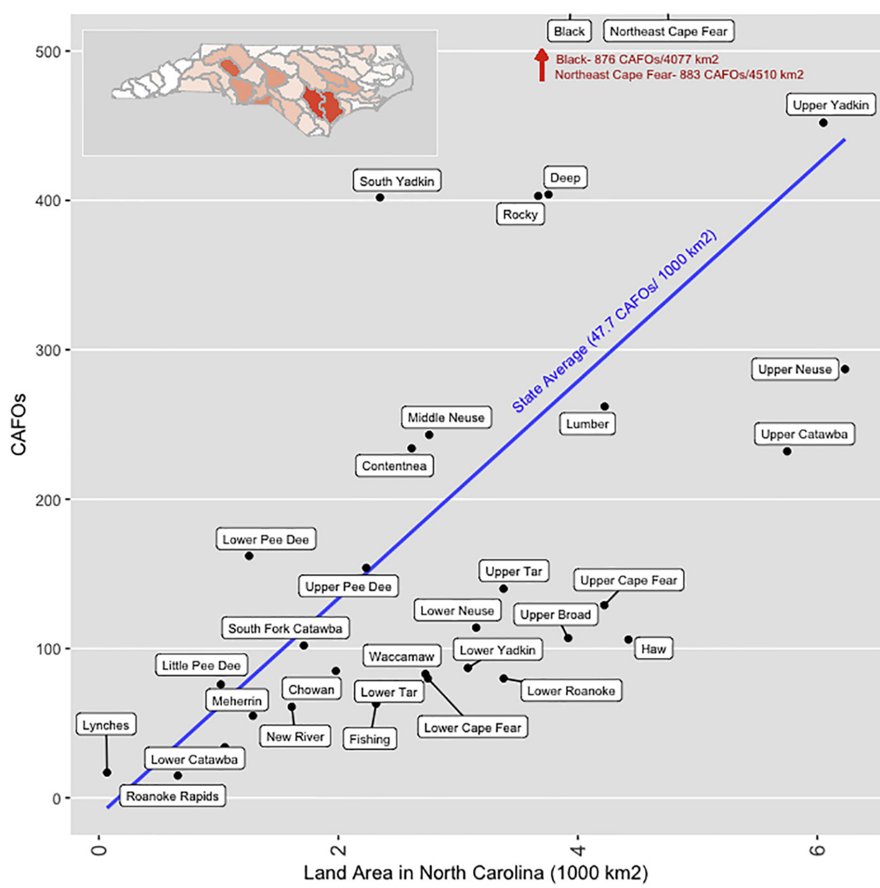

Fig. 2. CAFO density across North Carolina Watersheds. Boundaries are Hydrologic Unit Code (HUC) 8 boundaries designated by the National Hydrography Dataset. Inset map indicates increasingly concentrated CAFO density with darker colors.

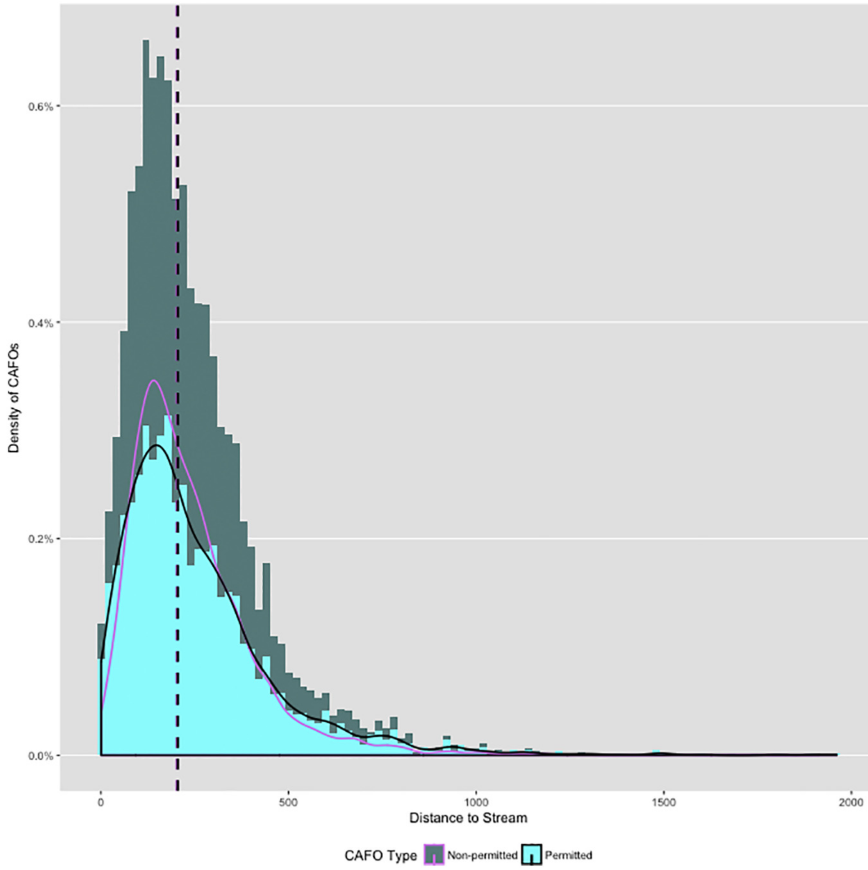

Fig. 3. Distribution of CAFO spatial data points to the nearest stream identified in the NHD database.

assess CAFOs at variable distances to the stream and in multiple land cover types in order to develop and implement BMPs that mitigate environmental impacts.

Our data indicate half of the 6646 CAFO points in the state of North Carolina are within at least 200 m of the nearest stream, and some are within 15 m; therefore, manure-based pollutants may pose a risk to water quality. The magnitude of the risk is not clear, particularly as the location of the points may vary within operations and the overall size of individual operations is not currently quantified. Our analysis included an implicit assumption that the point data are unbiased estimates of facility centroids, but does not include any analysis on the size of operations, which varies and was not available. Much of the data on individual permitted operations are protected as private in the state of North Carolina (Patt, 2017). Non-permitted operations in our dataset were identified using imagery and points are located on barns or within groups of barns, but the number of barns vary for each operation.

Landscape scale studies of the current environmental and human health risks posed by CAFOs would provide the foundation to understand and mitigate impacts in the context of global change. CAFO air quality impacts are known to pose human health risks, causing effects ranging from respiratory symptoms, headaches, nausea, eye irritation (Greger and Koneswaran, 2010; Heederik et al., 2007; Ogneva-Himmelberger et al., 2015; Schiffman et al., 2005). Climate change may exacerbate air quality-related human health risks associated with CAFOs (Fran et al., 2016; Pachauri et al., 2014). Livestock farming emits ammonium (NH_3) and nitrogen oxides (NO_x), which contribute to the formation of particulate matter and tropospheric ozone (Leip et al., 2015), both of which are expected to be problematic under climate change scenarios (Fran et al., 2016). There is a high level of confidence that extreme heat events will increase across the Southeast and that precipitation events will become more extreme. At the same time, tropical cyclones will include heavier precipitation, and likely be more intense (Carter et al., 2014; O'Gorman and Schneider, 2009; Pachauri et al., 2014; Robertson et al., 2013; Wuebbles et al., 2017). Further, regional water stress is expected to increase due to the combination of declining water yields and increasing demand from a rapidly expanding population (Carter et al., 2014; Emanuel, 2018; McNulty

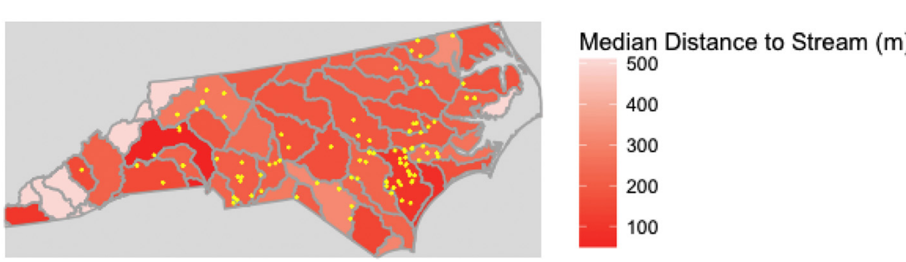

Fig. 4. HUC 8 watersheds coded by median distance between CAFO spatial data points and the nearest stream identified in the NHD database. Points indicate CAFO spatial data points within 15 m (50 ft) of streams, a commonly used riparian buffer distance.

et al., 2013; Sun et al., 2008). To minimize the environmental impacts of CAFOs under increasingly an increasingly extreme climate future, an increased understanding of their distinct environmental impacts is required, including impacts of agricultural stormwater runoff from CAFO barns, lagoons, and waste fields.

In North Carolina, CAFOs are concentrated in the Coastal Plain (Fig. 1), which is particularly vulnerable to catastrophic flooding following hurricanes (Mallin et al., 2002). Since 1990, 15 named tropical cyclones have made landfall in coastal North Carolina and an addition 20 have affected the state without a direct hit. Some of these storms have resulted in flooding and breaching of swine waste lagoons, particularly in the Northeast Cape Fear watershed (Fig. 2), which has one of the highest concentrations of CAFOs in the country (Mallin et al., 2002) and our data indicate many of these CAFOs are very near streams. Mallin et al. (2002) estimated that over 10% of permitted CAFOs were within the area inundated by Hurricane Fran in 1996.

Recognition of the environmental risk posed by CAFOs prompted a moratorium on new or expanding swine operations in 1997, following Hurricane Fran, and this moratorium was made permanent in 2007 (McDonald, 2016). During the most recent tropical cyclone, Hurricane Matthew (October 7–9, 2016), rainfall totals exceeded 38 cm in portions of Eastern North Carolina and the best information available suggests 14 industrial scale swine and poultry operations were flooded, with only two manure lagoons reported breached (McDonald, 2016; Musser et al., 2017). Animal waste contains high concentrations of nutrients as well as antibiotic and pharmaceutical contaminants (Burkholder et al., 2007). Recently, low levels of antimicrobial resistance have been detected in enteric bacteria collected from surface and ground water monitoring sites near CAFOs in the North Carolina Coastal Plain (Casanova and Sobsey, 2016). Catastrophic flooding associated with hurricanes and tropical storms can distribute these and other contaminants far downstream. Therefore, it is important to be able to model the impact of the entire operation, including spray fields that may be flooded or continue to operate during periods of saturated soil (Wing et al., 2002). Substantial data gaps concerning the spatial distribution and size of CAFOs limit the development of projections and other research projects to evaluate the potential impacts of CAFOs associated with catastrophic flooding. Such flooding not only affects surface water quality but poses risk to the large number of residents who depend on private groundwater wells for drinking water (Gibson and Pieper, 2017; Wing et al., 2002). It is possible that the moratorium on swine operations provided some mitigation or at least stabilized the risk. However, there are not sufficient data to determine the impacts of a stable number of swine operations and expanding poultry operations (Patt, 2017). Beyond flood events, long term monitoring is also needed, as animal waste lagoons can leak nutrients into soils and groundwater, reaching problematic levels gradually (Huffman and Westerman, 1995; Mallin, 2000; Ritter and Chirnside, 1990).

Given the widespread use of NLCD data as a spatial proxy for nutrient loading and other water quality parameterizations in environmental models(Almasri and Kaluarachchi, 2007; Karcher et al., 2013; Lehning et al., 2002; Nejadhashemi et al., 2011; Tran et al., 2010), the rate at which CAFOS are considered as natural systems by the NLCD is concerning. Forests and grasslands absorb nutrients and dampen hydrologic extremes by allowing for water percolation into soils, which are not functions provided by CAFOs. Even more than forests, the classification of CAFOs as wetlands in the NLCD is a significant concern for efforts to understand landscape scale nutrient pathways.

Wetlands concentrate and retain excess nutrients, sediments, and other contaminants associated with human activity, and wetland biogeochemical processes such as denitrification and adsorption can transform or sequester potential water contaminants (Zedler and Kercher, 2005). In some ways, CAFOs function as opposites of wetlands; they produce excess nutrients in the form of animal waste, and they often disperse these contaminants over wide areas using wastewater irrigation (Burkholder et al., 2007). Although only 12% of CAFOs in North Carolina are classified as wetlands by the NLCD, these operations could have outsized impacts on water quality in aquatic ecosystems.

5. Conclusions

In the US, there are approximately 450,000 CAFOs, and this form of industrialized agriculture is common in Europe and increasingly being adopted globally (Mallin et al., 2015). A growing body of work highlights tradeoffs between CAFO production and risks to the environment and human health (Burkholder et al., 2007; Donham et al., 2007; Greger and Koneswaran, 2010; Heederik et al., 2007; Mallin, 2000; Mallin and Corbett, 2006; Mallin et al., 2015; Nicole, 2013). However, a full understanding of both the risks and mitigation practices cannot be achieved without finer scale information about this emerging land use type. To this end, we encourage the development of tools and procedures to identify and incorporate CAFOs as a distinct land cover type within landscape datasets such as the NLCD.

The NLCD is produced by the Multi-Resolution Land Characteristics Consortium, a federal interagency organization with missions in science and environmental quality management issues related to land use and land cover https://www.mrlc.gov/about.php. The consortium has taken an adaptive approach since the original NLCD product in 1992, continually updated methods and datasets to improve its representation of land cover. The emergence and expansion of industrial agriculture since 1992, the large size of individual CAFO sheds compared to NLCD pixels, and the distinct waste footprints of these operations all point toward CAFOs as a land cover category that could be incorporated into NLCD. For this to happen, we suggest first steps: 1. Remote sensing studies to identify the spectral signatures of CAFO structures, feedlots, waste lagoons, and wastewater spray fields 2. Field studies to identify the diffusive waste footprints of CAFOs and determine whether there are differences between CAFO type (i.e., swine, poultry, beef) or management techniques to delineate the boundaries of CAFO agroecosystem footprints. These steps can help researchers, managers, and decision-makers move forward with watershed and regional studies of potential CAFO impacts under current conditions as well as scenarios of potential future climate.

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ATTACHMENT 38

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Contents

Part 2 Breeding, animal husbandry and manure management

Series list

Title	Series number
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Achieving sustainable production of milk - Vol 2 Safety, quality and sustainability Edited by: Dr Nico van Belzen, International Dairy Federation (IDF), Belgium	009
Achieving sustainable production of milk - Vol 3 Dairy herd management and welfare Edited by: Prof. John Webster, University of Bristol, UK	010
Ensuring safety and quality in the production of beef - Vol 1 Safety Edited by: Prof. Gary Acuff, Texas A&M University, USA; and Prof. James Dickson, Iowa State University, USA	011
Ensuring safety and quality in the production of beef - Vol 2 Quality Edited by: Prof. Michael Dikeman, Kansas State University, USA	012
Achieving sustainable production of poultry meat - Vol 1 Safety, quality and sustainability Edited by: Prof. Steven C. Ricke, University of Arkansas, USA	013
Achieving sustainable production of poultry meat - Vol 2 Breeding and nutrition Edited by: Prof. Todd Applegate, University of Georgia, USA	014

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Introduction

Recent IPCC reports have highlighted the environmental impact of livestock production as a major source of non-CO_2 emissions: methane (CH_4), nitrous oxide (N_2O) and ammonia (NH_3). The livestock sector must react to these reports and develop or implement methods that can reduce greenhouse (GHG) emissions from livestock production.

Part 1 of this volume focuses on the analysis of greenhouse gas emissions from livestock, specifically drawing attention to the range of methods that can be used to reduce these emissions. Chapters in Part 2 discuss breeding, animal husbandry and animal management and how improving these elements can help to reduce the environmental impact of livestock production. Part 3 concentrates on nutritional approaches such as improving feed efficiency, forage quality and using plant bioactive compounds to reduce GHG emissions. Chapters also review the use of feed supplements and how modifying the rumen environment can also help to reduce GHG emissions.

Part 1 Analysis

Chapter 1 looks at the key techniques used for measurement of CH_4 and other gas emissions from livestock production, ranging from individual animal measurements to herd scale measurements for grazing animals and whole farm emissions such as feedlots. Individual animal measurement techniques discussed include whole-animal respiration chambers and head capture measurement. Herd scale measurements include micrometeorological methods and the eddy covariance technique.

Expanding on topics previously covered in Chapter 1, Chapter 2 discusses greenhouse gas emissions in livestock production, focusing specifically on modelling methods, methane emission factors and mitigation strategies. The chapter begins by reviewing systems analysis and how it can be used to quantify GHG emissions from livestock. It then looks at the various stages of life cycle assessment and how it can be used to analysis the environmental effects of livestock production. Modelling applications and the importance national greenhouse gas inventory submissions are also considered in the chapter.

Part 2 Breeding, animal husbandry and manure management

The first chapter of Part 2 analyses the contribution of animal breeding to reducing the environmental impact of livestock production. Chapter 3 begins by addressing the impact of livestock production on the environment. It then goes on to discuss the environmental impact of broilers, layer hens, pigs and dairy cattle and how improving breeding techniques for all of these species can help to reduce the emissions they produce. The chapter also highlights future research directions and provides resources for further information on the subject.

Chapter 4 focuses on the environmental impact consequences of endemic livestock health challenges that lead to deterioration in animal health, and on the potential impacts arising from their mitigations. The first part of the chapter concentrates on the potential of animal health to affect the environmental impact of livestock systems. It also reviews the literature to date which has quantified the impact of health challenges for the environmental impacts of livestock systems. The potential of successful health interventions to mitigate negative environmental impacts represents a point of synergy between concerns around environmental sustainability and animal welfare, both of which represent 'hot topics' in the discourse surrounding the livestock industry and its sustainability. The chapter concludes by highlighting the challenges associated with modelling health interventions and their potential to mitigate environmental impacts.

The subject of Chapter 5 is sustainable nitrogen management for housed livestock, manure storage and manure processing. The chapter begins by discussing the various forms nitrogen can take, focusing specifically on ammonia, nitrous oxide and di-nitrogen. It then goes on to review livestock feeding and housing for dairy and beef cattle, pigs and poultry. The chapter also examines manure storage, treatment and processing by discussing the principles of emissions produced from these processes as well as mitigation measures that can be used. It also addresses the best practices and priority measures for livestock feeding, housing and manure storage, treatment and processing.

Chapter 6 discusses developments in anaerobic digestion (AD) to optimize use of livestock manure, particularly the use of livestock manure in the production of biogas. The chapter begins by reviewing the quantities and risks of livestock manure, which is then followed by a discussion of the biogas potential of livestock manure. The chapter also examines mono- and co-digestion and the various factors that can affect the efficiency of anaerobic digestion. It also discusses the use of biogas slurry and residues. The chapter shows how AD can play an important role in promoting circular agriculture. A case study on the use of AD in practice in Henan Province in China is also included.

Part 3 Nutrition

Part 3 opens with a chapter that examines the impact of improving feed efficiency on the environmental impact of livestock production. Chapter 7 starts by discussing the relation between greenhouse gases and dairy production, highlighting how important it is to the dairy sector to find ways of decreasing greenhouse gas output. The chapter then moves on to discuss the origins of methane and reactive nitrogen excretions in ruminants. A section on improving feed conversion efficiency is also included, which is then followed by a review of the nutritional practices that can be used to enhance feed conversion efficiency and decrease methane excretion. The chapter also examines the nutritional practices that can be used to increase milk protein efficiency and nitrous oxide excretion as well. Discussions on genetics and feed conversion efficiency and postabsorptive metabolism and feed conversion efficiency are also provided.

Chapter 8 reviews grazing management strategies that can contribute to reducing livestock greenhouse gas emissions. Strategies discussed include grazing season length and timing as well as sward structure and quality, including dry matter and clover content. The chapter also discusses the use of condensed tannin legumes such as chicory and plantain, as well as measurement issues including life cycle assessment.

Chapter 9 focuses on the opportunity to use plant bioactive compounds in ruminant diets for their potential to mitigate greenhouse gas emissions, particularly enteric methane. Nitrous oxide emissions related to urinary nitrogen waste are addressed when information is available. The main families considered are plant lipids and plant secondary compounds (tannins, saponins, halogenated compounds and essential oils). The effects of these compounds in vivo, their mechanisms of action, and their potential adoption on farms are discussed, and future trends in this research area are highlighted.

The next chapter looks at the use of feed supplements to reduce livestock greenhouse gas emissions, specifically focusing on direct-fed microbials. Chapter 10 outlines the strategy of using feed supplements for the reduction of greenhouse gas emissions in ruminants, including methane (CH_4), carbon dioxide and nitrous oxide, given that feed intake is an important variable in predicting these emissions. The chapter focuses on direct-fed microbials, a term reserved for live microbes which can be supplemented to feed to elicit a beneficial response. The viability of such methods is also analysed for their use in large scale on-farm operations.

Chapter 11 focuses on modifying the rumen environment to reduce greenhouse gas emissions. Ruminants were among the first domesticated animals and have been providing food, leather, wool, draft and by-products to humanity for at least 10 000 years. However, rumen methanogens reduce CO_2 to CH_4 in association with other rumen microbes that generate substrates for methanogenesis. Consequently, other rumen microbiota can directly and indirectly impact the abundance and activity of methanogens. Enteric methanogenesis from ruminants accounts for approximately 6% of total anthropogenic greenhouse gases emissions and can represent from 2% to 12% of the host's gross energy intake. A myriad of strategies to mitigate CH_4

emissions have been investigated, but few have been adopted by industry. This chapter reviews rumen- and feed-associated factors affecting CH_4 production and outlines the challenges associated with achieving a reduction in enteric CH_4 emissions. The pros and cons of these strategies are discussed in an attempt to define the best approaches to mitigate CH_4 emissions from ruminant production systems.

Part 1

Analysis

Chapter 1

Measuring methane emissions from livestock

Trevor Coates, Agriculture and Agri-Food Canada, Canada; and Deli Chen and Mei Bai, University of Melbourne, Australia

1 Introduction

Methane (CH_4) gas was first isolated by the Italian physicist Alessandro Volta in 1776 and described as the 'inflammable air native of marshes'. Although recognized as a local gas associated with decaying biological matter, CH_4 was thought to be a relatively static and minor component of the atmosphere. It would be another 200 years before advances in gas chromatography (GC) allowed Rasmussen and Khalil (1981) to show that CH_4 concentration in the atmosphere was not static but was increasing by an estimated 2% per year. With continued atmospheric monitoring and a lengthening historic record derived from ice-core data, the nature of rising CH_4 concentration and its relevance to global warming became increasingly apparent.

The establishment of the Intergovernmental Panel on Climate Change (IPCC) in 1988 and a growing awareness of the need to curb emissions sparked a flurry of research related to cattle CH_4 emissions beginning in the 1990s. Mitigation of enteric CH_4 emissions and the development of measurement techniques to validate the effectiveness of mitigation practices continue to be ongoing areas of research. Strategies that alter the rumen environment and the digestion process (Hristov et al., 2013) through the use of feed additives

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(Grainger et al., 2008) can improve feed efficiency and decrease enteric CH_4 emissions per kg of meat produced. Farm management and grazing strategies can also influence CH_4 emissions and the genetic selection of more efficient cattle (kg CH_4 per kg live weight) is also an ongoing promising area of research (Basarab et al., 2013).

Mitigation of CH_4 emissions requires emission measurements that are sensitive enough to measure the difference between standard practices and proposed mitigation strategies. Emission measurement methods also need to be operational at a range of spatial scales from the individual animal to *in situ* measurements under typical animal management conditions. This chapter reports on key techniques used for measuring CH_4 emission in agriculture at a variety of spatial scales. Advances in these techniques and promising new approaches to measure CH_4 emissions are also discussed.

2 Individual animal measurement techniques: whole-animal respiration chambers and head capture measurement

2.1 Whole-animal respiration chambers

The first estimates of CH_4 emissions from ruminants came from chamber studies, long before CH_4 had ever been measured in the atmosphere. The animal nutrition laboratory of the Pennsylvania State College constructed a respiration calorimeter in 1902 as a key component to better understanding animal physiology and ruminant nutrition (Armsby and Fries, 1903). Animal CH_4 production was recognized as a loss in feed energy and the Armsby respiration chamber, as it came to be known, was instrumental in generating feed ration and nutrition guidelines for America's expanding cattle industry. Measurement of CH_4 emissions was accomplished by routing a known volume of chamber air to a combustion furnace (Fries, 1910) and recording the change in combustion end products. After many years of experiments, Bratzler and Forbes (1940) developed a simple model relating animal CH_4 production with carbohydrate intake.

Modern chambers are more sophisticated, offering fine control of temperature, humidity and airflow, and advances in sensor technology have allowed for analysis of more components. Construction of chambers represents a considerable expense, but the importance of CH_4 as a source of agricultural greenhouse gas (GHG) emission has hastened their development. Whole-animal respiration chambers are now found at animal research facilities throughout the world. Chamber results have contributed to the current understanding of animal energetics and are considered a 'gold standard' measurement technique. Chambers offer a direct measure of emissions with few assumptions and a methodology that can be easily validated through gas release and recovery tests (McLean and Tobin, 1988). While whole-animal respiration chambers have been extremely valuable for mitigation work

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and quantifying treatment effects, the chamber represents a constrained environment, and it is less certain to what extent results can be extrapolated to actual cattle production systems (Johnson et al., 1994).

2.2 Head capture measurement

Measurement techniques capable of operating within real agricultural production environments are necessary for validating methane mitigation measures under typical animal management conditions. As an alternative to large animal chambers, a variety of systems have been designed to measure emissions, principally through focused airflow and concentration measurements of the area around the animal's head. These methods include:

- sniffer methods, where a sampling unit is incorporated into feed troughs;
- ventilated hood or headbox systems, which provide a more controlled environment but allow the animal access to food and water; and
- mask systems, which are fitted to the animal's nose and mouth.

The latter two techniques are also known as flux methods since they involve greater control of the airflow to capture emitted gases and measure CH_4 fluxes. A different approach is the use of handheld laser methane detectors (LMD) which are pointed by an operator at an animal's nostrils to measure methane column density along the length of the laser beam.

These techniques can be used within existing barn facilities and, depending upon the design, can be used to measure emissions continuously over a 24-h period or through spot measurements over the course of the day (Hammond et al., 2016a; Kebreab, 2015). Similar to respiration chamber measurement, these techniques can be affected by decreased feed intake, and intensive training is required for animals to become familiar with the hood apparatus, making it impractical to measure large numbers of animals (NASEM, 2018).

Sniffer methods are based on continuous breath analysis of exhaled air from animals using feed troughs in environments such as automated milking systems. A sampling unit is placed in the feed trough, and the air around the animal's muzzle is continuously monitored during feeding. Sensor systems detect the animal and activate breath analyzers located in the troughs, including Fourier transform infrared (FTIR) and non-dispersive infrared (NDIR) techniques. Measurements can then be used to develop an index of CH_4 emissions during milking as a product of peak frequency and mean peak area of CH_4 concentration (Garnswothy et al., 2012), or using the ratio of CO_2 to CH_4 (Lassen et al., 2012; Lassen and Løvendahl, 2016; Bell et al., 2014). Sniffer methods may be more affected by variable air-mixing conditions due to factors such as the geometry of the feed trough, muzzle position and movement, suggesting that flux techniques are more reliable (Huhtanen et al., 2015). However, recent research suggests that results from sniffer and flux methods are both comparable with each other and with respiration chambers, suggesting a growing degree of accuracy (Sorg et al., 2018; Difford et al., 2018).

The GreenFeed system (GF) (C-Lock Inc, Rapid City SD, USA) incorporates elements of the ventilated hood chamber into an automated feeder that dispenses a programmed amount of pelletized feed as bait to encourage visits to the GF. The GF is a robust system and can be incorporated into the production environment with one GF unit capable of measuring many animals consecutively. A proximity sensor in the head chamber identifies the visiting animal through its ear tag and initiates a gas sampling routine during which bait pellets are dispensed to keep the animals head in the feeder for 3–7 min during which time an emission rate is calculated. The procedure for deriving emission rates is reported by McGinn et al. (2021).

The GF unit can be programmed to limit the number of permitted visits per day but this measurement method is dependent on the animal's desire for the bait in the feeder, and the actual number of visits each animal makes per day will vary as will the number of animals that visit the device. For this reason, measurements are typically accumulated over several weeks to establish a daily emissions pattern for each animal that regularly uses the GF (Hammond et al., 2016b; Hristov et al., 2015; Huhtanen et al., 2019). When using spot measurements to determine daily emissions, care must be taken to prevent sampling bias by ensuring sampling times are appropriate for the daily feeding cycle of the animals using the device (Hammond et al., 2016a). As with ventilated hood chambers and head masks, the GF system is unable to capture the small emission eructed through the rectum, which has been reported to be between 4% and 8% of the total emission from cattle nose, mouth and rectum (Grainger et al., 2007b; McGinn et al., 2006a; Ulyatt et al., 1999). Muñoz et al. (2012) assumed 3% ± 1.5% emission was from the rectum.

3 Individual animal measurement techniques: tracer techniques

Tracer techniques rely on the co-location of a tracer gas source (with a known release rate) and the source to be measured, based on the assumption that both gases will be transported in the atmosphere in the same manner. Concentration measurements of the tracer gas and the source gas are made at some distance downwind. The ratio of gas concentrations is used with the known release rate of the tracer to determine the emission rate. Tracer techniques offer the advantage of a strictly ratiometric measure independent of meteorological conditions.

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3.1 Sulfur hexafluoride tracer technique

The sulfur hexafluoride (SF_6) tracer technique (Johnson et al., 1994) was developed to overcome the limitations of chamber measurements, providing individual animal emission estimates without constraints on the animal's typical behavior in the production environment. The technique requires placing a small permeation tube in the animal's rumen which emits SF_6 at a pre-calibrated rate. This tracer gas is expelled through eructation along with the CH_4 produced in the rumen, while a collection device attached to the animal slowly draws air, usually for 24 h, from the nose and mouth region through an intake affixed to a halter. This technique provided the first measurements of animal emissions from grazing systems (Lassey et al., 1997; McCaughey et al., 1997). It also provided a means to monitor the dynamics of pasture conditions by conducting short measurement programs over a year (Pavao-Zuckerman et al., 1999; Ulyatt et al., 2002).

The technique gained in popularity as many agricultural research facilities already had the capacity for gas chromatographic analysis of the air samples collected with the SF_6 technique. With a modest outlay to construct the sampling apparatus and prepare the permeation tubes, a technique was now available to obtain measurements of animal CH_4 emissions from their typical production environment. With an increasing number of users, the technique was improved through a better understanding of animal emission variability compared with chamber measurements (Grainger et al., 2007a), the release characteristics of the permeation tubes with time (Lassey et al., 2001), the effect of permeation rate on emissions estimates (Vlaming et al., 2007), the effect of background measurements (Williams et al., 2011) and the importance of the sample collection rate (Deighton et al., 2014).

The SF_6 tracer technique has proven valuable for mitigation work and is well suited to measurements on small groups of animals, particularly within dairy systems where animals are accustomed to daily handling and sampler changes can be coordinated with daily milking. Implementation in grazing systems is more problematic as cattle require extensive training to become accustomed to handling and the fitting of yokes (DeRamus et al., 2003). The technique is also limited by higher between-cow variability in measurement accuracy (Pinares-Patiño et al., 2011). In addition to the permeation tube approach, SF_6 has also been used as a tracer gas to estimate CH_4 emissions from whole farms by releasing gas along barn vents and pen railings (McGinn et al., 2006b) and collecting downwind air samples for GC analysis of SF_6 and CH_4.

3.2 Other tracer techniques: nitrous oxide-tracer Fourier transform infrared spectroscopy

Tracer studies have also been carried out using open-path FTIR and nitrous oxide (N_2O) as tracer gas (Griffith et al., 2008). The open-path FTIR has proven to be a robust instrument for trace gas studies with stable performance and high precision of concentration measurement. Concentrations of CH_4 and N_2O can be measured by one FTIR concentration sensor at a short interval (e.g. 3-min interval). The CH_4 emission rate (Q_{CH4}) is calculated following Eq. 1:

$$Q_{CH_4} = Q_{N_2O} * \left(\Delta CH_4 / \Delta N_2O\right) \tag{1}$$

where Q_{N2O} is the known N_2O release rate and ΔCH_4 and ΔN_2O are the enhanced mixing ratios of CH_4 and N_2O above the local background level.

Griffith et al. (2008) monitored grazing dairy animals by affixing release points along a fence line of a grazing paddock. Animals were confined within a narrow paddock with respect to the prevailing wind so as to minimize the tracer to animal distance and reduce errors arising from the separation of the emission source and the tracer. This technique has also been utilized with gas release canisters affixed to halters on individual animals (e.g. dairy cattle and sheep) (Bai, 2010; Jones et al., 2011) to better collocate the tracer with the emission source. The tracer canisters emit tracer gas N_2O at a known rate similar to SF_6 permeation tubes, and the concentrations of CH_4 and N_2O are measured simultaneously downwind of the animals with an open-path FTIR instrument. This provides a herd-emission rate rather than individual animal emissions. Similar to the SF_6 technique, the method requires daily animal handling for canister replacement. The N_2O tracer-FTIR approach and its need for animal confinement and daily animal handling make this technique more suited to short-term intensive field campaigns. Care must also be taken to ensure the measurement paddock is not a significant source of N_2O as might arise with an irrigated/fertilized grazing paddock.

Another application of the ratiometric technique was reported by Laubach et al. (2014) and McGinn et al. (2019). The ratio of the above background concentration of a target gas measured downwind from two different sources (a control and a treatment) is used to infer the emission reduction of the treatment (numerator) as a fraction of the control (denominator). The assumption used was that the wind flow and physical dimensions of the two sources were identical, thus satisfying the requirement of a single gas transfer coefficient for the calculation of both emissions.

4 Herd-scale measurement techniques: micrometeorological methods

Micrometeorological methods (MM) are by nature non-interference techniques and the most applicable tool for herd-scale emission measurement. In principle, they can be used to study animals at the farm scale without the requirement for animal handling. MM require measurements of atmospheric CH_4 concentration and meteorological variables above or downwind of the animals to infer emissions.

McGinn (2006), in a review of MM techniques, suggested that the mass difference (MD) technique, the integrated horizontal flux (IHF) technique and inverse dispersion (ID) methods were all appropriate choices for measuring emissions of free-ranging animals. The MD and IHF techniques are both mass balance (MB) methods where differences in concentration profiles upwind and downwind of the source are used with a wind speed profile to infer emissions. The MD technique was first proposed by Denmead et al. (1998) and subsequently used for estimating CH_4 emissions from sheep (Leuning et al., 1999) and from grazing animals (Harper et al., 1999). Instrumentation and set-up demands were significant in each of the studies, requiring pumps and switches to manage multiple sampling lines on each face of the pen running to a central CH_4 analyzer. The IHF technique similarly requires logistically complex profile measurements, and as with MD, it is spatially constrained by the downwind profile height, and the source area must be kept small. Application on a grazing landscape requires the animals to be confined, and this effectively restricts both the number of animals and the length of time for measurement.

ID methods rely on atmospheric dispersion models to compute the theoretical relationship between concentration and emission rate from a defined source. Flesch et al. (1995) described a backward-time Lagrangian stochastic model and its application for deriving emissions requiring only a single-concentration measurement of the plume, a known background concentration and wind statistics recorded with a three-axis sonic anemometer. This model has since been incorporated into a commercial software platform WindTrax® (http://www.thunderbeachscientific.com/) providing a convenient tool for mapping source and sensor locations and visualizing model runs. A model run consists of thousands of ‘particles’ (fluid elements) released from the sensor and followed backward in time with trajectories that are consistent with the averaged wind and turbulence statistics of the surface. Particles that ‘touch down’ within the defined source area are accumulated and their velocity at touchdown is used in the final calculation of the ratio of concentration to flux. The model can also operate in a forward mode where particles are ‘released’ from the source and any trajectories that intercept with the sensor volume are accumulated. This technique has since become the most popular MM technique for estimating animal methane emissions (Table 1).

Open-path sensors such as tunable diode lasers (TDL) and FTIR spectrometers have been used because of their ability to spatially integrate concentration measurements and are ideally paired with the WindTrax software. Similar to the MB methods, ID modeling requires a known background measurement and sensors with sufficient sensitivity to accurately sense the increase above background due to the source. Confinement of animals may be

Table 1 Micrometeorological techniques used for monitoring cattle methane emissions

Author	Technique†	Sensor	# of animals	Confinement	Stocking density*	Duration
Judd et al. (1999)	FG	GC	55‡	Paddock (3 ha)	18.6	5 d
Leuning et al. (1999)	MD	FTIR (closed path)	14‡	Pen (500m^2)	289	5 d
Harper et al. (1999)	MD	Gas analyser/tubing	4	Pen (500m^2)	80	3–4 d
Laubach and Kelliher (2005)	ID/IHF	Open-path laser	269–287	Paddock (1.7 ha)	164	2 wks
McGinn et al. (2006b)	ID	Open-path laser	321	Dairy farm	186	1 mon
Griffith et al. (2008)	Tracer/IHF	FTIR (closed path)	23–50	Paddock (210 m^2)	1095–2381	2 wks
McGinn et al. (2009)	ID	Open-path laser	60	Pens (2 × 252 m^2)	1190	3 wks × 3
Bjorneberg et al. (2009)	ID	Open-path FTIR	780	Dairy farm	167	1 wk × 4
Laubach (2010)	ID	Open-path laser	58	Paddock (0.35 ha)	166	4 d × 4
Dengel et al. (2011)	EC	Open-path analyser	10–120‡	Paddock (5.4 ha)	1.8–22	60–277 d
Gao et al. (2011)	ID	Open-path laser	700	Feedlot	316–346	4 wks × 2
Tomkins et al. (2011)	ID	Open-path laser	18	Paddock (1ha)	18	1wk × 5
Tallec et al. (2012)	EC	Closed-path analyser	5	Pen (400 m^2)	125	8 d
Laubach et al. (2013)	MB/ID ID	Closed-path analyser, Open-path FTIR Open-path laser	61	Paddock (3772 m^2)	162	1 wks × 3
Laubach et al. (2014)	ID	Closed-path analyser	30	Paddock (2000 m^2)	150	16 d
McGinn et al. (2014)	ID	Open-path laser	40	Paddock (1 ha)	40	1 wk × 4
VanderZaag et al. (2014)	ID	Open-path laser	149–245	Dairy farms	-	3 wks × 4
McGinn et al. (2015)	ID	Open-path laser	60–40	Paddock (6 × 1 ha)	30–10	5 d × 3
Tomkins and Charmley (2015)	ID	Open-path laser	60	Pen (600 m^2)	1029	2 wks

Bai et al. (2015)	ID	Open-path FTIR	17500	Feedlot	738	2 wks
Felber et al. (2015)	EC	Closed-path analyser	17–20	Paddock (0.6 ha)	28–33	7 mon
Bai et al. (2016)	ID	Open-path FTIR	28	Pen (400 m^2)	700	6 wks
Dumortier et al. (2017)	EC	Closed-path analyser	~30	Paddock (4.2 ha) Confined 1.7 ha	~7 >17	19 mon
Taylor et al. (2017)	EC	Closed-path analyser Open-path analyser	264	Paddock (6–30 ha)	3–18	12 mon
Prajapati and Santos (2017)	EC	Closed-path analyser	58000	Feedlot (59 ha)	983	10 mon
Prajapati and Santos (2018)	EC/ID	Closed-path analyser	24116	Feedlot (59 ha)	455–529	10 mon
Coates et al. (2018) McGinn	EC	Open-path analyser	20	Paddock (15 ha)	1.3	7 wks
and Flesch (2018)	ID	Open-path laser	5190	Feedlot A	12.5–16.2	90 d
			8110	Feedlot B	10.0–13.6	20 d
McGinn et al. (2019)	ID	Open-path laser	666-678	Pens (2 × 1200 m^2)	560	23 wks
Prajapati and Santos (2019)	EC	Open-path FTIR Closed-path analyser	24116	Feedlot (59 ha)	455–529	7 d 10 mon
Todd et al. (2019)	EC/Tracer ID	Open-path analyser Open-path laser	12 50–12	Paddocks	1.89 1.95 75	8 9 7
Dumortier et al. (2019)	EC/Tracer	Open-path analyser	9	Paddock (4.2 ha)	2.2	27 d

* Stocking density (cattle/ha) refers to stocking density of measurement area (i.e. pen or paddock) † EC = eddy covariance, MD = mass difference, MB = mass balance, FG = flux gradient, IHF = integrated horizontal flux, ID = inverse dispersion ‡ denotes sheep.

required on a grazing landscape to achieve a sufficient rise in concentration and to ensure an uncontaminated background (prevention of wandering animals). McGinn et al. (2015) demonstrated that ID was effective with confined grazing animals at a stocking density of ten animals/ha but measurement of grazing cattle emissions remains a challenge due to the spatial limitations of sensor footprints and the need to manage cattle according to the changing pasture conditions over the grazing season (Todd et al., 2019).

Overall, reviews of MM suggest that results are comparable with other emission monitoring methods and verification of MM can be achieved through controlled gas release/recovery experiments (McGinn, 2013). The use of MM to distinguish treatment effects is more challenging due to the number of variables that must be considered but with care this can be achieved (Laubach et al. 2013; McGinn et al., 2019).

5 Herd-scale measurement techniques: the EC technique

The theory behind the EC technique is based on fundamentals of turbulent transport based on the early work of Reynolds (1894) and Taylor (1915) (Taylor, 1938) that set the theoretical framework for EC. However, it would be many decades before sensors were developed that could confirm the theoretical functions and explore potential applications. The flux of a given scalar quantity (F_s) can be described simply by Eq. 2:

$$F_s = w\rho_s \tag{2}$$

where w is the vertical wind velocity and ρ_s is the molar density of the trace gas of interest. Applying Reynolds decomposition, the above can be separated into a mean component (product of the mean vertical wind and the mean gas density over an averaging period) and a fluctuating component (the product of the deviations from the mean for the same variables $w'\rho_s'$) to give Eq. 3:

$$F_s = \overline{w\rho_s} + \overline{w'\rho_s'} \tag{3}$$

where $\overline{w\rho_s}$ represents the product of the mean vertical wind and the mean molar gas density, and $\overline{w'\rho_s'}$ represents the mean product of the instantaneous deviations from the mean for the same components.

One of the assumptions made to simplify the above equation is that, over a set averaging period, the mean vertical wind speed should be zero and that, as part of processing, the turbulence components are processed through a coordinate rotation such that the mean vertical wind velocity for the measurement period is zero. With the first component removed, the basic equation for EC becomes (Eq. 4):

$$F_s = \overline{w'\rho_s'} \tag{4}$$

where the flux of the gas component is calculated as the mean covariance between the instantaneous fluctuations of the vertical wind velocity and the molar density of the trace gas.

5.1 The use of the technique in agriculture

Although the theoretical underpinnings of EC were in place early in the 20th century, the instrumentation to measure the fluctuating components of wind and concentration with a time response sufficiently fast to capture the range of eddy sizes would not be available for several decades. The first EC measurements associated with agriculture explored fluxes of heat, momentum and water (Swinbank, 1951). Early attempts were made to measure CO_2 fluxes over an agricultural crop using a propeller-type anemometer to record vertical wind fluctuations (Desjardins, 1974) but a loss of high-frequency data associated with the slow response times of the instrumentation led to an underestimate of fluxes.

EC techniques increased in popularity throughout the 1980s as the development of the microprocessor facilitated the manufacture of three-axis sonic anemometers. This coupled with sensor improvements and digital data acquisition systems led to further use of EC to generate flux data over crops (Anderson et al., 1984; Desjardins et al., 1984). As the availability of robust commercially available sonic anemometers and stable fast-response sensors increased, so too did the number of monitoring sites. The accumulation of data from carbon balance studies and the need for spatial integration across various ecosystems led to the development of regional and global flux networks throughout the ’90s (Baldocchi, 2003).

With the rapid expansion of flux networks and the accumulation of long-term flux data, there came a need for a more standardized approach to data analysis to allow meaningful comparison between sites. Quality control testing and assessment of data quality became an increasing concern (Foken and Wichura, 1996; Vickers and Mahrt, 1997). This emphasis on the documentation of analysis procedures led to the convergence of analysis protocols and to the development of a number of software packages designed to incorporate analytical processing options, quality-control tests and sensor-dependent corrections (Table 2). The use of a standardized analysis platform allows for a more consistent approach to data processing. Data can be shared in a manner that clearly documents the analysis steps taken. A programmed approach also aids in the reanalysis of datasets when software is updated to incorporate improvements in processing steps or additional corrections.

5.2 The use of EC techniques to measure livestock emissions

The vast flux networks that began in the ’90s were focused on characterizing ecosystem productivity through measurement of the carbon, water and

Table 2 A selection of freely available software packages for eddy covariance analysis

Software Package	Maintained by	Website
AltEddy	Wageningen University, The Netherlands	http://www.climatexchange.nl/projects/alteddy/
EdiRe	University of Edinburgh, Scotland	http://www.geos.ed.ac.uk/homes/rclement/micromet/EdiRe/
TK3	University of Bayreuth, Germany	https://zenodo.org/record/20349#.WKfwAG995hE
EddyUH	University of Helsinki, Finland	https://www.atm.helsinki.fi/Eddy_Covariance/EddyUHsoftware.php
ECO_2S	University of Tuscia, Italy	http://gaia.agraria.unitus.it/eco2s
EddyPro	LI-COR Biosciences, Lincoln Nebraska, USA	https://www.licor.com/env/products/eddy_covariance/compute.html#eddypro

energy exchanges between the homogenous extensive surfaces (grasslands, forests, etc.) and the atmosphere. This decade, however, also saw the first measurements of CH_4 fluxes using EC with closed-path CH_4 analyzers (Denmead, 1991; Verma et al., 1992). These first-generation analyzers were lab-based instruments requiring temperature control, mains power and pumps to draw air through the analyzer, and field measurements were limited to short-term intensive testing. The development of a low-power open-path CH_4 analyzer (McDermitt et al., 2011) has since enabled the deployment of EC systems to monitor CH_4 fluxes from a variety of landscapes including wetlands (Matthes et al., 2014), rice paddies (Alberto et al., 2014) and tundra (Raz-Yaseef et al., 2016).

Although EC has grown to be a well-tested micrometeorological technique for measuring exchanges between the surface and the ground, this is typically done in association with an extensive homogenous source, and fluxes are calculated on an area basis. In fact, this calculated flux will represent a spatially weighted average of surface fluxes from a defined portion of the underlying surface. Areas of the surface that contribute to the calculated flux (and their relative contributions) constitute the flux footprint (Schuepp et al., 1990). Typically, the footprint encompasses the immediate vicinity of the sensor and extends upwind. The shape of the footprint and its upwind extent vary with sensor height, aerodynamic roughness of the surface and atmospheric conditions but are generally in the range of a few hundred meters. Atmospheric stability plays a large role, however, with the footprint contracting substantially in unstable conditions and increasing potential to hundreds of meters during stable conditions (Göckede et al., 2004). When EC is used over a homogenous extensive surface, the calculated vertical flux is representative of the surface-emission rate, expressed in terms of g m^{-2} s^{-1}. When EC is used to infer emissions

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from point sources (or spatially limited area sources), the relationship between emission and flux is more complex as the contribution of these distinct sources to the EC flux will be dependent on the source location within the footprint.

Grazing systems represent a challenging and complex environment for the application of EC. The sensitivity of EC to the presence of grazing animals on the landscape has created difficulties for those who are interested in ecosystem fluxes. Herbst et al. (2011) found CH_4 peaks in their flux data from wetland emissions and utilized a camera system to show that these anomalies were correlated with grazing animals upwind of their measurement tower. Baldocchi et al. (2012) found that the presence of cows on the landscape was easily detected in their flux measurements and became a challenge in monitoring the emissions from a peat landscape as the influence of nearby cows resulted in fluxes many times higher than the typical landscape fluxes they sought to measure. A tower-mounted camera was installed so that periods with animals in the vicinity could be eliminated from the dataset, although the authors suggested they could, at a future date, use this data to estimate animal emissions.

Intentional measurement of ruminant emissions using EC was first performed by Dengel et al. (2011) who used an EC flux tower to capture CH_4 fluxes of a sheep-grazed paddock. The movement of sheep in and out of the measurement footprint led to high variability in the individual analysis periods, and no means of recording animal locations were available. However, by integrating the fluxes over a 7-month measurement period and dividing by the stocking density, a rough estimate of emissions was obtained. Grazing animals introduce extreme temporal and spatial heterogeneity on the landscape, and CH4 fluxes measured by EC fluxes will fluctuate widely depending on the number of animals in the footprint and their locations relative to the sensor (Fig. 1).

The application of EC to estimate emissions in the grazing environment will generally require an analysis of the footprint and its overlap with animal positions. Tallec et al. (2012) accounted for animal position by confining groups of animals in small pens around a central tower. Felber et al. (2015) used EC with free-ranging cattle and tracked animal position with global positioning system (GPS) collars. Both Tallec et al. (2012) and Felber et al. (2015) used the footprint weighting tool of Neftel et al. (2008), based on the 2D analytical footprint model of Kormann and Meixner (2001), as the basis of interpretation of EC fluxes to generate emission estimates. While the authors concluded that EC was sufficiently accurate for animal emission studies, a systematic underestimation of emissions when animals are far from the tower was noted. Felber et al. (2015) proposed that the use of a more sophisticated Lagrangian footprint model (as used in the present work) could yield more accurate estimates. Recent studies highlight the value of the technique in

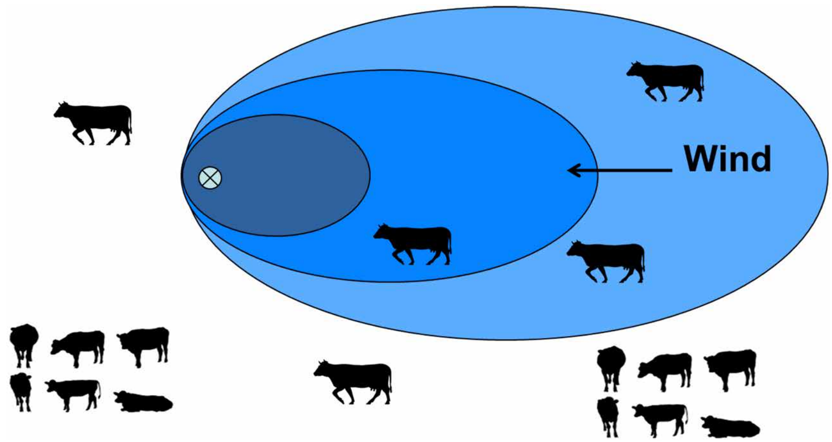

Figure 1 Representation of the flux footprint on a grazing landscape. In this representation only three animals contributed to the flux. Animals closer to the measurement point (marked by the x) contributed more than those further away.

assessing emissions in grazing environments but the challenges in accounting for variables such as wind and other atmospheric variables as well as animal distribution and movement continued (Prajapati and Santos, 2017, 2018; Coates et al., 2018).

6 Conclusion and future trends

Measuring methane emissions from livestock remains challenging yet there is a compelling urgency to continue to refine existing techniques and explore new opportunities as technologies evolve. The ability to reliably and repeatedly make valid measurements under a range of animal production scenarios is vital to support ongoing GHG mitigation efforts from the livestock sector. The expanding options for measurement are encouraging and the research community will continue to explore techniques best suited to their unique access to available equipment and expertise. The precision of respiration chambers will continue to play an important role in demonstrating effectiveness of new diet strategies for methane mitigation. Other techniques more applicable to actual farm conditions and a higher throughput of measurements have a role to play by accounting for animal variability. Recent studies comparing respiration chambers with the LMD, sniffer, SF_6 and GF techniques have found that results from the latter techniques correlated well overall with those from respiration chambers, but there was a lower correlation between techniques and degree of repeatability within individual techniques (with head enclosure techniques such as GF and SF_6 performing best) (Garnsworthy et al., 2019; Zhao et al., 2020). This suggests that a way forward is to combine techniques with appropriate weightings and target them for specific applications, whether emissions monitoring (using techniques such as respiration chambers and head enclosure methods) or using applications such as genetic evaluation in breeding cattle with lower emissions (using techniques such as GF, sniffer or LMD methods). MM methods will continue to be important for herd scale emissions and in understanding the relationship between animals and the rangeland ecosystem. Results from analytical methods can also be used both to validate and inform the range of increasingly sophisticated models predicting GHG emissions from livestock, allowing rapid analysis of a range of scenarios and parameters to help farmers improve their operations (Jose et al., 2016).

7 Where to look for further information

The following books provide helpful overviews on respiration chamber measurements, hood measurements and the SF_6 tracer technique:

- Animal and Human Calorimetry (McLean, J. A. and Tobin, G. 1988. Cambridge University Press, Cambridge).
- Measuring methane production from ruminants (Makkar, H. P. and Vercoe, P. E. 2007. Springer).

The following references provide valuable insights into meteorological approaches for flux measurement:

- Harper, L. A., Denmead, O. T. and Flesch, T. K. 2011. Micrometeorological techniques for measurement of enteric greenhouse gas emissions. *Animal Feed Science and Technology*, 166–67: 227–239.
- Rochette, P. and McGinn, S. M. 2005. Methods for measuring soil-surface gas fluxes. In Alvarez-Benedí, J. and Munoz-Carpena, R. (eds), *Soil-water-solute process characterization: an integrated approach*. CRC Press, Boca Raton: Florida, USA.

For up to date information on research initiatives and international conferences and workshops related to greenhouse gas emissions from agriculture:

- https://globalresearchalliance.org/.

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Chapter 2

Greenhouse gas emissions from livestock production: modelling methods, methane emission factors and mitigation strategies

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1 Introduction

An integral component for quantifying greenhouse gas (GHG) emissions from livestock production systems is whole-farm modelling. In contrast to the International Panel on Climate Change (IPCC) method, whole-farm modelling does not specify the estimation of GHG emissions by sector, but by the definition of system boundaries (Schils et al., 2005). This allows a complete analysis of methane and GHG emissions that is not possible within the framework of the IPCC method, as on-farm GHG emissions emanating from livestock farming systems are reported in three different sectors (Soussana et al., 2010). These sectors are agriculture, land-use change and forestry, and energy. Furthermore, the IPCC method is limited to national GHG emissions. Thus, even if GHG emissions from national sectors are combined to quantify total emissions, any emissions generated outside of the national boundaries are not included (Cerri et al., 2009). For example, a large proportion of concentrate feeds used within winter milk production systems are sometimes produced outside a nation's borders, but the associated GHG emissions from

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cultivation and harvesting are included in the inventories of the nation(s) that produce the concentrate feeds instead of the inventory where they are consumed.

An alternative to the production-focussed IPCC method is consumption-based GHG accounting. Peters and Hertwich (2008) outlined the methodology in detail and argued that emissions are driven by consumption, not by production. Estonia has demonstrated it is possible to use the consumption approach at a national scale to quantify GHG emissions (Gavrilova and Vilu, 2012). Quantifying national GHG emissions using both methods, that is, production and consumption could potentially identify any major transfers of GHG emissions from one nation to another (carbon leakage). This is an issue as some nations have less focus on reducing GHG emissions than others. The consumption approach, however, requires the development of harmonized farm models capable of quantifying GHG emissions associated with the life cycle of goods and services, that is, carbon footprint and methane emission intensity (Peters, 2008). Whole-farm models often use LCA methodologies to quantify emissions, and LCA methodologies often use farm models to complete the LCAs. Both of these modelling methods use a systems approach to quantify GHG emissions. When these modelling approaches are applied using the same set of assumptions (e.g. boundaries, unit of expression, methane emission factors), the results should be similar. This chapter will discuss the systems analysis and life cycle assessment modelling approaches and will go on from there to look at a range of model applications. These applications include use within the national inventories of various countries across species, including discussions around the use of different emission factors. The chapter concludes with applications to quantify emissions at the farm level and a discussion around some of the mitigation strategies that have been modelled.

2 Systems analysis

Systems analysis has been widely used by researchers to quantify GHG emissions from the ruminant system and assess methane mitigation strategies (Beukes et al., 2010; O'Brien et al., 2011; Clarke et al., 2013). The main stages of system analysis focus on the definition of a conceptual framework, with the precise aim and boundary of the model and its application.

2.1 Conceptual framework

In general, the first step of systems analysis is to formulate a conceptual model of the farming system of interest. The delimitation of the system boundaries of conceptual models is determined by the objective of the study and the precise aim of the model use. The objective of the analysis required should be used when decided on an appropriate model to complete an analysis. Generally, whole farm models that have quantified GHG emissions from livestock production systems estimate emissions from on- and off-farm sources related to the livestock product(s) up to a point it is exported from the farm (Olesen et al., 2006). Off-farm emissions from the production of external farm inputs such as concentrate feed and fertilizers are included with methane and other GHG emissions.

2.2 Model development

The second step of systems analysis entails creating a mathematical model of the production system defined. During this stage a series of algorithms are assembled and connected to mathematically model the farming system. In most cases, these equations are based on empirical relationships from representative field studies (Shalloo et al., 2004). Occasionally, experimental data from the farming system under study is used to estimate methane and other GHG emissions (Schils et al., 2005; del Prado et al., 2013). Within livestock system simulation models, emission factors in conjunction with livestock activity data are used to compute farm emissions. Sometimes these emission factors are obtained from the IPCC (2006) guidelines for on-farm emission sources. However, the IPCC emission factors are designed to enumerate national-level emissions. Thus, they often lack the refinement, model functions and emission factors, necessary to quantify the effect that changes to the production systems have on GHG emissions from individual farms (Schils et al., 2006). As a result, direct emission measurements, published in the scientific literature, are sometimes used to generate emission factors for subsequent inclusion in models in the pursuit of assessing on-farm emissions. In the case of off-farm emissions, almost all emission factors are obtained from literature sources or databases, for example, Ecoinvent (2010). Normally, data collected on-farm or representative information such as regional statistics are used as input data to operate whole-farm GHG models. For bovine production systems, several models have been developed to quantify GHG emissions from conventional pasture-based or confinement systems and organic systems (Beukes et al., 2010; Rotz et al., 2010; O’Brien et al., 2011). Results from previous livestock GHG models have been expressed per farm, per hectare of farmland and per unit of product, for example, per kg of fat and protein corrected milk (FPCM) or kg carcass weight (Thomassen et al., 2008; Clarke et al., 2013).

3 Life cycle assessment

Life cycle assessment considers the environmental effects of a product or service system (ISO, 2006a). The method also adopts a systems approach, but in contrast to systems analysis, the general step and principles of the LCA methodology are internationally standardized (ISO, 2006a,b). The International Organisation for Standardisation (ISO) originally developed standards for LCA in 1997, which were subsequently revised in 2006 (ISO 14040-14044). The main phases of LCA accordingly are goal and scope definition, life cycle inventory analysis, life cycle impact assessment and life cycle interpretation.

3.1 Goal and scope definition

This stage requires clearly stating the aims and objectives of an LCA project and the intended audience (ISO, 2006a). The goal definition determines what level of detail and what accuracy is needed for prediction/analysis. The scope of an LCA study should clearly describe the system under study and define the boundaries of the studied system (ISO, 2006a). Typically, the system boundaries of livestock LCA studies are defined to assess GHG emissions from all processes up until the point the primary product is sold from the farm (Beauchemin et al., 2011). This is commonly referred to as a 'cradle to farm gate' LCA. Some studies have also analysed further production stages, for instance, the processing stage and distribution to the retailer (Berlin, 2002; Hessle et al., 2017). The main environmental impacts evaluated in previous LCA studies of livestock are GHG emissions (global warming potential), acidification potential, eutrophication potential, land use and energy use (de Vries and de Boer, 2010; de Vries et al., 2015).

3.2 Life cycle inventory analysis

The second phase of LCA involves the compilation of inputs, outputs and emissions for a given product system throughout its lifecycle (ISO, 2006b). The aim of this stage of LCA modelling is to develop a model that quantifies the different resources used and the amount of waste and emissions generated per functional unit (Rebitzer et al., 2004). Resources used on-farm are normally collected directly or computed using relevant data sources. Emissions from on-farm processes are mainly estimated by applying emission factors from the literature or the IPCC (IPCC, 1997, 2006). For most LCA studies of livestock systems, international databases, for example, Ecoinvent (2010) or literature sources are used to estimate the resources used and emissions generated from processes that are indirectly related to the production system of interest, for example, data on fuel and fertilizer production.

3.3 Life cycle impact assessment

The inventory analysis phase lists the various substances used and pollutants emitted from a livestock production system (Thomassen et al., 2008). These results are generally difficult to interpret. Thus, a further stage known as life cycle impact assessment is needed to complete LCA of livestock systems (ISO, 2006a,b). This phase aggregates resources and emissions from the inventory analysis phase and computes (characterization) various potential environmental effects (Guinee et al., 2002). Environmental impacts are computed by converting the results of the inventory analysis stage using relevant characterization or equivalency factors. For instance, the global warming potential metric in CO_2-equivalents is applied during this stage to assess the climate impact of methane and other GHG emissions (Basset-Mens et al., 2009). The life cycle impact assessment stage allows the environmental effects of a livestock system to be assessed in a more interpretable way.

4 Modelling applications

Recent LCA and whole-farm GHG models from cool or temperate livestock regions were assessed. Table 1 provides a description of the modelling methods and emission factors used for analysis in 11 dairy studies, 9 beef studies, 3 sheep studies and 2 pig studies. In many situations, models calculated GHG emissions according to the approaches reported in national GHG inventories and the IPCC guidelines. A few studies used alternative equations for these sources such as Capper et al. (2009) or measured emissions directly as part of an on-farm research trial, for example, Doreau et al. (2011). Whole farm or LCA models were used for the following purposes:

- Quantify the environmental performance of farm systems and the effect of mitigation strategies on GHG emissions;
- Estimate the environmental sustainability of commercial farmers, for example, Bord Bia sustainable livestock assurance schemes; and
- Investigate the effect modelling decisions have on GHG emissions through sensitivity analysis.

4.1 Farm systems considered

Studies modelling GHG emissions from livestock farms have compared a variety of different systems including organic and conventional production systems, extensive and intensive systems, and confinement and grazing systems. These comparisons were generally representative of farms for a particular region. Many modelling studies that compare contrasting production systems aimed to assess the effect of intensification, defined as increased use of inputs per ha (e.g. fertilizer) have on GHG emissions. The results of these studies highlight that the effect of intensification on GHG emissions varies depending on the unit of expression. For instance, when dairy-farm GHG emissions are quantified per

Table 1 Summary of studies modelling greenhouse gas (GHG) emissions from livestock systems since 2009

Livestock species	Study	Farm description	Study goal(s) and methodology	GHG emission factors
Beef	Beauchemin et al. (2011)	Average farm in Western Canada	Life cycle assessment – multi-year model that assessed strategies to mitigate GHG emissions intensity from suckler beef	IPCC (2006)
Beef	Capper (2011)	Average US farm	Whole farm model – compared 1977 beef system GHG emissions to modern conventional systems	IPCC (2006) and literature sources
Beef	Clarke et al. (2013)	Irish research farm	Life cycle assessment – combined LCA model with a suckler beef bio-economic model and examined emissions from farms differing in stocking rate and finishing system	Ireland national GHG inventory method
Beef	Doreau et al. (2011)	French research farm	Life cycle assessment – completed an LCA of finishing or feedlot beef bull systems and estimated the effect different diets have on their GHG emission	Enteric methane emissions were measured. IPCC (2006) was used for manure
Beef	Hessle et al. (2017)	Average Swedish farm in Vastra Gotaland region	Life cycle assessment – assessed the effect four environmental improvement scenarios have on GHG emissions, nutrient loss and energy use from beef and milk	Swedish national GHG inventory method and IPCC (2006)
Beef	Murphy et al. (2017)	Irish research farm	Whole farm model – coupled a beef GHG model with a dairy beef bio-economic model to quantify effect of diet and slaughter age on GHG emissions	Ireland national GHG inventory method and literature sources
Beef	Ridoutt et al. (2013)	Commercial Australian farms	Life cycle assessment – assessed carbon, water and land-use footprints of contrasting cow-calf beef systems	Australian national GHG inventory method
Beef	Stackhouse et al. (2012)	Industry-simulated Californian farm	Life cycle assessment – quantified effect a growth promoter has on GHG emission from suckler beef	

Beef	Veysset et al. (2014)	Commercial French farms	Life cycle assessment – conducted a technical-economic survey on 53 suckler beef farms and quantified GHG emissions and fossil fuel use	French literature sources
Dairy	Bell et al. (2011)	UK research farm	Life cycle assessment – a Markov chain was used with LCA to compare GHG emissions from cows offered different levels of forage	IPCC (1997) and IPCC (2006)
Dairy	Beukes et al. (2010)	Average New Zealand farm in Waikato region	Whole farm model - integrated three models to investigate the effect different management scenarios have on dairy farming systems GHG emissions.	New Zealand national GHG inventory method
Dairy	Capper et al. (2009)	Average US farm	Whole farm model – compared GHG emissions from mid-1940 dairy system and modern conventional system	IPCC (2006) and literature sources
Dairy	Chobtang et al. (2016)	Commercial New Zealand farms in Waikato region	Life cycle assessment – evaluated the cradle-farm gate environmental performance of 53 commercial farms using dairy-base dataset.	New Zealand national GHG inventory method
Dairy	Del Prado et al. (2013)	Commercial farms in northern Spain	Life cycle assessment – coupled LCA and nutrient cycling models to estimate 17 dairy farms GHG emissions and nutrient-use efficiency	IPCC (2006) and literature sources
Dairy	Dolle et al. (2009)	Average farms for four French regions	Life cycle assessment – evaluated the effect different allocation methods have on dairy farms GHG emission intensity	Literature sources
Dairy	Mc Geough et al. (2012)	Average farm in Eastern Canada	Life cycle assessment – multi-year model that assessed the effect different allocation methods have on the GHG emissions intensity of raw milk	IPCC (2006)
Dairy	O'Brien et al. (2011)	Irish research farm	Whole farm model - combined a bio-economic dairy model with a GHG model to examine the effect genetic potential and grazing system has on dairy GHG emissions	Ireland national GHG inventory method

(*Continued*)

Table 1 (*Continued*)

Livestock species	Study	Farm description	Study goal(s) and methodology	GHG emission factors
Dairy	O'Brien et al. (2015)	Commercial Irish farms	Life cycle assessment – related the GHG emission intensity of 221 dairy farms to economic performance using the Teagasc national farm survey	Ireland national GHG inventory method
Dairy	van Middelaar et al. (2013)	Average Dutch farm	Life cycle assessment – coupled LCA with an economic model to assess the effect different feed strategies have on GHG emissions	Dutch national GHG inventory method
Dairy	Zehetmeier et al. (2012)	Average German farm	Whole farm model – assessed the effect increasing dairy cow milk yield has on GHG emissions from the dairy and beef industries	German national GHG inventory method
Pig	McAuliffe et al. (2017)	Commercial Irish farms	Life cycle assessment – compared GHG emissions and environmental metrics of average and top-performing pig farms	Ireland national GHG inventory method
Pig	Reckmann et al. (2013)	Average farm in northern Germany	Life cycle assessment – quantified GHG emissions and environmental metrics for a typical pig farm	IPCC (2006) and literature sources
Sheep	Jones et al. (2014)	Commercial sheep farms	Whole farm model- analysed GHG emissions from 64 sheep farms and identified areas for improvement	IPCC (2006) and literature sources
Sheep	O'Brien et al. (2016)	Average and commercial sheep farm	Life cycle assessment – evaluated the effect of intensification on GHG emissions and resource use of sheep farms	IPCC (2006) and literature sources
Sheep	Wiedemann et al. (2015)	Average Australian, UK, and New Zealand sheep farms	Life cycle assessment – evaluated the effect allocation methods have on GHG emissions from typical sheep farms	Literature sources

hectare of land, whole-farm models usually show that reducing the intensity of dairy systems reduces GHG emissions (Beukes et al., 2010; O'Brien et al., 2011). However, when GHG emissions are assessed on per unit of product basis (GHG emission intensity), intensification usually reduces GHG emissions (Capper et al., 2009; O'Brien et al., 2011). Given the rising demand for livestock products such as milk, this implies that GHG emissions should not be assessed in isolation from milk or meat production. The impact of livestock systems on emissions should also be contextualized with other production systems and countries.

A potential option for reducing GHG emissions from livestock, according to (Capper et al. 2009; Capper, 2011), is sustainable intensification. Capper et al. (2009) modelled GHG emissions from US beef and dairy systems and demonstrated increasing livestock output reduces GHG emissions through improved productive efficiency, defined as 'units of milk or meat produced per unit resource inputs'. Improving productive efficiency facilitates the dilution of maintenance effect, whereby the total resource cost per unit of product is reduced (Bauman et al., 1985). The land the strategy spares can potentially further reduce emissions by sequestering carbon in soil or woody biomass. The effect of sustainable intensification and livestock productivity on GHG emissions is not accounted for in national inventories that use the basic IPCC tier 1 approach but can be partially captured by inventories that use higher tiers. Many inventories could better reflect the influence livestock efficiency has on GHG emission intensity by updating important nutritional parameters. Further improvements may be possible to make once appropriate emission factors are established to determine the benefits of methane-reducing feed additives. Including additives or supplements within the inventory calculations as well as having the activity data captured could cut methane emissions from enteric fermentation of feed by 15–30% (Hristov et al., 2013).

Intensification may have undesirable effects on emissions as well and can lead to declines in soil organic matter or carbon levels (Zehetmeier et al., 2012; van Middelaar et al., 2013). For instance, van Middelaar et al. (2013) reported that converting grassland to arable land to support higher stocking rates improved dairy-farm production levels, but also dramatically increased soil carbon losses. Consequently, intensification nearly doubled GHG emissions per unit of product. A production system in a steady state, in terms of on-farm land use, is less likely to cause a land-use change effect, unless the origin of feed ingredients contained in compound concentrate feeds brought into the system change. O'Brien et al. (2016) noted that GHG emissions from livestock systems were sensitive to changes in these ingredients and showed including carbon sequestration in grassland resulted in extensive hill sheep farms emitting less GHG emissions per unit of output than intensive lowland farms. The inconsistent effect of intensification on livestock farms GHG emissions indicates that emissions and carbon sequestration should be assessed together. Within this context, the short-term effect of, for example, CO_2 versus the long-term effects captured through the GWP coefficients need to be taken into account in any computations.

Beef modelling studies report that GHG emission intensity can be mitigated by rearing beef from the dairy herd instead of the suckler herd. Switching to a dairy calf to beef production system eliminates suckler cows from the beef herd, which significantly reduces the GHG emission intensity of live weight or carcass weight as 80–90% of dairy cows' emissions are allocated to milk (Zehetmeier et al., 2012). The reduction is lower in terms of live weight because the terminal traits of surplus dairy cattle are inferior to suckler cattle. Some farmers generally select sires that are easy calving and have shorter gestation, which has a negative influence on carcass weight production. This may be possible to change by using new animal breeding technologies, for example, genomic selection.

5 National greenhouse gas inventory

Livestock methane emission factors reported in national GHG inventory submissions to the United Framework Convention on Climate Change (UNFCCC) were reviewed for annex 1 (developed) countries and 5 non-annex 1 countries, that is, Brazil, China, India, South Africa and Uruguay. The review was carried out on 2017 annex 1 national inventory submissions, which estimated GHG emissions for the period 1990–2015 (UNFCCC, 2017a). Unlike annex 1 countries, non-annex 1 nations do not report emissions annually. The latest reports available for these nations were 2–3 years older and estimated GHG emissions generated prior to 2013 (UNFCCC, 2017b). The appraisal of annex 1 and non-annex 1 GHG inventories considered the methods that nations use to estimate emission factors for enteric fermentation in livestock and livestock manure. The findings of this international methodological evaluation were summarized for livestock categories using the IPCC tier(s) (tier 1, tier 2 or 3) that each nation uses for emission calculations. The categories of livestock evaluated were: dairy and non-dairy cattle (bovine), sheep, pig, poultry and other livestock (e.g. rabbits, horses, mules). The results of the review for each category were compared to pertinent Irish livestock methane emission factors.

5.1 Bovine

Differences in dairy cow methane emission factors were related to the calculation and reporting methodology used, as well as cow productivity. For example, Ireland's tier 2 methane emission factors for bovines were generally lower than nations with heavier and higher-yielding cows such as the United States and higher than nations with lighter and lower-yielding cows (Table 2).

Most countries used tier 2 enteric methane emission factors for dairy and non-dairy cattle (Table 3). In official inventory terminology, anything more detailed than a tier 2 methodology is termed tier 3. However, it is not uncommon for people to use the term tier 2+ to describe situations where the tier 2 equations are used, but country-specific parameters are used. The tier 3 model can be an empirical or mechanistic model describing the fermentative and digestive processes in the gastrointestinal tract of dairy cattle. Tier 3 models operate with greater levels of activity data allowing regional, system and animal-based simulations. Tier 1 emission factors were not applied to compute dairy cow's enteric methane emission and were only used by Cyprus and the United Kingdom (UK) for mature beef cattle and younger stock (together known as non-dairy). Tier 3 emission factors were used by Ukraine and France (not implemented yet) to quantify enteric methane emissions from dairy cows and non-dairy cattle. Tier 3 emission factors were not used for estimating methane emission from dairy cow manure (exception being the Netherlands having a model capable of simulating methane, ammonia and manure methane at a tier 3 level) and rarely used for this source for non-dairy cattle.

Generally, nations that reported a higher tier method to estimate bovine enteric methane emissions used more data-intensive and detailed emission algorithms. Tier 3 equations were normally derived from published national research projects that measured and/or modelled methane. For example, a French project derived tier 2 and 3 methane emission factors for several categories of dairy cows and other cattle, considered as representative of the nation's breeding situations. Each category was associated with a breed, an average mass, a milk yield if necessary, as well as energy needs.

The Swiss tier 2 approach estimates cow feed and gross energy requirements using recommended national feeding standards that are widely used by Swiss farmers because they are a basis for their basic support payment. The Swiss Ym, methane conversion rate expressed as a fraction (i.e. the fractional loss of GEI as combustible CH_4), depends on the diet. The Ym value comes from national projects that measured methane from dairy cows in open and closed calorimeter chambers. The Swiss and French methods to estimate enteric methane emission factors for dairy cows aligned better with inventories like the Irish one rather than the complex tier 3 approaches applied by the Netherlands and Germany. Briefly, the Netherlands used Bannink et al. (2011) mechanistic, dynamic model of the rumen fermentation process to estimate methane from enteric fermentation in dairy cows. The inputs required to operate the Dutch model were feed intakes, the chemical composition of feed and degradation characteristic of the constituents of feed (e.g. crude protein). Cow feed intakes were estimated according to national feeding standards, and nutritional data was provided by a widely used Dutch agricultural laboratory.

Table 2 Live weight, milk yield, energy requirement and methane emission factors from selected annex 1 parties to the United Nations Framework Convention on Climate Change reported for dairy cows in production year 2015 (UNFCCC, 2017a)

	Ireland	United Kingdom	USA	New Zealand
Average live weight, kg	535	608	680	448
Milk yield, kg/cow per year	5458	7705	10 268	4362[a]
Gross energy required, MJ/d	261	300	IE[b]	IE
Methane from enteric fermentation, kg/year	117.2	130.0	146.0	84.3[c]
Methane from manure, kg/year	11.2	17.4	74.0	5.8[c]

[a] Obtained from dairy NZ (2016) https://www.dairynz.co.nz/media/5788611/quickstats_new_zealand_web_2017.pdf.
[b] Not reported.
[c] Includes dairy heifers.

Detailed nutrition data was used in the German approach to estimate dairy cow enteric methane as well. Their method, described by Rosemann et al. (2017), accounted for the effects of feed composition and feed properties using a German model developed by Kirchgessner et al. (1994). This model, like the Dutch model, was data-intensive and required information that is unlikely to be feasible to collect in the short term at a national level in some other countries that practice extensive grazing and/or where detailed quality information is not available most of the time.

Bovine feed intakes and OM digestibility estimates were required to estimate methane emissions from manure in most national inventories reviewed. The tier 3 method was only used to estimate methane from the manure of Australian beef cattle fed on feedlots. The calculation used measured methane conversion factors from Redding et al. (2015) for Australian manure storage systems. Australia was not the only nation that used country-specific methane conversion factors for manure management. Other examples included New Zealand, Denmark and Austria. A few national inventories such as the UK reported the proportion of cattle slurry systems that form a natural crust cover. In reality, it can be rather arbitrary whether a country declares themselves as operating at a tier 2 or tier 3 level within their inventories.

Several countries recognize that methane emissions from bovine and livestock manure are linked to other GHG emissions from this source, for example, nitrous oxide. For consistency, some countries use a comprehensive model that simultaneously quantified GHG and ammonia emissions from livestock. This approach was recommended in the EU 2017 submission to the UNFCCC for its member states but is only applied by a few nations, for example, the German GAS-EM model, Denmark's IDA model and Ireland.

Table 3 Method[a] applied to quantify bovine methane emission factors for annex 1 and non-annex 1 nations

			Dairy		Non-dairy	
Annex	Continent	Nation	Enteric	Manure	Enteric	manure
Annex 1	North America	Canada	T2	T2	T2	T2
		USA	T2	T2	T2	T2
Annex 1	Asia	Japan	T2	T2	T2	T2
		Kazakhstan	T2	T2	T2	T2
		Turkey	T2	T1	T2	T1
Annex 1	Asia/Europe	Russia	T2	T2	T2	T2
Annex 1	Europe	Austria	T2	T2	T2	T2
		Belarus	T2	T2	T2	T2
		Belgium	T2	T2	T2	T2
		Bulgaria	T2	T2	T2	T2
		Croatia	T2	T2	T2	T2
		Cyprus	T2	T2	T1	T2
		Czech Republic	T2	T2	T2	T2
		Denmark	T2	T2	T2	T2
		Estonia	T2	T2	T2	T2
		Finland	T2	T2	T2	T2
		France	T2, T3	T2	T2, T3	T2
		Germany	T3	T2	T2	T2
		Greece	T2	T2	T2	T2
		Hungary	T2	T2	T2	T2
		Iceland	T2	T2	T2	T2

(*Continued*)

Table 3 (*Continued*)

			Dairy		Non-dairy	
Annex	Continent	Nation	Enteric	Manure	Enteric	manure
		Italy	T2	T2	T2	T2
		Latvia	T2	T2	T2	T2
		Liechtenstein	T2	T2	T2	T2
		Lithuania	T2	T2	T2	T2
		Luxembourg	T2	T2	T2	T2
		Malta	T2	T2	T2	T2
		Monaco	NO	NO	NO	NO
		The Netherlands	T3	T2	T2	T2
		Norway	T2	T2	T2	T2
		Poland	T2	T2	T2	T2
		Portugal	T2	T2	T2	T2
		Romania	T2	T2	T2	T2
		Slovakia	T2	T2	T2	T2
		Slovenia	T2	T2	T2	T2
		Spain	T2	T2	T2	T2
		Sweden	T2	T2	T2	T2
		Switzerland	T3	T2	T2	T2
		Ukraine	T3	T2	T3	T2
		United Kingdom	T2	T2	T1, T2	T2

Annex 1	Oceania	Australia	T2	T2	T2	T2, T3
		New Zealand	T2	T2	T2	T2
Non-Annex 1	South America	Brazil	T2	T2	T2	T2
		Uruguay	T2	T2	T2	T2
	Africa	South Africa	T2	T2	T2	T2
	Asia	China	T2	T2	T2	T2
		India	T2	T1	T2	T1

[a] T1 = Tier 1, T2 = Tier 2, T3= Tier 3, NO = Not occurring, NE = Not estimated.

5.2 Sheep

Methane emissions from sheep are generally estimated by national inventories using tier 2 or tier 1 emissions factors (Table 4). The tier 2 method was used more often than the tier 1 approach. Generally, countries that applied the tier 2 method used the IPCC (2006) equations to estimate sheep feed requirements, manure excretion and methane emissions. France used a more advanced method to calculate enteric methane emission factors for sheep and New Zealand reported applying a country-specific tier 2 method. The French method for sheep is similar to the approach described earlier for French bovines, except for the calculation of methane from enteric fermentation. The New Zealand approach to calculate emissions from enteric fermentation and manure management in sheep was similar to that used for bovines. The tier 2 emission factors for both species were developed by Clark et al. (2003) and are regularly improved using new experimental research. The New Zealand computations for sheep enteric methane are carried out on a monthly basis and use country-specific data for sheep populations, pasture quality and productivity (e.g. milk yield and live weight). This data is generally available only at a national scale for sheep and beef cattle.

5.3 Pigs

The IPCC tier 1, enteric methane emission factors for pigs were used by 30 of the 47 nations reviewed (Table 4). The remaining countries used a tier 2 method for this source, except France which used tier 3. Tier 2 emission factors from the IPCC (2006) were primarily used to estimate methane from pig manure. Only 10 nations used a tier 1 method for this source. Most nations that used tier 2 emission factors entail computing pig's gross energy and protein intakes to meet feed requirements and estimating manure excretion from feed intake and diet digestibility. Methane emissions from manure are computed using survey data on manure storage systems and relevant IPCC conversion factors. This tier 2 method is slightly more advanced in some nations because manure excretion is based on national data instead of default excretion estimates provided by the IPCC.

The tier 2 and tier 3 methods that nations used to estimate enteric methane from pigs do not differ to that described for bovines and sheep. The Ym for pigs was normally derived from the references of the IPCC (2006) guidelines. Country-specific Ym values were also developed by a few nations, for example, Germany, but these estimates were similar to the IPCC estimate differing by <0.1 units.

5.4 Poultry

Nations are not required to estimate enteric methane emissions from poultry. This category is considered by the IPCC to emit negligible emissions. Thus, there are currently no tier 1 emission factors for this source. Nevertheless, 12 nations estimated enteric methane emissions from poultry and 3 reported using tier 2 emission factors (Table 5), because the method was developed from national research in their countries. The emission factors from such research, for example, Wang and Huang (2005) could potentially be applied to estimate enteric methane from poultry in other countries. A slightly higher number of nations estimated methane from poultry manure using tier 2 emission factors instead of tier 1. Australia was the only nation that reported a tier 3 method. The quantity of manure excreted by poultry was estimated using national data or IPCC default values.

5.5 Other livestock

The other livestock category enteric fermentation methane emissions were usually calculated by nations using tier 1 emission factors (Table 5). Nations seldom applied the tier 2 method to estimate enteric methane emissions from all species that are part of this livestock category. Most nations only used tier 2 emission factors to estimate methane emissions from a few economically important production species within the other livestock category, for example, for goat and deer. In general, the tier 2 approach in computing enteric methane emissions from other livestock species was very similar to the IPCC method for cattle and sheep, apart from the Ym parameter, which was not necessarily species-specific. For example, the South African Ym parameter for goats was based on sheep measurements. This approach to estimate Ym is generally acceptable as long as species have similar digestive systems. This approximation method is also recommended by the IPCC for other livestock species not listed in their guidelines and can be applied based on live weight for tier 1 emission factors.

Almost half of the nations reviewed used a tier 2 method to estimate methane from the manure of a species contained within the other livestock category. Of these nations, 13 used tier 2 methane emission factors for manure for all species within this category. The IPCC tier 2 equations and default manure excretion rates were generally used to estimate methane emissions from other livestock species manure.

5.6 Methodology conclusions

Livestock methane emissions factors can be computed more accurately using a tier 2 method instead of a default tier 1 approach because there is country-specific information available around methane emissions for the major production species. This approach is widely used by annex 1 nations to estimate methane from key livestock categories, for example, bovine, pigs and sheep. In addition, several annex 1 nations use this method for all species and integrate

Table 4 Method applied to quantify sheep and pigs methane emission factors for annex 1 and non-annex 1 nations

			Sheep		Pigs	
Annex	Continent	Nation	Enteric	Manure	Enteric	Manure
Annex 1	North America	Canada	T1	T2	T1	T2
		USA	T1	T2	T1	T2
Annex 1	Asia	Japan	T1	T1	T2	T2
		Kazakhstan	T2	T1	T1	T1
		Turkey	T1	T1	T1	T1
Annex 1	Asia/Europe	Russia	T1	T1	T2	T2
Annex 1	Europe	Austria	T1	T1	T1	T2
		Belarus	T1	T1	T1	T2
		Belgium	T1	T1	T1	T2
		Bulgaria	T2	T2	T1	T2
		Croatia	T2	T2	T2	T2
		Cyprus	T1	T1	T1	T1
		Czech Republic	T1	T1	T1	T1
		Denmark	T2	T2	T2	T2
		Estonia	T1	T1	T2	T2
		Finland	T2	T2	T2	T2
		France	T2, T3	T2	T2, T3	T2
		Germany	T1	T2	T2	T2
		Greece	T2	T2	T1	T1
		Hungary	T1	T2	T1	T2
		Iceland	T2	T2	T1	T1
		Italy	T2	T1	T1	T2

		Latvia	T1	T1	T1	T2
		Liechtenstein	T2	T2	T2	T2
		Lithuania	T2	T2	T2	T2
		Luxembourg	T2	T2	T1	T2
		Malta	T2	T1	T1	T2
		Monaco	NO	NO	NO	NO
		The Netherlands	T1	T1	T1	T2
		Norway	T2	T1	T1	T2
		Poland	T1	T1	T1	T1
		Portugal	T2	T2	T2	T2
		Romania	T2	T2	T2	T2
		Slovakia	T2	T2	T1	T1
		Slovenia	T1	T2	T1	T2
		Spain	T2	T1	T2	T2
		Sweden	T1	T1	T1	T2
		Switzerland	T2	T2	T2	T2
		Ukraine	T2	T2	T1	T2
		United Kingdom	T1	T2	T1	T2
Annex 1	Oceania	Australia	T2	T2	T2	T3
		New Zealand	T2	T2	T1	T2
Non-Annex 1	South America	Brazil	T1	T1	T1	T2
		Uruguay	T1	T1	T1	T1
	Africa	South Africa	T2	T2	T2	T2
	Asia	China	T2	T2	T1	T2
		India	T1	T1	T1	T1

Note: T1 = Tier 1, T2 = Tier 2, T3 = Tier 3, NO = Not occurring, NE = Not estimated.

with other methods to estimate air pollutants, for example, ammonia. The tier 2 method that nations applied were generally similar (i.e. emission factor and an estimate of intake to drive percentage losses), but the equations and parameters that nations used to calculate important inputs required to derive methane emissions, such as feed intakes, Ym and manure excretion were often based on national research. These country-specific equations are usually more accurate than the IPCC alternatives and should be used by the inventory where possible, for example, sheep. There may also be potential to develop more advanced methane emission factors for bovine and sheep by using national measurements of enteric methane emissions from these species.

6 Mitigation strategies

Whole-farm GHG models have assessed the effect of several farm practices on emissions. This section describes what can be captured, how they can be captured within models. Examples include selecting higher-yielding livestock, improving fecundity, reducing replacement rates, varying the grazing period, treatment of managed manure and improving nutrient management. The effect of breeding higher-yielding livestock on GHG emissions has varied between studies. Murphy et al. (2017) showed that a 13% increase in beef cattle live weight gain per day reduced GHG emission per unit of carcass weight by 19%. Similar results were reported by Bell et al. (2011) for milk production. In both studies, the reduction was explained by a decrease in methane emission due to improvements in feed conversion efficiency (kg of milk or meat/kg of feed) and decreasing maintenance costs. In contrast, O'Brien et al. (2011) reported Holstein-Friesian cows with higher milk yields increased emissions per unit of the product relative to lower-yielding cows. This was primarily caused by higher-yielding Holstein-Friesian cows having lower herd fertility rates and a shorter lifespan, which increased replacement heifer requirements and thus GHG emissions (O'Brien et al., 2011). The study indicates that it is important to consider a combination of genetic traits to reduce GHG emission intensity.

Beukes et al. (2010) modelled the effect of reducing herd replacement rates on GHG emissions from dairy systems. The analysis showed that a 10% reduction in replacement rate reduced GHG emission per unit of a product by an average of 14%. This decrease in GHG emissions occurred due to a decline in the number of non-productive animals. Bovine fertility is possible to improve through genetic selection and better reproductive management, that is, heat detection aids (Vellinga et al., 2011). Improving reproductive management can also reduce emissions from sheep farms. For instance, Ledgard et al. (2011) reported higher lambing percentages were partly responsible for a 22% reduction in GHG emission from New Zealand sheep production. Increased animal yields and farm profitability were the main drivers of this change.

The influence of the length of the grazing period on GHG emissions was evaluated by Schils et al. (2005) and O'Brien et al. (2015). Schils et al. (2005) investigated the effect of reducing the grazing period and reported that the strategy decreases nitrous oxide emissions from animal excreta deposited on pasture, but increases methane and nitrous oxide emissions from manure storage to a similar extent. Thus, the strategy had no net effect on GHG emissions. Furthermore, Schils et al. (2005) reported that the strategy has undesirable effects on other types of pollution, for example, ammonia. Thus, this demonstrates that models should also consider the undesirable secondary effects (pollution swapping) of GHG mitigation strategies (del Prado et al., 2013). The effect of extending the grazing period was examined by O'Brien et al. (2015). The study showed the strategy caused a greater reduction in methane and nitrous oxide emissions from manure storage than an increase in nitrous oxide emissions from animal excreta deposited on pasture. In addition, the strategy reduced dairy systems' GHG emission intensity by 0.17% per day and improved economic performance.

The potential for feed additives and growth promoters to mitigate methane emissions were assessed by Beauchemin et al. (2011) and Stackhouse et al. (2012). Beauchemin et al. (2011) reported that dietary oils mitigated the GHG emission intensity of western Canadian beef systems by 2–8%. However, Hristov et al. (2013) reported that this strategy was economically risky and highlighted that the long-term mitigation efficacy of oils and fats was uncertain. It also may be difficult to record this mitigation potential in national GHG emission inventories. Stackhouse et al. (2012) showed that antibiotic growth promoters reduced GHG emission intensity by 5–9% for California beef cattle, but these growth implants are proscribed in the EU. Few models directly assess strategies to mitigate methane from manure storage. McAuliffe et al. (2017) highlighted anaerobic digestion as a promising technology to mitigate the environmental impact of pig manure. The strategy can also capture bioenergy and produce electricity but requires more favourable tariffs.

6.1 Commercial farm

The environmental sustainability of commercial livestock farms often has been estimated in developed nations using whole farm or LCA models. O'Brien et al. (2015) estimated the milk carbon footprints of 221 Irish dairy farms using a life cycle approach and evaluated economic performance. The research showed a wide range in Irish dairy farms carbon footprints (0.50–1.97 kg of CO_2-equivalent/kg of fat and protein corrected milk). Generally, dairy farms with lower milk footprints were more profitable. For example, the milk carbon footprint of the top-third of farms in terms of net margin per hectare was 15% less than the bottom-third. Partial least square regression indicated that

Table 5 Method applied to quantify poultry and other livestock[b] methane emission factors for annex 1 and non-annex 1 nations

			Poultry		Other livestock	
Annex	Continent	Nation	Enteric	Manure	Enteric	manure
Annex 1	North America	Canada	NE	T2	T1	T1
		USA	NE	T2	T1	T2
Annex 1	Asia	Japan	NE	T2	T1	T1
		Kazakhstan	NE	T1	T1	T1
		Turkey	NE	T1	T1	T1
Annex 1	Asia/Europe	Russia	NE	T1	T1	T1
Annex 1	Europe	Austria	T2	T1	T1	T1
		Belarus	T1	T1	T1, T2	T1,T2
		Belgium	T1	T1	T1	T1
		Bulgaria	NA	T2	T1	T1
		Croatia	T1	T2	T1	T2
		Cyprus	NE	T1	T1	T1
		Czech Republic	NE	T1	T1	T1
		Denmark	T1	T2	T1	T2
		Estonia	NE	T1	T1	T1
		Finland	NE	T2	T1	T2
		France	NE	T2	T2	T2
		Germany	NE	T2	T1	T2
		Greece	T1	T1	T1	T1
		Hungary	T1	T2	T1	T2
		Iceland	T1	T1	T1	T1
		Italy	NE	T1	T1, T2	T1
		Latvia	NE	T1	T1	T1

		Liechtenstein	T2	T2	T2	T2
		Lithuania	NE	T1	T1	T1
		Luxembourg	NO	T2	T1	T2
		Malta	T1	T2	T1	T1
		Monaco	NO	NO	NO	NO
		The Netherlands	NE	T2	T1	T1
		Norway	T1	T2	T1, T2	T1
		Poland	NE	T1	T1	T1
		Portugal	NE	T2	T1	T1, T2
		Romania	NE	T2	T2	T2
		Slovakia	NE	T1	T1	T1
		Slovenia	NE	T2	T1	T2
		Spain	NE	T2	T1, T2	T1, T2
		Sweden	NE	T1	T1	T1
		Switzerland	T2	T2	T2	T2
		Ukraine	NE	T2	T1	T1, T2
		United Kingdom	NE	T2	T1	T1, T2
Annex 1	Oceania	Australia	NE	T3	T1	T2
		New Zealand	NE	T1	T1, T2	T1, T2
Non-Annex 1	South America	Brazil	NE	T1	T1	T1
		Uruguay	NE	T1	T1	T1
	Africa	South Africa	NE	T2	T1, T2	T1, T2
	Asia	China	NE	T1	T1, T2	T1, T2
		India	NE	T1	T1	T1

Note: T1 = Tier 1, T2 = Tier 2, T3= Tier 3, NO = Not occurring, NE = Not estimated.
Note: Other livestock includes goats, horses, mules, donkeys, camels, rabbits, deer, llamas, ostriches, reindeer and alpacas. Bovines, sheep, pigs and poultry were not part of this category.

farm practices which increase milk solids yield per hectare and the utilization of grazed grass mitigated GHG emission intensity and improved economic performance. The average methane emission factor for dairy cows (110 kg methane/cow) fell within the range of recent Irish inventory estimates.

Chobtang et al. (2016) and Del Prado et al. (2013) carried out similar research as O'Brien et al. (2015) for New Zealand and Spanish dairy farms. Del Prado et al. (2013) reported similar levels of variability among farms' carbon footprints as O'Brien et al. (2015), but Chobtang et al. (2016) found less variability for New Zealand dairy carbon footprints. This may be due to farms operating at a similar level of efficiency, but there was significant farm performance variability. Chobtang et al. (2016) also assessed other environmental measures such as acidification and health measures such as effects on human health. The variability across farms for these measures was generally much higher than the carbon footprint. Off-farm impacts were important for several environmental and health measures assessed and could be improved by increasing forage utilization. Veysset et al. (2014) also recommended the latter to mitigate GHG emissions from commercial French suckler beef farms. The authors highlighted that farms which produced the best forage and heaviest cattle mitigated GHG emission intensity by 50% compared to the least productive in the group. The least productive French farms were larger in the area and operated mixed crop-livestock systems.

Jones et al. (2014) estimated lamb carbon footprints for lowland, upland and hill sheep farms in the UK and reported lowland farms had the smallest mean footprint. There was substantial variability across lamb carbon footprints, and this was largely driven by animal and grassland productivity. The authors recommended a suite of management practices to reduce the carbon footprint of lamb, including improving lambing rates (lambs/ewe), increasing lamb growth rate and optimizing mating rates and concentrate usage. These improvements are likely to be problematic for hill farms where local conditions are much more challenging. Nevertheless, some improvement should be possible as Jones et al. (2014) reported top-performing hill farms outperformed the mean performance of lowland farms.

6.2 Modelling decisions

Quantifying GHG emissions from livestock systems involves making important modelling decisions such as the definition of system boundaries and allocation of emissions to co-products. O'Brien et al. (2011) examined the influence that different system boundaries have on GHG emissions from dairy farms varying in cow genotype and concentrate supplementation. This analysis showed that high-concentrate dairy systems reduced on-farm GHG emissions per kg of milk solids, but the opposite occurred for the New Zealand strain of Holstein Friesian when the system boundary was expanded to include off-farm GHG emissions. Further research by O'Brien et al. (2012) demonstrated similar results for grass-based and confinement dairy systems and highlighted that reducing national GHG emissions to comply with EU targets is likely to increase GHG emissions from dairy production through carbon leakage. This research demonstrates that livestock systems GHG emissions should be assessed using a complete approach like LCA to ensure changes to dairy systems reduce GHG emissions. O'Brien et al. (2011) recommended integrating the LCA method into the Irish inventory framework. This may be possible using datasets like the Teagasc national farm survey.

Various criteria or methods were evaluated for allocating GHG emissions to the products of sheep and dairy systems. Wiedemann et al. (2015) compared seven different criteria to allocate GHG emissions between sheep products, that is, live weight and greasy wool. The case study results were relatively similar when GHG emissions were expressed per kg of total products but varied widely when emissions were split between products based on their economic value. The study recommended allocating GHG emissions between sheep products according to their protein requirements or using the mass of protein sold. Dolle et al. (2009) and Mc Geough et al. (2012) showed similar variability in dairy systems GHG emission intensity due to the allocation method. The authors identified advantages and disadvantages with several methods. Mc Geough et al. (2012) recommended basing the allocation decision on the needs and availability of data, which agrees with the conclusions of Rice et al. (2017) for Irish dairy systems. Rice et al. (2017) reported simple mass allocation was the best approach in terms of the pedigree of data but advised using a range of allocation methods to understand the uncertainty associated with the decision.

7 Conclusion

Most environmental modelling studies use a life cycle approach to quantify livestock's GHG emissions and overall environmental performance. Generally, modelling studies estimated livestock methane emissions according to the calculations provided in national inventory reports or the IPCC guidelines, which ensure the compliance of approaches between different entities modelling at a national level. These calculations vary widely from country to country due to local conditions and livestock research capabilities, but normally show improvements in animal productivity and switching from suckler to dairy beef systems mitigate GHG emission intensity. There is significant potential to use models at a national level to estimate GHG emissions from commercial bovine systems to generate national footprints. Moreover, there is scope to use LCA and national inventory models to benchmark performance across countries and over time. Some national sustainability initiatives, for example, Origin Green are already employing these tools to improve farmers and the wider community understanding of GHG emissions from livestock production. Farm-scale models could be used to pin point the direction in relation to emissions reduction strategies, but it is important that models use an LCA approach to ensure that the analysis completed is robust and not facilitating carbon leakage. Models that are used in these situations should be peer reviewed, validated and evaluated against detailed experimental data.

8 Future research

The key future research required in the area of methane centres on biogenic methanes impact on global warming. Given the fact that methane is a short-lived gas there is increasing questions around its impact on climate change. There is a requirement for the IPCC and LCA methodologies to be relooked at in this context. It would be expected that the over-riding conclusions from this research will be that the impact of ruminant agriculture on climate change maybe over-estimated. Further information can be found at Cain et al. (2019).

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Part 2

Breeding, animal husbandry and manure management

Chapter 3

The contribution of animal breeding to reducing the environmental impact of livestock production

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1 Introduction

The world human population is expanding, and the demand for food in the year 2050 is expected to be approximately 70% greater than in 2010 (FAO, 2009). Demand for meat and other livestock products is highly elastic to income (Delgado et al., 1999), and therefore as population affluence improves, the demand for livestock products will increase further. Also, the global human population is ageing, and older people typically consume larger quantities of animal-derived protein than children (Steinfeld et al., 2006). The expected 70% increase in food demand requires an annual increase in food production of

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1.3% per annum. This increase in food demand requires an increased efficiency of food production, both animal and crop derived. Moreover, competition for land from other industries (e.g. biofuels) implies this increased animal and crop production must be achieved from an ever-declining land base. Although feed efficiency, as currently defined, is not synonymous with production efficiency, it undoubtedly will be a major contributor to increasing production from an ever-decreasing food-producing land base. The global production of red meat is expected to increase from 229 million tonnes in 1999–2001 to 465 million tonnes in 2050, while global milk production is expected to increase from 580 to 1043 million tonnes in the same period (Steinfeld et al., 2006). This increased production must, however, be achieved in an environmentally responsible and sustainable manner.

There is considerable commentary nowadays on climate change and its implications, as well as possible mitigation strategies. Animal production generates greenhouse gas (GHG) emissions as methane (CH_4) from enteric fermentation and manure, nitrous oxide (N_2O) from the widespread use of nitrogenous fertilizers and animal manure and carbon dioxide (CO_2) from the fossil fuels for energy usage plus land-use change. Methane, however, is not only an environmental hazard but is also associated with a loss of carbon from the rumen and therefore an unproductive use of energy (Johnson and Johnson, 1995). There is a wide variation in documented calculations of animal agriculture contributions to GHG (Herrero et al., 2011). O'Mara (2011) stated that animal agriculture is responsible for 8.0% to 10.8% of global GHG emissions based on calculations from the Intergovernmental Panel on Climate Change (IPCC), but if complete lifecycle analysis (i.e. accounting for the production of inputs to animal agriculture as well as the change in land use such as deforestation) is undertaken, this figure can be up to 18%. Cattle are the largest contributors to the global emission of CH_4 from enteric fermentation (O'Mara, 2011).

In December 2015, 195 nations signed the Paris Agreement. The Paris Agreement's central aim is to strengthen the global response to the threat of climate change. Therefore, in this century, the global temperature rise should at least be kept well below 2°C above pre-industrial levels. To pursue efforts to limit the temperature increase even further, a more stringent aim is set to a threshold of 1.5°C, instead of 2°C. Additionally, the agreement aims to increase the ability of countries to deal with the impacts of climate change and to make finance flows consistent with low GHG emission and climate-resilient pathways. All sectors have to reduce their environmental impact, including the livestock sector. In this chapter, we will review the current environmental impact of livestock species, show historical trends and quantify the contribution of animal breeding in reducing the environmental impact of livestock species.

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2 The environmental impact of livestock production

Animal production is responsible for 14.5% of total anthropogenic GHG emissions (Gerber et al., 2013). Approximately half of these emissions originate directly from animal production, whereas the other half comes from feed production. Of the total of 7.1 gigatonnes CO_2-equivalents produced by the global livestock sector, beef production is responsible for 41%, cattle milk production for 20%, pig meat production for 10%, chicken meat production for 6% and chicken egg production for 3%. The remaining 21% is produced by other species and purposes, like buffalo and small ruminants for milk and meat, and by draft power, manure as fuel, and so on (Gerber et al., 2013).

Several tools have been developed to assess the environmental impacts of animal agriculture on a product basis. One of these tools is FeedPrint (Vellinga et al., 2013; Wageningen Livestock Research, 2015). FeedPrint is a life cycle analyses (LCA) tool for the assessment of environmental impacts of animal production, especially GHG emissions, developed primarily for the Dutch animal feed industry. FeedPrint, therefore, contains a full database of animal feed ingredients and their respective impacts from cradle to farmgate, including, for example, fertilizer production and application, field emissions, energy use for field work, transport and feed production. Based on data available in this tool, we could show that the GHG emissions for the production of 1 kg of pig meat are higher than for 1 kg of broiler meat, followed by chicken eggs and dairy production (Fig. 1a). Although results differ between studies, this ranking is in line with, for example, De Vries and De Boer (2010) or Williams et al. (2006). The relatively low numbers of GHG emissions for the production of milk and eggs are mainly due to the high water content of these products compared to meat and the fact that animals do not need to be slaughtered to acquire the product. When values of GHG emissions are recalculated to per kg protein basis, numbers come much closer together, with broiler meat being among the best-performing products (De Vries and De Boer, 2010; Williams et al., 2006). Beef production is outside the scope of this chapter, but, in general, calculated GHG emissions per kg product and kg protein are much higher for beef than for the other animal products.

There are different sources of GHG emissions related to animal production. The best-known and most-investigated one is enteric methane (CH_4) emission from ruminants. For dairy production, animal-related CH_4 emissions account for over one-third of total GHG emissions (Fig. 1b). However, in general, about half of the GHG emissions related to animal production come from feed-production-related processes (Gerber et al., 2013). For poultry in The Netherlands, even more than 80% of GHG emissions are related to feed production (Fig. 1b), with CO_2 being the predominant contributor, followed by nitrous oxide (N_2O).

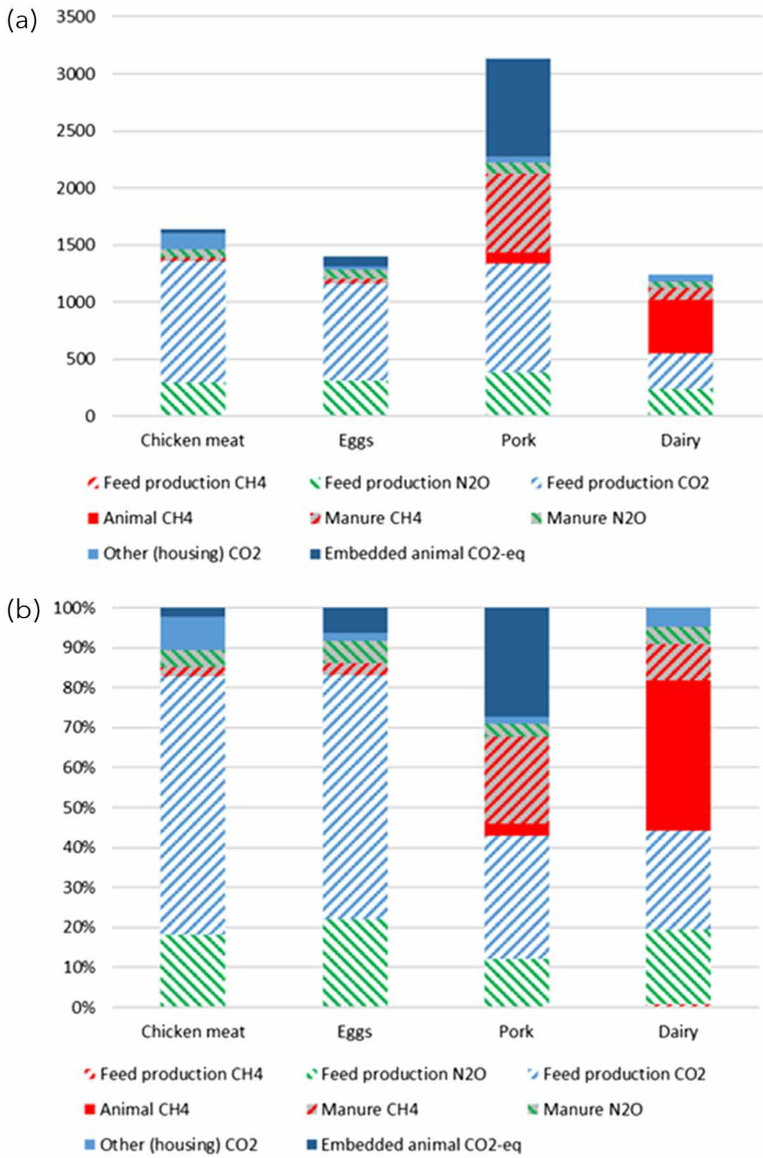

Figure 1 Greenhouse gas emissions (kg CO_2-equivalents) per tonne product (a) and relative contribution (%) to total greenhouse gas emissions (b) for four main animal products in The Netherlands based on FeedPrint (Vellinga et al., 2013; Wageningen Livestock Research, 2015). (Embedded animal = emissions caused by parent and young stock which are accounted to the final product).

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Next to the emission of GHG, nitrogen (N)– and phosphorus (P)–related environmental impacts, like acidification, eutrophication and depletion of P resources, are important when animal products are concerned (e.g. Thomassen and de Boer, 2005). These impacts could be assessed using lifecycle analysis but more straightforward by nutrient balances and derived efficiency parameters (Mu et al., 2016). The efficiency of N and P usage at animal level seems to be a relevant indicator for animal breeding, as animal breeding aims at improving animal production and efficiently using resources, resulting in a reduction of the environmental impact. The objective of this study was to quantify the contribution of animal breeding to reducing the environmental impact of the four major livestock species in The Netherlands, namely broilers (meat), laying hens (eggs), pigs (meat) and dairy cattle (milk).

In this chapter, we will show both the historical trends and the quantification of the contribution of animal breeding to the environmental impact of several livestock species: broilers, layers, pigs and dairy cattle. At the end of the chapter is a summarizing conclusion with recommendations.

3 Broilers: environmental impact and the contribution of breeding

3.1 Historical trends

Havenstein et al. (2003) performed a study where they compared the feed conversion of broilers genetically representative of animals for the years 1957 versus 2001 when fed broiler diets representative for diets fed in 1957 and 2001. Average body weights on 42 and 84 days of age for the 1957 broiler on the 1957 diets were 539 and 1430 g versus 2672 and 5520 g for the 2001 broiler on the 2001 diets, respectively. The 42-day feed conversion (FC) for the 1957 broiler on the 1957 diets was 2.34, and for the 2001 broiler on the 2001 diet was 1.63. The 2001 broiler on the 2001 feed was estimated to have reached 1815 g BW at 32 days of age with an FC of 1.47, whereas the 1957 broiler on the 1957 feed would not have reached that BW until 101 days of age with an FC of 4.42. Their conclusion was that 85–90% of the improvement in the performance of broilers is due to genetic selection, and 10–15% due to nutrition (Havenstein et al., 2003).

Havenstein et al. (2003) also showed that the slaughter weight increases over the years. This distorts the trend of the feed conversion ratio (FCR) over the years. Therefore, LEI (currently Wageningen Economic Research) calculated the trend in FCR corrected to a slaughter weight of 2.15 kg based on Dutch data, which showed that the FCR decreases with almost 1% per year (Fig. 2).

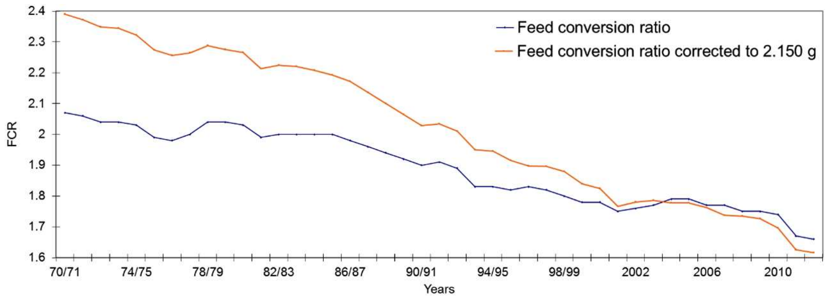

Figure 2 Trend in the feed conversion ratio of Dutch broilers (based on data from LEI (currently Wageningen Economic Research)).

Table 1 Summary data of broiler dataset with reference year, date of first and last placement, number of flocks, average number of animals per flock (Avg. numb.), average age at slaughter (Avg. age), average final body weight (after 1 day fasting) (Avg. weight) and average feed conversion ratio (Avg. FCR)

Year	First	Last	# flocks	Avg. numb.	Avg. age	Avg. weight	Avg. FCR
2014	07-05-13	28-08-14	12	33 105	37.2	2.2	1.66
2018	16-06-17	07-08-18	10	63 716	40.3	2.7	1.56

3.2 Quantification of contribution of animal breeding

3.2.1 Materials and methods

Data on the performance of Dutch commercial flocks were obtained from a broiler breeding company (Cobb Europe B.V., Boxmeer, The Netherlands) and contained 12 flocks with placement date in the period from May 2013 to August 2014, referred to as 2014, and 10 flocks from the period June 2017 to August 2018, referred to as 2018 (Table 1).

Feed composition was assumed to be equal in both periods and was derived from FeedPrint 2015.03 (Vellinga et al., 2013; Wageningen Livestock Research, 2015). Emissions of GHG related to the production of feed ingredients, and their N and P content, were collected from the database of FeedPrint 2018 (Wageningen Livestock Research, 2018).

Emissions of GHG are expressed in CO_2 equivalents (CO_2-eq.), which is a unit to express the contribution of different GHGs to their global warming potential (GWP), relative to CO_2. Methane (CH_4) and nitrous oxide (N_2O) have a GWP of respectively 36.75 (34 for biogenic CH_4) and 298 CO_2-eq (Myhre et al., 2013).

Efficiencies of N and P are expressed in percentages and calculated as N and P in output, for broilers in live animals, over input with feed. To express the output in kg N and P, body composition after 1 day fasting was based on Caldas (2015).

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3.2.2 Results

The environmental impact caused by GHG emissions from broiler production decreased with 1.6% per year, and N and P efficiency increased with 1.7% per year (Table 2). This decrease in environmental impact was fully caused by a decrease in FCR (Table 1, 1.5%), which was larger than expected based on a literature overview of the development of FCR, where FCR decreased with about 1%.

3.2.3 Discussion

As data were available only for the broiler growing phase, data analysis focused on this phase. This phase, however, is responsible for the vast majority of the GHG emissions of the whole broiler production chain (Fig. 1), with feed production in the broiler growing phase being responsible for 83% of total GHG emissions. Results, therefore, could be considered representative for the whole broiler production process.

The dataset used for this study was based on commercial flocks, instead of a research dataset. Therefore, it showed some peculiarities, for example, a considerable increase in flock size and age at slaughter, the latter also causing an increase in final body weight. These differences represent a global shift in slaughter weights for the broiler industry. However, for purposes of comparing GHG emissions, slaughter age and flock size differences could have contributed to an underestimation of the decrease in FCR, resulting in high reduction percentages. However, even with a more conservative estimation of 1% decrease in FCR, we could conclude that GHG emissions decrease and N and P efficiency increases with the current breeding goal for broilers.

4 Layers: environmental impact and the contribution of breeding

4.1 Historical trends

Pelletier (2018) provided an evaluation on the extent of and plausible reasons for the change in the life cycle environmental footprint of conventional Canadian

Table 2 Emissions of GHG and N and P efficiency of broiler production in 2014 and 2018

Year	GHG emissions (kg CO_2-eq/kg BW)	N efficiency (%)	P efficiency (%)
2014	1.43	53.2	46.0
2018	1.34	56.7	49.1

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egg production over a 50-year interval spanning from 1962 to 2012. For this, the historical trajectory of the industry toward more sustainable practices was elucidated. Supply chain acidifying emissions declined by 61%, eutrophying emissions by 68% and GHG emissions by 72%. Despite the >50% increase in annual egg production volumes – from 43.4 million dozen to 65.7 million dozen eggs in 1962 and 2012, respectively, the industry's overall environmental footprint actually decreased across all emissions and resource use domains considered. These observed changes are attributable to a combination of factors, including improved feed efficiency, changes in diet composition and manure management.

A similar study was performed by Pelletier et al. (2014) on the comparison of the environmental footprint of the egg industry in the United States in 1960 and 2010. They showed a similar reduction in GHG emissions by 71%.

The increase in egg production and the decrease in FCR were also present in Dutch data (Fig. 3), based on national data collated yearly (KWIN, 2011, 2013, 2017).

4.2 Quantification of contribution of animal breeding

4.2.1 Materials and methods

For laying hens a full LCA model was available (Van Winkoop, 2013), which takes into account parent stock and layers, both including their rearing phase.

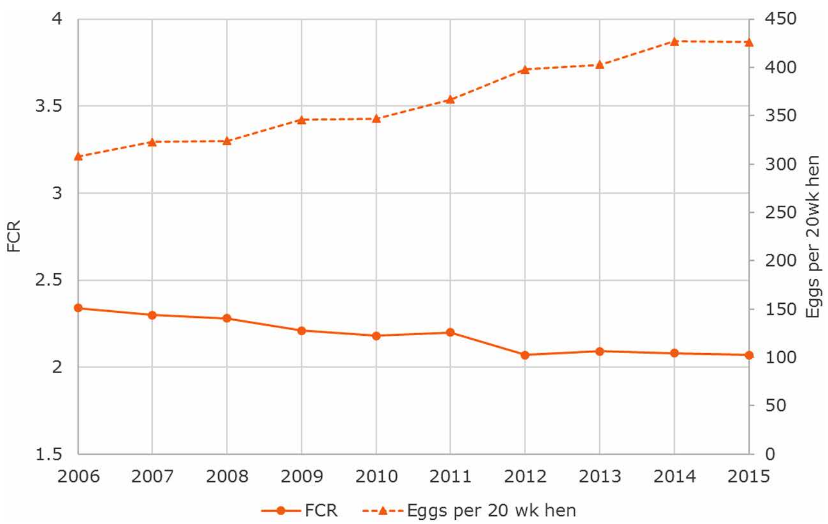

Figure 3 The trends in egg production and feed conversion ratios per Dutch laying hen from 2006 to 2015.

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For the calculation of genetic progress, however, only changes in the laying period were incorporated in the calculations.

Data on the development of modern commercial brown and white layer lines were obtained from a breeding company (Institut de Sélection Animale B.V. a Hendrix Genetics Company, Boxmeer, The Netherlands) and contained for the brown layers data for the years 2008 and 2017 (Table 3) and for the white layers for the years 2009 and 2017 (Table 3). These data should be considered to be from top-performing flocks and were available for the egg production phase only. These data were used to assess the rate of improvement, as this was anticipated to be representative for top-performing and average-performing commercial flocks. For a full LCA assessment, however, more data were needed and were derived from the 'commercial product' guides (Institut de Sélection Animale B.V. a Hendrix Genetics Company, Boxmeer, The Netherlands; Table 4). These data were also used for calculating environmental impacts for the commercial situation. It should be noted that these guides should hold for a broad range of commercial settings, including more challenging environments.

Feed composition was assumed to be equal in all periods and was derived from FeedPrint 2015.03 (Vellinga et al., 2013; Wageningen Livestock Research,

Table 3 Summary data of top-performing brown and white layers with reference year, total life time, mortality rate during laying period (from 18 weeks onwards), number of eggs per housed hen, total egg mass and average feed conversion ratio (Avg. FCR) during the laying period

Line	Year	Lifetime (weeks)	Mortality rate (%)	Eggs per housed hen	Total egg mass (kg)	Avg. FCR
Brown	2008	75	6	324	20.6	2.25
	2017	90	5	429	27.0	2.14
White	2009	75	6	329	20.7	2.16
	2017	90	5	433	27.3	2.05

Source: Institut de Sélection Animale B.V. a Hendrix Genetics Company, Boxmeer, The Netherlands.

Table 4 Summary data of commercial brown and white layers, total life time, mortality rate during laying period (from 18 weeks onwards), number of eggs per housed hen, total egg mass and average feed conversion ratio (FCR) during the laying period

Line	Lifetime (weeks)	Mortality rate (%)	Eggs per housed hen	Total egg mass (kg)	Avg. FCR
Brown	80	7.8	353	22.1	2.29
White	90	7.5	411	25.9	2.24

Source: Product guides, Institut de Sélection Animale B.V. a Hendrix Genetics Company, Boxmeer, The Netherlands.

2015). Emissions of GHG related to the production of feed ingredients, and their N and P content, were collected from the database of FeedPrint 2018 (Wageningen Livestock Research, 2018).

Emissions of GHG are expressed in CO_2 equivalents (CO_2-eq.), which is a unit to express the contribution of different GHG to global warming, their global warming potential (GWP), relative to CO_2. Methane (CH_4) and nitrous oxide (N_2O) have a GWP of, respectively, 36.75 (34 for biogenic CH_4) and 298 CO_2-eq (Myhre et al., 2013).

Efficiencies of N and P are expressed in percentages and calculated as N and P in output over input with feed. For laying hens, only eggs were considered as output and N and P contents were calculated with N and P content of raw egg (edible part; Finglas et al., 2015) applied to total egg mass (Tables 3 and 4) corrected for 15% shells (pers. Comm., Institut de Sélection Animale B.V. a Hendrix Genetics Company).

4.2.2 Results

Results for commercial brown and white layers (Table 5) show that impacts do not differ much.

The environmental impact caused by GHG emissions from egg production decreased by 0.7% per year for brown layers and by 0.9% per year for white layers (Table 6). N and P efficiency increased with 0.5% per year for brown layers and with 0.7% per year for white layers. The decrease in environmental impact was only partly caused by a decrease in FCR as also the production period was extended, due to genetic progress (Table 3). Especially for the assessment of GHG emissions, for which a full LCA including parent stock and rearing was used, the extended production period considerably contributed to the reduction of environmental impacts.

4.2.3 Discussion

As data for calculating genetic progress were only available for the laying period, total improvements could be expected to be larger when also improvements

Table 5 Emissions of GHG and N and P efficiency of egg production by commercial brown and white laying hens

Line	GHG emissions (kg CO_2-eq/kg egg)	N efficiency (%)	P efficiency (%)
Brown	2.18	30.2	15.5
White	2.09	30.9	15.8

Table 6 Emissions of GHG and N and P efficiency of egg production by top-performing brown laying hens in 2008 and 2017

Line	Year	GHG emissions (kg CO_2-eq/kg egg)	N efficiency (%)	P efficiency (%)
Brown	2008	2.18	30.6	15.7
	2017	2.03	32.0	16.4
White	2009	2.10	31.9	16.3
	2017	1.95	33.5	17.1

in parent stock and rearing would be taken into account. The laying period, however, is responsible for the vast majority of the GHG emissions of egg production (Fig. 1), with feed production in the laying period being responsible for 83% of total GHG emissions.

Based on this analysis we conclude that genetic progress is considerable in both brown and white layers, where white hens currently perform better and also improve faster than brown hens with respect to the environmental impact of production. As most brown hens produce brown eggs and most white hens produce white eggs and consumers in some countries prefer brown over white eggs, both types of layers still exist.

5 Pigs: environmental impact and the contribution of breeding

5.1 Historical trends

The feed efficiency of growing pigs has been a matter of serious commercial and scientific interest since at least 1970, but early recording technology made it difficult to produce accurate feed intake data at the individual level (Knap, 2009). Since electronic feeders were introduced, the pig breeding industry has been making good genetic improvement in feed conversion ratio (FCR), but this has been mainly due to genetic improvement of growth and body composition traits. A 35-year time trend illustrated by Knap and Wang (2012) shows very clearly that the average commercial FCR has come down from 3.3 to 2.6, with a quite considerable bandwidth among terminal crosses which does not show any signs of narrowing over time (Fig. 4).

5.2 Quantification of contribution of animal breeding

5.2.1 Materials and methods

Data from an experiment, which is described in Sevillano et al. (2018), were obtained from a pig breeding company (Topigs Norsvin Research Center B.V.,

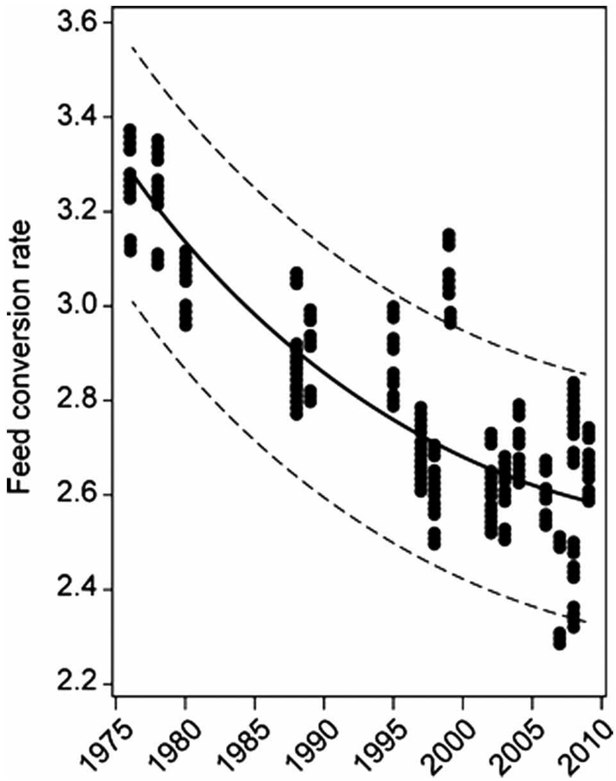

Figure 4 Time trends of feed conversion ratio in grower-finisher pigs of 103 terminal crosses, as recorded in public commercial product evaluation trials in Denmark, France, Germany, The Netherlands, UK and the United States. Unadjusted phenotypic population means, data from 18 literature and internet sources. The trend line is a spline interpolation plot through the data, with its 95% confidence limits. Figure copied from Knap and Wang (2012).

Beuningen, The Netherlands). In this experiment a corn/soybean meal (CS) diet and a cereals/alternative ingredient (CA) diet was fed to intact boars and gilts. The CS diet resembles American practice, but the impact of feed ingredients was calculated as fed in The Netherlands, which means, for example, that soybean meal mainly originated from Argentina and Brazil. The CA diet resembles Dutch practice, with many by-products in the diet. For calculating genetic progress, data from 400 pigs in 2014 (December 2013–May 2014) and 401 pigs in 2016 (November 2015–March 2016) were used (Table 7). Data only considered the growing-finishing phase (i.e. from 22 kg to approximately 120 kg of live weight). Feed composition was derived from Sevillano et al. (2018).

Emissions of GHG related to the production of feed ingredients, and their N and P content, were collected from the database of FeedPrint 2018

(Wageningen Livestock Research, 2018). Emissions of GHG are expressed in CO_2 equivalents (CO_2-eq.), which is a unit to express the contribution of different GHG to global warming, their global warming potential (GWP), relative to CO_2. Methane (CH_4) and nitrous oxide (N_2O) have a GWP of respectively 36.75 (34 for biogenic CH_4) and 298 CO_2-eq (Myhre et al., 2013). Efficiencies of N and P are expressed in percentages and calculated as N and P in output over input with feed. Protein deposition was calculated as described by Sevillano et al. (2018) and was used to calculate N deposition, whereas P deposition was calculated based on Pettey et al. (2015).

5.2.2 Results

The environmental impact caused by GHG emissions from pig production decreased by 0.6–0.7% per year dependent on the diet (Table 8). Nitrogen efficiency increased by 1.6–1.7%, whereas P efficiency increased by 0.4–0.6% over the two years. On all environmental indicators, boars performed slightly better than gilts. The N efficiency could be calculated more precisely than P efficiency because back fat thickness was measured in the experiment and used to calculate protein deposition. Therefore, the decrease in environmental impact was not only caused by a decrease in FCR (lower feed intake at same growth rate; Table 7) but also by higher protein deposition at similar growth.

5.2.3 Discussion

Data analysis focused on the grower-finisher phase because good-quality, detailed information was available for this phase. Furthermore, the chosen method for GHG emission calculation accounted for the effect of feed production on GHG emissions only. The full analysis of the whole production cycle could have given different results, as Groen et al. (2016) showed that CH_4 emissions from manure management, crop yields and reproduction performance are important processes determining whole chain GHG emissions from pig production. These results correspond well with data, shown in Fig. 1, where reproduction and rearing phase (27%) and emissions from manure (25%) contribute considerably to the total impact of pig production. Groen et al. (2016), however, also showed that FCR is the most important factor, with the highest influence on whole chain GHG emission from pig production. This is in agreement with the data shown in Fig. 1, where feed production alone explains more than 40% of GHG emissions from pig production.

Differences in the environmental impact of pigs, fed either a CS or CA diet, were most clear in P efficiency, caused by low digestibility of P in some by-products in the CA diet (e.g., rapeseed and sunflower meal). When the CS diet would have been calculated as fed in the country where corn and soybean

Table 7 Summary of pig dataset on corn/soybean meal (CS) and cereals/alternative ingredients (CA) diet with sex (Male = M, Female = F), reference year, average daily feed intake (ADFI) on starter, grower and finisher feed, average daily gain during the growing-finishing phase (ADG), average final empty body weight at slaughter and protein deposition during the growing-finishing phase

Diet	Sex	Year	ADFI starter (kg/d)	ADFI grower (kg/d)	ADFI finisher (kg/d)	ADG (g/d)	Empty body weight (kg)	Protein deposition (kg)
CS	M	2014	1.33	2.18	2.94	955	92.7	16.4
CS	M	2016	1.22	2.16	3.01	960	91.7	16.5
CS	F	2014	1.38	2.21	2.92	938	95.4	16.3
CS	F	2016	1.27	2.19	3.00	943	94.4	16.4
CA	M	2014	1.36	2.21	3.04	961	92.4	17.1
CA	M	2016	1.25	2.19	3.12	967	91.4	17.2
CA	F	2014	1.43	2.24	3.02	929	93.7	16.6
CA	F	2016	1.33	2.22	3.09	934	92.7	16.7

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Table 8 Emissions of GHG and N and P efficiency of male (M) and female (F) pigs on corn/soybean meal (CS) and cereals/alternative ingredients (CA) diet in 2014 and 2016

Diet	Sex	Year	GHG emissions (kg CO_2-eq/kg BW gain)	N efficiency (%)	P efficiency (%)
CS	M	2014	1.93	44.7	36.8
CS	M	2016	1.90	46.2	37.3
CS	F	2014	1.99	43.2	36.9
CS	F	2016	1.96	44.7	37.3
CA	M	2014	1.70	43.7	25.8
CA	M	2016	1.68	45.1	26.1
CA	F	2014	1.78	41.2	25.3
CA	F	2016	1.76	42.5	25.5

were grown, GHG impacts of pigs fed the CS diets probably would have been lower.

From this analysis we could conclude that current breeding goals decrease the environmental impact of producing pig meat and that boars are more efficient and, therefore, have a lower environmental impact than gilts.

6 Dairy cattle: environmental impact and the contribution of breeding

6.1 Historical trends

Over the past 100 years, the range of traits considered for genetic selection in dairy cattle populations has progressed to meet the demands of both industry and society (Miglior et al., 2017). At the turn of the twentieth century, dairy farmers were interested in increasing milk production; however, a systematic strategy for selection was not available. Organized milk performance recording took shape, followed quickly by conformation scoring. Methodological advances in both genetic theory and statistics around the middle of the century, together with technological innovations in computing, paved the way for powerful multi-trait analyses. As more sophisticated analytical techniques for traits were developed and incorporated into selection programs, production began to increase rapidly, and the wheels of genetic progress began to turn. By the end of the century, the focus of selection had moved away from being purely production oriented toward a more balanced breeding goal. This shift occurred partly due to increasing health and fertility issues and partly due to societal pressure and welfare concerns. Traits encompassing longevity, fertility, calving, and health, have now been integrated into national selection indices.

With these indices, milk production is still increasing per year. As shown in Fig. 5, milk production in The Netherlands has increased by 46% between 1990 and 2017. Because of the increased milk production, the feed intake has increased but to a lesser extent than milk production; therefore, the efficiency of dairy production (kg milk/kg feed) has improved over the years.

6.2 Quantification of contribution of animal breeding

6.2.1 Selection indices

Selection indexes are utilized by livestock breeders of many species around the world and aid in the selection of animals for use within a breeding program where there are several traits of economic or functional importance. Selection indexes provide an overall 'score' of an animal's genetic value for a specific purpose and are calculated based on weightings placed on individual traits that are deemed to be important for that purpose. Selection indexes assist producers in making 'balanced' selection decisions. The derivation of a selection index starts with the definition of the overall breeding objective.

The next stage in developing a selection index is to calculate economic values for each trait, generally with a bio-economic model, where the economic value is the increase in revenue from a unit change of a trait while everything else is held constant. Then, selection index theory (Hazel, 1943) is commonly used to calculate the most appropriate index weights and responses to

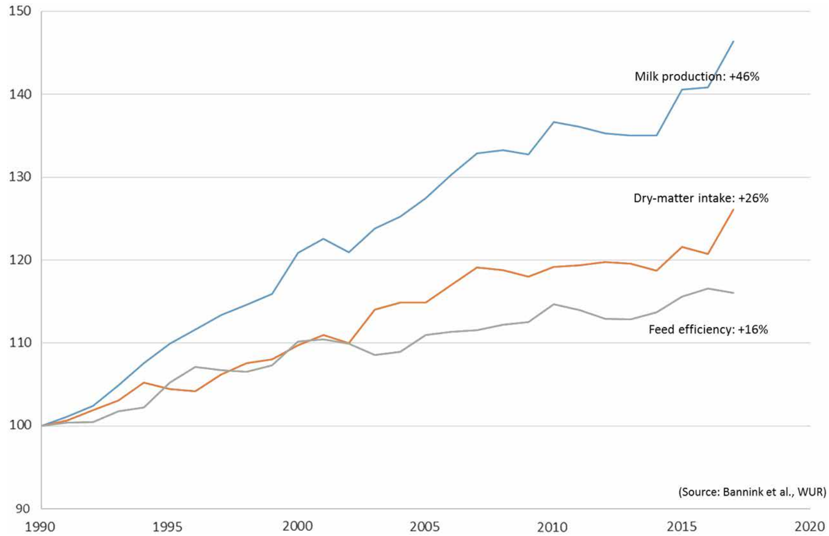

Figure 5 Trends in milk yield, feed intake and feed efficiency of the Dutch dairy cattle population between 1995 and 2017 (extended and based on Bannink et al. (2011)).

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selection for a set of traits given the genetic and phenotypic (co)variances and the economic values of traits in the index. The resulting selection index is the sum of n estimated breeding values (EBVi) for each trait multiplied by their respective index weights b_i.

$$\text{Index} = b_1\text{EBV1} + b_2\text{EBV2} + \ldots + b_n\text{EBVn}$$

6.2.2 Quantification effect of breeding

In our case, we started with a selection index representing the national breeding goal for dairy cattle of The Netherlands (CRV, 2018). The Dutch national breeding goal consists of milk component traits, longevity, health traits (udder health, claw health), fertility traits (interval first-last insemination, calving interval), conformation traits (for udder and for feed and legs), calving traits (calving ease and vitality of calves) and feed efficiency (Table 9). We added enteric CH_4 emissions to this index as a correlated trait. Genetic parameters were obtained from the literature (Lassen and Lovendahl, 2016). The heritability for enteric CH_4 production is 0.21, and genetic correlations with milk lactose, protein, fat and dry matter intake are 0.43, 0.37, 0.77 and 0.42 (−0.42 for feed saved), respectively. Correlations between enteric CH_4 production and other traits in the breeding goal were set to zero. All phenotypic correlations were also set to zero.

Selection index calculations show how much the traits are predicted to change per year. This is plotted in Fig. 6 for both CH_4 production (g/d) and for CH_4 intensity (CH_4 production expressed per kg of milk). It shows that CH_4 production per cow will steadily increase as a correlated response to selection for the current breeding goal. However, the methane intensity drops. Further reductions could be achieved when actively selecting on lower methane-emitting cows, by adding more weight on CH_4 in the national breeding goal. Selecting actively against methane would result in healthy, fertile, long-living cows that emit less CH_4. Actively selecting against CH_4 emission, however, requires large-scale recording of individual CH_4 emissions.

6.2.3 Discussion

The predicted future trends in enteric methane production are based on the genetic parameters used in the selection indices. The correlation of 0.77 between protein yield and enteric methane production is strong and impacts the results. Further research is needed to estimate reliable genetic parameters between enteric methane production and other traits of interest (e.g., the traits in the breeding goal). Estimating these genetic parameters requires that a large enough dataset is built, which includes records of enteric CH_4 emission of many individual cows.

Table 9 Heritabilities of (on diagonal) and genetic correlations (below diagonal) between the 15 traits in the Dutch national breeding goal (CRV, 2018)

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
1. Lactose	**0.43**														
2. Fat	0.38	**0.58**													
3. Protein	0.88	0.58	**0.50**												
4. Longevity	0.36	0.35	0.42	**0.12**											
5. Udder health	−0.03	−0.02	−0.06	0.36	**0.09**										
6. Interval first-last insemination	−0.34	−0.24	−0.29	0.25	0.27	**0.08**									
7. Calving interval	−0.44	−0.33	−0.37	0.11	0.21	0.85	**0.15**								
8. Udder conformation	−0.08	−0.04	−0.10	0.11	0.27	−0.05	0.00	**0.34**							
9. Feet and leg conformation	0.02	0.04	0.05	0.25	0.21	0.00	0.00	0.35	**0.17**						
10. Direct calving ease	0.07	0.15	0.11	0.24	0.15	0.20	0.24	0.00	0.00	**0.07**					
11. Maternal calving ease	0.00	0.00	0.00	0.16	0.09	0.25	0.24	0.10	0.10	0.19	**0.05**				
12. Direct vitality	0.05	0.09	0.02	0.14	0.05	0.10	0.14	0.00	0.00	0.60	0.14	**0.04**			
13. Maternal vitality	−0.04	−0.07	0.03	0.16	0.07	0.32	0.24	0.00	0.00	0.11	0.34	−0.16	**0.09**		
14. Claw health	0.00	0.15	0.07	0.33	0.09	0.10	0.14	0.15	0.65	0.16	0.06	0.03	0.10	**0.18**	
15. Saved feed costs	0.20	0.35	0.30	0.50	−0.03	−0.10	−0.30	−0.09	−0.29	0.41	−0.20	0.17	−0.05	0.11	**0.25**

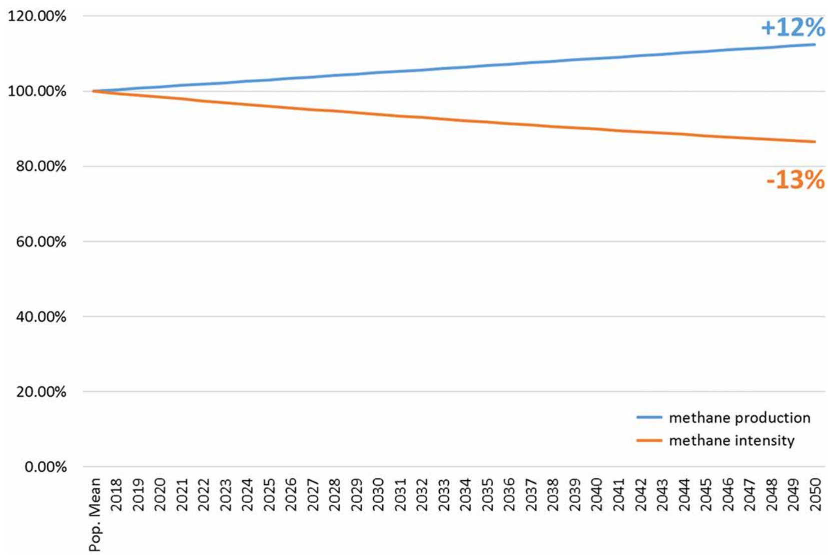

Figure 6 Expected future trends in CH_4 production (g/d) and CH_4 intensity (g/kg milk) for the Dutch dairy cattle population with breeding on the current national breeding goal.

7 Conclusion

Animal production is responsible for 14.5% of total anthropogenic GHG emissions. Approximately half of these emissions originate directly from animal production, whereas the other half comes from feed production. Animal breeding aims at improving animal production and efficient use of resources, which results in a reduction of the environmental impact. The objective of this study was to quantify the contribution of animal breeding in reducing the environmental impact of the four major livestock species in The Netherlands, namely broilers (meat), laying hens (eggs), pigs (meat) and dairy cattle (milk).

A literature review was performed to assess the current status of and historical trends in environmental impact, mainly focused on GHG emissions, and general performance criteria, like feed efficiency and lifetime production. Emissions related to the feed production dominate the impacts by broilers and laying hens. For pigs, the emissions during feed production and from manure are important contributors. For dairy cattle, as being ruminants, enteric methane emission is a large contributor to total GHG emissions. Historical trends show considerable improvements in efficiency over the last decades, in which breeding has an important role. The literature review showed that the contribution of breeding to reducing the environmental impact of animal production is led by an indirect response through selection on increased efficiency.

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Also a quantitative assessment was made on the current environmental impact of the four animal products and the effect of recent genetic improvements. For broiler meat, chicken eggs and pig meat the focus was on GHG emissions, and nitrogen (N) and phosphorus (P) efficiency, whereas for dairy the focus was on enteric methane emissions, an important contributor to GHG emissions. Data were partly provided by breeding organizations, that is, the partners in the Breed4Food consortium (www.breed4food.com).

The analyses in this chapter demonstrate that animal breeding can provide a mitigation tool to lower the environmental impact of livestock species. Genetic improvement of livestock is a particularly cost-effective technology, producing permanent and cumulative changes in performance:

- For broilers, it was shown that GHG emissions decreased with 1.7% and N and P efficiency increased by 1.6% per year due to the current breeding goals.
- For laying hens, white and brown hens were considered and it was concluded that white hens currently have a lower GHG impact and better N and P efficiency than brown hens and that improvements over the past 10 years went faster for white hens.
- For pigs, data were available from a well-controlled study with two diets and animals divided by sex over a time frame of two years. Results showed that in the growing-fattening phase of pigs, GHG emissions decrease and N and P efficiency increase with the current breeding goal. Furthermore, boars had a lower environmental impact than gilts.
- For dairy cattle, results showed that with the current breeding goal, CH_4 production per cow per day increases but CH_4 intensity (i.e. CH_4 production per kg milk) decreases.

All reported results are achieved without specific selection on environmental traits, but as an indirect response of the current breeding goals for each species, which is a combination of health, growth and (feed) efficiency. If direct selection of environmental traits is desired, recording of new traits is required, for example, N and P contents of meat and eggs and methane emission of individual dairy cows.

Direct measurement of GHG impact of animal production is difficult, but not impossible, which hampers active selection against GHG emissions of animals. In the short run, indirect selection against GHG emissions could be further optimized by putting more selection pressure on efficiency traits, while accounting for the effects on other important traits, for example, health, longevity and reproduction. In the long run, recording schemes could be set up to either record the desired traits on commercial farms (for dairy) or in parental stock (for pigs and poultry).

The LCA analyses performed in this study could be further improved by also including information of the parent stock and rearing phases. It is expected that when including genetic progress in parent stock and rearing phases of parents and commercials, the contribution of genetics to reduce GHG emissions per kg product has an even bigger impact.

8 Where to look for further information

8.1 Further reading

Selection index theory: Hazel, L. N. (1943) The genetic basis for constructing selection indexes. *Genetics* 28, 476–490. & Lush, J. L. (1960) Improving dairy cattle by breeding. I. Current status and outlook. *Journal of Dairy Science* 43, 702–706.

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Report on 'The contribution of breeding to reducing environmental impact of animal production', https://library.wur.nl/WebQuery/wurpubs/549934.

8.2 Key conferences

WCGALP (world conference of genetics applied to animal production) is well attended by members of the animal breeding community (industry and scientists).

GGAA (greenhouse gas of agriculture and animal) is well attended by scientists in all disciplines (nutrition, breeding, microbiology, etc.) working on the reduction of environmental impact of livestock production.

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Chapter 4

Quantifying the contribution of livestock health issues to the environmental impact of their production systems

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1 Introduction

There is an increased interest about the consequences of animal health on the environmental impact of livestock systems. This should not be surprising given the association between animal health and the efficiency with which they utilise the important resource inputs to livestock production, which are key factors in determining the environmental impact of these systems (de Vries and de Boer, 2010; McAuliffe et al., 2016; Tallentire et al., 2018). Animal health has major consequences on how animals utilise their resources, including feed, increasing inputs, including medication, and reducing outputs, such as amount of milk or meat produced per animal, or per unit of input (Perry et al., 2018).

While the consequences of animal health for the environmental impacts of livestock systems have been addressed to a limited extent, for example (ADAS, 2015; Mostert et al., 2018; Skuce et al., 2016), to date the focus has been almost exclusively on the implications of conditions which impact on ruminant systems and the implications of these for greenhouse gases (GHGs). This focus is natural, due to the high profile nature of both the issue of anthropogenic GHGs and the large contribution of ruminant systems to this issue (Gerber

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et al., 2013). However, the environmental issues facing the livestock industry are much broader than this: livestock systems have a major impact on global land use (Weindl et al., 2017), water use (Doreau et al., 2012) and the acidification of soils and waterways (Leip et al., 2015), to name just some of the issues. In addition, the majority of meat consumed globally is from pigs and poultry (FAO, 2019), but these production systems cause less GHGs than ruminants (Gerber et al., 2013). With consumption of non-ruminant livestock products expected to increase dramatically to 2050 (FAO, 2011), non-ruminant animal farming faces concerns around the impact of feed production (particularly the associated land and water use), and the pollution caused by their manure (de Vries and de Boer, 2010; Poore and Nemecek, 2018). These issues represent important future challenges that the industry must overcome to maintain its public acceptability.

Climate change is expected to exacerbate animal health issues, which in turn would affect the environmental impact of livestock systems, thus creating a vicious circle that links animal health, environmental impact and climate change. A combination of changes in air temperature, precipitation, as well as the frequency and magnitude of extreme weather events are all expected to have a negative impact on outcomes for livestock health and welfare (Lacetera, 2019). Climate change may also affect the quantity, quality and composition of animal feed, both directly and indirectly. The increase in the incidence of fungal contamination of animal feedstuffs is one such indirect effect on animal health due to increased concentration of mycotoxins in the feed (Bernabucci et al., 2011).

The focus of our chapter is on the environmental impact of endemic livestock health challenges that lead to deterioration in animal health, and on the potential impacts arising from their mitigations. This is because, while epidemics may have devastating effects on livestock systems, these would be usually controlled through stamping out by eradication of affected herds and flocks (Geering et al., 1999). In some instances, where large-scale eradication is unacceptable, other control measures, such as vaccination programmes supplemented by other disease control measures, can be used to eliminate many epidemic health and welfare challenges (Roth, 2011). The dearth of information on this issue in relation to the environmental impacts of epidemic health challenges for livestock means we are unable to give the latter issue detailed consideration in our chapter.

The first part of our chapter concentrates on the potential of animal health to affect the environmental impact of livestock systems. Because in several ways animal health affects environmental impact of livestock, that is, through decreases in productivity and efficiency and increases in culling, applies to all livestock systems, we review the consequences of animal health on resource inputs and outputs across livestock systems. This includes the implications for a variety of important variables for environmental impacts, such as the utilisation of nutrients in feed, mortality/culling rates, breeding performance and also milk quality in the case of dairy systems (ADAS, 2015). Subsequently, we review the literature to date which has quantified the impact of health challenges for the environmental impacts of livestock systems.

The potential of successful health interventions to mitigate negative environmental impacts represents a point of synergy between concerns around environmental sustainability and animal welfare, both of which represent 'hot topics' in the discourse surrounding the livestock industry and its sustainability. This is a topic that has been under-represented in the literature, both because of the difficulty to address it in practical terms, and because of a lack of framework that will allow us to do so (Rushton, 2017). The challenges associated with modelling health interventions and their potential to mitigate environmental impacts constitute the last section in our chapter. This part has a heuristic value, given the current lack of information in the literature.

2 Consequences of health challenges on resource inputs and outputs of the animal and production system

Given that a large part of the environmental impact assessment of livestock systems is done on the basis of the principle of mass flows (FAO, 2016a, 2018), below we review the consequences of health challenges on how an animal uses the nutrient resources it consumes. One of the most profound consequences of health challenges is a decrease in the efficiency of how nutrient resources are used (Kyriazakis and Houdijk, 2007; Sandberg et al., 2007), which in turn will have consequences on the fate of nutrient excretion. However, in addition to this, health challenges may have indirect consequences on the resource inputs required to produce the key outputs of livestock production systems, such as milk, eggs and meat. These indirect consequences would arise from the fact that system outputs may need to be discarded because of their quality or for safety reasons; individual animals may need to be culled because their performance is no longer considered economically viable by their keepers; and ultimately some animals may succumb to the consequences of the health challenges. Alternatively, some health challenges would require inputs from pharmaceuticals or specialist management, which in turn utilise resources for their production. For these reasons, both the direct and indirect consequences of health challenges on resource inputs into animals and their production system are considered below.

2.1 Anorexia during health challenges

A reduction in the voluntary food intake, henceforth called anorexia, is associated with most challenges that affect livestock health (Kyriazakis, 2014). During subclinical infections, anorexia manifests as a ~20% reduction in voluntary food intake and is frequently the only indication that an animal's health is being challenged (Hite et al., 2020; Kyriazakis et al., 1998). In more severe infections, there may be a complete cessation of eating, but this is usually associated with severe infection outcomes, such as death (Lough et al., 2015). Anorexia is also associated with metabolic diseases, such as acidosis or ketosis in ruminants, and is often used as an indicator that the management of the animals has not been appropriate and requires attention (González et al., 2008). Ketosis in lactating cows, which results from a severe energy gap between intake and output, may lead to complete cessation of eating, but intake returns almost immediate to its previous levels after appropriate treatment (Andersson, 1988; Berge and Vertenten, 2014).

With all other things being equal, anorexia during health challenges would be expected to lead to a reduction in animal performance, and therefore, the animal would need to take more time to achieve the same production output. This in itself would imply that the animal would need to divert proportionally more resources in maintaining itself for a longer period of time, and therefore the efficiency with which its food is being utilised would be reduced. However, studies that compared the performance of infected animals with that of uninfected animals that have been offered the same amount of food (usually called pair-fed animals), suggest that the former utilise the same amount of food less efficiently than the latter (Holmes, 1993). A frequent observation is that infected animals grow more slowly and deposit less protein and fat in their bodies than the uninfected pair-fed animals (Holmes, 1993; Escobar et al., 2004). This implies that infection per se is associated with other effects in feed utilisation, which will be detailed below.

2.2 The effects of infectious challenges on digestion, absorption and utilisation of nutrient resources

Due to their significance, the effects of infectious challenges on the digestion, absorption and utilisation of resources has been studied most extensively during gastrointestinal parasitism. Depending on the site of infection, digestion and absorption of nutrients may be significantly affected (Sandberg et al., 2007). In ruminant animals parasitized with gastrointestinal parasites, a reduction in nutrient digestion has been observed, especially when the parasites affect the organs of digestion. This is the case of parasitism with the abomasal parasites, *Ostertagia ostertagi* in cattle and *Teladorsagia circumcincta* in sheep, as the gastric glands of the abomasum are damaged (Holmes, 1993; Ceï et al., 2018). Similarly, whether the process of nutrient absorption is affected would depend on the site of gastrointestinal parasitism and its effect on organ integrity. Several gastrointestinal parasites in ruminants are associated with gastrointestinal damage, such as epithelial and mucus loss (Coop and Kyriazakis, 1999; Mavrot et al., 2015). In the case of blood-sucking parasites, such as *Haemonchus contortus*, blood and plasma loss is also the outcome of the infection. All these effects are associated with the loss of considerable quantities of protein into the digestive tract of the host animal (Holmes, 1993).

Similar observations have been reported in non-ruminant animals parasitized by gastrointestinal parasites. The apparent digestibility of protein was reduced during pathogen challenges with parasitic worms in pigs and was smaller for pathogens that affected mainly the stomach, compared with those affecting latter parts of the gastrointestinal tract (Hale et al., 1985; Midha et al., 2018) . For instance, in broilers infected with the small intestine coccidia *Eimeria acervulina*, the apparent ileal digestibility of every single amino acid in the food was reduced by ~5% (Rochell et al., 2016). As well as damaging parasitized enterocytes, *Eimeria* parasitism increases plasma protein leakage and mucogenesis, thus increasing endogenous amino acid flow.

Infectious challenges are expected to increase the (maintenance) requirements for nutrient resources through a variety of routes: (1) animals need to repair damage to its tissues or replenish lost fluids as a direct consequence of infection. (2) Fever, which accompanies several infections would increase energy expenditure and hence requirements. (3) Infected animals need to mount an immune response in order to cope with the infection and eventually overcome it. Hosts may respond with innate or acquired immunity, or a combination of these, depending on the type of challenge and the stage of a particular infection (Sandberg et al., 2007).

The Sandberg et al. (2007) investigations were the most comprehensive attempt to quantify such effects on nutrient resource requirements. It is difficult to make generalisations about these quantitative effects, given that different infectious challenges would increase the different sources of increased requirements to different extents, but some suggestions can be made. As far increases in energy requirements are concerned, it has been suggested that maintenance requirements to be 1.7–2.2 times as great in challenged as opposed to non-challenged animals. There appears to be highly specific pathogen differences, with the greatest energetic cost occurring for infectious challenges associated with fever, especially during the stages of acute infection (Sandberg et al., 2006). Otesile et al. (1991) challenged pigs with *Trypanosoma brucei* and found that they had a significant increase (1.7 times) in energy maintenance requirements than healthy controls.

In the case of increased protein requirements due to infection challenges, it has been estimated that protein maintenance animals challenged by a varied of pathogenic and non-pathogenic challenges varied from 1.3 times to 4 times that in the healthy. Webel et al. (1998) found that chicks challenged by a non-pathogenic antigen had increased lysine maintenance requirements, tended to have increased threonine maintenance requirements, but there was no effect for arginine requirements. Different effects on maintenance may suggest that amino acid requirements are affected to different extents and that this needs to be accounted for the changes in their utilisation during infectious challenges.

2.3 The effects of non-infectious challenges on digestion, absorption and utilisation of nutrient resources

Production diseases, which arise from the management of usually high-producing animals, are a good case to consider the effect of non-infectious challenges on digestion, absorption and utilisation of nutrient resources. They include the group of diseases usually referred to as 'metabolic diseases', such as rumen acidosis and ketosis in dairy cows, pregnancy toxaemia in ewes and ascites in broilers. Most of these conditions, at least in their subclinical form, may predispose animals to susceptibility to infectious challenges, and therefore, it is difficult to disassociate their consequences from them. For example, while it would be expected that the nutrient resource requirements which are associated with lameness would be very small, several metabolic diseases with significant resource requirements are a predisposing factor for lameness. Their subclinical forms are also very widespread, for example, sub-clinical ketosis was found to have an average prevalence of 24.1% in a global study of the issue in dairy herds (Brunner et al., 2019).

Sub-acute ruminal acidosis (SARA) in dairy cows is a good case in point for its effects on nutrient digestion and use. SARA is estimated to have an overall prevalence in dairy herds of 11–33% in studies that have investigated the issue (Kleen and Cannizzo, 2012). It arises from feeding of high-energy density diets to meet the energy requirements of high-yielding cows, which leads to reduced rumen pH and increased risk of acidosis. Ruminal acidosis can cause erratic fluctuations in feed intake, and low rumen pH causes the cow to go 'off-feed', which reduces the production of fermentation acids, allowing the pH to recover. Changes in the rumen environment lead to reduced digestion of fibrous ingredients and lowers the efficiency of microbial protein production in the rumen (Beauchemin, 2007), thus affecting directly digestion and absorption of nutrient resources. At the same time, low rumen pH leads to inflammatory changes both locally, including damage to the gastrointestinal tract wall, and systemically, associated with an acute phase response (Bertoni et al., 2008). This acute phase response may not only be associated with increased nutrient resource requirements, but also with significant changes in the energy and lipid metabolism in different body tissues. Due to the nature and the focus of the condition, and the difficulty in disassociating these effects, they have not been quantified separately.

Because most production diseases are associated with negative nutrient resource balance and the animal needs to sacrifice its body or tissue mastitis reserves for the sake of continuing the production of milk or eggs, there would be less nutrient resources available for the maintenance of ‘integrity’ functions, such as defences to pathogens. This would make the animal more vulnerable to challenges, previously easy to control. This explanation has been put forward for the occurrence of mastitis in high-yielding dairy cows (O’Rourke, 2009), which are less able to direct resources towards the functions of innate and acquired immunity.

2.4 Indirect consequences of health challenges on system inputs and outputs

In growing animals, the usual consequences of health challenges are increases in their mortality and decreases in their growth rate and feed efficiency. This is especially the case with endemic viral diseases, affecting non-vaccinated animals. For example, infection with the porcine reproductive and respiratory syndrome virus (PRRSv) resulted in an additional 11% increase in mortality among nursery pigs and 6% increase among growing pigs in the United States pig herds (Kliebenstein et al., 2004). The same reports suggest increased use of medication among PRRSv-affected herds, although the increase is not sufficiently quantified.

In reproducing animals, health challenges lead directly or indirectly to reproductive failure/fertility and culling. In herds endemically infected by PRRSv, there is a substantial increase in the percentage of abortions per sow year, and decreases in weaned pigs produced per sow year and number of farrowings per sow year (Valdes-Donoso et al., 2018). Similarly, it has long been established that there is a clear association between health and fertility for cows affected by conditions such as mastitis, lameness and ketosis (Bertoni et al., 2008). In all these cases, the percentage of animals which are prematurely culled or die from the conditions may be substantial.

Finally, health challenges may lead to product loss from affected animals. Such losses may be associated with the condemnation or discard of milk, and the partial or whole condemnation of carcasses from normal or emergency slaughter (Garcia et al., 2019). Specifically, during mastitis the condemnation of milk is associated with increased inflammatory cell content, which affects milk quality, and antibiotic residues in the milk due to treatment of infected udders. This is on top of the reduction in milk yield and any culling of cows.

Above, we developed a framework which enables to view the effects of health challenges on nutrient resource requirements, which in turn enables to quantify their effects on mass flows. The framework should allow us to review and account for the effect of health challenges on environmental impact, and this is done below.

3 Quantifying the environmental impact of health challenges

In this section, we review the literature published to date presenting quantitative results on the environmental impacts of health issues in livestock production systems. The life cycle assessment (LCA) framework is the widely accepted way to holistically assess the environmental impact of livestock systems (FAO, 2016b, 2016c, 2018), although there are no specific guidelines when applying the approach to health issues. While there have been a large number of environmental impact studies of livestock systems as reviewed by Poore and Nemecek (2018), a relatively small proportion of these focus on the implications of animal health issues for the environmental impacts of livestock production. These studies have begun to quantify the link between animal health issues and the environmental impacts of livestock systems, and they can broadly be grouped into two categories:

1. Studies that model the environmental impacts caused by the existence of a health issue.
2. Studies that investigate the potential of specific treatment strategies to mitigate a health issue and the potential implications for the environmental impacts of the production system.

Both approaches can be useful exercises when used appropriately. In the first category, approaches can be entirely hypothetical and take a broad view across multiple different diseases to understand where the greatest potential benefits for targeted treatment strategies lie, both in terms of environmental impact mitigation, and in traditional economic terms. Using the second approach, studies can provide a realistic picture of the real-world benefits in terms of impact mitigation that specific treatment or prevention strategies can have. In this section, we focus on studies that have taken the first approach.

Table 1 lists the studies found to date that have quantified the environmental impacts caused by specific health issues in livestock production systems. Almost all studies to date that have investigated the impact of health issues are on ruminant production systems, the exception being Li et al. (2015), who investigated the effect of vaccination for PRRSv on key sources of GHG emissions for pig production (see Section 4 for further details on this). Only two studies considered environmental impact categories beyond GHGs in relation to health issues (Chen et al., 2016; Hospido and Sonesson, 2005). The most wide-ranging and comprehensive study found was the ADAS (2015) report on the impact of endemic diseases and conditions for the GHGs and economic performance across the UK sectors of beef, dairy and dual purpose cattle systems. The ADAS study quantified both the GHGs caused by 10 important

Table 1 A summary of quantitative results reported in literature on the environmental impact of livestock health issues. Where a study addressed more than one disease, its results are reported separately in the table below

Study	Study type	Species (system)	Disease/s studied	Impact of health issue for GHGs (%)	Functional units and other methodological details	Other notes
(Skuce et al., 2016)	Modelling	Sheep	Gastrointestinal nematodes	+10	1 kg carcass weight	Theoretical calculation based on three levels of infection. +10% is based on going from 20% of flock infected to 0.
(Fox et al., 2018)	Experimental trial	Sheep (Lambs)	Parasitic worm infection (Teladorsagia circumcincta)	+33 methane emissions	Methane per dry matter intake	No overall GHG assessment.
(Skuce et al., 2016)	Modelling	Cattle (beef)	Neosporosis	+2.2–4.7	1 kg carcass weight	Theoretical calculation based on three levels of infection. Highest increase is based on going from 20% of flock infected to 0.
(Macleod et al., 2018)	Modelling	Cattle (dairy and beef)	Trypanosomiasis	+9	1 kg of edible protein at the farm gate	Milk and meat protein are equivalent in this calculation.
(Chen et al., 2016)	Modelling	Cattle (dairy)	Lameness	+8	1 kg of fat and protein-corrected milk (FPCM). FPCM milk = milk volume × (0.25 + 0.122 × fat% + 0.077 × protein%)	Study considered other impact categories, namely eutrophication potential, acidification potential and abiotic depletion.

(*Continued*)

Table 1 (*Continued*)

Study	Study type	Species (system)	Disease/s studied	Impact of health issue for GHGs (%)	Functional units and other methodological details	Other notes
(Mostert et al., 2018)	Modelling	Cattle (dairy)	Subclinical ketosis and related conditions	+2.3 per case	1 ton of FPCM milk	Calculation includes assumptions of probabilities for associated conditions including mastitis, metritis, displaced abomasum, lameness and clinical ketosis.
(Özkan et al., 2015)	Modelling	Cattle (dairy)	Mastitis	+2	1 L milk produced	
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Overall impact of disease on UK herd GHGs	+6 dairy and +6.6 beef	1000 L FPCM milk and 1 tonne edible carcass weight	The overall impact of all health issues considered for the UK cattle sector compared to theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Bovine viral diarrhoea (BVD)	+20 dairy and +130 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Johne's disease	+25 dairy and +40% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Salmonella	+20 dairy and +30 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.

(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Liver flukes	+10 dairy and +10 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	IBR	+8 dairy and +20 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Lameness	+7.5 dairy only	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Mastitis	+7 dairy only	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Infertility	+16 dairy and +21% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Calf pneumonia	+1 dairy and +4 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Calf diarrhoea	<1 dairy and +4 beef	1000 L FPCM milk and 1 tonne edible carcass weight	Increase for cases where farms impacted by the condition compared to a theoretical healthy herd.

health issues in UK systems and the potential of associated treatments and prevention strategies to mitigate them.

The 10 diseases and conditions investigated by ADAS were: infectious bovine rhinotracheitis (IBR), bovine viral diarrhoea (BVD), calf pneumonia, calf diarrhoea, liver fluke, mastitis, lameness, infertility, paratuberculosis and salmonella (ADAS, 2015). Other health issues for cattle considered in literature to date include: subclinical ketosis (Mostert et al., 2018), trypanosomiasis (Macleod et al., 2018) and neosporosis disease (Skuce et al., 2016). The studies show a wide range in the GHG emissions attributable to different health issues, the most dramatic results including an estimated 130% increase in GHG emissions from beef farms affected by bovine viral diarrhoea (BVD), and an increase of 25% in GHGs for dairy farms impacted by Johne's disease (paratuberculosis) compared to a 'healthy farm' scenario (ADAS, 2015).

The underlying data behind these models are able to shed light on why certain health issues cause greater GHG emission increases than others. Previously in this chapter we listed various pathways through which health issues can negatively affect animal performance in ways likely to increase GHGs from livestock production, but the studies in Table 1 quantify this link directly. Table 2 (reproduced from ADAS (2015)) summarises the major assumptions used in that study regarding changes in dairy cow performance when affected by different health issues compared to a 'healthy herd'. For example, on average, BVD increases mortality from 2% to nearly 5%, increases levels of abortion/still birth rates and reduces conception rates (Gunn et al., 2004; Houe, 1999). In addition to the increased mortality risk, the issues with reduced productivity mean that on average culling rate increases from 17% to 20%. All of these changes are likely to cause an increase in the GHGs of the farming system by decreasing the productive outputs. As can be seen from Table 2, the presence of Johne's disease (+20%) and *Salmonella* (+16%) were deemed to result in the largest increases in GHGs per cow per year compared to a healthy herd scenario. In the ADAS report calculations, BVD only produced a 10% increase in GHGs per cow per year, but due to losses in productive outputs (particularly reduced milk yield) its overall impact on GHGs per 1000 litres of milk produced was similar to that of salmonella. The differences between the data for Johne's disease and salmonella/BVD demonstrate how different mechanisms can drive GHG increases from health conditions. The increase in GHGs from salmonella and BVD are driven by increase in mortality, decrease in milk yield (this is more dramatic for BVD) and a reduction in the calving rate. However, for Johne's disease it is a dramatic increase in the culling rate from 17% to 33% along with some losses in milk yield which drive the increase in GHGs. This demonstrates that how important management responses can be for determining the impact of health conditions on GHGs, and increased GHGs do not just emanate from direct impacts on animal performance.

While conditions such as BVD and Johne's disease have been identified as causing the highest increases in GHGs for UK cattle in beef and dairy systems respectively, the literature beyond the ADAS report has tended to focus on conditions that the industry considers the most important from an economic perspective. As mastitis and lameness have previously been identified as some of the most economically important diseases in dairy herds (ADAS, 2015), they are associated with some of the highest losses in cattle welfare (Bennett and IJpelaar, 2003), and mastitis is the most likely condition to cause an animal to be culled (Liang, 2013; Ózsvári, 2017). It is thus unsurprising that these health issues are the subject of multiple studies with respect to their environmental impacts. The increase in GHGs caused by widespread problems such as mastitis, lameness and subclinical ketosis are calculated to be in the range of 2–8% in several studies (ADAS, 2015; Chen et al., 2016; Hospido and Sonesson, 2005; Mostert et al., 2018; Özkan et al., 2015). As can be seen in Table 2, instances of mastitis are linked to increased mortality, while also decreasing lactation yield, with the complications associated with the condition leading farmers to increase the culling rate (Rollin et al., 2015). The condition also leads to milk being discarded, due to treatment of the condition and instances of raised somatic cell counts. All of these factors along with its very high prevalence in dairy herds (estimated at 37% of all cows over 1 year in the ADAS report), mean the condition inevitably has a large effect on animal performance in dairy herds and results in GHG increases of around 7% (ADAS, 2015). Both Chen et al. (2016) and the ADAS report attributed an increase of around 8% in GHGs to scenarios of increased lameness on farm. These increases are caused by reduced feed intake resulting in weight loss and reduction in milk production, and lameness experienced early in lactation is also associated with reduced fertility, which further increases GHG intensity of the milk produced (Collick et al., 1989; Lucey et al., 1986). Chen et al. (2016) also reported increases of around 10% in acidification, eutrophication and fossil resource depletion from dairy production due to lameness. They concluded that for lameness, the same mechanisms of reduced feed intake and milk production were driving the increase in all impact categories considered. The lack of consideration of impacts beyond GHGs in this area of study to date means it is difficult to state if this is the case for other health issues beyond lameness.

Another interesting case study is that of subclinical ketosis (SCK), the implications of which for GHGs was investigated in a detailed LCA study by Mostert et al. (2018). This study took into account the full impact of SCK on Dutch dairy herds including the likelihood of complications such as mastitis, metritis and lameness, as well as progression to clinical ketosis. While SCK is prevalent in up to 49% of European dairy herds causing significant economic losses, the study suggested that overall the impact of SCK in terms of increased

Table 2 A summary of the estimated impact of various health issues on livestock performance parameters and greenhouse gas (GHG) emissions in UK dairy systems (adapted from ADAS (2015)).

	Mortality (%)	Lactation yield (litres)	Cull rate (%)	Calving interval (days)	Productive life (lactations)	Calves per cow per year	GHG emissions per cow
Healthy	2.0	7831	17	368	6.1	0.89	6995
Industrial baseline	3.0	7539	26	362	3.9	0.86	7105
Bovine viral diarrhoea (BVD)	4.9	7204	20	451	5.0	0.67	7670
Calf pneumonia	2.0	7831	17	368	6.1	0.89	7039
Calf scour	2.0	7831	17	368	6.1	0.89	7003
Infectious bovine rhinotracheitis	4.0	7560	17	401	6.0	0.78	7326
Infertility	2.0	7194	20	449	5.2	0.73	7485
Lameness	2.0	7534	19	387	3.2	0.85	7245
Liver fluke	3.9	7804	17	376	6.0	0.87	7693
Mastitis	3.2	7766	19	368	5.2	0.89	7415
Paratuberculosis	2.0	7590	33	405	3.1	0.81	8423
Salmonella	5.0	7584	17	402	6.0	0.77	8104

GHGs is relatively small at 0.9% per case of SCK, increasing to 2.3% when predicted complications are accounted for (Mostert et al., 2018). Importantly, however, the Mostert study presented detailed data on exactly what caused the predicted increases in GHGs, which should serve as a template for other studies in this area. For cases of SCK only, with no further complications, 72% of the predicted increase in GHGs is caused by a prolonged calving interval with the rest caused by reduced milk production. When individual cows experience complications relating to SCK then the potential increases in GHGs increase and the source of these increases change. In those cases involving mastitis-displaced abomasum and clinical ketosis, the increased removal of heifers for culling was the second largest contributing factor to increased GHGs. Contrastingly, in cases involving lameness, reduced milk production was the largest contributing factor to increased GHGs followed by prolonged interval between calving. The Mostert et al. (2018) study predicted that individual cases of culled cows with SCK increased GHG emissions by around 20%, whereas any cases of cows which died on farm increased GHGs by 50% per case compared to healthy cows. This was because meat from culled cows was considered to be viable and replace other low-value meat products, thus off-setting the need to produce these, whereas meat from cows that died on farm was assumed to be discarded. While important queries may be raised about the validity of this methodological approach, the detail in such calculations opens up a potential future pathway in bringing environmental impact considerations into the existing economic models which optimise culling decisions in dairy herds based on economic outcomes (Liang, 2013).

4 A framework to evaluate the environmental impact of health interventions

A more interesting and challenging question than what impacts are caused by the existence of health issues, is how to quantify the potential of prevention and treatment strategies to mitigate environmental impacts in livestock systems. The added interest here is that rather than just stating the extra environmental burden caused by certain health conditions, researchers can use modelling to identify the most effective strategies that can reduce this. The challenges they face in doing so are actually similar to those faced by farmers, vets and economists when trying to establish which health interventions are actually cost-effective. As described by Rushton (2017) there are still huge issues with the availability of baseline level data on the prevalence of endemic diseases and welfare issues in livestock populations, even in OECD countries. This hampers our ability to understand how cost-effective health interventions are likely to be at the scale of a national or transnational population.

4.1 Studies that have modelled the environmental impact of health interventions

A key difference between quantifying the potential environmental impact mitigation from health interventions and the economic consequences of doing so is that the environmental impact of several interventions, for example, producing vaccines or medicines, is likely to be small in comparison to any significant projected benefits through improved animal performance. The relatively small environmental impact of interventions means there is unlikely to be any meaningful trade-off in terms of environmental impacts if a treatment or prevention strategy has been shown to be effective (ADAS, 2015). This completely contrasts with the economic modelling where there is often a fine balance between such costs and benefits. For example, vaccinations only become cost-effective when the risk from disease is deemed to be above a certain threshold (Linhares et al., 2015). In reality, economic and animal welfare considerations still drive decisions around whether to implement health interventions on farm. As such, the question of how any cost-effective interventions are in terms of environmental impact reduction is of great interest to researchers in this area looking to incorporate multiple objectives in their modelling to establish why particular health interventions should be prioritised.

This is reflected in work such as the ADAS (2015) report where marginal abatement cost curve (MACC) analysis was performed to identify the most cost-effective strategies to reduce GHGs in UK cattle through health interventions (ADAS, 2015). As such, it was modelled on the national scale and linked epidemiological, LCA and economic modelling. Their use of MACC analysis to relate the treatment/prevention of health issues, GHGs and economic in this way was innovative and to our knowledge has not been replicated elsewhere for livestock systems. However, the MACC methodology is known to have important limitations. For example, it is not able to properly rank cost-negative mitigation measures as traditional MACC methodology will tend to favour scenarios that actually produce lower emission savings (Taylor, 2012; Ward, 2014). More generally, the methodology is viewed as lacking transparency with respect to the calculations in some cases, with limited ability to interpret uncertainty in the underlying data and being unable to account for interactions between individual treatment options (Kesicki and Ekins, 2012). Despite these criticisms it will be important that economic considerations are integrated into future work on health issues and environmental impacts in livestock systems, whether using MACCs or other relevant methods to account for environmental externalities in economic models of livestock systems (e.g. Jongeneel et al., 2014).

Table 3 summarises the studies we found that assessed the impact of specific treatment and prevention strategies to reduce GHGs caused by health issues in animal production systems. Fewer studies have addressed the more difficult task of assessing the impact of treatment and prevention strategies, than those quantifying the impacts caused by health conditions discussed in Section 3. The majority of the studies were modelling exercises applying empirical evidence within an LCA modelling framework. However, some experimental studies, which directly measured GHG emissions were also identified (Fox et al., 2018; Houdijk et al., 2017; Li et al., 2015). The predicted benefits for GHGs from specific treatment/prevention strategies based on the evidence of their effectiveness are naturally more conservative than the theoretical reductions calculated based on eliminating health issues. This is because they need to be based on evidence that the predicted improvements to system performance can be achieved through these treatment and prevention strategies.

The shared challenges between modelling the potential environmental impact mitigation from health interventions and their economic consequences, combined with increased interest in understanding the potential environmental mitigation from health interventions, mean a guideline framework for researchers constructing such models would be an important resource.

4.2 Integrating epidemiological data in environmental impact assessment

Figure 1 sets out the important components of modelling the environmental impacts of health issues and health interventions in livestock systems. An essential component for understanding the potential of interventions to mitigate health issues is to model the expected changes to production parameters and animal health outcomes from any individual or combination of interventions. This includes mortality rates, fertility rates, the utilisation of feed and quality characteristics of products from the farming system. This is consistent with the frameworks currently used by LCAs to assess the consequences of management changes on the environmental impact of a livestock system. The level of data required for the epidemiological component of the exercise is entirely dependent on the scale at which researchers model on, and in many cases appropriate data may only be available at the farm or herd level, which can be very difficult to scale-up to model interventions at the regional or national level (Garner and Hamilton, 2011; Rushton, 2017). This is a problem for LCA models, too, which often set out to present the impact of health conditions at the national or regional level. Work to date modelling the

Table 3 A summary of quantitative results reported in literature on animal health issues and associated treatment/prevention strategies. Each reported outcome represents the potential mitigation of GHGs shown using the treatment/prevention strategies. Where a study addressed more than one disease, its results are reported separately in the table below.

Study	Study type	Species (System)	Disease/s studied	Health intervention/s	Reported GHGs outcome	Functional units and other methodological details	Other notes
(Houdijk et al., 2017)	Experimental trial	Sheep (Ewes)	Parasitic worm infection (*Teladorsagia circum- cincta*)	Anthelminthics	−14%	1 kg Live weight gain	Increases and decreases shown for different sources of GHG production.
(Kenyon et al., 2013)	Experimental trial	Sheep (Lambs)	Gastrointestinal nematodes	Anthelminthics (four strategies – 'monthly' (NST), 'strategic' (SPT), 'targeted' (TST) or based on 'clinical signs' (MT)	−9% for best treatment case compared to worst case	1 kg Live weight gain	All groups in trial treated with Anthelminthics just using different strategies.
(Li et al., 2015)	Experimental trial	Pigs	Porcine reproductive and respiratory syndrome virus (PRRSv)	Vaccination for PRRSv	+11% CO_2 +300% N_2O NH_3 CH_4 and H_2S all unchanged	Figures represent increase in emissions from manure storage when unvaccinated pigs infected with PRRSv	No overall GHG assessment.
(Skuce et al., 2016)	Modelling	Cattle (dairy)	Infectious bovine rhinotracheitis (IBR) disease		−1.5 to 3%	1 kg carcass weight	Reduction based on any one of control measures listed being implemented.

(Hospido and Sonesson, 2005)	Modelling	Cattle (dairy)	Mastitis	Scenarios for blanket and selective dry cow therapy	−2.5% GWP in 'improved herd' scenario	1 L of milk	Improved scenario was based on a reduction in discarded milk and culling. Higher reductions reported for other impact categories which included eutrophication, acidification and ozone depletion.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Bovine viral diarrhoea (BVD)	Vaccination, double fencing, identification of PI animal	−14% dairy and −50% beef	1 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Johne's disease	Vaccination, colostrum management and hygiene, buying policy, test and cull	−13% dairy and −21% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Salmonella	Vaccination, vector control, hygiene	−15% dairy and −13% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Liver flukes	Strategic treatment, grazing management	−7% dairy and −5% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	IBR	Vaccination, double fencing, identification of latently infected carrier animal	−6% dairy and −13% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.

(*Continued*)

Table 3 (*Continued*)

Study	Study type	Species (System)	Disease/s studied	Health intervention/s	Reported GHGs outcome	Functional units and other methodological details	Other notes
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Lameness	Change in cow 'hardiness', mobility management, cow comfort	−4% dairy only	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Mastitis	Milking routine, dry cow therapy, nutrition, housing and milking machine maintenance	−1% dairy only	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Infertility	Fixed time AI, tail paint/kamar and activity metres, nutrition	−10% dairy and −12% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Calf pneumonia	Vaccination, colostrum intake, buildings, stocking density and buying policy	−<1% dairy and −1% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.
(ADAS, 2015)	Modelling	Cattle (dairy and beef)	Calf diarrhoea	Housing and calf comfort, prophylactic therapy, vaccination and colostrum screening	−<1% dairy and −1% beef	1000 L FPCM milk and 1 tonne edible carcass weight	Values represent the impact of implementing all treatments in model compared to no treatment.

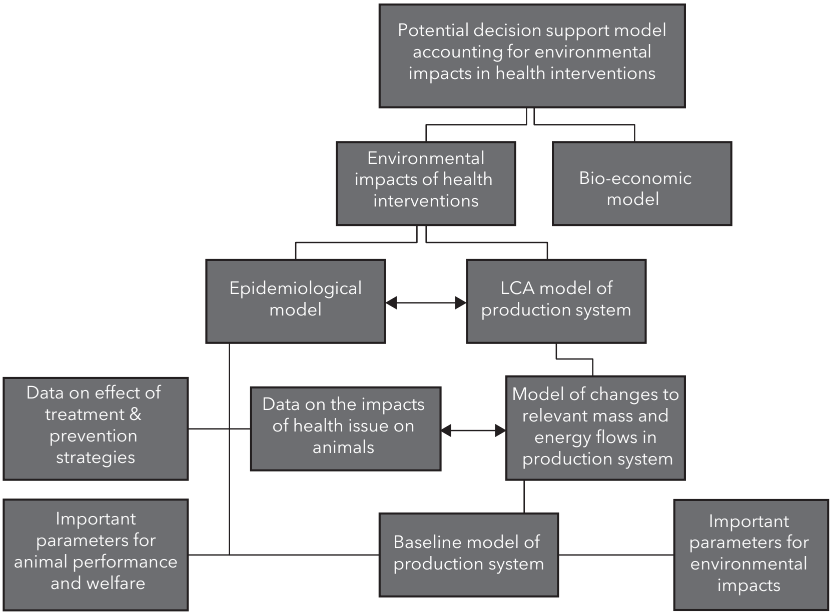

Figure 1 The important components for modelling the environmental impacts of health issues and health interventions in livestock systems through Life Cycle Assessment (LCA).

environmental impact of health interventions has acknowledged its limitations with regard to the robustness of the scenarios presented for animal health outcomes (e.g. ADAS, 2015). However, this may be an unfairly high standard to judge early work in this area, which can only work with the best data available to further our understanding of the potential for successful health interventions to reduce environmental impacts.

More challenging, when dealing with health issues, are the spatial and temporal scales that need to be taken into account to assess their effectiveness. Setting boundaries for the technical system being modelled is necessary in LCA. Commonly for LCA of livestock systems the model may be set up to represent a representative farming scenario within a region, without any precise definition of its location or spatial properties (McAuliffe et al., 2016). However, the effectiveness of many treatment and prevention strategies can only be considered at larger regional or national scales. In general, epidemic diseases affect multiple farms, and therefore have a strong spatial component. For example, it may be necessary to assume sufficient uptake of a preventative vaccine within a region to establish herd immunity without the need for extra controls on travelling animals, for example, in the case of bovine TB (Smith and Delahay, 2018). The same applies to endemic diseases whose transmission depends on a biological vector, such as trypanosomiasis. For this reason, several researchers have modelled the eradication of a disease as opposed to controlling a health challenge, as discussed above (Macleod et al., 2018; Skuce et al., 2016). This approach also comes with complexity, an example being the models of preventing IBR in cattle in both the ADAS (2015) report and later by Skuce et al. (2016). The latter study acknowledged that it may not be possible to achieve the eradication of the condition in the UK as has been shown to be possible elsewhere (Ackermann et al., 1990), without a full national programme of compulsory control including testing, culling vaccination and restriction of movement. As such, the implementation of control strategies on individual farms, such as vaccinations or double fencing, is predicted to have a reduced effect in disease reduction when introduced individually. In contrast, some treatment and prevention strategies can be pursued successfully in individual farms and can thus have 'isolated' consequences. For example, such is the case of hygiene measures: the washing of udders and general improvements to cleanliness of housing to reduce instances of mastitis (Lam et al., 2013). The challenges here are faced in all epidemiological modelling of transboundary diseases (FAO, 2016d): researchers need to justify why the scale at which they are modelling the treatment and prevention of health issues is appropriate and ensure that where possible they have considered that any issues they face with upscaling data from trials done at the farm level. They should also consider relevant interactions between different treatment strategies where possible.

Similarly, there are key temporal considerations in epidemiological modelling, with temporal scale being key to understand the effectiveness of prevention and treatment strategies (Onstad, 1992). For example, if one is interested in quantifying the environmental impact of a health challenge control over time, then the temporal effects on the epidemiology of the challenge will need to be taken into account. Currently, there is an increased interest in showing the reductions in environmental impact of a livestock sector achieved through management or breeding (Tallentire et al., 2018; Ottosen et al. 2020), and this exercise has analogies with what we are discussing here in relation to health challenge controls.

4.3 Identifying the impact of health issues and interventions on important mass and energy flows

Translating the outcomes of any epidemiological model of a health intervention into a model of the mass and energy flows for an animal production system is a significant challenge. One of the particular challenges here is that sensitive aspects of the LCA model may not be routinely recorded, even in experimental trials measuring the effectiveness of treatment and prevention strategies in reducing specific health conditions. For example, while aspects of animal performance such as feed intake, growth rate and milk or egg production would be routinely measured, the effect of health conditions on enteric methane production or the amount of volatile solids contained in manure would not be. While predictions of the environmental impact mitigation achieved from prevention and treatment strategies can be made just from traditional performance data, factors such as enteric methane and volatile solid excretion are extremely sensitive in models of environmental impacts in animal production (Opio et al., 2013). Most calculations relating GHGs and health issues in animal production use a fixed equation based on the energy and dry matter intake of animals to calculate enteric methane emissions. The methods either directly follow, or are derived from the tier 2 IPCC protocols for calculating methane emissions from livestock (IPCC, 2006). Several studies reference the fact that this is a sensitive assumption within their models which could be improved (Macleod et al., 2018; Mostert et al., 2018). Importantly, these emission factors and the predicted enteric methane emissions are not altered when considering the scenarios impacted by health conditions. This may actually not be true. Evidence of this can be seen from studies that have the relevant measurements in experiments (Fox et al., 2018; Houdijk et al., 2017). In the Houdijk et al. (2017) study, peri-parturient parasitism reduced feed intake by 9% and litter weight gain by 7% for rearing ewes, while also doubling maternal weight loss, and these were, at least partly, caused by parasite-induced anorexia. While net methane production was actually reduced, the methane yield per kg of digestible organic matter intake increased by 14%. Infection caused reduced feed intake depriving the animal of needed resources, reduced the efficiency with which feed was being used and increased the proportion of organic material being released as volatile organic compounds. The manure methane and nitrous oxide yields also increased 46% and 31% respectively per kg of digestible organic matter intake. These increased yields were a result of reduced nutrient absorption, which is a symptom of parasite infection (Coop and Kyriazakis, 1999; Shea-Donohue et al., 2017), and also contribute to the increased GHGs caused by untreated infection in this study.

These recorded increases in the yield of methane and nitrous oxide are important points for considering the environmental impact of health issues. Models that assume a constant relationship between feed intake and enteric methane emissions, as well as emissions from manure, may underestimate the increase in GHGs and other important emissions caused by health issues affecting the digestive system. To date these detailed experimental studies in relation to health issues and the production of GHGs have been conducted on sheep systems (Fox et al., 2018; Houdijk et al., 2017). However, similar findings could be reasonably predicted in dairy and beef systems for conditions involving parasitism and/or instances of diarrhoea. Increases in the methane and nitrous oxide yields per kg of digestible organic matter intake would increase the predicted increase in GHGs caused by such health issues produced by studies such as by ADAS (2015) on dairy and beef systems. Notably, Houdijk et al (2017) also found that release of respiratory CO_2 increased per kg intake of digestible organic matter by 17% and in the LCA results overall by 15%. CO_2 released this way is almost never taken into account in LCA models of livestock systems and certainly not modelled to increase in any of the other studies on health issues found in this review. These results suggest that the impact of some health issues on GHGs in livestock systems may be underestimated in many of the current LCA models produced. If possible, LCA models of health issues in livestock and potential mitigations should identify the relevant mechanisms within animals by which health issues may increase GHGs produced on farm from livestock systems, as well as accounting for expected changes to herd performance as is already typical.

Another important consideration in terms of mass and energy flows when modelling the environmental impact of health issues in livestock impact is any impact they may have on the quality of the output products. This is a different issue to some of the examples discussed earlier where milk may fail quality tests or for other reasons have to be discarded as a result of health issues for this is easily accounted for as waste within an LCA model. A relevant example in this case would be an alteration in carcass composition that reduced the quality of meat produced by sheep or beef cattle as a result of being challenged with gastrointestinal parasites (Holmes, 1993). Accounting for such issues is linked to a wider methodological challenge that researchers face when constructing LCA models of livestock systems that can account for the quality of the product produced. The key issue being the importance of the chosen functional unit in LCA modelling (Hospido et al., 2010).

The currently accepted convention, and standard way of representing functional unit in LCA of meat production systems is to use a variation on edible meat or carcass weight with no further properties describing the nutritional quality of the meat produced (e.g. ADAS, 2015; FAO, 2016c, 2018). Essentially its chemical composition and physical characteristics are usually assumed to be either unrelated to whatever treatments (changes to diet, health treatment, management changes) a livestock LCA has considered or it is acknowledged (not always explicitly) that the model was simply not able to take the relationship between these two things into account. This situation contrasts slightly with the standard functional unit used in dairy LCA of fat and protein-corrected milk (FPCM) (FAO, 2016a) using methods such as that set out by Bartl et al. (2011):

For conversion of kg milk to kg FPCM the yield was adjusted for energy to a standard milk with 4% fat, 3.2% protein and 4.8% lactose using the formula FPCM (kg/day) ¼ milk (kg/day) [0.038 fat (g/kg) þ 0.024 protein (g/kg) þ 0.017 lactose (g/kg)]/3.14 (Bartl et al., 2011).

A few studies of dairy systems build further on this to include either FPCM produced per land occupied, or to include the meat produced and sold from dairy systems in the functional unit, but the vast majority use a mass or volume measurement of FPCM (FAO, 2016a). More broadly in LCA of food systems there has been a trend towards developing nutritional functional units based on a set of nutritional properties for food products, which match guidelines for human dietary requirements (Saarinen et al., 2017). For LCA of livestock production systems McAuliffe et al. (2018) presented such a methodology based on using important nutritional properties for high-protein foods adapted from the methods of Saarinen et al. (2017) for Finnish food systems. To adopt such an approach requires confidently linking the consequences of production diseases for the chemical properties of livestock tissue. If such data is available for the health issues and treatments/prevention strategies being considered in LCA models, then it should be accounted for using an appropriate functional unit as discussed above, but to date this has not been the case for any studies of health issues in meat production systems.

4.4 Beyond health issues for ruminants and greenhouse gas emissions

Work to date on the environmental impacts of health issues in livestock has focussed almost exclusively on ruminant systems. We found only one study that specifically focussed on the environmental impact of health issues in non-ruminant systems. Li et al. (2015) assessed the implications of PRRSv infection on growth, dietary nutrient utilisation efficiency, manure output, as well as emissions of CO_2, CH_4, H_2S, N_2O and NH_3 gases from stored manure in both vaccinated and unvaccinated pigs. This study was able to show that in addition to the expected reduction in growth performance, exposure to PRRSv significantly increased CO_2 and N_2O released both directly from pigs and their manure when stored, with no observed impact on NH_3 or CH_4 emissions. Manure from infected pigs had a greater C:N ratio, and infection with PRRSv-increased manure output did not have an observed impact on the utilisation and retention of N in infected animals. These increased levels of organic matter excreted explained the increased CO_2 emissions from their manure. The increased C:N ratio was also observed to stimulate more fermentation and microbial activity during manure storage. Reaction routes as part of this increased microbial activity during storage were thought to cause the discrepancy between N_2O (increased) and NH_3 (no change) emissions from the manure of infected pigs.

Ultimately, the study was unable to demonstrate that prior vaccination reduced the negative impact of PRRSv on animal performance or GHG emissions (Li et al., 2015). While no overall GHG assessment was performed, the results of their work show that PRRSv caused an increase in the GHGs (increased CO_2 and N_2O) from pig production systems beyond the expected drops in animal performance. As with the case of parasite infection in sheep production discussed above, failure to take into account the link between infection and increase in on-farm GHG emissions may lead to researchers underestimating the impact of these conditions in LCA studies. Moreover, while their results were not fed into an LCA model, it was notable that neither NH_3 nor H_2S emissions from the pigs and during manure storage increased when animals were infected with PRRSv nor did infection have any detectable impact on nutrient utilisation. Nutrient utilisation efficiency and NH_3 emissions in particular, both in barn and during manure storage, are a major factor in determining the acidification and eutrophication caused by pig farming systems (Mackenzie et al., 2015; McAuliffe et al., 2016).

Most studies on the environmental impact of health issues (with exceptions, Chen et al., 2016; Hospido and Sonesson, 2005) did not consider other forms of environmental impact beyond GHGs, such as eutrophication, acidification and abiotic resource use. The focus on ruminant systems to date in this area is understandable as discussed above. However, when considering the environmental impacts of livestock production in broader terms beyond just GHGs, it becomes much more important to account for the role of non-ruminant production systems (de Vries and de Boer, 2010). Focussing on a single environmental impact does allow researchers to concentrate on translating the outputs of epidemiological models into fewer important mass flows. For example, LCA models which consider only GHGs will account solely for CH_4, N_2O, CO_2 and any hydrofluorocarbon emissions released during by the production system considered (Opio et al., 2013). Accounting for further impact categories such as acidification potential or water use, as recommended in most of the FAO technical guidelines on livestock LCA (FAO, 2016c, 2016b, 2018), would mean that researchers are more likely to capture potential trade-offs in the environmental impact benefits of health interventions. While each impact category added to the model increases uncertainty, as researchers have to establish the potential impact of health issues and treatment strategies on a greater number of physical flows in the system, it will be important to overcome these issues in future work in this area. As a case in point, the results from the Li et al. (2015) study are indicative that certain health conditions may have greater implications for some forms of environmental impact over others. Depending on the context that studies are carried out in, researchers may deem health issues that decrease nutrient utilisation efficiency, thus increasing NH_3 and other emissions are more problematic from an environmental impact perspective than those which do not in pig and poultry farming systems. For example, within the EU pig production represents around 25% of NH_3 emissions (Philippe et al., 2011) and EU has now defined standards which dictate limits for NH_3 emissions to air in housing as well as N and P excretions in manure (EU Science Hub, 2017). As such those looking to quantify which treatment and prevention strategies could yield the most important mitigation of environmental impacts in EU pig and poultry systems may wish to prioritise the reduction of nutrient excretion in their models. They would need to be able to identify the relevant mechanisms for health issues in pig and poultry systems in relation to nutrient utilisation and in barn emissions set out for PRRSv in the Li et al. (2015) study. The LCA models would also need to account for multiple impact categories in order to interpret these data into useful outputs for decision-making.

5 Conclusions

We reviewed important concepts for quantifying the relationship between health issues in livestock systems and their environmental impacts, as well as the studies published on this issue to date. In doing so we outlined a general framework and guidelines for constructing such models, highlighted a number of important methodological considerations for modelling in this area and outlined some significant gaps in current knowledge of this topic that should be addressed in future research. As this area of modelling requires the integration of epidemiological models with environmental impact models, there are unique temporal and spatial considerations to account for when modelling livestock health issues.

It is also important when integrating these two areas of modelling that the impact of health issues on sensitive mass flows within the animal and on farm, such as methane or ammonia emissions are taken into account. In most cases, researchers assume a constant relationship between the feed intake or energy intake of animals and on-farm emissions in their models of health issues and GHGs (Macleod et al., 2018; Mostert et al., 2018). However, while initial experimental work in this area to date suggests this may not always be the case, studies that have measured the relationships between on-farm emissions and health issues are sparse, presenting a challenge in developing effective models on this subject. Wherever possible they should be accounted for in models of the environmental impact of health issues, otherwise this will likely lead to inaccuracies in reporting the environmental impact of health issues in livestock. Likewise, no studies have thus far utilised relevant methods to account for the implications of health issues for product quality with respect to their environmental impacts. New methods being developed in this area for nutritional functional units in LCA studies of food systems (McAuliffe et al., 2018; Sonesson et al., 2017) should allow researchers to account for this in future where relevant data is available, for example, with respect to health challenges and product composition.

In most cases, the implications of health issues for environmental impacts beyond GHGs of livestock systems have not been considered or quantified to date. This contradicts several FAO guideline documents on how to quantify the environmental impact of livestock systems and represents a significant gap in our knowledge of the issue. Surprisingly few studies have considered the impact of health issues in pig and poultry production systems at all, with these sectors making up an ever-increasing proportion of livestock products consumed around the world, and this is an area that will warrant future exploration by researchers.

Ultimately, in order to actually be integrated into decision-making processes, environmental impact assessment models of health interventions will need to integrate economic considerations to produce recommendations for the most cost-effective mitigation measures. The ADAS (2015) report demonstrated the possibilities for this while modelling at a national scale. Work at this level could be used to justify funding or subsidies for particular health interventions at the national level on the combination of animal welfare and environmental grounds. However, there may also be scope for impactful work in this area to be done through farm level modelling. This could integrate health interventions and their potential benefits for environmental impacts into decision support tools as is being currently being developed in other areas of livestock husbandry such as feeding (Garcia-Launay et al., 2018; Mackenzie et al., 2016), breeding (Ottosen et al., 2020; Tallentire et al., 2018) and environmental management (Pexas et al., 2020; Ten Hoeve et al., 2014).

6 Where to look for further information

- The Oxford University Research Archive (https://doi.org/10.5287/bodleian: 0z9MYbMyZ) stores a streamlined global database of GHGs associated with food and beverage production systems.
- For interesting discussion on the challenges of economics in animal health modelling – Rushton, J. 2017a. Improving the use of economics in animal health – Challenges in research, policy and education. Prev. Vet. Med. 137, 130–139. https://doi.org/10.1016/J.PREVETMED.2016.11.020.
- The latest FAO technical guidelines published on developing LCA models of livestock systems can be found here – http://www.fao.org/partnerships/leap/publications/guidelines/en/.
- For an overview of basic principles and important methodological issues in LCA modelling – Curran, M. A. 2012. Life cycle assessment handbook: a guide for environmentally sustainable products. Scrivener Publishing LLC, Beverley, MA.
- ADAS, 2015. Study to Model the Impact of Controlling Endemic Cattle Diseases and Conditions on National Cattle Productivity, Agricultural Performance and Greenhouse Gas Emissions. Final Report to Defra/AHVLA on Project FFG1016. 210.

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Chapter 5

Sustainable nitrogen management for housed livestock, manure storage and manure processing

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1 Introduction

Nitrogen can take various forms (Fig. 1), including atmospheric di-nitrogen (N_2) and a wide range of reactive nitrogen (N_r) compounds, including all forms of nitrogen that are biologically, photochemically and radiatively active. Compounds of nitrogen that are reactive include ammonia (NH_3) and ammonium (NH_4^+), nitrous oxide (N_2O), nitrogen oxides (NO_x), nitrite (NO_2^-), nitrate (NO_3^-), nitric acid (HNO^-_3) and a wide range of organic nitrogen compounds (R-NH_2). Reactive forms of nitrogen are capable of cascading through the environment and causing an impact through smog, acid rain, biodiversity loss and so on as well as affecting climate (Butterbach-Bahl et al., 2011b). The design of abatement/mitigation measures requires a sound knowledge of the processes that influence formation and emission of all N_r compounds and N_2 into the environment, where nitrogen is lost to a wide range of atmospheric and aquatic pathways.

1.1 Ammonia

The principles of ammonia formation and its influencing factors are well known. Degradation of nitrogen-containing organic substance results in

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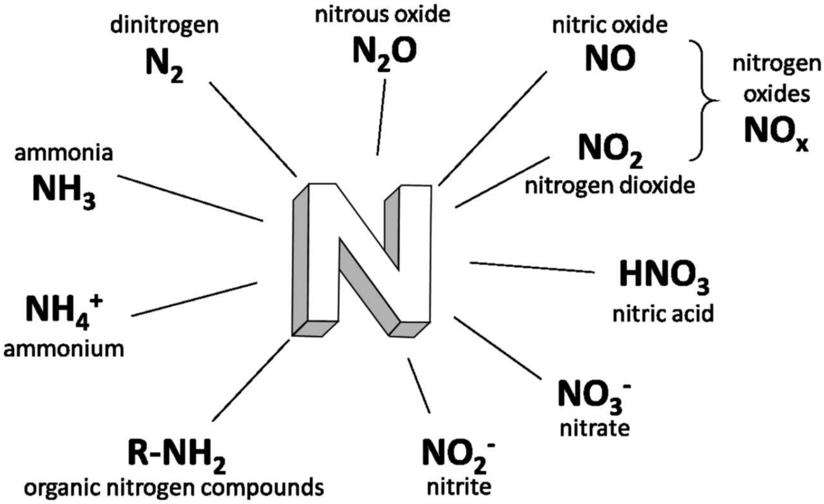

Figure 1 Major forms of nitrogen occurring in the environment. The sum of all forms except N_2 is often termed fixed or reactive nitrogen (Nr).

ammonium formation (NH4⁺). There is an equilibrium between ammonium and ammonia:

$$H_2O + NH_3 \rightleftharpoons OH^- + NH_4^+$$

The degree to which ammonia forms the ammonium ion depends on the pH of the solution. If the pH is low, the equilibrium shifts to the right: more ammonia molecules are converted into ammonium ions. If the pH is high, the equilibrium shifts to the left: the hydroxide ion abstracts a proton from the ammonium ion, generating ammonia.

Ammonia emissions are governed by the difference between solution and atmosphere NH_3 partial pressure. High NH_3 concentrations in the solution and low NH_3 concentrations in the surrounding atmosphere increase NH_3 emissions. According to Henry's Law, ammonia emissions are also temperature dependent with rising temperatures increasing emissions (Fig. 2). Denmead et al. (1982) give the following equation:

$$NH_{3(\text{solution})} = \left(NH_{3(\text{solution})} + NH_{4(\text{solution})}\right) \Big/ \left(1 + 10^{0.09018 + (2729.92/T) - \text{pH}}\right)$$

where

$NH_{3(solution)}$ = NH_3 concentration in the solution
$NH_{3(solution)} + NH_{4\,(solution)}$ = The sum NH_3 and NH_4^+ in the solution

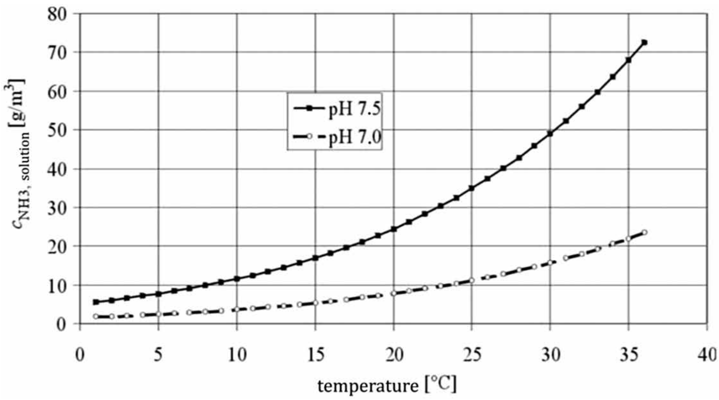

Figure 2 NH_3 concentration in the solution depending on temperature for pH 7.0 and pH 7.5 (Denmead at al., 1982).

T = Temperature in the solution [K]
pH = pH value in the solution

Ammonia emissions associated with livestock housing, manure storage, management and processing result from the degradation of urea by the ubiquitous enzyme urease which results in $NH4^+$ formation. Urea is mainly excreted in the urine and once it is hydrolysed it is much more prone to ammonia losses than organic nitrogen excreted in the faeces. In the case of poultry nitrogen is excreted largely in the form of uric acid, which hydrolyses like urea to produce ammonia. Where it is possible to dry excreta (e.g. in poultry litter), strategies may focus on reducing the hydrolysis rate of uric acid and urea. Once ammoniacal nitrogen (the sum of NH_3 + NH_4^+) is formed, strategies in animal housing and manure management focus on avoiding its volatilization to the atmosphere, for example, by reducing access to air, by reducing pH or by keeping the manure surface cool (cf. Fig. 2).

1.2 Nitrous oxide and di-nitrogen

The gases N_2O, NO_x and N_2 are formed both during the nitrification and the denitrification processes in the environment. The 'Leakage' model developed by Firestone and Davidson (1989) shows N_2O and NO_x losses as leakage flows during nitrification and denitrification (Fig. 3).

Nitrification oxidizes ammonia via nitrite to nitrate. This process is strictly aerobic. Autotrophic-nitrifying bacteria belong to the widespread group of nitrosomonas, nitrospira and nitrobacter, which are capable of growing on CO_2, O_2 and NH_4^+. Availability of NH_4^+ is mostly the limiting factor as CO_2 and O_2

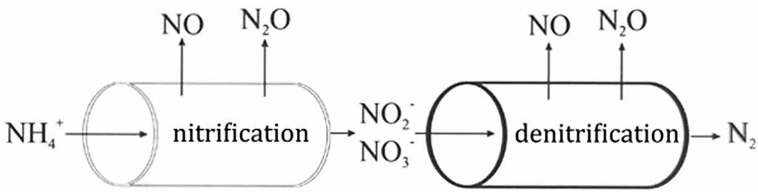

Figure 3 Leakage model for N_2O and NO_x losses during nitrification and denitrification (Firestone and Davidson, 1989).

are available in abundance. Low pH, lack of P and temperatures below 5°C or above 40°C lead to a reduction in nitrification activities. A water content of 60% of soil water-holding capacity is optimal for the nitrification process.

At low pH values, nitrification is carried out by bacteria and fungi. In contrast to the autotrophic nitrifiers, they need carbon sources for their growth. Their turnover rate is much lower compared to the autotrophic nitrifiers, but a substantial total turnover can still be achieved as a wider range of species have the ability for heterotrophic nitrification. N_2O production during nitrification is around 1%, NO_x production ranges between 1% and 4% of inputs N (Butterbach-Bahl et al., 2011a)

Denitrification reduces nitrate to N_2O, NO or N_2 when oxygen availability is low. NO_3^-, NO and N_2O serve as alternative electron acceptors when O_2 is lacking, and hence the denitrification occurs only under strictly anaerobic conditions. Molecular N_2 is the last part of the denitrification reaction chain and it is the only biological process that can turn reactive nitrogen into molecular N_2. Denitrifying bacteria are heterotrophic and facultative anaerobic. This means that they use O_2 as electron acceptor and switch to alternative electron acceptors (NO_3^-, NO and N_2O) when oxygen availability is low. Denitrifying bacteria are wide spread and show a high biodiversity.

Controlling factors for denitrification have been extensively investigated, mainly under lab conditions. Complex interactions exist between the various influencing factors which make a prediction of N_2O emissions difficult under practical conditions.

Denitrification is mainly governed by oxygen availability. Denitrification starts when the O_2 concentration decreases to below 5% (e.g. Hutchinson and Davidson, 1993). This may be the case in poorly aerated soils (e.g. high water content, in excess of 80% water-filled pore space), but also in soils where a high biological turnover consumes the oxygen faster than the supply. Easily degradable carbon sources and high nitrate concentrations also enhance the denitrification rate, while low temperature and low pH limit denitrification activity.

The relationship between N_2 and N_2O formation is mainly governed by the relationship between electron acceptor and reducing agent, and by the

O_2 concentration in the substrate. N_2 is only formed under strictly anaerobic conditions and a wide C: NO_3^- ratio. High nitrate concentrations increase the rate of N_2O production. These differences have effects in practice concerning N losses from housed livestock and manure storage, according to the extent of oxygen and carbon availability in different systems.

1.3 Nitrate and other nitrogen leaching and run-off

Diffuse pollution of groundwater and surface waters with N (and P) is a problem in many regions of the world, especially in areas with high livestock production. Animal manures contain substantial quantities of organic matter, N and P that if managed inappropriately may be lost from animal housing, manure storages or after field application.

Nitrogen and organic matter losses to aquatic systems mainly occur by leaching through the soil profile and through surface run-off when the infiltration capacity of the soil is exceeded. Point-source emissions can also be acutely damaging to local environments, for example, in the case of slurry store leakages. In surface waters, the losses cause problems with eutrophication and algal bloom, and in areas that rely on the use of groundwater, high nitrate concentrations can be a problem for the potable water quality. For drinking water the EU limit has been set at a nitrate (NO_3^-) concentration of 50 mg L^{-1} (EU Drinking Water Directive, 98/83/EC). Once leached to surface waters this N may also become a source of emissions of nitrous oxide, which is a potent greenhouse gas. In addition, significant loss of N resources is also an economic cost for the farmer, and N fertilizer production uses substantial amounts of fossil energy, causing global warming and other environmental emissions. Appropriate management and use of manures is therefore essential for minimizing nutrient leaching and the environmental impact of agriculture.

1.4 Consideration of nitrogen flows

Measures to reduce nitrogen losses from livestock feeding, housing and manure processing need to be seen in relation to other measures described in this guidance document. 'Manure management is a continuum from generation by livestock to storage and treatment and finally to land spreading' (Chadwick et al., 2011). This means that there is the potential for nitrogen, carbon and phosphorus losses at each stage of this continuum. A 'mass flow' approach has been used by Webb and Misselbrook (2004) to estimate NH_3 emissions from the manure management continuum. This approach allows effects of measures to reduce emissions and conserve manure N at one state to be considered as the manure passes to the next stage in the continuum. Similarly, other gaseous N losses, including N_2O, NO_x and N_2, may be assessed using a mass flow approach in a manner similar to that of Dämmgen and Hutchings (2008).

The importance of such a *whole system approach* is that effects of abatement methods at one stage are considered in downstream stages (Sommer et al., 2009, 2013), including losses of nitrogen to water through leaching and run-off.

1.5 Approach used to describe abatement measures

The following sections present the main management practices and abatement/mitigation measures that will influence N utilization and losses from housed livestock, manure storage, manure treatment and manure processing. Some measures will mitigate all forms of N loss, whereas others may mitigate a specific N loss pathway with either little impact or a negative impact on other N loss pathways. Enhanced abatement may be possible through the combined implementation of certain packages of measures. Linked management of nutrient cycles is required for effective environmental protection (Sutton et al., 2013)

The sections are based on information given in 'Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen' (Bittman et al., 2014), 'Best Available Techniques (BAT) Reference Document for the Intensive Rearing of Poultry or Pigs' (Santonja et al., 2017), 'Code for Good Agricultural Practice for Reducing Ammonia Emissions' (UNECE, 2015) and 'Framework Code for Good Agricultural Practice for Reducing Ammonia Emission' (Economic Commission for Europe, 2015). The information shall also be published as part of a Guidance Document on Integrated Sustainable Nitrogen Management developed by members of the Task Force on Reactive Nitrogen and submitted for adoption as UNECE document by the Executive Body of the UNECE.

2 Livestock feeding and housing

2.1 Livestock feeding

Ammonia emissions result from the degradation of urea by the ubiquitary enzyme urease which results in NH_4^+ formation. Urea is mainly excreted in the urine and once it is hydrolysed it is much more prone to ammonia losses than organic nitrogen excreted in the faeces. The crude protein content and composition of the animal diet is the main driver of urine excretion. Excess crude protein (CP) that is not needed by the animal is excreted and can easily be lost in the manure management chain. Adaptation of crude protein in the diet to the animals' needs is therefore the first and most efficient measure to mitigate nitrogen emissions. This measure reduces the loss of all N forms (Fig. 1) because it reduces the amount of excreted nitrogen. Groenestein et al. (2019) compared ammonia emissions related to nitrogen-use efficiency of livestock production in Europe. They come to the conclusion that expressing ammonia losses as a fraction of feed N would be preferable in expressing it per animal place as it give a clearer picture of the actual fraction of N in the feed that is lost to the environment.

Reduction of CP in animal feed is one of the most cost-effective ways of reducing N emissions throughout the entire manure management chain. For each per cent (absolute value) decrease in protein content of the animal feed, NH_3 emissions from animal housing, manure storage and the application of animal manure to land are decreased by 5–15% depending also on the pH of the urine and dung. Low-protein animal feeding also decreases N_2O emissions, and increases the efficiency of N use in animal production. Potential trade-offs with CH_4 emissions from enteric fermentation are not yet fully researched and need to be assessed. However, efficient N use is crucial for environmental-friendly milk production. Moreover, there are no animal health and animal welfare implications as long as the requirements for all amino acids are met. As there is much natural variation in nitrogen use efficiency (NUE) between individual animals, targeted breeding for better NUE can also be an option.

Low-protein animal feeding is most applicable to housed animals. It is less applicable for grassland-based systems with grazing animals, because grass is eaten by the animals in an early physiological growth stage and thus is typically high in degradable protein. It should be noted that grassland with leguminous species (e.g. clover and lucerne) also have a relatively high protein content, and so may be associated with provision of excess dietary N for livestock. Strategies to lower the protein content in herbage include: balanced N fertilization, grazing/harvesting the grassland at later physiological growth stage, and so on, and alteration of the ration of grassland-based systems, such as use of supplementary feeding with low-protein feeds.

2.1.1 Feeding strategies for dairy and beef cattle

Adapt protein intake in diet: Lowering CP of ruminant diets is an effective strategy for decreasing NH_3 loss. The following guidelines hold:

1 The average CP content of diets for dairy cattle should not exceed 15–16% in the dry matter (DM) (Broderick, 2003; Swensson, 2003). For beef cattle older than 6 months this could be further reduced to 12%;
2 Phase feeding can be applied in such a way that the CP content of dairy diets is gradually decreased from 16% of DM just before parturition and in early lactation to below 14% in late lactation and the main part of the dry period; and
3 Phase feeding can also be applied in beef cattle in such a way that the CP content of the diets is gradually decreased from 16% to 12% over time.

More information and associated costs can be found in the TFRN costs assessment (Chapter 3.4 'Low nitrogen feeding strategies in dairy cattle' in Reis et al., 2015).

In general, increasing the energy/protein ratio in the diet by using 'older' grass (higher sward surface height) or swathed forage cereal and/or supplementing grass by high energy feeds (e.g. maize silage) is a well-proven strategy to reduce levels of crude protein. However, for grassland-based ruminant production systems, the feasibility of these strategies may be limited, as older grass may reduce feeding quality, especially when conditions for growing high-energy feeds are poor (e.g. warm climates), and therefore such feeds have to be purchased. Hence, full use of the grass production would no longer be guaranteed. In the absence of other measures, such a strategy may also risk increasing methane emissions.

In many parts of the world, cattle production is grassland-based or partly grassland-based. In such systems, protein-rich grass and grass products form a significant proportion of the diet, and the target values for CP may be difficult to achieve, given the high CP content of grass from managed grasslands. The CP content of fresh grass in the grazing stage (2000–2500 kg DM/ha) is often in the range of 18–20% (or even higher, especially when legumes are present), whereas the CP content of grass silage is often between 16% and 18% and the CP content of hay is between 12% and 15% (e.g. Whitehead, 2000). In contrast, the CP content of maize silage is only in the range of 7–8%. Hence, grass-based diets often contain a surplus of protein and the magnitude of the resulting high N excretion strongly depends on the proportions of grass, grass silage and hay in the ration and the protein content of these feeds. The protein surplus and the resulting N excretion and N losses will be highest for grass-only summer rations (or grass-legume rations) with grazing of young, intensively fertilized grass or grass legume mixtures.

Urine excreted by grazing animals typically infiltrates into the soil. This means that NH_3 emissions per animal are reduced by extending the periods in which animal graze compared with the time spent with animals housed, where the excreta is collected, stored and applied to land. It should be noted that grazing of animals may increase other forms of N emissions (e.g. nitrate-N leaching and N_2O emissions). However, given the clear and well-quantified effect on NH_3 emissions, increasing the period that animals are grazing all day can be considered as a strategy to reduce emissions.

Increase productivity: Overall, increasing the productivity of dairy cattle in terms of milk or meat can decrease emissions per unit of animal production. Optimized productivity will also result in a reduction of enteric methane emissions. However, optimum productivity levels vary according to breed and region and must also consider that ruminants can only cope with a certain amount of concentrates and require sufficient roughage in their diet to stay healthy.

Increase longevity: Productivity can be increased though increasing milk production per year and through increasing the amount of milk production cycles per animal. Optimized diet and housing conditions enable a higher longevity of dairy cattle. Improving the longevity of dairy cattle also decreases the number of young cattle necessary for replacement. Reducing endemic disease and genetic gain through targeted breeding can also offer value.

2.1.2 Feeding strategies for pigs

Adapt protein intake in diet: Feeding measures in pig production include phase feeding, formulating diets based on digestible/available nutrients, using low-protein amino acid-supplemented diets and feed additives/supplements. Further techniques are currently being investigated (e.g. different feeds for males (boars and castrated males) and females) and might be additionally available in the future.

The crude protein (CP) content of the pig ration can be reduced if the amino acid supply is optimized through the addition of synthetic amino acids (e.g. lysine, methionine, threonine, tryptophan, typically limiting amino acids, which are too low in normal grain rations) or special feed components, using the best available information on 'ideal protein' combined with dietary supplementation. Lassaletta et al. (2019) performed a global analysis for pig systems that included the simulation of changes in CP. More information and associated costs can be found in the TFRN Costs Assessment (Chapter 3.2 'Low nitrogen feeding strategies in pigs', in Reis et al., 2015).

A CP reduction of 2–3% in the feed can be achieved, depending on the pig production category and the current starting point (Cahn et al., 1998). It has been shown that a decrease of 1% CP in the diet of finishing pigs results in a 10% lower total ammoniacal nitrogen (TAN) content of the pig slurry and 10% lower NH_3 emissions (Cahn et al., 1998). The inclusion of processed household and industry residues or wastes in the feed rations with a controlled energy/protein ratio is a complementary measure that reduces dependence on imported feedstuff. This measure also represents a reduction of upstream N_r emissions associated with feed production and downstream emissions associated with waste management (Lassaletta et al., 2019; zu Ermgassen et al., 2016).

2.1.3 Feeding strategies for poultry

Adapt protein intake in diet: For poultry, the potential for reducing N excretion through feeding measures is more limited than for pigs because the conversion efficiency currently achieved on average is already high and the variability within a flock of birds is greater. A CP reduction of 1–2% may be achieved depending on the species and the current starting point, but is already a well-proven measure for growers and finishers. Further applied nutrition research is currently being carried out in EU member states and North America and this may support further possible reductions in the future.

2.2 Livestock housing

When using measures to abate emission from livestock houses, it is important to minimize loss of the conserved nitrogen during downstream handling of the manure, in storage and spreading to maximize the benefit from the cost of abatement.

2.2.1 Cattle housing

Housing systems for cattle vary across the ECE region. While loose housing is most common, dairy cattle are still kept in tied stalls in some countries. In loose housing systems all or part of the excreta is collected in the form of slurry. In systems where solid manure is produced (such as straw-based systems), it may be removed from the house daily or it may remain there for up to the whole season, such as in deep litter stables. The system most commonly researched is the 'cubicle house' for dairy cows, where substantial NH_3 emissions arise from fouled slatted and/or solid floors and from manure in pits and channels beneath the slats/floor. There has been much less research to measure NO_x, N_2O and N_2 emissions from cattle housing, so recommendations in some cases have to be based on general principles, and are therefore subject to larger uncertainty than for NH_3 emissions from such systems.

Housed-cattle systems are generally set on stone or concrete bases, so direct nitrate leaching is not expected, unless there are cracked bases associated with poor maintenance. Run-off of N_r compounds from cattle housing systems may occur if ponded excreta is not correctly drained into storage tanks (e.g. associated with flooding events).

While 'hard standings' (typically concrete areas adjacent to dairies) provide a significant source of ammonia emissions outside of animal houses, in some parts of the UNECE region, cattle are kept in confined areas outside (e.g. feed lots), where N_r leaching, run-off and gaseous N losses may be substantial.

Animal welfare considerations tend to lead to an increase of soiled walking area per animal, increased ventilation and an overall increase in emissions. Changes in building design to meet the new animal welfare regulations in some countries (e.g. changing from tied stall to cubicle housing) will therefore increase NH_3 emissions unless abatement measures are introduced at the same time to combat this increase.

Solid versus slurry manure systems: Straw-based systems producing solid manure for cattle are not likely to emit less NH_3 in the animal houses than slurry-based systems. Further, N_2O, NO_x and N_2 losses due to (de)nitrification tend to be larger in litter-based systems than slurry-based systems (Webb et al., 2012; Groenestein and van Faassen, 1996).

While straw-based solid manure can emit less NH_3 than slurry after surface spreading on fields (e.g. Powell et al., 2008), slurry provides a greater opportunity for reduced emission application methods. Verification of any NH_3 emission reductions from using solid-manure versus slurry-based systems and from solid-liquid separation should consider all the stages of emission (housing, storage and land application).

2.2.2 Mitigation measures for cattle housing

Mitigation options for cattle housing can be grouped into the following types:

- Floor-based systems and related management techniques (including scrapers and cleaning robots);
- Litter-based systems (use of alternative organic material);
- Slurry management techniques at pit level;
- Indoor climate control techniques; and
- End-of-pipe techniques (hybrid ventilation + air cleaning techniques) and GHGs abatement/mitigation techniques.

Several pathways can be identified to further optimize existing and develop new abatement techniques. In this respect emission reduction techniques at animal housing level should aim to affect one or more of the following important key factors and/or driving forces of the nitrogen emission process:

- Draining capacity of the floor for direct transportation of urine to the manure storage;
- Residence time of open urine/manure sources;
- Emitting surface area of open urine/manure sources;
- Urease activity in urine puddles;
- Urine/manure pH and temperature (see Housing Measures 6 and 8);
- Indoor air temperature;
- Air velocities at emitting surfaces (urine puddles and manure surface in the pit);
- Air exchange between pit headspace and indoor air; and
- Exhaust of indoor air.

Immediate segregation of urine and faeces: A physical segregation (i.e. keeping separately) of faeces, which contain *urease*, and urine in the housing system reduces hydrolysis of urea, resulting in reduced emissions from both housing and manure spreading (Burton, 2007; Fangueiro et al., 2008a,b; Møller et al., 2007). Both acidification and alkalization of the in-house segregated urine reliably inhibits urea hydrolysis. The duration of the inactivation period can be adjusted by the dosage of acid or alkali addition

Verification of any NH_3 emission reductions from using solid-manure versus slurry-based systems and from solid-liquid separation should consider all the stages of emission (housing, storage and land application). Additional advantages of solid-liquid separation can also be expected during land-application, where urine (containing most of the available ammoniacal N) infiltrates more easily due to its lower dry-matter content than slurry, reducing NH_3 emissions. Although solid manure does not infiltrate, it mainly consists of organic N forms, which are much less liable to NH_3 emissions. Less is known about the consequences of solid-liquid separation on the emissions of N_2O, NO_x, N_2 and nitrate leaching, although substantial adverse effects are not expected.

Regular cleaning of floors in cattle houses by 'toothed scrapers': The 'grooved floor' system for dairy and beef cattle housing, employing 'toothed' scrapers running over a grooved floor, is a reliable technique to abate NH_3 emissions. Grooves should be equipped with perforations to allow drainage of urine. This results in a cleaner, low-emission floor surface with good traction for cattle to prevent slipping. Ammonia emission reduction ranges from 25% to 46% relative to the reference system (Smits, 1998; Swierstra et al., 2001). In the absence of measurement data, it is expected that use of the grooved floor system would have little impact on other N_r and N_2 losses since it is mainly directed to reduce immediate exposure to air of ammonium-rich excreta.

Regular cleaning of floors in cattle houses: Thorough cleaning of walking areas in dairy cattle houses by mechanical scrapers or robots has the potential to substantially reduce NH_3 emissions. The automatic cleaning should be performed at regular intervals (e.g. on an hourly basis) to achieve full benefits of the measure.

Frequent slurry removal: Regular removal of liquid manure from under the slats in the house to an outside store can substantially reduce NH_3 emissions by reducing the emitting surface and the slurry storage temperature. A reduced storage temperature will also result in a reduction of methane emissions.

Increase bedding material: Bedding material in animal housing can affect NH_3, N_2O, NO_x and N_2 emissions. The physical characteristics (urine absorbance capacity, bulk density) of bedding materials are of more importance than their chemical characteristics (pH, cation exchange capacity, carbon to nitrogen ratio) in determining NH_3 emissions from dairy barn floors and pig houses (Misselbrook and Powell, 2005; Powell et al., 2008; Gilhespy et al., 2009; Groenestein et al., 2007). However, further assessment is needed on the effect of bedding on emissions for specific systems while taking into account the whole manure management path. The approach can have a positive interaction with animal welfare measures.

Barn climatization to reduce indoor temperature and air flow: In houses with traditional slats (either non-sloping, 1% sloping, or grooved), optimal barn climatization with roof insulation (RI) and/or automatically controlled natural ventilation (ACNV) can achieve a moderate emission reduction (20%) of NH_3 due to the *decreased temperature* (especially in summer) and *reduced air velocities* (Braam et al., 1997a,b; Smits, 1998; Monteny, 2000). To the extent that such systems cool stored manure, emissions of methane will also be reduced.

Use of acid air scrubbers: Chemical or acid air scrubbers are effective in decreasing NH_3 emissions from force-ventilated pig housing. However, they cannot yet be generally implemented in cattle housing because these are mostly naturally ventilated across the ECE region. Also, there are few data for scrubbers on cattle (Ellen et al., 2008). In any situations where cattle are housed with forced-ventilation, this measure can be considered as Category 1. Recent developments consider combining targeted ventilation of naturally ventilated barns with air scrubbers. More research and development are needed here.

Floor type: Different improved floor types based on slats or solid, profiled concrete elements have been tested. These designs combine emission reduction from the floor (increased run-off of urine) and from the pit (reduction of air exchange by rubber flaps in the floor slots). The emission abatement efficiency depends on the specific technical characteristics of the system.

Increased grazing: Decreasing the amount of animal excrement in animal housing systems through increased grazing is an effective measure to decrease NH_3 emissions. Total annual emissions (including housing, storage and spreading) from dairy systems may decrease by up to 50% with nearly all-day grazing, as compared with animals that are fully confined. While increased grazing is a reliable NH_3 emission reduction measure for dairy cows, the amount of emission reduction depends on the daily grazing time and the cleanliness of the house and holding area. In some cases grazing may also contribute to increased run-off and leaching of NO_3^- and other N_r compounds, as well as N_2O and NO_x emissions. Grazing can also be associated with increased pathogen mobilization.

2.2.3 Pig housing

Designs to reduce NH_3 emissions from pig housing systems have been described in detail in Economic Commission for Europe (2015) and in the IPPC document on Best Available Techniques (BAT) (Santonja et al., 2017). These apply the following main elements:

- Reducing manure surfaces such as soiled floors, using channels for slurry holding surfaces and using sloped walls. Partly slatted floors (~50% area), generally emit less NH_3, particularly if the slats are metal- or plastic-coated rather than concrete, allowing the manure to fall rapidly and completely into the pit below. Emissions from the non-slatted areas are reduced by inclined, smooth surfaces, by locating the feeding and watering facilities to minimize fouling of these areas, and by good climate control in the building;
- Removing the slurry from the pit frequently to an external slurry store with vacuum or gravity removal systems or by flushing systems at least twice a week;
- Additional treatment, such as liquid/solid separation; provided that the storage of the separated fractions maintains low emissions;
- Circulating groundwater or other cooling agents in floating heat exchangers or walls of slurry pits to cool the surface of the manure in the under-floor pit to at least below 12°C. Constraints include costs and need to locate a source of groundwater away from the source of drinking water;
- Changing the chemical/physical properties of the manure, such as decreasing pH;
- Using surfaces which are smooth and easy to clean (see above);
- Treatment of exhaust air by acid scrubbers or biotrickling filters;
- Lowering the indoor temperature and ventilation rate, taking into account animal welfare and production considerations; and
- Reducing airflow over the manure surface.

For a given floor slat width, manure drains from concrete slats less efficiently than from steel- and plastic-covered slats and this is associated with greater emissions of NH_3. Note that steel slats are not allowed in some countries for animal welfare reasons. Cross-media effects have been taken into account in defining BAT for the various housing designs. For example, frequent flushing of slurry (normally once in the morning and once in the evening) causes nuisance odour events. Flushing slurry also consumes energy unless manually operated passive systems are used.

Use of straw litter in pig housing is expected to increase due to concern for the welfare of the pigs. In conjunction with (automatically controlled) naturally ventilated housing systems, straw allows the animals to self-regulate their temperature with less ventilation and heating, reducing energy consumption. In systems with litter, the pen is sometimes divided into solid areas with litter and slatted dunging areas. However, pigs do not always use these areas in the desired way, using the littered area to dung and the slatted area to cool off in warm weather. Generally, pens should be designed to accommodate desired excreting behaviour of pigs to minimize fouling of solid floors. However, this is more difficult in regions with a warm climate. Note that integrated evaluation of straw use should consider (a) the added cost of the straw and mucking out the pens, (b) the possible increased emissions from storage and application of manure with straw and (c) the benefit of adding organic matter from straw to the soil.

2.2.4 Mitigation measures for pig housing

The reference system, used commonly in Europe, is a fully slatted floor with a deep manure pit underneath and mechanical ventilation; emission ranges from 2.4 kg NH_3 to 3.2 kg NH_3 per finisher pig place per year. Since growers/finishers are always housed in a group, most systems used for group housing of sows are applicable to growers. Emissions from different abatement/mitigation approaches are compared with this reference system in terms of the emission reduction amount (Bittman et al., 2014). Most data are available for NH_3, with little data concerning effects on N_2O, NO_x, N_2 and nitrate leaching. The underlying principles are largely similar for these losses as for cattle housing systems, recognizing the different housing needs of pigs and the particular characteristics of pig excreta.

Acidification of slurry: Reductions in NH_3 emissions can be achieved by acidifying slurry to shift the chemical balance from the gaseous NH_3 to the ionic and soluble NH_4^+. The manure (especially the liquid fraction) is collected into a tank with acidified liquid (usually using sulphuric acid, but organic acids can be used as well, though at higher cost) maintaining a pH of less than 6 (Bittman et al., 2014; Fangueiro et al., 2015). In pig housing systems, emission reduction of 60% or more have been observed (Kai et al., 2008). The measure is not anticipated to affect other N_r or N_2 losses. Acidification of slurry is anticipated to be effective for both cattle and pig slurry, though measurements have so far concentrated on investigating pig slurry. One study (Petersen et al., 2012) showed acidification of cattle slurry to pH 5.5 reduced the NH_3 emissions by more than 90% and at the same time reduced emissions of the GHG CH_4 by 67–87%. Attention should be given to monitoring soil pH and metal content if acidified slurry is to be used in agriculture. In-house acidification will reduce NH_3 emissions throughout the manure management chain. Furthermore, slurry acidified with sulphuric acid is not suitable as the sole feedstock for biogas production (but can be used as a smaller proportion).

Reduce emitting surface: Ammonia emissions can be reduced by 25% by decreasing the surface area of the emitting floor through frequent and complete vacuum-assisted drainage of slurry from the floor of the pit. Where this is possible, this technique has no cost. Partly slatted floors covering 50% of floor area generally emit 15–20% less NH_3, particularly if the slats are metal or plastic-coated which is less sticky for manure than concrete. Decreasing the risk of emissions from the solid part of the floor can be achieved by: (a) using an inclined (or convex), smoothly finished surface, (b) by appropriate siting of the feeding and watering facilities to minimize fouling of the solid areas and (c) by good climate control (Aarnink et al., 1996; Guingand and Courboulay, 2007; Ye et al., 2008a,b). Further reduction of the emitting area can be achieved by making both the partly slatted area and the pit underneath smaller. With the smaller slatted area, the risk of greater fouling of the solid area can be mitigated by installing a small second slatted area with a water canal underneath at the other side of the pen where the pigs tend to eat and drink. The canal is filled with about 2 cm of water to dilute any manure that might eventually drop into it. This slatted area will have low emissions because any manure dropped here will be diluted. This combined manure-canal and water-canal system can reduce NH_3 emissions by 40–50% depending on the size of the water canal. This approach is not expected to have a significant effect on emissions of N_2 or other N_r compounds.

Reducing the emitting surface area by having one or two slanted pit walls, in combination with partly slatted floors and frequent manure removal, can reduce emissions by up to 65%. Reducing the emitting surface area with shallow V-shaped gutters (maximum 60 cm wide, 20 cm deep) can reduce emission in pig houses by 40–65%, depending on the pig category and the presence of partly slatted floors. The gutters should be flushed twice a day with the liquid (thin) fraction of the slurry rather than water; flushing with water dilutes the manure and increases the cost of transporting and applying it in the field.

Increase bedding material (solid manure housing): Bedding material in animal housing can affect NH_3, N_2O, NO_x and N_2 emissions. The physical characteristics (urine absorbance capacity, bulk density) of bedding materials are of more importance than their chemical characteristics (pH, cation exchange capacity, carbon to nitrogen ratio) in determining NH_3 emissions from dairy barn floors (Misselbrook and Powell, 2005; Powell et al., 2008; Gilhespy et al., 2009). However, further assessment is needed on the effect of bedding on emissions for specific systems while taking into account the whole manure management path. The approach can have a positive interaction with animal welfare measures. However, approaches benefiting animal welfare can also be operated as slurry-based systems, with only little straw supply.

Regular cleaning of floors: Cleaning of floors in pig houses by mechanical scrapers or robots has the potential to substantially reduce NH_3 emissions. The automatic cleaning should be performed at regular intervals to achieve full benefits of the measure (Amon et al., 2007). It is worth mentioning that in warm countries (e.g. Mediterranean region), for sanitary reasons, floor cleaning is done more frequently with consequences in the slurry composition, which may reach up to 98% water.

Frequent slurry removal: Regular removal of slurry from under the slats in the house to an outside store can substantially reduce NH_3 emissions by reducing the emitting surface and the slurry storage temperature. A reduced storage temperature will also result in a reduction of methane (Amon et al., 2007).

Barn climatization to reduce indoor temperature and air flow: Surface cooling of manure with fans using a closed heat exchange system is a technique with a reduction efficiency of 45–75% depending on animal category and surface of cooling fins. This technique is most economical if the collected heat can be exchanged to warm other facilities such as weaner houses (Huynh et al., 2004). In slurry systems this technique can often be retrofitted into existing buildings. However, this system is not applicable when straw bedding is used or when the feed contains a lot of roughage. This is because a layer of floating residue may develop on top of the slurry.

Use of acid air scrubbers: Treatment of exhaust air by acid scrubbers (mainly using sulphuric acid) or biotrickling filters has proven to be practical and effective for large-scale operations in Denmark, Germany, France and the Netherlands (e.g. Melse and Ogink, 2005; Guingand, 2009). This is most economical when installed in new houses, because retrofitting in existing housing requires costly modification of ventilation systems. Acid scrubbers have demonstrated NH_3 removal efficiencies of more than 90%, depending on their pH-set values. Scrubbers and biotrickling filters also reduce odour and particulate matter by 75% and 70%, respectively (Guingand, 2009). Further information is needed on the suitability of these systems in South and Central Europe. Operational costs of both acid scrubbers and trickling filters are especially dependent on the extra energy use for water recirculation and to overcome increased back pressure on the fans. Optimization methods are available to minimize costs (Melse et al., 2012) and costs will be lower for large operations. The approach may also contribute to reducing N_2O and NO_x emissions, but more research is needed here.

Use of biological air scrubbers: Biological air scrubbers operate with bacteria that remove ammonia and odours from the exhaust air. Ammonia captured in biological air scrubbers typically undergoes nitrification and denitrification associated with increased emissions of N_2O, NO_x and N_2. Recovery of the collected N_r in bio-scrubbers may help offset this increase by reducing the need for fresh N fixation and production of chemical fertilizers.

2.2.5 Poultry housing

Designs to reduce NH_3 emissions from poultry housing systems have been described in detail in Economic Commission for Europe (2015) and in the document on BAT under the EU Industrial Emissions Directive (Santonja et al., 2017), and applying the following principles:

- Reducing the open surface area of emitting manure;
- Removing the manure frequently from the poultry house to an external slurry store (e.g. with belt removal systems);
- Quickly drying the manure to reduce hydrolysis of uric acid to ammonia;
- Using surfaces which are smooth and easy to clean;
- Treatment of exhaust air by acid scrubbers or biotrickling filters (i.e. biological air scrubbers); and
- Lowering the indoor temperature and ventilation as animal welfare and/or production allow, reducing microbial processes that mobilize N_r losses.

Many of the measures listed for cattle and pigs are also applicable for poultry systems, especially reduction of emitting surface, barn climatization to reduce indoor temperature and air flow), and acid air scrubbers. This section therefore focusses on additional considerations for poultry housing. Further information can be found in European Commission (2015), in the IPPC BREF document (Santonja et al., 2017) and in the UNECE Ammonia Guidance Document (Bittman et al., 2014).

Where poultry houses are disconnected from the ground (e.g. concrete base), emission reduction measures for NH_3 are not directly expected to affect nitrate and other N_r leaching and run-off. For smaller farms, which are not required to comply with national legislation (e.g. BAT) for layers, and for free-range poultry, pathways to the soil can also be anticipated. In such cases, NH_3 emission reduction including rapid drying and dry storage of poultry litter may also have benefits to reduce N_r leaching. In addition, expert observations have shown that downward pointing air exhausts onto porous ground surfaces surrounding poultry houses can lead to localized increases of N_r leaching and run-off into ground waters. Reduction of NH_3 emissions (and N_r-containing dusts) can therefore also contribute to reducing such hotspots of N_r leaching and run-off.

2.2.6 Mitigation measures for housing systems for laying hens

A wide range of regulations and minimum standards for protecting laying hens exist across the UNECE region. For example, in the EU, regulations apply under Council Directive 1999/74/EC. This Directive has prohibited the use of conventional cage systems since 2012. Instead, only enriched cages (also called 'furniture cages'), or non-cage systems, such as litter (or deep litter) housing systems or aviary systems, are allowed.

Rapid drying of poultry litter: Ammonia emissions from battery deep-pit or channel systems can be lowered by reducing the moisture content of the manure by ventilating the manure pit. The collection of manure on belts and the subsequent removal of manure to covered storage outside the building can also reduce NH_3 emissions, particularly if the manure has been dried on the belts through forced ventilation. The manure should be dried to 60–70% DM to minimize the subsequent formation of NH_3. Manure collected from the belts into intensively ventilated drying tunnels, inside or outside the building, can reach 60–80% DM content in less than 48 h, but in this case exposure to air is increased, risking an increase in NH_3 emissions. Weekly removal from the manure belts to covered storages reduces emissions by 50% compared with bi-weekly removal. In general, emission from laying hen houses with manure belts will depend on: (a) the length of time that the manure is present on the belts, (b) the drying systems, (c) the poultry breed, (d) the ventilation rate at the belt (low rate = high emissions) and (e) the feed composition. Aviary systems with manure belts for frequent collection and removal of manure to closed storages reduce emission by more than 70% compared with the deep litter housing system. While the primary drying poultry litter has been on reducing NH_3 emissions, keeping excreted N in the form of uric acid can be expected also to reduce N_2O, NO_x and N_2, since this will also reduce nitrification and denitrification. Dried poultry litter will therefore have a higher fertilizer value for farmers, which should be compensated by using reduced doses during land application, as compared with decomposed poultry litter.

Use of acid or biological air scrubbers: Treatment of exhaust air by acid or biological scrubbers (=biotrickling filters) has been successfully employed in several countries (Melse and Ogink, 2005; Ritz et al., 2006; Patterson and Adrizal, 2005; Melse et al., 2012). Hahne et al. (2016) in Germany counted 179 installed air scrubbers in poultry installations and 1012 scrubbers installed in pig houses, respectively. The main difference from pig systems is that poultry houses (especially with dried litter) typically emit a much larger amount of dust. Acid or biological scrubbers remove 70–90% of NH_3, and also remove fine dust and odour. To deal with the high dust loads, multistage air scrubbers with pre-filtering of coarse particles have been developed (Ogink and Bosma, 2007; Melse et al., 2008).

2.2.7 Mitigation measures for housing systems for broilers

To minimize NH_3 emission in broiler housing, it is important to keep the litter dry. Litter moisture and emissions are influenced by:

- Drinking-water design and function (leakage and spills);
- Animal weight and density, and duration of the growing period;
- Ventilation rate, use of in-house air purification and ambient weather;
- Use of floor insulation;
- Type and amount of litter; and
- Feed.

Reducing spillage of water from the drinking system: A simple way to reduce spillage of water from the drinking system is using a 'nipple drinkers' instead of 'bell drinkers'. This approach should be integrated into wider systems designed to keep poultry litter dry.

Air scrubber technologies to remove NH_3 from ventilation air are highly effective, but not currently widely implemented because of high installation and running costs. Packed-bed filters and acid scrubbers currently available in the Netherlands and Germany remove 70–90% of NH_3 from exhaust air. Comprehensive measuring of air scrubbers is done by the German Agricultural Association (DLG, 2020) based on a scientific standard testing frame. As with such systems for laying poultry, questions about long-term reliability due to high dust loads need to be further clarified. Various multi-pollutant scrubbers have been developed to also remove odour and particulate matter (PM_{10} and $PM_{2.5}$) from the exhaust air (Zhao et al., 2011; Ritz et al., 2006; Patterson and Adrizal, 2005). Implementation of both acid scrubbers and biological air scrubbers for broiler housing is largely similar to that for laying hens

3 Manure storage, treatment and processing

3.1 Principles of emissions from manure storage, treatment and processing

For livestock agriculture to become sustainable, an optimal and efficient use of manure nutrients and organic matter is essential. However, manure nitrogen may be easily lost via gaseous emissions (NH_3, N_2O, NO_x and N_2) and leaching of nitrate (NO_3^-) and other N_r compounds. Besides nitrogen losses, animal and manure emissions of methane (CH_4) to the atmosphere must be reduced as far as possible, to limit climate change impacts.

Significant N losses may occur during storage of urine, faeces or mixtures (slurries and farmyard manures/deep litters), and simple treatment (e.g. solid-liquid separation) or more advanced processing (e.g. anaerobic digestion, ultrafiltration) may enable more appropriate manure management with lower N losses.

The *treatment of manures* typically involves a one-step operation to improve the properties of the manure. Expected effects include the improvement of the fluid properties (by adding water or by separating solids), the stabilization of volatile nutrients (by acidification) or a reduction in odour nuisance (e.g. aeration). Single-stage treatment of manures is typically applied on farms in the proximity of livestock buildings. The mass and ingredients of manures are not or only slightly changed by treatment systems.

The *processing of manures* generally describes more complex and multi-step processes, which are used specifically to produce new products, for example,, higher nutrient content, less water content, free of undesirable odours and hygienically safe. In most cases, manure processing is used to produce marketable products that can be used as fertilizers and soil conditioners, as well as secondary raw materials (e.g. fibres). Manure processing technologies may either be located on farms or operated as central/decentral plants.

Manure treatment and processing always come at a cost, both in economic, energy and environmental terms, so the simplest option fulfilling the goal(s) should always be the priority option: (1) direct land application, (2) simple treatment second, (3) advanced processing (with (1) first, according to local limitations, including those related to pollution). Simple treatment and advanced processing are most relevant when conditions (e.g. high regional livestock density, large manure N surplus relative to local crop demand) favour overall environmental benefits from treatment or processing. Such systems should be designed with awareness of the need to avoid pollution swapping (e.g. reducing ammonia loss, but increasing nitrate leaching somewhere else and vice versa).

Animal slurry composition is typically not ideal with regard to low emission handling and crop fertilizing properties. In particular, the high DM and carbon content pose several problems during slurry storage, application and crop utilization (Table 1). This points to the opportunity for increased development of systems to collect and store urine and dung separately, or to apply manure treatment by solid-liquid separation.

High slurry DM tends to result in crust formation on the slurry surface and/or to sedimentation on the bottom of the slurry tank. In order to achieve an even distribution of nutrients in the slurry, slurry must be mixed/homogenized prior to application. Homogenization of slurry with high DM content is energy-consuming and increases NH_3 emissions as a larger volume of the slurry comes in close contact with the atmosphere.

Slurry contains considerable amounts of easily degradable carbon that serves as substrate for microbes. During slurry storage a continuous degradation of organic matter can be observed. Degradation intensity is strongly dependent on the slurry DM content. Amon et al. (1995) investigated changes in slurry composition over a 200-day storage period for stored cattle, beef and pig slurry. Degradation of organic matter was found to be significantly greater with higher slurry DM content. Such slurry degradation will include mineralization to form of ammonium (NH_4^+) from organic matter. This points to the opportunity for increasing the immediate fertilizer value of the slurry, provided that storage is covered, thereby avoiding NH_3 emissions and benefiting from increased slurry NH_4^+ content.

As conditions in slurry are anaerobic, degradation of organic matter is always dominated by anaerobic pathways. This means, that both CH_4 and CO_2 are formed as end-products of the degradation process. It is thus to be assumed that high DM slurry bears a greater risk for CH_4 emissions, contributing

Table 1 Challenges and benefits resulting from slurry high dry matter and carbon content, low nutrient content

	Problems
Storage	• natural crust formation and sedimentation of solids, giving heterogeneous concentration of nutrients • high energy consumption per unit of nutrient for pumping and mixing • potentially higher emissions of NH_3, N_2O, N_2, CH_4 and odour
Field application	• high potential risk of NH_3 losses due to slow infiltration • high technical effort required (at high economic cost) for even and low emission application • suffering of crop plants due to scorching by broadcasted slurry
Crop utilization	• less effective crop uptake of slurry N than from mineral fertilizer • increased temporary N immobilisation in the soil, increasing risk of lower crop N effect • higher risk of denitrification and subsequent N_2O and N_2 emissions • crop N effect less predictable/more variable than from mineral fertilizer
	Benefits
Storage	• natural crust formation may serve as a natural barrier, inhibiting NH_3 transport to the atmosphere; furthermore, the crust may have significant capacity for CH_4 oxidation, due to its partial aerobic conditions and high microbial activity
Field/soil	• high dry matter and carbon content contributes to maintenance of soil organic matter content and biologically active soil

significantly to climate change. This also points to the opportunity for CH_4 and CO_2 recovery, for example, linked to anaerobic digestion for production of biogas.

Environmental-friendly slurry application in the field requires that the slurry is more evenly applied near or below the soil surface. It is much more complicated to fulfil this requirement when the slurry has a higher DM content, causing a higher viscosity and less easy flow through band spreading hoses. Following application of slurry, NH_3 emissions can be substantial and are found to increase with an increase in slurry DM content, due to slower soil infiltration (Sommer et al., 2013; Bittman et al., 2014). This emphasizes the importance of maintaining low DM contents of slurries. By reducing NH_3 and other nitrogen losses, available N resources on farms are increased, decreasing the need for additional N to be bought as manufactured inorganic fertilizer.

The N availability to plants is difficult to calculate with high DM slurry, because a high DM content drives increased microbial immobilization right after application. The more narrow the C/N-ratio, and the higher NH_4-N content, the more slurry N is potentially available to plants, whereas with a wide C/N-ratio,

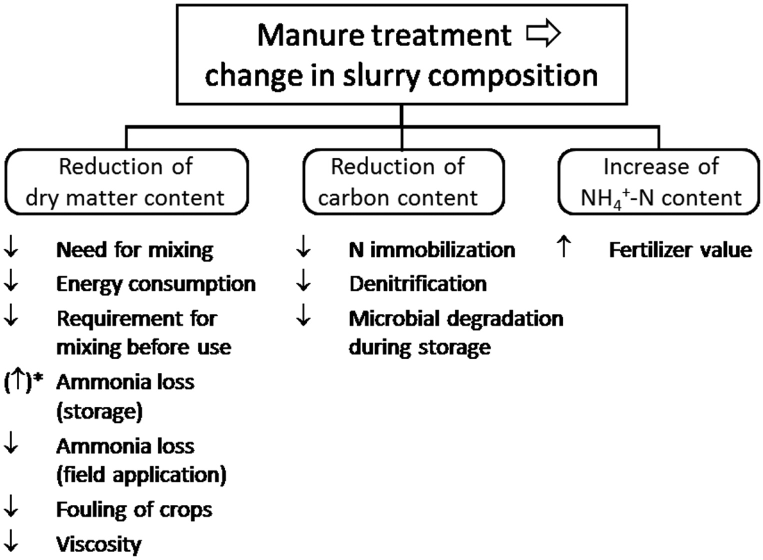

Figure 4 Effect of changes in slurry composition achieved by manure treatment. Arrows indicate decrease (↓) or increase (↑) in the listed property. * If depending on natural crusting of manure to reduce emissions rather than other types of cover.

part of the slurry N is immobilized in the soil N pool and becomes available only at a later and often unpredictable or even too late stage, causing increased risk of nitrate leaching. In addition, an increase in slurry DM and subsequent soil N content has the potential to increase rates of nitrification and denitrification, increasing subsequent N_2O, NO_x and N_2 losses (e.g. Dosch, 1996). It may thus be beneficial to reduce slurry DM and carbon content at an early stage of manure management. This leads to several manure treatment options that can be evaluated in relation to the requirements listed in Fig. 4.

In line with the objectives of the EU Circular Economy Action Plan,[1] there is an opportunity to encourage the use of recycled nutrients that can replace nutrients otherwise obtained from primary raw materials. The main challenge is to use recycled nutrient resources that have an equal or better environmental performance than the primary nutrient resources they replace. Efforts are ongoing across the EU to develop manure-processing technologies that allow manure to be turned into a safe and agronomically valuable resource that could be more widely used.[2]

Techniques for simple manure treatment can be classified as physical, chemical or biological (Fig. 5, Bernal et al., 2015). Furthermore, a number of

1 https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52015DC0614
2 https://ec.europa.eu/jrc/en/research-topic/waste-and-recycling

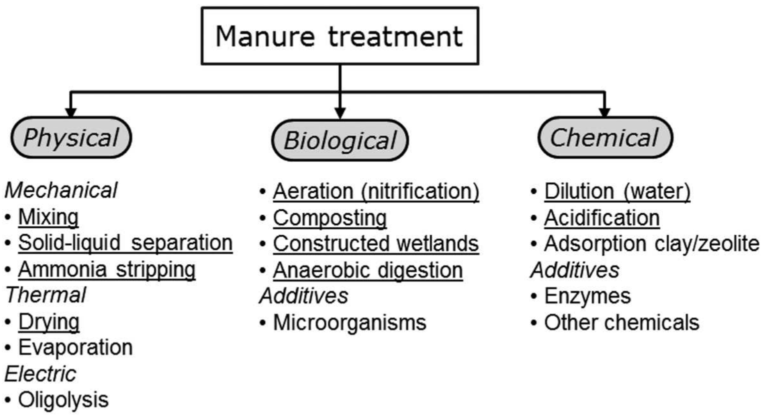

Figure 5 Options for simple manure treatment. Options underlined are in some regions commonly applied in full scale on commercial farms (mainly pig farms); other options are applied either rarely or only in experimental/pilot scale – these are not dealt with further here, pending the availability of proof-of-concept and documentation.

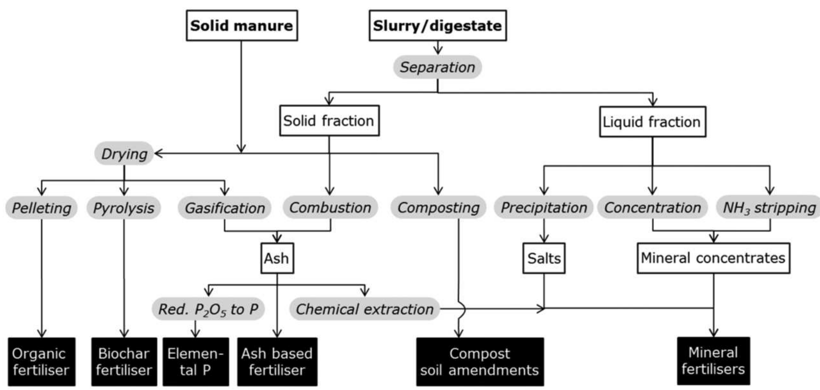

Figure 6 Options for combining simple treatment with more advanced processing of manures to recover and upgrade nutrient and energy, resulting in widely different bio-based fertilizers (modified from Jensen, 2013). Only a few are yet applied in full commercial scale; other are still in experimental/pilot stage (and are therefore not dealt with further here).

different options/technologies are available for further and more advanced processing of raw or treated manures for recovering and upgrading nutrients and organic matter from different manure types (Fig. 6). For slurries or other liquid manures, such as digestate from anaerobic digestion of manure and other bio-waste, all treatment steps start with mechanical separation into (a) a *solid fraction* which is relatively rich in organic N and P, and (b) a *liquid fraction*, with low P, but relatively high mineral N and K contents. Different simple techniques can be combined with each other. This allows a wide variety of by-products to be combined, resulting in highly variable distribution of organic nitrogen, ammoniacal nitrogen, phosphorus, carbon and other nutrients, which must be taken into account when managing the different fractions.

There may be additional possible treatments of the liquid phase. In order to save water without increasing the amount of nitrogen supplied to the soil, and to favour the circular economy of water, it is common to carry out successive treatments of the liquid phase, so that the resulting product can be used in fertigation. For example, in the south of Spain wetlands are being constructed to allow the reuse of water for irrigation, in areas of scarce availability. In addition to nitrogen, many other characteristics have an influence in the decision of choosing a procedure, such as the contribution of organic matter, the formation of methane and other GHGs, the presence of other nutrients, type of agricultural systems, salinity, weather and, very important in the countries of southern Europe, the water footprint.

Each of these processing pathways and resulting products (Fig. 6) has certain advantages and disadvantages, and the net environmental benefits/impacts and economic costs/profits differ greatly. A number of factors must be considered when prioritizing the processing options (Jensen, 2013):

- The primary aim should be nutrient recycling, mainly N and P; N is consumed in the largest quantities, is expensive and has impacts on energy consumption and GHG emissions, while P is a scarce and non-renewable resource, with the highest price.
- Splitting N and P into different fractions is generally beneficial, as this enables more flexible and balanced fertilization in accordance with the needs of many crops.
- The technology or combination of technologies applied should preferably also produce energy or consume relatively little energy, so net energy production should be taken into account for both environmental and economic reasons.
- Local solutions should be preferred, avoiding too high transport cost and impacts; regional or more central solutions are therefore only justified if the economy of scale via higher efficiency outweighs the negative impacts of transporting the manure to a common facility.
- The quality of end-products and byproducts is assessed differently depending on the perspective of the user. For instance, a manure combustion ash, where the majority of the N has been lost, will not be appreciated by an organic farmer, while a compost is highly appreciated for its soil ameliorating effect and slow release of N, even if some N has been lost in the process.
- Biochars and compost may be valued highly by orchard and vineyard producers for its effects on soil-water-holding capacity and nutrient

retention, whereas conventional crop production farmers may value mineral concentrates and salts more highly. Production of recovered, bio-based fertilizer products should not be supply-driven (trying to solve a waste problem), but rather demand-driven (bio-based fertilizers that the farmers want).

3.2 Mitigation measures for manure storage, treatment and processing

3.2.1 Covered storage of slurry (natural crust)

Where slurries have a high DM content, these may form a natural crust during storage, which is associated with substantially reduced ammonia emission (Bittman et al., 2014). There is large agreement that crusting impacts the gas release in many ways: enhanced resistance to mass transfer (Olesen and Sommer, 1993), oxidation of NH_3 (Nielsen et al., 2010) and CH_4 (Petersen et al., 2005) and formation of N_2O related to nitrification and denitrification occurring in liquid-air interfaces near air-filled pores present in crusts (Petersen and Miller, 2006). NH_3 and CH_4 may be consumed due to microbial activity in the crust leading to an emission reduction (Petersen and Ambus, 2006; Nielsen et al., 2010) while N_2O production may be enhanced (VanderZaag et al., 2009). A comprehensive assessment of the current knowledge on the effect of natural crusts can be found in Kupper et al. (2020). The reference is taken as uncovered storage, including on an impermeable surface, which explains the benefit for nitrate leaching.

3.2.2 Covered storage of manure (solid cover)

A wide range of options are available for covered manure storage using solid covers, including use of metal tanks with solid lids, floating covers on lagoons and use of slurry bags, most of which are associated with negligible ammonia emission if well operated. Further details of such systems are provided by Bittman et al. (2014). Less focus has been given to ensuring that solid manure (e.g. farmyard manure and poultry manure) are covered, for example, through use of plastic sheeting. The reference is taken as uncovered storage, including on an impermeable surface, which explains the benefit for nitrate leaching.

3.2.3 Covered storage of manure (dispersed coverings)

Ammonia emissions can be significantly reduced when covering solid organic fertilizers with dispersed coverings such as peat, clay, zeolite and phosphogypsum. The basis of the approach is to prevent contact of NH_3 emitting surfaces with the air, especially when covering them with ammonium absorbing substances. Lukin et al. (2014) found that total NH_3 emissions from poultry manure amounted to 5.9% when it was covered with peat, 4.7% when it was covered with loam, 1.3% when it was covered with zeolites and 16.9% when it was covered with phosphogypsum. These values are relative to NH_3 emissions in the reference variant with no covering. Use of these simple materials to cover piles of organic fertilizers thereby substantially reduces NH_3 emissions into the atmosphere (Lukin et al., 2014). Protocols are needed to specify minimum thickness of each type of covering material. Further testing is needed to assess the effect on N_2O, NO and N_2 emissions. Unless an impermeable base is used, the approach risks significant nitrate leaching. In combination with an impermeable base, it can reduce both N_r emissions to air and leaching losses to water.

3.2.4 Storage of solid manure under dry conditions

Simply storing manure in a dry place, out of the rain, can also reduce nitrogen emissions from a range of N_r compounds and N_2. This is even more important for dried poultry litter, where keeping manure dry and out of the rain helps to avoid hydrolysis of uric acid to form ammonia. However, poultry litter is hydroscopic and will emit some ammonia when in humid atmospheres, even when kept free of rain (e.g. Elliot and Collins, 1982). Keeping solid manure dry during storage minimizes mineralization and denitrification which can give rise to N_2O, NO_x and N_2 emissions, as well as reducing nitrate and other N_r leaching.

3.2.5 Storage of manure on a solid concrete base with walls

Investments in this approach have been motivated out of the need to reduce nitrate leaching and other N_r leaching by avoiding run-off and infiltration into the soil. The approach has the benefit of being low-cost, but risks substantial NH_3 emissions, while also being ineffective at avoiding nitrification and denitrification, which contribute to N_2O, NO_x and N_2 emissions. The approach is preferable to open-field storage of solid manure on a permeable surface. Storage of solid manure on concrete areas is considered good agricultural practice for nitrate pollution, but makes no contribution to reducing NH_3 emissions.

3.2.6 Slurry mixing

Slurry mixing in the storage is one of the most commonly applied manure treatment technologies. Slurry is thereby homogenized, typically shortly prior to field application, in order to achieve a more homogenous distribution of nutrients across the field(s) to which the volume of the slurry storage is applied. Apart from this, mixing does not offer any additional benefits compared to untreated slurry. Neither DM nor carbon content is reduced, and the C/N-ratio is not altered. No significant changes in N_2O or CH_4 emissions are expected, but NH_3 may tend to increase, depending on the extent and timing of mixing (mixing will tend to increases pH by promoting CO_2 loss from slurry), so mixing should only be done shortly before field application.

3.2.7 Adsorption of slurry ammonium

Slurry additives can act on a chemical, physical or biological basis. Clay/zeolite mineral additives have been shown to adsorb NH_4-N and can thus potentially reduce NH_3 losses. However, this can only be achieved effectively with high amount of additives, for example, 25 kg of zeolite per m^3 slurry have been shown necessary to adsorb 55% of NH_4-N (Kocatürk-Schumacher et al., 2017, 2019). On most commercial farms it is neither logistically possible nor economically profitable to add such high amounts of slurry additives. Addition of biochar may also reduce NH_3 emissions from stored manure.

3.2.8 Slurry acidification during storage

An obvious way to minimize ammonia emissions from stored slurry is to decrease pH by addition of strong acids or other acidifying substances. This can also be done in the animal house (Housing Measure 8). Care must be taken that the low pH is maintained to get the full benefit of this measure. Slurry with a sufficiently reduced pH will also emit less methane. This solution has been used commercially since 2010 in countries such as Denmark (by 2018, around 15–20% of all slurry applied in Denmark was acidified; Birkmose, pers. comm.), and its high efficiency for minimizing NH_3 emissions has been documented in many studies (see review by Fangueiro et al., 2015, with emission reductions by >80% possible). It is most typical to acidify slurry using sulphuric acid (cheapest industrial acid; also, the sulphate added serves as a relevant plant nutrient source), though use of other acids is also possible. Acidification also reduces methane formation very effectively, by up to 67–87% (Petersen, 2018). Reduced nitrification and denitrification decrease the potential for N_2O and N_2 emission, though further studies are required to demonstrate efficiency for this. In one novel variant of this method, electricity is used to produce a plasma, which oxidizes N_2 to NO and thence to nitrogen dioxide (NO_2), which converts in slurry to produce nitric acid (HNO_3). In this way, slurry acidification is achieved while augmenting the nutrient value of the manure (Graves et al., 2019). More research is needed to assess this option fully.

Costs for in-house acidification systems can be higher than acidification during field application (Manure Measure 9), but are counteracted by additional benefits including improved in-house air quality benefiting animal and staff, which may influence productivity, retention of more slurry

N throughout the manure management chain, and associated savings in fertilizer costs.

3.2.9 Slurry aeration

Slurry aeration introduces oxygen rapidly into the slurry in order to allow aerobic microbes to develop. Oxidation of organic matter to CO_2 and H_2O increases, and thus CH_4-production and emission is reduced. Odorous compounds are degraded. Slurry DM content decreases. Thus, less mixing is needed and technical properties of slurry are often improved. However, successful aeration requires 200 m^{-3} oxygen per t of slurry (Burton, 1998).

Slurry aeration increases NH_3 emissions and energy consumption. The potential for NO_x emissions is also expected to increase, as increased oxygen availability promotes nitrification, while subsequently higher levels of nitrate availability may increase other oxidized N_r losses and denitrification. The extent of these increases has so far been quantified only in few studies (Amon et al., 2006) and more research is necessary to allow a complete evaluation. In the present context, an increase in denitrification to form N_2 is considered a waste of available N_r resources.

3.2.10 Mechanical solid-liquid separation of slurry fractions

During slurry separation, solids and liquids are mechanically separated from each other. This results in two fractions: a liquid slurry fraction, with relatively low DM content compared with the slurry, and a solid fraction that can be stored in heaps. Energy consumption for slurry separation is relatively low, but depends on the technology used for separation. DM content in the liquid fraction is reduced by 40–45%, and vice versa for the solid. Carbon content in the liquid is typically reduced by 45–50%, with the C/N-ratio of the liquid decreasing from about 10:1 to about 5:1 (Amon et al., 1995; Sommer et al., 2013). As carbon is removed from the slurry, microbial degradation of organic matter during slurry storage is reduced. However, the opposite may be the case for the solid fraction, depending on storage conditions.

The removal of solids reduces crust formation and sedimentation of the liquid fraction compared to raw slurry. Thus, less-intensive mixing is necessary to homogenize the slurry prior to application. Efforts for low-emission application techniques are also reduced as separated slurry has a lower viscosity and flows more easily through band spreading hoses (Owusu-Twuma et al., 2017). Slurries with very low DM content can be spread with simple nozzle-beam-dischargers that can be operated on slopes >10%, which is not possible with other band-spreading techniques. Furthermore, separated slurry liquid fraction has a low viscosity and infiltrates rapidly into the soil. Thus, plants get less dirty, and ammonia emissions after liquid fraction spreading are typically reduced.

A substantial reduction of ammonia emissions by slurry separation is therefore possible for the liquid phase, especially following land application (e.g. Amon et al., 2006).

The liquid fraction of separated slurry has a narrow C/N-ratio which reduces the potential for microbial N immobilization in the soil and the potential for N_2O emissions. Crop N availability of the liquid fraction is therefore more predictable and can be better calculated in order to match nutrient requirements of crops to actual fertilization. Dosch (1996) investigated fertilization with untreated and separated slurries and found significantly higher denitrification rates with untreated slurry. Separated slurry liquid fraction on the other hand resulted in significantly higher crop yield. However, the solid fraction needs to be handled with care during storage to avoid elevated ammonia emissions. Furthermore, the solid fraction may become a source of methane emissions, if not properly treated. Alternatively, if the solid fraction is used as feedstock for biogas production, this methane potential may be recovered and utilized as renewable energy source. After application, the solid fraction serves mainly as soil improvement and slow release N fertilizer.

Slurry separation fulfils most requirements of appropriate manure treatment. Costs could be further reduced if the technology was more widespread and more separators were on the market and available to farmers. As the fertilizer value of the liquid fraction from separated slurry is improved, mineral N fertilizer input can be reduced. The slurry liquid fraction can be applied at the soil surface in a growing crop with very simple low-cost slurry band spreaders with a high uptake efficiency and fertilizer replacement value. The main caveat of the method is the difficulty of appropriate storage, handling and utilization of the solid fraction; this needs to be low emission, in order not to compromise benefits of the liquid fraction. An alternative is to use the solid fraction as a feedstock in nutrient anaerobic digestion with nutrient recovery.

3.2.11 Anaerobic digestion

Anaerobic digestion of animal manures is mainly implemented for bio-energy production reasons. Improvement of manure quality is therefore typically considered to be a 'by-product' of anaerobic digestion. However, when combined with nutrient recovery methods (Fig. 6), nutrient management can be considered as fully integrated as a key goal in implementation of anaerobic digestion.

Biogas production from animal manures through anaerobic digestion aims at maximizing the bio-methane yield. Where no biogas recovery system is available, unintended anaerobic degradation of organic substances into methane during manure storage should be limited as far as possible, to prevent emission to the atmosphere of this strong GHG. This also maximizes the resource availability for subsequent biogas production when facilities are available. Under these circumstances (including heating of the manure to promote digestion) methane production is enhanced, allowing its collection and use (e.g. in combined heat and power production). Anaerobic digestion not only reduces methane emissions from subsequent storage of the manure digestate, but the energy produced typically substitutes consumption of use of fossil energy. Both effects reduce anthropogenic GHGs.

Anaerobic digestion reduces manure carbon and DM content by about 50% (Amon and Boxberger, 2000). NH_4^+ content and pH in digested slurry are higher than in untreated slurry. Thus, potential for ammonia emissions during subsequent slurry storage are increased. Digested slurry therefore has to be stored in covered slurry stores. These should be connected to the gas-bearing system of the biogas plant, because methane is still formed after the main digestion phase has taken place in the heated digester.

Due to the reduced DM content, biogas slurry can infiltrate more rapidly into the soil, which tends to reduce ammonia emissions after slurry application. However, the increased NH_4^+ content and pH give rise to higher potential for ammonia loss especially after surface application. It is therefore strongly recommended to apply biogas slurry with low-emission techniques near or below the soil surface (e.g. band application or injection).

The combined implementation of anaerobic digestion (reducing DM content, increased NH_4^+ and pH) and low-emission land-spreading application (e.g. trailing hose, injection) considerably reduces ammonia emissions. In addition, N immobilization and N_2O losses are likely to be smaller than from untreated slurry, due to the removal of easily degradable organic substances during the anaerobic digestion process. Energy consumption for pumping and mixing is considerably reduced due to the reduced DM content. When combined with appropriate methods for low-emission land-spreading of the digestate, anaerobic digestion therefore has multiple benefits. In addition, it provides the opportunity for further processing for more advanced forms of nutrient recovery, including nutrient precipitation, concentration and ammonia stripping (Fig. 6).

3.2.12 Manure composting

Composting of manure is done in order to create a stable and odourless bio-based fertilizer product, with lower moisture content, while containing most of the initial nutrients, free of pathogens and seeds (Jensen, 2013). Composting significantly reduces mass (as a result of water evaporation and volatile solids decomposition to release CO_2) and hence transport costs. However, it is difficult to avoid some loss of manure N in the form of NH_3 and the process also emits greenhouse gases, with potential for increased N_2O and CH_4 emissions, in addition to NO_x and N_2 (Chowdhury et al., 2014). The N fertilizer value of composts is often significantly lower than the N-rich manure components it is made from, which is largely a result of associated NH_3 and N_2 emissions (Jensen, 2013). Composting on porous soil surfaces may also be associated with significant leachate, including NH_4^+, NO_3^- and other N_r compounds. Composting is typically a low-cost technology, but implies space requirements and energy consumption. Overall, it can therefore not usually be recommended to mitigate nitrogen losses, but may be preferred on other criteria (e.g. volume and weight reduction, compost product stability, reduced odour, improved marketability and soil amelioration).

3.2.13 Nutrient recovery by drying and pelletizing of manure solids

Drying and pelletizing of solid manures, slurry or digestate solids can be done to create a more stable and odourless bio-based fertilizer product. Drying is energy-intensive and thereby relatively expensive, unless excess energy (e.g. from the combined heat and power plant engine on a biogas plant) is freely or cheaply available. Increased ammonia loss is inevitable in the process, unless exhaust filtering or scrubbing and recovery is applied, or the solids are acidified prior to drying. Drying is usually combined with a pelletizing process to facilitate handling. The pelleted material can be marketed as an organic matter and P-rich soil amendment; if acidified prior to drying, the resulting product may also be rich in plant available N (Pantelopoulos et al., 2017).

3.2.14 Nutrient recovery by combustion, gasification or pyrolysis

Combustion, thermal gasification or pyrolysis of manure and digestate solids can be used to generate a net energy output for heat and/or electricity production. However, at present the method leads to a more or less complete loss of the manure N, which is converted into gaseous N_2, as well as NO_x and NH_3. Available advanced technologies (e.g. selective non-catalytic reduction) focus on denitrifying these N_r gases to N_2. Until systems are implemented to minimize N_2 formation and recover the N_r gases, this measure cannot be considered appropriate for abating overall N loss.

At the same time, the approach produces ash or biochar residuals. These ashes contain the non-volatile nutrients, concentrated relative to the solids. They can be used as an ash-based, P- and K-rich soil amendment or bio-based fertilizer. The availability of the remaining nutrients in the ash is generally much lower than for the raw manure, whereas for biochar it is in-between ash and raw manure. Organic compounds in the biochar that are produced are very recalcitrant to biological decay and have a very large specific surface area, being potentially charged. This means that such biochar may be used for soil amendment, ameliorating soil pH and organic matter positively.

3.2.15 Nutrient recovery by precipitation of nitrogen salts

Struvite ($MgNH_4PO_4{\cdot}6H_2O$) can be precipitated from liquid manures, provided that the appropriate conditions are present (pH ~9, a molar ratio 1:1:1 of Mg^{2+}:NH_4^+:PO_4^{3-}, conducive physical settling conditions). As such the precipitation of struvite is a method for removal and recovery of both N and P from liquid manures. The method has been developed for wastewater treatment, where P removal can easily reach more than 70% and it is commercially available for sewage treatment plants, although not yet widely applied. For manures, the struvite technique is particularly relevant for anaerobically digested slurries and the liquid fraction from digestate separation; hence, it has been the subject of massive research in the past decade and quite high removal efficiencies have been achieved (56–93%; see further review in Jensen, 2013). However, it only works for the N already present as NH_4^+ and further development is needed for appropriate application to liquid manures and digestates. So far, only a few commercial-scale plants are in operation worldwide. The main advantage of struvite is its high concentration and similarity in physical-chemical properties to conventional mineral N fertilizer.

3.2.16 Nutrient recovery by concentration of nitrogen salts and solutions

Mineral concentrates are highly nutrient-rich solutions that may be obtained via ultrafiltration, evaporation or reverse osmosis of the liquid fraction from separation of slurry or digestate. These mineral concentrates (the retentate) may be directly applied to agricultural land and the byproduct water low in nutrients (the permeate) may be directly discharged to surface waters or the sewage system. The greatest experience with these technologies in Europe are from the livestock regions of the Netherlands and Belgium, where a number of centralized and large-scale manure processing plants utilizing a range of technologies in combination (e.g. anaerobic digestion, solid-liquid separation, ultrafiltration/reverse osmosis/solids drying). Provided that the losses can be kept to a minimum, the mineral fertilizer replacement value of the mineral concentrates can be relatively high, as they resemble commercial liquid fertilizers, with nearly all the nutrients in a mineral, plant-available form. However, to avoid gaseous NH_3 losses this may require prior acidification or injection of the concentrate into the soil (Jensen, 2013). As these technologies are still under investigation, the UNECE Categories are currently uncertain (i.e. Category 3, pending further assessment).

3.2.17 Nutrient recovery by ammonia stripping

Air stripping of NH_3 is a process whereby the liquid fraction after manure separation is brought into contact with air, upon which NH_3 evaporates and is carried away by the gas. Instead of ambient air, 'steam stripping', can be used where steam is replaces use of air as the ammonia carrier. Since evaporation occurs from the liquid surface, it is advantageous to ensure that the liquid has a large surface area. This can be achieved in a stripping column with structured packing, where it spreads over the packing material in a thin film and therefore has a considerably larger surface. The mass transport also increases with the concentration of $NH_{3(aq)}$ in the liquid phase, hence, if pH and/or temperature is increased, an increasing part of total ammoniacal nitrogen is in $NH_{3(aq)}$ form and the mass transport of NH_3 increases (Sommer et al., 2013). Altogether this makes the technology relatively energy demanding and costly, though cheap/free surplus energy from, for example, a biogas- combined heat and power plant may reduce energy costs. Alternatively, using selectively permeable membrane contact systems at lower temperatures may offer a cheaper solution, if membrane fouling can be avoided. Ammonia released from an NH_3 stripping column or from a manure drying facility can be collected using wet scrubbing with an acid solution, typically sulphuric acid to make ammonium sulphate (most common), but also has been reported with nitric acid to make ammonium nitrate. Both compounds can serve as raw materials for mineral fertilizers, and thus provides opportunity for circular economy development as part of the fertilizer industry's commitment to include recovered and recycled N_r. In general, this is a well-known and generally effective technology. The main barriers are the relatively low N scrubber-liquid concentrations achievable (and thus high logistic costs), and the quality requirements for introduction of the scrubber-liquid into the raw materials market for the fertilizer industry. Actual developments are working on process improvement through applying sulphur or nitrate instead of an acid and on developing a system with CO_2 to generate ammonium carbonate which can also be used as fertilizer and bring C back into the soil. The ammonia-low liquid fraction is then used to flush the manure channel to reduce NH_3 and CH_4 emissions in the house.

4 Best practices and priority measures

Best practices and priorities for the selection of abatement/mitigation measures must be based on the following criteria: (i) implementability; (ii) effectiveness; (iii) impact on environmental emissions; (iv) secondary effects; (v) controllability; and (vi) cost efficiency. Based on these criteria, we suggest the following priority measures:

4.1 Livestock feeding

The following priorities through livestock feeding help to reduce nitrogen losses:

- Avoid N surplus from the very beginning of the manure management continuum;
- Adjust animal diet to animal performance (in line with existing guidance in the UNECE Ammonia Framework Code);
- Adapt animal diet to shift N excretion from urine to faecal excreta;
- Dairy cattle:
    - Reduction of crude protein content in the diet;
    - Adapt diet and dairy production system to site specific conditions;
    - Increase milk yield with moderate level of concentrates;
    - Increase production cycles per cow.
- Pigs:
    - Reduction of crude protein content in the diet;
    - Multiphase feeding;
    - More use of food wastes (inc. from processing and retail) as a way to reduce upstream and downstream emissions.

4.2 Livestock housing

The following priorities help to reduce nitrogen losses from livestock housing:

- Reduction of indoor temperature;
- Reduction of emitting surfaces, reduction of soiled areas;
- Reduction of air flow over soiled surfaces;
- Use of additives (e.g. acidification);
- Frequent removal of slurry to an outside store; and
- In the longer term: smart barns with optimized ventilation (open housing) or ventilation air scrubbing (closed housing), immediate segregation of urine and faeces components, in-house acidification of slurry (pigs and cattle).

4.3 Manure storage, treatment and processing

The following priorities help to reduce nitrogen loses and to mobilize nitrogen recovery and reuse from manure storage, treatment and processing:

- Store solid manures outside the barn on a solid concrete base in a dry/covered location;
- Ensure tight slurry stores, and cover either by a solid cover, or by ensuring sufficient natural crust formation;

- Use manure treatment where relevant to:
  - Homogenize nutrient content for more even field spreading to ensure that all available nutrient resources are used effectively for crop growth;
  - Reduce slurry DM content, for example, by solid-liquid separation, to enhance soil infiltration and limit NH_3 loss;
  - Increase slurry NH_4^+ content to maximize crop N availability;
  - Lower pH by acidification to reduce NH_3 volatilisation and enhance fertilizer value; and
  - Apply manure treatment methods to enable combined energy and nutrient recovery, that is, anaerobic digestion, where relevant.

The use of manure advanced processing for N recapture and production of value-added nutrient products from recycled manure N resources should be focused on situations where other effective options are not available. Examples of such advanced processing technologies are high-tech separation by filtration, reverse osmosis and NH_3 scrubbing, drying of manure and digestate solids for organic fertilizer production.

Production of recovered, bio-based fertilizer products should not be supply-driven (trying to solve a waste problem), but rather demand-driven (bio-based fertilizers the farmers want).

5 Conclusion and future trends in research

It is clear that manure management impacts quantities of N_r emissions (NH_3, direct and indirect N_2O emissions, NO_x emissions, NO_3^- leaching) and N_2 emissions, as well as emissions of CH_4 and CO_2. This applies at each stage of the manure management continuum (Chadwick et al., 2011). Since production of these gases, as well as of leachable N_r, is of microbial origin, the DM content and temperature of manure and soil are key factors for farm manure management decisions that influence the magnitude of N and GHG losses. There remains a degree of uncertainty in emission rates of N and GHGs from different stages of manure management, and researchers continue to investigate interactions of the management and environmental factors that control emissions. Some specific approaches to reducing N and greenhouse gas emissions from livestock housing and manure storage include optimizing diet formulation, low-emission housing technologies, manure processing and nutrient recovery. The technologies include air scrubbers, covered manure storage, slurry separation and anaerobic digestion, nitrogen concentration and stripping methods.

Existing legislation across the UNECE region offers opportunities to find 'win–win' scenarios, with benefits in reducing multiple forms of pollution. An example is the EU Nitrates Directive (91/676/EEC), which has led to development of Nitrate Vulnerable Zone action plans to prevent application of animal manure, slurry and poultry manure (with high available N content) in autumn, a practice which reduces N losses, as well as direct and indirect N_2O losses. Care is needed to ensure that legislation does not lead to potential 'pollution swapping' (e.g. unadjusted use of slurry injection to reduce NH_3 emissions at the expense of an increase in N_2O emissions, with no modification of N inputs). A core principle is that measures that reduce one form of N loss need to either be accompanied by a reduction of fresh nitrogen inputs, or be accompanied by an increase in harvested products, to maintain mass consistency. In this way, what may first seem a trade-off at the field-scale, can be seen at the landscape and regional scale as an opportunity to move towards a more circular system with lower overall N losses.

The nature of the N cycle and its interaction with the C, P and other nutrient cycles demands a holistic approach to addressing N and GHG emissions and mitigation research at a process level of understanding. Systems-based modelling must play a key role in integrating the complexity of management and environmental controls on emissions. Progress has been made to this end (Sommer et al., 2009), with some studies producing whole farm models encompassing livestock production (del Prado et al., 2010).

Concepts for best practices to reduce adverse environmental impacts depend on the following integrated concepts:

- Relationship between nitrogen and GHG emissions;
- Influence of climate change on nitrogen emissions;
- Interaction between abatement/mitigation and adaptation measures;
- Interaction between nitrogen emissions and animal welfare;
- Integrated assessment of the whole manure management continuum;
- Integrated assessment considering the three pillars of sustainability: economy, environment, society;
- Interaction between consumer demand and nitrogen emissions;
- Development of region-specific concepts for sustainable intensification;
- Modelling of livestock production at regional, national and global scale; and
- Economic impact of both the cost of the techniques and the benefit to the farmer of reducing emissions and retaining nitrogen as a fertilizer.

Concepts to reduce adverse environmental impacts depend on the *understanding at a process level* of the following:

- Assessment of emissions from naturally ventilated barns;
- Assessment of emissions from new, animal friendly housing systems;

- Development of abatement/mitigation measures especially for naturally ventilated dairy barns (e.g. targeted ventilation and air scrubber, manure acidification);
- Interaction between climate change and heat stress/animal behaviour/emissions;
- Interaction between low protein diets and N and GHG emissions;
- Interactions between N and GHG emissions during housing, storage and application to field;
- Life-cycle assessment: for example, grass-based dairy feeding versus low protein dairy feeding;
- Feed and manure additives for improved N use efficiency; and
- Manure treatment for higher N use efficiency (increase of nutrient availability, decrease of emissions) and potential of processing to recover manure N into bio-based fertilizers in a circular economy.

Concepts to reduce adverse environmental impacts depend on the development of *flexible concepts* for environmental improvement:

- Climate and site-specific conditions vary across the UNECE region and globally;
- All three columns of sustainability must be considered: economic, environmental and social sustainability;
- Conflicts of interest must be addressed; and
- Targeted approaches according the needs of different regions.

Concepts to reduce adverse environmental impacts depend on *effective communication and interaction*:

- Establishing networks to exchange manure management information, connect people, and forge partnerships;
- Launching an on-line knowledge hub – on best practices for livestock housing and manure management; and
- Establishing a roster of experts to provide targeted technical assistance and training, analysis and practical implementation and policy support, relying heavily on co-financing and in-kind resources from partners.

The development of best practice concepts is challenging. Climate and site-specific conditions are highly variable. It is essential to consider the three columns of sustainability: economy, environment and society and to address synergies and potential conflicts of interest. This inevitably leads to the conclusion that there will be no 'one size fits all solution'. Best-practice concepts provide the basis to guide on the development of flexible measures targeted for each specific region and context.

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Chapter 6

Developments in anaerobic digestion to optimize the use of livestock manure

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1 Introduction

For thousands of years, agricultural production has provided humans with the food needed for survival. However, the fact that agriculture can provide energy has received less attention, starting with burning firewood and crop straw to produce heat and energy. Non-renewable energy sources such as oil and coal are declining, which means alternative energy sources such as biomass energy are urgently needed. As a renewable resource, biomass energy is also environmental-friendly.

Anaerobic digestion (AD) is an important method for converting biomass into bioenergy. AD can use a range of substrates including kitchen waste, crop straw, energy crops, municipal solid waste, sewage sludge and livestock manure. Although biogas production from livestock and poultry manure cannot achieve zero carbon emissions, it does not produce the level of carbon

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emissions produced by fossil fuels. Livestock manure is still a widely used biomass material for production of bioenergy.

The production of biogas through AD of livestock manure is a complex process. It involves a variety of complex physiological and biochemical metabolic pathways, the essence of which is the material and energy metabolism of microorganisms under anaerobic conditions. AD is typically into three stages according to the utilization and transformation of organic matter (Fig. 1):

- hydrolysis;
- acidogenesis; and
- methanogenesis.

In the hydrolysis step, macromolecular organic matters (fat, carbohydrate, protein, etc.) are hydrolyzed into small molecules such as monosaccharides, amino acids, fatty acids and so on by the action of extracellular enzymes. Then the small molecule organic compounds are converted to a volatile organic acid, ethanol and so on by the acidified bacteria. H_2, CO_2 and acetic acid are then formed under the action of hydrogenic bacteria and acetogenic bacteria during this acidogenesis step. Finally, the methanogenic bacteria synthesize methane using acetic acid, H_2, CO_2 and so on in the methanogenesis stage.

It can be seen from the above that the AD process can produce a variety of substances, including H_2, CH_4, alcohol, lower fatty acids and so on. These substances can also be reused as biofuels, biodiesel, biogas, power generation and so on. The production of bioenergy by AD is thus very clean and efficient. There are many kinds of substrates in AD. Livestock and poultry manure is one of the commonly used raw materials, but a large amount can cause pollution, a key issue to address in its effective use.

The purpose of this chapter is to discuss developments in AD to optimize the use of livestock manure, particularly the use of livestock in the production of biogas. It also discusses the use of biogas slurry and residues. The chapter shows how AD can play an important role in promoting circular agriculture. It includes a case study on the use of AD in practice in Henan Province, China.

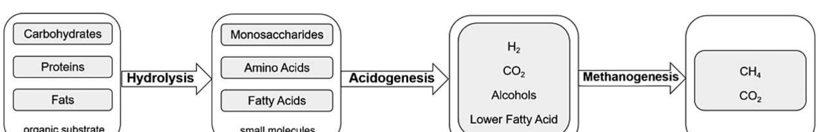

Figure 1 Steps of anaerobic digestion process.

2 Livestock manure: quantities and risks

Since ancient times, pigs, cattle, sheep and poultry have been kept by humans. However, it has not been possible to measure the amount of that feces produced by these livestock species exactly. Fortunately, the numbers of livestock and poultry raised in various countries and regions are now more accurately counted, so that the amount of manure can be estimated. Numbers of livestock in countries and regions are counted by the Food and Agriculture Organization of the United Nations (FAO), and can be found through its database (http://www.fao.org/home/en/; accessed November 10, 2019). A method for estimating the amount of manure by the quantity of livestock and poultry has been proposed using the following formula that is as follows (Gao and Li, 2015):

$$M_i = \sum_{i}^{n} Q_i \cdot d_i \cdot m_i \tag{1}$$

where Q_i is the total number of livestock, d_i is the excretion coefficient (the daily excretion of each animal) and m_i is the animal slaughter period.

This formula has been used to estimate the quantity of livestock manure produced in China (Lin et al., 2012; Li et al., 2010; Liu et al., 2005) (Fig. 2). The estimate for 2017 is up to 4 trillion tons of livestock manure that is produced

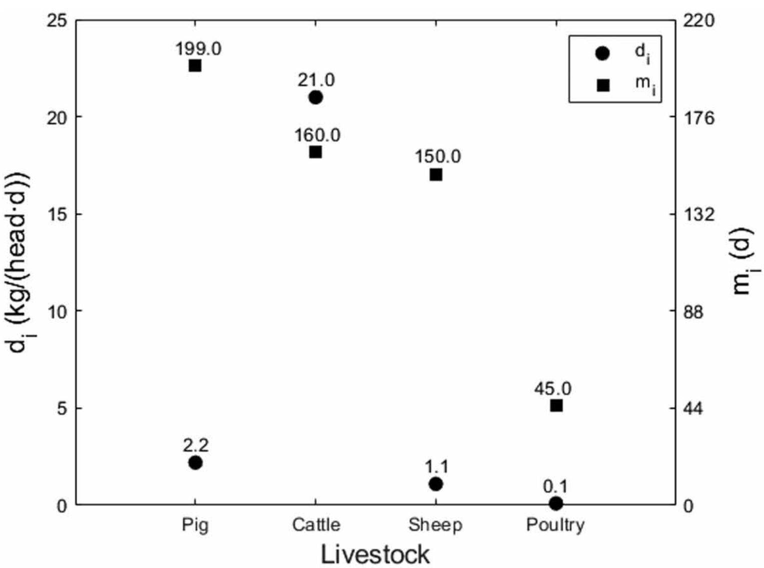

Figure 2 The estimated parameters of livestock manure quantity (Note: d_i is the excretion coefficient; m_i is the animal slaughter period).

in China. If such large amounts of manure were directly discharged without proper treatment, they would cause a huge pollution risk.

The first pollution consequence is air pollution. Livestock and poultry manure contain harmful gases such as hydrogen sulfide and amine, which can produce methyl mercaptan, dimethyl sulfide, methyl sulfide, dimethylamine and a variety of low-grade fatty acids or other odorous gases. If not treated in time, they cause a relatively low oxygen content in the air and increase the degree of turbidity (Liu et al., 2010). Moreover, greenhouse gases from livestock manure are also an important factor in global climate change. In addition to emitting CH_4, it is estimated that N_2O produced by livestock manure accounts for 30–50% of global agricultural N_2O emissions (Oenema et al., 2005). These harmful gases are caused by the unique environment of the feces and the large amount of microbial activity. It is striking that since the pre-industrial era, the concentration of CH_4 in the atmosphere has increased by 2.5 times and the concentration of N_2O has increased by about 20% (IPCC, 2006). Although this situation is caused by many factors, the treatment of livestock manure is a significant issue in controlling greenhouse gas (GHG) emissions (https://www.ipcc.ch/; accessed November 6, 2019).

The second consequence is water pollution. The effects of pollution of livestock and poultry manure to water bodies can be divided into three types:

- the release of acid gas into the air and then precipitation in the form of acid rain;
- direct discharge of pollution into surface water which flows into rivers and lakes; and
- entry into groundwater by osmosis.

Manure contains a lot of nitrogen and phosphorus. Once manure enters a water body and the quantity exceeds the water's purification ability, it causes algae to multiply in the water, and aquatic animals die of oxygen deficiency. Physical and chemical water properties will change followed by microbial composition, which finally makes the water toxic (Hooda et al., 2000).

The third consequence is soil pollution. Heavy metals are ingested through feed during livestock and poultry farming. Most are not used but excreted. When these manures are used as fertilizers, heavy metals including copper, zinc, cadmium, chromium, lead, mercury and arsenic will accumulate in the soil and crops, posing a potential risk to soil and human health (Feng et al., 2018b). In addition, when excessive sodium and potassium are in the feces, it will decrease soil micropores, destroy soil structure and reduce plant growth if applied directly to the soil. In addition, animal antibiotics are commonly used to treat and prevent animal diseases. Excessive antibiotics cannot be completely absorbed in animals but are excreted, accumulating in the soil and posing a significant environmental risk (Pu et al., 2018).

Finally, pollution from livestock also causes harm to human health. Air quality is reduced and the odor causes potentially fatal diseases of the respiratory system. Unclean drinking water systems are associated with waterborne disease outbreaks (Oun et al., 2014). Moreover, there are many pathogens in feces and, if humans eat plants grown in contaminated land, they are also likely to get sick. Thus, livestock manure can directly or indirectly affect human health. In summary, the large amount of livestock manure produced and unsuitable handling creates enormous environmental and health risks. Rational use of livestock and poultry manure to reduce these hazards and turn them into useful products is an important area of research.

3 The biogas potential of livestock manure

The biogas potential of a region refers to the amount of biogas produced by the waste in the region under appropriate conditions. Since not all waste is used for AD, and production conditions are not always optimal, biogas potential is an estimate under ideal conditions. Nevertheless, it is of great significance as a guide to the development of biogas production in a particular area.

Different types of animal manure have different capacities to produce biogas. The difference between them is called the biogas conversion coefficient of manure. Many studies have shown that livestock manure has moderate biogas potential (Nasir et al., 2012). Many factors involved in production processes also affect the production of biogas from livestock manure. Previous studies have suggested the following livestock manure biogas conversion parameter and formula for estimating manure biogas potential (MBP) (Yao, 1988; Zhang et al., 2012) (Fig. 3). Eq. (2) is one of the methods to calculate the biogas potential.

$$\mathrm{MBP} = \mathrm{M_i} \cdot \mathrm{b_i} \tag{2}$$

where M_i is the amount of livestock manure produced (kg) from livestock i, and b_i is the livestock manure biogas conversion parameter.

This method is a simple calculation but subjected to a high potential level of error. In order to make up for the shortcomings of this method, some scholars have proposed a new method to improve biogas potential estimation. Biogas production is affected by the total solids (TS) of livestock manure because the presence of manure and urine in the excrement means it cannot be used entirely to produce methane. The total solid coefficients (TSC) of the feces can be further calculated by Eq. (3) (Zhang et al., 2019):

$$\mathrm{TSC}(\%) = \frac{\mathrm{F_i} \cdot \mathrm{F_s} + \mathrm{U_i} \cdot \mathrm{U_s}}{\mathrm{F_i} + \mathrm{U_i}} \tag{3}$$

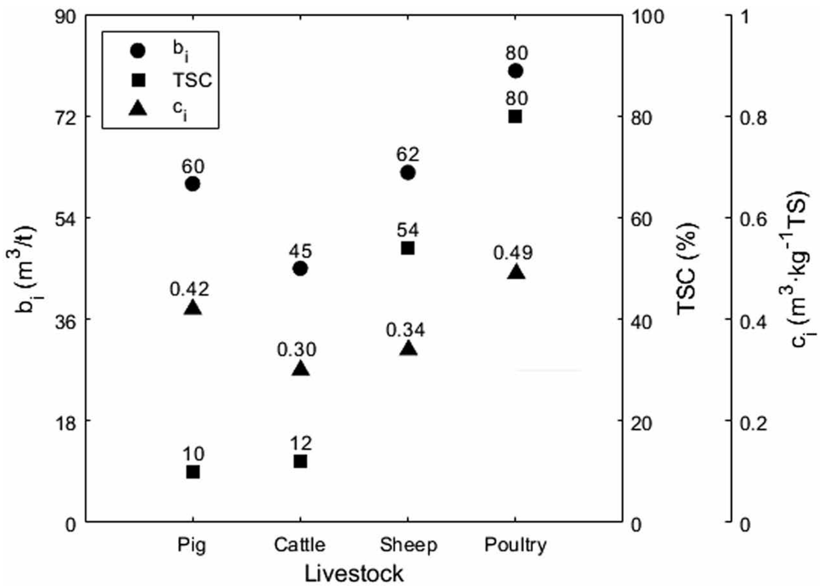

Figure 3 The estimated parameters of livestock manure biogas potential (Note: b_i is the livestock manure biogas conversion parameter. TSC is the total solid coefficients of livestock manure. c_i is raw biogas production rate per kg of dry matter).

where F_i represents the daily output of feces, F_s represents the total solid of the feces, U_i is the daily yield of urine and U_s is the urine's total solid.

Eq. (4) is used to calculate the biogas potential:

$$MBP = TSC \cdot M_i \cdot c_i \quad (4)$$

where M_i is the amount of livestock manure, c_i is the raw biogas production rate per kg of dry matter ($m^3 \cdot kg^{-1}$TS).

For reference, the TSC and c_i of some livestock have been listed in Fig. 3. It should be noted that, if the statistical data of animals is used to directly estimate the potential (where the amount of manure used for gasification, returning to the field and other industrial production has not been removed), the estimated value is relatively large.

4 Anaerobic mono-digestion and co-digestion

Depending on the different feeding modes of the substrate, the AD of biogas is divided into two types:

- anaerobic mono-digestion; and
- anaerobic co-digestion.

These are discussed in the following sections.

4.1 Anaerobic mono-digestion

At present a diverse range of livestock manure can be used in anaerobic mono-digestion raw material, in which feces from swine, cattle and chicken are richer. In addition, there are some AD that uses the droppings of ducks, sheep and rabbits (Song et al., 2010). The growth characteristics of cattle mean a large amount of excretion. Its abundance makes cattle manure one of the most commonly used raw materials for AD. Fodder is excreted after being digested through the rumen, and so cattle manure is rich in microorganisms, making cow dung a good raw material for AD. However, the content of cellulose and hemicellulose in cattle dung is high, and microorganisms are difficult to decompose and utilize, so there is still much space for improvement of biogas production from this source.

Swine manure is also a good AD substrate, with rich organic matter and high biogas yield. But the rapid hydrolysis caused by the large amount of carbohydrates can also have a negative effect. The accumulation of volatile fatty acids (VFA) in the system leads to acidification, imbalance of microbial growth and even digestion failure. Swine manure also contains ammonia, which is prone to ammonia inhibition. Chicken manure contains ammonia nitrogen which interferes with digestion because ammonia poisoning affects the performance of reactor. However, chicken feed is mainly based on grains and protein, and chicken manure has high nutrient content and high utilization value.

As this suggests, the performance of different manures in anaerobic mono-digestion varies because of their different characteristics. Gao and Li (2015) found methane yields for swine, cattle and chicken manure of 410, 270, 377 mL CH_4/g VS respectively, at initial volatile solid loading (VSL) of 8 g VS/L. Due to the variable presence of nutrients, anaerobic manure mono-digestion often faces problems such as poor stability, long lag time and low yield. However, anaerobic co-digestion can balance the carbon-nitrogen ratio (C/N) of the system to make microorganisms grow. It can also supplement the trace elements needed for microorganism growth and metabolism which further improves the fermentation effect.

4.2 Anaerobic co-digestion

Single manure substrate fermentation often suffers from C/N imbalance. A high C/N ratio results in a low protein dissolution rate. The total ammonia nitrogen (TAN) and free ammonia nitrogen (FA) concentrations are both low (Xue et al., 2019). However, an excessively high C/N ratio is not sufficient to maintain cell biomass, resulting in a reduction of biogas production. Substrates with a low

C/N ratio increase ammonia inhibition, which is noxious to methanogens and decreases underutilization of carbon sources (Mao et al., 2015). Optimizing the substrate C/N ratio by mixing suitable raw materials thus plays an important role in the AD process. The optimal C/N ratio for AD is between 20 and 35, with the most commonly used ratio at 25 (Yen and Brune, 2007; Zhang et al., 2013; Punal et al., 2000). It is worth mentioning that, whether anaerobic mono- or co-digestion is used, the optimal pH in systems is in the range of 6.8–7.4, because pH can directly affect microbial activity.

Mixing raw materials is an effective way to reduce the negative impact of anaerobic mono-digestion and expand the production of biogas. The mixing of raw materials for AD is not limited to feces, which can also extend to other substances like municipal solid waste, food waste, sewage sludge and crop straw. Through the mixing of substrates, nutrients such as C and N in the digestion system can be balanced, the microbial community in reactor can be enriched and the methane conversion efficiency of the organic substance can be improved. Studies have shown that the synergistic effect of mixed fermentation has a positive effect on the production of biogas.

Co-digestion of dairy manure, chicken manure and wheat straw performed better in methane production than digestion using a single substrate, with a maximum potential of CH_4 production predicted to be 394 mL/g VS (Wang et al., 2013). A synergistic effect was found when corn straw and chicken manure was mixed in proportions of 3:1, regardless of pretreatment (Feng et al., 2018a). The co-digestion of pig manure and dry maize straw is superior to the mono-digestion of pig manure, improving the C/N ration and enhancing the buffer capacity of the system (Song et al., 2016). When dairy manure was co-digested with food wastes, the highest cumulative biogas yield reached 459.4 mL/g VS (Batool et al., 2020). Livestock manure can also be co-digested with a variety of organic matter, such as the *Pennisetum* hybrid (Lianhua et al., 2020), mango leaves (Abudi et al., 2020) and animal carcasses (Tapparo et al., 2020). As these examples show, in general, mixed fermentation is better than single fermentation.

5 Factors affecting the efficiency of anaerobic digestion

There are many factors affecting AD which can be divided into the following categories:

- Substrate regulation;
- Raw material pretreatment;
- Process control; and
- Accelerators.

These are discussed in the following sections.

5.1 Substrate regulation

Each kind of animal excrement has unique physicochemical characteristics. Substrate regulation includes factors such as substrate concentration, C/N ratio and inoculum substrate ratio. The digestion system can be affected by changing these conditions. The fact that different livestock and poultry manures produce different biogas yields is one example of the way substrate properties and their combinations influence AD. The effect of inoculum on AD is also multifaceted. It can increase the number of microorganisms and ensure a suitable pH, but also accelerates the start of fermentation and shorten the lag phase. Selecting the proportion of inoculation to match the properties of the raw materials is important for optimizing AD.

5.2 Raw material pretreatment

The importance of pretreatment in substrate regulation needs to be emphasized. Low biodegradability usually occurs because the substrate contains something which is difficult to degrade, for example, the undigested plant fiber in cow dung (Sawatdeenarunat et al., 2016). Pretreatment of the substrate before fermentation is a good solution to this problem (Mosier et al., 2005). Different pretreatments have different effects on substrates.

Physical pretreatment mainly consists of mechanical pulverization and grinding to change the external morphology or internal structure of the raw material in order to improve biodegradability and biogas production (Elliott and Mahmood, 2012; Carrere et al., 2010). Larger granules of substrate reduce the degradation rate of chemical oxygen demand (COD) in the fermentation process, thereby reducing biogas production (Esposito et al., 2011). Chemical pretreatment mainly uses chemical reagents such as acid, alkali or oxidizing agent. Alkali reagents like ammonium hydroxide and sodium hydroxide are often used as alkaline pretreatment. Wahid et al. (2020) have shown that mixing wheat straw, solid cattle manure and solid slaughterhouse carcass material can improve anaerobic biodegradability after alkali and ultrasonic pretreatment. Acetic acid and organic acids provide the desired organic acid components to some microorganisms. Hydrogen peroxide and especially ozone are common oxidants, which avoid excessive accumulation of intermediate products in the digestion process and eliminate pathogens (Kianmehr et al., 2010). However, oxidant pretreatment is relatively costly. Other pretreatments include thermal pretreatment (Kaar and Holtzapple, 1998) and biological pretreatment (Zhong et al., 2011). All pretreatment methods have advantages and disadvantages. Various factors such as likely costs and benefits need to be considered, after which a single pretreatment can be chosen or combined to achieve the desired effect.

5.3 Process control

Digestion temperature, pH, hydraulic retention time (HRT) and organic loading rate (OLR) are all factors that can be regulated during the fermentation process. Some of these factors are discussed in more detail below. In addition, factors such as the type of reactor (Fig. 4), management, construction time and power generation equipment will impact the generation and utilization of biogas.

The usual temperature for thermophilic AD is 55°C and 35°C for mesophilic digestion. Compared to mesophilic AD, thermophilic AD has a fast response speed and high carrying capacity, resulting in higher yield. But it also has shortcomings, including lower system stability, acidification, more polluting discharge, higher net energy input and more investment, given the temperature requirements (Bowen et al., 2014). In actual production, many biogas digesters on farms depend on seasonal temperature without additional heating, reducing process stability and gas production due to ambient temperature changes.

pH directly affects the digestion process. The growth rate of microorganisms is very sensitive to changes in pH. The relative abundance of microorganism species is 6 when pH is 4.0 and increases to 14 when pH rises to 7.0 (Fang and Liu, 2002). Suitable pH ranges for methanogenic and acidogenic microorganisms are different. At pH 6.5–8.2, the efficiency of methanogenesis is optimal with the best pH at 7.0 (Lee et al., 2009). Methanogen growth slowed down visibly when the pH is less than 6.6

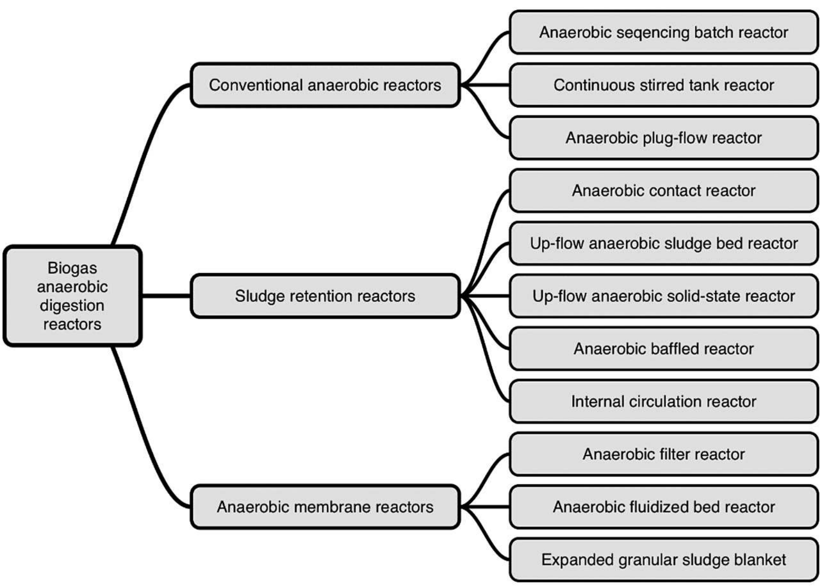

Figure 4 Types of biogas anaerobic digestion reactors.

(Zhang et al., 2009b). For acidogenesis, 5.5–6.5 is the most suitable pH range (Kim et al., 2003). Separating AD into a two-stage process is the preferred mode of operation with separate hydrolysis/acidification and acetogenesis/methanogenesis stages.

The time required for the decomposition of organic matter is defined as the retention time. Decomposition is affected by microbial growth rate, the process temperature, OLR and the matrix substance (Aboudi et al., 2015). The quantity of volatile solids in the digester is represented by OLR. Biogas production will improve to some extent with higher OLR (Mao et al., 2015).

5.4 Accelerators

AD additives can be divided into inhibitors and accelerators. Artificial inhibitors are not generally used in most biogas production processes, and so this section focuses on the use of accelerators. Accelerators can be further subdivided into biological accelerators and inorganic chemical accelerators.

Biological accelerators include several kinds of fungal microbial agents, which are mainly used for the treatment process of lignocellulosic biomass (Zheng et al., 2014). These include rumen fungi and protozoa (Zheng et al., 2019; Yildirim et al., 2017). Other biological accelerators include functional enzymes, for example, extracted from 3-day cultures of *T. harzianum* (Enzyme T) and *Aspergillus* spp. to improve methane yield (Zhao et al., 2018). But the cost of enzyme preparation is high, limiting its use.

Inorganic chemical accelerators are widely used, including acids, bases, inorganic salts and elements such as N, P, K and S. These substances are essential nutrients for the growth and metabolism of microorganisms that can be directly utilized in the AD process with significant effects (Weiland, 2010). However, inappropriate doses of these elements may result in reduced system stability. Trace elements such as Fe, Ni, Mg and Ca can also promote biochemical reactions (Demirel and Scherer, 2011). However, if too acidic or alkaline, and not utilized by microorganismss, these accelerators may affect the stability of the process.

6 Products from biogas digestate

During the AD process, livestock and poultry manure are decomposed through the action of microorganisms into biogas slurry containing proteins, amino acids and other water-soluble substances. Substances that cannot be completely decomposed and other impurities are deposited at the bottom of the reactor by gravity and become biogas residue after drying. The effluent of many biogas digesters is a solid-liquid mixture referred to as biogas digestate.

6.1 Digestate value-added products

The components of biogas slurry are complex, including nutrients such as N, P and K, as well as trace elements like Fe, Zn, Mn and Mo, which can be utilized by animals and plants. Microorganisms, antibiotics, enzymes and other unknown substances may also be present in the slurry. This means that the reuse value of biogas slurry is very high. In addition, the use of digestate to produce value-added products can reduce odor by up to 80% and make positive changes in odor composition (Weiland, 2010). The AD process is also able to partially inactivate weed seeds, bacteria, viruses, fungi and parasites in feedstock from livestock manure, improving safety (Mata-Alvarez et al., 2000).

Biogas slurry and biogas residue can be used directly or processed to produce value-added products such as biogas fertilizers and soil improvers (Nkoa, 2014; Edwards and Daniel, 1992) and animal feed (Zhang et al., 2008). Because biogas digestate can contain plant hormones including heteroauxin, gibberellins (GA4, GA19, GA53), plant growth agents can be produced (Huo et al., 2011). Biogas residue can also produce bedding materials for animal housing.

Biogas digestate is widely used as a base fertilizer, foliar fertilizer, top dressing and in soilless cultivation. As an example, the recommended application rate of biogas slurry on coastal-reclaimed farmland is 480 and 9.00–11.25 m^3 ha^{-1} for rice and wheat, respectively (Tang et al., 2019). The optimal proportion of chemical fertilizer nitrogen substitution by biogas slurry was 70% with a biogas slurry application rate of 278.56×10^3 kg hm^{-2} (Wang et al., 2018). Digestate as a fertilizer has a positive function on plant biomass (Barbosa et al., 2014). Digestate promotes the activities of various enzymes such as nitrate reductase and glutamine synthetase (Jabeen and Ahmad, 2017). It has also been shown to improve yield (Holm and Heinsoo, 2013, 2014). The use of digestate also promotes recycling (Tampio et al., 2016). Indirect effects include promotion of soil microbial activity (Hupfauf et al., 2016). Biogas fertilizer also changes soil composition through processes such as ammonia and nitrogen oxide emissions (Nkoa, 2014), nitrate leaching potential (Riva et al., 2016), transformation of organic carbon and GHG emissions (Knudsen et al., 2014), as well as leaching and precipitation of phosphorus. These cause changes in elements such as Mg, Fe, Mn and so on (Zirkler et al., 2014). Biogas digestate also increases pH, electrical conductivity, air porosity and bulk density (Meng et al., 2018). In addition to biogas fertilizer, biogas slurry can also promote germination and seedling by soaking of seeds (Ni et al., 2015).

Biogas slurry can also be used for aquaculture and livestock feed. Biogas residue as an alternative feed saves feed costs and improves digestion without affecting pork quality (Zhang et al., 2009a). In aquaculture, biogas digestate nutrients can be eaten by fish or promote the growth of plankton as a source of nutrition.

6.2 Disadvantages of biogas digestate

There are some shortcomings in using biogas digestate. Harmful substances such as heavy metals may still be present in biogas fertilizer and will still contaminate the soil (Duan et al., 2012). The bioavailability of heavy metals in AD is an important criterion to predict the ecological risk of heavy metals, which is determined by the form of each metal (Thanh et al., 2016). AD operational parameters (pH, redox potential and VFAs concentration) determine bioavailability (Knoop et al., 2017). Heavy metals are not biodegradable and can accumulate to potentially toxic concentrations as digestion proceeds (Chen et al., 2014). In the process of AD, S^{2-}, CO_3^{2-}, PO_4^{3-} and so on will cause reactions and precipitation will change the solubility and availability of heavy metals (Insam et al., 2015). The toxic effects of heavy metals depend on their solubility and availability (Choong et al., 2016). Vegetables and grains produced after irrigation with heavy metals such as zinc, lead and cadmium are harmful to human health (Bian et al., 2015). Although the maximum allowable concentrations of heavy metals in the digestate are currently set in many countries, they still present an environmental risk (Gusiatin and Kulikowska, 2014). However, proper management can reduce this risk.

7 Ecological agriculture models for biogas utilization

Agriculture needs to reduce its environmental impact by adopting ecological principles such as reducing waste and creating circular, more self-sufficient systems which recycle waste into useful by-products such as fertilizer and biogas which can then be fed back into the system. This section introduces four typical ecological agriculture models: ecological farms, ecological orchards, ecological greenhouses and 'five supporting' mode.

7.1 Ecological farms

Ecological farms are a widely used model. They can be divided into two types:

- Small-scale household farms; and
- Large-scale 'engineering' farms.

The household ecological farm consists of a single household or several neighboring households on a small scale. Processes are simple: human feces and livestock manure are added to a fermentation tank to produce biogas for human. The 'engineering' type is mainly for farms that cultivate animals and crops on a large scale. Biogas digesters are constructed to eliminate agricultural waste such as straw and manure. Equipment and capacity of biogas digester are usually well developed. Biogas is not only used for heating but also for generating electricity which can be used on the farm with any surplus supplied into the local power grid to generate additional value too.

7.2 The ecological orchard and other three-in-one models

The ecological orchard model is one example of a circular agricultural system. The three-in-one model (Fig. 5) combines animals, microorganisms and plants. First, animals can eat plant stalks, weeds, fruits and so on to grow. Secondly, biogas is produced by AD when animal manure is fed to biogas digesters to produce biogas for heat and power. Thirdly, biogas slurry can be applied to the plants as organic fertilizer to stimulate plant growth, and biogas residue can also be used as animal feed. This creates a circular, more self-sufficient system with fewer inputs and outputs. Plants can be orchard trees, arable crops, vegetables, tea and so on. Animals can be livestock, poultry and aquatic species. Typical examples of this three-in-one model are the 'biogas-rice-duck', 'chicken-biogas-fruit' and 'pig-biogas-fish' models (Chen, 1997; Yang et al., 2018; Yang et al., 2012). Successful models of this type save feed and energy costs, increase the variety and quantity of products produced, and can also improve product quality while reducing environmental impact.

7.3 Ecological greenhouse: the four-in-one model

Temperature is a key limiting factor for the production of biogas. Both ecological farms and ecological orchard models can use their own production of biogas as an energy source for heating. However, in some areas, the winter is cold and low temperatures prevent the fermentation tank from operating properly (Pham et al., 2014). Adding thermal insulation can be expensive. For the circular agriculture model to operate properly and economically, a four-in-one model has emerged. This model has a greenhouse in addition to biogas digesters and provides an effective ecological cycle from livestock feeding to

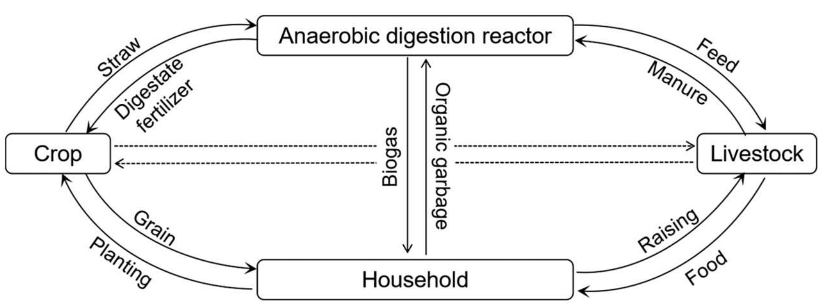

Figure 5 Eco-agriculture model of biogas anaerobic digestion.

vegetable production, resulting in a higher conversion efficiency in nutrient cycles and energy flows (Qi et al., 2005). An example is combined pig and vegetable greenhouse production, by combining a plastic solar greenhouse, an underground 6–10 m^3 biogas digester, a pigsty of about 10–20 m^2 and toilet facilities, parallel to the vegetable greenhouse. This forms an energy ecosystem (Fig. 6).

This model has the following advantages:

1 Due to the presence of the greenhouse, the biogas digesters are warmed up by the solar energy, solving the problem of gas production in cold winter temperatures.
2 The temperature of the pigsty can also be increased by 3–5°C in winter, providing suitable growth conditions for pigs.
3 The toilet (for human personnel) and pigsty provide sufficient fermentation feed for the biogas tank.
4 The pigs exhale a large amount of CO_2, and increase CO_2 concentration in the greenhouse by a factor of 4–5 which can greatly improve vegetable growth conditions in the greenhouse, and increase yield and quality.

7.4 Five-in-one model

The four-in-one model has improved ecological agricultural production in cold areas, but what kind of ecological agricultural model should be developed in areas of water shortage? As an example, the northwestern region of China has sufficient sunshine and a large temperature variation between day and night

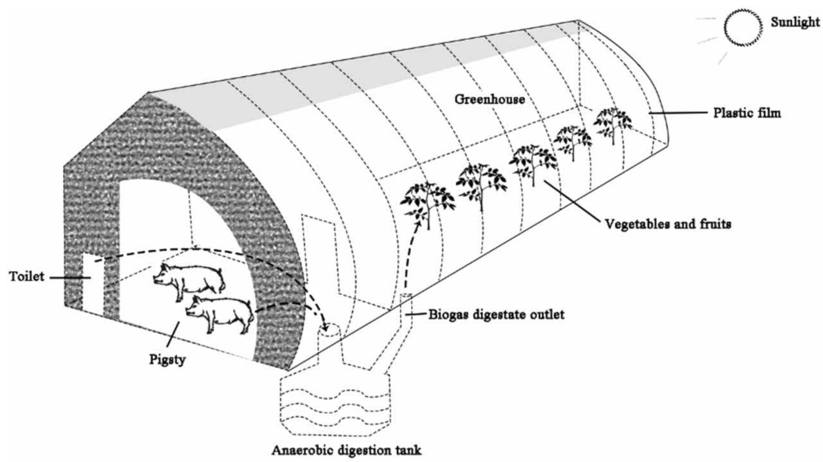

Figure 6 Structure of ‘four-in-one’ eco-agriculture model.

which could favor planting high-quality fruit and cash crops. However, the regional climate is arid and semi-arid climates with problems such as drought and soil erosion due to lack of water. The five-in-one model incorporates the components of the four-in-one system but adds water storage capacity (to capture the available natural precipitation) to enable drip irrigation (Qiu et al., 2001; Wu et al., 2015). The biogas slurry can, for example, be applied to the roots of fruit trees using drip irrigation. As with the four-in-one model, each component supports the others.

8 Case study: biogas production in Henan Province, China

China is a country with abundant resources and well-developed agriculture, including the production of large volumes of agricultural, forestry and domestic waste as well as sewage sludge. Bio-gasification is a promising way of recycling this large amount of waste. Henan is the country's major agricultural province. In 2014, the output of major cereal and meat products in Henan Province reached 55.227 million tons and 69.91 million tons, respectively, as recorded by the Henan Statistical Yearbook (http://www.ha.stats.gov.cn/hntj/lib/tjnj/2015/indexch.htm; accessed November 8th 2019). Henan Province also has a large number of biogas projects combining the development of biogas and agriculture.

8.1 Biogas potential of Henan Province

The 2014 China Statistical Yearbook shows the number of pigs, cattle, sheep, chickens and ducks raised in Henan Province during the year. The total number of these animals reached 959.62 million. The amount of cattle manure and pig manure produced is 88.77 million ton and 61.68million ton, respectively, sheep manure is 11.12 million ton and poultry is 4.05 million ton, respectively. Figure 7 shows the numbers of sheep, cattle, pigs and poultry, the volume of manure and MBP. It can be seen that biogas potential and fecal volume do not increase with the number of animals in a linear way. Although poultry is the largest group, accounting for 84% of the total, the biogas potential is the smallest, accounting for only 4% at 324 million m^3. Biogas potential also corresponds to the kinds of manure produced: the MBP of cattle, pig, sheep and poultry reached 3.99 billion m^3, 3.70 billion m^3, 0.68 billion m^3 and 0.32 billion m^3, respectively. This shows that agricultural waste resources have great potential to supply energy via biogas. However, the biogas production of current biogas projects is far below the potential value (Gao et al., 2019). This may be due to geographical conditions, technical equipment, management methods, and policy and economic constraints. However, this estimate shows the biogas potential of agricultural waste.

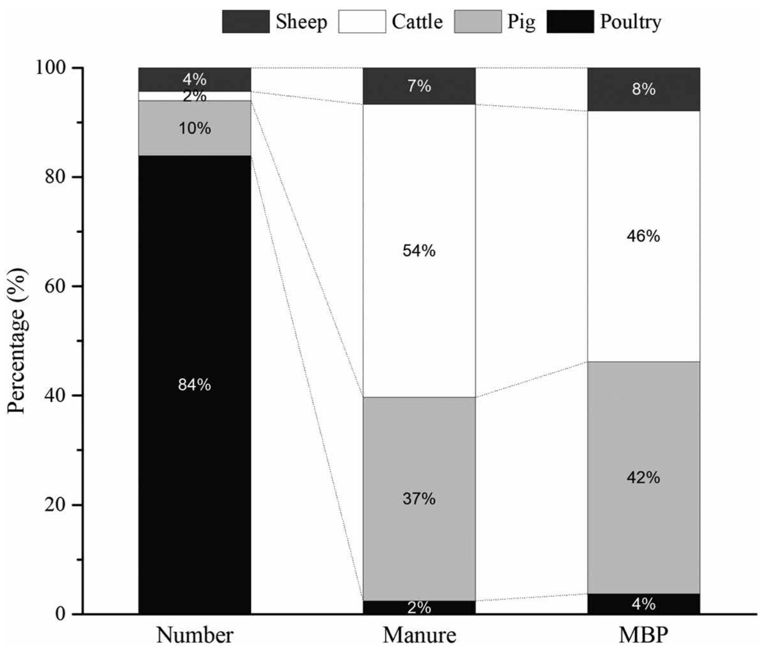

Figure 7 Livestock number, manure and MBP volume of Henan Province in 2014.

8.2 Eco-agriculture model

Henan Province has pioneered an eco-agriculture model called '100 mu-1000 heads ecological quadrant' (mu is a unit of area of about 0.0667 ha). It consists of a 3 mu livestock production line and 100 mu of farmland as a unit with 1000 pigs produced. Pig manure is digested and turned into organic fertilizer to use on the farmland, and the feed grain is processed into organic grain in situ. Such a 'pig-fertilizer-grain' recycling production line has the characteristics of low investment, rapid results and environmental benefits. It not only solves the problem of pollution in pig production, but also helps farmers out of poverty.

There are a number of requirements needed to develop this model. First, the government must improve policies related to biogas. Good policy guidelines can lead, supervise, guarantee, promote and support high-quality and sustainable production of biogas locally. High-quality biogas projects are economically and ecologically beneficial while supplying large amounts of bioenergy. Policies are an effective guarantee to achieve these benefits. The types of policies should include guidance, incentives, research, measure to promote markets, as well as standards governing constructive and operation (Feng et al., 2012).

Second, biogas companies must be responsive to policies and regional conditions, particularly in developing technologies that match local conditions. A key aspect is the intelligent control technology to improve the gas production rate of AD. By continuously monitoring and adjusting the actual fermentation environment, it is possible to find the best substrate and conditions for AD. Selecting a fermentation device to suit local conditions can improve resource utilization and biogas production efficiency. There is also a need for supporting infrastructure and maintenance capabilities to support biogas power generation, including skills training (Gao et al., 2019b).

Finally, each region needs to adapt biogas production to local resources and conditions. As an example, it might select crops which benefit most from local conditions while selecting livestock with the greatest regional market potential. Establishing successful, long-running biogas engineering enterprises are the path to promote large-scale, professional development of biogas. Circular agriculture and ecological agriculture models provide the basis for fully utilizing biogas resources, for example, by integration with solar greenhouses and water-saving irrigation systems, and combining biogas production with crop and aquaculture production to achieve sustainable biogas production. These models provide a path toward a sustainable future for agriculture.

9 Summary and future trends

Livestock and poultry manure is a huge potential problem which exposes the environment to pollution and other risks. However, as an organic substance, livestock manure has huge biogas potential in the production of clean energy in the face of energy shortages and environmental change. Developments in biogas production technology have improved the utilization of animal manure through AD, increasing the economic and environmental benefits. However, at present biogas production from animal manure is below its potential. Areas for further study include:

1. Establishing a more accurate biogas estimation method, for combining various wastes with actual research data.
2. Innovating high-efficiency, low-cost biogas production technology and equipment.
3. Developing a variety of eco-agricultural models suitable for different regions to find more efficient ways of using animal manure.
4. Developing evaluation methods to determine the most scientific and effective way to use manure in different areas.

Advances in these and other areas offer the opportunity for a bright future for this sector.

10 Where to look for further information

Further reading:

- A website that can provide information on the size of livestock and poultry farming in various countries and regions around the world is the Food and Agriculture Organization of the United Nations (FAO) database available at: (http://www.fao.org/home/en/).
- A good way to assess the energy distribution and utilization of different countries and regions is the BP Statistical Review of World Energy available at: (https://www.bp.com/en/global/corporate.html).
- *A Comprehensive Review of Anaerobic Digestion Is Review on Research Achievements of Biogas from Anaerobic Digestion by Mao Chunlan* (2015).
- *The Study of Biogas Potential Estimation Is Biogas Potential, Utilization and Countermeasures in Agricultural Provinces by Gao Mingxue* (2019).

Key journals/conferences:

- *The World Energy Congress Organized by the World Energy Council (WEC)*, Is the Most Important Meeting for the Development of the Energy Industry.
- *Renewable and Sustainable Energy Review* is a journal that contains reviews of renewable and sustainable energy.
- *Bioresource Technology* publishes articles about bioresource fundamentals, applications and management, including biogas.

11 Acknowledgements

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Part 3

Nutrition

Chapter 7

The impact of improving feed efficiency on the environmental impact of livestock production

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1 Introduction

As ruminants, cattle are marvellous bioreactors that, through symbiotic rumen fermentation, convert cellulosic plant biomass and other organic materials inedible to humans into high-quality animal proteins for human nutrition. Nevertheless, the conversion is of course not 100% efficient, and so varying quantities of waste products such as carbon dioxide (CO_2), methane (CH_4), and reactive nitrogenous compounds are emitted. As such, producers of ruminant livestock must strive to maximise output for each unit of input, both to enhance enterprise profitability and to minimise the environmental impacts of dairy and meat production. A key metric of this system efficiency is feed conversion efficiency (FCE), which for milk production is usually defined as energy-corrected

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milk divided by feed dry matter intake (DMI) and for meat production is live weight gain divided by feed DMI. FCE per se also has a genetic component, which can be measured by residual feed intake (RFI). RFI is defined as the actual intake minus the feed intake expected to meet requirements for milk production, growth, reproduction and maintenance (Koch et al., 1963).

FCE has been widely used in beef production, as well as in pork and poultry production, to monitor the efficiency of feed utilization for growth. The dairy industry also recognizes the importance of the metric in management systems, but in addition to milk yield, there is also a need to account for body tissue loss and gain in calculating the efficiency of a lactating dairy cow (VandeHaar, 1998). Maximising the output of saleable product per unit of resource input is a standard principle of all manufacturing industries that relate directly to profitability. Another way of stating this relationship is that producers must minimise their unit cost of product and optimise their total unit output (Colman et al., 2011).

Relative to the reduction of greenhouse gases and contaminants of water, the simple concept is that the more carbon and nitrogen (N) in feedstuffs captured in the product, the less carbon and N are available for conversion into waste products (e.g. CO_2, CH_4 or urea N). By this principle, increasing milk or meat output from the same feed input requires changes in digestibility or postabsorptive nutrient metabolism with the result that less greenhouse gases and other waste products are produced per unit of milk or meat. The same principles apply to phosphorus and other nutrients that may become pollutants when they escape the animal through feaces or urine. This chapter will focus on the efficiency of milk production by dairy cattle related to nutrition and genetics focussing on how improving FCE can decrease the greenhouse gas burden of milk production and how FCE can be improved.

2 Greenhouse gases and dairy production

In the United States during 2008, the dairy sector's contribution to greenhouse gas emissions was estimated to be 134 Tg CO_2 equivalents, which equated to 1.9% of the total U.S. output (Thoma et al., 2013). Of this, CH_4, nitrous oxide (N_2O), and CO_2 contributed 44%, 13%, and 41%, respectively, of total emissions of the sector (Thoma et al., 2013). Emissions of CH_4 attributed to the dairy sector are primarily enteric emissions arising from digestive tract fermentation and stored manure, whereas N_2O emissions are largely attributable to N fertiliser application for the production of feedstuffs and manure application to farm land, including direct deposition by grazing livestock (Uweze et al., 2020; Rotz, 2018). In contrast, direct emissions of CO_2 arising from rumen fermentation and animal metabolism are not considered, as these emissions are a consequence of the digestion and metabolism of plant material incorporating atmospheric

CO_2 captured by photosynthesis. These estimates of the contribution of dairy production systems to global greenhouse gas emissions have been revisited recently because of the shorter half-life of CH_4 relative to CO_2. Methane from ruminants is also derived from the digestion of plant material, and over time CH_4 in the atmosphere is converted to CO_2, which reduces the long-term global warming effect of enteric CH_4 (Cain et al., 2019). In light of these considerations, dairy's contributions to CO_2 and N_2O emissions through fossil fuel consumption, fertilizer production and use, as well as manure management, are a greater concern for global warming in the longer term.

While the dairy sector is a small contributor relative to other industries such as oil and gas, the industry faces pressure to decrease greenhouse gas output. From the viewpoint of dairy producers, this pressure should not be viewed necessarily as burdensome because methane amounts to a 3.8–7.4% (5.6% on average) loss of gross energy intake from feeds (Kebreab et al., 2008). Mitigating greenhouse gas emissions should help not only the environment but also the financial bottom line of dairy production, unless the costs of mitigation become excessive. This is true especially if the feed energy not emitted as methane is captured as additional milk or body tissue (Reynolds et al., 2011). However, in most studies to date decreases in methane emission resulting from feeding methane inhibitory compounds such as 3-nitrooxypropanol (Reynolds et al., 2014; Hristov et al., 2015) or nitrate (Olijhoek et al., 2016) have not been associated with increases in milk energy yield or body energy balance. In this case, a part of the energy not emitted as methane is emitted as hydrogen, but the fate of the remainder of the methane energy not emitted is not certain (Olijhoek et al., 2016). In contrast, dietary strategies such as supplemental fat consistently decrease methane per kg feed DMI and often increase milk yield (Beauchemin and Grainger, 2011), but in this case, the effects on milk yield are through mechanisms independent from the effects on methane emission.

An issue with striving to decrease methane emissions is a decrease relative to what – in other words, what is the denominator of the equation? A reduction in total methane produced by the dairy sector will be difficult to achieve without reductions in animal numbers or effective mitigation strategies (e.g., methane-inhibiting feed supplements). Reductions in methane formation per unit of feed dry matter consumed (methane yield) may require shifts in the microbial community's overall metabolism or methanogenic enzyme inhibition through feed additives such as nitrate (Olijhoek et al., 2016) or 3-nitrooxypropanol (Hristov et al., 2015) as discussed above. Reductions in methane yield can also be achieved by feeding supplemental fat, which provides dietary energy that is not fermented in the rumen to yield hydrogen for methane synthesis (Grainger and Beauchemin, 2011). In contrast, decreases in methane production per unit of milk produced (methane intensity) are possible by boosting production efficiency (milk per unit of feed DMI). Much of this benefit arises from a dilution of the greenhouse gas production associated with the maintenance intake of the animal (Knapp et al., 2014). Cows require a certain amount of feed nutrients to maintain critical life functions even in the absence of milk production; think of it as the 'overhead' digestion and metabolism needed to sustain the basic functions of cows. Increasing the amount of milk produced per unit of additional feed intake thus serves to dilute the amount of maintenance greenhouse gas production over a larger number of milk production units. Other ways that improving feed efficiency may decrease methane output per unit of milk produced relate to increasing rates of nutrient passage through the rumen, shifting site of digestion to the intestine, feeding supplemental fat, or reducing heat production by the animal by alterations in metabolism that result in greater milk yield (discussed in a later section).

3 Origin of methane and reactive nitrogen excretions

In ruminants, enteric methane is mostly produced by the reduction of CO_2 in the rumen and hindgut. Greenhouse gas (primarily methane) output by dairy cattle represents 24.2% of enteric emissions from livestock (USEPA, 2021). The majority of methane production (ca. 85%) occurs in the reticulorumen of cattle, with normally only 13% being produced in the lower gut and rectal emissions constituting only 2–3% of total animal emissions (Murray et al., 1976; Munoz et al., 2012). Thus, factors relating to feed quality, feed consumption, feed degradation, and ruminal metabolism are paramount for the determination of methane output by dairy cattle.

The rich and extensive microbiota in the rumen constantly degrades complex carbohydrates in the diet (cellulose, hemicellulose, pectin, and starch) and metabolizes the constituent monosaccharides to the principal short-chain or volatile fatty acids (VFA), which are acetate, propionate, and butyrate. Conversion of monosaccharides to VFA generates ATP that can be used to drive nucleic acid and protein biosynthesis, that is, the principal components of new microbial cells. Degradation of feed protein and nonprotein N, including urea N recycled to the rumen, provides substrates (NH_3, amino acids, peptides) for microbial amino acid and protein synthesis. Ammonia N not captured as microbial protein is absorbed, converted to urea N in the liver, and either recycled to the gastrointestinal tract or excreted in the urine. Urea recycled to the rumen and hindgut is degraded by microbial urease to CO_2 and NH_3, and the NH_3 is either used for microbial protein synthesis or is reabsorbed. Any microbial protein synthesized in the hindgut is excreted as faeces, and microbial protein can account for more than 50% of faecal non-ammonia N of lactating dairy cattle (Larsen et al., 2001). Urea N in urine is very reactive and can be quickly volatilised as NH_3, especially when urine is mixed with faeces containing microbial urease. Faecal protein N is less reactive than urinary urea N, but as noted above, can contribute to N_2O and nitrate losses to the environment, depending on manure management practice.

Methane production serves a critical role within the microbial community. Production of acetate from fibre fermentation is associated with the co-production of CO_2 and H_2. The partial pressure of H_2 must be kept very low in the rumen to avoid poisoning the fermentation, and the primary means of doing so is the conversion of H_2 and CO_2 to CH_4 and H_2O (Hungate, 1966). The organisms responsible belong to the Archaea, but changes in Archaeal populations correlate only weakly with methane production (Tapio et al., 2017). Rather, methane production is related more to the populations of hydrogen-producing bacteria (Wallace et al., 2017). Production of propionate is not associated with the production of CO_2 or H_2; in fact, propionate uses hydrogen reducing equivalents in its synthesis. Although butyrate production also produces CO_2, it uses hydrogen reducing equivalents, so methane formation is not necessary (Hungate, 1966).

A fundamental relationship exists then in the fermentative breakdown of dietary components within the rumen, which can be appreciated by writing a simplified (non-chemically balanced) equation for rumen fermentation:

$$\text{dietary C and N} \rightarrow \text{microbial biomass} + \text{VFA} + CO_2 + CH_4 + H_2O + NH_3 + \text{heat} \quad (1)$$

From the standpoint of microbial growth efficiency, microbes strive to maximise biomass growth at a minimum VFA production, thus capturing as much of the starting feed substrate in new microbial cells as possible. The mixed rumen microbial population is able to generate 3–4 moles of ATP per mole of glucose fermented, which contrasts with simple monoculture fermentations that may generate only 2 moles of ATP (Hungate, 1966). Therefore, as rumen nutritionists, we understand that maximising microbial efficiency in the rumen ultimately leads to improved efficiency of milk production. Another way to think of this relationship is that the more dietary C and N we can keep in VFA and microbial protein, the less CH_4 and NO_2 are released into the environment. Similarly, the more efficiently dietary protein can be converted to milk protein through microbial protein synthesis and the metabolism of absorbed amino acids, the less excess N is excreted in urine and faeces. This idea forms the basis for why improving FCE decreases methane loss and the total greenhouse gas emissions attributable to dairy production.

4 Feed conversion efficiency

Improvements in milk production over the last several decades provide a clear demonstration of the benefits of improved FCE. Increases in FCE have arisen from increases in milk production, which dilutes the proportion of feed used for maintenance (VandeHaar and St-Pierre, 2006). Milk production has increased from intense genetic selection, improved nutrition and cow health, and other advances in modern management techniques. The improved FCE has benefited dairy producers' profitability, decreased the environmental impact of milk production, reduced the amount of land required for milk production, and decreased the greenhouse gas emission per unit of milk produced (Knapp et al., 2014). Continued gains in FCE by these methods will diminish, however, because the effect diminishes with each increment of dilution of maintenance (VandeHaar, 1998; Reynolds et al., 2011; VandeHaar et al., 2016).

Many environmental factors also affect FCE, including nutrition. Diet digestibility, or more specifically forage digestibility, is a major component of FCE. Dietary protein and carbohydrate balance, supplemental fats and various additives may also impact FCE through effects on digestive function and milk energy yield. Other factors include the stage of lactation, body reserve changes, physical activity, mastitis or other diseases, acidosis and heat and cold stress. From a dairy system standpoint, the total FCE of the farm unit is impacted by numerous factors, including calving interval, days dry, age of first calving, reproductive efficiency, periparturient health disorders, and death loss in calves and heifers. These latter factors centre on the effects of decreased productive days of life for those animals that are not producing milk, or not producing large amounts of milk, but continue to consume the feed.

Herd and Arthur (2009) reviewed the biological basis for differences in RFI for beef cattle. Possible components included feed intake, digestion, metabolism associated with fat and protein deposition, physical activity, thermoregulation, and other components of basal metabolism. As reviewed by Reynolds et al. (2011), digestible energy is the most variable proportion of gross energy intake in ruminants. It accounted for 86% of the variation in dietary net energy for lactation in dairy cattle in the calorimetry studies on which the initial NEmilk system for energy requirements of lactating dairy cows in the USA was based (Moe et al., 1972). This largely reflects differences in forage quality and the digestibility of fibre, which is affected by forage maturity at harvest and processing during conservation and feeding, as well as associative effects of other diet components that impact the rate and extent of fibrolytic activity in the rumen.

Beever and Drackley (2013) calculated methane production per kilogram of milk production as a function of increasing milk production efficiency and found a negative relationship between the two (Fig. 1). That this theoretical calculation was borne out in practice was demonstrated by an analysis of 323 measurements of methane emission by lactating dairy cows at the University of Reading (Fig. 2). When expressed on the basis of MJ of methane emitted per MJ of milk energy production, there was a reduction in methane emission

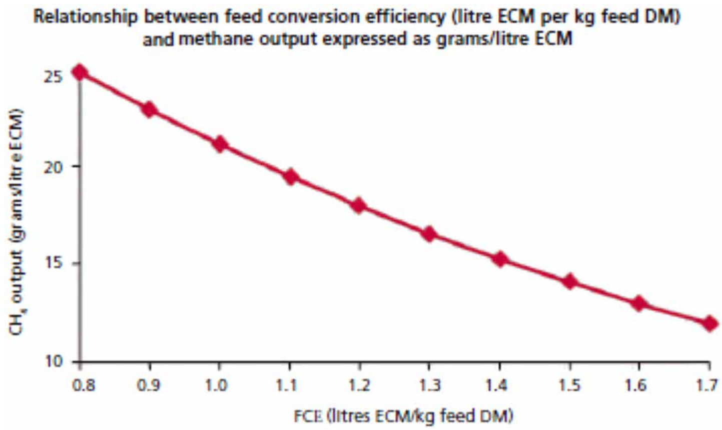

Figure 1 Theoretical relationship between methane output (g/L ECM) and FCE (L ECM/kg DM). Reprinted from Beever and Drackley (2013) with permission.

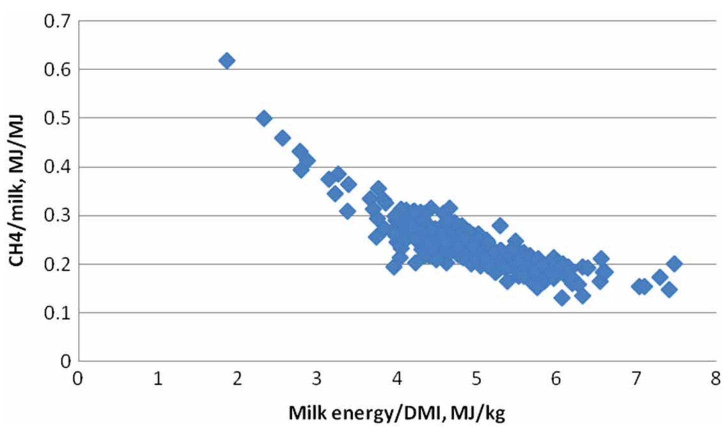

Figure 2 Methane energy/milk energy (MJ/day) as a function of milk energy (MJ/day) per unit feed dry matter intake (kg/day) where $y = 0.833*\exp^{(-0.259*x)}$. Individual observations (n = 323) for cows fed various dietary treatments in experiments conducted at the University of Reading with correction for the random effect of experiment on the intercept.

per unit milk energy yield with increasing FCE, expressed as milk energy per kg feed DMI (MJ/kg). An exponential decay curve provided the best fit of the data after correction for the effect of experiment, clearly showing the higher methane 'overhead' of lower-yielding cows and the dilution of maintenance greenhouse gas 'costs' with higher milk yield and FCE. Similar relationships between increased milk yield per se and reduced methane emission per unit milk yield have been noted previously (Reynolds et al., 2011).

5 Nutritional practices to enhance feed conversion efficiency and decrease CH_4 excretion

Because methane formation occurs from microbial fermentation of dietary constituents, it follows that nutrition of the ruminant impacts FCE and methane production (Table 1). The most important aspect of nutrition is feed intake, represented in ruminants as DMI. A meta-analysis of methane measurements from growing and lactating cattle (Mills et al., 2009) revealed that feed DMI was the primary predictor of methane emission (Reynolds et al., 2011). The effect of DMI is explained by the increasing provision of fermentable substrates, although the use of digestible energy only marginally improved the prediction (Mills et al., 2009).

A commonly used equation to estimate methane production (Moe and Tyrrell, 1979) is as follows:

$$\begin{aligned}\text{methane production, MJ/d} = {} & 1.837 + 1.142\,(\text{digested soluble residue, kg/d}) \\ & + 2.142\,(\text{digested hemicellulose, kg/d}) \\ & + 5.828\,(\text{digested cellulose, kg/d}).\end{aligned} \tag{2}$$

Since this early work of Moe and Tyrrell (1979), there have been numerous subsequent meta-analyses of measurements of methane emission relative to

Table 1 Feeding management variables and effects on methane (CH_4/ECM), dry matter intake (DMI), energy-corrected milk (ECM), and feed conversion efficiency (FCE). Modified from Knapp et al. (2014)

Feeding alteration	CH_4/ECM	DMI	ECM	FCE
Increased DMI	Decrease	Increase	Increase	Increase or no change
Increased forage quality	Decrease	Increase	Increase	Increase
Decreased forage particle size	No change	Increase	Increase or no change	No change
Grain processing	Decrease	No change	Increase or no change	Increase or no change
Increased concentrate feeding	Decrease	Increase	Increase	Increase or no change
Rumen pH <5.5	Decrease	No change or decrease	Decrease	Decrease
Brown midrib corn	Decrease	Increase	Increase	Increase or no change
Fat feeding	Decrease	Decrease or no change	Increase	Increase

diet intake and composition, which have largely confirmed the overriding role of intake of digestible structural and nonstructural carbohydrates, as well as lipids (see below), in determining methane emission and yield. For example, a recent global analysis of 3183 individual observations concluded that prediction models should include diet intake, digestibility, and concentrations of structural and nonstructural carbohydrates and ether extract for greatest accuracy (Benaouda et al., 2019).

According to Equation 2, increments of digested soluble residue, largely comprising starch and sugars, will increase methane to a lesser extent than digested hemicellulose or cellulose from forages. In turn, digested hemicellulose has a much smaller effect than digested cellulose. This is consistent with the fact that starch fermentation yields a greater proportion of propionate compared to fibre fermentation (Hungate, 1966). Increasing concentrate supplementation in the diet increases digestible energy, usually increases FCE, and decreases methane output per unit of milk produced, as predicted by the analysis in Fig. 2 and by the diluting effect of greater productivity on methane production per unit of milk yield. Concentrates containing cereal grains rich in starch will increase the metabolisable energy available to the animal, usually increasing milk yield unless excessive amounts are fed that cause acidosis (Van Soest, 1994).

In contrast, increasing the digestibility of forages, which will usually increase FCE by boosting milk yield, might not reliably change methane production per unit of milk produced because of the increase of fermented cellulose and hemicellulose. However, in practice, improving forage digestibility usually increases FCE and, by increasing milk production, the yield of methane per unit of ECM decreases due to the dilution of the maintenance concept. Variations in concentrations of cellulose, hemicellulose, and starch, as well as their digestibility, interact with differences in DMI that can make it difficult to predict the resulting methane production from differences in FCE (Reynolds et al., 2010; Livingstone et al., 2015). For example, diets higher in maize silage contained more starch and less NDF than diets higher in grass silage, and resulted in greater DMI and milk yields, but FCE was greater for the diets high in grass silage. However, methane intensity was greater for the high grass silage diets (Hammond et al., 2016).

Forage digestibility can be increased in several ways. Harvesting at younger maturity increases digestibility because of greater concentrations of nonstructural carbohydrates, such as sugars and fructans, along with lower lignification of plant cell walls (Van Soest, 1994). Some genetic variants have greater digestibility. For example, the stover from the *bm-3* or brown midrib genetic variant of maize and sorghum is more highly digested than typical varieties (Oba and Allen, 2000). Cellulose, hemicellulose, and NDF were less digestible in diets containing 40% (DM) maize silage and 10% alfalfa silage than in diets containing 10% maize silage and 40% alfalfa silage (Ruppert et al., 2003). Finally, grass forage is typically more digestible than legumes, such as alfalfa, although the rate of fermentation is faster for legumes (Van Soest, 1994).

Processing forages can have contradictory effects on methane yield. Fine grinding or pelleting of forage increases the surface area available for fibrolytic microbiota to attach for fermentation, but the small particle size increases its passage from the rumen (Russell and Hespell, 1981), thus limiting the digested dry matter and so limiting methane formation (Moss et al., 2000). Such effects could decrease methane by 20–40% per unit of DM at high intakes, although at restricted intakes the effects are less (Johnson and Johnson, 1995). However, this also decreases potential energy delivery to the animal, which might decrease milk production, although compensatory hindgut digestion can recover some of the lost energy (Van Soest, 1994). With fine grinding of the forage and the resulting faster passage, however, the animal will likely eat more (Van Soest, 1994), which would tend to increase methane production.

Grains can be processed by grinding, rolling, and steam flaking, all of which increase surface area for digestion and decrease particle size. Grain processing can decrease the amount of starch that passes to the small intestine. Digestion in the small intestine should be more efficient energetically, but results have not borne this out in practice (Huntington et al., 2006). The effect of processing on ruminal degradation of starch varies among cereals and depends on the processing methods used (Svihus et al., 2005). Extensive rolling of barley maximised ruminal and postruminal digestion of starch (Yang et al., 2001). More extreme processing tends to increase total VFA concentration in the rumen, with a greater proportion of propionate that would decrease methane formation. Grinding maize grain increased starch digestibility in the total tract compared with rolling (Knowlton et al., 1998), but ground maize resulted in greater methane production (Wilkerson et al., 1997). Steam flaking maize grain resulted in greater FCE than grinding, regardless of particle size of grind (Ahmadi et al., 2020). However, finely ground maize can produce results similar to steam-flaked maize (Mathew et al., 2011). Heat treatment through pelleting, flaking, extruding, and toasting can change the ruminal degradation rates of protein and carbohydrates and decrease the acetate:propionate ratio in the rumen (Van Nevel and Demeyer, 1996). However, the relationship between such treatments and CH_4 emissions depends on the feed, composition of the total diet, and intake (Knapp et al., 2014).

Supplemental fats and oils are often added to diets for dairy cattle to increase the energy density of diets for high milk production (Palmquist and Jenkins, 2017). According to meta-analyses (Rabiee et al., 2012; Hu et al., 2017), fats usually maintain or decrease DMI while increasing milk yield and milk energy output. Consequently, supplemental fat sources often increase FCE and, as expected, decrease methane output per unit of milk energy (Ruppert et al., 2003; Kliem et al., 2019). The effect of lipids on methane production is dependent on the source, fatty acid profile, inclusion rate, form of lipid, and diet composition (Beauchemin et al., 2008). Lipid sources replace fermentable substrates, and the unsaturated fatty acids provide an alternate sink for hydrogen disposal. Unsaturated fatty acids also directly impact fibre-fermenting microbiota and methanogens in the rumen (Czerkawski et al., 1966; Blaxter and Czerkawski, 1966). In practice, however, changes in methane production are not always observed. Nevertheless, a meta-analysis conducted by Knapp et al. (2014) found that increasing fat content of the diet decreased methane per unit of ECM, and that the type of lipid source affected the response. Free vegetable oils and endogenous lipids resulted in a greater reduction of methane per unit of ECM than did inert fats or seed lipids. Some seeds, vegetable oils, and medium-chain fatty acids (such as those found in coconut oil; Hristov et al., 2009) further decrease methane, but often at the expense of DMI, which could be detrimental to productivity and reproduction over the long term (Reynolds et al., 2011; Knapp et al., 2014).

In addition to 3-nitrooxypropanol and nitrate mentioned previously, a variety of potential nutritional additives have been tested for their ability to improve FCE and decrease methane losses, including ionophores, yeasts, certain seaweeds, and plant bioactive compounds with antimicrobial or digestive effects, such as tannins and essential oils (Beauchemin et al., 2008; McAllister and Newbold, 2008). These compounds have had varying efficacies, especially in the longer term, perhaps due to differences in dose rate and adaptation of the rumen microbes (Beauchemin et al., 2008). Although some have shown potential to decrease methane excretion, for these approaches to be widely adopted their effects will need to be consistent, sustained, without deleterious effects on feed intake, production, FCE or product quality, economic to adopt, and sustainable (Reynolds et al., 2011).

6 Nutritional practices to increase milk protein efficiency and decrease N_2O excretion

In ruminant production systems, enteric CH_4 production is the greatest contributor to greenhouse gas emissions, followed by CH_4 from manure and in beef feedlot systems, N_2O from pen surfaces, and N_2O emissions from soils (Montes et al., 2013). Improving the efficiency of dietary N use by increasing N incorporation into milk protein and decreasing N losses in faeces and urine will have positive effects on N_2O excretion and environmental contamination with N (Uwizeye et al., 2020). Many of the same principles discussed to this point relative to increasing FCE will have beneficial effects on increasing efficiency of dietary N use. From an environmental standpoint, animal practices to decrease urinary N loss have the biggest impact because urinary N is more susceptible to leaching and volatile losses than faecal N and contributes directly to N_2O loss from livestock facility surfaces (Dijkstra et al., 2013a; Montes et al., 2013).

The main driver of N losses from cattle is N consumed in feed. Dairy cows secrete in milk an average of 21–33% of consumed N (Calsamiglia et al., 2010), with almost all the remaining N excreted in faeces and urine. In agreement, Reed et al. (2015) calculated an average total manure N excretion of 69% of N intake from a large database. Dijkstra et al. (2013b) calculated the theoretical upper limit of dietary N incorporation into milk protein to be 43% at maximal milk secretion for a cow weighing 650 kg and producing 40 kg/day of fat and protein corrected milk.

Huhtanen and Hristov (2009) concluded from a meta-analysis that dietary crude protein (CP) concentration is the most important dietary factor influencing milk N efficiency, with ruminal degradation of CP being of lesser importance. Differences in amount and, to a smaller extent, digestibility of N in feed affect not only the total amount excreted but also the partitioning of N among milk, urine, and faeces (Castillo et al., 2001; Kebreab et al., 2002). Partitioning of manure N excretion into faecal and urinary N excretion is important because differences in N intake largely affect urinary N output, which is of greater importance to reduce environmental impact (Dijkstra et al., 2013a).

The carbohydrate portion of the diet may impact the efficiency of dietary N use through differences in the amount or efficiency of N capture in the rumen. Ruminants must have sufficient ruminally degradable protein to maximise fermentation with the amount of fermentable carbohydrates supplied. Wilkinson and Garnsworthy (2017) showed that replacing dietary grazed grass or grass silage with corn silage at similar milk yield increased N efficiency and reduced the C footprint. Grazed grass is particularly problematic in terms of urinary N loss because of the high soluble CP content of the grass and concurrent lower content of rapidly fermentable carbohydrates (Hristov et al., 2013).

Dairy producers have often overfed dietary protein in the mistaken assumption that it would boost milk production (Broderick, 2018). Broderick (2003) compared diets with CP increasing from 15.1% to 16.7% and 18.4% by adding soya meal to the diet. Milk and milk protein yields increased with the first increment of CP but not the second. The only result from increasing dietary CP to 18.4% was the increased excretion of urinary N, which accounted for nearly all of the increment of dietary N. Recently, a trend for lower dietary CP has been observed in high-producing herds. Diets with as little as 14.9% CP can support milk production in excess of 45 kg/day when properly balanced for N and carbohydrate fractions and supplemented with key ruminally protected amino acids (Fessenden et al., 2020). In 2010, the five highest producing herds in the state of Wisconsin were being fed diets containing an average of 16.9% dietary CP, with the lowest at 16.3% (Broderick, 2018). Over the period from 2004 to 2010, CP content of dairy diets in Wisconsin decreased 1.1 percentage units, but milk and protein production increased by 1700 and 51 kg per lactation. Clearly, minimising dietary CP needed to support high milk production pays dividends in the way of decreasing ration cost and environmental excretion of N, without sacrificing productivity.

7 Genetics and feed conversion efficiency

Using RFI as a measure of FCE has the advantage that it is independent of maintenance requirements and is not an efficiency determined by the level of production per se. A disadvantage of the measurement is that animals that are more efficient have negative RFI, which is non-intuitive and difficult for producers to accept. Reported heritability estimates of RFI in dairy cattle generally are low to moderate, with estimates ranging from 0.01 to 0.40 among lactating cows (Connor, 2015). Tempelman et al. (2015) determined that RFI had a heritability of 0.15–0.18. This means that the trait could be improved through genetic selection but that much of the variation in phenotype must be attributable to other environmental factors. Genetic correlations suggest no undesirable relationships detected between RFI and fat-corrected milk yield, productive life or feeding behaviours, and desirable relationships between RFI and predicted methane production in lactating cows (Connor, 2015).

There is currently interest in the potential selection of dairy cattle for improved feed efficiency (e.g. Lin et al., 2013; VandeHaar et al., 2020) and lower methane emission or methane yield (e.g. Breider et al., 2019; Lassen and Løvendahl, 2016). However, these traits must not be considered in isolation; for example, reductions in methane yield in sheep were shown to be driven by higher rates of feed passage through the rumen attributable to smaller rumen volume and thus reduced total feed intake capacity (Goopy et al., 2013).

The benefits of selection for RFI make the prospect of its application promising. For example, Holstein–Friesian heifers in New Zealand and Australia fed a forage-based diet that were in the bottom 10% of the sample population for RFI (i.e. most efficient) consumed 15%–20% less feed relative to heifers in the top 10% for RFI (least efficient; Williams et al., 2011; Waghorn et al., 2012). Differences are similar to differences in DMI of 12–13% reported in low versus high RFI groups of growing Angus–Hereford steers (Cruz et al., 2010). The dairy heifers cited showed no differences in feed intake, yields of milk or milk components, change in BW or body condition score when evaluated during days 75–195 of their first lactation (Macdonald et al., 2014). These results indicate that considerable savings in feed costs can be achieved by maintaining only the most feed-efficient growing heifers in the herd, with no negative consequences on future lactation performance.

In addition to selection for RFI, it might be possible to use genetic selection to improve N efficiency. Marshall et al. (2020) used breeding values for milk urea N (MUN) to create high and low lines of Holstein–Friesian dairy cows. There was a positive relationship between MUN breeding value and MUN, with MUN decreasing 1.61 mg/dL per unit decrease in MUN breeding value. Urinary N concentration decreased 0.67 g/L per unit decrease of MUN breeding value, with no difference in urine volume or urinary frequency, which resulted in a 165 g/day decrease in urinary urea N excretion between animals with the highest and lowest MUN breeding value. At the same milk yield, milk protein percentage increased by 0.09 per unit of MUN breeding value. Such preliminary results are positive both for decreasing environmental excretion of N and for improving N partitioning into milk for greater producer profitability.

Relative to greenhouse gas emissions, cattle with lower RFI have a lower DMI than less efficient animals at similar production levels (Connor, 2015), thus decreasing one major contributor to CH_4 production. By definition, animals with lower RFI have less manure output for a given level of production, which should decrease the release of CH_4 and N_2O from stored manure (Montes et al., 2013). Recent research demonstrated that cows with greater FCE (lower RFI) also used protein more efficiently, which would help reduce N excretion in urine (Liu and VandeHaar, 2020).

8 Postabsorptive metabolism and feed conversion efficiency

Heat energy lost in the conversion of ME to net energy (NE) typically accounts for 20–30% of gross energy intake in dairy cattle. But, in contrast to digestibility, the variation and opportunities to decrease these losses are more limited. The effects of forage type, processing, and forage to concentrate ratio on the efficiency of ME utilisation for milk and meat production (and FCE) have been extensively researched (Reynolds, 2011). Such effects are a greater concern in extensive systems relying heavily on poorer quality forages. Because the digestive system and liver account for as much as 50% of body heat production in ruminants, their metabolism has an impact on the partial efficiency of ME use for production that is disproportionate to their mass. Forage amount and digestibility impact ME utilisation for NE, which is largely attributable to the tissues of the gut and may reflect differences in the work of digestion and gut mass (McLeod and Baldwin, 2000). In contrast, nutrient re-partitioning agents such as growth hormone impact energy utilisation by altering metabolism in the muscle, adipose tissue, and mammary gland to increase milk production, which improves FCE by dilution of maintenance.

Reducing losses of dietary energy as heat may be one of the main factors affected by selection for improved RFI. Only 19% of the variation in RFI among animals may be attributable to differences in diet digestion and heat of fermentation, with the remainder likely due to differences in physical activity, body composition, protein turnover, and other metabolic processes associated with maintenance requirements and partitioning of nutrients between protein and lipid deposition (Herd and Arthur, 2009).

The effects of excess protein on energy utilisation in cattle have generated much discussion. Tyrrell et al. (1970) reviewed results of calorimetry studies and reported a reduction in ME or NE of diets when digestible protein was fed in excess of requirements. This effect was included in the calculation of NE for lactation (Moe et al., 1972). Some have attributed this effect of excess protein to the energy costs of urea synthesis in the liver, but studies in sheep and cattle catheterised for measurements of liver metabolism have consistently failed to show an effect of increased ammonia absorption and subsequent urea production by the liver on hepatic oxygen consumption (Reynolds, 2006). In contrast, supplemental protein fed in excess of requirement does increase liver oxygen use (i.e. heat production), as well as oxygen use by the gut, suggesting that the effect of excess protein on the efficiency of energy utilisation is associated with amino acid catabolism, rather than urea synthesis per se (Reynolds, 2006). Over 100 years ago, Rubner showed that the postprandial rise in heat production in dogs, termed 'specific dynamic action', associated with consumption of protein was greater than for carbohydrates and fats (Brody, 1939), which also reflects the catabolism of amino acids in excess of requirements. These effects might be attributable to stimulation of protein turnover and greater activation of the sympathetic nervous system. Regardless of the mechanism, the potential benefits of feeding protein and ruminally protected amino acids more precisely to cattle extend beyond just the environment and economics to improving FCE.

9 Conclusion

Methane intensity of dairy production is related inversely to FCE. By diluting methane associated with cow maintenance, increasing milk production and increased FCE serve to decrease methane production per unit of milk energy produced. Nutritional practices that boost productivity usually boost FCE and will help to decrease methane intensity. Some of these include improving forage quality, increasing concentrate supplementation, and using supplemental fats and oils. Genetic selection for improved FCE (measured by RFI) will decrease methane intensity. Minimising dietary CP will help decrease urinary N losses, which contribute to increased N_2O and decreased efficiency of N capture into milk protein.

10 Future trends in research

Increasing FCE is directly related to profitability and will be a continued goal of research for improvement in the dairy industry. Decreased methane intensity should be a collateral benefit of such improvement. Determining the genetic basis for FCE, as measured by RFI, may eventually lead to improvements in the overall efficiency of milk production. Improving genetic aspects of forage digestibility and methods to improve use by cows will help decrease methane intensity. The research will continue to find compounds or strategies that will decrease methane yield without compromising DMI or productive efficiency. Novel mitigation strategies might include inhibitors or early life microbial modulation to change the balance between hydrogen producers and methanogens. In addition, the possibility that synergistic effects between strategies might occur, such as combining 3-nitrooxypropanol with lipids, should be investigated. Research will continue to identify genetic opportunities to minimise the environmental burden of livestock production as well as urinary N excretion.

11 Where to look for further information

A comprehensive discussion of strategies to mitigate methane emissions in dairy cattle can be found in Knapp, J. R., Laur, G. L., Vadas, P. A., Weiss, W. P., and Tricarico, J. M. (2014), 'Invited review: Enteric methane in dairy cattle production: Quantifying the opportunities and impact of reducing emissions', *J. Dairy Sci.,* 97, 3231–3261.

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Chapter 8

Improving grassland/forage quality and management to reduce livestock greenhouse gas emissions

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1 Introduction

Pasture-based systems of production face multiple challenges, including global food demand, continuing to balance environmental sustainability, and ensuring product quality meets the highest consumer standards of nutrition and health. The challenge ahead for grassland farmers is imposing, but system sustainability and reducing methane emissions by improving management and forage quality is an aspect of pasture farming that needs to be addressed intensely by all elements of the industry. European grasslands sustain a large number of domestic herbivores, 150 million cows and 150 million sheep, which is roughly 15% of the global animal production. Grazers impact the cycling of C and N within pastures via defoliation, excretal returns and mechanical disturbance. They emit CO_2 via their metabolic activity and methane (CH_4) through enteric fermentation. Improving grassland management practise on farms is a key avenue of addressing further reductions of CH_4 emissions at farm level. There are many ways of

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completing this successfully, and the challenge is to ensure these methods are employed at farm level through:

i optimising the use of home-grown grass;
ii increasing the proportion of grazed grass in the overall diet of the grazing ruminant;
iii extending the grazing season, especially in spring and autumn, thereby reducing the indoor feeding period;
iv adopting a strategy of grazing lower pre-grazing herbage masses during the grazing season (adapting sward morphology);
v grazing swards with higher levels of clover content, targeting levels of >20% in the sward, on average, across the grazing season;
vi improving grass dry matter intake (DMI), milk solids and average daily gain at grass; and
vii measurement of pasture quality at farm level across the growing season and appropriate supplementation patterns based on these figures.

These themes are discussed in the rest of the chapter.

2 Grassland areas and productivity in Europe

Europe maintains one of the highest livestock densities in the world. In 2018, the 27 member states of the European Union (EU-27) produced: 26%, 13%, 22%, 12% and 11% of the world's milk, beef, pork, poultry and eggs, respectively (FAO, 2009). Following considerable growth in the 1960s and 1979s, cattle numbers in Europe have been decreasing since 1980. More recently, reductions in animal production in Europe are related to market developments as well as to changes in agricultural and environmental policies. Between 1970 and 2013, 5.9 million ha of grassland in the EU9 were lost to other purposes (Table 1), equating to a 15.3% reduction in the proportion of permanent grassland. The proportions of permanent grassland reduced significantly in Germany (31.8%), the Netherlands (30.1%), France (28.1%), Denmark (25.9%) and Belgium (21.7%), while the reduction in Italy was much lower (13.3%). There was a small reduction in Ireland (2.2%) and Luxembourg (0.8%), while there was an increase in the United Kingdom (6.4%). Similar reductions in grazing land areas accrued in Austria, Greece, Spain and Hungary.

Grassland productivity will be affected by botanical composition, soil characteristics, climate conditions, altitude, latitude and management (De Vliegher and Carlier, 2007). Lee (1983) reviewed grassland potential productivity data from most European countries, and it was divided into five major geographical/climatic regions:

Table 1 Changes in the proportion of utilised agricultural area in permanent grassland area in EU9 between 1970 and 2013

		1970		2013	
	Total UAA	Permanent grassland (ha)	Proportion (%)	Permanent grassland (ha)	Proportion (%)
Belgium	1 307 900	621 253	47.5	486 539	37.2
Denmark	2 619 340	264 553	10.1	196 451	7.5
France	27 739 430	11 456 385	41.3	8 238 611	29.7
Luxembourg	131 040	67 486	51.5	66 961	51.1
Germany	16 699 580	6 780 029	40.6	4 625 784	27.7
Ireland	4 959 450	4 007 236	80.8	3 917 966	79
Italy	12 098 890	3 823 249	31.6	3 315 096	27.4
The Netherlands	1 847 570	1 104 847	59.8	772 284	41.8
United Kingdom	17 096 170	10 138 029	59.3	10 787 683	63.1
Total	84 499 370	38 263 066	46.9	32 407 374	40.5

Source: Eurostat; authors own calculations.

1 North-west and west Europe;
2 Central Europe;
3 South-east Europe;
4 Mediterranean Europe; and
5 Northern Europe.

Factors that are considered to influence the grassland productivity are altitude, water stress, temperature and aspects such as slope and soil depth.

A coordinated experiment organised under the auspices of the FAO Lowland Grassland Sub-network measured the production and productivity of cutting grassland according to a standardised protocol in 32 European sites. The average annual production ranged from 10 t DM ha^{-1} to 14 t DM ha^{-1} (Peeters and Kopec, 1996). The extremes in grass production were very different ranging from 2 t DM ha^{-1} in Portugal to 20 t DM/ha in Germany (Kiel) (Peeters and Kopec, 1996; Fig. 1). The most productive sites (>15 t DM ha^{-1}) were located on the Atlantic side of Europe between 52°N and 57°N latitude.

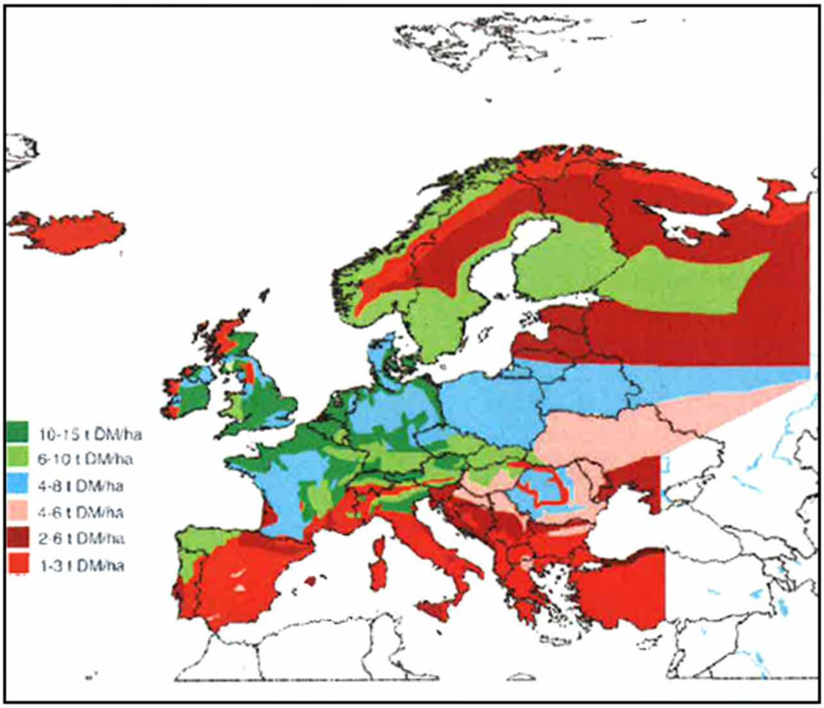

Figure 1 Production potential (annual yields in t DM/ha) of mown and heavily fertilised grasslands (source: Peeters and Kopec, 1996).

These included the Netherlands, Great Britain, Ireland, Belgium, north-western France and northern Germany. The less-productive sites were situated at high or low latitudes in Europe.

The EU (28 countries) currently has a permanent grassland area of about 60 million ha (Eurostat, 2017). Permanent and temporary grasslands represent 40% of the total utilised agricultural area in Europe (Huyghe et al., 2014) and a large acreage of these grasslands is exclusively used as ruminant feed, as either grazed grass or grass silage/hay. This asset of grasslands is extremely important for the human population since ruminants deliver food for humans by converting human-inedible plant biomass into high-quality human edible proteins. By providing feed to ruminants, grasslands contribute to the feeding of member state populations. Grass-based ruminant production delivers a number of other services to society, like carbon (C) sequestration (e.g. Soussana et al., 2010; Conant et al., 2017) and biodiversity (e.g. Isselstein et al., 2005; Van den Pol-van Dasselaar et al., 2019)).

Under climatically and topographically favourable conditions, the European grasslands area has been significantly reduced during the last 30 years (Huyghe et al., 2014). According to the 3rd report of the EU MAES initiative (Mapping of Ecosystems and Ecosystem Services), between 2006 and 2012 the main causes for this process were the conversion of grasslands into arable crops like maize (including for the production of biofuels) and other crops (32% of the lost area), the sprawl of urban areas, economic sites and infrastructures (30%), and the withdrawal from farming (17%) (Erhard et al., 2016). In many countries, the number of dairy cows decreased in the last 30 years but the milk yield of individual cows increased during the same period, with the number of cow reductions mainly driven by the implementation of the milk quota regime. Between 2010 and 2016, however, the bovine population slowly grew again by 1.4% (Eurostat, 2017).

The improvement in individual animal milk production is achieved based on an increasing amount of concentrates and maize in cow rations and declining herbage use from grassland (e.g. Isselstein et al., 2005). More and more farmers have changed to all-year housing and do not provide access to grazing for their cows, for example, and only

- 42% of German dairy cows have access to pasture (Gurrath, 2011); and
- 25% of Danish dairy cows have access to pasture (Van den Pol-van Dasselaar, 2016).

Such a decline in the access to grass has led to increases in milk production costs at farm level, and a worrying dependence on imported feed inside the farm gate; if continued it will lead to more GHG from the ruminant population.

3 The challenge of greenhouse gas emissions from livestock

Greenhouse gas emissions (GHG) from livestock are closely related to ruminant numbers. Factors other than ruminant numbers also have an impact. The size and productivity of animals affects their feed intake and enteric CH_4 emissions. Rarely, if ever, have the levels of grassland management, and the type of grassland offered to the grazing animal, been considered as a mitigation factor for CH_4 production. We know clearly that ruminants produce CH_4 during enteric fermentation of feed and CH_4 and N_2O are released from stored manure. Farm-based studies indicate that there are large differences among farms in terms of animal productivity and environmental impacts. These differences are often related to the management skill of the farmer, technologies applied and environmental conditions. We will discuss the possibilities to reduce CH_4 levels through better grazing management practises and the choice of more appropriate pasture species, given the changes in European grassland area and practise over recent years.

Ruminants lose between 2 and 12% of ingested energy as CH_4. Improvement in forage quality and more specifically forage digestibility has been investigated as a means of enteric CH_4 mitigation (Hristov et al., 2013). Structural carbohydrates have been reported to be more methanogenic than soluble carbohydrates. In ruminants with high feed intakes, reductions in enteric CH_4 emissions per unit intake with increased digestibility of feeds have been reported (Hristov et al., 2013). Greater digestibility is associated with a fermentation profile in the rumen that is unfavourable to CH_4 production. Hristov et al. (2013) stated a more digestible feed is associated with greater intake and production, diluting maintenance energy requirements and resulting in less CH_4 per unit of animal product. The literature is full of studies having evaluated feeds in indoor feeding systems based on concentrates and forage diets. Grazing studies with CH_4 emissions measured are currently scarce but increasing.

4 Grazing management to combat climate change: grazing season

4.1 Grazing season length and impact of climate change

Implementing good grazing management practises to improve the quality of pastures will increase animal productivity and lower CH_4 per unit of product (Boadi et al., 2004). A long grazing season can increase the annual proportion of grazed grass in ruminant diets, which can reduce feed costs and increase profitability (Dillon et al., 2005; Finneran et al., 2012). Grazing is generally positively perceived by consumers when compared to indoor feeding systems

(Van den Pol-van Dasselaar et al., 2019). Phelan et al. (2015) investigated the spatial variation in grazing season lengths from 32 European countries obtained from the results of the EUROSTAT Survey on Agricultural Production Methods (SAPM) and bioclimatic variables for dairy farms. The reference year was 2012 for all countries with the exception of Spain and Portugal which had 2009 as the reference year. Grazing season length was positively correlated with mean temperature during the coldest quarter and negatively correlated with precipitation in the wettest month. Figure 2 illustrates the observed and predicated grazing season lengths for dairy farms in all 32 European countries. The predicted grazing season lengths were longer than observed in Belgium, Estonia, Germany, Hungary and the Netherlands but shorter in Bulgaria, France, Latvia and Lithuania.

4.2 Early season grazing

Grazed grass can be increased in the overall diet of the dairy cow by allowing cow's access to grass early in spring; this is an opportunity for all member states. Many studies have shown an improvement in milk production and composition with this practise (O'Donovan et al., 2004; Kennedy et al., 2006). As well as improving animal performance, early spring grazing can have beneficial effects

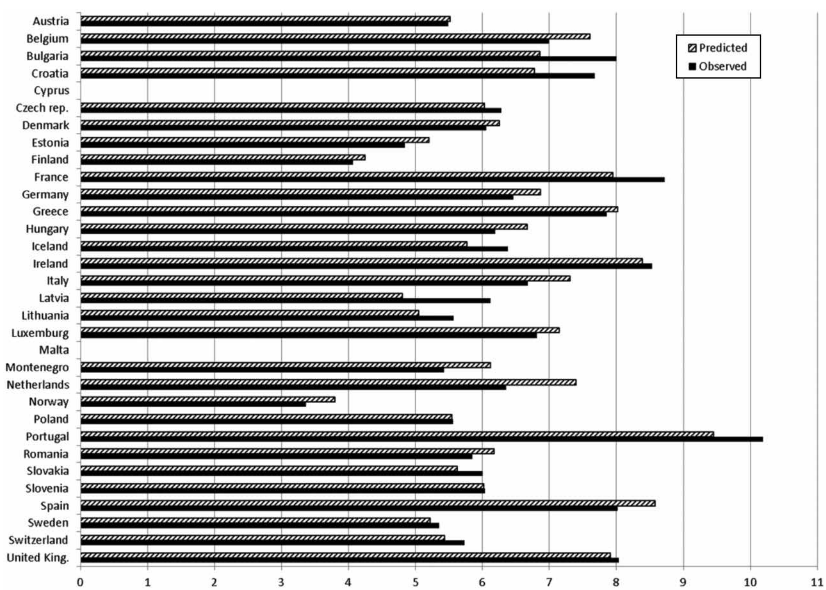

Figure 2 Observed and predicted grazing season length (months) for 32 European countries (source: Phelan et al., 2015).

including increasing grass utilisation, sward quality and simplifying grazing management. Late turnout to grass can lead to under-grazing of pastures for a variety of reasons, for example, excessively high pre-grazing herbage mass, low-grazing stocking rates or poor grass utilisation conditions. O'Donovan et al. (2004) found that early spring grazing can act as a sward conditioner, that is, avoids build-up of excessively high pre-grazing yield. They found high milk production from early grazed swards even with a low grass allowance compared to late-grazed swards, clearly showing the beneficial effects of early grazing on sward structure and quality. The early use of grass reduces the need for supplementary feed, making better use of home-grown feed. Early spring grazing reduces the large requirement for machinery, fuel and fertilisers, and facilitating less GHG production on farms.

O'Neill et al. (2011) and Robertson and Waghorn (2002) found that in early lactation pasture-fed cows produced less CH_4 emissions per day than cows offered a total mixed ration (TMR) diet. The higher DMI exhibited by TMR cows is likely to have caused the increased CH_4 emissions by these cows.

Increasing DMI increases CH_4 production as greater DMI provides a greater intake of fermentable substrate, including both structural and non-structural carbohydrates (Moe and Tyrrell, 1980). In addition, saliva is an important rumen buffer (Owens et al., 1998) and higher saliva production may give rise to increased rumen pH which would maintain favourable rumen fermentation conditions for fibre digestion and methanogenesis (Krause et al., 2002). Pinares-Patiño et al. (2007) found that in their study saliva production was positively correlated with higher daily CH_4 emissions and feed intake.

5 Grazing management to combat climate change: sward structure and quality

5.1 Sward structure characteristics

The presented herbage mass very much dictates the grazing intensity and level of DMI achieved by the grazing animal. Pre-grazing herbage mass also dictates the source from which the animal selects its diet (Wade, 1991). The presented herbage mass influences the sward characteristics presented to the grazing animal. Grazing animals prefer living to dead material, younger to older material, leaf to stem and legume leaves to grass leaves (Leaver, 1985). Michell and Fulkerson (1985) showed that pre-grazing herbage mass increased at the sward base in the latter part of the season, due to stem and dead material accumulation (Mayne et al., 1987). High levels of death and decay accompany poor herbage utilisation. Curran et al. (2010) agreed with those findings, and his work showed that high pre-grazing herbage mass swards had greater accumulations of dead material in the second half of the season due to a lax post-grazing height and low levels of grass utilisation. Korte et al. (1984) suggested that a high grazing intensity reduced the production and development of reproductive tillers. Fulkerson and Donaghy (2001) stated that severe defoliation removes too much water soluble carbohydrate (WSC) storage capacity and reduces regrowth, while under lax defoliation, the loss of DM through leaf senescence and reduced rates of tillering are not compensated for by the increased growth rate that results.

Criteria for determining when to defoliate pastures have been based on rotation length, sward height and pre-grazing herbage mass (HM) (Mayne et al., 2000). Pre-grazing herbage mass takes account of sward height and density, and is an animal sward interface indicator of when the sward is ready to graze. Previous studies by O'Donovan et al. (2004) and Kennedy et al. (2006) showed that high HM swards supported greater stocking rates; however, grazing low HM swards had a positive effect on herbage quality, milk production and grass DMI. Curran et al. (2010) found that low herbage mass swards supported high stocking rates due to a greater number of grazing rotations and improved sward quality due to intense grazing. O'Donovan et al. (2004) found that herbage from early grazed swards (February/March) was of higher quality (increased organic matter digestibility (OMD) and UFL value) relative to late grazed swards (April). This in part reflects seasonal trends in the accumulation of dead herbage. Wade (1991) found sward stem (true and pseudostem) as a barrier to increasing grass DMI, and this is true when cows are forced to graze to low post-grazing residuals. Sheath resistance increases in importance as a barrier to intake as pre-grazing herbage mass and pre-grazing height increase.

Wims et al. (2010) found that dairy cows grazing high pre-grazing herbage mass swards increased their CH_4 production per cow per day (+42 g), per kg of milk yield (+3.5 g/kg), per kg milk solids (+47 g/kg) and per kg grass DMI (+3.1 g/kg). The main difference in these swards was a 10-day difference in rotation length. In both early and late season measurements, Wims et al. (2010) found that cows grazing high HM swards lost a greater proportion of their gross energy intake as CH_4 during both measurement periods (+0.9% and +1% for summer and autumn, respectively). Wims et al. (2010) offered low HM swards to cows which maintained higher grass DMI and milk output; however, while no significant differences were found in CH_4 production, the conclusion was that grazing lower pre-grazing herbage mass swards tended to reduce CH_4 per unit of DMI (−8.2%) and energy corrected milk yield (−10%) compared to grazing high HM swards. Both these studies agreed that offering lower HM swards to grazing cows constituted a viable CH_4 mitigation strategy.

5.2 Maintaining sward quality

Ruminant CH_4 originates from the digestible fraction of the diet rather than the whole diet and fermentation of cell wall carbohydrates (NDF) produces more CH_4 than fermentation of soluble sugars (Moe and Tyrrell, 1980). Blaxter and Clapperton (1965) reported relationships between dietary factors and CH_4 emissions for indoor-fed animals. These authors stated that absolute emissions (g d^{-1}) and CH_4 yield (% of gross energy intake) increase and decrease, respectively, as feed intake increases above maintenance requirements, but both absolute CH_4 emissions and CH_4 yield decreases with increasing digestibility. Plant maturity is the most important factor affecting the morphology and forage quality. As maturity stage increases, the proportion of cell wall (neutral detergent fibre (NDF)) contents increases, whereas the proportion of cell contents decreases. The NDF contents are negatively correlated to rates of digestion and passage and herbage intake (fill effect).

Hammond et al. (2011) found 0.87 of the variation in total enteric CH_4 emissions of grazing sheep was predicted by OM intake, and the relationship between forage chemical composition and total CH_4 emissions and CH_4 yield (g/kg of DM intake) were weak. O'Neill et al. (2011) found that cows grazing a high-quality perennial ryegrass diet had lower CH_4 per unit of feed intake than cows that offered a TMR of lower digestibility. The effect of forage quality on CH_4 emissions most likely depends on the extent of contrast in forage quality between treatments in the completed studies.

Figure 3 shows the impact of grass quality on the actual post-grazing sward height achieved over two grazing seasons by dairy cows grazing sward plots. It is clear that grass varieties with greater grass quality values have subsequently lower post-grazing sward height, meaning improved grass utilisation. Since 2013, Ireland has introduced the Pasture Profit Index (McEvoy et al., 2011), and key traits of this index are seasonal DM yield, pasture quality, silage DM yield and persistency. New traits such as grazing utilisation will be introduced into this index in the next year.

5.3 Grass dry matter intake at pasture

Achieving high grass utilisation consistently takes a considerable amount of grazing management discipline, and its application can be poor on farms. Within the typical range of herbage allowance (HA) in grazing systems, herbage intake increases on average by 0.10–0.15 kg/kg HA at ground level and 0.20–0.25 kg/kg HA above 4–5 cm (Delagarde et al., 2011). This means that the marginal response of pasture utilisation rate when increasing HA is very small (15–25%). There is a consensus that increasing feed intake reduces CH_4 yield as g CH_4/kg DMI although its effect is larger with high- versus low-quality diets

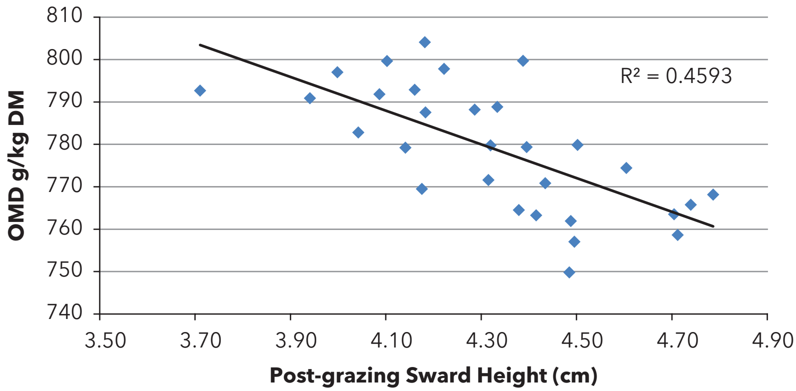

Figure 3 The effect of pasture quality (OMD) on post-grazing sward height over the grazing season (Tubritt, personal comm).

(Blaxter and Clapperton, 1965). Combined with good grassland management, allocating the correct daily HA and pre-grazing HM, CH_4 from grazing dairy cows can be reduced. It is important to maintain a high-quality diet, as possible, when grazing.

5.4 Grass clover swards with dairy cows

Higher animal performance from swards containing white clover can be expected because of its superior nutritional value compared to perennial ryegrass due to higher crude protein content (Butler, 2000), lower structural fibre values (Thomson, 1984) and higher OMD (Wilman and Riley, 1993). White clover is a potential option to reduce CH_4 emissions because of higher grass DMI and higher milk production (McClearn et al., 2019). Grass-white clover has lower aNDF and high voluntary DMI compared to grass-only (Ulyatt, 1970; Enriquez-Hilalgo et al., 2014). Lower aNDF concentration in white clover may lead to rapid ruminal degradation and passage rate, which should lower CH_4 yield as g CH_4/kg DMI.

A balance between the optimum sward white clover content for milk production and pasture production must be achieved to optimize both animal and pasture performance. Within the sward, white clover proportion changes seasonally; to achieve consistent clover content is a real challenge in grazing management. It is difficult to maintain optimum levels of white clover in the sward because of climatic factors (drought, waterlogging, colder soils), poor grazing managements, suboptimal soil fertility and pests. Dineen et al. (2018) completed a meta-analysis from a number of studies which had white clover included in the diet of grazing dairy cows. The mean sward white clover content was 31.6%, mean daily milk yield and milk solids yield per cow were increased by 1.4 kg and 0.12 kg, respectively, milk and milk solids yield were unaffected when cows grazed grass clover compared to grass-only swards.

McClearn et al. (2019) created swards with an average annual sward white clover content of 23.6% and 22.6% in tetraploid perennial ryegrass and white clover swards and diploid perennial ryegrass and white clover, respectively. Milk production did not differ between grass ploidies during a 4-year study, but cows grazing the perennial ryegrass-white clover treatments had significantly greater milk yield (+597 kg/cow per year) and milk solids yields (+48 kg/cow per year) compared with cows grazing the perennial ryegrass-only swards. Increased milk output has been associated with higher herbage nutritive value for perennial ryegrass-white clover swards, especially mid-season, compared to perennial ryegrass-only swards (Soegaard, 1993) and an increase in voluntary herbage DMI (Ribeiro Filho et al., 2005) with numerous studies having shown selective grazing of white clover over perennial ryegrass (Rutter et al., 2004). McClearn et al. (2019) found the difference in milk production from the perennial ryegrass-white clover swards was observed from May onwards in each year. This pattern was consistent with white clover content in the sward increasing as the season progressed and is similar to what Woodward et al. (2001) and Egan et al. (2018) reported. Andrews et al. (2007) suggested that sward-white clover content greater than 20% is required to establish an animal production response. Establishing such white clover proportions on farms will be and is a real challenge. Egan (personal comm) has shown high clover germination rates on farms but low establishment. He found that successful rates of establishment were only 60% across all farms, 12 months after over seeding. Direct reseeding perennial ryegrass-white clover swards is the only reliable way of establishing these swards on farms.

Methane emissions related to gross energy intake of animals-fed legumes are lower than animals-fed grasses (Beauchemin et al., 2008). Some previous research has shown that clover inclusion in pasture can reduce dairy cow CH_4 emissions and others have found no effects (van Dorland et al., 2007). Enriquez-Hilalgo et al. (2014) found that an average annual white clover content of 20% was not sufficient to improve overall sward production, quality or dairy cow productivity. The cows grazing the grass-white clover swards had a tendency to consume more and emitted less CH_4 than cows grazing the grass-only swards.

Structural carbohydrates are fermented at slower rates than non-structural carbohydrates such as starch and sugars to yield more CH_4/unit substrate fermented. Slower rumen fractional outflow rates and higher rumen volumes increase rate of CH_4 emission, most probably by allowing increased digestion of structural carbohydrates and providing a better environment for the growth of methanogens. The higher readily fermentable carbohydrate:structural carbohydrate ratio in white clover compared with perennial ryegrass may decrease the rumen acetate:propionate ratio, which is expected to lower CH_4 yield.

6 Grazing management to combat climate change: legume forages

Legume forages have a substantially higher nutritive value than grasses, biologically fix N and the condensed tannin (CT) containing legumes birdsfoot trefoil and sainfoin (*Onabrychis vicifolia* Scop.) and may have further advantages over non-tannin containing forages such as alfalfa. Sainfoin contains higher concentrations of condensed tannins (50–80 g/kg DM) compared to 5–47 g/kg DM for birdsfoot trefoil. Condensed tannins bind strongly to proteins and it has been proposed that some plants evolved CT production as a chemical defence, first against invasion by pathogenic microorganism, then against being eaten by insects and finally against being eaten by grazing herbivores. Originally it was thought that CT-containing forages were the Lotus species, sulla (*Heddysarium coronarium*) and sainfoin. Newer technologies have shown the presence of CT in grasses, legumes and herbs (Table 2).

Bloat is caused by very high solubility of forage proteins leading to the development of stable foam in the rumen, and it is prevalent in cattle and sheep-fed legumes. Because of their protein-precipitating properties, grazing CT-containing legumes has long been known to eliminate bloat. Recently it has been proposed that the plant CT concentration needed to make forage bloat safe was 5 g CT/kg DM or greater. Most common legumes and grasses used in temperate agriculture have CT concentrations well below this value (Table 2). It will be a challenge to raise CT levels through plant breeding, but it is on the agenda as a breeding goal of the major grass and legume breeding companies worldwide. In temperate systems the challenge will be to integrate such legumes consistently into a sward growth pattern.

Both chicory (*Cichorium intybus L.*) and plantain (Plantago lanceolata L.) are suitable herbs for inclusion into swards managed under intensive grazing because of their high yield potential and ability to maintain sward quality mid-season (Cranston et al., 2015). The use of multispecies swards containing chicory and plantain is of particular interest on sites which are prone to soil moisture deficits, because of their deeper root systems and greater drought tolerance (Lee et al., 2015). Swards containing chicory and plantain can support higher levels of animal performance particularly in summer and autumn because of its ability to maintain sward quality in comparison to grass-white clover swards (Cranston et al., 2015). Grazing management, such as rotation length and post-grazing sward height, affects the growth, persistence and nutritive value of swards containing chicory and plantain (Lee et al., 2015). Multispecies swards containing clover and herbs can persist under grazing for 3–5 years, but are

Table 2 The extractable and bound condensed tannin content of legumes, grasses and herbs fed to ruminants in temperate grazing systems, measured by the butanol-HCl method

Forage	Extractable	Condensed Protein-bound	Tannin (g/kg DM) Fibre-bound	Total
Legumes				
Big trefoil *(Lotus pedunculatus)*	61	14	1	77
Birdsfoot trefoil *(Lotus corniculatus)*	36	9	2	47
Sulla *(Hedysarum coronarium)*	33	9	3	45
Sainfoin *(Onabrychis vicfolia)*	29			
Red clover *(Trifolium pratense)*	0.4	0.6	0.7	1.7
Lucerne *(Medicago sativa)*	0.0	0.5	0.0	0.5
Grasses				
Perennial ryegrass (*Lolium perenne*)	0.8	0.5	0.5	1.8
Herbs				
Chicory (*Chicorium intybus*)	1.4	2.6	0.2	4.2
Sheep's burnet (*Sanguisorba minor*)	1.0	1.4	1.0	3.4

suited to a 3–4-week rotation length with lax grazing to 8 cm (Cranston et al., 2015). These grazing guidelines are very much in contrast with those necessary for modern perennial ryegrass varieties. Their role in more modern grazing systems is currently under evaluation in a number of countries.

7 Grazing management to combat climate change: measurement issues

7.1 CH_4 measurement at grazing

Respiration chambers (RC) have been considered the gold standard for measuring enteric CH_4 emissions from farm animals, but this is only the case if RC are operated properly and recoveries are fixed and preferably close to 100%. Animals in RC must have stable daily feed intake. Approximately 30% of today's CH_4 emissions are as a result of the previous day's DMI. Daily variation in DMI can cause variation in CH_4 emissions.

Compared to housed animals, grazing animals have higher energy requirements due to the added cost of walking and grazing. There is continual debate about measurement methods – the standard method for CH_4 measurement is open-circuit respiration calorimeter chambers – yet this bears no relation to the activity of the grazing animal, when grazing the animal selects their chosen herbage, is exposed to the variances in climate, and has to forage, this is the opposite to the behaviour in an enclosed chamber, where the diet selection is somewhat compromised and supply is regular. Data from chambers cannot be applied to all farm situations; this is why the SF_6 and Green Feed (GF) system need to be more widely used in grazing systems.

Variability with the SF_6 method has been notoriously high but modifications by Deighton et al. (2014) addressed the most important sources of error, and the modified technique produced CH_4 measurements with accuracy similar to measurements using RC. Some of the variation with SF_6 seems intrinsic to the technique because the estimated CH_4 emission rate appears sensitive to factors that affect the proportions of exhaled and eructated air in the air samples and distance of the sampling point from mouth to the mouth/nostrils (Berends et al., 2014). A recently introduced technique for direct measurement of enteric CH_4 emissions is the automated head chamber system, GF, which was developed for spot sampling for exhaled and eructated gases (Zimmerman and Zimmerman, 2012). When used properly with repeated animal measurements, GF can be a reliable technique for measuring enteric CH_4 emissions from ruminants. An important prerequisite for decreasing uncertainty of the measurement when using GF is that all animals visit the unit at times that enable estimation of the diurnal pattern of CH_4 emission over successive 24 h period. For accurate daily emissions estimates, animal visits need to be distributed appropriately over the 24-h feeding cycle and a number of repeated days of measurements are required for each animal. Both GF and SF_6 methods are established techniques, and they can provide accurate estimates of enteric CH_4 emissions when properly used and calibrated. Direct comparisons of both techniques have shown acceptable agreement (Grainger et al., 2007; Huhtanem et al., 2018), and both provide valuable data required for grazing systems.

7.2 Life cycle assessment (LCA) with pasture-based diets

Comparisons of results between life-cycle assessment (LCA) studies is difficult due to differences in computation of life cycle inventories and choice of functional unit (de Boer, 2003; Van der Werf et al., 2009). In many previous LCA studies of grass and confinement systems, CH_4 from enteric fermentation was identified as the main cause of on-farm GHG emission (Casey and Holden, 2005). O'Brien et al. (2012) found that GHG emissions from cultivation of purchased concentrate and forage, emissions associated with the manufacture of inputs, that is, fertiliser used in the production of on-farm forages, were the main contributors to off-farm GHG emissions. Off-farm GHG emissions from confinement systems were more than double the off-farm GHG emission from grass-based system in the O'Brien et al. (2012) study, because of the greater quantity of concentrate feed. Consequently, total GHG emission per unit of milk produced was greater for the confinement system relative to the grass-based system.

Grassland systems can be further improved with strategic use of concentrate feed. Cederberg and Mattison (2000) suggested that the environmental impact of concentrate feed could be reduced by using domestic or regionally produced rapeseed meal rather than imported soya bean meal. Concentrate impacts can be reduced further by lowering the crude protein levels in concentrate by more appropriate diet formulation, in line with the chemical value of the diet, for example, avoiding the use of higher concentrate feeds. This would in turn have positive impacts on reducing N losses and further enhance N-use efficiency. There is much more to be learned about grassland and its seasonal quality fluctuations if there is more emphasis on pasture quality assessment across the growing season.

8 Conclusion

The continual decline in the area of permanent grassland in Europe is not assisting farmers in developing better grassland management practise and many countries are growing their dependence on imported feed (both forage and concentrate). The focus on farm needs to be to optimise the level of home-grown feed in the ruminant's diet.

None of the aspects of grassland management solely will deliver key movements in reducing CH_4 emissions; however, some of the gains that can be made are interacting with one another. High-grazed grass utilisation systems can be effective in this role, not all countries can adapt totally to this system type, but at different stages in the growing season, grass can be capitalised upon. Table 3 shows the effects of new and improved innovation and applications to reduce CH_4 emissions. Some of the improvements are gradual, improvement in animal productivity (based on proper national breeding goals), adapting grassland management, using appropriate grass varieties and clovers. The key aspect of such changes is that they have to be established inside the farm gate, herein is a crucial challenge for the grassland/ruminant sectors. Using home-produced feed, much of the efficiency in CH_4 can be harnessed as set out at the start of this chapter. Diet manipulation through adopting different grazing strategies that improve the forage quality available to the herds is

Table 3 Summary of methane mitigation strategies for grazing animals

Strategy	Potential CH_4 reduction	Technology availability/ feasibility	Production cost benefits
Improving animal productivity	20–30%	Feasible and practical	Increased feed cost Increased milk production Use of fewer animals Less feed per kg of milk
Forage species and maturity	20–25%	Feasible	Increased feed efficiency Increased milk production
Rotational grazing of animals/early grazing	9% or more	Feasible	Increased feed intake Increased milk production
Managed grazing of animals versus confined feeding	25% or more	Feasible needs more investigation	Cheaper feed costs Supplement use Reduced milk fat/ protein content Higher net return
Use of high-quality forages/pastures	25% or more	Feasible	Increased feed intake Increased milk production
Genetic selection (use of high net feed efficiency animals)	21%	Long-term feasibility	Decreased feed intake Increased feed efficiency

Source: Boadi et al. (2004).

readily available to farmers; however, many times it is not clearly advocated by the wider industry. Including grass and legumes in the diet of the grazing ruminant does not require major investment, but an overall change in mind-set. It is clearly a CH_4 mitigation factor and is positively viewed by the wider consumer population.

9 Where to look for further information

While the number of organisations across the world and indeed in Europe completing grazing research combined with measurement of GHG emssions is relatively small, there is a recently assembled research infrastructure consortium in Europe called the Smartcow network (www. Smartcow.eu). Ireland (Teagasc), United Kingdom (SRUC and University of Reading), France (INRAE), Belgium (CRAW), Denmark (Aarhus University), Germany (FBN), the Netherlands (Wageningen University) and Spain (IRTA) are collaborating partners of the consortium, their objective to ensure a commonality in research approach, measurements and protocols in methane research and other research intiatives. This type of infrastructure model will become more important into the future as it will allow a more smooth comparison of research techniques across countries, and this is really important with the techniques involved in GHG measurement at grazing.

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Chapter 9

The use of plant bioactive compounds to reduce greenhouse gas emissions from farmed ruminants

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1 Introduction

Livestock farming activities account for 14.5% of global greenhouse gas (GHG) emissions of anthropogenic origin (Gerber et al., 2013). Concerning the ruminant livestock sector, the largest contribution is from cattle and sheep mainly in the form of methane (CH_4) and nitrous oxide (N_2O) emissions, which represent, in carbon equivalent, 44% and 30% of GHG emissions, respectively. Methane has a digestive (enteric) origin in ruminants and is mostly eliminated into the atmosphere by eructation. During the microbial fermentation process of feeds in the rumen (bacteria, protozoa and fungi), hydrogen (H_2) is produced and is immediately used by archaea methanogens to reduce carbon dioxide into CH_4. Nitrous oxide is produced in the soil during microbial processes (nitrification and denitrification) of urinary nitrogen (N) (urea and ammonia) excreted by ruminants (De Klein and Eckard, 2008; Selbie et al., 2015). Both enteric CH_4 emissions and urinary N waste represent loss of dietary energy (2–15%) and N (75–95%) ingested, which could be otherwise available for animal production (Hristov et al., 2013b). Therefore, decreasing enteric CH_4 emissions and N excretion from ruminants is important for reducing the environmental impact of ruminant production and for improving feed efficiency and the sustainability of this sector.

Compounds produced by the secondary metabolic processes of plants have been used for medicinal purposes by humans since antiquity (Wink, 2015).

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Research on the use of compounds in animal production increased when the use of antibiotics as growth promoters was banned in Europe and other parts of the world in the mid-2000s. The potential use of plant bioactive compounds in animal nutrition to reduce CH_4 emissions and N waste is the subject of renewed interest as they are seen as a natural alternative to chemical additives and are well perceived by consumers. Some plant compounds have marked biological activity and, depending on their concentration in ruminant diets, can have positive or negative effects on animal responses. Plant bioactive compounds are promoted as improving health (are antiparasitic, reduce bloating and are antioxidant) and performance (N use efficiency). Conversely, they can decrease intake and diet digestibility (Mueller-Harvey, 2006) and can be toxic to animals (Reed, 1995).

We reviewed the current information on the use of plant bioactive compounds in ruminant nutrition to promote livestock farming that is not only more environment-friendly and efficient in the use of feed but also compliant with consumer demands for quality and safety in animal products. We focussed on the potential of plant bioactive compounds to mitigate enteric CH_4 production in ruminants and, when information is available, N waste. The main families of compounds considered as plant lipids are secondary compounds that are tannins, saponins, halogenated compounds and essential oils. Priority was given to information from *in vivo* studies by exploring the ability of plant compounds to positively modulate not only ruminant responses but also their mechanisms of action and utilization on farms. We selected two case studies showing the value of growing and using sainfoin forage in ruminant diets to decrease GHG emissions at the farm scale and combining dietary strategies with different modes of action to increase enteric CH_4 abatement. Future research on the use of plant bioactive compounds to reduce GHG emissions from farmed ruminants is also considered.

2 Families of plant bioactive compounds

2.1 Lipids

Lipids have a high nutritional value. The primary reason to use lipids in the diet of ruminants was to increase the potential production of animals and to improve the quality of meat and dairy products in terms of fat content and composition. However, an excessive dietary fat supplementation (> 7% dry matter [DM]) can affect microbial fermentation and fibre digestion in the rumen (Palmquist and Jenkins, 2017). The variable effects of lipids on ruminal fermentation are usually attributed to differences in their lipid structure (Bayat et al., 2018; Vargas et al., 2020). One factor is their degree of unsaturation because unsaturated fatty acids (from oleaginous oils or seeds and by-products such as residues from food processing plants) inhibit fermentation more than saturated fatty acids.

Commercial inert lipids (e.g. calcium salts of saturated fatty acids) are rumen bypass fats, which do not affect fibre digestion in the rumen at normal levels of supplementation in the diet.

Lipids are a proven dietary strategy for reducing CH_4 emissions from ruminants whose effectiveness depends on many factors, such as the dose, the source and the mode of distribution (Martin et al., 2010; Beauchemin et al., 2020).

Meta-analyses agree that the CH_4-mitigating effect of lipids are dose-dependent (Giger-Reverdin et al., 2003; Eugène et al., 2008; Grainger and Beauchemin, 2011; Doreau et al., 2011); the decrease in CH_4 emissions (g/kg DM intake [DMI]) varies between 1% and 7% per 10 g/kg of fat added DM in the diet. For low doses of added dietary lipids (< 2% DM), the mitigating effect was not systematic in cattle (Chung et al., 2011; Veneman et al., 2015). For high doses of added dietary lipids, the decrease is linear in cattle with rapeseed oil (0%, 5.4% and 9.5% added lipids; Jentsch et al., 1972) and coconut oil (0%, 1.3%, 2.7% and 3.3% added lipids; Hollmann et al., 2012) or quadratic in sheep with coconut oil (0%, 3.5% and 7% added lipids; Machmüller and Kreuzer, 1999). In dairy cows, many trials have shown the decreasing effect of lipids on methanogenesis (Martin et al., 2016; Bayat et al., 2018; van Gastelen et al., 2017) with different forage-based diets. The dose-response effect of extruded linseed (0%, 1.8%, 3.6% and 5.4% added lipids) on CH_4 emissions was more substantial with a corn silage-based diet compared to a hay-based diet (Martin et al., 2016). This more marked effect on methanogenesis was related to the adverse effects of lipids on animal performance (intake, digestibility and milk yield) with high doses of lipid supplementation.

The form of presentation of lipids also greatly influenced CH_4 output from dairy cows: inhibition of methanogenesis increased with the theoretical availability of linseed lipids in the rumen (oil > extruded seed > whole seed) (Martin et al., 2008). In practical conditions, extruded linseed is the most used form, because it is more readily available, easy to use, and less costly than oil and more efficient than crude linseed.

Concerning the source of lipids, conclusions of meta-analyses are not consensual: Grainger and Beauchemin (2011) did not find an effect of the source of lipids on CH_4 emissions, whereas medium chain and polyunsaturated fatty acids were reported to be more potent than others, according to the study of Doreau et al. (2011). In addition, the CH_4-mitigating effect of extruded linseed (2–3% added lipids) persisted for up to 1 year in dairy cows fed diets based on grazed pasture (80%) or grass silage (60%) (Martin et al., 2011). The persistency of this effect is very important for practical use.

The modes of action of lipids in the mitigation of ruminal methanogenesis are multiple (Martin et al., 2010). A common effect for all lipids is that when carbohydrates are substituted by lipids, as lipids are not fermented in the rumen, they do not contribute to H_2 production, unlike the carbohydrates they replace. Moreover, lipids have a toxic effect on some microbial populations (Popova et al., 2011; Vargas et al., 2020), more particularly on H_2-producing microbes (cellulolytic bacteria and protozoa), and reduce the metabolic activity of archaea methanogens by limiting H_2 availability and consequently CH_4 production. Ruminal biohydrogenation of unsaturated lipids may also help decrease methanogenesis, but the H_2 sink function of this biohydrogenation process was estimated to be negligible (i.e. 1–2%) based on stoichiometric (Czerkawski, 1986) and modelling (Mills et al., 2001; Giger-Reverdin et al., 2003) approaches.

Adding fat supplements in a proper dose to ruminant diets is a real opportunity to persistently reduce enteric CH_4 emissions without altering animal performance. This dietary strategy can be immediately implemented on commercial farms, especially if the quality of the meat and milk produced is improved, as is the case with unsaturated lipid sources. Most of the time, lipids are added in mixed diets as oil, oilseeds or food by-products, which limits their utilization for grazing ruminants. Notwithstanding, the use of lipids as feed ingredients in ruminant diets is relatively costly; if they are not locally produced, they may have a higher carbon footprint, which should be considered before adopting this approach. The combination of lipids with other dietary strategies has the potential to further reduce enteric CH_4 emissions (see Section 3).

2.2 Secondary compounds

2.2.1 Tannins

Among the polyphenolic compounds, special emphasis has been placed on the effects of tannins as they may act at several levels to reduce GHG emissions from ruminants.

First, the well-known ability of tannins to bind dietary proteins and reduce rumen proteolysis results in an increase of N duodenal flow and a shift from urinary to faecal N excretion (Aufrère et al., 2008; Theodoridou et al., 2010). As urinary N deposition results in N_2O emissions that are much higher than those arising from faecal N deposition (Luo and Kelliher, 2010), incorporating tannins in ruminant diets has great potential to decrease these emissions. In addition, a direct application of tannin extract (rather than feeding it) to barns reduces urease activity, thereby decreasing ammonia loss from dairy barn floors (Powell et al., 2011). Finally, if tannins are from forage legume species, such as sainfoin (*Onobrychis viciifolia*), birdsfoot trefoil (*Lotus corniculatus*) or sulla (*Hedysarum coronarium*), their ability to fix and transfer atmospheric N into the soil reduces the use of N fertilizers, which are a source of N_2O emissions through microbial nitrification and denitrification processes (Bouwman, 1996).

Second, abundant literature reports show that tannins in ruminant diets decrease enteric CH_4 emissions (review of Piluzza et al., 2014). The effect of these molecules on methanogenesis is highly variable between studies according to their nature (condensed or hydrolyzable), their chemical structure (molecular weight), especially the dose ingested by the animal, and the form of presentation. Given the extremely diverse structure of tannins in the plant kingdom, efforts have recently been made to understand their structure/activity relationships to facilitate their applicability (Mueller-Harvey et al., 2019). This aim was achieved, thanks to remarkable progress in the chemical analysis of both condensed (Zeller, 2019) and hydrolyzable tannins (Engström et al., 2019).

Using purified hydrolyzable (from chestnut and sumach) and condensed tannins (from mimosa and quebracho), Jayanegara et al. (2015) showed *in vitro* that hydrolyzable tannins have a greater effect in reducing CH_4 emissions with a less detrimental effect on digestibility than condensed tannins. Rira et al. (2019) reported the same conclusion with tropical forages tested *in vitro*: hydrolysable tannin-rich sources (*Acacia nilotica*) were more effective in supressing methanogenesis than condensed tannins-rich sources (*Calliandra calothryrsus* and *Leucaena leucocephala*). In addition, a combination of these plants did not highlight synergies between these two types of tannins. The type of molecular interaction that drives the protein-binding capacity of tannins is highly related to the structure of both tannins and proteins. Protein precipitation increases consistently with the mean degree of polymerization and tends to be higher with prodelphinidin-rich condensed tannins due to a greater number of potential hydrogen-bond participants available to interact with proteins (Zeller, 2019). The ability of hydrolyzable tannins to form insoluble complexes with the protein seems to be related not only to their molecular weight (oligomers are superior to monomers) but also to the type and number of functional groups (e.g. galloyl groups) in monomers (Engström et al., 2019).

Using purified condensed tannins of different structures from eight plants, Huyen et al. (2016b) showed *in vitro* that the proportion of prodelphinidins in condensed tannins had the largest effect on CH_4 production and fermentation characteristics, followed by the average polymer size. Other *in vitro* studies have shown that condensed tannins with a high degree of polymerization are more potent in lowering CH_4 production and the diversity and abundance of rumen methanogens (Hatew et al., 2016; Saminathan et al., 2016). Similarly, Baert et al. (2016) investigated *in vitro* how the degree of oligomerization of purified ellagitannins, an important family of hydrolyzable tannins, can influence their ability to alter ruminal fermentation including CH_4 production. They showed that large oligomers have more detrimental effects on gas production and volatile fatty acids (VFA) than small oligomers, while being similarly effective in their ability to decrease CH_4 production.

The meta-analysis conducted by Jayanegara et al. (2012), including data from a total of 30 experiments (both *in vitro* and *in vivo*), helped to partly clarify the underlying mode of action of tannins on methanogenesis. These authors reported that related CH_4 reduction is associated with reduced OM digestibility, especially fibre, because of a decreased number/activity or impaired substrate adhesion of fibrolytic microbes. This inhibitor effect of tannins on fibrolysis was more marked for condensed tannins than for hydrolysable tannins in dairy ewes (Buccioni et al., 2015). Recently, Costa et al. (2018) reported in sheep that gram-positive specialized fibrolytic bacteria (*R. albus, R. flavefaciens* and *B. proteoclasticus*) were more affected by condensed tannins than gram-negative bacteria (*F. succinogenes*, *S. ruminantium* and *P. bryantii*), with a decrease in rumen volatile fatty acids concentration, mostly acetate.

Methane emissions also declined when expressed per kg of digested organic matter (DOM), suggesting that other mechanisms account for the anti-methanogenic activity of tannins (Jayanegara et al., 2012). Tannins have been shown to directly inhibit H_2 using methanogens in the rumen of sheep (Liu et al., 2011) and beef cattle (Yang et al., 2017). This direct effect of tannins on methanogens microbiota, without affecting fibre digestion, would be more specific to hydrolysable tannins as reviewed by Vasta et al. (2019).

The potential of tannins to reduce methanogenesis has been widely reviewed in both *in vitro* and *in vivo* studies, thus highlighting the large variability of data (reviews of Piluzza et al., 2014 and Vasta et al., 2019). Archimède et al. (2016) observed a linear relationship between the condensed tannins content of three tropical-rich plants (*Glyricidia sepium, Leucaena leucocephala and Manihot esculenta*) and CH_4 reduction *in vivo*. The potential of mitigation ranged between 13% and 36% in sheep fed a forage diet containing between 1.5% and 4.0% DM of condensed tannins. The authors also reported better palatability (and intake) of tannin-rich tropical plants and a strong decrease in CH_4 emissions in animals fed the plants as pellets. Concerning hydrolysable tannins, their potential of mitigation ranged between 10% and 25% with sheep (Liu et al., 2011) and 11–30% with beef cattle (Yang et al., 2017) fed diets containing 1–3% DM of tannins from chestnut and pure hydrolysable tannins, respectively.

Few general equations of CH_4 prediction concerning tannins, mostly derived from *in vitro* trials, are available because of the diversity of the chemical analysis methods and the types of tannins (Jayanegara et al., 2012). We conducted a quantitative review of the literature by meta-analysis to assess the specific effect of tannins (condensed or hydrolyzable) on *in vivo* CH_4 emissions in ruminants (Eugène et al., 2019). Using an existing database (Methafour, INRA, 2018) on the effect of forages fed to ruminants on CH_4 emissions, we were able to significantly improve the accuracy of Eq. [1] based on the animal-feeding level and forage diet composition to predict CH_4 emissions, by taking into account the tannin content of the forage diets as mentioned in Eq. [2]:

- Eq. [1] for forage diets

$$CH_4 / DOM = 34.95 - 4.05FL + 0.027NDF - 0.010DOM$$
$$(n = 412, nexp = 153, RMSE = 3.1)$$

- Eq. [2] for forage diets containing tannins

$$CH_4 / DOM = 34.26 - 3.96FL + 0.027NDF - 0.008DOM - 1.72Log_{10}(1 + TAN)$$
$$(n = 398, nexp = 147, RMSE = 3.1)$$

where CH_4/DOM is the CH_4 production (g/kg DOM), FL is the feeding level (kg DM intake, % liveweight), NDF is the NDF content (g/kg DM), DOM is the DOM content (g/kg DM) and TAN is the tannin content (g/kg DM) of the diet, which is transformed on a logarithmic basis to account for its largely abnormal distribution (Sauvant et al., 2018).

The coefficients of regression of all variables remain stable between Eq. [1] and Eq. [2], highlighting the specific effect of tannins on methanogenesis. Based on current scientific knowledge, we propose to use the coefficient of TAN in Eq. [2] to evaluate the average quantitative effect of tannins *in vivo* on CH_4 emissions in all types of diets (Fig. 1). Our results confirm that CH_4 mitigation increases with the dose of tannins in the diets (Jayanegara et al.,

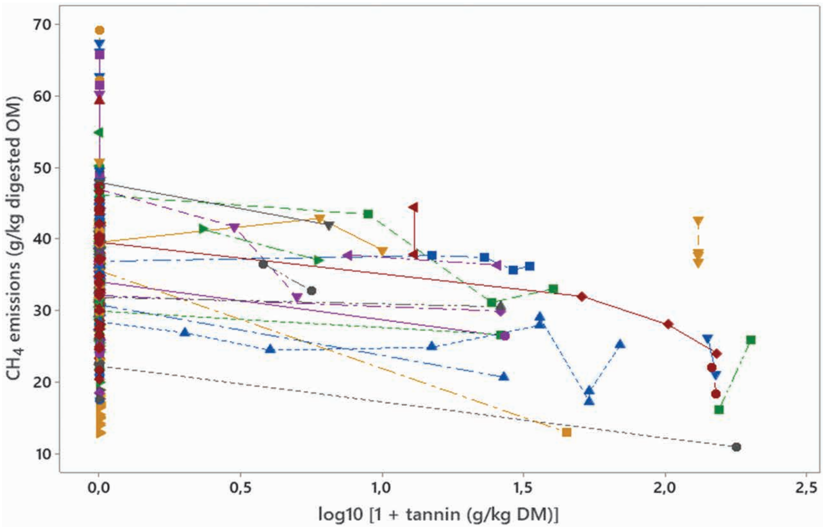

Figure 1 Relationship between methane emissions (g/kg DOM) and tannin content (g/kg DM) in the diet.

2012). Unfortunately it is not possible to give a minimum threshold of tannins content to observe an effect on methanogenesis because it is modulated by the FL, NDF and DOM contents of the diet.

Tannins are consumed by animals as plants or added as an extract to rations. The use of fodder-containing tannins is particularly relevant for grazing ruminants since many forage legumes are rich in tannins. New insights into the chemical structure of tannins help to explain the inconsistencies of the effects on protein-binding ability and on CH_4 emissions reported in the literature. Despite this progress, there are still questions to address concerning the use of tannin-rich resources on farms. Considering the variability in tannin activity, one solution could be the production of batches of feeds or plant extracts analysed before their marketing. Also, in the context of agro-ecological ruminant production systems, the use of integrated solutions such as tannin-containing legumes offers opportunities to act at multiple levels of GHG production (see Section 3).

2.2.2 Saponins

Saponins are secondary metabolites present in seeds, leaves and roots of a broad variety of plants. They are usually classified into two major classes, triterpenoids (soy, pea, garlic, sunflower and panama bark) and steroid glycosides (e.g. oat, eggplant, tomato, yucca and fenugreek), but Vincken et al. (2007) refined their classification in 11 main categories based on their carbon skeleton. Despite some negative effects upon feeding in animal nutrition (increased membrane permeability of erythrocytes and enterocytes, or impaired animal production and reproduction; reviews of Addisu and Assefa, 2016; Francis et al., 2007), saponins can have beneficial effects on rumen fermentation and animal health when used in a proper dosage. Among others, saponins can decrease *in vivo* degradability of feed protein, avoid N accumulation and increase efficiency of microbial protein synthesis in the rumen (Francis et al., 2007; Patra and Saxena, 2009). In addition, saponins from *Quillaja saponaria* (known as quillaja saponins), *Yucca schidigera* (known as yucca saponins) and *Camellia sinensis* or *assamica* (known as tea saponins) have been extensively studied for their mitigating effect on rumen methanogenesis. Other saponin sources have shown interesting CH_4-mitigating impacts: mangosteen peel powder (Wanapat et al., 2014) and alfalfa saponins (Klita et al., 1996), but these effects need to be confirmed.

The underlying mechanism mainly involves an inhibitory effect towards rumen microbes and more particularly protozoa, which produce large amounts of H_2 and are known to live in symbiosis with methanogenic archaea (Guyader et al., 2014; Morgavi et al., 2010). The membrane-disrupting activity of saponins would explain their toxic effect on protozoa through the formation of complexes with sterols present in the protozoal cell wall, thereby inducing cell lysis (Morgavi et al., 2010).

The CH_4 mitigation potential of saponins depends on the dose and source of saponins (Patra and Saxena, 2010). Most studies on this subject have been conducted *in vitro*. In a meta-analysis combining 23 studies, Jayanegara et al. (2014) reported a linear inhibiting dose-response effect of saponins towards methanogenesis (tested dosage between 0% and 0.6% DM). Compared with quillaja and tea saponins, yucca saponins induced the greatest reduction in CH_4 expressed as mL per unit of incubated substrate. However, when expressed as a percentage of total gas produced, all tested saponin sources were statistically similar and produced less CH_4 than the control. However, *in vivo* results are not as clear. Using up to 1.4% DM of quillaja saponins in the diet, Pen et al. (2007) and Holtshausen et al. (2009) did not observe a difference in CH_4 emissions of sheep and dairy cows, respectively. Patra and Saxena (2009) summarized published papers studying the *in vivo* effect of saponins, including yucca saponins, on fermentation parameters. Only two articles out of five showed a significant reduction (−14% in Santoso et al., 2004; −7% in Wang et al., 2009) in methanogenesis with sheep fed a diet containing 1.3% DM of yucca saponins. Similarly, the effect of tea saponins seems to be highly variable. With similar dosages comprising between 0.4% and 0.8% DM, three articles reported a decrease in CH_4 yield (g/kg DMI) on adult sheep (−17% in Yuan et al., 2007; −26% in Zhou et al., 2011) and on lambs (−69% in Mao et al., 2010). More recent papers showed an absence of effect on steers (Li and Powers, 2012) and non-lactating cows (Guyader et al., 2015), or even an increase in CH_4 emissions on lactating dairy cows (+18% in Guyader et al., 2017) with the same dosages. Concerning the effect of tea saponins on other digestion parameters, data are scarce but mostly indicate an absence of effect on nutrient digestibility or N balance. However, milk yield in dairy cows (Guyader et al., 2017) and average daily weight gain in growing steers (Li and Powers, 2012) decreased (−18% and −80%, respectively) as a consequence of lower intake in both studies (−12% and −27%, respectively). However, 0.4% tea saponin in a Chinese wild rye-based diet did not affect feed intake or growth of lambs (Mao et al., 2010).

Four main reasons might explain the variable effect of saponins on methanogenesis and limit their utilization in animal nutrition. The quality of saponins is an important criterion for their CH_4-mitigating efficiency (Jayanegara et al., 2014). Plant maturity, geographical area of production and extraction methods are three parameters affecting the final concentration and quality of saponins (Li and Powers, 2012). Denaturation of saponins might also be possible during pelleting processes. Indeed, a modification of the miscellaneous structure of quillaja saponins was observed after heating from 20°C to 60°C (Mitra and Dungan, 1997). Guyader et al. (2015) assumed that denaturation was one possible explanation for the lack of effect on CH_4 production of tea saponins fed to non-lactating dairy cows.

The transient effect of saponins on rumen microbes is another limitation for their utilization. After adaptation, rumen bacteria are able to separate the active compound of saponins (sapogenin) from the sugar moiety, leading to their inactivation (Ramos-Morales et al., 2017). Newbold et al. (1997) supplemented sheep with saponins from foliage of an African multipurpose tree, *Sesbania sesban*. Protozoa concentrations dropped by 60% after 4 days, but the population recovered after 10 days. However, the time for adaptation of rumen microbes seems to be dependent upon the source of saponins: the anti-protozoal effect of saponins from *Sapindus rarak* was persistent over 105 days in sheep (Wina et al., 2006). The chemical modification of their structure to avoid microbiota adaptation may maximize the CH_4-mitigating potential of saponins (Ramos-Morales et al., 2017).

The CH_4-mitigating response seems to be dependent on the composition of the basal diet. For instance, in a study with young Holstein males, Wang et al. (2019) concluded that changes in the ruminal microbial community with tea saponin supplementation were different between alfalfa-, hay- or soybean hull-based diets. Given that the protozoal community is strongly affected by the basal diet, Patra and Saxena (2009) assumed that the diet-dependent effect of saponins is related to their selectivity for specific protozoal species.

Finally, the last challenge in saponin utilization is their impact on feed intake. Reduced intake has been reported following dietary supplementation with tea saponins in lactating dairy cows (Guyader et al., 2017) and steers (Li and Powers, 2012). Nevertheless, this drawback is not systematic: among the 43 papers compiled by Patra and Saxena (2009), who did not include recent articles on tea saponins, only two reported a decrease in feed intake with yucca saponin supplementation.

The effects of saponin supplementation in the diets of ruminants are highly contrasted. The conditions in which yucca, quillaja and tea saponins reduce CH_4 emissions from ruminants must be refined (optimal dose, long-term persistency). In addition, given the wide variety of saponin structures, screening of other plants might highlight the beneficial effect of new sources available for localized markets. Before adoption by farmers, the potential effects of saponins on digestion efficiency and zootechnical performance should also be investigated in depth.

2.2.3 Halogenated compounds

Halogenated products (e.g. bromoform, dibromomethane, dichloromethane, bromochloroacetic acid, etc.) exist naturally in seaweed at different concentrations, and much more in red and brown algae than in green ones.

These compounds may be produced as defense against disease and marine herbivores, anti-oxidants or by-products of metabolic processes (Keng et al., 2020).

Different macroalgae have been shown to decrease *in vitro* CH_4 production effectively (Dubois et al., 2013; Machado et al., 2014; Kinley and Fredeen, 2015). Among 20 tropical species screened, the red macroalgae, *Asparagopsis taxiformis*, was identified as the most efficient (Machado et al., 2014). Low doses (2% OM incubated) of *A. taxiformis* almost eliminated *in vitro* CH_4 production (Machado et al., 2016a), without any effect on forage digestibility (Kinley et al., 2016) and without compromising other fermentation parameters at a 5% OM supplementation rate (Roque et al., 2019a).

The CH_4-mitigating effect of red seaweed *Asparagopsis* spp. *(A. taxiformis and A. armata)* was recently confirmed in three *in vivo* trials. Li et al. (2016) reported a consistent (over a 72-day period) and dose-dependent reduction in CH_4 emissions (−50% to −80%) when adding *A. taxiformis* at 1–3% of diet OM, respectively. In dairy cows, adding *A. armata* at 0.5% and 1% of diet OM reduced CH_4 emissions (−26% and −67%, respectively) over 21 days while compromising animal performances (milk yield and intake) only at the high dose (Roque et al., 2019a). A recent experiment in feedlot beef cattle, *A. taxiformis* was tested in a high grain diet at three inclusion levels (0.05%, 0.10% and 0.20% of diet OM) over a 90-day period (Kinley et al., 2020). Steers receiving 0.10% and 0.20% *A. taxiformis* demonstrated decreased CH_4 emissions up to −40% and −98% and demonstrated weight gain improvements of +53% and +42%, respectively. There was no negative effect on daily feed intake, feed conversion efficiencies or rumen function, and no residues or changes in meat-eating quality were detected.

Bromoform is the most abundant natural product in *Asparagopsis taxiformis* and thus has been identified as the compound involved in CH_4 reduction, even if a combination of the different compounds may play a role in this reduction (Machado et al., 2016b). Halogenated compounds in *Asparagopsis taxiformis* appear to act as structural analogues of coenzyme M and thus inhibit the final step of the methanogenesis pathway (Liu et al., 2011). It has been shown that the decrease in abundance of methanogens in the rumen was positively correlated with the decrease of methanogenesis and increase in H_2 emissions (Machado et al., 2018; Roque et al., 2019a). Emissions of bromoform into the atmosphere may occur during the growth of seaweed or during desiccation processes (Keng et al., 2020), which would prevent – or at least greatly hamper – the farming of red seaweed on a commercial basis.

Macroalgae have a tremendous potential to inhibit methanogenesis in ruminants at low doses of supplementation. *Asparagopsis* spp. are the most effective species. Further investigations are required to confirm a long-term persistency effect on methanogenesis and long-term safety in animal responses before adoption by farmers. In addition, widespread use of red seaweed for animal nutrition raises concerns about their contribution to biogenic halocarbon emissions and their impact on the environment (i.e. ozone depletion related to bromoform). The carbon footprint of each step of algae production (harvesting, drying process, delivery, etc.) also needs to be considered for upstream emissions.

2.2.4 Essential oils

In addition to the compounds considered in previous sections, there are other plant bioactive compounds, collectively known as 'essential oils', that have the potential to mitigate CH_4 and ammonia production in ruminants (Cobellis et al., 2016). The name essential oil is not specific; it mainly comprises a diverse group of terpene and phenylpropene compounds as well as organo-sulphur compounds (Benchaar and Greathead, 2011). There are thousands of compounds that are qualified as essential oils, and many of them have been tested *in vitro* (reviewed by Calsamiglia et al., 2007; Hart et al., 2008; Benchaar and Greathead, 2011; Cobellis et al., 2016). However, for multiple reasons only a handful of these compounds have been pursued in animal studies. Many of the compounds tested decreased methanogenesis through a general reduction in feed degradation and fermentation in the rumen and, therefore, are not further considered in this chapter. The effect of some compounds was observed at high doses not compatible with their incorporation (as an extract or as the plant containing the active component) in the diet. In addition, for some compounds or plants, there are issues of toxicity, palatability, cost and availability that preclude their utilization even for experimental purposes. Further, the majority of *in vivo* studies have tested the effects of essential oils on general production parameters and only a handful of them included measurements of enteric CH_4 emissions (Cobellis et al., 2016). In this section, we will focus on those plant secondary compounds that were tested *in vivo* for their anti-methanogenic activity.

Most metabolites tested to reduce CH_4 production in ruminants are naturally produced by plants to fend off microbial invasion. Compounds that are effective *in vitro*, such as eugenol, which is abundant in the essential oil of clove, and cinnamaldehyde, which is abundant in cinnamon (Macheboeuf et al., 2008; Patra and Yu, 2012), did not reduce CH_4 emissions when tested on dairy cows (Benchaar, 2015; Benchaar et al., 2015). Carvacrol is a monoterpene with a phenol ring structure that is abundant in oregano and thyme. Oregano (*Origanum vulgare*) leaves fed to lactating dairy cows at doses of 250, 500 or 750 g/d decreased CH_4 production by up to −40% for the medium dose (Tekippe et al., 2011; Hristov et al., 2013a). But the CH_4 measurements were done up to 8 h after feeding, and the authors noted that 24-h continuous measurement is needed to validate the results. In another study on dairy cows, the use of oregano extract mixed into the diet at 0.056% DM tended to reduce CH_4 yield (g/kg DMI) by −22% (Kolling et al., 2018).

Flavonoids are a class of plant secondary compounds that have antimicrobial, anti-inflammatory and anti-oxidative functions. These compounds have been extensively studied in animal nutrition (Olagaray and Bradford, 2019). In studies with ruminants, supplementation of diets with flavonoids from mulberry decreased CH_4 yield (g/kg DMI) by −11% in sheep (Ma et al., 2017). The main flavonoids of mulberry are quercetin glycosides and rutin, a glucorhamnoside of quercetin (Ju et al., 2018). In contrast, the use of pure rutin or rutin contained in buckwheat seeds did not have any effect on CH_4 emissions in dairy cows (Stoldt et al., 2016). Catechins, flavonoids contained in green tea leaves, decreased CH_4 emissions (g/kg digestible DMI) in dairy cows by −18% (Kolling et al., 2018). Green tea also contains saponins that may have a synergistic effect in reducing CH_4. Notwithstanding, a commercial purified catechin extract linearly decreased CH_4 emissions in sheep by 7–13% (Aemiro et al., 2016). Catechins have known antimicrobial activities including a toxic effect on protozoa (Aemiro et al., 2016), but they are also known H_2 sinks that can compete with CH_4 production in the rumen environment (Becker et al., 2014). For flavonoids, in general, it is noted that those that have anti-inflammatory functions in the host animal are not effective in reducing CH_4 emissions or modulating microbial fermentation in the rumen (Olagaray and Bradford, 2019).

Sinigrin is a glucosinolate found in some plants of the Brassicaceae family, such as black mustard and horseradish, which is naturally converted to allyl isothiocyanate when the plants are processed (Mohammed et al., 2004). The latter compound is responsible for the strong flavour of horseradish and low palatability if used as a feed additive (Mohammed et al., 2004). A coated additive would avoid the problem of palatability and provide a gradual release of the sinigrin. A cyclodextrin-coated horseradish oil added to the diet of steers decreased CH_4 emissions by −19%. Although the mechanism of action is not well understood, the number of methanogens also decreased significantly. A parallel *in vitro* study showed a large increase in H_2 associated with CH_4 reduction (Mohammed et al., 2004), similar to that observed with specific inhibitors of methanogens or methanogenesis, such as garlic or halogenated compounds.

A particular mention is made for organo-sulphur compounds from garlic (*Allium sativum*). Sulphur compounds in garlic have both general antimicrobial properties and are specific inhibitors of the enzyme hydroxymethylglutaryl-CoA (HMG-S-CoA) reductase, which is essential for the production of the cell wall of archaea methanogens. These compounds remarkably reduce CH_4 production *in vitro* (reviewed by Hart et al., 2008; Benchaar and Greathead, 2011). However, there are few reports describing the *in vivo* use of garlic oil or diallyl disulphide, the main component of garlic oil. A decrease of about

−8% (g CH_4/kg digestible OM intake) was reported in sheep supplemented with garlic extract (Ma et al., 2016), whereas garlic leaves, which are normally discarded after harvesting the bulbs, reduced emissions by −10% (g CH_4/kg DMI) in sheep (Panthee et al., 2017).

Garlic oil combined with linseed oil reduced emissions in lambs, but the effect cannot be ascribed solely to garlic oil (Saro et al., 2018). Similarly, dried garlic combined with mangosteen peel rich in tannins and saponins reduced CH_4 emissions in cattle (Manasri et al., 2012), but the effect is confounded. In contrast, no effect was observed in a study with diallyl disulphide, garlic oil or raw garlic (Klevenhusen et al., 2011; Patra et al., 2011). More recently, a commercial mixture of garlic and citrus extracts (Mootral) was tested in dairy cows with positive results (Roque et al., 2019b; Vrancken et al., 2019). These results are encouraging but should be confirmed with further studies. For instance, in the work of Roque et al (2019b) the reduction in CH_4 was observed in the last week of the 12-week study, but not before. In addition to the product mentioned previously, there are several commercial products based on mixtures of essential oils that have been tested for their CH_4-reducing activity. The most tested are CRINA Ruminants (DSM; mixture of essential oil components) and Agolin Ruminant (Agolin; mixture of coriander oil, eugenol, geranyl acetate and geraniol, among others) and XTRACT Ruminant (Pancosma; mixture of cinnamon, cloves and capsicum oleoresin from chili peppers). The first product showed no effect on CH_4 emissions in beef cattle (Beauchemin and McGinn, 2006; Tomkins et al., 2015). The effect of Agolin Ruminant on dairy cows was recently evaluated in a meta-analysis (Belanche et al., 2020). A total of 23 *in vivo* experiments and on-farm studies were identified in which the additive was supplemented at 1 g/d/cow. Of these, nine had records of enteric CH_4 that showed an average decrease of −8.8% in CH_4 production (g/d), of −12.9% in CH_4 yield (g/kg DMI) and −9.9% in CH_4 intensity (g/kg milk) without a negative effect on feed digestibility or milk yield. The effects were observed only after an initial period of adaptation of at least 4 weeks.

Anacardic acid is an akylphenolic compound that is found in the shell of the cashew nut. It has antimic activity, particularly against gram-positive bacteria. In the rumen, it decreased the numbers of H_2- and formate-producing bacteria such as *Ruminococcus flavefaciens*, *Butyrivibrio fibrisolvens* and *Treponema bryantii*, whereas succinate-producing bacteria such as *Prevotella* spp., *Selenomonas ruminantium*, *Anaerovibrio lipolytica* and *Succinivibrio dextrinosolvens* increased (Shinkai et al., 2012; Konda et al., 2019). Methanogen numbers also decrease with changes in the community composition (Shinkai et al., 2012; Kang et al., 2018). The use of cashew nut shell liquid (CNSL) as a feed additive reduced CH_4 emissions in Holstein cows by −19% and −38% in a dose-dependent manner (Shinkai et al., 2012). The anti-methanogenic effect of CNSL was also observed in Thai native cattle and buffaloes (Konda et al., 2019). These changes in CH_4 were observed along with increases in propionate and decreases in acetate in the rumen. The technical grade CNSL (t-CNSL) is the main by-product of the cashew industry that does not contain anacardic acid, as during the production process it is converted into cardanol. The utilization of t-CNSL was not as effective at reducing enteric CH_4 emissions in dairy cows (Branco et al., 2015), suggesting that anacardic acid is the main active component in CNSL.

Essential oils used to mitigate CH_4 emissions and N waste have been extensively studied mainly *in vitro*, but validation of the results *in vivo* is still scarce. The use of cocktails of molecules in most *in vivo* studies makes it difficult to identify active molecules and to understand the mechanisms of action. Work still needs to be done to demonstrate the effectiveness of certain essential oils and to consider their synergistic or antagonistic interaction with other compounds. For most compounds, there is also a need to assess their efficacy in the long term, not only in reducing CH_4 emissions but also in the production and welfare of animals, to facilitate adoption by farmers.

3 Case studies

3.1 Sainfoin, a traditional forage legume containing condensed tannins

Several methods can be used to deliver tannins to the animals, using crude extracts from plants, by-products, wood or whole plants. Agro-ecological and local solutions that integrate several dimensions of ruminant nutrition can decrease GHG emissions at multiple levels, while improving protein self-sufficiency and reducing inputs such as fertilizers and drug treatments (Soussana et al., 2015). In this context, one option is to include legume species containing tannins in animal diets. Recently, a European multidisciplinary research consortium of agronomists, plant breeders, ruminant nutritionists, veterinarians and experts in tannin chemistry focussed on sainfoin, a traditional forage legume (Mueller-Harvey et al., 2019). Here, we present the specific results of this project regarding the potential of sainfoin to decrease GHG emissions (Fig. 2).

A first interesting result was obtained at the field level, where symbiotic N fixation by sainfoin was shown to be comparable with major N-fixing species such as white and red clover without treatment with a commercial rhizobia product (Malisch et al., 2017). The authors concluded that sainfoin has great potential for cropping grass-legume mixtures with increased forage yields, especially when cutting frequency and N fertilizer input are low. These results indicate that sainfoin is relevant to local production of high-protein forage without applying excessive N fertilization leading ultimately to N_2O emissions.

When ensiled with grass, sainfoin preserves silage quality via increased fermentation intensity and reduction in protein degradation in the silos (Copani

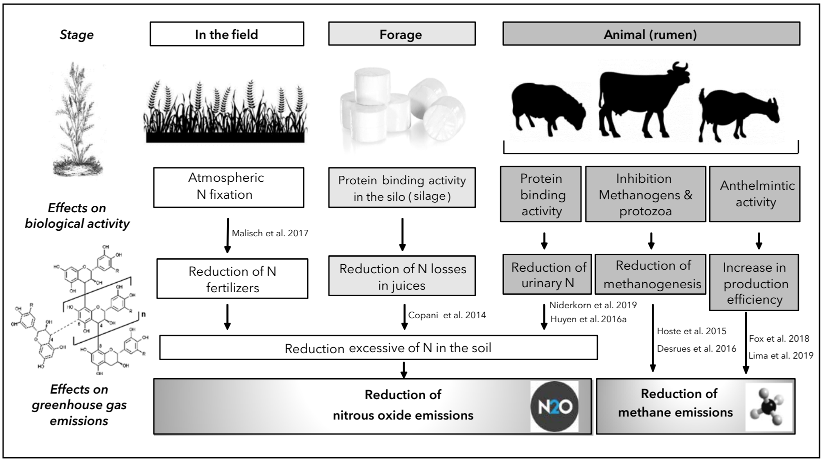

Figure 2 Multiple effects of growing and using sainfoin in ruminant diets to decrease greenhouse gas emissions.

et al., 2014). This was shown by a lower proportion of soluble N and ammonia (relative to total N) in silage compared to grass silage, which reduces N losses in fermentation juices and decreases the adverse impacts on the environment.

At the animal level, the decrease in CH_4 yield (g/kg DMI) and changes in N partition when sainfoin is incorporated in the diet were consistently observed in sheep (Niderkorn et al., 2019) and dairy cows (Huyen et al., 2016a). A particularly interesting result was obtained in dairy cows when a sainfoin-containing diet reduced CH_4 yield and diet digestibility of fibre but improved milk yield compared to the same diet in which sainfoin was replaced by grass. The authors hypothesized that sainfoin may redirect metabolism towards body protein accretion at the expense of body fat (Huyen et al., 2016a). Sainfoin-condensed tannins were shown to have anthelmintic activities in both small (Hoste et al., 2015) and large ruminants (Desrues et al., 2016), showing that a large spectrum of these compounds counteracts infection by gastrointestinal nematodes. This effect may help to decrease GHG emissions by animals, as shown by recent results indicating that parasitism increases CH_4 emissions in sheep (Fox et al., 2018; Lima et al., 2019).

3.2 Additive effect of different anti-methanogenic dietary strategies, a proof of concept

Lipids have emerged as a persistent option for mitigating enteric CH_4 emissions from ruminants (Doreau et al., 2014). However, their potential mitigation is moderate (~20%) if used at a suitable dose avoiding negative effects on animal performance (see Section 3.1).

As proof of concept, we tested whether it was possible to increase the CH_4-mitigation potential of lipids (linseed oil) by combining them with another dietary strategy (nitrate) with a different mode of action on the metabolism of H_2 in the rumen. In a meta-analysis, we reported that lipids may be relevant in reducing H_2 production (via reduction of protozoa), whereas nitrate may stimulate H_2 consumption (H_2 sink) by a competitive pathway to methanogenesis (Guyader et al., 2014). We assumed that simultaneous manipulation of H_2 production and H_2 utilization allows a greater reduction in CH_4 emissions than when acting on a single pathway. To test this hypothesis, we tested the effect of linseed oil and nitrate fed alone or in combination on CH_4 emissions and digestive processes in non-lactating cows.

The daily kinetics of CH_4 emission measurements clearly showed an additive effect of the dietary strategies (−32% reduction for linseed + nitrate vs. −17% and −22% reduction for linseed and nitrate fed alone, respectively). Linseed oil supplementation reduced CH_4 emissions throughout the day compared to the control diet, while nitrate had a transient but marked action for 3 h post-feeding. Combination of the strategies cumulated the two modes of actions (Fig. 3).

In addition, we showed that linseed oil plus nitrate fed to lactating cows for 2 months induced a constant reduction of CH_4 emissions (−29% g/kg DMI), without any effect on digestibility of nutrients, N balance and milk performance. This persistent effect showed the absence of adaptation of rumen microbiota. However, the energetic benefits from the decreased CH_4 emissions did not appear beneficial for dairy cows (Guyader et al., 2016).

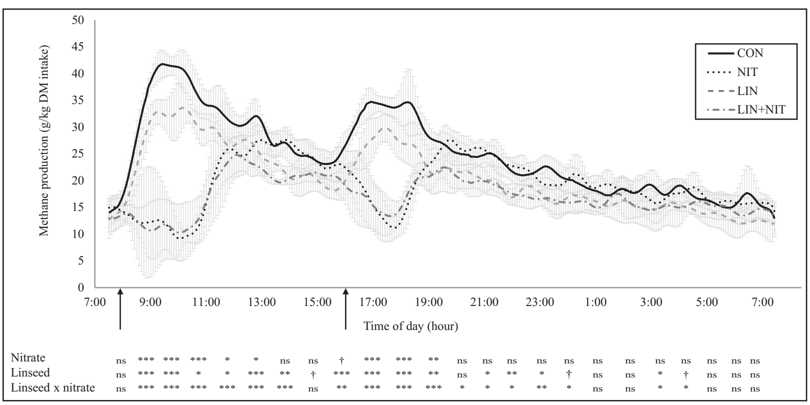

Figure 3 Daily methane production pattern of non-lactating cows fed four different diets containing linseed oil and calcium nitrate alone or in association (n = 4). Treatments consisted of control diet (CON), CON plus 3% calcium nitrate (NIT), CON plus 4% linseed oil (LIN) and CON plus 4% linseed oil and 3% calcium nitrate (LIN+NIT). The arrows indicate time of feeding. Error bars indicate SD. Adapted from Guyader et al. (2015).

Our work confirmed the initial working hypothesis that combining dietary strategies with different mechanisms of action to reduce H_2 availability in the rumen reduces methanogenesis more markedly than when lipids are fed individually. This opens up a range of possibilities for designing new strategies to increase CH_4 abatement (Beauchemin et al., 2020).

4 Outstanding questions and future trends in research

Considering the current health crisis, integrating animal production into a 'one health' approach is more relevant than ever: it s important to consider health care for humans, animals and the Earth in a systemic and integrated way at local, national and global levels. In this context, the use of plant bioactive substances from local resources in animal nutrition is a strong 'card to play' for promoting efficient and safe livestock farming to feed populations, while minimizing its environmental impact.

Many feed resources contain lipids and secondary compounds that are likely, if used properly, to improve animal performances and health, decrease enteric CH_4 emissions and N waste, and improve the quality of animal products. However, many questions have to be addressed before widespread application at the farm level. One demand of stakeholders concerns the possibility of standardized resources with a guaranteed content in active principle. This implies further research to characterize active molecules and their mode of action in order to design and evaluate new feeding strategies that are more efficient in minimizing GHG emissions from ruminants.

Evaluation of new resources based only on the traditional feed value is not sufficient. A multicriteria approach of these resources, as well as practices (crop growing, conservation, processing, feed delivery), is required to consider all animal responses, without neglecting the evaluation of the cost-benefit ratio for farmers. Another challenge is to develop resources with valuable properties for pasture-based systems in order to better integrate the context of agro-ecological ruminant production.

5 Where to look for further information

5.1 Key articles or books

- Vasta, V., Daghio, M., Cappucci, A., Buccioni, A., Serra, A., et al. 2019. Invited review: Plant polyphenols and rumen microbiota responsible for fatty acid biohydrogenation, fiber digestion, and methane emission: Experimental evidence and methodological approaches. *J. Dairy Sci.* 102, 3781–3804. https://doi.org/10.3168/jds.2018-14985.
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- Beauchemin, K. A., Ungerfeld, E. M., Eckard, R. J., Wang, M. 2020. Review: Fifty years of research on rumen methanogenesis: Lessons learned and future challenges for mitigation. *Animal*, 14:S1, s2–s16. https://doi.org/10.1017/S1751731119003100.

5.2 Key conferences

- International Symposium on the Nutrition of Herbivores, Clermont-Ferrand, FRA (2018-09-02—2018-09-06). Proceedings of the 10th International Symposium on the Nutrition of Herbivores in *Advances in Animal Biosciences*, 9(3), 337–786. doi:10.1017/S2040470018000146.
- International Symposium on Ruminant Physiology, Leipzig, DEU (2019-09-03—2019-09-06). Proceedings of the XIIIth International Symposium on Ruminant Physiology in *Advances in Animal Biosciences*, 10(3), 369–649. doi:10.1017/S2040470019000037.
- Greenhouse Gases and Animal Agriculture conference, Iguassu, BRA (2019-08-04—2019-08-10). Proceedings of the VIIth Greenhouse Gas and Animal Agriculture Conference. http://www.ggaa2019.org/sites/default/files/proceedings-ggaa2019.pdf.

5.3 Major international research projects and networks

- LegumePlus (2012–2015): European project aiming to optimize plant polyphenols in legumes for ruminant nutrition and health plus environmental sustainability (Project Number PITN-GA-2011-289377).

- Pro YoungStock (2018–2021): European CORE Organic Co-fund Project aiming to collect, develop and assess natural feeding strategies increasing dairy livestock welfare (Preject FiBL 50090).
- SmartCow (2018–2022): European project on infrastructures for increased research capability and innovation in the European cattle sector. Joint research activities are focussed on improving the quality and ethics of research services through advances in the capabilities to investigate feed efficiency and emissions in cattle at a large scale and to valorize data from sensors monitoring nutrition, health and behaviour.

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Chapter 10

The use of feed supplements to reduce livestock greenhouse gas emissions: direct-fed microbials

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1 Introduction

The agricultural sector contributes approximately 24% of all global greenhouse gas (GHG) emissions (IPCC, 2014). The main routes of production for GHG emissions are enteric fermentation and manure management (Haque, 2018). The main gases produced are methane and nitrous oxide, and to a lesser extent, carbon dioxide (McMichael et al., 2007). In 2010, total anthropogenic methane and nitrous oxide emissions accounted for approximately 20% and 5% of all emissions to date, respectively, based on the fifth assessment report (IPCC, 2014).

At a global level, livestock annually produce around 80 million tonnes (Tg) of enteric methane (Patra, 2012). Of these 80 million tonnes of enteric methane,

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an estimated 18.9 Tg are attributed to dairy cattle, 55.9 Tg to beef cattle, and 9.5 Tg to sheep and goats (Hook et al., 2010). Of all livestock, ruminants are the main producers of enteric methane (Lassey et al., 2007). These animals contain a four-chambered stomach, which generates methane primarily via eructation and belching as a result of the complex microbiological fermentation that occurs in the rumen. This enteric fermentation involves degradation of cellulose and other macromolecules (Boadi et al., 2004) to allow for adsorption into the bloodstream. The large and diverse microbial population ferments the polymers to volatile fatty acids (VFAs), carbon dioxide (CO_2) and methane (Kataria, 2015).

Due to methanogenesis, the ruminant suffers a loss of ingested feed-derived energy of approximately 6–14% depending upon the diet (Johnson and Johnson, 1995). Rather than losing resources to an ineffective microbial process, this energy could instead be used by the animal to produce better-quality milk for its own development (Tapio et al., 2017). It has been predicted that reducing methane generation in the rumen would mean that more energy would be retained by the animal, thereby enhancing its nutritional efficiency (Yang et al., 2016).

Methane production by ruminants is influenced by various factors such as the physical and chemical characteristics of the feed, the feeding schedule and the feed additives. Methane is derived from ingested feed and therefore diet composition, while intake can be used to manipulate fermentation by altering the microbial interactions through feed additives, that is, direct fed microbials (DFMs). This would also have the potential to positively impact animal production. Manipulating host diet may reduce methane emissions by decreasing fermentation of organic matter, therefore shifting the site of fermentation of organic matter from the rumen to the intestine, consequently diverting hydrogen away from methane production (McGinn et al., 2004). For the cattle industry, reducing methane losses can represent an improvement in feed efficiency. Therefore, mitigating methane losses from cattle has both long-term environmental and short-term economic benefits (McGinn et al., 2004). There is a challenge, however, to maintain a balance between productivity, household food security and environmental preservation (Wright et al., 2011).

2 Methane and agriculture

Methane is a prominent GHG which is found in natural wetlands, rice fields, livestock and biomass burning. It is emitted through human activities such as the production and transport of coal, natural gases and oil, as well as naturally, through animal fermentations and gas deposits, such as peatlands. As the second most abundant GHG, it has been projected that one tonne of methane will absorb 34 times more thermal energy than one tonne of carbon dioxide, over a 100-year period (IPCC, 2013). Due to its thermal conductivity, methane levels have the potential to influence climate change on short-time scales (Rice et al., 2016).

Since the beginning of the industrial revolution, the levels of methane observed in the atmosphere have increased more than two-fold, with a continued 1–2% rise per annum since the 1980s, as measured by the National Oceanic and Atmospheric Administration (Singh et al., 2018). Realising the catastrophic threat posed by climate change, the 2015 Paris Agreement, under the United Nations Framework Convention on Climate Change (UNFCCC) and backed by 195 countries, aims to limit the increase in global average temperatures to below 2°C and, where/if possible, limit it to 1.5°C. It is expected that methane emissions from domesticated ruminants will decline in developed countries, due to an ever-growing trend towards an animal-free diet. However, factors such as population growth, rising incomes and commercialisation of previously small-scale farms will result in increased methane production in developing countries (EPA, 2014). In this way, global methane emissions from enteric fermentations are estimated to increase 32% by 2020 (EPA, 2013).

The microbial composition of the rumen, the fore-stomach of the ruminant animal, has a major influence on the feed digestion and the release of end products, such as methane, into the environment. The rumen is home to a vast array of protozoa, anaerobic fungi, anaerobic bacteria and archaea. This diverse array of microorganisms are responsible for the degradation of lignocellulose, which is used as an energy source for the animal. Short-chain fatty acids (SCFAs) are produced from these soluble sugars, and absorbed into the rumen epithelium, resulting in by-products of hydrogen (H_2), carbon dioxide (CO_2), formate and methyl-containing compounds. These by-products are important substrates for methane-forming archaea.

Due to the high microbial diversity within the rumen, methane is formed by many types of methanogens, each using distinct metabolic pathways and precursors. Although methane production can also occur in the lower gastrointestinal tract, a surprising 89% of methane emitted from ruminants is produced in the rumen itself and exhaled through the mouth and nose (Hook et al., 2010). In general, methanogenic archaea use $H_2 + CO_2$, formate, methylated C1 compounds, or acetate as energy and carbon sources for growth (Deppenmeier, 2002). The majority of rumen methanogens have been shown to belong to the *Methanobrevibacter* genus, accounting for 74% of all archaea (Henderson et al., 2015). When combined with *Methanosphaera* spp. and two *Methanomassiliicoccaceae*-affiliated groups, there are five dominant methanogen groups that comprise 89.2% of the community (Henderson et al., 2015).

Methanogenesis is a complex process dependent upon a range of microbes, which contribute either indirectly by creating the appropriate environment required for the growth of methanogens or directly by producing the substrates used by methanogens. The production of methane in cattle is also influenced by diet composition (ingredient and chemical), feed intake, and digestibility (Hristov et al., 2018). It has long been established that an increase in concentrate levels in the diet results in a decrease in methane emission as a proportion of energy intake or expressed by unit of animal product such as milk or meat (Wanapat et al., 2015).

High-starch diets have been shown to decrease methane emissions in ruminants better than fibrous diets, such as those containing beet pulps. Non-structural carbohydrates such as starch and sugars are associated with higher ruminal fermentation rates and accelerated feed turnover which cause a change in the rumen physico-chemicals and a shift in the microbial population. A shift in VFA (volatile fatty acid) production from acetate towards propionate occurs with the development of starch-fermenting microbes. This results in lower methane production because the relative proportion of ruminal hydrogen sources declines whereas that of hydrogen sinks increases. As propionate production and methanogenesis are competing pathways, starch-fermenting bacteria can compete with methanogens for hydrogen, therefore less methane would be produced in the rumen (Moss et al., 2000). For this reason, maize silage or whole-crop silage can reduce methane production in the rumen (Haque, 2018).

Manipulating feed in a manner which will improve feed utilisation and ameliorate product yields while reducing methane emissions will be beneficial for farm production and preferable for the environment.

3 Nitrous oxide and carbon dioxide in agriculture

Nitrous oxide (N_2O) is a potent GHG with a 100-year global warming potential 298 times greater than carbon dioxide (EPA, 2017). Currently, the main sources of anthropogenic N_2 emissions are agriculture, industry, biomass burning and indirect emissions from reactive nitrogen, leaching and atmospheric degradation (Reay et al., 2012). When considering direct agricultural emissions, 38% is attributed to N_2O, 32% to methane from ruminants, 12% from biomass burning, 11% from rice production and 7% from manure management (Bellarby et al., 2008). Livestock-related nitrous oxide emissions are estimated to total between 1 and 2 million tonnes of nitrous oxide-N each year, mainly due to animal waste. Nitrous oxide from synthetic fertilisers, manure applications and crop residues left on farms account for over 40% of total agricultural emissions (WRI, 2014). Nitrous oxide is an intermediate gas for both nitrification (transformation from ammonium to nitrate) and denitrification (the biological reduction of nitrate to N_2 gas), and these processes are both facilitated by microbial action (Mosier et al., 1998). The amount of N_2O released depends on the system and duration of waste management.

Nitrogenous fertilisers and manure pits combine to drive the growth of these emissions. Fertilisers are in general applied in excess and not fully absorbed by the plants themselves, which leads to only 50% recovery of fertilizer N in global crop production (Eickhout et al., 2006). Consequently, a great proportion accumulates in soil and is either lost directly as nitrous oxide, or leaches into water courses, enhancing downstream, indirect N_2O emissions. The amount lost will greatly depend on many other factors such as climate, soil and management practices (Brentrup et al., 2004; Eickhout et al., 2006).

Fertilisers containing N compounds consume up to 10 times more energy and consequently result in more GHG emissions, than fresh manure which is a low C-emitting alternative. Fertilisers are commonly used in agriculture, with the production of fertilisers emitting ~1.2% of the world's total GHGs (Wood and Cowie, 2004). Efficacy in the manufacturing of fertilisers can contribute to a significant reduction in nitrous oxide levels. Improvements would be related to greater energy efficiency in ammonia production plants, introduction of new nitrous oxide reduction technologies and other general energy-saving measures in manufacturing. With an increasing population and a demand for greater food production, N_2O emissions are likely to continue to rise in the coming decades (Reay et al., 2012).

Carbon dioxide is a colourless, odourless gas, released through natural processes such as respiration and volcanic eruptions, as well as human activities of deforestation, land-use changes and the burning of fossils fuels. Increasing concentrations of atmospheric CO_2 and other radioactive greenhouse gases, will ultimately lead to profound effects in the ecosystem. CO_2 does not break down easily in the atmosphere and can persist for several centuries.

Carbon sequestration refers to the process by which atmospheric CO_2 is transferred to soil or vegetation (Teagasc, 2017). The earth's soils contain approximately 1500 Pg (Picogram) of carbon, making it the largest surface of terrestrial carbon (C) (Post et al., 1990). Agricultural soil can act as both a source and sink of atmospheric CO_2 because it not only produces C but can also store C in soil and vegetation (Baah-Acheamfour et al., 2016; Paustian et al., 2000). Soil organic carbon (SOB) is influenced by the physical and chemical environments of the soil (e.g. moisture, temperature, aeration, pH and nutrient availability), the characteristics of the organic matter (i.e. susceptibility to microbial decay) and the physical accessibility of the organic matter to microbes (Paustian et al., 2000). Reconstructions of global land-use change suggest that terrestrial ecosystems have contributed as much as half of the increases in CO_2 emissions from human activity in the past two centuries (Post et al., 1990; Houghton and Skole, 1990).

Current knowledge suggests that agricultural soils have the capabilities to act as CO_2 sinks, however, this is dependant on changes in management practices. Management adaptation strategies which may be incorporated to reduce CO_2 emissions from agricultural soils include: (1) reduced tillage (2) cropping intensification and increased production efficiency (Paustian et al., 2000).

Arguments in favour of using agricultural soil C sequestration as a mitigation option are that additional benefits such as improving soil and water quality, reducing erosion, enhancing better soil fertility and crop production will rise from increasing soil organic matter (Paustian et al., 2000). However, carbon sequestration may lead to important water and nutrient depletion and increased soil salinity and acidity (Jackson, 2005). Although sequestration reduces the levels of CO_2 in our atmosphere, these negative effects on crop yield are not favoured, with an ever-increasing world population and demand for food.

4 Direct-fed microbials (DFMs)

Direct-fed microbials (DFMs) refer to microorganisms which are used to supplement feed to exert a beneficial effect on the animal. They contain live, viable cells, rather than additives which may only contain bacterial constituents. The term probiotic and DFM can be used interchangeably. The US Food and Drug Administration (FDA) authority define DFMs as 'products that are purported to contain live (viable) microorganisms (bacteria and/or yeast)'. DFMs are regulated as feed ingredients by the American Association of Feed Control Officials (AAFCO) and the FDA. DFMs are provided to the ruminant in the form of a bolus or mixed in with feed (Khan and Oh, 2015). According to USDA's National Animal Health Monitoring System's (NAHMS) Dairy 2007 study, 20% of dairy and heifer operations used DFMs for preventative purposes, an increase from 14.4% in 2002. In the European Union, approximately 20 microbial feed additives are authorized for use (Meieregger et al., 2010).

DFMs have been shown to limit gastrointestinal infections and provide optimally regulated environments in the digestive tract (Seo et al., 2010). DFMs detoxify toxic compounds, modulate the innate immune system, and maintain optimal gut movement and mucosal integrity of the intestine (Kumar et al., 2015). These effects have been mainly shown in pre-ruminants, where their benefits include a reduction in the incidence of diarrhoea, a decrease in faecal shedding of coliforms, promotion of ruminal development, improved feed efficiency, increased body weight gain and reduction in morbidity (Krehbiel et al., 2003).

In adult ruminants, there is little research available in relation to the efficacy of DFMs containing lactic acid bacteria (LAB). The use of yeast as a DFM has shown varied results. *Saccharomyces cerevisiae* was shown to reduce methane by 6–10% with varying concentrations (Lila et al., 2004). Adversely, the use of three bacterial DFM treatments of *Propionibacterium freudenreichii* 53-W, *Lactococcus pentosus* D31 and *Lactococcus bulgaricus* D1 did not alter ruminal fermentation and failed to reduce methane emissions in lactating primiparous cows on a high-starch or high-fibre diet (Jeyanathan et al., 2019).

Due to the restriction of using antibiotics as animal supplements, DFM use has become more popular due to their potential to influence the rumen environment and enhance feed efficiency. There have been several recent studies (Table 1) which investigated the use of DFMs as methane mitigators, applied alone or in combination with other treatment methods.

4.1 Types of DFMs

Many types of DFMs exist, ranging from bacterial, to yeast and fungal sources. Natural methods for reducing GHGs also exist, such as seaweed and other organic feed supplements. The type of DFM used varies on its effectiveness and intended use. This review aims to outline both the most common DFMs used, as well as mentioning new feed supplement strategies that are also aiming to reduce our GHG levels.

4.1.1 Lactic acid bacteria

Lactic acid bacteria (LAB) are an order of gram-positive bacteria which have a G+C content below 55mol% (König et al., 2009) are acid-tolerant, generally non-sporulating, non-respiring, either rod or coccus shaped bacteria that share common metabolic and physiological characteristics. The LAB group is usually reserved for the genera such as *Lactobacillus, Leuconostoc, Pediococcus* and *Streptococcus*, as well as others.

LAB produce a variety of inhibitory compounds such as organic acids, hydrogen peroxide and ethanol, which may offer them a competitive advantage in the ruminal ecosystem by inhibiting pathogenic microbial species (McAllister and Newbold, 2008). In addition, LAB produce antimicrobial peptides such as bacteriocins, ribosomially synthesised proteins produced by a bacterium of one strain, which are active against those of a closely related strain (Yang et al., 2014). Bacteriocins are deemed safe, since they are non-hazardous to eukaryotic cells.

LAB are good candidates to use as DFM because they are environmentally robust and have a number of mechanisms whereby, they may alter or influence neighbouring microbial communities with a beneficial effect on the animal (McAllister et al., 2011). LAB used as DFMs may produce lactic acid, which results in a lower pH in the rumen environment.

4.1.2 Lactic acid-utilizing bacteria

Lactic acid-utilizing bacteria such as *Megasphaera elsedenii, Propionibacterium shermanii* and *P. jensenii* have also been proposed as DFMs and have been used

Table 1 Detailing the use of DFM as feed additives and their effect on enteric methane production

Study	Type of Diet	DFM used	Effect on methane	Effect on animal welfare/ productivity
Oh et al. (2019) Effects of *Saccharomyces cerevisiae*-based DFM and exogenous enzyme products on enteric methane emission and productivity in lactating dairy cows.	The basal diet consisted of (dry matter basis) 60% forage and 40% concentrates and contained 16.5% crude protein and 32.0% neutral detergent fibre.	*S. cerevisae* (SDM) and exogenous enzyme product (ENZ).	No effect.	SDM increased milk yield by 2 kg/d without affecting DMI or feed efficiency. Supplementation of the diet with ENZ did not affect DMI, milk yield or feed efficiency.
Meller et al. (2019) Potential roles of nitrate and live yeast culture in suppressing methane emission and influencing ruminal fermentation, digestibility and milk production in lactating Jersey cows.	Ground corn.	Live yeast culture and nitrate.	Nitrate decreased methane by 17% but decreased dry matter intake by 10% (from 19.8 to 17.8 kg/d) such that methane:dry matter intake ratio numerically decreased by 8%.	Milk and milk fat production were not affected, but NO_3- decreased milk protein from 758 to 689 g/d.
Deng et al. (2018) Ruminal fermentation, nutrient metabolism and methane emissions of sheep in response to dietary supplementation with *Bacillus licheniformis.*	Total mixed ration (TMR).	Spore-forming B*acillus licheniformis*.	Daily methane production in the treatment groups was lower than in the control.	Dietary *B. licheniformis* supplementation effectively increased energy and protein utilisation in the sheep.

Jeyanathan et al. (2019) Bacterial DFMs fail to reduce methane emissions in primiparous lactating dairy cows.	Cows were randomly divided into two groups that were fed a corn silage-based, high-starch diet (HSD) or a grass silage-based, high-fibre diet (HFD).	DFM treatments: *Propionibacterium freudenreichii* 53-*Lactobacillus pentosus* D31 *Lactobacillus bulgaricus D1*.	No mitigating effect of DFM was observed on methane emissions in dairy cows.	The effect of DFM on milk fatty acid composition was negligible. *Propionibacterium* and *L. pentosus* DFMs tended to increase body weight gain of cows.
Ellis et al. (2016) DFM inoculated silage with LAB.	Treatment with one of four grouping, with inoculant long term and short term. Diets consisted of grass silage and concentrate (75:25 on a dry matter basis).	*Lactobacillus plantarum, Lactococcus lactis* and *Lactobacillus buchneri*.	Methane levels were not affected.	Dry matter intake, energy, milk and fat composition were not affected.
Thota et al. (2017) Effect of probiotic supplementation on nutrient digestibilities, growth performance and enteric methane emissions in Deccani ram lambs.	12 Deccani ram lambs of uniform body weight (16.5±0.64 kg with 130.11±3.00 days of age) were randomly allotted to two treatments in a completely randomized design. Animals were fed basal diet (chopped sorghum stover), concentrate and chopped green fodder.	*S. cerevisiae47, S. boulardii, L. acidophilus* and *P. freudenreichii*	Mean enteric methane treatment was 21.9% less than control group.	Feed efficiency of the animals was improved.

successfully to decrease concentrations of lactate and maintain ruminal pH. Since propionate is the major precursor for gluconeogenesis in early lactation dairy cows (Reynolds et al., 2003), increments of propionate production in the rumen result in increases of hepatic glucose production (Stein et al., 2006), providing more substrates for lactose synthesis, improving energetic efficiency and reducing ketosis (Weiss et al., 2008). In addition, increased propionate may reduce hydrogen available for methane production in the rumen.

4.1.3 Yeast

The most commonly used DFM is the anaerobic yeast *Saccharomyces cerevisae* and the filamentous fungus *Aspergillus oryzae* (Phillipeau et al., 2017). Traditionally, yeast products were used as feed additives to improve animal health and welfare, thereby improving animal performance and its effect as a probiotic is well-established (Darabighane et al., 2019). As such, supplementation with yeast may indirectly reduce methane production per protein (milk and meat) produced through enhancing ruminal fibre degradation and overall feed conversion efficiency (Bayat et al., 2015).

Yeast can modify rumen fermentation in a manner that can potentially reduce methane formation by decreasing rumen pH or favouring the production of certain VFAs such as acetate, propionate and butyrate (Chung et al., 2011; Iqbal et al., 2008). This is dependent on the diet offered to the animal (Islam and Lee, 2019). It has been suggested that live yeast promote the use of hydrogen by ruminal acetogens and drive the fermentation process towards acetate production instead of methane formation (Kataria, 2015). Additional proposed mechanisms by which yeast reduce methane is by reducing protozoan numbers. High populations of protozoa generally are associated with higher ruminal ammonia concentrations and increased methane production.

5 Direct-fed microbials (DFMs) and greenhouse gas (GHG) reduction

Of all agriculture GHGs, methane is the most impactful. DFMs have been explored in terms of methane reduction; however, research is still lacking for the use of DFMs against CO_2 and N_2O.

5.1 Methane

There is an increasing interest in exploring the use of naturally occurring feed additives. DFMs are already used to improve productivity and health of ruminant livestock, therefore they can be used as a possible option to reduce methane (Jeyanathan et al., 2014). Although the exact mechanism for methane reduction by DFMs has not been elucidated, it is thought that DFMs are responsible for the redirection of H_2 away from the methanogenesis pathway, as well as decreased production of H_2 during feed fermentation (Jeyanathan et al., 2011). Rumen methanogens produce methane by reducing CO_2 using H_2; therefore H_2 is a limiting substrate for methanogenesis. The amount of hydrogen produced in the rumen is highly dependent on the diet and type of rumen microbes as the microbial fermentation of feeds produces different end products that are not equivalent in terms of hydrogen output (Mirzaei-Aghsaghali and Maheri-Sis, 2011).

Proposed mechanisms of action include: (1) increased butyrate or propionate production, which may result in reduced methane production due to the utilisation of metabolic H_2 by acetogenic bacteria to produce acetate (Lila et al., 2004); (2) decrease in the number of ciliate protozoa in the rumen (Broucek, 2018); high populations of protozoa generally are associated with higher ruminal ammonia concentrations and increased methane production; this suggests that protozoa themselves or associated bacteria actively degrade dietary proteins and are methanogenic; (3) increase in lactic acid-utilizing bacteria, resulting in a reduction of lactic acid, leading to a more stable ruminal environment (Boadi et al., 2004).

Although the mechanism of action is not fully understood, studies have been performed on a broad range of DFMs for methane mitigation, with varied results. It is proposed that because some strains of yeast increase rumen bacterial growth (Chaucheyras-Durand et al., 2008), less methane may be produced due to a shift in partitioning of hydrogen between microbial cells and fermentation products (Newbold and Rode, 2006).

The scientific literature points to the idea that although the concept of using DFMs as mitigators of enteric methane emission is not novel, scientists have not yet elucidated the exact mechanism by which this occurs. Doyle et al. (2019) summarises studies which have used LAB as DFM to reduce methane, and although methane inhibition *in vitro* is evident, these seem to yield mixed results where efficacy tends to be strain-specific and dependent on delivery mode. Furthermore, a limited number of available animal trials make it difficult to draw an outright conclusion, therefore more research is needed to identify whether the use of DFM supplements can the use of DFM supplements can be used effectively to mitigate methane generation in ruminant livestock.

The use of DFMs as feed additives and their effect on enteric methane production has been tested with a range of diets, organisms and ruminants, of which is summarised in Table 1. Many trials incorporate the use of a combination of DFMs, such as yeast and LAB (Thota et al., 2017).

Yeast such as *A. oryzae* has been seen to reduce methane by 50% which was correlated directly to a 45% decrease in the population of protozoa by

Frumhloz et al. (1989). More recently, Mwenya (2004) showed promising results when yeast was added as part of feed for sheep, where a reduction ($P<0.05$) in methane production (L per day) was observed in the supplemented diets compared to control. Conversely, Martin and Nisbet (1990) found that adding *S.cerevisae* and *A.oryzae* resulted in increased methane production.

Methane reduction by yeast may be strain-dependent and variable in the impact on the rumen, as indicated in a study by Chung et al. (2011), where 15 non-lactating Holstein cows were fed a diet containing two strains on *S.cerevisae* at $1x10^{10}$ CFU per cow daily. Results indicated that yeast strain 1 did not affect methane emission intensity compared with the control treatment, while strain 2 lowered methane emissions but increased the risk of acidosis. More research is needed to screen yeast strains which can confer a probiotic benefit when used in feed and reduce methane simultaneously.

In general, studies investigating the effect of *S.cerevisae* on enteric methane production have shown differing conclusions. A meta-analysis compiled by Darabighane et al. (2019) comparing 46 scientific publications on dairy and beef cattle, concluded that supplementation of yeast does not significantly reduce methane production or methane/DMI. A comparison of 11 studies of yeast- recipient versus control group indicated that the addition of yeast, when it reduced methane emissions was not statistically significant (standard mean deviation (SMD) = −0.051; P = 0.792). The authors of this meta-analysis suggested that the results should be interpreted with caution because they were based on a small number of studies. A compilation of studies investigating the effect using *S. cerevisiae* as a DFM for reducing rumen methanogenesis showed inconsistent results between *in vitro* and *in vivo* studies (Jeyanathan et al., 2014). This could be attributed to the discrepancies in experimental conditions.

In terms of LAB/LUB, a meta-analysis completed by Krehbiel et al. (2003), indicated a generally positive trend for the improved health of young cattle treated with DFM inoculants. The use of bacteriocin like particles from LAB and LUB is of particular interest. Bacteriocins have antimicrobial proteinaceous properties which are ubiquitous in nature and are produced by a range of gram-positive and gram-negative organisms. Bovicin, a bacteriocin produced by *Streptococcus bovis*, has been found to reduce methane in vitro by up to 50% (Lee et al., 2002). Bacteriocins can be useful in directly inhibiting methanogens and/or reducing H_2 to other reductive microorganisms (Garsa et al., 2019).

Nisin is composed of 34 amino acids with two structural domains and is classed as a lantibiotic. This bacteriocin produced by *Lactococcus lactis* species, with GRAS (generally regarded as safe) status, has also shown methane reduction potential. However, Nisin supports the propionate production and depicts selective activity against gram-positive bacteria of rumen origin; therefore, complete exploration must be carried out before its use as an animal additive (Garsa et al., 2019). *Megasphaera elsdenii* (one of the most important lactate-utilizing species in the rumen) has been reported to establish a successful DFM product, but ruminal pH and fermentation have been inconsistent (Klieve et al., 2003; Henning et al., 2010)

Bacterial DFMs do not always result in positive results. Jeyanathan et al. (2019) reported that DFM treatment results in no mitigation effect, as well as no effect on milk composition (Table 1). Similarly, it was seen that in a study using eight Holstein Friesian dairy cows, treated with both long- and short-term LAB inoculations of *Lactobacillus plantarum, Lactococcus lactis* and *Lactobacillus buchneri*, no methane reduction was observed (Ellis et al., 2016). Variances in strain and dose, as well as difference in basal silage conditions may be responsible for the lack of effects observed. There have been other attempts to inoculate the rumen with LAB and fungi (*Candida kefyr*) along with nitrate supplementation to both control methanogenesis and possibly prevent nitrite formation, but no consistent animal data, have been reported (Takahashi, 2011). Although fermentation of lactate to VFA would help prevent a decreased ruminal pH, introduction of lactate-producing DFM would require careful scrutiny in situations in which sub-acute rumen acidosis might occur (Hristov et al., 2013).

5.2 Nitrous oxide and carbon dioxide

Diet can play a fundamental role in manure emissions, as it influences the volume and composition of manure. In particular, diet affects the amount, form and partition of N excretion between urine and faeces, and the amount of fermentable organic matter excreted (Hristov et al., 2013). Reducing ruminally degradable protein concentration can reduce NH_3 (Ammonia) emissions from manure, through a marked reduction of urinary urea excretion, NH_3 concentration and potentially N_2O emissions from dairy manure (Forabosco et al., 2017). Feed additives and dietary manipulation options targeting nitrous oxide emissions are mostly studied in isolation, but can have unexpected synergistic or antagonistic effects. Further research in this field needs to be carried out to test this.

6 Strengths and challenges of direct-fed microbials (DFMs)

6.1 Consumer acceptance

Realising the GHG mitigation potential of agriculture is ultimately dependent on farm-level decisions based on how adoption will benefit the individual farmer (Chandra et al., 2016). Mitigation options that both reduce GHG emissions and increase farm productivity, that is, cost-effective practices, are more likely to be adopted (Smith et al., 2007, 2008) than practices which would negatively affect farmers' income.

6.2 Expenditure

Studies have shown that DFMs as probiotics offer economic advantages, with a cost savings of 1–5%. Initiatives such as Pasture Base Ireland aims to help farmers make better decisions around grassland management, thus ensuring that the grass offered to the animals is of the highest quality resulting in reduced methane emissions (Wims et al., 2010). This will reduce methane emissions by minimising the amount of silage and supplemental feed in the diet and improving feed quality (Analysis Ireland, 2020–2030). Table 2 summarises the possible benefits of using commercially available DFMs, which in the long run will provide financial benefits.

Table 2 Composition and benefits of commercially available DFM products

DFM name	Bacterial #'s	CFUs	Benefits
Generator D	Nine species of live bacteria	24 billion CFU per 2 g	Replaces digestive organisms, higher milk and meat production
Generator Elite	Fourteen species of live bacteria	264 billion CFU	Replaces beneficial organisms, supports appetite, helps maintain pH, aids DM intake
Generator ProSacc	Nine species of live bacteria	2.27 trillion CFUs	Lactic acid-producing bacteria, enzymes to break down feed, B vitamins
Generator ULTRA	Eight species of live bacteria	68.25 billion per feeding	Enzyme-producing bacteria, purified digestive enzymes
Generator PYK	Eight species of live bacteria	480 billion CFUs	Digestive supplement, ideal as a calf starter
ProP169	Propionibacterium freudenreicheii strain P169	60 billion CFUs per feeding	Increase rumen propionate yield
Bovine DFM Powder	Unknown	300 billion CFUs per pound	Improves palatability of feed and rumen function, increased fibre digestion, feed intake and milk/meat production
Blended Fermentation Product Series – COMBO	Five species of live bacteria	100 million – 4 billion CFUs per gram	Source of amino acids, cellular proteins, vitamins (B), minerals

6.3 Inconsistent results

There is inadequate evidence of the direct enteric methane mitigating effect of yeast and other DFMs. DFMs function as probiotics which stabilize pH and stimulate rumen function. These have been reported to result in significant improvement in animal productivity and enhanced feed efficiency. This ultimately reduces methane emission intensity. However the issue of maintaining antimethanogenesis in the rumen has not been fully studied and it seems that DFMs would have to be continuously administered to maintain their efficacy (Jeyanathan et al., 2011).

6.4 Alternative hydrogen sinks

It has been suggested that rumen function will be disrupted if methane production is significantly decreased by directly inhibiting methanogenic archaea without the provision of alternative hydrogen sinks (McAllister and Newbold, 2008). Of all the hydrogen sinks present in the rumen, methane production is the most important in that the activity of rumen methanogens lowers H_2 concentration to about 1 μM, which permits a more rapid fermentation of the feed, giving the animal access to more VFA in a given time (Wolin, 1979). Conversely, high concentrations of H_2 in the rumen slows the microbial activity of feed fermenters, potentially slowing down the conversion to VFAs (Buddle et al., 2011). It is therefore important to introduce a system that will ensure that the accumulation of hydrogen in the rumen is minimised.

It has been proposed that inhibiting methanogenesis could favour microbial biomass production as an alternative hydrogen sink (Henderson et al., 2015). Inhibiting methanogenesis could have consequences on microbial ATP generation. In anaerobic systems like the rumen, part of the negative Gibbs energy change associated with fermentation is used to generate ATP through substrate level- and electron transport-linked phosphorylation. Changes in the H_2 sink levels could also affect such energy generation pathways. Inhibiting methanogens without providing alternative electron acceptors commonly results in an overall reduction in feed digestibility and often, reduced animal productivity (Beauchemin, 2009).

7 Other methane mitigation methods

7.1 Rumen manipulation

Biochemical pathways and microorganisms that play a role in methanogenesis directly or indirectly can be targeted and modulated. The amount, type and rate of fermentation of dietary carbohydrates affect both the total amount and proportions of individual VFAs formed and, ultimately, the amount of methane produced. Methylotrophs are microorganisms that can use single carbon organic compounds such as methane and methanol (Jeyanathan et al., 2015). These are key players in the carbon cycle and can be used to compete against methanogens for substrates (Iguchi et al., 2015). Acetogens are microorganisms that generate acetate as an end product of anaerobic respiration, directly producing the H_2 required for methanogenic growth. Targeting acetogens to reduce their population levels in the rumen has been shown to divert hydrogen away from methane production during ruminal fermentation (McGinn et al., 2004). Nitrate can act as an alternative H_2 sink, which has the potential to reduce rumen methanogenesis. Nitrate-reducing bacteria such as *W.succinogenes* and *S.ruminantium* are present in concentrations of 10^6 cells/mL of rumen fluid (Jeyanathan et al., 2011); however, to compete with methanogenesis, they must be present in higher concentrations. Sulphate-reducing bacteria (SRB) can act both in a competitive or co-operative association with methanogens, depending on levels of sulphate available. When sulphate levels are unlimited, SRB compete with methanogens for H_2, formate and acetate. In depleted sulphate environments, SRB co-operate with methanogens by acting as net producers of H_2.

7.2 Methane inhibitors

Bromochloromethane (BCM) has been found to decrease methane production in ruminants and is considered one of the most effective mitigative strategies for methane inhibition (Martinez-Fernandez et al., 2016). The BCM dramatically increases H_2 expelled without affecting DM intake and feed digestibility. However, the use of BCM has been restricted due to its ozone-depleting capacity. Recently, another chemical, 3-nitrooxypropanol (3NOP), which has been developed synthetically has been shown to reduce methane emission levels by up to 30% without negatively affecting animal performance (Hristov et al., 2015). This non-toxic compound has shown promise as a feed additive as some studies have reported increase in animal weight gain in dairy cows (Haisan et al., 2014 and Hristov et al., 2015) while consistently decreasing methane production. 3NOP is a structural analogue which inhibits methanogens by binding to the active site of the enzyme methyl-coenzyme M reductase which is responsible for the final step in methanogenesis pathway (Martinez-Fernandez et al., 2018). This chemical methane inhibitor is still under development and requires registration as a zoo-technical feed additive under European Union legislation (Rooke et al., 2016).

In terms of its practicality as methane inhibitor on-farm, administering 3 NOP at doses of 111 mg/kg diet DM or less resulted in 21% reduction in methane. However, there was a large degree of uncertainty (95% confidence interval, 13–29%) due to the variation in dose and method of administering 3NOP. Current view of the manufacturers is that the method for administration would appear to be critical and currently recommendations include dosing of a total mixed ration to ensure coupling of feed intake to intake of methane inhibitor (Rooke et al., 2016).

The product could be supplied as a premix to be incorporated into diets on farm. The daily recommended dose of 106 mg/kg diet DM is small enough that it may be practicable to administer the compound as a bolus into the rumen to release 3NOP over an extended time period and therefore compatible with the grazing situation.

7.3 Archael phage

Bacteriophages, or phage, are viruses which attack bacteria. Although not yet isolated from the rumen, *Siphroviridae* phages have been reported to infect *Methanobrevibacter*, *Methanococcus* and *Methanobacterium* species (McAllister and Newbold, 2008). Phages are highly specific and their use as methane biocontrol agents would likely not disrupt the normal microbial composition of the rumen ecosystem.

7.4 Natural feed supplements

Recently there have been several studies investigating the effects of seaweed (or macroalgae) as a livestock feed supplement and its potential to reduce methane production in ruminants. Antimethanogenic compounds are known to exist naturally in seaweed. The tropical red algae species of *Asparagopsis taxiformis* produces the halogenated antimethanogenic compounds, bromoform and dibromochloromethane which have been shown to reduce methane production by up to 80% when fed to sheep (Li et al., 2016). Another red algae species, *Asparagopsis armata*, has been demonstrated to reduce methane emissions by around 20% in dairy cattle, which was increased to around 60% reduction with an increased inclusion level of 1% (Roque et al., 2019).

8 Conclusion

DFMs are a viable method for the reduction of GHG emissions in livestock. DFMs may be used in conjunction with existing technologies, that is, vaccines, and small molecule inhibitors. For this to work in on-farm operations, there needs to be a balance between the reduction of GHGs, cost-effectiveness, animal welfare and animal productivity. Essentially, improving feed quality and the overall efficiency of dietary nutrient can be an effective way of decreasing the intensity of GHG emissions from livestock.

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Chapter 11

Modifying the rumen environment to reduce greenhouse gas emissions

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1 Introduction

The dramatic increase in the human population – estimated to reach 9.7 billion people by the year 2050 – will require an approximately 25% increase in gross agricultural output between 2020 and 2050 to meet the global food demand of humanity (FAO 2018; Nations 2019). However, increased animal production has placed an added strain on the environment as a result of the production of greenhouse gases (GHGs) (Huws et al. 2018) and nutrient accumulation in intensive livestock systems (Tullo et al. 2019). Ruminant production produces a number of GHGs including carbon dioxide (CO_2) mainly due to the use of fossil fuels, methane (CH_4) from enteric fermentation and manure, and nitrous oxide (N_2O) from manure and nitrogen fertilizer (Lynch and Pierrehumbert 2019). CH_4 is a particularly prominent GHG, as it has a global warming potential that is 28 times greater than that of CO_2 (Jackson et al. 2019). Globally, it has been shown that ruminants contribute about 11% of total anthropogenic GHG production, with approximately 6% arising from enteric CH_4 from ruminants (Rojas-Downing et al. 2017; Grossi et al. 2019; Beauchemin et al. 2020). In addition to the negative impacts on the environment, enteric CH_4 emissions can also represent a 2–12% loss in gross energy intake (Johnson and Johnson

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1995). Theoretically, if this energy loss could be reduced through mitigation technologies, the energy saved could be redirected toward meat and milk, improving production efficiency. Although the carbon footprint on an intensity basis from livestock husbandry has declined in the past 50 years owing to improvements in animal production efficiency, a trend that is expected to continue, the increasing demand for animal protein may limit the achievement of the Paris accord target of limiting temperature increases to 1.5°C above pre-industrial levels (Ripple et al. 2014; Beauchemin et al. 2020; Leahy et al. 2020). Therefore, the mitigation of enteric CH_4 emissions is essential to a reduction of agricultural GHG emissions.

This chapter contains three sections: the first section highlights the rumen microbiome and its contribution to CH_4 emissions; the second section summarizes the main factors (rumen- and feed-associated) that influence CH_4 production in ruminants; the final section discusses the most promising current strategies used to reduce CH_4 emissions in ruminants.

2 Greenhouse gas production and the role of the rumen microbiome

2.1 Greenhouse gas production in ruminants

The major GHGs associated with ruminant production are CO_2, CH_4, and nitrous oxide (N_2O), which are emitted either directly from enteric fermentation (CH_4) and manure (CH_4, N_2O) or indirectly from land-use change, fertilizer use and burning of fossil fuels (CH_4, N_2O, CO_2) during feed production (Schils et al. 2005; Grossi et al. 2019). CO_2 emitted by respiration is not considered as a net contributor to GHGs as its release to the atmosphere (via respiration) is offset by plants that capture CO_2 to form biomass through photosynthesis (Steinfeld 2006). However, CO_2 generated as a result of the use of fossil fuels for transport of ruminant feed or products (e.g. milk, beef), generation of electricity and other on-farm practices may exceed 41 million tonnes per year (Steinfeld 2006). Thus, the two major GHGs emitted directly by ruminants are enteric CH_4 and N_2O generated from manure (Gerber et al. 2013). Combined, these two GHGs account for approximately 80% of the total emissions arising from the livestock sector and thus are significant contributors to the global GHG budget (Opio et al. 2013). It is estimated that 44% of the total GHG emissions produced by ruminants are in the form of CH_4 (Fig. 1a), which arises as a result of the reduction of CO_2 to CH_4 by archaea in the rumen and in manure (Gerber et al. 2013). The main sources of N_2O are derived from chemical fertilizer and N deposition from manure, which are influenced by manure management and indirect emissions that arise from the volatilization of NH_3 from intensive livestock systems and its downwind deposition. Nitrous oxide comprises 31% of the 7.1 Gt of CO_2 equivalents that are produced annually by the livestock sector (Gerber et al.

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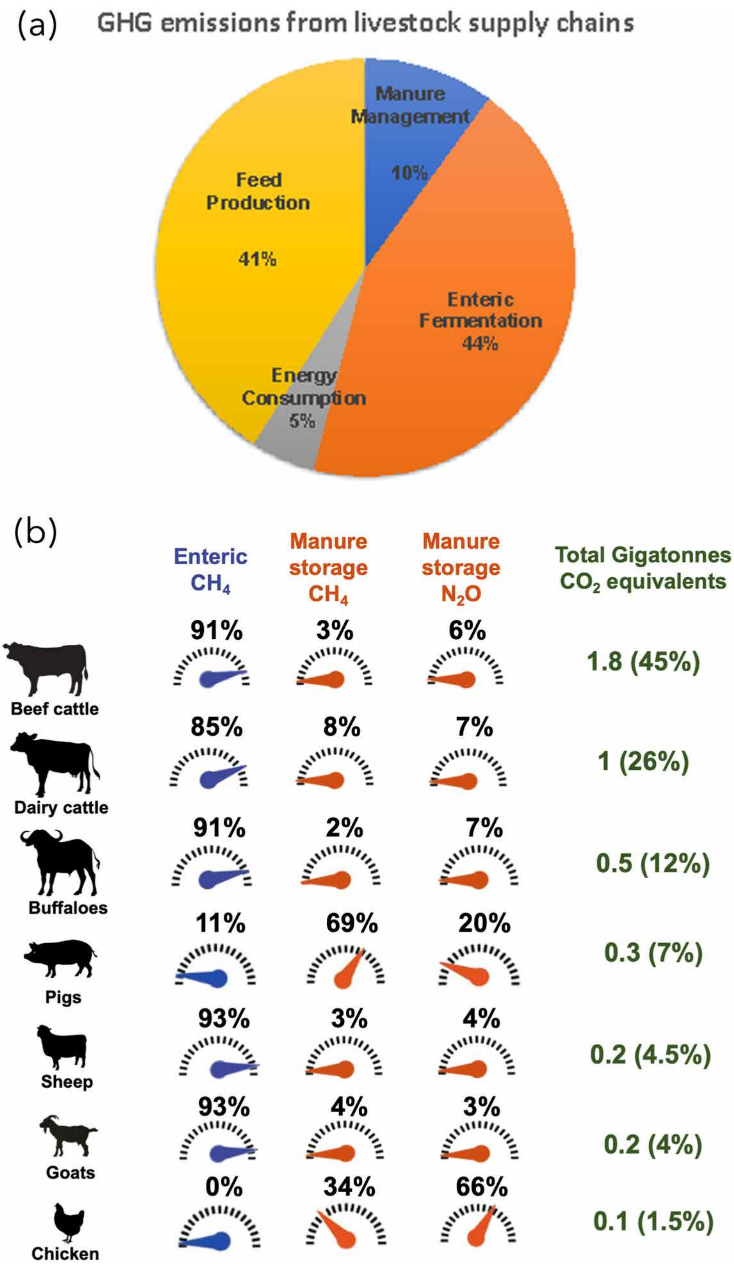

Figure 1 (a) Emissions in livestock supply chains from enteric fermentation, feed production, manure management and energy consumption (adapted from FAO 2017). (b) The farm-gate non-CO_2 emissions (%) from enteric fermentation and manure storage by livestock species in carbon dioxide equivalents (Grossi et al. 2019).

2013; Adler et al. 2015). On average, the enteric CH_4 in CO_2 equivalents accounts for over 90% of the farm-gate non-CO_2 emissions from ruminant production (Fig. 1b), with beef cattle producing the largest proportion of these emissions (Grossi et al. 2019). In all, these outcomes indicate that there is an urgent need for the development of strategies to mitigate GHG production by ruminants, while concurrently improving animal husbandry systems to meet the increasing demand for sustainable food production.

As enteric CH_4 is the largest contributor to GHG emissions from ruminants, research efforts have been primarily directed toward the development of strategies to reduce ruminal CH_4 production. CH_4 emissions are commonly expressed as CH_4 production (g/day), yield (g/kg dry matter intake) and intensity (g/kg average daily gain or energy corrected milk) (Niu et al. 2018; Ornelas et al. 2019). Rumen microbiota play a fundamental role in feed digestion and provision of energy for the host (Russell and Rychlik 2001), with CH_4 being a by-product of the rumen fermentation process. A variety of molecular, genomic, microbiological, nutritional and chemical approaches have been explored for their ability to reduce enteric CH_4 emissions, most of which have had limited success. The production and flow of reducing equivalents among the rumen microbiota is complex, and it is likely that any successful CH_4 mitigation strategy will be based on a fundamental understanding of the various roles of rumen microbiota in CH_4 production.

2.2 Rumen archaea and methanogenesis

In the rumen, archaea are one of the four main microbial groups. The domain archaea are divided into two different kingdoms: Euryarchaeota, consisting of methanogens and extreme halophytes, and Crenarchaeota, consisting of hyperthermophiles and nonthermophiles (Bayley et al. 1999). The assessment of the rumen archaeal community present in the ribosomal database project revealed 3516 archaeal sequences, representing an estimated 1500 species (Kim et al. 2011). Despite this diversity, studies have shown that 90% of rumen archaea are methanogens. Rumen methanogens are exclusively members of the Euryarchaeota, ranging from 10^6 to 10^8 cells per mL of rumen fluid, accounting for less than 4% of the rumen microbial population (Lin et al. 1997; Janssen and Kirs 2008). *Methanobrevibacter* (63.2% of the methanogenic population), *Methanomicrobium* (7.7%), *Methanosphaera* (9.8%), Rumen Cluster C (now referred as *Methanomassiliicoccaceae*, 7.4%), and *Methanobacterium* (1.2%) represent the majority of the methanogens in the rumen (Janssen and Kirs 2008; Patra et al. 2017).

Rumen methanogens produce CH_4 by transferring reducing equivalents formed by other microbiota (i.e. bacteria, fungi, and protozoa) to CO_2 (Deppenmeier 2002). Interspecies hydrogen transfer plays a key role in this process as it ensures a low partial pressure of H_2 (~162 Pa) in the rumen and avoids the inhibition of fermentation as a result of an accumulation of reduced co-factors (Janssen and Kirs 2008; Ungerfeld 2013; Rojas-Downing et al. 2017). During methanogenesis, reduced co-factors (e.g. NADH, NADPH, FADH and Fd) are oxidized and the reducing equivalents are transferred to methanogenic archaea through a series of biochemical steps to reduce CO_2 to CH_4 (Ungerfeld 2015). The hydrogenotrophic pathway is the predominant pathway for CH_4 production in the rumen, with *Methanobrevibacter* species accounting for the majority of this activity and typically accounting for over 90% of rumen archaeal 16S rRNA reads (Hristov et al. 2012). Most methanogens can also use formate as the electron donor for CO_2 reduction as a result of the activity of formate dehydrogenase. Formate is produced during fermentation by microorganisms that possess pyruvate-formate lyase or from the fermentation of plant metabolites like oxalic acid. Ruminal methanogens (e.g. *Methanosarcina, Methanosphaera, Methanimicrococcus*) can also utilize methanol, methylamines, and methylated sulphides as substrates for the synthesis of CH_4 (Ellis et al. 2008; Janssen and Kirs 2008; Poulsen et al. 2013; Tapio et al. 2017).

Recent studies have shown that the total number of methanogens in the rumen is not necessarily directly related to CH_4 yield, but rather it can be related to the abundance of particular archaeal species (e.g. *Methanobrevibacter gottschalkii*) (Zhou et al. 2011). However, it seems that the expression of genes involved in methanogenesis may be a more accurate predictor of rumen methanogenesis (Roehe et al. 2016) than the abundance of certain archaeal species (Zhou et al. 2011). The expression of genes within the methanogenesis pathway (e.g. *methyl-coenzyme M reductase alpha subunit – mcrA*) has been shown to be greater in high-CH_4-emitting cattle as compared to their low-emitting counterparts (Shi et al. 2014). It is worth noting that many factors influence rumen methanogenesis (e.g. diet, intake, rate of passage, rumen volume), and consequently the amount of CH_4 produced.

2.3 Other rumen microbiota and methane production

In the rumen, bacteria are the most abundant cellular microbiota (density of 10^{10}–10^{11} cells per mL rumen fluid), making up at least 50% of the microbial cell mass (Creevey et al. 2014). The rumen bacteria ferment feed to produce volatile fatty acids (VFA) and microbial protein, providing the main nutrients utilized by the host and other rumen microbiota (Ellis et al. 2008). Other bacterial fermentation end-products can also be used as substrates for methanogens (Zinder 1993).

Methanogens share a commensal relationship with rumen ciliates, and these protozoa-methanogen consortia account for approximately 30–40% of the enteric CH_4 produced in ruminants (Finlay et al. 1994). Protozoa are present at 10^4–10^6 cells per mL in rumen fluid and account for 20–50% of the rumen microbial biomass (McSweeney and Mackie 2012). They are involved in lipid hydrolysis and can produce large amounts of hydrogen via organelles known as hydrogenosomes (Tymensen et al. 2012), which provide substrates to methanogens via interspecies hydrogen transfer (Gijzen et al. 1988; Hobson and Stewart 1997; Kittelmann et al. 2015). Removal of protozoa from the rumen, a process known as defaunation, can suppress fiber digestion (Mosoni et al. 2011), but it is also consistently associated with a reduction in CH_4 production (Morgavi et al. 2012; Newbold et al. 2015). However, this reduction in CH_4 production is often transitory with emissions often returning to original levels within a month after defaunation as a result of an increase in hydrogen transfer between bacterial populations and methanogens (Guan et al. 2006).

Anaerobic rumen fungi are active players in the degradation of lignocellulosic feedstuffs and therefore produce reducing equivalents that are used by methanogens to reduce CO_2 to CH_4 (Gruninger et al. 2014). Anaerobic rumen fungi, such as *Neocallimastix frontalis*, have a synergistic relationship with methanogens through interspecies hydrogen transfer and also possess hydrogenosomes. This synergistic relationship increases the enzymatic activity of rumen fungi and their ability to degrade plant cell-wall carbohydrates and produce fermentation products (Hook et al. 2010). As a result of this relationship, greater fungal abundance has been associated with increased CH_4 emissions (Tapio et al. 2017). These findings suggest that rumen fungi are positively linked to methanogenesis through their provision of reducing equivalents for methanogens.

3 Factors influencing methane production in ruminants

As the largest forestomach in ruminants, the rumen is the primary site of CH_4 production within the digestive tract, with less than 5% of the emissions being emitted as flatulence (Lassey 2007; Hammond et al. 2016). Microbiological, physiological, nutritional and genetic factors all influence enteric CH_4 production in ruminants (Fig. 2).

3.1 Rumen-associated factors

3.1.1 Shifts in the rumen microbiome

It is clear that the rumen microbiome is associated with enteric CH_4 production and manipulating its structure and function through the diet is one of the simplest approaches in reducing CH_4 production in ruminants. For example, increasing the proportion of concentrate in the diet shifts the microbiota from

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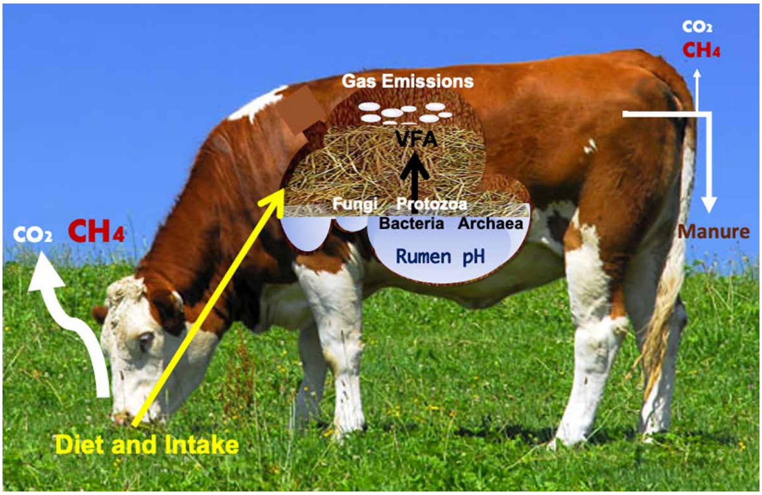

Figure 2 Rumen fermentation and factors associated with enteric CH_4 emissions. CH_4 is produced during rumen fermentation and influenced by rumen-associated factors (e.g. host genetics, rumen microbiota or physiological processes) as well as feed-associated factors (e.g. diet composition or feed intake). The weight of the white arrows represents the proportion of GHG emissions out of the mouth versus from manure and flatulence.

species that degrade plant cell-wall carbohydrates to those that degrade non-fiber carbohydrates (Zhang et al. 2017). This shift increases the production of propionate in the rumen and decreases the production of acetate. As propionate is a net sink of reducing equivalents, their availability to reduce CO_2 to CH_4 is decreased. A high-concentrate diet also has the capacity to change the rumen environment through shifts in ruminal pH, which in turn affects the diversity and composition of rumen microbiota. Unlike other members of the rumen microbiota, the diversity of archaeal communities appears to stay relatively stable during shifts from a high-forage to a high-concentrate diet (Kumar et al. 2015). This may reflect their role in the reduction of CO_2 to CH_4, a biochemical function that is common to both forage- and concentrate-based diets. It does appear that the composition of methanogenic archaea community does change with age of the host, with methylotrophic methanogens being predominant in the neonate, whereas hydrogenotrophic methanogens are responsible for the majority of CH_4 production in the mature rumen (Friedman et al. 2017).

3.1.2 Rumen pH

Rumen pH is one of the major factors influencing enteric CH_4 production. A rapid shift from a roughage-based to a grain-based diet can cause acute ruminal acidosis, shifting volatile fatty acid production toward lactate and dramatically reducing the diversity of rumen microbiota (Dijkstra et al. 2012; Petri et al. 2013). In this context, Slyter et al. (1966) studied the effect of ruminal pH on rumen fermentation and microbial population in vitro, finding a significant reduction in VFAs and CH_4 production at a pH below 6.0. Similarly, Van Kessel and Russell (1996) observed that CH_4 production was reduced in the rumen fluid from a cow fed a concentrate-based diet when the pH declined to 5.45, but if the pH was restored to 7.0 with NaOH, CH_4 production resumed to a level similar to that of the rumen fluid which was collected when the cow was fed forage. The increase in propionate production with concentrate-based diets could account for a portion of the decline in CH_4 production, as propionate-producing bacteria which are abundant in these diets also utilize reducing equivalents, while the growth of methanogens is inhibited (Janssen and Kirs 2008). The optimal pH for the growth of methanogens in vitro ranges from 6.0 to 7.2 (Paynter and Hungate 1968; Jarvis et al. 2000; Kumar et al. 2009; Hook et al. 2011). However, responses of methanogens to pH in vitro may differ from those observed in vivo. Studies with heifers fed a high concentrate diet found that CH_4 production was the highest during the period after feeding, when the ruminal pH was the lowest (Fig. 3; Hünerberg et al. 2015). Methanogens exist within biofilms on feed, protozoa and the rumen wall and metabolic conditions within these microenvironments may differ from that in free rumen fluid or in in vitro cultures.

Consequently, although high-concentrate diets can reduce the intensity of CH_4 production (i.e. CH_4 per kg of meat or milk) they may not result in a reduction in absolute emissions. Furthermore, care must be taken to ensure that ruminal pH does not decline to levels that promote subclinical or clinical acidosis.

3.1.3 VFA production and absorption capacity of the rumen wall

Carbohydrates are broken down into VFA during ruminal fermentation to supply the host animal with the energy required for growth and maintenance. In this process, feed particles are fermented by rumen microbes to simple sugars that are then converted to VFAs, CH_4 and CO_2. The formation of acetic acid and butyric acid releases reducing equivalents, while the formation of propionic acid consumes reducing equivalents. Therefore, the types of VFAs produced in the rumen are closely linked to methanogenesis, such that CH_4 production can theoretically be predicted by the following equation (Moss et al. 2000):

$$\begin{aligned} CH_4(\text{mol}) = {} & 0.45\ \text{acetic acid}(\text{mol}) + 0.40\ \text{butyric acid}(\text{mol}) \\ & - 0.275\ \text{propionic acid}(\text{mol}). \end{aligned}$$

This equation shows that the production of acetate and butyrate enhances CH_4 production while propionate formation inhibits methanogenesis by

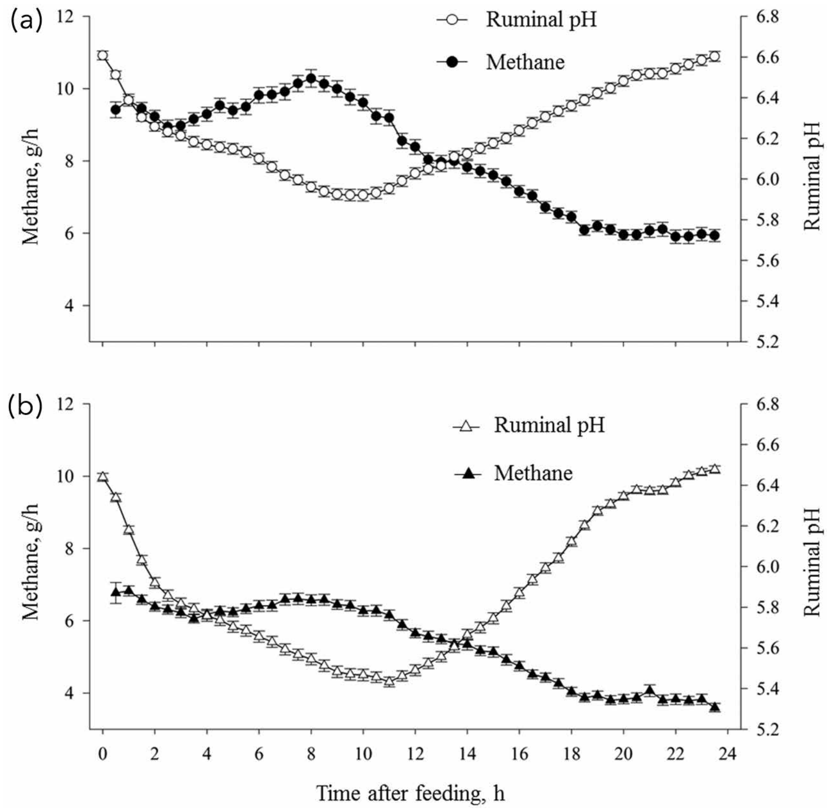

Figure 3 Diurnal CH_4 emission rate (g/h) and ruminal pH of beef heifers fed high-forage (a) or high-concentrate (b) diets. CH_4 was measured using 4 open-circuit respiratory chambers, with each chamber housing 2 heifers. Continuous rumen pH of individual heifers was measured using indwelling pH loggers. The ruminal pH and CH_4 emission rates were averaged between heifers housed in the same chamber (n = 8; means ± SEM) (Hünerberg et al. 2015).

competing for reducing equivalents in the rumen. The production of VFAs, however, is not necessarily directly related to their concentrations in rumen fluid due to variations in their absorption rate across the rumen wall and their rate of passage from the rumen. Indeed, Dijkstra et al. (1993) showed that concentrations of VFA in rumen fluid may not reflect VFA production, particularly at low rumen pH where rates of VFA absorption are increased. In this context, Penner (2014) reported that VFA absorption through the ruminal wall is linked to ruminal pH and is influenced by the passive diffusion of protons or by the secretion of bicarbonate through anion exchange mechanisms. In general, ruminal VFA concentrations have been shown to be positively correlated with VFA production (Leng 1970) and as a result some studies have attempted to predict ruminal CH_4 production based on VFA concentrations. Williams et al. (2019) developed a model that considered the production and removal of VFAs via absorption through the ruminal wall, and concluded that CH_4 yield (MY, g/kg DMI) in dairy and beef cattle could be predicted using the following equations:

$$MY = 4.08 \times (\text{acetate/propionate}) + 7.05;$$

$$MY = 3.28 \times (\text{acetate} + \text{butyrate})/\text{propionate} + 7.6;$$

$$MY = 316\,/\,\text{propionate} + 4.4;$$

where the unit of VFAs is mol/100 mol total VFA. However, VFA concentrations may not be suitable predictors of daily CH_4 production in all feeding situations, as it has been shown to be the case for grazing sheep (Robinson et al. 2010). The lack of information on molar proportions of VFAs, rumen pH and fractional rates of VFA absorption in pastured ruminants may account for this shortcoming (Bannink et al. 2006).

3.2 Feed-associated factors

3.2.1 Feed intake

It is well known that total daily CH_4 production is primarily affected by feed intake and differs widely among ruminant species (Kirchgessner et al. 1991; Shibata et al. 1993; Benchaar et al. 2001). Past studies showed that CH_4 production can be estimated from dry matter intake (DMI), particularly when feed intake is less than 1.5 times maintenance energy requirements (Moe and Tyrrell 1979). The positive correlation between CH_4 production and DMI at lower intake levels (< 1.5 times the maintenance) was used by Shibata et al. (1992) to develop an equation to estimate CH_4 production in ruminants:

$$CH_4\,(\text{L/day}) = 0.0305 \times \text{DMI}(\text{g/day}) - 4.441\,(r = 0.992).$$

However, CH_4 production per unit of feed intake is negatively correlated with an increase in feed consumption for cattle and sheep, although the total output of CH_4 may increase at higher intakes (Hammond et al. 2013; Warner et al. 2017; Goopy et al. 2020). Higher levels of intake are usually achieved by increasing the level of concentrate in the diet, which can increase the production of propionate production and decrease the production of CH_4 (Coppock et al. 1964). A reduction in retention time of feed in the rumen as a result of increased

DMI may also account for the decline in CH_4 production per unit of feed intake (Lechner-Doll et al. 1991). Therefore, maximizing feed intake can be an effective strategy to reduce CH_4 yield per kg DM consumed.

CH_4 production differs among ruminant species, and Shibata et al. (1993) were one of the first authors to use a simple quadratic equation to accurately describe the relationship of CH_4 production and DMI at different feeding levels in dairy cattle, beef cattle, sheep and goats:

$$CH_4 \text{ production} \left(\text{L/day}\right) = -17.766 + 42.793X - 0.8749X^2 \left(r = 0.966\right);$$

where X is DMI (kg/day).

Charmley et al. (2016) reported the equation to predict CH_4 production of forage-fed Australian cattle:

$$CH_4 \text{ production} \left(\text{g/day}\right) = 20.7 \left(\pm 0.28\right) \times \text{DMI} \left(\text{kg/day}\right).$$

Several equations to estimate CH_4 production have been developed based on diet composition across a number of ruminant species (Mills et al. 2003; Beauchemin and McGinn 2006; Ellis et al. 2007; Patra et al. 2016; Bell et al. 2016; Niu et al. 2018). These equations consider not only DMI, but also other nutritional traits such as digestible energy, non-fiber carbohydrate, and neutral detergent fiber. Consequently, the chemical composition of the feed (i.e. forage to concentrate ratio, feed type and quality) can also have a direct influence on CH_4 emissions in ruminants.

3.2.2 Diet composition

Moe and Tyrrell (1979) found that the production of CH_4 per unit of structural carbohydrates (i.e. cellulose and hemicellulose) was higher than that for soluble carbohydrates (i.e. starch and sugars). Later, Johnson and Johnson (1995) suggested to further separate non-cell wall carbohydrates into soluble sugar and starch to improve the estimation of CH_4 production per unit of feed. This differentiation is necessary as, according to Johnson and Johnson (1995), soluble sugars tend to produce more CH_4 than starch. For example, high grain, starch-rich diets can exhibit increased passage rates, propionate production and decreased ruminal pH. As a result, CH_4 production is usually lower with high concentrate than high forage diets. Chemical composition of the feed can also influence enteric CH_4 production. For example, corn grain results in lower CH_4 production than barley grain (Beauchemin and McGinn 2005), and likewise cattle fed corn silage produce less CH_4 than those fed barley silage (Benchaar et al. 2014). These differences likely reflect the high fiber and lower starch content of barley grain and barley silage as compared to corn grain and corn silage. Ruminal CH_4 yields have also been reported to be lower from legume forages (alfalfa, clover, etc.) than grass forages (McCaughey et al. 1999; Benchaar et al. 2001). This may also reflect the lower structural carbohydrate levels of legume forages versus grass forages and possibly their higher rate of passage through the rumen. Finally, plant maturity is also a critical factor in determining the amount of CH_4 produced by ruminants, as the fiber content of forages increases with maturity (Russell et al. 2007). Consequently, DMI decreases and CH_4 production per unit of forage typically increases with advancing maturity (Pinares-Patiño et al. 2007; Molano and Clark 2008). Additional diet components which may impact CH_4 yield (g/kg DMI) in ruminants include the level of digestible organic matter, lipid, neutral detergent fiber, acid detergent fiber and lignin content (Ellis et al. 2007; Bell et al. 2016). As a result, feed quality can be a major determinant of CH_4 emissions in ruminant production systems.

It is well known that CH_4 production is usually lower with high concentrate than high forage diets as has been shown in lactating cows (Kurihara et al. 1997; Lovett et al. 2005; Aguerre et al. 2011), beef cattle (Lovett et al. 2003; Doreau et al. 2011) and goats (Kurihara et al. 1997). Beauchemin and McGinn (2005) reported that CH_4 emissions as a percentage of gross energy intake dropped from 7.42% with a high-forage diet to about 3% with a high-grain diet fed to Angus cattle. However, this practice may result in subacute or acute rumen acidosis (Owens et al. 1998; Plaizier et al. 2008), as well as higher CH_4 emissions from manure (Hindrichsen et al. 2006). Moreover, grain supplementation did not influence daily CH_4 production from grazing beef steers (Boadi et al. 2002), and similar observations were also reported as there was no difference in enteric CH_4 emissions when goats were fed diets at maintenance that differed in forage to concentrate (Lima et al. 2013). This may reflect differences in the quality and quantity of the forage selected during grazing.

Since feed quality affects enteric CH_4 production, corn grain-based diets can be used to replace barley grain diets to lower CH_4 emissions (Beauchemin and McGinn 2005; Fellner et al. 2008; Yurtseven and Ozturk 2009). Similarly, the consumption of legume forages needs to be encouraged in regions where legume forages are suitable for cultivation as ruminant feed, as CH_4 emissions are generally lower for legume forages (McCaughey et al. 1999; Benchaar et al. 2001). Secondary compounds of plants such as condensed tannins may also contribute to the lowering of CH_4 emissions associated with legume forages (Woodward et al. 2002; Beauchemin et al. 2009). Other plants or plant extracts with high levels of bioactive compounds (e.g. saponins, flavonoids, and essential oils) can reduce ruminal CH_4 emissions by affecting metabolic processes and/or the activity of rumen microorganisms, but the effectiveness varies depending on the source, type and concentration of the bioactive (Patra and Saxena 2010; Bodas et al. 2012; Kim et al. 2015). In addition, feed intake and the maturity of the plants need to be considered in this strategy. Feeding

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alfalfa hay harvested at the mid-bloom stage is recommended compared to using vegetative-stage hay due to the 15% decrease in CH_4 production relative to gross energy intake (Benchaar et al. 2001).

In addition, forage processing methods may reduce enteric CH_4 production from ruminants if feed intake is not restricted. Grinding or pelleting forages leads to a faster rate of passage and lower fiber digestibility, reducing CH_4 production by 20–40% (Blaxter 1989; Johnson and Johnson 1995; Boadi et al. 2004). However, the extent of physically effective fiber needs to be considered while using forage processing methods to ensure the minimal quantity of fiber required to maintain rumen health and function. Increased ruminal passage rates with pelleted forages can also result in reduced ruminal fiber digestion.

4 Modifying the rumen environment to reduce methane emissions

A number of strategies have been investigated for their ability to reduce enteric CH_4 emissions, either by directly targeting methanogens or limiting the availability of reducing equivalents to convert CO_2 to CH_4. A number of approaches have been undertaken including diet manipulation, a variety of feed additives, immunization, defaunation, genomics technologies, and genetic selection of the host (Table 1).

4.1 Rumen manipulation

4.1.1 Specialty feed and additives

4.1.1.1 Fat supplementation

Addition of fat increases the energy density of the diet and has been shown to effectively mitigate CH_4 emissions in ruminants. Supplementing fat for fermentable carbohydrates can reduce rumen fermentation and decrease the growth of ruminal methanogens and protozoa (Prins et al. 1972; Czerkawski et al. 1975; Zhou et al. 2013). For example, CH_4 emissions (g/day) were reduced by 22% when sunflower oil (400 g/d, 5% of DMI) was added to the diets of Holstein steers (McGinn et al. 2004). In addition to sunflower oil, coconut oil, crushed whole oilseeds (rapeseed, sunflower and linseed) and rumen-protected crystalline fat have been assessed for their ability to lower CH_4 emissions. Machmüller et al. (2000) found that coconut oil and oilseeds decreased CH_4 production (ml/kg live weight) by 10–27% in lambs, a response that was associated with lower numbers of protozoa and a decline in the acetate to propionate ratio. Beauchemin and McGinn (2006) reported that adding canola oil at up to 4.6% of DMI reduced CH_4 emissions (g/d) by 32%, but care needs to be taken as an inclusion of total dietary fat at levels above 6% of diet DM can suppress feed intake and fiber digestibility.

The extent to which fat suppresses CH_4 production depends on the source (i.e. oilseed, tallow, nuts), the form (i.e. whole seed, extruded, extracted oil, protected fat), fatty acid profile, supplementation level, and the composition of the diet (Machmüller et al. 2000, 2003; Machmüller 2006; Jordan et al. 2006a; Petrie et al. 2009). Unsaturated fatty acids can serve as a minor hydrogen sink via biohydrogenation of the double bonds, decreasing the availability of reducing equivalents for methanogens (Johnson and Johnson 1995). Fat supplements that contain high levels of medium-chain fatty acids (e.g. coconut oil, palm kernel oil) are particularly effective at reducing CH_4 emissions as they appear to be toxic to ruminal methanogens (Machmüller and Kreuzer 1999; Machmüller et al. 2003). Long-chain fatty acids may also reduce CH_4 production in an undesirable fashion by lowering DMI and fiber digestion (McGinn et al. 2004; Jordan et al. 2006b). Beauchemin et al. (2007) reported that adding 3% of either saturated or unsaturated long-chain fatty acids in the form of tallow, sunflower oil or whole sunflower seeds to a forage diet reduced CH_4 emissions by 14%, 14% and 33%, respectively. Of these sources, sunflower oil inhibited fiber digestion the least. Fat supplementation has also been shown to reduce CH_4 emissions in grazing cattle (Pinares-Patiño et al. 2016; Beck et al. 2019). However, ensuring uniform intake of fat supplements among individuals in a herd is difficult and care must be taken to ensure that fiber digestion is not inhibited. Furthermore, oilseeds can be expensive and often do not fit into diets that are formulated on a least-cost basis.

4.1.1.2 Ionophores

Ionophores (i.e. monensin, lasalocid, salinomycin, narasin, maduramicin, laidlomycin and semduramicin) are antibiotics that are frequently added to ruminant diets to cause favorable changes in rumen fermentation (Novilla et al. 2017). Monensin, produced by *Streptomyces cinnamonensis,* is the most widely used ionophore and has been extensively used to improve ruminal fermentation and feed efficiency in beef and dairy cattle (Russell and Strobel 1989; Sauer et al. 1998). Monensin reduces CH_4 production by decreasing the ratio of acetate to propionate and inhibiting the growth of ruminal hydrogen-producing bacteria and disrupting the association between methanogens and protozoa (Russell and Houlihan 2003; Guan et al. 2006; Beauchemin et al. 2008). Some studies have shown that monensin can lower CH_4 production by up to 25% (Van Nevel and Demeyer 1995; Tedeschi et al. 2003), but reductions are usually less than this and often do not persist. Guan et al. (2006) tested the effect of monensin on steers fed either a low- or a high-concentrate diet and observed that its inhibitory effects on CH_4 emissions declined over time. Moreover, the effect

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of monensin on CH_4 production is dose-dependent. Beauchemin et al. (2008) conducted a meta-analysis (Sauer et al. 1998; McGinn et al. 2004; Van Vugt 2005; Guan et al. 2006; Odongo et al. 2007; Waghorn 2007) and concluded that a dose less than 15 ppm of monensin had no effect on CH_4 production (g/kg DMI).

4.1.1.3 Nitrate

Nitrate (NO_3^-) acts as an electron acceptor which utilizes H_2 as it is reduced to ammonia (NH_3) via nitrite (NO_2^-), a reaction that is thermodynamically more favorable than the reduction of CO_2 to CH_4 by methanogens (Ungerfeld and Kohn 2006). Nitrate can also be directly toxic to methanogens and thus stands out as a potential feed additive to suppress ruminal CH_4 production. However, the reduction of NO_3^- to NO_2^- occurs more rapidly than the reduction of NO_2^- to NH_3 and as a result NO_2^- in the rumen can enter the bloodstream and reach toxic levels (Latham et al. 2016). Nitrite can convert Fe^{2+} to Fe^{3+} within hemoglobin, forming methaemoglobin, which is no longer capable of transporting oxygen (Lee and Beauchemin 2014). The clinical symptoms of acute NO_3^- toxicity include reduced feed intake and gain, reproductive failure or even death (Bruning-Fann and Kaneene 1993).

Despite the risk of toxicity, several studies have evaluated the effect of NO_3^- on CH_4 emissions in ruminants. In general, NO_3^- has been shown to reduce CH_4 production in sheep (Sar et al. 2004; Van Zijderveld et al. 2010; Nolan et al. 2010), dairy cows (Van Zijderveld et al. 2011; Lund et al. 2014), and beef cattle (Hulshof et al. 2012; Velazco et al. 2014; Duthie et al. 2018). Van Zijderveld et al. (2011) reported that NO_3^- (21 g/kg DM) consistently decreased CH_4 emission by 16% in lactating dairy cows. Lund et al. (2014) also tested diets with NO_3^- (20 g/kg DM) and found a significant reduction in CH_4 production both in vivo (31%) with dairy cows and in vitro (10–16%). However, if NO_3^- was removed from the diet, CH_4 emissions returned to pretreatment levels. If the concentration of NO_3^- in the diet is gradually increased, rumen microbes can adapt and accelerate the conversion of NO_2^- to NH_3, reducing the risk of toxicity (Lee and Beauchemin 2014). However, even with this consideration, the risks of NO_2^- toxicity may be too high for NO_3 to be used to lower CH_4 emissions in ruminants under practical production conditions.

4.1.1.4 3-Nitrooxypropanol (3-NOP)

3-Nitrooxypropanol (3-NOP) is a small molecule that binds to methyl-coenzyme M reductase (MCR) and inhibits its active site nickel (I), thereby inhibiting the final step in methanogenesis (Duin et al. 2016). The main advantage of using 3-NOP is that it specifically targets *MCR* within methanogenic archaea without

Table 1 Summary of strategies for reducing enteric GHG emissions in ruminants

Strategy	Mechanism	Advantages/ Limitations	Feasibility of application
Diet Manipulation			
Increased concentrate	Alters VFA production toward propionate; decreases rumen pH and inhibits rumen microbes associated with methanogenesis	Can reduce CH_4 production; Improves production Decreases enteric CH_4 production but may not reduce total GHG emissions; Animals may have acidosis risk	Feasible for housing or grazing systems
Feed types and Forage quality/ maturity	Influences intake and digestibility; some feedstuffs contain secondary compounds that inhibit rumen microbes	Can reduce CH_4 production per unit of product; improves digestibility and animal production May increase feed costs; variable influence on CH_4 production	Feasible for multiple systems but highly dependent on feedstuff availability and environmental factors (e.g. rainfall, temperature)
Forage processing	Influences intake and digestibility	Can reduce CH_4 production immediately Animals may have acidosis risk; increases costs and energy to process forages	May not be feasible for grazing system
Feed Additives			
Fat supplementation	Inhibits microorganisms associated with methanogenesis	Can reduce CH_4 production; can increase diet energy density and animal performance Potential negative impacts on intake, fiber digestibility; needs more information on animal productivity	May not feasible for grazing system
Ionophores	Inhibits microorganisms associated with methanogenesis	Improves feed efficiency; mild to moderate reduction in CH_4 production Lack of long-term inhibitory effect; not approved in some countries	Feasible for housing or grazing system
Nitrate	Reduces H_2 availability, directly toxic to methanogens	Can reduce CH_4 production; can be used to replace urea as a protein source Risk of toxicity; may increase N excretion	Feasible for housing or grazing system

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3-NOP	Inhibits methanogens	Reduces CH_4 effectively Needs more information on animal performance, dose suggestions and food safety; not approved in the market	May not be feasible for grazing system
Algae	Inhibits methanogens	Reduces CH_4 effectively Lack of more in vivo studies to show animal performance and long-term effect; may require energy to process the algae	Experimental, but may not feasible for grazing system
Immunization	Inhibits methanogens	Applicable for grazing ruminants, easy to operate in farms Lack of effect information on CH_4 production, animal health and production	Feasible for different systems
Defaunation	Eliminates ruminal protozoa	Reduces CH_4 effectively Adequate level of defaunation depends on diet compositions; some chemical agents are toxic; Needs more research on animal health and performance	Difficult to implement for grazing system
Genomic technology	Inhibits microorganisms associated with methanogenesis	Important for developing vaccines or inhibitors to target microorganisms associated with methanogenesis; may improve feed efficiency and animal productivity by manipulating microbes and their pathways Research is in early stages	Experimental, may be feasible for different systems
Genetic selection	Animal breeding for low CH_4 production per unit of product by manipulating rumen microbiota	Increases the efficiency of production in multiple ruminant systems; may have long-term persistency on reducing CH_4 efficiently Needs to determine genotype and diet interactions; lack of ability to measure CH_4 production on a large population of animals; lack of information on long-term productivity and animal performance	Feasible for different systems

impacting the activity of other microbes involved in the fermentation of feed. To support licensing and application in ruminants, 3-NOP has been experimentally evaluated in dairy cows (Reynolds et al. 2014; Haisan et al. 2014; Hristov et al. 2015), beef cattle (Romero-Perez et al. 2014; Vyas et al. 2018; Martinez-Fernandez et al. 2018) and sheep (Martínez-Fernández et al. 2014) at various doses and with different diets. Early experiments studied the effect of 3-NOP (40 μL/L and 80 μL/L) in vitro using sheep rumen fluid and reported a reduction in CH_4 production of 86–96%, with no effect on total VFA concentration (Martínez-Fernández et al. 2014). Then, 3-NOP was tested in sheep at a dose of 100 mg/d per animal, resulting in a 26% decrease in CH_4 production (g/kg DMI). Haisan et al. (2014) also observed that CH_4 production was decreased by 60% in lactating dairy cows fed 3-NOP at 2.5 g/d, with no negative effect on DMI or milk production. A 30% reduction in CH_4 emissions was also reported in lactating dairy cows fed mixed rations with 40, 60 or 80 mg 3-NOP per kg of DM over 12 weeks (Hristov et al. 2015). In this study, body weight gain increased, indicating the potential repartitioning of energy that would have been lost as CH_4 toward tissue deposition (Hristov et al. 2015). 3-nitrooxypropanol was also shown to improve the feed efficiency of beef cattle fed high-forage diets and high-grain diets, with a 42% and a 37% reduction in CH_4 yield (g/kg DMI), respectively (Vyas et al. 2018). Based on these studies, the optimal dose of 3-NOP was proposed to range between 1 g/day and 2 g/day. Incorporation of 3-NOP into a pelleted supplement versus mixing it into a complete diet reduced CH_4 emissions in lactating dairy cows by 23% and 28%, respectively (Van Wesemael et al. 2019). As no significant difference was observed between the two supplementation methods, it is feasible to administer 3-NOP either in the complete diet or in a pelleted supplement.

When compared to other *MCR* inhibitors (e.g. bromoethane sulphonate – BES) (Grawert et al. 2014), 3-NOP did not reduce feed intake, digestibility or animal performance. Moreover, 3-NOP- (200 mg/kg DM for backgrounding or 125 mg/kg for finishing phase) mediated reductions in enteric CH_4 in beef cattle fed high-grain diets for at least 105 days (Vyas et al. 2018) and in dairy cattle for over 12 weeks (Hristov et al. 2015; Vyas et al. 2018). It has also been reported that 3-NOP decreases total VFA concentration, particular acetate, whereas it increases butyrate levels with no adverse effects on growth of the host (Melgar et al. 2020; Kim et al. 2020). While applications for registration of 3-NOP (Bovaer®) have been submitted to regulatory authorities in Europe and other regions of the world, further studies are required to define the optimum concentrations of 3-NOP that optimize reductions in enteric CH_4 emissions across a range of diets and ruminant production conditions. The development of systems to administer 3-NOP to grazing ruminants will be key, if meaningful reductions in CH_4 are to be achieved throughout the production system. Fortunately, recent studies have shown that administration of 3-NOP is unlikely to result in the release of residues into meat or milk (Thiel et al. 2019a) or cause toxicity (Thiel et al. 2019b).

4.1.1.5 Algae

Algae can be classified according to their size (i.e. microalgae, macroalgae), or based on their pigmentation (i.e. green, red, brown) and the habitat where they grow (i.e. freshwater, marine). All algae produce secondary metabolites that exhibit antimicrobial activity, such as halogenated methane analogues (HMAs) (Plaza et al. 2008; Blunt et al. 2011; Machado et al. 2018). The HMAs (e.g. bromochloromethane, bromoform or chloroform) exert antimethanogenic activity by binding to reduced vitamin B_{12}, blocking the cobamide-dependent methyltransferase reaction needed for the formation of methyl-coenzyme M, and thus consequently mitigating CH_4 emissions (Wood et al. 1968; Machado et al. 2018). Because of these antimicrobial properties, some algae (e.g. *Ascophyllum nodosum* and *Asparagopsis taxiformis*) have been tested in vitro and in vivo for their ability to reduce enteric CH_4 emissions (Wang et al. 2008; Machado et al. 2014; Li et al. 2018a). Machado et al. (2014) evaluated 20 species of tropical macroalgae through in vitro methods and concluded that marine algae were more effective than freshwater algae at inhibiting methanogenesis. Among the examined species, *Asparagopsis* spp. was the most effective at reducing enteric CH_4 production, with reductions as high as 98.9% in rumen fluid after 72 h-incubation (Machado et al. 2014). This result was attributed to secondary metabolites in *Asparagopsis* that inhibited both ruminal bacteria and protozoa (Paul et al. 2006; Genovese et al. 2009). Machado et al. (2018) reported that *Asparagopsis taxoformis* biomass containing 1.3 μM bromoform in vitro was similar to the addition of 5 μM bromoform in reducing CH_4 production, suggesting that bromoform was not the only CH_4 inhibitor in *A. taxiformis*. Li et al. (2018a) showed that addition of *A. taxiformis* at up to 3% of dietary DM lowered CH_4 emissions in sheep by as much as 80%, without impacting weight gain and by as much as 98% in steers with a 42% improvement in weight gain (Kinley et al. 2020). Supplementing *Asparagopsis armata* at 1% of dietary OM reduced CH_4 intensity (g/kg milk yield) in lactating dairy cows by 60% in the short term (Roque et al. 2019). Belanche et al. (2016) proposed that *Ascophyllum nodosum* possessed anti-protozoal activity and showed that it decreased CH_4 emissions from rumen fluid in vitro. Another study also reported that rumen bacteria and methanogens were linearly reduced, while protozoa were increased, when increasing levels (1%, 3% and 5 %) of sun-dried *Ascophyllum nodosum* were fed to sheep (Zhou et al. 2018). Although CH_4 emissions were not measured in this study, the 30–75% decline in total methanogens may have decreased enteric CH_4 emissions.

As can be observed, the effectiveness of algae in reducing rumen CH_4 emissions has been tested in several experiments, but investigations of their efficacy in large-scale beef and dairy production systems are needed. Harvesting of these aquatic plants from natural ecosystems may not be sustainable and scale-up of commercial production systems to a level that could satisfy demand could be both challenging and expensive. Drying and transportation of microalgae and macroalgae can also contribute to the GHG footprint of these additives.

4.1.2 Immunization

Vaccination against rumen methanogens has been studied for its ability to mitigate CH_4 emissions, an approach that is particularly appealing for use in grazing ruminants. Vaccines induce the immune system of the host animal to produce antibodies, which could potentially enter the rumen via saliva and bind to antigenic proteins located on the surface of methanogens, inhibiting their growth and activity (Williams et al. 2009; Wedlock et al. 2010, 2013; Subharat et al. 2016). Antibodies have been shown to be stable in the rumen for at least 8 h (Williams et al. 2008), but sufficient diversity and quantities of antibodies must enter the rumen so as to impact the diversity and the majority of the methanogens within the rumen.

Whole methanogen cells from different methanogen species have been investigated for their immunogenicity in grazing sheep in Australia and New Zealand, but results have been inconsistent (Wright et al. 2004; Leslie et al. 2008; Williams et al. 2009). Zhang et al. (2015) vaccinated goats with a new recombinant protein (*EhaF*, energy-converting hydrogenase A subunit F), which is a *Methanobrevibacter ruminantium* M1 adhesin-like protein responsible for hydrogenotrophic methanogenesis. The vaccine had no apparent impact on enteric CH_4 emissions or ruminal methanogens, although it did alter the composition of the bacterial community. Also, whole fixed protozoal cells have been evaluated as antigens in vaccines to control protozoa in the rumen of sheep, but this approach also was not shown to be effective (Williams et al. 2008). Improving the specificity of antibody targets against methanogens and enhancing immune responses so as to increase the concentration of antibodies that enter the rumen could improve the efficacy of anti-methanogen vaccines.

4.1.3 Defaunation

Defaunation is a process whereby ruminal protozoa are reduced or eliminated from the rumen by dietary, chemical or biological agents. Several chemicals (e.g. copper sulfate, dioctyl sodium sulfosuccinate and detergents) have been used to remove protozoa from the rumen (Becker and Everett 1930; Burggraaf and Leng 1980; Rowe et al. 1985). However, the most promising approach for removing protozoa from the rumen is through dietary manipulation as chemical agents may be harmful to host health (Williams and Coleman 1992; Ushida et al. 1997). Removal of protozoa from the rumen has been proposed as a strategy to reduce enteric CH_4, as 9–37% of total ruminal CH_4 emissions are attributed to interspecies hydrogen transfer between protozoa and methanogens (Newbold et al. 2015). Prolonged feeding of high grain and the resulting lower ruminal pH can inhibit the growth and reduce the diversity of protozoal populations (Whitelaw et al. 1984; Kreuzer and Kirchgessner 1987). This outcome could be one of the reasons why CH_4 emissions tend to be lower in ruminants fed high concentrate versus high forage diets. Addition of fat to the diet can also lower protozoa numbers. Coconut oil, in combination with high concentrate diets, has been recommended as an alternative feeding strategy that could reduce CH_4 emissions by up to 57% (Machmüller et al. 1998; Nguyen and Hegarty 2017). The effectiveness of defaunation at lowering CH_4 emissions depends on the composition of the diet, as suppression of protozoa in ruminants fed high-forage diets can significantly inhibit fiber digestion (Li et al. 2018b). Furthermore, most defaunation procedures do not remove all protozoa from the rumen, and if they do it is challenging to ensure that ruminants remain defaunated under practical production conditions. Some studies have suggested that programming microbial colonization in the neonate could result in life-long reductions in enteric CH_4 emissions (Hegarty et al. 2008; Yáñez-Ruiz et al. 2015). Although these studies have shown that these approaches can alter microbial community structure and/or activity post-weaning, they have yet to demonstrate a life-long reduction in enteric CH_4 emissions.

4.1.4 Genomic approaches in characterizing and reducing methanogenesis

It is challenging to find a universal approach for effectively mitigating enteric CH_4 production in different farming systems (housing or grazing) without influencing nutrient digestibility and animal productivity. Following the advancement of omics technologies, more in-depth information of the rumen microbiome has revealed remarkable host-microbe interactions and how they contribute to CH_4 production. Whole genome sequencing, as the basis of these technologies, plays a critical role in targeting methanogens and other ruminal microbes that produce substrates for methanogenesis. Culturomics is thought to be a useful technology for characterizing, as of yet, uncharacterized rumen microorganisms. A 'Hungate1000' project was launched in 2012 aiming to sequence 1000 cultured rumen microbial genomes to enhance our knowledge about the diversity and function of these microbes (Seshadri et al. 2018). Currently, 501 genome sequences of rumen microorganisms have been

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generated, of which 480 are bacterial genomes and 21 are archaea (Seshadri et al. 2018; Newbold and Ramos-Morales 2019). This includes members of 9 phyla, 48 families and 82 genera, with many species represented by only one or a few isolates. Although, multiple isolates of some polysaccharide-degrading genera such as *Butyrivibrio*, *Prevotella* and *Ruminococcus* were sequenced. Next-generation sequencing helps researchers predict rumen microbial function from metataxonomic data. In this context, the available rumen microbial genomics of a single species can be applied to identify and explain the process of methanogenesis based on microbial genomes (i.e. methanogens and other microbes-contributing substrates to methanogens) and the specific proteins that are expressed during methanogenesis (Huws et al. 2018). For example, Gilmore et al. (2017) reported that assembly and analysis of the methanogen genomes enabled the metabolism of four methanogens (*Methanobacterium bryantii*, *Methanosarcina spelaei*, *Methanosphaera cuniculi* and *Methanocorpusculum parvum*) to be predicted. They found all sequenced methanogens were enriched for genes in energy production and conversion, coenzyme transport and metabolism, but were lower in genes assigned to carbohydrate and lipid transport and metabolism. With a greater understanding of methanogen metabolism, CH_4 production may be modified by optimal designing of biochemical processes. Furthermore, with the revealing of the presence of a prophage (*peiR*) in the rumen methanogen (*M. ruminantium* M1) genome, another potential approach using the phage-encoded lytic enzyme (endoisopeptidase PeiR) to specifically target and lyse the pseudomurein cell walls of rumen methanogens could provide another viable strategy to reducing enteric CH_4 emissions (Leahy et al. 2013; Altermann et al. 2018; Gilbert et al. 2020). Microbial genomes can also enhance our understanding of microbial interactions that drive methanogenesis, such as the commensal relationship between methanogens and ciliates. However, to date, unlike rumen bacteria and archaea, our knowledge of the genomes of protozoa and fungi is limited as a result of their G-C rich genomes which are difficult to assemble.

Tapio et al. (2017) revealed that methanogenesis may be associated with the composition of rumen archaeal community but not archaea abundance, especially with the *Methanobrevibacterium gottschalkii* clade. Therefore, metagenomics would be a useful tool to understand methanogenesis mechanisms in the rumen by predicting microbial metabolic functions. By characterizing methanogens in the buffalo rumen through metagenomics, genes encoding key steps in the process of methanogenesis were identified (Singh et al. 2015). In a beef cattle study, the abundance of archaeal genes in rumen digesta differed between high and low CH_4-emitting steers, with eight genes being differentially abundant between these two populations (Wallace et al. 2015). *Succinovibrionaceae* were more abundant in the low CH_4 emitters, whereas *Methanobrevibacter* were more abundant in high CH_4 emitters.

Researchers have characterized the rumen microbiota of high CH_4-emitting and low CH_4-emitting sheep, finding that lactic acid-producing *Sharpea azabuensis* were more abundant in low CH_4 emitters (Kamke et al. 2016). These studies support the possibility of there being value in selecting ruminants for specific rumen microbiota that are associated with lower enteric CH_4 emissions.

McAllister et al. (2015) outlined several metagenomic and metatranscriptomic approaches that could be used to mitigate enteric CH_4 emissions. The chemogenomics approach is based on finding chemical compounds that inhibit specific proteins expressed by genes involved in methanogenesis. This methodology resulted in the development of 3-NOP (Duval and Kindermann 2012) as described above. Using genomics to identify extracellular proteins with improved antigenicity in the host could improve the efficacy of vaccines against methanogens.

4.2 Improved genetic selection

It has been reported that CH_4 yield (g/kg DMI) varies among individual ruminants fed the same diet. Pinares-Patiño et al. (2003) found that some pastured sheep were persistent-high or low CH_4 emitters in four separate measurement periods. This might be due to genetic differences in CH_4 production among individuals and variation in rumen methanogenic ecology (Zhou et al. 2009). This strategy aims to link genetic traits (e.g. weaning weight, dag score, muscle depth) with the rumen microbiota in order to predict CH_4 yield phenotypes across ruminant species (Kamke et al. 2016). Through the identification of rumen microbiome of these grazing sheep, they discovered that species closely related to the lactic acid-producing, *Sharpea azabuensis*, were more abundant in low CH_4-emitting sheep. In contrast, H_2 producing *Ruminoccocaceae*, *Lachnospiraceae* and *Verricomicrobia* were more prevalent in the rumen of high CH_4-yielding sheep (Kittelmann et al. 2014; Kamke et al. 2016). These findings suggest that the selection of ruminants that yield less CH_4 per unit of DMI may be a viable strategy to lower herd emissions. Residual feed intake (RFI), as an indicator of cattle feed efficiency, is the difference between actual feed intake and the expected feed intake estimated from body weight, growth rate and production over a specific period (Jones et al. 2011). Researchers have demonstrated that cattle with low RFI emit less CH_4 than animals with high RFI (Hegarty et al. 2007; Zhou et al. 2009). This finding promoted studies focusing on the genetic selection for low RFI animals (Weber et al. 2016). However, some studies have contradicted these findings, suggesting that CH_4 yields may actually be higher in low-RFI cattle (McDonnell et al. 2016; Velazco et al. 2017; Flay et al. 2019). It has been suggested that this strategy of selecting animals with low RFI for the suppression of CH_4 emissions could be more advantageous for systems that rely on the use of low-quality feeds in ruminant production (Patra 2012).

Nucleic acids have been used as genetic markers for rumen microbial processes (Huws et al. 2018) and those genetic markers related to ruminant CH_4 production have been studied in recent years. Golder et al. (2018) studied the genetic markers associated with rumen microbiome and metabolome of dairy heifers fed high grain diets. They observed that some metabolites (e.g. acetate to propionate ratio, lactate) and rumen microbial phyla (e.g. Actinobacteria, Euryarchaeota, Fibrobacteres) had associated markers and quantitative trait loci, which may enable selection for acidosis-resistant cattle with lower CH_4 emissions. A high genetic correlation between DMI and predicted CH_4 emissions was revealed by Pickering et al. (2015) using a data set that included 1726 dairy cows, where one single nucleotide polymorphism (SNP) on chromosome 7 was reported to be associated with CH_4 emission traits. Singh et al. (2015) identified the functional genes involved in rumen methanogenesis and acetogenesis pathways in Indian buffalo, which were key enzymes in CH_4 production. Their results also contributed new knowledge about acetogenic bacteria that compete with methanogens and thus may be useful to reduce CH_4 emissions. Roehe et al. (2016) suggested using archaeal abundance in ruminal digesta to genetically select animals for lower enteric CH_4 emission, avoiding the need for direct measurement of enteric CH_4 emissions. They also identified 20 genes (e.g. *mcrA* and *fmdB*) associated with CH_4 production through metagenomic analysis of rumen contents. These results contradicted with those of Carberry et al. (2014) where they found that diversity rather than abundance of methanogens determined the extent of enteric CH_4 emissions. Auffret et al. (2018) identified genes associated with the hydrogenotrophic methanogenesis pathway in beef cattle of different breeds fed different diets and suggested that they could be used as biomarkers of CH_4 emissions. The inconsistencies across these studies may reflect the challenges of comparing divergent data sets from different animals fed various diets as well as differences in sampling method, sample preparation and data analysis (Huws et al. 2018).

Nevertheless, as some rumen microbial features have been reported to be heritable and influenced by host genetics in beef cattle (Li et al. 2019), it has been suggested that it should be possible to genetically select for ruminants with low CH_4 emission traits. Wallace et al. (2019) collected samples from over 1000 dairy cows to investigate the relationship of host genomics and rumen microbiome as well as their interactions on CH_4 emissions. Difford et al. (2018) studied the individual variation in CH_4 production influenced by both host genetics and rumen microbiome in dairy cows. They reported that host genotypes can explain 21% of the individual variance while the cumulative effect of rumen bacteria and archaea was 13%, and these two were independent. The core rumen microbiome was found to be correlated with host genetics, and the possibly heritable rumen microbes raise the possibilities of selecting for low CH_4-emitting cattle. However, the complexity of measuring enteric CH_4 emissions may make it difficult to employ this approach in commercial ruminant production systems.

5 Conclusion

Research in the genetics, nutrition, microbiology, and physiology of ruminants has identified several strategies that are effective at reducing CH_4 emissions per unit of animal product (i.e. milk, meat or wool). Altering the composition of the diet and some feed additives are capable of reducing enteric CH_4 emissions, particularly in ruminants fed total mixed diets where additives can be easily administered. Some of these technologies only suppress CH_4 emission in the short term and almost all must be administered daily to be effective. Administration of these additives is notoriously difficult in more extensive grazing-based ruminant productions systems, which incidentally are often responsible for the largest portion of CH_4 emissions in ruminant production systems. Immunization and genetic selection are CH_4 mitigation approaches that could be applicable to grazing ruminants. However, evidence that vaccination can reduce enteric CH_4 emissions is still lacking and the complexities of measuring enteric CH_4 production has hampered the adoption of this selection criterion in commercial breeding programs.

Overall, a combined approach may be the best strategy to achieve a consistent decrease in CH_4 emissions in ruminants. For example, dietary management in a feed-efficient herd will help to reduce the intensity of enteric CH_4 emissions. Furthermore, novel strategies to reduce CH_4 production per unit of milk or meat in grazing systems need to be emphasized. Besides, consumers' preference regarding animal welfare, food quality and safety need to be taken into consideration while developing and selecting strategies that reduce enteric CH_4 emissions.

6 Where to look for further information

The Global Research Alliance harbors a livestock research group that is focused on reducing the greenhouse gas emissions intensity of livestock production systems and increasing the quantity of carbon stored in soils. It is home to a number of relevant networks including the Animal Health and Greenhouse Gas Emissions Intensity Network; Animal Selection and Genetic and Genomics Network; Feed and Nutrition Network; Manure Management Network and Rumen Microbial Genomics Network. The Global Rumen Census, Low Emission Livestock Development and Measuring, Reporting and Verifying Greenhouse Gases activities were also developed within this organization.

- https://globalresearchalliance.org/research/livestock/.

The Livestock Environmental Assessment and Performance Partnership has also produced a number of documents related to sustainability in livestock production systems. Particularly relevant documents would include guidelines on: Environmental performance of feed additives in livestock supply chains; Measuring and modelling soil carbon stocks and stock changes in livestock production systems; Environmental performance of animal feeds supply chains; Greenhouse gas emissions and fossil energy demand from small ruminant supply chains; Environmental performance of large ruminant supply chains.

- http://www.fao.org/partnerships/leap/resources/guidelines/en/.

The Intergovernmental Panel on Climate Change can also be a useful source on overall anthropogenic impacts on climate change. Some sections of the report are related to agricultural production and the role of livestock in greenhouse gas emissions. Searching the site using the term 'Agriculture' will bring up most relevant documents.

- https://www.ipcc.ch/.

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Index

ATTACHMENT 39

Ammonia Emissions from North Carolina Hog Operations' Animal Waste Management Systems to Produce Biogas

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As part of my on-going research on air quality issues surrounding concentrated animal feeding operations (CAFOs), I have been doing further analysis of ammonia emissions from hog operations, and have also revisited my own research papers. I wanted to share these findings with you promptly.

The hog industry in North Carolina is in the process of retrofitting existing animal waste management systems to produce biogas. The conversion of animal manure to biofuels (e.g. biogas) is often promoted as an environmentally beneficial management system, with a potential to reduce greenhouse gas emissions. Anaerobic digestion (AD) is a method for converting biomass into bioenergy. Livestock manure is a commonly used biomass material for production of bioenergy.

My 2008b research paper (attached) calculated the per-hog ammonia emission from combined sources (water-holding structure and barns) at a CAFO that used anaerobic digestion to produce biogas, and found that for the same ambient temperature, this emission increased by 12% relative to a comparable CAFO that does not produce biogas, that is, a CAFO employing traditional lagoon and sprayfield technology. My recent analysis has allowed me to determine the specific increase for the open lagoon (secondary lagoon) storing anaerobic digestate. I have determined that the per-hog ammonia emission from anaerobic digestate stored in an open water holding structure without additional treatment (i.e., an open secondary lagoon) increased by approximately 66 percent compared to the per-hog emission from a conventional anaerobic hog waste lagoon (primary lagoon).When a treatment module intended to reduce ammonia emissions was functioning in the cooler month of November 2002, the per-hog ammonia emission from the water holding structure decreased by approximately 59 percent compared to a conventional anaerobic hog waste lagoon. See table below.

Note that these figures represent the changes to the ammonia emissions rate from a given source (water-holding structure or house) as a percentage of the per-hog nitrogen-excretion rate (referred to in my paper (2008b) as %E, as discussed on page 1152) at conventional and biogas-producing facilities. The %E values in the table below represent the emissions at conventional and biogas-producing CAFOs, normalized to account for differences in the size of lagoons, the number and weight of hogs, and differences in feed and other factors affecting nitrogen excretion.

This information is drawn from my 2008b paper entitled "Characterizing Ammonia Emissions from Swine Farms in Eastern North Carolina: Part 2—Potential Environmentally Superior Technologies for Waste Treatment," which is discussed in more detail below. A copy of the paper is attached. This analysis updates the information I provided in an expert report in the N.C. Office of Administrative Hearings, which I understand was submitted to DEQ in December 2021 as an attachment to comments regarding the forthcoming biogas general permit.

The data from my 2008 research paper indicate that storage of waste from covered anaerobic digesters in open lagoons (i.e. secondary lagoon) without additional treatment increases lagoon emissions of ammonia significantly. Ammonia emissions are a serious concern for air quality and water quality. The data also indicate that additional treatment dramatically reduces these ammonia emissions.

Background

Anaerobic digesters do not significantly change the nutrient quantity in the waste as nitrogen and phosphorus are retained and only carbon is reduced through conversion and degassing of methane and carbon dioxide. However, the mass of organic nitrogen is decreased, and it is mineralized to TAN (Total Ammoniacal Nitrogen=NH_4^+ N + NH_3); i.e. NH_3 N expressed as a percentage of Total Kjeldahl Nitrogen (TKN) increases as manure is digested.

Thus, ammonia (NH_3) and ammonium (NH_4^+) are found at higher concentrations in liquid digestates than raw manure (Amon et al., 2005; Clemens et al., 2006; Weaver et al., 2012; Nkoa, 2014). Therefore, anaerobic digestates stored in open secondary lagoons and land-applied to fields have higher NH_3 emission than undigested animal manures and slurries (Strik et al., 2006; Aneja et al., 2008 a, b; (Harper et al., 2010); Alburquerque et al., 2012; Nkoa, 2014) especially as the temperature increases. Factors such as temperature and pH may alter the equilibrium between ammonia and ammonium. For example, increasing temperature and pH will enhance ammonia emissions (Angelidaki and Ellegaard, 2003; Weaver et al., 2012; Nkoa, 2014). Air movement across an open lagoon surface also enhances NH_3 loss (Dari et al., 2019).

An increase in NH_3 emissions relative to conventional open lagoons during the month of April from a secondary lagoon filled with digestate from a covered anaerobic digester at Barham Farm was documented in North Carolina in 2008 as part of the North Carolina State University Animal and Poultry Waste Management Center's evaluation of potentially environmentally superior technologies (ESTs) (Aneja et al., 2008 a, b).

Barham Farm (Aneja et al., 2008 b) (35.70 °N, 78.32 °W, 130 m mean sea level [MSL]) is located near Zebulon in Johnston County, NC. Field campaigns were conducted during April 1–12, 2002 and November 11–22, 2002, at this farm site. A schematic layout of the EST at Barham farm including the various sampling locations is given in Figure 2 of the attached paper (Aneja et al., 2008 b). This potential EST had an in-ground ambient digester comprised of a covered anaerobic waste lagoon. The primary lagoon was covered by an impermeable layer of 40-mm thick high-density polypropylene that prevented gaseous methane and other gases and odor from escaping into the atmosphere during the digestion process. Methane gas produced during the digestive process was extracted and burned into a biogas generator to produce electricity. Heat from the generator was captured and used to produce hot water that was used by the farm in its production activities. Effluent from the digester (covered lagoon) flowed into a storage pond with a surface area of 4459 m^2. This storage pond was formerly part of the primary anaerobic lagoon before the digester was built. A portion of this effluent was supposed to be further treated via biofilters, the purpose of which was to convert NH_4^+ to nitrate in the effluent. This nitrified effluent was then used to flush out the swine production facilities, and the excess effluent was channeled into the larger overflow pond with a surface area of 19,398 m^2. A heavy polymer baffle separated the overflow and storage ponds. The overflow pond was used to store rainwater and overspills from the storage pond. Water from the overflow pond was also pumped into a nitrification biofiltration system where the nutrients in the treated effluent were used to fertilize vegetables grown in greenhouses adjacent to the swine production facility. In this study, NH_3 flux measurements were made from the surfaces of the storage pond, the overflow pond, and from the covered anaerobic primary lagoon. Average NH_3 concentrations were measured using the OP-FTIR spectroscopy system across the forced ventilation fan openings, as well as along the sides of swine houses (barns) to estimate barn emissions during the experimental periods.

During the first measurement period in April, we were notified that the biofilter components of the EST were not operational during that time. These conditions and process make the Barham Farm results from April 2008 similar to the current biogas project developing in Eastern North Carolina. For the April results, the digestate was placed in an open water holding structure with no additional treatment, since the biofilters were not functioning. This means that the April 2002 sample results from the water-holding structure are functionally the same as an open secondary lagoon as is proposed for the recently-permitted North Carolina biogas project sites.

Analysis

Ammonia emission results for Barham Farm are presented in Table 4 of my 2008 paper (Aneja et al., 2008 b), “Summary of NH_3 emissions from the EST farms and percent reduction during the experimental periods,” and I have supplemented them with greater granularity as shown below in the right-hand column in bold. The percent reduction for the water-holding structure (lagoon) was calculated using the data in Table 4 of my 2008 paper and Formula 3 on page 1152 of the paper.

Barham Farm - Ammonia Emissions

Sampling Period	Emission Source	%E_{EST}	%E_{EST} (WHS + House)	%E_{conv}	%E_{conv} (WHS + House)	%Reduction (WHS + House)	%Reduction (WHS)
April 2002	WHS	18.8	39.4	11.3	35.2	-11.9	**-66.4**
	House	20.6		23.9			
November 2002	WHS	4.0	31.7	9.7	32.5	2.5	**58.8**
	House	27.7		22.8			

WHS = water-holding structure (lagoon)
House = animal confinement barns
E_{EST} = ammonia emissions at Barham Farms
E_{conv} = emissions at conventional CAFO employing lagoon and sprayfield technology

This detailed analysis shows the following:

1. Ammonia emissions for both the water-holding structure and the houses combined at Barham Farm during the warmer period, April 2002, when “we were notified that the EST was not fully functioning as designed, because the biofilters were not operational during that time,” shows an increase (i.e., a negative reduction) in ammonia emissions of 11.9% (WHS+House).

2. Ammonia emissions for the water-holding structure during April 2002 when “we were notified that the EST was not fully functioning as designed, because the biofilters were not operational during that time” showed an increase in ammonia emissions of approximately 66% (WHS only).

3. During the cooler period, November 2002, when the EST treatment system was functioning, there was a decrease in total ammonia emissions of 2.5% (WHS+House).

4. During the cooler period, November 2002, when the EST treatment system was functioning, there was a decrease in ammonia emissions from the water-holding structure of approximately 59% (WHS only).

This additional and detailed analysis of ammonia emissions leads me to conclude that the implementation of a biogas system using a covered anaerobic digester followed by an open secondary storage lagoon without additional treatment will have serious negative consequences at CAFOs in North Carolina by way of enhanced ammonia emissions and impact on the environment. These enhanced ammonia emissions could be eliminated, and the problem of overall ammonia emissions could be mitigated, by adding an effective treatment module.

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ATTACHMENT 40

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Ammonia and greenhouse gas emissions from slurry storage - A review

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ABSTRACT

Storage of slurry is an important emission source for ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4), carbon dioxide (CO_2) and hydrogen sulfide (H_2S) from livestock production. Therefore, this study collected published emission data from stored cattle and pig slurry to determine baseline emission values and emission changes due to slurry treatment and coverage of stores. Emission data were collected from 120 papers yielding 711 records of measurements conducted at farm-, pilot- and laboratory-scale. The emission data reported in a multitude of units were standardized and compiled in a database. Descriptive statistics of the data from untreated slurry stored uncovered revealed a large variability in emissions for all gases. To determine baseline emissions, average values based on a weighting of the emission data according to the season and the duration of the emission measurements were constructed using the data from farm-scale and pilot-scale studies. Baseline emissions for cattle and pig slurry stored uncovered were calculated. When possible, it was further distinguished between storage in tanks without slurry treatment and storage in lagoons which implies solid-liquid separation and biological treatment. The baseline emissions on an area or volume basis are: for NH_3: 0.12 g m^{-2} h^{-1} and 0.15 g m^{-2} h^{-1} for cattle and pig slurry stored in lagoons, and 0.08 g m^{-2} h^{-1} and 0.24 g m^{-2} h^{-1} for cattle and pig slurry stored in tanks; for N_2O: 0.0003 g m^{-2} h^{-1} for cattle slurry stored in lagoons, and 0.002 g m^{-2} h^{-1} for both slurry types stored in tanks; for CH_4: 0.95 g m^{-3} h^{-1} and 3.5 g m^{-3} h^{-1} for cattle and pig slurry stored in lagoons, and 0.58 g m^{-3} h^{-1} and 0.68 g m^{-3} h^{-1} for cattle and pig slurry stored in tanks; for CO_2: 6.6 g m^{-2} h^{-1} and 0.3 g m^{-2} h^{-1} for cattle and pig slurry stored in lagoons, and 8.0 g m^{-2} h^{-1} for both slurry types stored in tanks; for H_2S: 0.04 g m^{-2} h^{-1} and 0.01 g m^{-2} h^{-1} for cattle and pig slurry stored in lagoons. Related to total ammoniacal nitrogen (TAN), baseline emissions for tanks are 16% and 15% of TAN for cattle and pig slurry, respectively. Emissions of N_2O and CH_4 relative to nitrogen (N) and volatile solids (VS) are 0.13% of N and 0.10% of N and 2.9% of VS and 4.7% of VS for cattle and pig slurry, respectively. Total greenhouse gas emissions from slurry stores are dominated by CH_4. The records on slurry treatment using acidification show a reduction of NH_3 and CH_4 emissions during storage while an increase occurs for N_2O and a minor change for CO_2 as compared to untreated slurry. Solid-liquid separation causes higher losses for NH_3 and a reduction in CH_4, N_2O and CO_2 emissions. Anaerobically digested slurry shows higher emissions during storage for NH_3 while losses tend to be lower for CH_4 and little changes occur for N_2O and CO_2 compared to untreated slurry. All cover types are found to be efficient for emission mitigation of NH_3 from stores. The N_2O emissions increase in many cases due to coverage. Lower CH_4 emissions occur for impermeable covers as compared to uncovered slurry storage while for permeable covers the effect is unclear or emissions tend to increase. Limited and inconsistent data regarding emission changes with covering stores are available for CO_2 and H_2S. The compiled data provide a basis for improving emission inventories and highlight the need for further research to reduce uncertainty and fill data gaps regarding emissions from slurry storage.

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1. Introduction

Livestock production systems around the world generate slurry—a mixture of feces and urine from housed livestock, mixed with bedding material and cleaning water (Pain and Menzi, 2011). Storage of slurry is required to enable the spreading in the field at appropriate time to supply nutrients to crops. Thus, a major part of the slurry is transferred from housings to outdoor stores such as tanks (at or above ground level) or earthen lagoons. Stores have variable forms and dimensions (e.g. up to several hectares for lagoons) according to the required storage volume. They have been identified as important emission sources for ammonia (NH_3), hydrogen sulfide (H_2S) and greenhouse gases (GHGs) including nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) from livestock production. Slurry stores are complex systems which influence emissions in many ways (Sommer et al., 2006; VanderZaag et al., 2008; Sommer et al., 2013).

A thorough description on principal mechanisms influencing the release of NH_3, GHGs and H_2S from slurry stores can be obtained from several studies (Olesen and Sommer, 1993; Ni, 1999; Sommer et al., 2006; VanderZaag et al., 2008; Sommer et al., 2013). Some important basic principles are summarized here. Slurry stores have a defined area where the gas exchange with the atmosphere takes place. It is a diffusive process and is quantified by emission rate values with the unit mass per area and time. Dissolved species of the gases are produced through microbial breakdown of nitrogen or organic compounds in the bulk slurry. Depending on prevailing chemical equilibria (e.g. NH_3/NH_4^+ which shifts to NH_4^+ at a low pH-value) and absence of microbial consumption, the gases move towards the emitting surface driven by diffusion (i.e. movement due to concentration gradients) and convection where parcels of air or liquid induce a movement of the compounds in the slurry (Sommer et al., 2013). At the slurry-air interface, the compounds pass gas- and liquid-phase resistances and diffuse into the air where they are transported to the atmosphere by convection. Transport within the liquid phase is temperature dependent and the gas-phase transfer is dependent on both temperature and turbulence (VanderZaag et al., 2015). Depending on the dry matter content of the slurry or more precisely, the amount of particles in the slurry which is influenced by the slurry type, animal species, animal diets, the thickness of the slurry bulk layer in the stores and meteorological conditions (Smith et al., 2007), a natural crust at the slurry surface can develop. It constitutes a barrier to the gas molecules between the liquid and the air. NH_3 and CH_4 may be consumed due to microbial activity in the crust leading to an emission reduction (Petersen and Ambus, 2006; Nielsen et al., 2010) while N_2O production may be enhanced (VanderZaag et al., 2009).

Ammonia has a large variety of negative environmental impacts which encompass the quality of air, soil and water, ecosystems and biodiversity. Moreover, it contributes to the formation of particulate matter which impairs human health (Sutton et al., 2011). N_2O and CH_4 are strong GHGs (Myhre et al., 2013). H_2S is often related to odor nuisances and can be lethal to animals and humans at high exposure levels (Sommer et al., 2013). NH_3 and GHG emissions have been regulated by the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (UNECE, 1999) and by the Kyoto protocol arising from the UN Framework Convention on climatic change (UN, 1997), respectively. Member countries of these protocols are obliged to calculate and report their national emissions annually, to track changes and compare to national emission ceilings where applicable. The methods for emission reporting are defined in EEA (2016) for NH_3 and in IPCC (2006) for N_2O and CH_4.

EEA (2016); IPCC (2006) and UNECE (2014) provide emission factors for slurry storage or numbers for emission reduction related to mitigation techniques which are used for emission reporting in emission inventories. However, a considerable number of recent studies on emissions from slurry storage provide updated information. The present review paper aims therefore to collect the data on NH_3, GHGs (CH_4, N_2O, CO_2) and H_2S emissions from these recent but also from previous studies and to provide a comprehensive overview on emissions from cattle or pig slurry stored uncovered and emission changes due to slurry treatment and coverage of slurry stores. This information can be used for the purpose of guide values, e.g. for the evaluation of emission data, and for improving emission inventories (greater accuracy, reduced uncertainty), e.g. for the determination of baseline emissions or emission reductions due to slurry treatment or coverage of slurry stores. The compiled data is entirely provided in the Supplementary data 2 for tracking the present or conducting future analyses.

2. Material and methods

2.1. Data search and data selection

A literature research was carried out with Web of Science [5.3] using the following search terms: "storage", "slurry", "emission"; "lagoon", "slurry", "emission". These searches were done on January 10, 2018 and yielded 601 papers in total. In a first screening, 290 papers were eliminated because they did not encompass livestock slurry. The remaining 311 articles were retained. In addition, 58 papers were found in the reference list of the screened articles. Therefore, in total, 369 articles were retained for further screening according to the following criteria:

(i) The investigated slurry was produced in an animal operation and consisted of urine and feces excreted from the animals onto a floor of a barn, a hardstanding or a milking parlor. The slurry might contain solids like bedding material or feed residues and be diluted with water. The investigated slurry was untreated or submitted to a treatment such as solid-liquid separation, anaerobic digestion, addition of an acid (acidification), additives or co-substrates. The treatment occurred under real-world conditions or after slurry sampling in the laboratory. Studies based on synthetic slurry, e.g. urine and feces collected separately from animals and subsequently combined in the laboratory, were excluded since fresh animal excretions substantially differ in chemical composition from stored slurry (Table 6). Moreover, urine and feces deposited onto a floor can rapidly undergo processes leading to gaseous losses. Hence, synthetic slurries might induce different emission levels as compared to slurries submitted to real-world conditions.

(ii) The untreated or treated slurry was transferred from the animal operation to a storage tank or a lagoon outside of animal housings and then submitted to measurements under real-world conditions or the slurry as characterized under point (i) was collected from a floor, an underfloor pit or an outside store and subsequently transferred to an experimental vessel where emissions were measured at pilot- or laboratory-scale. Studies encompassing e.g. emissions from a pit below an animal confinement were excluded since such facilities provide an environment which substantially differs from outside stores (e.g. exposure to outdoor climate, disturbance of the slurry surface due to continuous addition of animal excretions over almost the whole area of a pit).

(iii) The reported emission data are based on experimental determination of emission rates as defined by VanderZaag et al. (2008). Studies providing gas concentrations only were excluded.

(iv) The article provides numerical data encompassing emission data or percent differences in emissions between a slurry submitted to a treatment or slurry stored with covering and a reference system with untreated slurry or uncovered storage, respectively.

After evaluation, 120 papers complied with criteria (i) to (iv). 93 papers did not provide numerical data or comply with these criteria but included substantial information on emissions from slurry storage, e.g. basic mechanisms driving emissions. The remaining 156 papers were excluded because they were out of topic or did not provide substantial information. An overview on the screened papers is in Supplementary data 1.

2.2. Data extraction

Data from the 120 papers were extracted. The parameters as shown in Table 1 were transformed, standardized or aggregated where necessary and then compiled in a database. Overall, 711 records were available for the analysis where one record is defined as an ensemble of entries listed in Table 1 (i.e. multiple records may be created from a single paper). Each record may differ in completeness according to the information provided in a paper.

2.3. Standardization of emission data

Emissions were reported in the papers using numerous units involving the gas molecule (i.e. NH_3, N_2O, CH_4, CO_2 and H_2S) or N, C or S included therein and various units for weight, time and surface or volume. Also, cumulative emissions were given over the entire experimental period. Overall, 36, 22, 31, 13 and 3 different ways for emission reporting were found for NH_3, N_2O, CH_4, CO_2 and H_2S, respectively. Standardization was performed in the present study to obtain comparable values over all records. For all emission rates, the unit g of molecules was used according to UNECE, (2015) and IPCC (2006). An emission on an area basis was applied for NH_3, N_2O, CO_2 and H_2S. For CH_4, the emission relative to the bulk volume was employed. Due to the availability of numerous additional records, data relative to the area were also provided for CH_4. For the area and the volume, the unit m^2

Table 1
Parameters extracted from the papers after transformation or standardization and transferred into the database. Explanations are given for parameters marked with symbols in the table footnote. The complete extracted data are provided in the Supplementary data 2.

Parameter	Explanation
Year	Date the study was published
Country	Location where the study was done
Slurry type	Cattle or pig
Slurry treatment	Untreated, solid-liquid separation, anaerobic digestion, acidification, aeration, addition of additives, dilution with water, addition of co-substrates (also denoted off-farm materials; mostly organic residues from e.g. food industry or energy crops) and combinations of treatments (e.g. solid-liquid separation and anaerobic digestion)
Slurry characteristics	Chemical analysis of the slurry: dry matter (DM), volatile solids (VS), total nitrogen (N_{tot}), ammonium (NH_4^+; TAN (total ammoniacal nitrogen) is often used instead of NH_4^+), total carbon (C), total sulfur (S) in g L^{-1}, pH
Type of study*	Farm-scale, pilot-scale, laboratory-scale
Type of store	For farm-scale studies: tank, lagoon according to Pain and Menzi (2011)**
Replicates	Number of replicates of real-world stores or experimental vessels
Store characteristics	Investigated store surface (m^2), depth (m), and volume (m^3); agitation of slurry (number of agitation events); other producer events or meteorological conditions; slurry temperature (°C)
Experimental conditions	Duration of storage of investigated slurry (days); duration of the study (days); number of measurement periods and total duration of the measurement (hours); season of measurements: cold, temperate, warm; for the determination of the season, the meteorological winter, spring or fall and summer were considered
Meteorological conditions	Air temperature during measurements (°C); air speed over the emitting surface during measurements (m s^{-1}); rainfall (cumulative amount during measurements in mm)
Measurement methods applied	Measurement method for the gases: dispersion modeling based on a backward Lagrangian stochastic (bLS) dispersion model or UK-ADMS atmospheric dispersion model, flux chamber method, flux gradient method, micrometeorological mass balance method (e.g. integrated horizontal flux, IHF; vertical radial plume mapping, VRPM), sampling at exhaust chimney, tracer gas method, method not further defined; instrument used for the concentration measurements of the gases
Cover type	Storage uncovered or covered; For covered storage: cover type according to VanderZaag et al. (2015): impermeable structural covers: lid (wood or concrete), tent covering; impermeable floating covers: plastic film; permeable synthetic floating covers: plastic fabrics, expanded clay, other materials such as expanded polystyrene, plastic tiles; permeable natural floating covers: peat, straw, vegetable oil, other organic materials (wood chips, sawdust etc.), other cover types such as storage bag
Occurrence of a natural crust at the store's surface	Formation of natural crust: yes or no, crust thickness (cm), time for natural crust formation (days)***
Measurement data****	NH_3 (g NH_3 m^{-2} h^{-1}, g NH_3 m^{-3} h^{-1}, g NH_3 AU^{-1} h^{-1}), NH_3-N in % TAN and in % N, N_2O (g N_2O m^{-2} h^{-1}, g N_2O m^{-3} h^{-1}, g N_2O AU^{-1} h^{-1}), N_2O-N in % TAN and % N, CH_4 (g CH_4 m^{-2} h^{-1}, g CH_4 m^{-3} h^{-1}, g CH_4 AU^{-1} h^{-1}), CH_4-C % VS, CO_2 (g CO_2 m^{-2} h^{-1}, g CO_2 m^{-3} h^{-1}, g CO_2 AU^{-1} h^{-1}), CO_2-C in % VS, CO_2eq (g CO_2eq m^{-2} h^{-1}, g CO_2eq m^{-3} h^{-1}, g CO_2eq AU^{-1} h^{-1}), H_2S (g H_2S m^{-2} h^{-1}, g H_2S m^{-3} h^{-1}, g H_2S AU^{-1} h^{-1}); Difference between untreated and treated slurry or between slurry stored uncovered and stored covered in percent for NH_3, N_2O, CH_4, CO_2, CO_2eq, H_2S

* Type of study: Farm-scale: measurements carried out at real-world storage facilities at a farm site. This information could be obtained from the description of the experimental setup given in the papers. Pilot-scale and laboratory-scale: measurements conducted under controlled conditions in experimental vessels. Due to a lack of definition for these study types, a discrimination according to the following characteristics was employed: Pilot-scale: volume of slurry investigated: ≥500 L with experimental vessels situated outdoors, with or without a shelter and submitted to ambient meteorological conditions. Laboratory-scale: volume of slurry investigated: < 500 L. Most of the studies defined as laboratory-scale studies were conducted indoors in a temperature-controlled room. Three studies deviated from the conditions regarding study situation or temperature control and for four studies, this information was not available (Supplementary data 2). Despite these gaps in information, the studies were retained.

** A tank is a large, normally open-top, in most cases circular vessel made from pre-fabricated vitreous enameled steel, concrete or wood panels charged from a reception pit and emptied using a pump. It is a facility constructed at or below ground level and may extend above ground with a depth of several meters. Earthen storage basins not designed for biological treatment of slurry are considered as stores equivalent to tanks. Like earthen storage basins, a lagoon is a large rectangular or square shaped structure with sloping earth bank walls and may be lined with water impermeable material. Lagoons are designed for both storage and biological treatment (Pain and Menzi, 2011). They are not emptied below a specific depth necessary for slurry treatment except for maintenance (Hamilton et al., 2001).

*** We did not consider a natural crust as a mitigation technique equivalent to covering of slurry stores. The significance of crusting and considerations regarding distinction between crusting and storage covering are specifically addressed in Section 4.2.4.

**** For units: see Section 2.3. Acronyms: AU: animal unit = animal with a live weight of 500 kg; CO_2eq: carbon dioxide equivalent. CO_2eq is a standardized unit for different greenhouse gases. The numbers reported rest on data provided by the authors of the papers which were mostly based on IPCC (2007); TAN: total ammoniacal nitrogen; N: nitrogen; VS: volatile solids.

and m^3 was used, respectively. For all gases, the time unit hour was applied (reasons are given in section 4.1). Where useful for inventories, the time unit year was additionally provided for emissions. In this paper, the emission data standardized as explained above are denoted emission on an area or volume basis.

Since emission inventories do usually not apply emissions on an area or volume basis but emission factors which express emissions as a proportion of a compound present in the slurry store, data were additionally scaled as follows: percent of TAN for ammonia (EEA, 2016), percent of N for N_2O (IPCC, 2006) and percent of VS for CH_4 (IPCC, 2006) and CO_2. To be consistent with the notion "emission on an area or volume basis" regarding terminology, we used the term flow-based emission. Flow-based emissions were either taken from the papers or determined based on the emission rate, the N, TAN or VS content of the slurry, the volume of the store and the duration of the experiment. Dividing the cumulative emission which was derived from the emission rate and the duration of the study by the amount of the compounds present in the store at the beginning of the experiment (derived from the slurry content of N, TAN or VS and the slurry volume) yielded the flow-based emission. It was only calculated if no slurry addition or discharge occurred during the experiment.

2.4. Data analysis

2.4.1. Descriptive statistics of the emission data

In a first step, descriptive statistics (number of records, minimum, 1 st quantile, median, average, 3rd quantile, maximum, standard deviation) were calculated over all records encompassing slurry stored uncovered. There were eight categories for data reporting resulting from the combination of two slurry types (cattle and pig) with four study types (farm-scale lagoon comprising solid-liquid separation and biological treatment of slurry; farm-scale tank, pilot-scale and laboratory-scale which include untreated slurry).

2.4.2. Baseline emissions

2.4.2.1. Definition. We define the term baseline emission as the average emission occurring with slurry storage according to the reference technology without emission control similar to VanderZaag et al. (2015). This implies uncovered storage in the following types of store: i) tanks or earthen stores without slurry treatment; ii) lagoons with solid-liquid separation and biological treatment occurring during storage (Hamilton et al., 2001). The baseline emission is considered as representative for average emissions over the whole course of a year. According to EEA (2016), baseline emissions are given separately for cattle and pig slurry. We further distinguished between storage in tanks (or earthen stores) and lagoons. Baseline emissions were calculated from uncovered slurry stores regardless of the occurrence of a natural crust because its formation can be only partially controlled and thus varies widely between stores (Smith et al., 2007). Moreover, there was insufficient information about the presence of crusts in the data impeding a distinction between crusted and non-crusted store's surfaces.

2.4.2.2. Determination. Baseline emissions were calculated using farm-scale and pilot-scale studies published in peer-review papers. For the calculation of representative emissions, important influencing factors should be considered such as the meteorological conditions (mainly air temperature, wind speed, precipitation) and operations at storage facilities (Sommer et al., 2013). Among these factors, we were able to include air temperature since the season used for emission measurements which can be used as surrogate for the temperature was available for more than 90% of the records. Records were dropped where conditions prevailed which are not representative for slurry storage in practice over a longer period, e.g. if daily agitation of slurry occurred. More detailed information on meteorological conditions and operations at storage facilities was not available and could not be included in the evaluation of emissions (e.g. only approx. 60% of records provided numerical air temperature data). Information on wind speed, precipitation and crust formation was available for less than half of the records.

We hypothesized that for generating emission data which are representative over the whole course of a year, emission values generated during the cold, the warm and the temperate season (spring, fall) should be equally covered. To achieve this, a weighting of the emission data for season was done. Values were aggregated according to the categorization "Season code" ("c": cold season = winter, "t": temperate season = spring or fall, "w": warm season = summer, "c,t": cold and temperate season, "c,w": cold and warm season, "t,w": temperate and warm season, "c,t,w": cold, temperate and warm season), "Slurry type" (cattle, pig), "Type of study" (farm-scale, pilot-scale) and "Type of store" (for farm-scale studies: lagoon, tank). For some papers, emission values for each individual season "c", "t", and "w" were provided and also the average value over the year, i.e. the "c,t,w" value. In these cases, the "c,t,w" value is denoted as redundant in the database (Supplementary data 2). It was used for the further calculations and not the values of the individual seasons. The aggregated values were averaged afterwards in the following manner:

i) Study duration varied considerably, i.e. individual experiments ranged from less than one day up to several months. The individual records were thus weighted according to measurement durations of records within each "Season code" category. The individual records were aggregated to four classes of measurement durations: a) ≥1 month, b) ≥1 week to < 1 month, c) ≥1 day to < 1 week, d) < 1 day. Weighting was done based on the square-root of the median of the measurement duration for each class to avoid over-emphasis of long-term measurements. The median values of the measurement duration for the 4 classes a, b, c and d were 146.5 days, 16.9 days, 4.5 days and 0.34 days, respectively. This implied the following respective weights 12.1, 4.1, 2.1 and 0.6. Therefore, a record based on a measurement of more than one month received a weighting 20.8 times higher than a record based on a measurement over less than a day.
ii) Average values for each season "c", "t" and "w" were calculated from all available values within one category (based on "Slurry type", "Type of study" and "Type of store"). Averaging was done in a way that values spanning over more than one season were attributed to the respective seasons, i.e. a value for "c,t" was attributed half to "c" and half to "t", a value of "c,t,w" was counted one fourth to seasons "c" and "w" and one half to season "t". For example, to average a "c" value based on a 2 weeks measurement (c_{2weeks}), a "c" value based on a 2 days measurement (c_{2days}) and a "c,w" value that based on a 2 months measurement ($cw_{2months}$) led to the following average "c" value: "c_{avg}" = (c_{2weeks}*$weight_{2weeks}$ + c_{2days}*$weight_{2days}$ + $cw_{2months}$*0.5*$weight_{2months}$)/($weight_{2weeks}$ + $weight_{2days}$ + 0.5*$weight_{2months}$).
iii) These average values were further averaged to annual emission rates "c,t,w" by weighting the value for season code "t" twice as high as the seasons "c" and "w" (i.e. "c,t,w_{avg}" = ¼*"c" + ½*"t" + ¼*"w") since the temperate season code "t" includes two seasons (spring and fall). These final averaged values are listed in column "Avg" in Tables 8, 9 and Supplementary data 4.

Numbers for baseline emissions are reported as average emission values if at least one record for each of the season "c", "t" and "w" was available. Included can be a record from an individual season (i.e. "c", "t" or "w"), or any kind of seasons combination (i.e. "c,t" "c,w" "t,w" or "c,t,w"). The lower and upper 95% confidence bounds (l95, u95) for baseline emissions were determined using bias-corrected and accelerated bootstrap intervals (Efron, 1987) if at least three individual records for each of the season "c", "t" and "w" were available. Again, this can be in the form of an individual season or any kind of seasons combination as for the calculation of the average. The bootstrapping was done as non-parametric bootstrapping with sampling stratified by season. To test whether there are significant ($p < 0.05$) differences in these baseline emission values, 95% confidence intervals were obtained from bootstrapping the differences between each combination of values. If a confidence interval of a difference did not include 0, the difference was marked as statistically significant.

The data resulting from this procedure related to emissions on an area or volume basis were aggregated according to the slurry type (cattle and pig) and the study types farm-scale tank and pilot-scale and for the two study types combined which were denoted as baseline emissions tank. The baseline emissions for lagoons are based on measurements carried out at farm-scale for lagoons. Baseline emissions expressed as flow-based emissions were given separately for cattle and pig slurry for tanks only due to a lack of appropriate data for lagoons. The calculation procedure is additionally illustrated based on an example in the Supplementary data 9.

2.4.3. Emissions and emission changes due to slurry treatment and covering of slurry stores

We determined the emission changes due to slurry treatment techniques and covering of slurry stores using records with a treatment or a cover and a reference system (uncovered storage with untreated slurry) to compare the emissions on an area or volume basis from both. Due to the limited number of available records, the restriction to peer review papers and exclusion of laboratory-scale studies was not applied. For storage covering, all records with less than 20 cm of slurry depth were excluded from the data analysis since it is likely that such conditions differ too much from the real-world and even more evident if the thickness of the cover material is similar to that of the bulk slurry layer. Studies where slurry depth was not provided were excluded.

Although a natural crust is often listed as abatement measure together with slurry store covers (Bittman et al., 2014) we did not consider it as a mitigation technique equivalent to covering of slurry stores. In contrast to coverings such as impermeable floating covers, it is not applicable for all stores since it does not form at each slurry type. Crusting was neither considered for the analysis on emission changes due to slurry treatment and covering of slurry stores because of insufficient information about the presence of crusts in the experimental data. The significance of crusting is specifically addressed in Section 4.2.4.

The numbers from different studies were aggregated without a weighting for season or measurement duration due to the limited number of records. We tested whether the differences between treatments or covers and the reference system (untreated slurry or uncovered storage) were significantly different from zero by a two-sided t-test.

3. Results

3.1. Characterization of the database

3.1.1. General characteristics

The literature review yielded a total of 711 records. Among them, 13% were from before 2000. The period between 2000 and 2010 contributed 43%, and 44% were published after 2010 (Table 2). US and CA generated 28% and 19%, respectively, of the records while 11 European countries provided 48%. Two countries from Asia and Oceania contributed 3% and 2% of the records. Ammonia was studied in 38% of the records, while 59% were on GHGs, and 3% on H_2S. Among GHGs, CH_4 was most often investigated with a share of 30% of all records. 47% of the records included one gas and 53% several gases.

Table 3 shows the types of studies. A share of 46% of the records are based on studies conducted at farm-scale. Pilot-scale studies contributed 31% and laboratory-scale 23% of the records. Records from pilot-scale studies are similarly represented over all three periods before 2000, between 2000 and 2010 and after 2010. In contrast, data from farm-scale studies and conducted in the laboratory occur more frequently from 2000 onwards.

An overview of the investigated slurry types is shown in Table 4. Cattle and pig slurry each account for about 50% of the investigated slurries. Cattle slurry mostly originated from dairy cows while for pig slurry fattening pigs and breeding pigs or a mixture of both was studied. Other types of slurry were included in measurements as well, but these occur much less. The proportion of untreated slurries is 65% and 87% for cattle slurry and pig slurry, respectively. Solid-liquid separation occurs for 16% (cattle slurry) and 3.7% (pig slurry) of the records. Anaerobic digestion of unseparated slurry applies for 7.2% (cattle slurry) and 3.7% (pig slurry) of the records while for anaerobically digested and separated slurry, the numbers are 8.1% for cattle slurry and 0.6% for pig slurry, respectively. Other treatments encompass acidification, aeration, supplementation with additives or dilution of slurry, but these treatments occur less.

Approximately 140 records compare the emissions between covered and uncovered storage. More than 80% of these data are from pilot and laboratory studies. Straw covers and other natural materials such as wood chips or maize stalks were most often investigated (51 in total). Also, cover types such as a lid, plastic film and fabrics were frequently addressed resulting in approximately 15 records for each.

Measurement methods employed in the experiments are shown in Table 5. Roughly, two thirds of all measurements were carried out using a flux chamber method. While this is almost the only option for pilot- and laboratory-scale studies, this system was also used for approximately 30% of the measurements conducted at farm sites. Methods like dispersion modeling or micrometeorological mass balance method make up about 60% of the records from farm-scale studies. Other methods e.g. using a tracer gas were rarely applied.

Slurry analyses are shown in Table 6. Not all studies provided analytical data of the slurry (e.g., only 84% of NH_3 studies presented TAN values). While most laboratory studies analyzed TAN, only 67% of the studies carried out at farm sites reported this parameter. Pilot-scale studies lie in between with 92% of records reporting TAN data. The availability of analytical data is similar for other parameters (e.g. DM) as for TAN but with somewhat lower numbers. The composition of the mixture of urine and feces as excreted by animals published by ASAE (2005) and Richner et al. (2017) is added at the bottom of Table 6. They provide numbers for cattle on DM, VS and TAN in the range of 80 to 90 g L^{-1}, 53 to 70 g L^{-1} and 1.4 to 2.1 g L^{-1}, respectively. For pigs, the values for DM, VS and TAN are in the range of 50 to 90 g L^{-1}, 36 g L^{-1} and 3.4 to 5.0 g L^{-1}. The slurry analyses given in the records show substantially lower numbers for DM and VS contents for untreated slurries which is most likely due to dilution with water from farm operation and rainfall at the farms (Table 6). Studies at farm-scale based on tanks, at pilot-scale and at laboratory-scale exhibit DM contents which are in a similar range within cattle and pig slurry. Numbers for DM are lower for pig slurries compared to cattle slurry except for laboratory scale studies. Pig slurry exhibits higher N_{tot} and TAN contents than cattle slurry. Within farm-scale studies, the numbers for all analytes strongly differ between slurry from tanks and from lagoons. Values for DM, VS, N_{tot} and TAN are lower for lagoons by a factor of approximately two to eight as compared to slurry stored in tanks. Slurries from lagoons compare better with slurries after solid-liquid separation (Table 6) than with untreated slurries.

3.1.2. Descriptive statistics of emission data from cattle and pig slurry stored uncovered

Descriptive statistics are shown in Table 7 for NH_3, N_2O, CH_4, CO_2 and H_2S over all records encompassing untreated cattle and pig slurry stored uncovered from studies conducted at farm-, pilot- and laboratory-scale (farm-scale studies with lagoons include biologically treated and separated slurry; see section 2.4.2.1). Data from measurements conducted during warm, temperate and cold seasons are unevenly

Table 2
Number of records listed by country and year of publication and share of total records by country.

Country	Before 2000		2000 to 2010		After 2010		total			Share of total
	Cattle	Pig	Cattle	Pig	Cattle	Pig	Cattle	Pig	Cattle and pig	
AT	0	0	15	6	0	0	15	6	21	3%
AU	0	0	0	0	0	4	0	4	4	1%
CA	1	1	47	10	72	5	120	16	136	19%
CN	0	0	0	0	0	17	0	17	17	2%
DE	8	9	4	2	0	0	12	11	23	3%
DK	20	17	0	0	6*	14	22	35	57	8%
ES	0	0	0	0	1	0	1	0	1	0.1%
FR	0	0	2	33	0	6	2	39	41	6%
IT	0	0	23	32	12	12	35	44	79	11%
JP	0	0	0	0	3	0	3	0	3	0.4%
LT	0	0	0	0	0	21	0	21	21	3%
NL	13	11	4	4	0	0	17	15	32	5%
NZ	6	0	1	1	2	0	9	1	10	1%
PT	0	0	4	0	4	8	8	8	16	2%
SE	0	0	1	0	6	3	7	3	10	1%
UK	1	3	13	2	6	13	20	18	38	5%
US	1	3	13	86	74	25	88	114	202	28%
Total	50	44	127	176	182	132	359	352	711	100%
Share of total	13%		43%		44%		100%			

* Cattle slurry with addition of other types of manure and feedstock materials.

Table 3
Number of records classified by type of study (farm-scale, pilot-scale, laboratory-scale) and time periods of publication and in percent of the total.

Type of study	Before 2000	2000 - 2010	After 2010	Total	Share of study types
Farm-scale	27	157	141	325	46%
Pilot-scale	54	75	90	219	31%
Laboratory-scale	13	77	77	167	23%
Total	94	309	308	711	100%
Publication of study types over time (in percent of total)					
Farm-scale	8%	48%	43%	100%	
Pilot-scale	25%	34%	41%	100%	
Laboratory-scale	8%	46%	46%	100%	

Table 4
Overview on investigated slurry types stored uncovered or covered: number of records listed by slurry treatments, slurry types and share of the total records in percent.

Slurry treatment	Cattle	Pig	Other*	Cattle	Pig	Other*
	n			Percent of total		
Untreated	233	302	–	65%	87%	–
Solid-liquid separation	57	13	–	16%	3.7%	–
Anaerobic digestion	26	13	–	7.2%	3.7%	–
Anaerobic digestion, solid-liquid separation	29	2	4	8.1%	0.6%	100%
Acidification	5	3	–	1.4%	0.9%	–
Acidification, anaerobic digestion	–	2	–	–	0.6%	–
Acidification, anaerobic digestion, solid-liquid separation	–	1	–	–	0.3%	–
Acidification, solid-liquid separation	–	1	–	–	0.3%	–
Dilution	5	2	–	1.4%	0.6%	–
Addition of additives	3	3	–	0.8%	0.9%	–
Aeration	1	4	–	0.3%	1.1%	–
Aerobic treatment	–	2	–	–	0.6%	–
Total	359	348	4	100%	100%	100%

* Cattle slurry with addition of other types of manure and feedstock materials.

Table 5
Number of records classified by the measuring method and by the type of study.

Measuring method	Farm-scale	Pilot-scale	Laboratory-scale	Total	
Dispersion modeling based on bLS* or ADMS**	107	2		109	15%
Dispersion modeling based on bLS* and VRPM***	8			8	1.1%
Flux chamber method	98	213	167	478	67%
Flux gradient method	4			4	0.8%
Micrometeorological mass balance method	92			92	13%
Sampling at exhaust chimney	4			4	0.6%
Tracer gas method	7			7	1.0%
Method not defined	5	4		9	1.3%
Total	325	219	167	711	100%

* backward Lagrangian stochastic (bLS) dispersion model.
** UK-ADMS atmospheric dispersion model (Hill et al., 2008).
*** Vertical Radial Plume Mapping.

distributed over all records (Supplementary data 3). The minimum and maximum emission values differ by one to several orders of magnitude for all gases. The average often exceeds the median by a factor of two or more which is most pronounced for N_2O. This indicates a distribution of data being right skewed by high values. The variability of data and the occurrence of high maximum values is most pronounced for laboratory-scale studies. Striking high values exceeding the median by at least one order of magnitude for NH_3, CH_4 and CO_2 are reported in the laboratory-scale study of Guarino et al. (2006). For N_2O, high values were found from three studies conducted at farm- and pilot-scale (Clemens et al., 2006; Amon et al., 2007; Leytem et al., 2011) (Supplementary data 2,8,11). For H_2S, one figure from a laboratory-scale study stands out which exceeds all other values by two orders of magnitude (Hobbs et al., 1999).

Table 6
Number of records of a slurry type (cattle, pig), type of study (f: farm-scale; p: pilot-scale; l: laboratory-scale), type of store for farm-scale studies, and slurry treatment (untreated, sol-liq sep: solid-liquid separation) in the database. Number of records (n) with analytical data on DM, VS, N_{tot}, TAN and average contents of DM, VS, N_{tot}, TAN in g L^{-1} for untreated slurry.

Slurry type	Type of study*	Type of store	Slurry treatment**	Total number of records	DM	VS	N_{tot}	TAN	DM	VS	N_{tot}	TAN
				n	n				g L^{-1}			
Cattle	f	lagoon	untreated	73	19	7	13	14	17	3.7	1.2	0.2
Cattle	f	tank	untreated	39	21	9	19	25	67	48	3.1	1.5
Cattle	p		untreated	106	97	36	93	97	62	53	3.2	1.6
Cattle	l		untreated	35	31	24	29	31	57	43	3.0	1.3
Pig	f	lagoon	untreated	109	19	23	50	76	9.7	4.5	0.8	0.6
Pig	f	tank	untreated	55	35	9	33	35	42	37	3.3	1.9
Pig	p		untreated	63	56	30	56	54	50	33	4.6	3.2
Pig	l		untreated	68	68	43	64	64	59	56	4.7	2.9
Cattle	f tank, p, l		sol-liq sep	23	19	10	17	17	39	29	2.4	1.2
Pig	f tank, p, l		sol-liq sep	14	10	5	8	12	29	23	3.8	2.3
Cattle	Contents of mixture of urine and feces obtained from ASAE (2005)								*80*	*53*	*3.0*	*1.4*
Pig									*61-90*	*n.a.*	*4.7-7.0*	*3.4-5.0*
Cattle	Contents of mixture of urine and feces obtained from Richner et al. (2017)								*90*	*70*	*3.9*	*2.1*
Pig									*50*	*36*	*6.5*	*4.6*

n.a.: not available.
* f: farm-scale; p: pilot-scale; l: laboratory-scale.
** sol-liq sep: solid-liquid separation.

Table 7
Emissions from cattle and pig slurry stored uncovered in tanks at farm-scale, pilot-scale and laboratory-scale without slurry treatment and in lagoons with solid-liquid separation and biological treatment; descriptive statistics for NH_3, N_2O, CH_4, CO_2 and H_2S in g m^{-2} h^{-1} or g m^{-3} h^{-1}. n: number of records; Min: minimum; 1 st Qu: first quartile; 3 st Qu: third quartile; Max: maximum; Std: standard deviation. Additional information is provided in Supplementary data 3.

Slurry type	Study type		n	Min	1 st Qu	Median	Average	3rd Qu	Max	Std
				NH_3 g m^{-2} h^{-1}						
Cattle	Farm-scale	lagoon	35	< 0.01	0.03	0.10	0.13	0.21	0.36	0.11
Cattle	Farm-scale	tank	20	0.02	0.04	0.06	0.13	0.16	0.68	0.15
Cattle	Pilot-scale		53	< 0.01	0.03	0.07	0.09	0.11	0.44	0.08
Cattle	Laboratory-scale		19	< 0.01	0.02	0.04	0.26	0.33	1.4	0.43
Pig	Farm-scale	lagoon	74	< 0.01	0.04	0.08	0.15	0.18	0.68	0.18
Pig	Farm-scale	tank	23	0.03	0.06	0.10	0.22	0.27	1.0	0.26
Pig	Pilot-scale		22	0.01	0.06	0.20	0.24	0.26	0.92	0.23
Pig	Laboratory-scale		20	< 0.01	0.03	0.23	0.69	0.71	4.5	1.16
				N_2O g m^{-2} h^{-1}						
Cattle	Farm-scale	lagoon	13	< 0.001	< 0.001	< 0.001	0.002	0.001	0.02	0.006
Cattle	Farm-scale	tank	3	< 0.001	0.001	0.002	0.002	0.003	0.003	0.002
Cattle	Pilot-scale		46	< 0.001	< 0.001	0.001	0.003	0.004	0.04	0.007
Cattle	Laboratory-scale		6	< 0.001	< 0.001	< 0.001	0.005	0.001	0.03	0.01
Pig	Farm-scale	lagoon	6	< 0.001	< 0.001	< 0.001	0.003	0.002	0.01	0.005
Pig	Farm-scale	tank	5	Not detected						
Pig	Pilot-scale		17	< 0.001	< 0.001	< 0.001	0.01	0.001	0.06	0.02
Pig	Laboratory-scale		4	< 0.001	< 0.001	< 0.001	0.003	0.003	0.01	0.006
				CH_4 g m^{-3} h^{-1}						
Cattle	Farm-scale	lagoon	3	0.27	0.29	0.30	0.77	1.0	1.7	0.83
Cattle	Farm-scale	tank	7	< 0.01	0.26	0.75	0.83	1.3	1.9	0.71
Cattle	Pilot-scale		46	0.01	0.07	0.42	0.56	0.75	3.6	0.69
Cattle	Laboratory-scale		15	< 0.01	0.15	0.64	10	16	51	16
Pig	Farm-scale	lagoon	2	< 0.01	0.88	1.8	1.8	2.6	3.5	2.5
Pig	Farm-scale	tank	10	0.02	0.25	0.55	1.6	3.1	5.0	1.8
Pig	Pilot-scale		21	0.01	0.13	0.18	0.77	1.0	3.4	1.1
Pig	Laboratory-scale		18	0.02	1.3	2.9	7.4	6.6	33	10
				CO_2 g m^{-2} h^{-1}						
Cattle	Farm-scale	lagoon	18	0.27	1.9	2.3	4.7	5.3	27	6.4
Cattle	Farm-scale	tank	3	11	11	11	16	18	25	8.1
Cattle	Pilot-scale		15	0.17	2.8	4.3	5.6	6.3	21	5.2
Cattle	Laboratory-scale		14	0.45	2.4	8.0	86	189	332	120
Pig	Farm-scale	lagoon	7	< 0.01	< 0.01	0.03	0.89	0.74	4.7	1.8
Pig	Farm-scale	tank	1	5.7	5.7	5.7	5.7	5.7	5.7	–
Pig	Pilot-scale		7	3.2	3.6	4.4	6.6	9.0	13	4.1
Pig	Laboratory-scale		14	1.0	6.3	9.1	52	80	217	75
				H_2S g m^{-2} h^{-1}						
Cattle	Farm-scale	lagoon	3	0.02	0.04	0.06	0.05	0.07	0.07	0.03
Cattle	Laboratory-scale		3	< 0.01	< 0.01	0.01	0.01	0.01	0.02	0.01
Pig	Farm-scale	lagoon	14	< 0.01	< 0.01	< 0.01	0.01	0.03	0.08	0.02
Pig	Laboratory-scale		6	< 0.01	< 0.01	< 0.01	0.47	0.02	2.8	1.1

Table 8
Emissions on an area or volume basis from cattle and pig slurry stored uncovered in tanks at pilot-scale and at farm-scale without slurry treatment and in lagoons with solid-liquid separation and biological treatment. Baseline emissions for storage in tanks and lagoons given in g m^{-2} h^{-1} / kg m^{-2} y^{-1} for NH_3, N_2O and CO_2 and in g CH_4 m^{-3} h^{-1} / kg CH_4 m^{-3} y^{-1}. n: number of records after aggregation; Avg: average; l95, u95: lower and upper 95% confidence bounds; cells denoted with "-": value is not available; #: values denoted with different letters are significantly different ($p < 0.05$). Detailed information is provided in Supplementary data 4.

Slurry type	Study type/baseline emissions		n	Avg	l95	u95	#	Avg yearly amount
				NH_3 g m^{-2} h^{-1}				NH_3 kg m^{-2} y^{-1}
Cattle	Pilot-scale studies		34	0.08	0.07	0.09	a	–
Cattle	Farm-scale studies	tank	11	0.09	0.05	0.13	ab	–
Cattle	**Baseline emissions***	**lagoon**	**28**	**0.12**	0.10	0.15	**bc**	**1.1**
Cattle	**Baseline emissions****	**tank**	**45**	**0.08**	0.07	0.09	**a**	**0.67**
Pig	Pilot-scale studies		15	0.24	0.15	0.38	def	–
Pig	Farm-scale studies	tank	8	0.23	0.13	0.37	cdef	–
Pig	**Baseline emissions***	**lagoon**	**40**	**0.15**	0.12	0.19	**ce**	**1.3**
Pig	**Baseline emissions****	**tank**	**23**	**0.24**	0.17	0.34	**f**	**2.1**
				N_2O g m^{-2} h^{-1}				N_2O kg m^{-2} y^{-1}
Cattle	Pilot-scale studies		33	0.002	0.001	0.002	a	–
Cattle	Farm-scale studies	tank	–	–	–	–	–	–
Cattle	**Baseline emissions***	**lagoon**	**11**	**< 0.001**	–	–	–	**< 0.01**
Cattle	**Baseline emissions****	**tank**	**35**	**0.002**	**0.001**	**0.002**	**a**	**0.02**
Pig	Pilot-scale studies		12	0.002	< 0.001	0.005	a	–
Pig	Farm-scale studies	tank	2	< 0.001	–	–	–	–
Pig	**Baseline emissions***	**lagoon**	–	–	–	–	–	–
Pig	**Baseline emissions****	**tank**	**14**	**0.002**	**< 0.001**	**0.005**	**a**	**0.01**
				CH_4 g m^{-3} h^{-1}				CH_4 kg m^{-3} y^{-1}
Cattle	Pilot-scale studies		35	0.49	0.38	0.70	a	–
Cattle	Farm-scale studies	tank	6	1.2	0.88	1.5	b	–
Cattle	**Baseline emissions***	**lagoon**	**3**	**0.95**	**0.40**	**1.5**	ab	**8.3**
Cattle	**Baseline emissions****	**tank**	**41**	**0.58**	**0.46**	**0.76**	a	**5.1**
Pig	Pilot-scale studies		16	0.67	0.38	1.1	a	–
Pig	Farm-scale studies	tank	3	0.76	–	–	–	–
Pig	**Baseline emissions***	**lagoon**	**1**	**3.5**	–	–	–	**31**
Pig	**Baseline emissions****	**tank**	**19**	**0.68**	**0.41**	**1.1**	a	**6.0**
				CO_2 g m^{-2} h^{-1}				CO_2 kg m^{-2} y^{-1}
Cattle	Pilot-scale studies		6	7.0	–	–	–	–
Cattle	Farm-scale studies	tank	–	–	–	–	–	–
Cattle	**Baseline emissions***	**lagoon**	**14**	**6.6**	**2.6**	**17**	–	**58**
Cattle	**Baseline emissions****	**tank**	**8**	**8.0**	–	–	–	**70**
Pig	Pilot-scale studies		4	8.8	–	–	–	–
Pig	Farm-scale studies	tank	1	5.7	–	–	–	–
Pig	**Baseline emissions***	**lagoon**	**3**	**0.30**	–	–	–	**2.7**
Pig	**Baseline emissions****	**tank**	**5**	**8.0**	–	–	–	**70**

Cells denoted with "-": value is not available.
* Baseline emissions lagoon are entirely based on values from farm-scale studies lagoon.
** Based on the average from studies at farm-scale tank and pilot-scale.

3.2. Baseline emissions

3.2.1. Emissions on an area or volume basis

Table 8 shows emissions on an area or volume basis from cattle and pig slurry stored uncovered in tanks at farm-scale and at pilot-scale without slurry treatment and in lagoons with solid-liquid separation and biological treatment for NH_3, N_2O, CH_4 and CO_2. Average NH_3 emissions from farm-scale studies conducted at lagoons are higher than those from tanks for cattle slurry but lower for pig slurry. Pilot-scale studies exhibit similar emissions as farm-scale studies conducted at tanks, but they differ when compared to measurements from lagoons. The range between the lower and upper 95% confidence bounds is relatively small for cattle slurry but large for pig slurry with the greatest range for farm-scale studies from tanks (0.13 to 0.37 g NH_3 m^{-2} h^{-1}). The baseline emission for lagoons is 0.12 g NH_3 m^{-2} h^{-1} and 0.15 g NH_3 m^{-2} h^{-1} for cattle and pig slurry, and for tanks 0.08 g NH_3 m^{-2} h^{-1} and 0.24 g NH_3 m^{-2} h^{-1} for cattle and pig slurry, respectively. Baseline emissions given as a yearly average emitted amount for lagoons are 1.1 kg NH_3 m^{-2} y^{-1} and 1.3 kg NH_3 m^{-2} y^{-1} for cattle and pig slurry, and for tanks 0.67 kg NH_3 m^{-2} y^{-1} and 2.1 kg NH_3 m^{-2} y^{-1} for cattle and pig slurry, respectively. The differences between baseline emissions for cattle slurry and pig slurry, and the difference between lagoons and tanks are both statistically significant ($p < 0.05$).

Values for N_2O emissions mostly originate from pilot-scale studies. The data from the three studies which exhibit high values mentioned in section 3.1.2 were excluded for the calculation of baseline emissions. The N_2O losses shown in Table 8 are very low and often close to the limit of detection. Negative fluxes are reported e.g. in VanderZaag et al. (2009) or values lower than the limit of detection in Misselbrook et al. (2016). Pig slurry exhibits a large range between the lower and upper 95% confidence bounds (< 0.001-0.005 g N_2O m^{-2} h^{-1}). Baseline emissions are 0.002 g N_2O m^{-2} h^{-1} for cattle and pig slurry stored in tanks. Storage in lagoons for cattle slurry is 0.0003 g N_2O m^{-2} h^{-1} while for pig slurry no baseline value is available. Statistically significant differences were not found for N_2O.

Farm-scale studies exhibit higher CH_4 emissions than pilot-scale studies (Table 8). For both study types, pig slurry has a higher emission level as compared to cattle slurry. The baseline emission values for lagoons are 0.95 g CH_4 m^{-3} h^{-1} (cattle slurry) and 3.5 g CH_4 m^{-3} h^{-1} (pig slurry), and for tanks 0.58 g CH_4 m^{-3} h^{-1} (cattle slurry) and 0.68 g CH_4 m^{-3} h^{-1} (pig slurry), respectively. The baseline emission for lagoon storage of pig slurry is based on one record only. But its distinctly higher emission level as compared to the baseline for tank storage and relative to the baseline emissions of cattle slurry stored in lagoons and tanks is confirmed by the area based CH_4 emissions where the data basis is much larger and statistically significant differences ($p < 0.05$) were found (Supplementary data 4).

For CO_2, the number of observations is relatively small. Some studies exhibit high values for cattle slurry which are greater than 20 g CO_2 m^{-2} h^{-1} (Leytem et al., 2011; Minato et al., 2013; Misselbrook

Table 9
Flow based baseline emissions for tanks from untreated cattle and pig slurry stored uncovered for NH_3 given in percent of total ammoniacal nitrogen (TAN), N_2O in percent of nitrogen (N), CH_4 and CO_2 in percent of volatile solids (VS). The average (Avg) and the lower and upper 95% confidence bounds (l95, u95) are shown. The numbers are mainly based on pilot-scale studies. Cells denoted with "-": value is not available; #: values denoted with different letters are significantly different ($p < 0.05$). Detailed information is provided in Supplementary data 4.

	n	Avg	l95	u95	#
		NH_3% TAN			
Cattle	31	16%	14%	19%	a
Pig	17	15%	9.2%	23%	a
		N_2O% N			
Cattle	16	0.13%	0.08%	0.18%	a
Pig	8	0.10%	0.01%	0.18%	a
		CH_4% VS			
Cattle	27	2.9%	2.3%	3.7%	a
Pig	14	4.7%	2.1%	10%	a
		CO_2% VS			
Cattle	4	11%	–	–	–
Pig	3	9.2%	–	–	–

Cells denoted with "-": value is not available.

et al., 2016). The baseline emissions for lagoon storage are 6.6 g CO_2 m^{-2} h^{-1} and 0.30 g CO_2 m^{-2} h^{-1} for cattle and pig slurry, respectively, and for tank storage, 8.0 g CO_2 m^{-2} h^{-1} for both slurry types. Data on H_2S emission are sparse and a calculation of baseline emissions is only feasible for lagoon storage which are 0.04 g H_2S m^{-2} h^{-1} for cattle slurry and 0.01 g H_2S m^{-2} h^{-1} for pig slurry (Supplementary data 4).

3.2.2. Flow-based emissions

Flow-based emissions, i.e. emissions given in percent of TAN, N or VS present in the store are shown for NH_3, N_2O, CH_4 and CO_2 in Table 9. Almost all data originate from pilot-scale studies (Supplementary data 4) which can be used for baseline emissions for tanks but not for lagoons. Baseline emission values for NH_3 are 16% of TAN for cattle slurry and 15% of TAN for pig slurry, respectively. N_2O emissions are 0.13% of N for cattle slurry and 0.10% of N for pig slurry. Baseline emissions for CH_4 are 2.9% of VS for cattle slurry and 4.7% of VS for pig slurry. Emissions for CO_2 reach 11% of VS and 9.2% of VS for cattle and pig slurry, respectively, but the data basis is limited. The ranges between the lower and upper 95% confidence bounds are large in most cases and are partially skewed to high values, especially for N_2O and CH_4 from pig slurry. There were no statistically significant differences.

3.3. Emission changes due to slurry treatments

Acidification clearly reduces NH_3 emissions by ca. 70% during storage compared to untreated cattle and pig slurry (Table 10). The effect is even higher for CH_4 (61%–96%) but lower for CO_2. For NH_3 and CH_4, the differences are statistically significant ($p < 0.05$). An emission reduction also occurs for digested slurries and slurries after solid-liquid separation combined with acidification for all gases except for N_2O (Supplementary data 6). In contrast, the emissions are enhanced for N_2O emissions compared to untreated cattle and pig slurry although limited data are available and the differences not statistically significant. Data on H_2S emissions are sparse. Fangueiro et al. (2015) state in their review that H_2S emissions were either unaffected or decreased following acidification.

The number of studies on emission changes due to anaerobic digestion is limited. Where more than one observation is available, both an increase and a decrease in emissions occur for storage after anaerobic digestion as compared to untreated slurry (Supplementary data 6). NH_3 and N_2O exhibit on average greater emissions from anaerobically digested slurry. Most studies comparing anaerobically digested and untreated slurry exhibit lower emissions of CH_4 for the former. An emission increase is observed for N_2O and CO_2 for pig slurry, although this is based on only one observation for both gases. Statistically significant differences do not occur for anaerobic digestion.

Average NH_3 emissions during storage from the liquid fraction are significantly ($p < 0.05$) higher as compared to untreated cattle slurry (Table 10). But for pig slurry, only a slight effect of solid-liquid separation on NH_3 release can be observed which is statistically insignificant. CH_4 and CO_2 exhibit lower emissions from the liquid fraction as compared to untreated slurry with a statistically significant difference for CH_4. A statistically significant reduction ($p < 0.05$) in N_2O emissions occurs for cattle slurry. But the release of N_2O is greater for pig slurry compared to untreated slurry where the difference is statistically not significant.

Five studies examined the effect of slurry dilution with water and found an average reduction of all investigated gases in the range of approximately 30–50%. Statistically significant effects occurred for cattle slurry for NH_3 and N_2O. Maximum abatement effects of 88% and 86% were found for N_2O and CH_4, respectively (Supplementary data 6).

3.4. Emission changes due to covering of slurry stores

The average NH_3 emission percent reduction due to covers ranges between approximately 50% up to ca. 90% for most cover types (Table 11). However, the variability of values is large. Minimum values can be around 15% and maximums higher than 95% (Supplementary data 7). The emission mitigation does not systematically differ between cattle and pig slurry on a percentage basis. Emission reductions lie in a similar range for structural covers, impermeable floating covers, permeable floating covers and the other cover materials. The differences are statistically significant ($p < 0.05$) for the following covers and

Table 10
Percentage emission change (i.e. % change of emissions on an area or volume basis) during storage due to acidification, anaerobic digestion, solid-liquid separation and dilution of cattle and pig slurry for NH_3, N_2O, CH_4 and CO_2 relative to untreated slurry. Positive figures indicate a decline, negative numbers an increase in emissions. n: number of records, Avg: average emission change; Std: standard deviation; cells denoted with "-": value is not available. Detailed information is provided in Supplementary data 6.

		NH_3			N_2O			CH_4			CO_2		
		n	Avg	Std	n	Avg	Std	n	Avg	Std	n	Avg	Std
Acidification	Cattle	5	71%*	17%	1	−4%	–	5	61%*	36%	5	7%	23%
	Pig	3	77%*	22%	1	−39%	–	3	96%*	3%	1	67%	–
Anaerobic digestion	Cattle	3	−59%	64%	3	−16%	29%	5	−2%	129%	1	53%	–
	Pig	1	45%	–	1	−363%	–	1	99%	–	1	−22%	–
Solid-liquid separation	Cattle	12	−23%*	21%	6	43%*	36%	10	32%*	27%	7	18%	24%
	Pig	7	−1%	18%	1	−258%	–	7	39%*	39%	5	13%	12%
Dilution	Cattle	5	48%*	29%	5	57%*	38%	5	39%	33%	–	–	–
	Pig	–	–	–	–	–	–	2	47%	15%	2	30%	11%

Cells denoted with "-": value is not available.

* Numbers with an asterisk indicate a statistically significant difference ($p < 0.05$) between the treated and the untreated slurry.

Table 11
Percentage emission change (i.e. % change of emissions on an area or volume basis) from storage of cattle and pig slurry due to different types of covers relative to uncovered storage for NH_3, N_2O, CH_4, CO_2 and H_2S. Positive figures indicate a decline, negative numbers an increase in emissions. n: number of records, Avg: average emission change; Std: standard deviation; cells denoted with "-": value is not available. Detailed information is provided in Supplementary data 7.

		Slurry type	NH_3			N_2O			CH_4			CO_2			H_2S		
			n	Avg	Std	n	Avg	Std	n	Avg	Std	n	Avg	Std	n	Avg	Std
Impermeable structural covers	Lid (wood or concrete)	Cattle	6	73%*	29%	2	−4%	23%	2	15%	2%	–	–	–	–	–	–
		Pig	7	64%*	35%	4	31%	56%	4	45%*	17%	–	–	–	–	–	–
	Tent covering	Cattle	2	77%	9%	–	–	–	–	–	–	–	–	–	–	–	–
		Pig	2	89%	7%	–	–	–	–	–	–	–	–	–	–	–	–
Impermeable synthetic floating covers	Plastic film	Cattle	4	66%*	22%	–	–	–	–	–	–	–	–	–	–	–	–
		Pig	6	88%*	18%	2	100%	0%	2	62%	54%	–	–	–	–	–	–
Permeable synthetic floating covers	Plastic fabrics	Cattle	1	89%	–	1	68%	–	1	−2%	–	1	15%	–	–	–	–
		Pig	5	39%*	15%	–	–	–	3	−17%	18%	–	–	–	4	50%*	20%
	Expanded clay	Cattle	4	59%	39%	–	–	–	2	11%	7%	2	0.1%	1%	–	–	–
		Pig	12	74%*	20%	1	−8%	–	6	8%	17%	5	29%*	8%	–	–	–
	Expanded polystyrene	Cattle	2	79%	2%	–	–	–	–	–	–	–	–	–	–	–	–
		Pig	4	64%*	32%	–	–	–	2	−26%	41%	2	26%	35%	–	–	–
	Plastic tiles	Cattle	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
		P	2	88%	11%	1	−7%	–	1	25%	–	–	–	–	–	–	–
Permeable natural floating covers	Peat	Cattle	2	90%	13%	–	–	–	–	–	–	–	–	–	–	–	–
		Pig	3	59%	31%	–	–	–	1	−33%	–	1	−31%	–	–	–	–
	Straw cover	Cattle	8	71%*	19%	2	−79%	30%	4	3%	30%	4	−6%	10%	–	–	–
		Pig	8	73%*	22%	–	–	–	7	0.2%	36%	2	13%	9%	–	–	–
	Other organic material#	Cattle	4	51%*	32%	–	–	–	4	−13%	37%	4	−46%	71%	–	–	–
		Pig	4	45%	44%	–	–	–	4	−9%	37%	4	20%	17%	–	–	–
	Vegetable oil	Cattle	4	71%*	16%	–	–	–	2	39%	6%	2	27%	9%	–	–	–
		Pig	4	94%*	10%	–	–	–	2	11%	2%	–	–	–	–	–	–

Cells denoted with "-": value is not available; # materials like maize stalks or wood chips; cells denoted with "-": value is not available.
* Numbers with an asterisk indicate a statistically significant difference ($p < 0.05$) between storage with a cover and uncovered storage.

both slurry types: lid, plastic film, straw cover, vegetable oil; other organic materials for cattle slurry; plastic fabrics, expanded clay, expanded polystyrene for pig slurry.

For N_2O, an increase in emissions is observed in many cases. But reduced emissions occur as well (Supplementary data 7). However, the number of records providing emission changes from slurry storage due to store covers is sparse and the effects are statistically insignificant. CH_4 emissions being lower by approximately 10% to 60% occur for impermeable covers (lid and plastic film), plastic tiles and vegetable oil compared to uncovered storage (Table 11). For plastic fabrics, expanded polystyrene and peat, the emissions are higher by 2% to 33%. The other cover types (expanded clay, straw and organic materials such as corn stalks or wood chips) show both increases and reductions in CH_4 emission (Supplementary data 7). On average, CH_4 emissions from slurry stores covered with permeable materials moderately differ in emission levels as compared to uncovered storage. The differences in CH_4 emissions are statistically not significant ($p < 0.05$) except for pig slurry covered with a lid. Stores covered with plastic fabrics, expanded clay and expanded polystyrene emit less CO_2 while higher emissions are observed for peat and straw covers than for the uncovered controls, but the differences are statistically not significant. Plastic fabrics induce a significant ($p < 0.05$) emission reduction for pig slurry by 50% for H_2S. Data on both CO_2 and H_2S emissions are sparse.

4. Discussion

4.1. Variability in emissions

The high variability of emission levels as shown by descriptive statistics (Table 7) may be due to different meteorological conditions, disturbance of the slurry surface induced by operations at the stores and slurry characteristics. The enhanced variability in laboratory-scale studies compared to the other study types is striking. In laboratory-scale studies, the environment is expected to be largely uniform since the experiments were mostly conducted in a temperature-controlled room with ambient temperatures lying in a narrow range and the slurry being undisturbed. As most of the laboratory-scale studies aimed at a comparison of different techniques or systems, the representativeness of the resulting emission rates for real-world conditions was not the primary focus and discrepancies between different approaches are very likely present. A thorough evaluation of potential biases of the laboratory studies is not possible due to missing information on the measuring systems and is beyond the scope of this paper (see Liu et al. (2020) and the related discussion). Also, for other study types, the occurrence of methodological biases cannot be ruled out which may lead to implausible results. Detection of striking values might be hampered due to the multitude in units used in the papers. Therefore, a standardization as used here and providing guide values are important issues. For this, a favorable option is the unit g m^{-2} h^{-1} or g m^{-3} h^{-1} of a molecule. It is equally suitable to illustrate an emission pattern within one day, also in combination with important influencing factors such as temperature or wind speed which can change over short time periods and to compare them with e.g. average emissions over one year. If a yearly amount of a gas release is required, data can be obtained from Table 8. Alternatively, the unit mol m^{-2} h^{-1} could be used to facilitate the comparability between different molecules, even if to date, it is generally not used in the context of emission inventories.

4.2. Important factors influencing emissions

The relevance of important influencing factors on emissions from slurry stores is discussed in this section in order to support interpretation and understanding of the data used to determine baseline emissions and emission changes due to slurry treatment and coverage of slurry stores. It should be noted that a part of these influencing factors could not be included in the data processing such as the weighting or the statistical analysis of emission data due to insufficient information in the records. This data limitation applied for operations at stores, the meteorological parameters rain and wind speed and the natural crust.

4.2.1. Type of slurry

Records from the same study where both cattle and pig slurry have been investigated using the same approach were compared. Eight studies (De Bode, 1991; Sommer et al., 1993; Husted, 1994; Kaharabata et al., 1998; Balsari et al., 2007; Dinuccio et al., 2008; Mosquera et al., 2010; Misselbrook et al., 2016; Baral et al., 2018) with a total of 14, 2, 8 and 3 pairs of records on NH_3, N_2O, CH_4 and CO_2 emissions, respectively, were available. For NH_3, 85% of data pairs, exhibited higher emissions for pig slurry than for cattle slurry. Similar for N_2O, CH_4 and CO_2, pig slurry exceeds emissions of cattle slurry in most cases. These findings agree with the data reported in Table 8 (except for CO_2) and with data previously published by Sommer et al. (2006) and VanderZaag et al. (2015).

4.2.2. Operations at stores

Operations at the storage tank, such as agitation, filling and removing of slurry are necessarily related to real-world storage systems. Their effects are usually reflected in farm-scale measurements using non-intrusive methods but rarely included in pilot studies or farm-scale studies using chamber systems. A series of studies specifically investigated such processes (see Supplementary data 10). They showed consistent results and provided evidence that disturbance of the manure surface due to slurry agitation, filling and discharging of the stores induces large episodic emissions for NH_3, CH_4, CO_2 but not for N_2O. While emissions of CH_4 and CO_2 rapidly decline after cessation of the operations and can even drop to levels below the previously undisturbed stores, increased emission levels persist for NH_3. Due to the relatively short time duration of agitation over the year and the subsequent drop below average levels for CH_4, this operation per se does not substantially contribute to annual NH_3 and GHG emissions from slurry (VanderZaag et al., 2009). A more detailed overview on emissions during and following operations at stores is given in the Supplementary data 10.

4.2.3. Meteorological conditions

Increasing air temperature and wind speed enhance the emissions since they directly affect diffusion and convection of gases near the emitting surfaces (Sommer et al., 2013). The relationship between the temperature as represented by the season of measurements and the emission level could be demonstrated in the present study (Supplementary data 5). It must be considered however, that the air temperature is a simplistic surrogate for the slurry temperature which is a determinant factor for GHG emissions. Rennie et al. (2018) demonstrated that slurry store design and operations (i.e. filling level, agitation) influence the slurry temperature and the emission level of gases such as CH_4. In 25 studies, slurry temperatures during different seasons are available. Slurry temperatures increase as expected in the order cold < temperate < warm for 94% of the cases. The effects of temperature and wind speed are not discussed further in the present study because this topic has been previously covered by e.g. Ni (1999) or Sommer et al. (2006). In contrast, emission changes related to the influence of rain events and thawing of the slurry surface are summarized here since they have been less frequently addressed in the literature. Petersen et al. (2013) found lower NH_3 emission from uncovered storage of pig slurry with precipitation than from the treatment without rain although the differences are not statistically significant. It was shown that ammonia emissions can decline towards zero during rain events after slurry spreading (Hafner et al., 2019) due to sorption of NH_3 onto wet surfaces. Moreover, the TAN-concentration at the emitting surface may decrease with precipitation due to dilution or transport of TAN from a crusted slurry surface into the bulk liquid. Overall, it can thus be assumed that NH_3 emissions from slurry storage during rain events are low. In contrast, an increase in emissions of CH_4 has been observed. Balde et al. (2016b) reported average emissions of 1.8 g CH_4 m^{-2} h^{-1} for digested slurry while peak emissions during rain events reached 10 g CH_4 m^{-2} h^{-1}. This was likely due to bursting of bubbles at or near the surface. Elevated emissions were also observed by Balde et al. (2016a) from storage of the liquid fraction of cattle slurry which confirms earlier findings from Kaharabata et al. (1998) and Minato et al. (2013) on slurry stored in open tanks or lagoons. Kaharabata et al. (1998) suggested that the emission increase is due to more disturbance at the slurry surface induced by rain and thus enhancement of the CH_4 exchange through the liquid surface area and of incidental outburst of gas bubbles (ebullition). Petersen et al. (2013) found a drop of N_2O emissions to zero as a result of rewetting of the crust after rainfall inducing a shift towards anaerobic conditions. Grant and Boehm (2015) did not find a relationship between H_2S emissions and rain events.

VanderZaag et al. (2011) observed important bubble flux events in late winter/early spring that coincided with surface thawing which were probably due to a release of previously produced CH_4 that was trapped under the frozen slurry surface. In the study of VanderZaag et al. (2010a) which encompassed winter and spring, N_2O release was only recorded during spring thaw. A moderate increase in CH_4 emissions was observed during the same period at slurry temperatures above 0 °C while NH_3 and CO_2 flows were unaffected by spring thaw. Elevated CH_4 emissions due to thawing of the manure store were also reported by Leytem et al. (2017).

4.2.4. Natural crust

There is agreement that crusting impacts the gas release in many ways: enhanced resistance to mass transfer (Olesen and Sommer, 1993), oxidation of NH_3 (Nielsen et al., 2010) and CH_4 (Petersen et al., 2005) and formation of N_2O related to nitrification and denitrification occurring in liquid–air interfaces near air-filled pores present in crusts (Petersen and Miller, 2006). Several studies investigated the effect of a natural crust on the emission level (Sommer et al., 1993; Misselbrook et al., 2005; Aguerre et al., 2012; Wood et al., 2012). All studies showed that a natural crust provided an efficient barrier leading to an emission reduction for NH_3. Baldé et al. (2018) confirmed these findings by measurements conducted under farm conditions at tanks and earthen basins containing slurries with differing ability to form natural crusts (i.e. raw cattle slurry, the liquid fraction of cattle slurry produced by solid-liquid separation, digested slurry with and without solid-liquid separation). They confirmed that slurry stored with a thick surface had lower NH_3 losses. Grant and Boehm (2015) found that crusting of a lagoon surface containing dairy cow slurry reduced NH_3 but not H_2S emissions. Nielsen et al. (2010) showed that NH_3-oxidizing bacteria may contribute to a significant reduction of NH_3 emissions if a natural crust is present on a slurry store. Grant and Boehm (2018) found emissions from a tank containing pig slurry to be greater by 10% when the surface was covered with a crust than without crusting (difference not statistically significant). They explained their findings by the higher TAN content of the crusted slurry surface as compared to a non-crusted one. Sommer et al. (2000) and Husted (1994) found higher emissions of CH_4 from slurry without than from slurry with a natural surface crust. Wood et al. (2012) investigated the emissions from dairy slurry with varying DM contents and thus natural crusts with different thicknesses and coverage of the storage surfaces. They were not able to relate the CH_4 fluxes to the presence of a natural crust. N_2O production was found to be enhanced after build-up of a natural crust (VanderZaag et al., 2009).

In the literature (e.g. Vanderzaag et al., 2015), a natural crust is often classified as abatement measure for NH_3 similar as slurry store covers. However, crust formation can only be controlled to a limited extent. Crusts are of variable thickness, coverage of the store and durability. Their effectiveness for emission abatement has therefore be considered as inconsistent (Vanderzaag et al., 2015). Crusts only develop for slurry types with a high content of fibrous material (Bittman et al., 2014). This applies mainly for cattle slurry (Smith et al., 2007) and less for pig slurry (Sommer et al., 2006). Crusting is likely to occur at a slurry DM content of more than 20 g L^{-1} (Sommer et al., 2006; Wood et al., 2012) which mostly does not apply for slurry stored in lagoons (Table 6). Consequently, they have much less ability to form a natural crust as shown by e.g. Balde et al. (2018).

We therefore did not consider crusting as an emission mitigation technique equivalent to slurry store covers but rather as a parameter influencing emissions from stored slurry and thus excluded it from the analysis of emission changes due to covering of stores. But we stress that if a natural crust is present, it is likely to significantly contribute to an emission reduction and, therefore, should be preserved by e.g. reducing slurry agitation and addition of manure below the surface.

The limited information in the data impedes our ability to clearly distinguish between crusted and non-crusted stores. This may be relevant for (i) the calculation of baseline emissions and (ii) emissions changes due to slurry treatments and covering of stores: (i) baseline emissions determined here may include stores with variable occurrence of a natural crust. For lagoons, information on crusting was available for 45% (cattle slurry) and 19% (pig slurry) of the records, respectively. Among these, 62% of the lagoons containing cattle slurry were fully or partly covered by a crust during the emission measurements. For pig slurry, this applies for 21% only. Among records used for the determination of baseline emissions for tanks, 78% and 50% included information regarding crusting for cattle and pig slurry, respectively. Of these, 83% (cattle slurry) and 48% (pig slurry) had a fully or partly crusted surface. This complies with findings that crusting occurs less on lagoon surfaces and stores containing pig slurry. The proportion of crust occurrence for cattle slurry stored in tanks is in line with earlier findings (Smith et al., 2007). We thus suggest that the baseline emissions determined here are based on studies which appropriately reflect the range of store surface crusts occurring at farms. (ii) A natural crust may occur in combination with a storage cover and thereby be enhanced (Chadwick et al., 2011) since the slurry surface is less exposed to wind turbulence. In experiments comparing uncovered and covered storage, it is thus difficult to stringently distinguish between the effect of covering and of crusting. Moreover, this information is not always available: only 60% of the records used to determine the emission change due to covering included information on crusting. From these, about half had crusted surfaces and the other half not. This might partly explain the variability of emission changes due to covering found here. These considerations should be taken into account for the discussion in the Sections 4.4 and 4.6.

4.3. Study types to be included for baseline emissions

Data should only be included for the calculation of baseline emissions if they can be considered as representative or typical for gas flows occurring at farm conditions. In principle, this applies for farm-scale studies. Pilot-scale studies imply some aspects of farm-scale studies due to measurements conducted in outdoor facilities and a slurry volume in the order of several cubic meters. But there are concerns extrapolating data from pilot-scale studies to real-world systems. VanderZaag et al. (2009, 2010a, 2010b) who performed pilot-scale studies state that although measured fluxes were reported, emission trends and treatment differences or temporal trends were the focus of their analysis. Moreover, almost all pilot-scale studies are based on flux chambers. VanderZaag et al. (2010b) argued that steady-state chambers alter the enclosed environment and concluded that absolute fluxes measured might deviate from emissions that would occur without chambers. Nevertheless, several studies conducted in pilot-scale facilities similar to that of VanderZaag et al. (2009, 2010a, 2010b) quantified emissions of NH_3 and GHGs and derived emission factors for slurry storage (e.g. Amon et al., 2006, 2007; Rodhe et al., 2012). Petersen et al. (2009) presented a pilot-scale facility and suggested to use the obtained results for better documentation of emission data for GHG and ammonia inventories. Pilot-scale studies have occasionally been conducted with simulation of real-world conditions by including mixing of the slurry or filling of tanks during the experiment (VanderZaag et al., 2009; Rodhe et al., 2012).

Emission peaks for CH_4 were observed in several studies (VanderZaag et al., 2011; Balde et al., 2016a) due to ebullition. They may remain unrecorded (Rodhe et al., 2012) unless the gas measurements are continuous with a high temporal resolution. This shortcoming may apply for pilot-scale studies where e.g. a flow chamber is used which is moved between several experimental tanks (e.g. Amon et al., 2006). Intermittent gas sampling can hamper measurements at a farm-scale as well. Grant et al. (2015) assumed differences in emission levels between two locations due to under-sampling of ebullition events given the short measurement periods. Sampling large storage areas using chambers might be hampered if the sampled surface areas are not representative for the entire store. Balde et al. (2016b) found average emissions of CH_4 measured at an earthen storage containing liquid digestate with a floating chamber which were about four-fold greater than measured at the same time with a non-intrusive bLS technique. The authors explained this by the limited area covered by the chamber and by disturbances induced by the chamber causing bubble formation and bursting thereby increasing emissions.

To summarize, it can be hypothesized that farm-scale measurements using non-intrusive methods are a preferential option. Still, data from such studies are limited at present time. Therefore, inclusion of records from pilot-scale and farm-scale studies appears to be the best opportunity for the determination of baseline emissions. This approach provides a larger data basis as if only farm-scale studies were included. Moreover, Table 8 shows that emissions from pilot-scale studies comply with farm-scale studies tank for NH_3 but less for CH_4. On the other hand, we excluded laboratory-scale studies for the determination of baseline emissions. They are mostly not designed for generating emission rates. Their experimental conditions strongly deviate from an environment that occurs under practical conditions. The enhanced variability found in emissions level from laboratory-scale studies (section 3.1.2) points to severe methodological shortcomings which might bias baseline values.

4.4. Baseline emissions

4.4.1. Emissions on an area or volume basis

NH_3 emissions from pig slurry are higher as expected due to its higher TAN content and its lower ability to form a natural crust compared to cattle slurry. Sommer et al. (2006) and VanderZaag et al. (2015) suggested lower emissions on an area basis from pig slurry stored in lagoons than from storage in tanks. This complies with the results of this study (Table 8). Lagoons are the prevailing system for slurry storage in the US (Sorensen et al., 2013). They usually have a greater surface area than tanks which would imply more exposure to the ambient air turbulence suggesting a higher emission potential. Slurries from lagoons have on average a lower dry matter and TAN content as compared to tanks. This might be due to a stronger dilution with water: e.g. five out of six lagoons investigated by Leytem et al. (2017) collected parlor wash water and not slurry from a pit of a livestock housing. We assume that solid-liquid-separation was applied at the farms studied which have lagoons although this was not always clearly defined in the papers. This is supported by the low contents in DM and TAN in slurry from lagoons as shown in Table 6. The lower solids content would enhance the emission potential due to less ability for formation of a natural crust at the slurry surface (Wood et al., 2012). But the lower TAN content induces the opposite effect on NH_3 emissions (Sommer et al., 2006). The overall impact of these effects combined on the emission level is difficult to assess. The present data suggest higher emissions from lagoons than from tanks containing cattle slurry but the opposite for pig slurry.

The baseline emissions (lagoons: 0.12 g NH_3 m^{-2} h^{-1} and 0.15 g NH_3 m^{-2} h^{-1} for cattle and pig slurry, tanks: 0.08 g NH_3 m^{-2} h^{-1} and 0.24 g NH_3 m^{-2} h^{-1} for cattle and pig slurry, respectively), are mostly lower than numbers given by VanderZaag et al. (2015), Sommer et al. (2006) and Bittman et al. (2014). VanderZaag et al. (2015) suggested emissions for crusted and non-crusted cattle slurry of 0.11 and 0.19 g NH_3 m^{-2} h^{-1}, respectively, from tanks or lagoons. For pig slurry stored in a lagoon, they give 0.12 g NH_3 m^{-2} h^{-1}, and stored in a tank, 0.40 g NH_3 m^{-2} h^{-1}. Sommer et al. (2006) provided similar values. Bittman et al. (2014) gave baseline emissions between 0.19 and 0.40 g NH_3 m^{-2} h^{-1}. They attributed the lower value to slurry which is frozen in the store for several months, and the higher value applies to warm countries. For N_2O, most studies exhibit emissions clearly below 0.01 g N_2O m^{-2} h^{-1} (Supplementary data 8). In contrast, three papers reach values from 0.02 to 0.06 g N_2O m^{-2} h^{-1} and N_2O losses ranging between 25% and 160% of the NH_3 emissions determined concomitantly (Clemens et al., 2006; Amon et al., 2007; Leytem et al., 2011). Unless at very low levels of NH_3 emissions, flows of both NH_3 and N_2O in the same order of magnitude do not occur in other records and have not been reported in the livestock sector (e.g. EEA, 2016). Therefore, the data from the three studies were excluded for the calculation of baseline emissions. If they were kept, the baseline emissions for N_2O would be higher by a factor of two and three for cattle and pig slurry stored in tanks, respectively. Chadwick et al. (2011) stated in their review that N_2O emission from slurry stores without a surface cover are negligible which supports the baseline emissions shown in Table 8.

For CH_4, higher emissions occur for farm-scale studies than for pilot-scale studies (Table 8; statistically significant differences for emissions on an area basis; $p < 0.05$; Supplementary data 4). This could be due to the temperature dependency of methanogenesis (Elsgaard et al., 2016). Pilot-scale studies exhibit lower slurry volumes as compared to farm-scale stores which suggests faster cooling of the slurry and therefore a lower methane conversion rate. Another reason could be the batch-filling of vessels used for pilot-scale studies which differs from continuous filling and incomplete removal of slurry at farm-scale stores. Under such conditions, aged slurry may act as inoculum which was shown to enhance emissions of CH_4 (Wood et al., 2014). Overall, the lower emission level for CH_4 measured at pilot-scale included for the determination of baseline emissions tank could lead to an underestimation thereof.

The review of Owen and Silver (2015) reported CH_4 emission data from lagoons and tanks of dairy systems being 2.3 and 2.7 g CH_4 m^{-2} h^{-1}, respectively. This is higher than data from farm-scale studies reported here which are 1.2 and 1.3 g CH_4 m^{-2} h^{-1} for cattle slurry stored in lagoons and tanks, respectively (Supplementary data 4). However, the data basis of Owen and Silver (2015) is smaller and measurements carried out in the warm season tend to be overrepresented. The higher CH_4 emissions from pig slurry as compared to cattle slurry are expected due to the higher methane production potential of pig slurry (Triolo et al., 2011). Both cattle and pig slurry exhibit lower losses from tanks than from lagoons. Moreover, lagoon storage produces a solid fraction which includes a large proportion of the slurry VS generating additional emissions. According to VanderZaag et al. (2010b), a CO_2-C:CH_4-C ratio of 50:50 is expected from stores. Looking at records which include emission data of both CH_4 and CO_2, a large variation occurs. The average CO_2-C:CH_4-C ratio is approximately 65:35 which also differs from the CH_4 to CO_2 relationship expected from anaerobic digestion of livestock slurries (ca. 55%–70% CH_4 content of dry biogas; Triolo et al., 2011). This could be due to a tendency for greater CO_2-C:CH_4-C ratios in pilot-scale studies which increases the average CO_2-C:CH_4-C ratio of all records included. The greater ratios were also linked to studies with low CH_4 fluxes. This is likely because pilot-scale studies had less ability to provide appropriate conditions for CH_4 production as mentioned above. On the other hand, CO_2 seems to be emitted more consistently in all studies.

As the aim of all studies considered for the calculation of baseline values in the present paper was the determination of emission rates we think that the baseline emissions are robust and reflect the current state of knowledge. But the confidence intervals shown in Table 8 may be substantial. This suggests an inherent variability in the systems which can be due to differing conditions regarding meteorological conditions, operations at stores and occurrence of a natural crust. Baseline values must be considered as average numbers. In a specific situation and e.g. for representative regional values, deviations from the presented baseline values can occur. Moreover, methodological biases cannot be ruled out and different experimental approaches might entail systematic differences in results (e.g. possibly CH_4 emissions from pilot-scale studies). Such effects have been observed for experimental data on NH_3 emissions from slurry application (Hafner et al., 2018).

4.4.2. Flow-based emissions

The determined baseline emission values for NH_3 of 16% of TAN and 15% of TAN for cattle and pig slurry, respectively, which are mostly based on pilot-scale studies exhibit similar values as the emission factors of EEA (2016) which give 20% of TAN and 14% of TAN as Tier 2 default values for cattle and pig slurry. Data from farm-scale studies have comparable numbers for storage in tanks for cattle slurry: 16% of TAN (Baldé et al., 2018) and 13% of TAN (McGinn et al., 2008) but lower emissions for pig slurry (Dinuccio et al., 2012) with 2% and 5% of TAN. Flow-based emissions are not available for lagoons. IPCC (2006) and EEA (2016) suggest an N_2O emission factor being zero for slurry storage without a natural crust. For a crusted store, IPCC (2006) and EEA (2016) give EFs of 0.5% of N and 1% of TAN entering the store, respectively. These values are higher than the values determined in this study which are 0.13% of N and 0.10% of N for cattle and pig slurry, respectively (Table 9). The eight highest values for flow-based N_2O emissions originate from records that include slurry stores with a crust which supports the occurrence of N_2O emissions with crusted store surfaces. This complies with Sommer et al. (2000) who suggest that N_2O is produced in drying of natural crusts where aerobic and anaerobic zones exist. Drying enhances convection of liquid upward through the cover, where dissolved ammonium can be oxidized by nitrifying bacteria in an aerobic environment and under such conditions, molecules produced from nitrification can be denitrified. During ammonium oxidation and denitrification, N_2O is released as an intermediate or final product. At limited oxygen availability, formation of N_2O is enhanced.

For CH_4, a direct comparison between the suggested baseline emissions with default emission factors used in models for emission inventories is not possible. A simplified application of the approach of Mangino et al. (2001) and IPCC (2006) using the methane conversion factor (MCF) for slurry in a cool climate with an annual average temperature of 14 °C results in a CH_4 emission of ca. 4.0% of VS and 7.5% of VS for cattle slurry and pig slurry, respectively. These figures are somewhat higher than the baseline emissions suggested (Table 9) which are 2.9% of VS and 4.7% of VS for cattle and pig slurry, respectively. It should be noted that emission values for CH_4 derived from such model approaches can strongly deviate from measured values as shown by several studies (e.g. Kariyapperuma et al., 2018).

For the determination of flow-based emissions, analytical data of the slurry, the flow volume of slurry into storage and its residence time in the store must be known. Determining these three parameters is not straightforward which might explain the high degree of absence regarding flow-based emissions in farm-scale studies. This particularly applies for lagoons where extended slurry residence times, accumulation of solids over long time periods and repeated recycling of liquids used for flushing or recharging pits of livestock housings represent additional challenges. Generally, lagoons have a greater surface to volume ratio and longer slurry residence times as compared to tanks. These two factors will lead to higher flow-based emissions for lagoons as compared to tanks if identical emissions on an area basis are assumed for both storage systems.

For inventory purposes, emission factors could be calculated using the baseline emissions on an area or volume basis and an assumption of the surface or volume of the storage system, the average values for the residence time of the slurry in the store and the slurry contents of TAN, N or VS. These values are specific for different countries and production systems. A further assessment thereof is outside the scope of this paper. The calculation of flow-based emissions (and emission factors) is subject to additional uncertainties as compared to emissions on an area basis due to the requirement of further parameters.

4.5. Emission changes due to slurry treatments

The pH value has a strong effect on gaseous emissions from slurry stores (Sommer et al., 2013). This is appropriately reflected by the data on emission changes due to slurry acidification through addition of inorganic acids shown in Table 10. The variability in the achieved reduction is likely related to the degree of acidification and the different pH values in slurry (Dai and Blanes-Vidal, 2013). The emission reductions found for NH_3 and CH_4 are in line with the review of Fangueiro et al. (2015). Similarly, Petersen et al. (2012) observed significant reduction effects for NH_3 and CH_4 due to acidification. The data point at an increase in N_2O emission but this is based on limited data. Bastami et al. (2016) concluded that self-acidification of slurry induced by addition of substrates rich in carbon may be a promising alternative to slurry acidification using concentrated acids for abatement of CH_4 emissions. Additives other than acids to reduce gaseous emissions or odor nuisance from manure storage have been investigated in some studies (Martinez et al., 2003; Sun et al., 2014; Owusu-Twum et al., 2017). A clear emission reduction due to other additives did not occur. Similarly, Van der Stelt et al. (2007); Wheeler et al. (2011) and Holly and Larson (2017a) found little evidence that manure additives other than acids have a clear influence on the release of ammonia and GHGs. Still, individual investigations have shown an emission reduction potential for certain additives (Bastami et al., 2016).

The number of studies allowing a direct comparison of emissions from storage of untreated slurry and anaerobically digested slurry is limited since biogas plants are mostly fed with manure and off-farm organic feedstock material which hampers a direct comparison with unamended untreated slurry. The increase of NH_3 emissions due to anaerobic digestion complies with studies which include anaerobic digestion with addition of organic feedstock material. Baldé et al. (2018) measured NH_3 emissions from two stores at different farms containing untreated livestock slurry and liquid digestate obtained from livestock slurry and organic feedstock materials under farm conditions. Emissions from the untreated slurry were lower. Koirala et al. (2013) suggested that anaerobic digestion of dairy slurry significantly increased the NH_3 volatilization potential. The most important factor was the enhanced ammonium dissociation. Anaerobic digestion seems to reduce CH_4 emissions during slurry storage. Maldaner et al. (2018) found lower CH_4 losses from the liquid fraction of anaerobically digested slurry amended with organic feedstock material compared to unamended raw slurry from the same farm before the installation of anaerobic digestion. This is likely due to the reduction of the VS load after digestion but also a consequence of solids removal with solid-liquid separation of the major part of the digestate. Furthermore, Maldaner et al. (2018) suggested that VS remaining in the digestate was less degradable which leads to a reduced CH_4 production. VanderZaag et al. (2018) showed the CH_4 emission potential (B_0) from digestate was 35% lower than the B_0 of untreated manure. In contrast, Sommer et al. (2000) and Rodhe et al. (2015) measured higher emissions from anaerobically digested slurry as compared to untreated cattle slurry. They explained this by the presence of a larger and more active microbial community in digested slurry. However, most storage tanks are never completely empty. Residual aged slurry may act as inoculum and enhance the production of CH_4 (Sommer et al., 2007; Ngwabie et al., 2016). Although, the microbial population in aged slurry may be less efficient for methane production as compared to microbes present in anaerobically digested slurry, the higher amount of degradable organic carbon available in untreated slurry might compensate this. It can therefore be hypothesized that untreated slurries as occurring in real-world stores imply a higher potential for CH_4 emissions than anaerobically digested slurry.

Solid-liquid separation reduces the solids content of the slurry and thus the potential to develop a natural crust. This enhances NH_3 emissions during storage which complies with the increasing emissions of cattle slurry due to solid-liquid separation. Pig slurry exhibits a lower ability to form a natural crust which could explain why almost no effect of solid-liquid separation on NH_3 emissions can be observed (Table 10). Baldé et al. (2018) investigated the NH_3 emissions from two stores situated at different dairy farms containing untreated slurry and separated liquids under farm conditions. They measured higher emissions from the liquid fraction than from the untreated slurry and reported similar findings for the separated liquids from digestate derived from livestock slurry and organic feedstock materials. In contrast, Hjorth et al. (2009) found significantly higher NH_3 emissions from raw and digested slurries than from the corresponding liquid fractions. They explained their findings by the higher ammonium and N contents of the unseparated raw and pre-digested slurries compared with the liquid fractions, which increased the potential for NH_3 volatilization. The lower N_2O storage emissions of the liquid fraction for cattle slurry as compared to raw slurry is in line with the conclusions of the review paper published by Chadwick et al. (2011). The reduced emissions of CH_4 due to solid-liquid separation results from the reduction of the total solids content in the slurry which can be considered as a surrogate for the available VS pool. This leads to a lower amount of organic matter which can be degraded to CH_4 and CO_2 (Wood et al., 2012). However, yearly average CH_4 emissions of 1.4 g CH m^{-3} h^{-1} and 2.2 g CH m^{-3} h^{-1} for the first and second year of measurements were reported for dairy slurry from a farm-scale study conducted at a tank (Balde et al., 2016a). These numbers exceed the baseline emission and emissions from farm-scale studies from tanks for cattle slurry given in Table 8. Balde et al. (2016a) explained the elevated emission levels by the high biodegradability of the liquid fraction and the limited crust development. VanderZaag et al. (2018) showed that the speed of CH_4 production was much higher for the separated liquid fraction, compared to untreated slurry. Grant et al. (2015) found CH_4 emissions on an area basis from the liquid fraction of cattle slurry stored in a lagoon which are similar to the baseline emission for cattle slurry (Supplementary data 4). The discrepancy between the emission changes given in Table 10 and the high emissions found in these two studies is difficult to explain.

Dilution of slurry with water changes its DM content. DM of slurry can be considered as an indicator for the N/TAN- and VS-content which influences the potential production of NH_3, N_2O, CH_4 and CO_2, respectively (Wood et al., 2012). However, DM affects the formation of a natural crust as well (section 4.2.4). Overall, dilution leads to a reduction of all investigated gases which complies with the findings of Ni et al. (2010) for NH_3 and CO_2 and of Habetwold et al. (2017) for CH_4. But this conclusion is based on pilot-scale studies where the slurry volume is identical for the diluted and untreated slurry. Under practical conditions, addition of water leads to a higher amount of slurry. If the area of manure stores is thereby increased due to requiring larger storage capacities, the reduction might be overcompensated due to a rise in emitting surface (Ni et al., 2010).

Aeration of slurry is a technique which is used to remove excess N from slurries. It induces nitrification and denitrification that converts TAN in the slurry to nitrite/nitrate with the aim of a complete denitrification to N_2. If the process is not properly controlled aeration can produce substantial amounts of NH_3 and N_2O (Loyon et al., 2007). The effect of aeration was investigated in several laboratory- and pilot-scale studies. Amon et al. (2006) found a strong increase of NH_3 emissions by up to a factor of five. Molodovskaya et al. (2008) reported NH_3 emissions of up to 50% of total slurry N. Losses were increased at greater aeration rates. Many studies found a strong increase in emissions for N_2O with aeration (Beline et al., 1999; Beline and Martinez, 2002; Amon et al., 2006; Loyon et al., 2007). Low emissions for both NH_3 and N_2O were achieved from low flow phased oxic/anoxic treatment (Molodovskaya et al., 2008). A reduction of CH_4 by ca. 50% to almost 100% emissions was observed by Martinez et al. (2003) and Amon et al. (2006) if slurry aeration was applied. Concomitantly, CO_2 emissions were reduced (Martinez et al., 2003).

An increase in emissions of a specific gas during slurry storage due to a treatment technique does not necessarily indicate a conflict related to emission mitigation. Enhanced losses during storage can be reduced by e.g. storage covering and might be overcompensated by reduced emissions during subsequent field application. The overarching goal of manure management is the reduction of gaseous losses between the excretion by livestock and uptake by arable and fodder crops. Therefore, the discussion on effects of slurry treatments on emissions from slurry storage must consider the context of good management practices along the whole manure management chain (Sajeev et al., 2018).

4.6. Emission changes due to coverage of slurry stores

Almost all types of covers induce a substantial emission reduction for NH_3 which complies with the review of VanderZaag et al. (2008). Emission reductions lie in a similar range for all categories of covers and for both cattle and pig slurry. This contrasts to Bittman et al. (2014) who give distinct values for the different cover types and lower values for “Low technology” floating covers such as permeable natural floating covers. VanderZaag et al. (2015) give an emission reduction of 80% for impermeable structural covers and for impermeable synthetic floating covers which is in the range of the values given in Table 11. A larger layer thickness of natural floating covers leads to a higher emission reduction (Guarino et al., 2006; VanderZaag et al., 2009). This is probably due to a more efficient barrier for the gas transport between the slurry and the ambient air (VanderZaag et al., 2009). Other cover types not included in the data analysis according to section 2.4.3 also efficiently reduce NH_3 emissions. Organic materials such as steam-treated wood or biochar were shown to exhibit similar effects as floating covers consisting of straw or peat (Holly and Larson, 2017b). Minerals like perlite or zeolite were also found to be efficient in NH_3 emission reduction (Hörnig et al., 1999; Portejoie et al., 2003).

The increase in emissions observed for N_2O in many cases agrees with the previously published literature (VanderZaag et al., 2008; Chadwick et al., 2011). Petersen et al. (2013) observed lower N_2O flows with a straw cover exposed to precipitation as compared to straw covers where wetting by precipitation was excluded. Sommer et al. (2000) suggested that N_2O is only produced in periods with drying surface layers. Storage covers can influence the formation of a natural crust (Chadwick et al., 2011). A natural surface crust on slurry can provide sites with aerobic conditions where nitrification occurs which produces N_2O (Sommer et al., 2000). Therefore, the variability in emission changes due to slurry storage covering could be driven by differing moisture contents of the manure surfaces and differing formation of a natural crust due to covering. However, the number of records related to emission changes for N_2O due to store covers is sparse and these findings can be uncertain.

The observed increase in CH_4 emissions for plastic fabrics, expanded polystyrene and peat complies with the review of VanderZaag et al. (2008). Straw covers provide additional carbon but might reduce ebullition and increase aerobic microbial activity at the upper storage layer (VanderZaag et al., 2009). This induces contrasting effects on CH_4 net emissions. VanderZaag et al. (2009) suggested that the reduction of CH_4 emissions due to a straw cover is related to areas in a crust where microbial breakdown of CH_4 might occur. It can be assumed that the enhanced CH_4 consumption overcompensates the increased potential for CH_4 production due to the additional carbon supply with straw. An opposite effect can occur if straw is incorporated into the bulk slurry during storage due to e.g. agitation. Petersen et al. (2013) found that an elevated CH_4 concentration in the gas phase above the slurry surface is required for a significant stimulation of methane oxidation. This would support the preponderant emission reduction found for impermeable covers. For other cover types, the CH_4 concentration above the emitting surface might have been inconsistent in the experiments which could explain the contrasting emission changes for CH_4 due to storage covering. Similar to NH_3, a larger layer thickness of straw covers leads to a higher emission reduction for CH_4 although the differences of the emissions are low (Guarino et al., 2006; VanderZaag et al., 2009).

The contribution of the different gases to the total of GHG emissions is largest for CH_4 with a proportion of ca. 80% (VanderZaag et al., 2009; Petersen et al., 2013). Therefore, the changes of GHG equivalents due to effects of covers is moderate with a slight trend towards lower total GHG emissions.

VanderZaag et al. (2010b) have shown that a permeable synthetic floating cover was more efficient regarding emission reduction of NH_3 and CH_4 when slurry is agitated as compared to undisturbed slurry. VanderZaag et al. (2009) found that agitation increased NH_3 losses from straw covered tanks less than from the uncovered reference. CH_4 emissions from covered and control tanks were similarly changed.

Although found efficient in emission reduction, low cost floating covers such as straw covers are probably not efficient for emission mitigation in practice since they may be destroyed when the slurry surface is disturbed due to strong winds or operations at a store. Therefore, we consider impermeable structural covers or synthetic floating covers as most reliable for emission mitigation.

4.7. Recommendations for further research

The emission data provided in records from different studies range over several orders of magnitude even for the same slurry type, the same type of study and identical seasons of measurement. This may be partly due to varying conditions related to manure management and meteorological conditions occurring during the measurements. In addition, different study designs and measuring methods are likely to contribute to the variability in emissions. An important issue in future research should thus focus to identify and quantify potential experimental biases. This aspect requires the simultaneous use of independent approaches to determine emissions.

Farm-scale studies using non-intrusive methods such as micrometeorological mass balance (Wagner-Riddle et al., 2006) or dispersion modeling (Flesch et al., 2009) are likely to be a preferential option. Such approaches avoid interactions with emitting processes and determined emission rates best reflect the emissions occurring under conditions at farm-scale. They have the ability to cover large area sources (Gao et al., 2008) and can thus integrate the large inhomogeneity of emissions over space and time. For dispersion modeling, the limiting factor is the requirement of a simple topography allowing for representative turbulence measurements. Most of the micrometeorological methods require a minimum wind speed. Many sensors, e.g. for NH_3, have a minimum detection limit which may be higher than gas concentrations occurring under conditions with low emissions (Balde et al., 2019). Consequently, farm-scale studies using non-intrusive methods have a risk to overestimate the true average emissions (Baldé et al., 2018). This risk can be minimized by using recently developed sensors such as DOAS systems (Volten et al., 2012; Bell et al., 2017). Moreover, it would be important to quantify such potential biases by an assessment of gap filling procedures used for missing data due to e.g. non-detection at low concentration levels as done by Voglmeier et al. (2018). For reliable results from farm-scale studies, extended measurement periods are required covering all seasons of a year. Moreover, recent research has demonstrated that the history of the storage may play an important role for CH_4 emissions (Kariyapperuma et al., 2018) pointing at the necessity of measurement campaigns over several years for an adequate determination of representative emission rates. This implies a large effort in labor and costs. In addition, thorough recording of the operations at the storage facilities (agitation, filling, discharging of the stores by using e.g. a webcam, continuous measuring of the slurry volume stored), of crusting at the stores surface (thickness, structure, coverage of the surface), of slurry temperature at several depths, of meteorological conditions as well as slurry sampling and analyses are required. A few studies comply with these requirements (e.g. Baldé et al., 2018; Kariyapperuma et al., 2018). Still, collection of such data can be challenging or even hardly feasible (e.g. representative slurry sampling at large lagoons). Also, operations at stores can largely differ between individual farms and consequently, it is difficult to select an experimental site at farm-scale which is appropriate to generate baseline emissions. Therefore, several measurement campaigns that consider the variety of different conditions occurring at slurry stores are required.

Pilot-scale studies are indispensable for studying principal mechanisms and influencing factors driving emissions or to evaluate the effectiveness of emission mitigation techniques. Facilities allowing for continuous measurements e.g. as presented by Petersen et al. (2009) are probably the best option. Further advantages of pilot-scale studies are the possibility to conduct experiments in replicates and a better control of the experimental conditions. There is also a potential to generate bases for modeling which could be used to complement data from farm-scale studies. Further progress for the quantification of emissions from slurry storage could be achieved by an analysis of individual measurement intervals from several experiments and model construction on this basis as e.g. done for slurry application by Hafner et al. (2019). The measurement intervals should include the relevant information regarding influencing factors. For this, we recommend that the researchers provide the emission data along with parameters as given in the Supplementary data 2. For indistinct parameters such as crusting, we suggest the elaboration of a standardized procedure to achieve a definition which reliably reflects its influence on the emission level.

5. Conclusions

The present article provides a comprehensive overview on published emission data from slurry storage which serves as a basis to determine guide values and baseline emissions for NH_3, GHGs and H_2S. Standardization of the emission data is an important issue in the present study due to the use of a large variety of units in the studies. Accompanying parameters (e.g. data on slurry analyses) were only partly available in the papers and could thus not be used for a more advanced data analysis. However, the season of the experimental period which served as a surrogate for the temperature was provided in most studies. Descriptive statistics of the emission data revealed a large variability for all gases. Data generated during warm, temperate and cold seasons are unevenly distributed over all records. Therefore, the calculation of an average annual value completed with a confidence range based on a weighting of the emission data according to the season and measurements duration was done. The baseline emissions on an area or volume basis determined for cattle and pig slurry stored in lagoons and tanks (Table 8) are mostly lower than existing reference values. NH_3 baseline emissions for tanks related to TAN are 16% of TAN (range: 14%–19% of TAN) and 15% TAN (range: 9.2%–23% of TAN) for cattle slurry and for pig slurry, respectively, and thus similar to emission factors used in emission inventory models. The flow-based baseline emissions for N_2O and CH_4 are lower than current emission factors. Total GHG emissions from slurry stores based on the global warming potential using a 100-year time horizon are dominated by CH_4.

Techniques for slurry treatment exhibit contrasting effects on emission levels during storage. Acidification was found to be efficient in reducing the emissions of NH_3 and CH_4 but less for CO_2 while the release of N_2O was enhanced in few studies. Solid-liquid separation causes higher losses for NH_3 and a reduction in CH_4, N_2O and CO_2 emissions. Anaerobic digestion promoted NH_3 emissions in most studies. In contrast, emission changes during slurry storage were less explicit for CH_4, although there is evidence toward an emission reduction. The effect of anaerobic digestion on N_2O and CO_2 emissions is unclear. It is essential to consider the context of good management practices along the whole manure management chain when the effect of slurry treatments on emissions from slurry storage is assessed.

All storage cover types reduce emissions of NH_3 while the effect is small for CH_4 and CO_2 with a trend toward a reduction. Permeable covers increase emissions of N_2O. Total GHG emissions tend to be lower with coverage of slurry stores. Overall, coverage of slurry is efficient to abate NH_3 emissions involving a minimum risk of pollution swapping.

The present study provides a robust data basis for the determination of baseline emissions except for flow-based baseline emissions for lagoons which could not be calculated. The emission data in the records from different studies may vary over several orders of magnitude even for the same slurry type, the same type of study and identical seasons of measurement. For future research, appropriate study designs are required to generate baseline emissions appropriate to improve emission inventories. For this, farm-scale studies using non-intrusive methods are likely to be a preferential option. Pilot-scale studies are important to complement results from farm-scale studies.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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ATTACHMENT 41

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Article

Agricultural Biogas Production—Climate and Environmental Impacts

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Abstract: Livestock manure is a major source of the greenhouse gases (GHGs) methane (CH_4) and nitrous oxide (N_2O). The emissions can be mitigated by production of biogas through anaerobic digestion (AD) of manure, mostly together with other biowastes, which can substitute fossil energy and thereby reduce CO_2 emissions and postdigestion GHG emissions. This paper presents GHG balances for manure and biowaste management as affected by AD for five Danish biogas scenarios in which pig and cattle slurry were codigested with one or more of the following biomasses: deep litter, straw, energy crops, slaughterhouse waste, grass–clover green manure, and household waste. The calculated effects of AD on the GHG balance of each scenario included fossil fuel substitution, energy use for transport, leakage of CH_4 from biogas production plants, CH_4 emissions during storage of animal manure and biowaste, N_2O emissions from stored and field applied biomass, N_2O emissions related to nitrate (NO_3^-) leaching and ammonia (NH_3) losses, N_2O emissions from cultivation of energy crops, and soil C sequestration. All scenarios caused significant reductions in GHG emissions. Most of the reductions resulted from fossil fuel substitution and reduced emissions of CH_4 during storage of codigestates. The total reductions in GHG emissions ranged from 65 to 105 kg CO_2-eq ton^{-1} biomass. This wide range showed the importance of biomass composition. Reductions were highest when straw and grass–clover were used as codigestates, whereas reductions per unit energy produced were highest when deep litter or deep litter plus energy crops were used. Potential effects of iLUC were ignored but may have a negative impact on the GHG balance when using energy crops, and this may potentially exceed the calculated positive climate impacts of biogas production. The ammonia emission potential of digestate applied in the field is higher than that from cattle slurry and pig slurry because of the higher pH of the digestate. This effect, and the higher content of TAN in digestate, resulted in increasing ammonia emissions at 0.14 to 0.3 kg NH_3-N ton^{-1} biomass. Nitrate leaching was reduced in all scenarios and ranged from 0.04 to 0.45 kg NO_3-N ton^{-1} biomass. In the scenario in which maize silage was introduced, the maize production increased leaching and almost negated the effect of AD. Methane leakage caused a 7% reduction in the positive climate impact for each percentage point of leakage in a manure-based biogas scenario.

Keywords: biogas; anaerobic digestion; manure; greenhouse gases; methane; nitrous oxide; environmental impacts

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1. Introduction

Emissions of nitrous oxide (N_2O) and methane (CH_4) from livestock manure management contribute around 10% of the total non-CO_2 greenhouse gas (GHG) emissions globally calculated as CO_2 equivalents [1]. N_2O and CH_4 have global warming potentials for time horizons of 100 years (GWP100) of 298 and 25 times, respectively, higher than

GWP100 per kg of carbon dioxide (CO_2) [2]. Ammonia (NH_3) emission and leaching losses of nitrate (NO_3^-) are important indirect sources of N_2O [3,4].

Globally, manure management contributes about 10% of agricultural CH_4 emissions [5], but in confined livestock production systems (e.g., dairies and piggeries) with liquid manure management, this proportion can exceed 50% depending on climate [1]. Animal manure applied to soil contributes to maintenance of soil carbon (C) insofar as a fraction of the manure C is sequestered. The Danish Energy Agency has calculated that of the Danish GHG emissions from livestock production, manure management contributes 22%, manure and mineral fertilizers applied to soil contribute 39%, and enteric fermentation contributes 39% to total agricultural GHG emissions [6].

Emissions of CH_4 and N_2O are regulated and accounted under the UNFCCC as part of the Paris Agreement. The reduction target for the EU on GHG is 55% by 2030 with reference to the year 1990 [7]. The Danish parliament has decided that GHG emissions from Denmark must be reduced by 70% with reference to 1990 by 2030 [6], and agriculture must contribute to this reduction. This calls for the implementation of technologies that cost-effectively reduce GHG emissions from the livestock slurry management chain. Slurry is in focus because 80% of Danish livestock manures are managed in the form of slurries [8].

GHG emissions from slurry management systems can be reduced by AD treatment, frequent export of slurry from livestock buildings to colder outside stores, acidification, or separation of slurry combined with incineration [9–13].

In Denmark, more than 25% (weight basis) of animal manure is today anaerobically codigested on centralized biogas plants with organic wastes from the food industry, slaughterhouses, dairies, and the fish industry with the aim to produce CH_4 for bioenergy. The residues from AD must be recycled as fertilizer and meet the requirements for content of pathogens, heavy metals, and environmentally harmful substances. Biogas plants use almost all industrial residues available in Denmark, and increasing amounts of straw, grass, deep litter, etc. are used in the codigestion of slurry.

The economy is critical when making decisions about the introduction of technologies in farming aiming to reduce GHG emissions, and socioeconomic impacts of different types of biomass for AD have been calculated [14,15]. In scenarios with codigestion of slurry with fibre fraction from slurry separation, maize silage, grass, and sugar beet, NO_3^- leaching was assessed, but the effects of the AD treatment on GHG emissions and NO_3^- leaching were not well documented [14]. More recently, a refined model of CH_4 from manure management that accounted for different storage conditions was used to calculate the effects of biogas and frequent export of slurry from livestock housing to an outside storage tank [16]. However, no biomasses other than slurry were accounted for, nor were any other effects, such as energy and environmental impacts.

When assessing the effect of AD as a potential mitigation measure, a whole-farm approach is needed to estimate GHG emissions from the pig or dairy farm, and calculations must include evaluation of side effects in the form of increased NH_3 emissions, reduced C sequestration, leakage of CH_4 from the biogas plant, etc. [17]. It is relevant to improve estimates of the potential of AD to reduce the negative GHG balance of livestock farming. Therefore, in the present study, the effect of AD on the GHG gas balance was calculated using a "system analysis approach", which included substituting CO_2 emission from power and heat production using fossil fuel; leakage of CH_4 from biogas production plants; CH_4 emissions during storage of animal manure and organic waste; N_2O emissions from stored and field applied manure, organic waste, and digestate; N_2O emissions related to NO_3^- leaching and NH_3 emission; N_2O emission from cultivating energy crops; and effects on soil C sequestration (Figure 1).

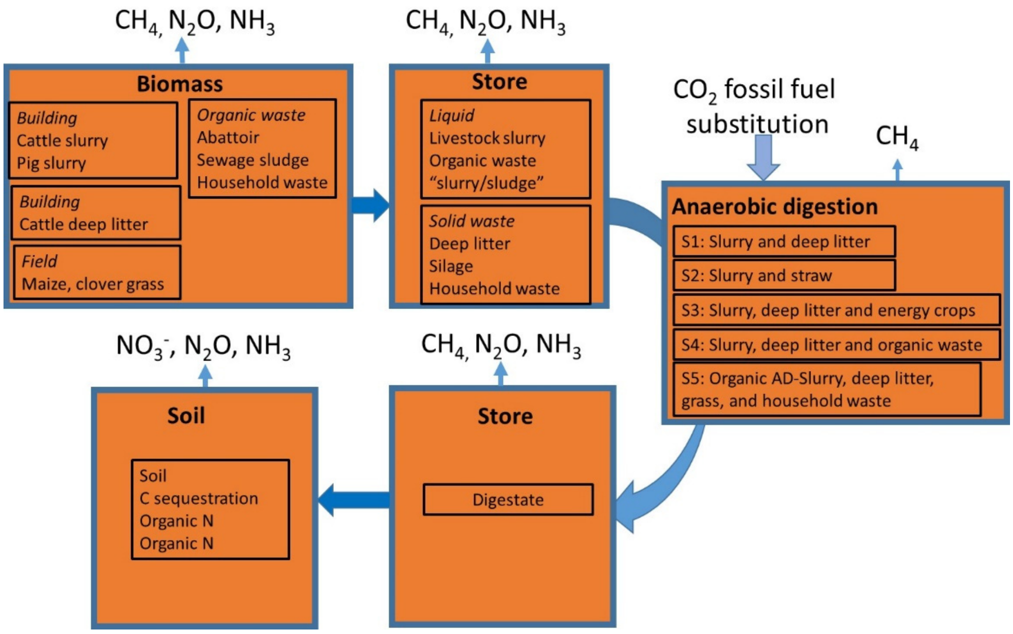

Figure 1. Flow diagram depicting the transfer of biomass among the five compartments of the biomass transfer continuum or system.

This quantification of the climate and environmental effects of biogas production constitutes an important basis for designing and targeting future biogas subsidies to optimize the climate and environmental benefits of production. In this study, assessments are reported for five biomass scenarios in a Danish biogas context with different retention times and biomass compositions. The assessment was based on the best technologies currently used by the Danish biogas sector.

2. Materials and Methods

The calculated climate and environmental effects of introducing centralized codigestion AD in the livestock sector were compared with typical reference management of slurry and waste biomasses. Environmental impacts of introducing AD in five manure management systems (Figure 1; Table 1) were calculated with a whole-system calculation approach for each scenario using one ton of dry matter of biomass as the functional unit. The calculations included CH_4 and N_2O emissions from each scenario (AD and reference farm without AD), the effect on soil C storage, and AD effects on N_2O emissions related to NO_3^- leaching and NH_3 emission. GHG and environmental effects were calculated using the models applied in the Danish national inventory. Global warming potentials for a 100-year time horizon (GWP100) of CH_4 and N_2O at 25 and 298 CO_2-eq kg^{-1}, respectively, were used [18].

2.1. Slurry and Biowaste Management

In this study, the starting point was the collection of excreta and urine in slurry channels. The retention time was set at 20 days for cattle and 19 days for pigs [16], and the average slurry temperature was set at 13.8 °C for cattle and 18.6 °C for pigs. From the channels, slurry was transferred to an outside store and subsequently applied to grassland or arable land between March and August. In the outdoor storage tank, the temperature of untreated slurry was calculated based on monthly mean temperatures [16]. Untreated slurry and digestate must be stored according to national regulations to minimize ammonia emissions [19]. About 80% of untreated slurry was covered by a floating layer of natural crust, straw, or clay pebbles, and 20% was covered by a tent cover. Solid manure was stored for an average of 5 months in heaps covered by a PVC sheet. The deep litter in the reference system was covered with PVC and stored for an average of 5 months.

Table 1. Model plants used in the five scenarios. Distribution of biomass input is given in weight percentage. The slurry input was a mixture of 50% cattle slurry and 50% pig slurry, except in S5, where only cattle slurry was used.

Scenario	Input	Input g DM kg^{-1} Biomass	Reactor g DM kg^{-1} Biomass	Reference Scenario
1	Slurry (80%) + deep litter (20%)	112	95	Animal slurry is stored in a slurry tank and then applied by injection or trailing hose. Deep litter is stored in covered stacks/manure piles for five months and applied before sowing spring cereals.
2	Slurry (80%) + straw (20%)	220	95	Straw is cut and incorporated.
3	Slurry (80%) + deep litter (8%) + energy crops (12%)	94	51	The land farmed with energy crops is used for cereal crops.
4	Slurry (70%) + deep litter (10%) + organic waste (20%)	141	53	The organic waste is stored as slurry and then spread directly on the field (slaughterhouse waste), incinerated (glycerine), or composted and then applied (biowaste).
5	Organic grass–clover (25%) + cattle slurry (50%) + deep litter (20%) + biowaste (5%)	97	95	At an organic farm without a biogas plant, the grass–clover is managed as green manure.

Slurries were applied by injection to bare soil and grassland and by trail hoses to autumn-sown crops such as winter cereals, whereas deep litter had to be incorporated within 4 h after application to soil.

In the reference system, slaughterhouse waste was stored together with livestock slurry and then spread on fields in the growing season. Glycerine was used for energy production by incineration, and source-separated household and industrial organic wastes were composted and used as fertilizer. The maize grown for codigestion in S3 was assumed to substitute a cereal crop in the reference system, and the grass–clover grown for codigestion in the organic farming scenario (S5) was assumed to be cut and mulched in the reference system.

In the AD concepts, liquid manure was transferred from livestock buildings to stores on the farm, as in the reference system (Figure 1). Every 3–30 d, the slurry in the outside store and slurry channels within the building were assumed to be emptied and transferred to stores on the AD plant and covered according to regulations. Deep litter was assumed to be transferred from the farms to biogas plants, where it was stored in covered heaps until used for biogas production. Straw was used for biogas production, and in the organic farm scenario, grass–clover crops were fed to the digester as silage. Organic waste in the form of abattoir waste was fed to the biogas reactors after storage in concrete stores similar to those used for animal slurry, and glycerine was assumed to be stored in containers until use. Source-separated household waste and industrial organic waste were assumed to be stored in covered heaps.

In the five scenarios examined in this study (Table 1), biomass was fed to centralized biogas plants with 45 d hydraulic retention times (HRT) in primary thermophilic reactors (53 °C) and an average retention time of 45 d. In newly built biogas plants in Denmark, HRT tends to be longer, and therefore, the calculations included scenarios with retention times of 60 and 90 days. Digestate was defined to be cooled by heat exchange to 25 °C and then stored at the plant for 20 d in a storage tank with CH_4 gas collection. In all model plants, digestion took place by serial operation in two reactors with the same retention time in each unit, and the biogas was assumed to be upgraded and transferred to the natural gas distribution network by removal of CO_2 in the biogas.

The digestate was transferred from the AD plant to farmer concrete stores, where 50% of the digestate was assumed to be covered with a tent and the rest by a floating cover, straw, or natural surface crust. The digestate was applied by injection to bare soil and grass crops (mandatory in Denmark) and by trail hoses to other growing crops.

2.2. Calculations

Greenhouse gases, NH_3 emissions, and leaching of NO_3^- from different farm compartments in the reference and AD biomass management continuum are depicted in Figure 1. An overview of the calculation of biogas production, emissions, leaching, and soil C storage is given in the following. Short reviews about the algorithms used to calculate emissions, transformation, and leaching losses are presented in Supplementary Materials, which also contain tables with parameters for the algorithms and emission factors.

2.2.1. Biogas Production

The CH_4 production from the different biomasses was estimated from our own experiments and other studies (Supplementary Materials, Table S1). The ultimate CH_4 yield in terms of volatile solids (VS) is the yield achieved at a retention time of more than 90 days, and to determine the yield at shorter retention times, the CH_4 produced in the biogas reactor was determined for each biomass through modelling using the Gompertz equation [20]. The gas potential at a given time was calculated as follows:

$$M(t) = B_0 \cdot \left(1 - e^{-k \cdot t}\right) \tag{1}$$

where M is the cumulative CH_4 yield (mL g^{-1} (VS)), B_0 is the theoretical CH_4 yield (mL g^{-1} (VS)), k is a first-order kinetic rate constant representing the hydrolysis constant, and t is retention time (days). The biogas production was calculated from this equation and thus determined by the amount and quality of biomasses and the retention time in the biogas reactor.

2.2.2. Energy Production and Consumption

The calculations of CO_2 substitution from energy production were based on the displacement of natural gas, in which CH_4 substituted CO_2 corresponding to 0.057 kg CO_2-eq MJ^{-1} [21]. The electricity demand for agitators, pumps, etc. at the plant was assumed to be covered by a mix of the Danish electricity production, which in 2019 was estimated at 0.150 g CO_2 kWh^{-1}, as calculated using data from [22]. The volume of biomass transported in one truckload to and from the biogas plant varied among the different biomass types, and diesel consumption was given per kilometre driven. The distances were calculated as the average additional transport of biomass compared to the reference scenario without AD and varied from 0.8 kg CO_2-eq ton^{-1} for grass and maize silage to 11.6 kg CO_2-eq ton^{-1} for glycerol. The effect on GHGs of replacing mineral N fertilizer due to higher N availability in the digestates was estimated by assuming that the long-term N availability equivalent of total N was 5% higher in the treated than in untreated manure [23]. The potential reduction in N fertilizer use was assumed to give a reduction in GHGs of 5.6 kg CO_2 kg^{-1} N [24], equivalent to 0.28 kg CO_2 kg^{-1} treated N in the biogas plant.

2.2.3. Methane Emission from Slurry and Digestate

Daily methane emissions from slurry and digestate during storage were estimated from the volumes of readily degradable (VS_d, kg kg^{-1}) and slowly degradable organic matter (VS_{nd}, kg kg^{-1}) as proposed by Sommer et al. [9]:

$$F_t = (VS_d + 0.01VS_{nd})e^{(lnA - \frac{E_\alpha}{RT})} \tag{2}$$

where F_t is the methane production rate (g CH_4 kg^{-1} VS h^{-1}), E_α is the process activation energy (J mol^{-1}), lnA (g CH_4 kg^{-1} VS h^{-1}) represents the methanogenic potential of the substrate, R is the universal gas constant (J K^{-1} mol^{-1}), and T is the temperature (K). Equation (1) assumes that the amount and degradability of biomass organic matter, and storage temperature, are main controlling variables.

E_α was set to 81,000 J mol^{-1} [25]. The parameter lnA was highly variable and depended on slurry origin, treatment, storage conditions, and age [4]. Petersen et al. [26] estimated

lnA for slurry collected in pig and cattle barns by measuring CH_4 production rates at the storage temperature and calculating lnA from Equation (1) after rearrangement:

$$lnA = ln\left[\frac{F_t}{(VS_d + 0.01VS_{nd})}\right] + \frac{E_\alpha}{RT} \quad (3)$$

The degradability of VS in slurry changes during storage, and no data were available from outside storage tanks. Instead, a different approach was used in which lnA estimates were related to total VS:

$$lnA' = ln\left[\frac{F_t}{VS_{total}}\right] + \frac{E_\alpha}{RT} \quad (4)$$

Note that the parameter value derived from total VS is referred to as lnA' to distinguish from the original calculation of lnA with reference to degradable VS. A limited number of studies were identified for which information about storage temperature and VS content were available to allow estimation of lnA' in pig and cattle slurry, as well as digestate (Supplementary Materials, Table S4). For the present study, the lnA of pig and cattle slurry in barns reported by Petersen et al. [26] were recalculated to lnA'; Table 2 summarizes the values used.

Table 2. Values of the parameter lnA' used in scenario analyses to represent methanogenic potential; for derivation, see text. In the table, $\bar{x} \pm$ s.e. refers to average and standard error.

Category	Storage Period	lnA' g CH_4 kg^{-1} VS h^{-1}	Reference
Cattle slurry	Barn	30.1	[20]
	Outside store	29.2 ± 0.1	[25,27,28]
Pig slurry	Barn	30.6	[20]
	Outside store	30.3 ± 0.4	[25,27,28]
Digestate	Outside store	27.9 ± 0.4	[28,29]

Methane emissions were calculated separately for cattle and pig slurry in barns and for untreated slurry, digestate, and other biomasses assumed to be stored in outside storage tanks. Assumptions regarding retention time, storage temperatures, etc. are given in Supplementary Materials.

After field application, manure environments are predominantly at a redox level at which little, if any, CH_4 is produced. Transient emissions have sometimes been reported; these are probably due to release of dissolved methane produced during storage [30,31].

2.2.4. Methane Emission from Solid Manure

Methane may be emitted from solid manure (deep litter, fibre fraction from separated slurry or digestate) during storage. The emission level is determined by VS degradability, air- and water-filled porosity, and coverage, which in turn determine biological oxygen demand, gas exchange rates, temperature, and anaerobic volume developing during storage. The Danish emission inventory estimates this source with a model proposed in [4]:

$$EF = BMP * MCF * 0.67 \quad (5)$$

where EF (kg CH_4 kg^{-1}(VS)) is the CH_4 emission factor, BMP (m^3 CH_4 kg^{-1}(VS)) is the biochemical methane production potential, and MCF (%) is a country-specific CH_4 conversion factor. For deep litter, an MCF of 3% is assumed if manure is exported at 1-month intervals or less, and an MCF of 17% is assumed if manure is removed at longer intervals. With a BMP of 0.24 m^3 kg^{-1}(VS), the overall CH_4 emission from barns and during outside storage were as shown in Table 3.

Table 3. Emission factors for CH_4 and N_2O from deep litter in housing and storage facilities (IPCC 2006) as well as emission factors used for the storage period in this report. BMP was 0.240 m^3 (CH_4) kg^{-1} (VS).

Categories	Methane		Nitrous Oxide
	IPCC (housing and outside storage)		
	MCF (% of BMP)	kg CH_4 kg^{-1} (VS)	N_2O-N % of total N
<1 month in housing	3	0.005	1
>1 month in housing	17	0.027	1
	This study (outside storage)		
	kg CH_4 kg^{-1} (C)	kg CH_4 kg^{-1} (VS)	% of total-N
Storage in covered heaps	0.015	0.0075	0.5
Composting	0.03	0.015	2.2

We assumed that half of these emissions would come from outside stores, corresponding to 0.005 and 0.027 kg CH_4 kg^{-1}(VS) for short- and long-term storage periods, respectively (Table 3). Recent studies have shown that emission of CH_4 from uncovered and uncompacted manure heaps are at a level of 0.027 kg CH_4 kg^{-1}(VS) [32–34], and we therefore assumed that the emission factor was 0.03 kg CH_4 kg^{-1}(VS) for uncovered manure heaps and half as much for heaps with PVC tent covers (i.e., 0.015 kg CH_4 kg^{-1}(VS)) (Table 3) [34].

In organic farming, animal manure is often actively composted by turning the heap. This reduces the development of anaerobic volumes in the heap and contributes to lower CH_4 emission compared to undisturbed heaps. Based on a CH_4 emission from actively composted organic waste corresponding to 3% C [32], and assuming the same C/VS ratio in organic waste and deep litter, CH_4 emissions from deep litter were estimated (Table 3).

2.2.5. Nitrous Oxide Emission

Nitrous oxide may be emitted from slurry and digestate, as well as solid manure, during storage and after field application. Nitrous oxide emissions are associated with nitrification and denitrification, two interdependent processes occurring under aerobic and anaerobic conditions, respectively. Oxic–anoxic gradients occur in slurry storages with surface crusts and in the outer layers of manure heaps.

Nitrous oxide emissions during storage of slurry or digestate depend on the development of a floating crust where populations of nitrifying and denitrifying bacteria live. The IPCC guidelines give a default emission factor for storage tanks with a surface crust of 0.5%, i.e., 0.5% of total N entering the storage tank is converted to N_2O [4]. Danish pilot-scale measurements indicated lower emissions, 0.2–0.4% [35], but the level of emissions is influenced by climatic conditions, especially the water balance (rain and evaporation). The emission factor of 0.5% of N in the total flow of slurry was used here even though only the surface can be a source of N_2O. Without a surface crust, the emission factor for N_2O was set to 0 for both untreated slurry and digestate.

For cattle deep litter, IPCC [4] recommends a N_2O emission factor for barn and storage of 1% regardless of retention time in the barn. Assuming half of these emissions occur during outdoor storage, this effectively corresponded to 0.5% of total N. In a review by Pardo et al. [32], N_2O-N emissions from compost heaps with organic waste corresponded to 2.2% of total N (Table 3).

The default emission factor for nitrous oxide emissions from N in field-applied liquid and solid manure is 1% [4]. In soil with organic amendments, be they manure, digestate, or crop residues, the balance between oxygen (O_2) demand and O_2 supply is an important control of denitrification and N_2O emissions [36]. While anaerobic digestion reduces the availability of degradable VS, and hence O_2 demand, the net effect on N_2O emissions depends on the interaction with specific soil conditions. A review of field studies [37] reported mostly reductions in N_2O emissions with AD, but increases have also been reported. In the present study, no effect of anaerobic digestion on N_2O emissions was assumed. Nitrous oxide emissions related to NH_3 emissions and nitrate (NO_3^-) leaching were accounted for with emission factors of 1% and 0.75%, respectively, in accordance with the national inventory of Denmark [18].

2.2.6. Ammonia Emission

The emissions of NH_3 from slurry storage tanks and solid manure heaps were calculated using emission factors estimated by [38]. These were within the ranges given in the recent review by Kupper et al. [39]. The NH_3 emission factor for composting of source separated organic waste was taken from the review by Pardo et al. [32]. When deposited, the emitted NH_3 contributes to N_2O emissions [4], and this indirect source of N_2O was included in the calculations. Ammonia emission factors for applied livestock liquid manure given by [38] were used in this study, while emission factors for each month were calculated with the ALFAM model [40] using average monthly weather conditions and average slurry compositions for Denmark. Based on a review of recent studies, we assumed that NH_3 emission from digestate would be higher than emission from untreated slurry (Supplementary Materials, Table S6).

Data from studies on emissions of NH_3 from deep litter applied to soil were limited in 2008 [38], and only one emission factor for application during different months was given. The evidence about the effects of climatic conditions on emission of NH_3 from solid manure applied to soil is still limited, and emission factors cannot be assessed at a monthly scale [41]. We therefore used the same emission factor for applied deep litter regardless of whether the application took place in the spring or autumn. Deep litter must, in Denmark, be incorporated into the soil within 4 h of application, and it was assumed that within this timespan, 25% of total ammoniacal N (TAN) was emitted as NH_3.

2.2.7. Crop Production and Nitrate Leaching

AD processing of animal slurry and biowaste affects NO_3^- leaching from crop production in both direct and indirect ways. The direct effects are through the effects of applied N in both the first and following year after application of fertilizer or manure; here, AD affects the quality and quantity of N applied. The indirect effects of AD on NO_3^- leaching occur through AD effects on NH_3 volatilization, which affects the N available for plants through both the N loss and the deposited N.

Nitrate leaching during the first year after fertilizer or manure application was assumed to be proportional to the amount of total N (mineral N + organic N) applied [23,42,43]. Total N application was assumed to be similar before and after AD, in accordance with Danish fertilizer regulations, despite more N being plant available after AD. This means that more N is taken up in the first crop and less organic N is left in the soil from digestates. In the scenario with an energy crop (S3), total N application increased by AD, as the energy crop contributed with extra organic N to the system, and the N derived from the energy crop was assumed to replace only mineral N fertilizers with an efficiency of 40% as in the Danish legislation. In the other scenarios, total N application remained the same before and after digestion.

The effect of AD on NO_3^- leaching over a 10-year period was calculated using a model based on the principles described by Sørensen et al. [23]. It was assumed that about 40% of the organic N input would be mineralized during years 2–10 after application, and that 34% of the mineralized N would be leached as nitrate [23]. Because of the lower N mineralization after AD, NO_3^- leaching was also slightly reduced after digestion. The model uses information on soil and climatic conditions, and it was assumed that 80% of the manures were used on sandy soils with precipitation above average Danish levels [44].

The direct and indirect effects of increased NH_3 volatilization after AD on NO_3^- leaching were calculated separately. It was assumed that increased NH_3 emission caused a net reduction in NO_3^- leaching [8] as estimated with the following assumptions. The empirical NLES5 model estimates an average marginal leaching of 17% during the first three years after mineral N application in spring under Danish conditions and at N rates near the economic optimum [44]. In Denmark, 80% of livestock manure is applied to sandy soils [45] with higher-than-average marginal leaching, and therefore, 20% of NH_4^+-N in applied manure/digestate was assumed to be lost by leaching over a 3-year time period. Furthermore, NH_4^+-N applied to soil contributes to organic N from plant residues that may give an extra NO_3^- N leaching equivalent to 2% of the N input in years 3–10 after application [8]. The total reduction in NO_3^- N leaching from NH_4^+-N was therefore 22% of the increase in NH_4^+-N volatilization loss over a 10-year period. However, part of the lost NH_3 would be deposited on agricultural land where part of that pool is leached. It was estimated that 10% the NH_4^+-N lost by volatilization was land deposited and leached as nitrate [8]. Thus, the net effect of increased NH_3 loss on reduction in NO_3^- N leaching was set to 22% − 10% =12% of the increase in NH_3-N loss over a 10-year period. This factor was used to calculate the leaching reduction due to increased ammonia volatilization.

2.2.8. Soil Carbon Storage

The effect of biogas treatment of slurry and other livestock manure on soil C storage is still relatively poorly quantified, but a study based on laboratory incubations measured slightly smaller soil C storage in connection with AD treatment [46]. Based on Thomsen et al. [46], the amount of C digested in the biogas plant was assumed to have contributed to C storage by 25% of the effect achieved when adding C in fresh plant material and straw, i.e., 0.25 × 15% = 3.75% of the C transformed to CO_2 and CH_4 in biogas during the digestion process would alternatively have been stored after a 20-year period, assuming retention in the soil of 15% of the C added in plant material over 20 years [46].

3. Results

3.1. Biogas Production

The largest biogas production was achieved in the scenarios in which slurry was codigested with straw or grass, which were the scenarios with largest energy output. In the energy balances showing reduced CO_2-eq emissions, it was assumed that the CH_4 produced substituted natural gas (Figure 2). Power is used in biogas plants for heating, pumping, and mixing, which reduces the net energy production, but most Danish biogas plants limit the energy consumption using heat exchangers, reducing the digestate temperature to 25 °C. Without heat exchange, the digestate would be stored at higher temperatures and be a significant source of CH_4 emissions. In the calculations, the use of heat exchangers reduced the need for process heat and thereby enhanced the CO_2 balance by around 10% (Figure 2).

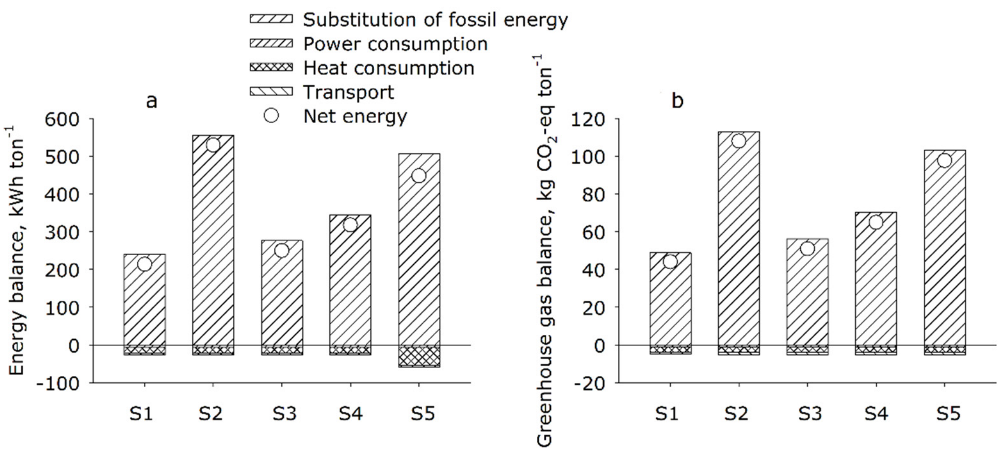

Figure 2. The energy (**a**) and GHG (**b**) balances of the energy production at model plants with heat exchangers installed. Scenarios were (S1) slurry and deep litter; (S2) slurry and straw; (S3) slurry, deep litter, and maize silage; (S4) slurry, deep litter, and organic waste; and (S5) slurry, deep litter, organic waste, and organic grass–clover.

The use of diesel for transport of biomass had little influence on GHG balances (Figures 2 and 3) in the biogas plants and constituted less than 3% of the energy produced.

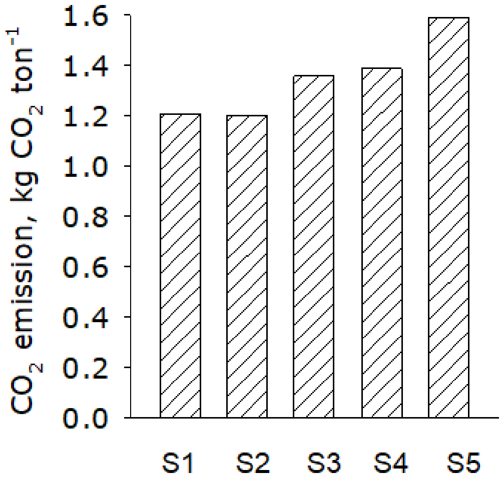

Figure 3. CO_2 emissions from biomass transport in model plants in which slurry was codigested with the following biomasses: (S1) deep litter, (S2) straw, (S3) deep litter and maize silage, (S4) deep litter and organic waste, and (S5) deep litter, organic waste, and organic grass–clover.

3.2. Methane Emissions

Whereas experimental data were available for the estimation of daily CH_4 emissions from cattle and pig slurry in barns, parameters representing the methanogenic potential during storage had to be extracted from published storage experiments. The lnA' values (Table 2) indicated that the methanogenic potential in stored digestate was around 70% lower than that in cattle slurry and 90% lower than that in pig slurry. This was in accordance with the assumption that anaerobic digestion would remove 90% of the degradable VS. With these differences in methanogenic potential, the five biogas scenarios showed reductions in CH_4 emissions from barns and subsequent outside storage that varied between 41 and 56% (Figure 4).

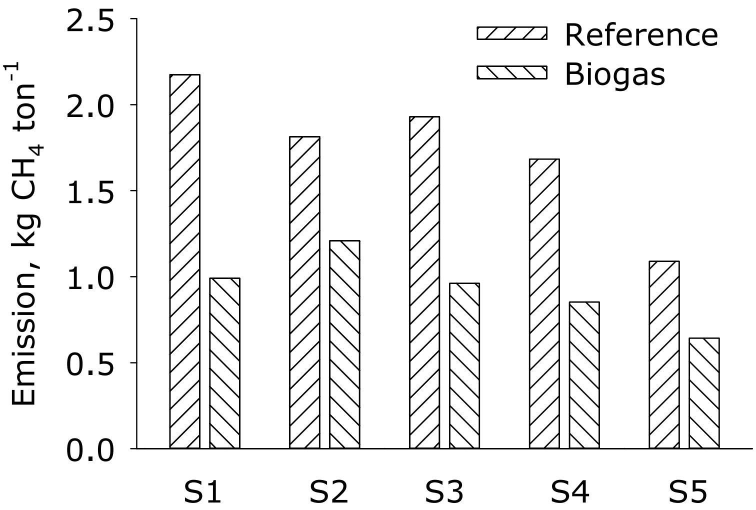

Figure 4. Calculated CH_4 emissions from barn and slurry storage tanks for the five reference and biogas scenarios. The model plants included slurry codigestates as follows (S1): deep litter, (S2) straw, (S3): deep litter and maize silage, (S4) deep litter and organic waste, and (S5): deep litter, organic waste, and organic grass–clover.

In the model calculations, the relationship between VS degradation and CH_4 production depended on assumptions regarding the proportions of C in VS converted to CH_4 and

CO_2, a ratio that is subject to large uncertainty (cf. Supplementary Materials). In the scenario calculations, the CH_4/CO_2 ratio was set at 25:75 for untreated slurry and codigestates stored anaerobically and 10:90 for digestates. The importance of these ratios was evaluated in a sensitivity analysis calculating the effects of reducing by half, or doubling, the share of CH_4 produced in pig and cattle slurry (Table 4). Untreated pig slurry was sensitive to the assumption regarding CH_4 share, with 21% lower CH_4 emissions if the assumed CH_4 share was reduced by half and 17% higher emissions if the assumed CH_4 share was doubled. All other relative changes were negligible. Experimental data on CH_4 emissions from slurry and digestate were the reference for model calculations, and as a consequence, a lower or higher CH_4/CO_2 ratio would lead to less or more residual VS, respectively, being exported and available as substrate for CH_4 emissions from the outside storage. The limited effect of changing the CH_4/CO_2 ratio for digested slurry was due to the fact that pretreatment emissions in barns were identical, and the CH_4 emissions following biogas treatment were greatly reduced.

Table 4. Sensitivity analysis of the importance of the CH_4/CO_2 ratio in the gas produced from VS degradation of untreated and digested slurry for the cumulated CH_4 emissions from barn and outside slurry storage (relative differences with scenario results as basis).

Untreated Slurry			Digested Slurry		
CH_4/CO_2	**Cattle**	**Pig**	**CH_4/CO_2**	**Cattle**	**Pig**
12.5:87.5	0.99	0.79	5:95	0.98	0.99
25:75	1.00	1.00	10:90	1.00	1.00
50:50	1.01	1.17	20:80	1.01	1.00

For each scenario, the CH_4 emissions were calculated using 45, 60, and 90 days of HRT in the reactor, which may affect the predicted CH_4 emissions during subsequent storage of the digestate. This is exemplified in Figure 5b, which shows the sources of CH_4 for scenario S5 (organic biogas) with cattle slurry, deep litter, grass–clover silage, and biowaste, as well as CH_4 emissions without treatment (HRT 0 d) for reference. It was mainly the emission of CH_4 from cattle slurry that was affected by increasing HRT, the emission being 16% less at 90 than at 45 days HRT. The reduction for deep litter was 5–6%, and the changes for grass silage and biowaste were <1%.

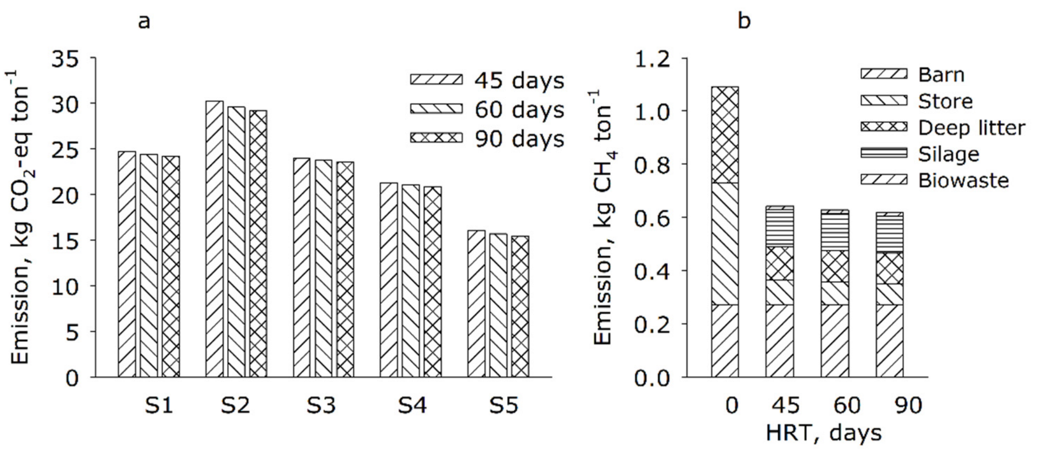

Figure 5. (**a**): Total GHG emissions in each of the five scenarios with 45, 60, and 90 d HRT calculated as CO_2-eq. (**b**): Methane emissions from stored biomasses without biogas treatment (HRT 0 d) or (S5) with biogas treatment at increasing hydraulic retention time (HRT). In the biogas scenarios, livestock slurry was codigested with (S1) deep litter, (S2) straw, (S3) deep litter and maize silage, (S4) deep litter and organic waste, and (S5) deep litter, organic waste, and grass–clover.

The total CH_4 emissions for scenarios with HRT at 45, 60, or 90 d, expressed as CO_2 equivalents, are shown in Figure 5a. All scenarios showed the same trend, with 2–3% lower GHG emission at 90 than at 45 d HRT.

Overall GHG balances were calculated for biogenic CH_4 and N_2O emissions from barns and outside storage, and after field application (Figure 6). While biogas treatment gave a substantial reduction in CH_4 emissions, ranging from 41% to 56%, as described above, no effect on N_2O emissions was assumed, and N_2O emissions are therefore directly proportional to the N content of the biomasses used in each scenario, which were identical in reference and biogas scenarios. As a result, the overall GHG balances corresponded to reductions through biogas treatment ranging between 21% and 40%.

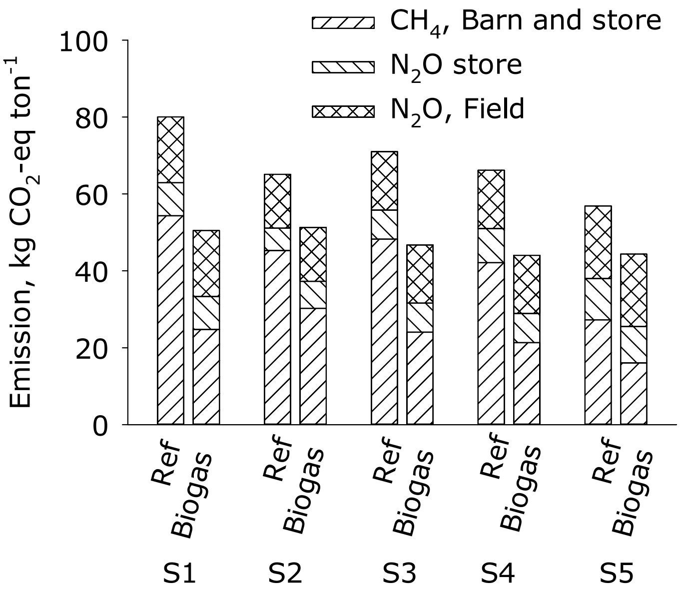

Figure 6. Total biogenic CH_4 and N_2O emissions, calculated as CO_2-eq, from stored slurry and other biomasses in the reference (Ref) and biogas scenarios (Biogas). The scenarios were (S1) slurry and deep litter; (S2) slurry and straw; (S3) slurry, deep litter, and maize silage; (S4) slurry, deep litter, and organic waste; and (S5) slurry, deep litter, organic waste, and organic grass–clover.

3.3. Ammonia Emission

The digestion of animal slurry, crops, and biowaste increases emission of NH_3 because of the higher emission potential of digestate due to its higher pH and TAN to N ratio. Consequently, AD increased NH_3 emission in all five scenarios. The highest increases were in scenarios S3 and S5, in which slurry was codigested with maize silage and grass–clover, which increased the N content in the digestate. This was, in the scenarios with deep litter, to some extent counteracted by reduced emissions from the solid manure management, as emissions were lower from the fraction of digested solid manure than from the solid manure managed in the reference scenarios (Table 5).

Table 5. Changes in NO_3^- leaching, NH_3 emission, and NO_x emission by introducing AD in manure management. In the model plants, pig and cattle slurry was codigested with (S1) deep litter, (S2) straw, (S3) deep litter and maize silage, (S4) deep litter and organic waste, and (S5) deep litter, organic waste, and organic grass–clover.

Source	Scenarios				
	S1	**S2**	**S3**	**S4**	**S5**
NO_3^- (kg NO_3^--N ton^{-1} (biomass))	−0.19	−0.13	−0.04	−0.18	−0.45
NH_3 (kg NH_3-N ton^{-1} (biomass))	0.19	0.18	0.21	0.14	0.30
NO_x (g NOx ton^{-1} (biomass))	2.49	2.48	2.30	3.97	2.13

3.4. Crop Production and Nitrate Leaching

The change in NO_3^- leaching by implementing AD was estimated for a mixture of the most common crops in Denmark and calculated with and without the inclusion of NH_3 emission after digestion. The increase in NH_3 emission reduced NO_3^- leaching by only 0.02–0.04 kg NO_3^--N ton^{-1} (biomass) (data not shown). In the reference scenario, slaughterhouse waste was assumed to be applied to crops as untreated biofertilizer hygienized and mixed with slurry. The source-separated organic household waste would in the reference system be composted and applied to a spring-sown crop as an alternative to digestion. The reduction in NO_3^- leaching by AD was 0.04–0.45 kg NO_3^--N ton^{-1} (biomass), with the lowest reduction in the scenario with an energy crop (S3) and the largest in the organic system with digestion of a grass–clover green manure crop that would alternatively be cut and mulched in the field (S5). The reduction in NO_3^- leaching was lowest in S3 because the overall input of N increased in this scenario. We did not include the effect of a crop change to maize in the leaching effect because of the uncertainty of which crop was being replaced by maize, although it was expected to be mainly cereals. Nitrate leaching is higher from maize than from most other crops [44], and if this effect had been accounted for, or if a higher proportion of maize had been applied in S3, then an increase in NO_3^- leaching would be expected by AD [47].

3.5. Soil Carbon Storage

For the scenario including straw as a cosubstrate, the alternative was assumed to be incorporation of straw into the soil. For one ton of straw with a dry matter content of 85%, and 45% C in the dry matter, this corresponded to reduced soil C storage of 7.7 kg C, equal to 28.1 kg CO_2-eq. It was assumed that substituting cereals for silage maize would not affect soil C storage [48] and that changes from mulching grass–clover to a cutting regime with return of the digestate would also have little effect [49].

3.6. GHG Emissions

Introducing AD in the five different scenarios reduced the net GHG emissions of livestock farming by 67–111 kg CO_2-eq ton^{-1} of biomass at 60 d HRT (Table 6). The corresponding effect in terms of DM was 479–613 kg CO_2-eq ton^{-1} DM. Most of this reduction was ascribed to substitution of fossil energy in energy production and avoided CH_4 emission during storage of biomass. The difference in biogas production in the five scenarios was due to the different amounts of dry matter and biogas potentials in the substrates supplied to the plants.

The biogas plants in scenarios S2 and S5 were supplied with biomasses with high dry matter concentrations, which contributed to a high amount of energy produced per ton of biomass and thus a considerable GHG reduction. If the comparison were to be made independently of these differences in dry matter content, the assessment should be based on the GHG effect per GJ or dry matter input. For example, scenario S2 with straw had a significantly higher GHG effect per ton of biomass than S1 with deep litter. The reason is that considerably more dry matter was supplied in the straw scenario. If the same amount of dry matter had been supplied in the straw scenario as in the deep litter scenario, then the straw scenario would have had a lower GHG effect. This was reflected in the lower GHG effect per GJ of straw than of deep litter. The model plant with the greatest GHG reduction per ton of biomass was the scenario in which 20% straw was added (S2). However, this was also the scenario with the lowest GHG effect in terms of dry matter.

Table 6. Calculated GHG emissions per ton of biomass and per kg of dry matter (DM) (values in brackets) for five model plants at 60 d HRT and 1% methane leakage from the biogas plant. The model plants were (S1) slurry and deep litter; (S2) slurry and straw; (S3) slurry, deep litter, and maize silage; (S4) slurry, deep litter, and organic waste; and (S5): slurry, deep litter, organic waste, and organic grass–clover. Positive values indicate lower emissions, and negative values indicate higher emissions, from biogas.

Source	Unit	S1	S2	S3	S4	S5
Energy	kg CO_2-eq ton^{-1} biomass or kg CO_2-eq ton^{-1} DM	50.44 (450.4)	117.05 (532.1)	57.84 (507.4)	71.83 (509.4)	105.30 (534.5)
Glycerol for heating					−13.80 (−97.9)	
Process energy		−4.08 (−36.4)	−4.08 (−18.5)	−4.08 (−35.8)	−4.08 (−28.9)	−4.08 (−20.9)
Transport		−1.21 (−10.8)	−1.20 (−5.5)	−1.15 (−10.1)	−1.62 (−11.5)	−1.20 (−6.1)
Fertilizer production, N		1.61 (14.4)	1.32 (6.0)	1.43 (12.5)	1.43 (10.1)	1.77 (9.0)
Methane leakage from biogas plant		−4.50 (−40.1)	−10.3 (−46.9)	−5.42 (−47.5)	−6.34 (−44.9)	−9.29 (−47.2)
Methane from storage *		29.91 (267.1)	15.75 (71.6)	24.50 (214.9)	21.04 (149.2)	11.54 (58.6)
Nitrous oxide from storage *		0.00	−1.26 (−5.7)	0.00	1.32 (9.3)	1.32 (6.7)
Nitrous oxide after application		0.00	0.00	0.00	0.00	0.00
Nitrous oxide from nitrogen leaching		0.40 (3.6)	0.27 (1.3)	0.04 (0.8)	0.40 (2.8)	1.01 (4.9)
Nitrous oxide from ammonia emission		−0.69 (−6.2)	−0.66 (−3.0)	−0.76 (−6.6)	−0.51 (−3.6)	−1.11 (5.63)
Nitrous oxide from maize cropping		0.00	0.00	−0.74 (−12.1)	0.00	0.00
Soil C storage (digested biomass)		−3.14 (−28.0)	−6.16 (−28.0)	−2.12 (−18.6)	−2.11 (−15.0)	−2.64 (13.4)
Total impact		68.8 (613)	110.7 (503)	69.6 (604)	67.6 (479)	102.6 (520)
Energy production	GJ gross energy ton^{-1} biomass	0.90	2.07	1.02	1.27	1.86
Total impact	kg CO_2-eq GJ^{-1} gross energy	76.47	53.61	68.19	53.29	55.21
Nitrate leaching	kg NO_3-N ton^{-1} biomass	0.19 (1.7)	0.13 (0.6)	0.04 (0.4)	0.18 (1.3)	0.45 (2.3)
NH_3	kg NH_3-N ton^{-1} biomass	−0.19 (−1.7)	−0.18 (−0.8)	−0.21 (−1.8)	−0.14 (1.0)	−0.30 (1.5)
NO_x	g NO_x ton^{-1} biomass	−2.49 (22.2)	−2.48 (11.3)	−2.30 (20.2)	−3.97 (28.1)	−2.13 (10.8)

* Methane and N_2O from storage relate to emissions from storage of biomasses, especially slurry, deep litter, and slaughterhouse waste.

4. Discussion

The analysis presented here showed that anaerobic codigestion of slurry with biowaste, crop residues, and crops primarily reduced GHG emissions by substituting fossil fuel for power and heat production and reducing CH_4 emission during postdigestion storage. The main environmental benefits from biogas energy systems compared to fossil fuel energy systems occurred in terms of reduced GHG emissions and reduced resource consumption [50]. The impact of utilizing animal manure for biogas production is important in this respect, since avoided emissions from the reference system of conventional manure management could be credited to the biogas system [50], which was not the case for energy crops and straw that would not be sources of GHGs in a non-AD scenario.

In our study, AD was assumed not to affect N_2O emissions, in contrast to calculations presented in a previous study by Sommer et al. [9], in which the removal of degradable organic matter during AD was assumed to reduce N_2O emissions from digestate applied to soil by reducing the potential for denitrification. However, experimental results on this aspect have conflicted [37], and recent studies have made it clear that the amount and composition of denitrification products depend on complex interactions between digestate and soil properties. Thus, accounting for the effect of AD on N_2O emissions would probably require considering the composition of residual VS in digestates, as modified by codigestates and soil gas exchange controlling the exchange of oxygen and denitrification products [51]. The consideration of specific site and weather conditions was beyond the scope of this study, but this should be investigated further.

In our study, the total impact on GHG was 67–111 kg CO_2-eq ton^{-1} of biomass at 60 d HRT, which demonstrates that the biomass mix played an important role. In a study by Poeschl et al. [52] the GHG impact was 75 kg CO_2-eq ton^{-1} for a small-scale plant and 120 kg CO_2-eq ton^{-1} for a large-scale plant. In this study, liquid manure accounted for 55% in the small-scale plant, while in the large-scale plant, only wastes from industry and household were included [53]. Including corn silage had a high positive GHG impact, while grass silage had a negative GHG impact. This was in contrast to our study, in which the scenario with maize had the lowest GHG impact while the scenario with glass–clover had significantly higher impact. However, in a study by Hijazi et al. [50], nonleguminous perennial grass was used. In organic farming, leguminous perennial grass is used as a source for N fertilizer because of its ability to fix atmospheric N [53]. This type of grass might be better than nonleguminous perennial grass in terms of the savings of direct N_2O emission from N input in the form of mineral or organic fertilizers [53].

4.1. Biomass Sources for Biogas

The GHG impacts calculated in our study were based on a Danish territorial perspective; in general, only impacts on Danish national GHG emissions were included. However, the production of commercial N fertilizer was taken into account with a minor climate impact corresponding to 0.28 kg CO_2-eq per kg biomass N, or about 1.5 kg CO_2-eq per ton of biomass. The fertilizer replacement value of biomass is increased after digestion. However, Danish legislation with quotas on mineral N fertilizer application does not account for this, and it is uncertain whether farmers take the higher N availability into account. Therefore, this effect is uncertain in practice but has potential to be utilized. Since there is no fertilizer production in Denmark, such emissions are not included in Denmark's national GHG inventory and therefore were not considered. Neither were the possible effects of changed land use elsewhere on the planet (iLUC) considered. This was relevant only for scenario S2, in which the cultivation of maize as an energy crop was set to replace the production of cereals and could, thus, potentially have iLUC impacts. The iLUC impacts are uncertain, but they may potentially exceed the calculated positive climate impacts of biogas production [53]. In the inventory of biofuels under the EU's Renewable Energy Directive (RED), iLUC impacts are included, although they are not included in the EU requirements for compliance with the RED II [54]. For maize for biogas production, the iLUC impact in a RED context was most recently calculated as 21 kg CO_2-eq MJ^{-1} [55].

Besides livestock manure, the model plants used different types of biomasses from the agricultural sector. Addition of 20% straw to the slurry (S2) gave the largest GHG reduction per ton, but the smallest reduction in terms of DM. The Danish biogas sector is considered a cornerstone in Danish green energy production, and there is increasing demand for more biomass for codigestion with slurry. While there is plenty of unused straw available as cosubstrate, the amount of straw added in scenario S2 cannot be managed with existing biogas technology because of problems related to pumping and agitating the biomass. However, the Danish biogas industry has projected that future technologies in form of pretreatment, pumps, and agitators will be able to manage this amount of dry matter. The calculations in this report assumed that the alternative to the use of straw for biogas was incorporation in the field. If the alternative had been incineration for combined heat and power, the climate impact would have been considerably lower. On the other hand, biogas from straw has the advantage that plant nutrients and part of the slowly degradable C in the straw is returned to the field.

Substituting some of the deep litter (S1) with maize silage (S3) improved the GHG emission reduction per ton slightly. The high degradability of the organic matter in maize contributes to high biogas production per ton of biomass, and energy production was therefore higher for S3 than for S1. In a study from 2013 [14], the GHG emission by codigestion of slurry and 10% maize was reduced by 72 kg CO_2-eq ton^{-1} (wet basis), which was about 10 kg CO_2-eq ton^{-1} (wet basis) more than in our study. In the mentioned study [14], the fibre fraction from separation of slurry was included in the calculation, and it was assumed that it had a higher CH_4 emission potential during storage than deep litter.

Glycerol can be used for energy production in power plants and Otto engines [56], and it is also a useful raw material for biogas production. When glycerol is used in conventional heat and power (CHP) production, the CO_2 reduction effect is 690 kg CO_2 ton^{-1} when it substitutes natural gas, and when producing biogas substituting natural gas, it reduces GHG emissions with an equivalent of 558 kg CO_2 per ton. When compared to incineration in power plants, biogas is considered a high-value energy carrier that can be stored and converted to electricity.

Among the biogas plants, the GHG effect was lowest in the plant with deep litter (S1), since CH_4 emissions during storage increased because of the higher CH_4 emission from digestate than from heaps of deep litter. In S4, with added glycerol, it was assumed that the glycerol would otherwise have been used efficiently for heat and power production. This assumption has not previously been used and is one of the reasons why the GHG reduction potential of an "industrial waste plant" was lower than in the study by Nielsen et al. [57], in which the total GHG effect was a reduction at 90 kg CO_2-eq ton^{-1}.

The organic farming biogas plant (S5) had, next to the deep litter and straw model plant (S2), the largest GHG reduction potential per ton (102 CO_2-eq ton^{-1}), which may be attributed to the high biogas yield as a result of the high proportion of grass, deep litter, and biowaste. In a previous analysis from 2013, the GHG reduction calculated for the introduction of AD in an organic farming system was 83 kg CO_2-eq ton^{-1} [15], but lower effects of both energy production and CH_4 emission during storage was assumed. In our study, the effect of substituting fossil fuels contributed more to the overall GHG impact than in previous studies, in which the importance of reducing CH_4 emissions was greater and a reduction in N_2O emissions was included in the calculations [57].

Energy crops, such as maize, are still a significant source of substrates used by Danish biogas plants, but the amount that can be used is constrained by restrictions under subsidy schemes. Compared with cereal crops such as wheat, there are only limited negative effects of growing maize and other energy crops. However, grass and sugar beets have better environmental and GHG profiles for biogas production than maize [58], and in the future, cover crops are also expected to be used for biogas production. This may reduce the N_2O emissions currently seen after the incorporation of cover crops [59] while at the same time maintaining and possibly improving soil C storage potential. Cover crops and straw together provide a promising source of biogas while at the same time increasing the recycling of nutrients in crop production [60]. Utilization of the expanding area with cover crops as a source of biomass for biogas production would not have the negative iLUC effects that are associated with the cultivation of energy crops for biogas.

4.2. Biogas Plant Configuration

Increasing the reactor size and hydraulic retention time (HRT) reduced net GHG emissions via an increase in the production of biogas and a reduction in the amount of degradable VS in digestate transferred for downstream storage, which will reduced CH_4 emissions from the digestate (Figure 7). The effect of increasing HRT was related to the degradability of the organic matter in the biomass used, and the highest effect was calculated when adding straw or deep litter, which feature high concentrations of slowly degradable biomass. There were increases in the GHG reduction potential by increasing HRT from 45 to 60 days in all scenarios, whereas the effect of extending HRT further was positive only for scenarios S1 to S4, because the positive effect in S5 was outweighed by greater consumption of process energy.

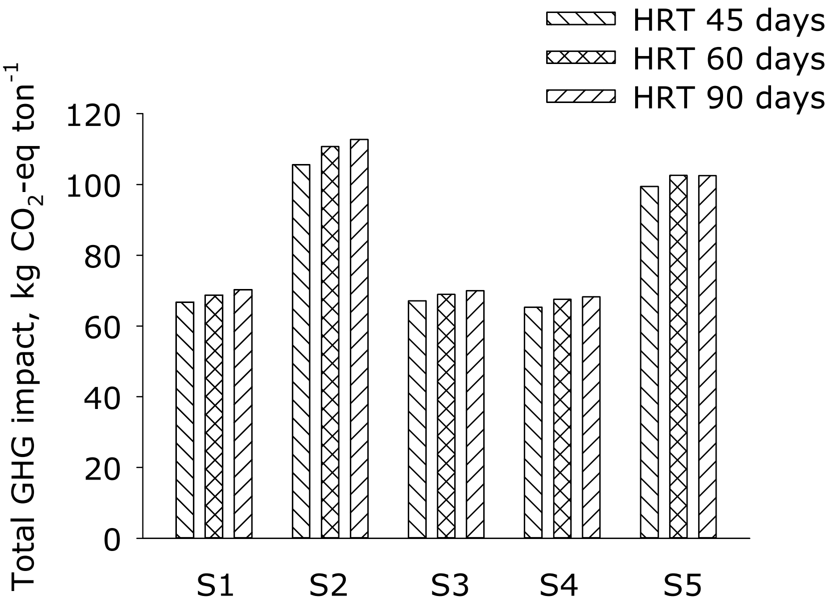

Figure 7. Effect of hydraulic retention time in AD reactor on total GHG impact.

Methane leakage may have a significant effect on total climate impact, which would mainly be due to the global warming potential of CH_4 and, to a lesser extent, the unrealized energy production (Figure 8). The total climate impact was almost linearly reduced with increasing CH_4 leakage (Figure 8). The total impact was reduced by about 5 kg CO_2-eq per ton of biomass from scenario S1 at a methane leakage of 1% to about 10 kg CO_2-eq ton^{-1} of biomass at a methane leakage of 2%. This means that about 7% of the positive climate impact of the plant was lost for each percentage point of leakage. For a leakage of 15%, biogas no longer had a positive effect for scenario S1. Release and leakage of CH_4 from small, unheated digesters may, in a scenario in which biogas energy substitutes coal, negate the GHG-reducing effect of biogas production if 40% of the biogas produced is emitted to the atmosphere [61].

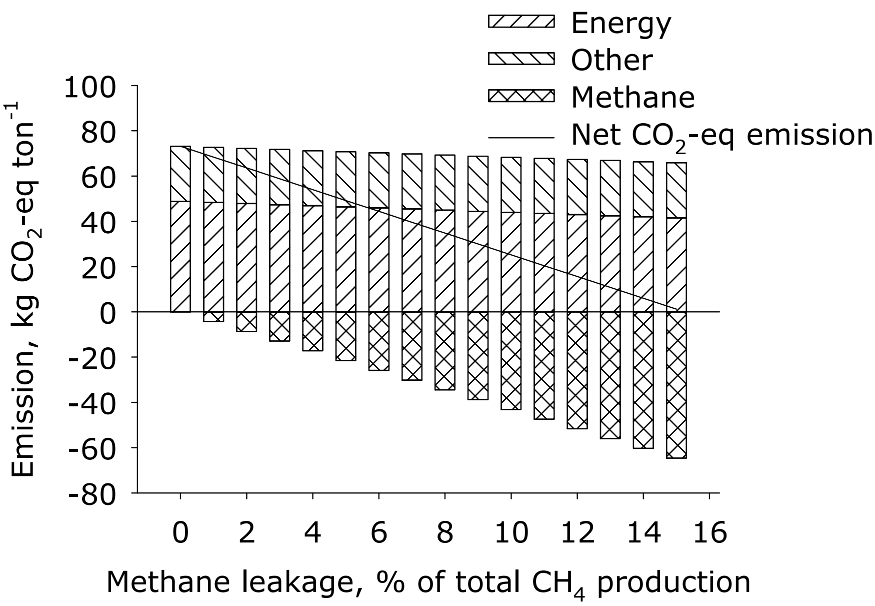

Figure 8. Effect of methane leakage on the climate impact for model plant (S1) with 45 days HRT. Methane and nitrous oxide emissions shown are from storage of livestock manure and biogas slurry.

Nielsen et al. [57] assumed that the organic waste used as codigestate would be stored under anaerobic conditions in the reference scenario and therefore be a significant source of CH_4 emissions. Compared with [57], the biogas scenarios in the present study used a greater diversity of biomasses as codigestates. The relevant alternative management of these biomasses was incorporation (straw), ensiling (maize), storage in a heap (deep litter), or composting (biowaste), and only the slaughterhouse waste included in scenario S4 was assumed to be stored under anaerobic conditions if not digested. To the extent that codigestates were not stored anaerobically in the reference scenarios, this reduced the CH_4 mitigation potential of biogas treatment compared with that in the previous analyses.

4.3. Nitrogen Losses

The reduction in NO_3^- leaching from the AD scenarios was lowest in the system with energy crops (S3), because the overall N input increased in this system with the introduction of more organic N from plant material. If 12% maize silage was used, the effect of digestion on leaching was close to zero, in accordance with previous estimates by Sørensen and Børgesen [23]. Nitrate leaching is typically greater for maize cultivation than for most other crops [44], and if this effect could be included, or if a higher proportion of energy crops were applied in S3, then an increase in NO_3^- leaching would be expected with AD compared with the corresponding reference scenario. However, since the effect is much affected by the assumption about which crop would be replaced by maize, we did not include the effect of a crop change to maize. On organic farms, AD of plant biomass from green manure crops, as in S5, increased the average plant availability of N [60]. This meant that less organic N was left in the soil to contribute to leaching by mineralization in the following years, and the effect of AD in such a system was estimated to reduce NO_3^- leaching by 0.45 kg N ton^{-1} biomass.

The NH_3 emission potential of digestate applied in the field was higher than that of cattle slurry and pig slurry because of the higher pH of the digestate. This effect, and the higher content of TAN in digestate, contributed to higher emission from the AD systems. The effect of higher TAN concentration contributed 60–70% of the increase in NH_3 emission from AD systems, and most of this increase was due to higher emission from digestate applied to soil. The relatively small increase in NH_3 emission in scenario S3 was due to the emissions from storage and application of deep litter and biowaste being large in the reference system. Ammonia emission in the organic scenario, S5, was high because of the increase in the TAN content of digestate originating from N in the codigested grass–clover and the low NH_3 emission from the reference system. This increase in NH_3 can be avoided if organic farms inject slurry into the soil, which is feasible because of the high share of spring crops in organic crop rotations for which slurry can be injected prior to seeding or on grassland during the growing season.

In Denmark, slurry acidification is an alternative to incorporation and injection, but this technology is not suitable for digestates, because the high pH buffer capacity of the digestate results in high demand for acid and thus high cost for this treatment; furthermore, acidification with sulfuric acid is not allowed in organic farming.

4.4. Uncertainties

The calculations were based on the current knowledge about energy production in the form of biogas from different types of biomasses and their related environmental and GHG impacts. Biogas technology is constantly developing, and so is the alternative use of biomasses. This leads to uncertainty with respect to the representativeness of the biogas plants defined and their composition of biomasses. However, the model plants analysed in this study represent the types of biomasses currently used for biogas production in Denmark.

Other uncertainties are associated with the way in which impacts were quantified. The study was based on the models and data used in Denmark's inventories of environmental and GHG impacts. These models are constantly being developed, particularly to better account for variation in environmental controls of the biological processes that determine the impacts and in the properties of biomasses and how they are managed in practice. With the current knowledge, it is not possible to quantify those uncertainties, but a qualitative discussion follows.

The gas potentials of the different biomasses, and the rate at which the gas was produced, are significant sources of uncertainty, especially in the assessment of the effect of retention time in the biogas reactor. The degradation profiles used were, therefore, essential for the estimates of gas yield with different retention times and for the assessment of residual VS in the digestates. The degradation profiles further influenced the CH_4 emissions during the subsequent storage. There is a considerable need for documentation of the rate at which biogas is produced and identification of any interactions between biomasses that may give rise to synergies and/or antagonism. Moreover, it is necessary to provide better documentation of the correlation of CH_4 production between batch tests and continuous systems. These sources of uncertainty may have affected the reported differences among the effects of 45, 60, and 90 d retention time.

Our study assumed that electricity for the process energy used in biogas production was covered by a mix of Danish electricity production, which was estimated to be 0.150 g CO_2 kWh^{-1} in 2019. It may be argued that such emissions could have been 0 g kWh^{-1} if only renewable energy had been used. If it were assumed that no CO_2 was emitted from the production of process energy, the total positive climate impact would increase by about 0.97 kg CO_2 ton^{-1} of biomass at 45 d retention time, increasing the total climate effect by a maximum of 1.7%. Hence, the emission factor assumed for electricity for process energy was less important.

The estimation of CH_4 emissions and the related degradation of organic matter during storage of biomasses was based on simplified input data and assumptions. Firstly, CH_4 emissions were assumed to be a product of VS and temperature alone, but this does not always well explain temporal dynamics [62], and the composition and growth of methanogenic communities would probably be part of an improved model [63]. Furthermore, CH_4 emissions were calculated from total VS and not degradable VS because of a lack of relevant experimental data regarding VS composition. In the analysis presented here, increasing the retention time in biogas reactors from 45 to 90 days showed only a limited effect on CH_4 emissions during subsequent storage, with a 15% reduction in posttreatment emissions from cattle slurry being the most significant effect. A possible reason for the limited sensitivity to HRT could be that CH_4 emissions were estimated on the based on total VS and using the same lnA' value for digestate regardless of retention time. This parameter also represents the slowly degradable parts of VS for which anaerobic degradation was small whether HRT was 45, 60, or 90 days. It is likely that a model in which the VS degradability of each biomass was defined would better capture differences among digestates from the five scenarios with very different feedstocks.

Biogas treatment of pig and cattle slurry (and codigestates) reduces the availability of easily degradable organic matter. All other factors being equal, the resulting decline in demand for oxygen to degrade residual organic matter after field application should reduce the extent and lifetime of anoxic conditions with a potential for N_2O production in well-drained soil. There are, however, confounding effects. For example, soil compaction and periods with rainfall reduce soil oxygen status and change the conditions for N_2O production in untreated manure and digestates applied to soil [36]. Furthermore, fibre-rich codigestates such as maize silage or deep litter may change the distribution of degradable VS in the soil of digestates compared with that of untreated manure [64]. A systematic investigation is needed to elucidate interactions among manure and digestate properties, soil conditions, and N_2O emissions before the effect of anaerobic digestion on N_2O emissions can be included in biogas assessments.

The calculations included an assumption that digestion of slurry and biomasses reduced soil C storage compared with direct field application of those biomasses. Only very limited documentation exists of this effect [46], which is very difficult to determine experimentally. Therefore, the effect is also uncertain, and further studies are needed.

In our calculations, the higher content of TAN in the digestate caused a significant increase in NH_3 emissions in the scenario calculations, corresponding to 60–70% of the higher NH_3 emission. However, there is great uncertainty in these estimates due to a lack of knowledge about how the different combinations of substrates and slurry in the biogas scenarios would affect digestate physical characteristics and infiltration in the soil. Ammonia emissions from slurry or digestates are reduced with faster infiltration, which is a function of viscosity, dry matter content, and adhesiveness. It is poorly understood how infiltration of digestate is affected by the substrates used for AD.

Regardless the types of biomasses used for biogas production, the following measures are important to achieve the potential environmental and climate benefits:

- CH_4 leaks from the biogas installation should be minimized.
- Digestate storage should be covered, and low-NH_3-emission technology should be used for field application.
- Heat exchangers should be employed to cool down the digestate to ambient temperature before storage to improve the energy balance and reduce GHG and NH_3 emissions.

5. Conclusions

Environmental and climate assessments were conducted for different biogas scenarios to evaluate the sustainability of this treatment technology and identify potential improvements of environmental and climate impacts. The scenarios were analysed considering (i) biomass composition; (ii) process temperatures; (iii) hydraulic retention time; (iv) methane leakage from biogas installations; and (v) digestate storage and field application. With respect to energy production, only upgrading for the natural gas grid and substitution of natural gas with biogas were considered.

On the basis of this study, the following conclusions were drawn:

(1) The scenarios investigated resulted in GHG mitigation ranging from 65 to 105 kg CO_2-eq ton^{-1} biomass. Reductions per ton of biomass were greatest when straw or grass–clover was used for codigestion, whereas reductions per unit energy produced were highest with deep litter and deep litter plus maize silage.
(2) The ammonia emission potential of digestate applied in the field was higher than that from untreated cattle and pig slurry because of digestates' higher pH, resulting in an increase in ammonia emission of 0.14 to 0.3 kg NH_3-N ton $^{-1}$ biomass. The use of low-emissions application technology for a larger share of the digestate should limit these higher emissions.
(3) All scenarios reduced nitrate leaching (0.04 to 0.45 kg NO_3-N ton^{-1} biomass). However, introducing maize silage almost eliminated this reduction.
(4) Increasing the hydraulic retention times led to higher climate impact via increased energy production and lower amounts of volatile solids available for degradation and subsequent CH_4 emission during digestate storage.
(5) Methane leakages can have a significant effect on the total climate impact, with about 7% of the positive climate impact being lost for each percentage point of leakage in a manure-based biogas scenario.
(6) The methodology used predicted significant reductions in CH_4 emissions but assumed there was no reduction in direct emissions of N_2O from digestates, which is not always true. Furthermore, iLUC, which was ignored which for bioenergy use, may have a negative impact on the GHG balance.

These and other examples given above show the importance of the assumptions chosen for this type of analysis. Still, it was concluded that biogas treatment of livestock slurry and biowastes has the potential to reduce GHG emissions, improve N use efficiency, and reduce nitrate leaching losses. However, the risks of higher ammonia emission and CH_4 leakage during AD need to be managed.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/su14031849/su14031849/s1, Table S1: Assumptions on dry matter content and gas potential of different biomasses used in biogas production. The ultimate CH_4 yield is the yield achieved at a retention time of more than 90 days. The CH_4 yields after 45 and 60 days and ultimate gas yield are based on data from tests at the Foulum biogas plant Aarhus University. Sources; (1) Average of 50 analyses of slurry supplied to two biogas plants, (2) Olesen et al. (2018), (3) Data from Foulum biogas plant, Aarhus University and (4) Data from tests at Foulum biogas plant, Aarhus University; Table S2: Data used for the calculation of energy consumption on a standard Danish biogas plant; Table S3: Energy use and CO_2 emission due to transportation of biomass. CO_2 emissions of 2.7 kg per litre of diesel is assumed; Table S4: The methane production potential values, lnA', for digestate, cattle slurry and pig slurry were calculated based on information extracted from published studies about methane production rate, total VS and temperature. In the table, $\overline{x} \pm$ s.e. refers to average and standard errors, Table S5: Ammonia emission factors for stored liquid and solid manure (Hansen et al. 2008) and organic food waste (Pardo et al. 2015); Table S6: Ammonia emission factors for cattle and pig slurry applied to soil (Hansen et al. 2008), and the novel estimates emission from digestate; Table S7: Assumptions about plant available N (NH_4^+-N) in biomasses before and after biogas treatment during the first crop growing season after application of manure, required N use efficiency for manures and organic wastes (by Danish legislation), and calculated reduction in NO_3^- leaching due to AD of manure—not accounting for changed NH_3 loss. The share of NH_4^+-N in manure is based on Sørensen and Børgesen (2015). Organic N expected to be transformed to NH_4^+ within the first season is included in the NH_4^+-N share of total N; Figure S1: Average cumulative net N mineralisation from organic N applied in livestock manure over a 10-year period after application.

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ATTACHMENT 42

Dinitrogen and methane gas production during the anaerobic/anoxic decomposition of animal manure

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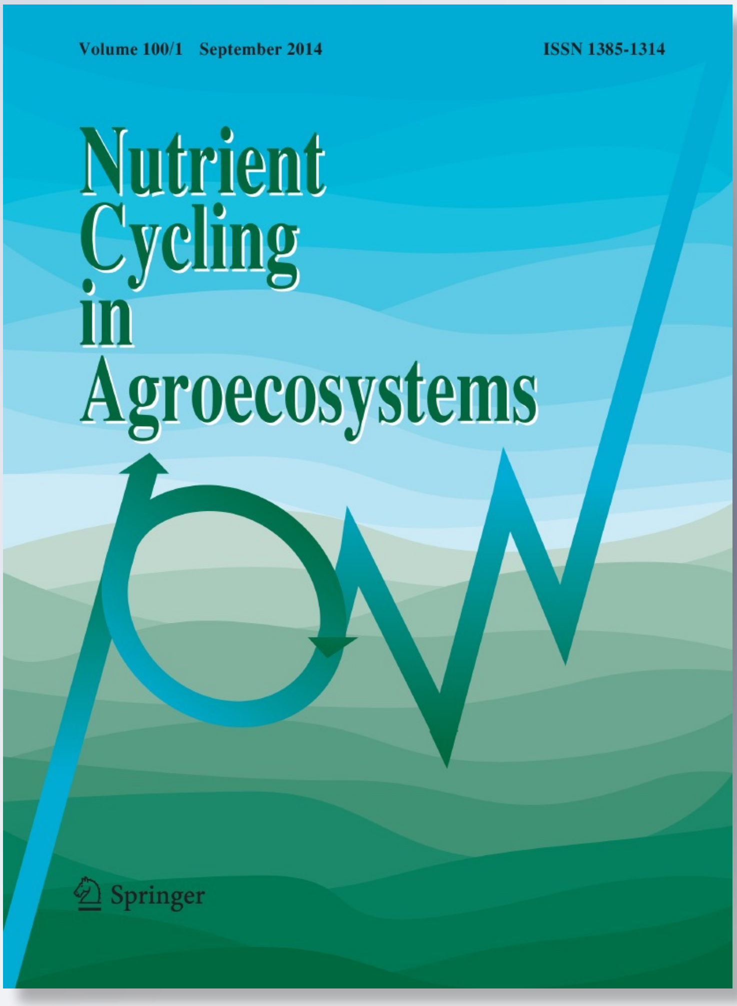

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ORIGINAL ARTICLE

Dinitrogen and methane gas production during the anaerobic/anoxic decomposition of animal manure

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Abstract Trace-gas emissions from animal feeding operations (AFOs) can contribute to air quality and global change gases. Previous and current estimated gas emissions from AFOs vary widely and many do not consider all forms of carbon (C) and nitrogen (N) emissions. Studies have found that as methanogenesis in the lagoons increased, conversion of ammonium (NH_4^+) to dinitrogen (N_2) also increased. The purpose of this research was to measure N_2 and CH_4 emissions from swine AFOs in three locations of the U.S. and to evaluate the possible universal relationship between lagoon methanogenesis and the conversion of NH_4^+ to N_2 gas. This relationship was tested by measuring N_2 and CH_4 emissions in two climates at 22 different farms. Methanogenesis was correlated with NH_4^+-to-N_2 conversion by a near-constant N_2 to CH_4 emissions ratio of 0.20, regardless of C loading and climatic effects. The process is shown to be thermodynamically favored when there is competition between NH_4^+ oxidizing reactions. Under methanogenic conditions (redox potentials of methanogenesis) N_2 production is favorable and nitrification/denitrification is not. Thus, N_2 production is stimulated in methanogenic conditions. Evaluation of NH_3 gas emissions from AFOs must consider other N emissions than NH_3. Finally, a statistical model was developed to estimate methane and N_2 emissions (kg gas ha^{-1}) given feed input per lagoon surface area (kg feed ha^{-1}) and local air temperature. Further studies are needed to investigate the mechanisms involved in manure processing and isolate the favorable mechanisms into engineering improved manure processing.

Keywords Ammonium · Methane · Methanogenesis · Thermodynamics · Lagoon · Dinitrogen

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Introduction

Ammonia (NH_3) is a significant air pollutant, especially in combination with acid gas production from fossil fuel combustion, because the resulting acid–base reaction potentially leads to an air quality problem in the form of haze and respirable particulate matter (PM). The link between PM and increased mortality is well established (Pope et al. 2002; Cohen et al. 2005). Ammonia emissions' estimates from swine manure treatment lagoons, as a percent of feed nitrogen (N) input, have been reported to vary from 36 to 71 % (Doorn et al. 2002a; Hatfield et al. 1993; USEPA. 2004). From a systems' analysis approach using the *USEPA National Emissions Inventory* (USEPA, 2004), the addition of all NH_3 emissions' components, such as from housing (22 %), lagoons (43 %), field application of manure (23 %), N leaving as animal protein (30 %, from host data), suggest that more than 100 % of the N entering the farm system is leaving the farm as NH_3 volatilization plus animal product. Recent studies in North Carolina (NC) (Harper et al. 2004b), the Georgia Coastal Plains (GA) (Harper and Sharpe 1998; Harper et al. 2000), and the Central Great Basin (CGB) (Harper et al. 2010; Weaver et al. 2012) regions have shown that swine lagoons emit significantly less NH_3 than previously and currently thought. Much of the N estimated as NH_3 gas emissions has been found to be converted to dinitrogen gas (N_2) (Harper et al. 2000, 2004b; Weaver et al. 2012), representing an even larger discrepancy for the N balance of farm systems suggested by the USEPA. This aspect of dinitrogen emissions, not considered in most of the estimates of NH_3 emissions from animal feeding operations (AFOs), highlights the fact that the N cycle in lagoons is not fully understood. Benign N_2 emission from lagoons is a pathway of N emissions is that is significant and must be considered in the total N balance of AFOs. When the National Emissions Inventory (USEPA 2004) NH_3 emissions values are combined with published (measured) N_2 emissions (Harper et al. 2000, 2004a, b; Weaver et al. 2012), in many cases more N as NH_3 plus N_2 is emitted than is excreted by the animals, suggesting the need to reevaluate emissions' estimates.

Many of the current NH_3 emissions' estimates are based upon chamber measurements. A number of studies using dynamic chamber measurements (Aneja et al. 2000; Blunden and Aneja 2008) have led to higher emission estimates than found by micrometeorological measurements (Harper and Sharpe 1998; Harper et al. 2000, 2004b, 2010). Doorn et al. (2002b) pointed out that studies with dynamic chambers led to emission factors 2.3 times higher than studies with micrometeorological techniques, while others (Shah et al. 2006; Rochette et al. 1992; Harper 2005; Harper et al. 2010; 2011) stated that chamber techniques are not even suitable for developing emission factors as they create conditions at the water surface that overestimate NH_3 emissions. Based on all of the evidence (Harper et al. 2000, 2004b; Weaver et al. 2012) and discussions regarding the physical chemistry of highly anaerobic systems (van Clemput 1972, 1997), it seems very plausible that NH_3 emissions from lagoons are lower than indicated by current emission factors and a significant fraction of N is emitted as N_2.

There are complex interactions between carbon (C) and N compounds during manure processing by microbial and chemical processes. While little emissions' research for methane (CH_4) and carbon dioxide (CO_2) has been accomplished (Sharpe et al. 2001; DeSutter and Ham 2005) in AFOs, Harper et al. (2000; Table 1; 2010) found interesting correlations between emissions of NH_3, CH_4, nitrous oxide (N_2O), and CO_2 from manure-processing lagoons. These and other studies (Harper et al. 2010) show that manure management aimed at reducing the emissions of one gas could have the undesired consequence of increasing emissions of other gases. In these studies, manure lagoons with a high rate of methanogenesis also converted significant amounts of ammonium (NH_4^+) to benign N_2 gas with little or no N_2O produced [in the lagoons with the highest rate of methanogenesis, atmospheric N_2O was actually absorbed by the lagoon (Harper et al. 2000)]; however, when methanogenesis decreased, smaller emissions of N_2 occurred and higher rates of N_2O were produced. Harper et al. (2010) also showed that removing organic material from swine production farms for biogas production reduced CH_4 emissions by 47 % (the reduction resulted in a 44 % decrease in radiative forcing gases) from the biogas farms while increasing NH_3 emissions from the biogas farms by 46 %, a substantial increase in air-quality emissions. Weaver et al. (2012) also showed similar results. The above studies suggest there is a relationship between the amount of methanogenesis and conversion of NH_4^+ to N_2 gas in manure-processing lagoons. Thus, the main purpose of this study was to measure biological gas emissions from six manure-processing lagoons within a three-county area of eastern NC, a farm in GA, and in a large swine operation in the CGB (15 farms), and to evaluate the relationship between methanogenesis (CH_4 production) and conversion of organic and inorganic N to N_2 gas (N_2 production).

Table 1 Summary of lagoon ebullition studies for methane, carbon dioxide, and dinitrogen gases used in this report

Study location	Study period	Farm type	Lagoon size (ha)	Number animals	Average weight of animals (kg)	Feed N input (kg N $Farm^{-1}$ d^{-1})	Feed input per surface area (kg feed d^{-1} ha^{-1})	Lagoon ammonium (mg NH_4^+-N L^{-1})	CH_4 emissions (kg CH_4 ha^{-1} d^{-1})	CO_2 emissions (kg CO_2 ha^{-1} d^{-1})	N_2 emissions (kg N_2 ha^{-1} d^{-1})
NC (20)[a]	1999–2001	FW #1[b]	2.26	2,000	196.6	117	2,227	175	26.2	0.8	10.2
NC (2149)	1999–2001	FW #2	1.09	1,350	237.0	136	4,960	389	100.3	5.8	25.8
NC (318)	1999–2001	FW #3	0.69	7,644	61.3	111	6,827	422	103.7	5.3	16.4
NC (2704)	1999–2001	F #1	1.61	2,400	197.0	148	4,338	348	73.3	3.2	18.5
NC (3101)	1999–2001	F #2	0.77	3,248	61.3	127	6,205	427	107.3	4.8	17.8
NC (10)	1999–2001	FF	2.55	1,200	643.3	502	7,247	545	169.8	21.7	43.8
CGB (1, 2, 3)[c]	2002–2003	S	1.93	5,000[d]	185.0[d]	423[d]	9,500[e]	1,081	74.8	17.5	36.0
CGB (4, 5, 6)[c]	2002–2003	N	0.50	12,000[d]	13.6[d]	102[d]	8,900[e]	1,475	71.4	20.5	18.2
CGB (7, 8, 9)[c]	2002–2003	F	1.69	11,520[d]	68.0[d]	442[d]	11,400[e]	1,851	72.1	20.2	21.2
The following published emissions from GA (Harper et al. 2000) and CGB (Weaver et al. 2012) were compared in this study:											
GA (1)	1994–1998	F	NA[f]	NA	NA	NA	NA	NA	125.8	7.3	23.1
CGB (10, 11, 12)[e]	2004–2006	F(B)[g]	1.69	10,187	85.0	399	10,800	1,725	87.0	34.5	16.7
CGB (13, 14, 15)[e]	2004–2006	F(C)[g]	1.69	11,954	78.0	399	10,800	1,752	95.7	36.8	20.3

[a] Farm number in parentheses

[b] Letters represent: *F* ~~Farrow~~, *FW* Farrow-to-Wean, *S* Sow, *N* Nursery F = Finisher

[c] Average of three farms

[d] Estimated from Harper et al. (2006)

[e] Calculated from feed N input and 2.3 % N in feed

[f] *NA* not applicable, four cascading lagoons

[g] Biofuel (B) and control (C) farms

Materials and methods

In the 22 swine lagoons studied from all regions, undecomposed organic material (manure) from animal production houses is pumped to lagoons where the organic materials settle to the bottom forming a layer of semi-solid organic material which is anaerobically decomposed producing gas. Gas bubbles emitted from the sludge layer in each of the lagoons, were trapped in six collectors (Fig. 1) randomly located within each of six areas of the lagoon. These gas collectors do not interfere with the emission process, as with NH_3 chambers (Harper, 2005). On a short-term basis ($<$ $\sim$2 weeks), ebullition gases, the result of biological and thermodynamic processes, are emitted from the lagoon bottom and are not affected by climatic events at the lagoon surface; however, NH_3 emissions are highly influenced by the physical processes of water surface turbulence and temperature. The collectors were made of 20-L, open-bottom carboys (0.275 m diameter) with flotation collars at the top of the carboys (Fig. 1) and tethered to the lagoon bottom to collect the mass-flow gases (bubbles) before they reached the water–air interface. All air was removed from the collectors at placement. Water in the collectors was displaced by the ebullition gases over time, visually measured on a graduated scale on the collector periodically to determine gas mass-flux. Gases were transferred from the collectors using sample lines flushed with the gases from the collectors and then subsequently attached to evacuated six-L SUMMA canisters. The SUMMA canister samples were then transported to a laboratory where gas samples were analyzed by gas chromatography (Harper et al. 2004b; Weaver et al. Weaver et al. 2012). No N_2O was found in the collectors via GC. In other studies no N_2O emissions were found from anaerobic lagoons using atmospheric transport techniques and tunable diode laser spectroscopy (Harper et al. 2000, 2004a, b). Samples of helium (He) injected into the collectors showed a sampling procedure error of about 1 % due to atmospheric N_2 contamination (see Harper et al. 2004b). Further, modeling studies showed the theoretical maximum contamination from the atmosphere would be <5–10 % (De Visscher and Harper 2005, unpublished data). Gas fluxes were determined by measuring the amount of gases collected divided by the time between measuring intervals (collection volumes were measured as ebullition necessitated, normally from two to three times per week in summer and weekly or bimonthly in winter) and then multiplying the emissions by the measured concentrations of each gas. This sampling protocol has been used extensively and further description of the measurement technique may be found in Harper et al. (2000, 2004b) and Weaver et al. (2012).

A summary of all farms in this study is included in Table 1. Fifteen farms of four different types in the CGB were sampled during 2002–2006: two sets (2002–2003 and 2004–2006) of three each F farms; another set of three F farms with organic matter removed for biogas production (2004–2006); one set of three each of nursery (N) farms (2002–2003) and sow (S) farms (2002–2003). Data from 2004 to 2006 are from an earlier published study (*after* Weaver et al. 2012). Six farms were sampled during 1999–2001 for N and C emissions in NC including three farrow-to-wean (FW), two finisher (F), and one farrow-to-finish (FF) farms. Data from an F farm in GA during the period of 1994–1998 were included (*after* Harper et al. 2000). Farm animal numbers ranged from 1,400 to 12,000.

Farms were selected in three geographical areas: fifteen in the CGB, six in three NC counties, and one in GA, to evaluate the effect of management on biogas emission rates (subject to host availability). The farm types included F, FW, and FF farms with input feed protein ranging from 13 to 17 % (feed N from 2.1 to 2.7 %). Three sow farms in the CGB were selected for comparison to production farms. Feed input, feed analysis, animal numbers and weights, number of animals sold, and other management information were supplied by the host owners/managers where available. Lagoon temperature was measured 2.5 cm below the water surface and within the sludge layer with micro temperature-loggers [Onset Computer Corp, Bourne, ME (Note: commercial names are included for the benefit of the reader and do not imply endorsement by the authors or their host institutions)]. The lagoons typically never formed crusts on the surface and were well mixed as demonstrated by near uniform temperatures from the top to the top of the sludge layer in lagoons of the CGB study. Since it is not appropriate to calculate NH_3 emissions from chamber systems (Harper 2005), lagoon NH_3 emissions were calculated from pH, NH_4^+ measurements of effluent samples (collected in bottles at the surface of each lagoon), surface lagoon temperatures, and wind speeds measured at 1.5 m height (from a metereological station on site), and a lagoon NH_3 emissions model by De Visscher et al. (2002). Housing NH_3 emissions were estimated from a model developed by Harper et al. (2004a) for North Carolina swine farms.

Effluent and sludge layer samples collected were frozen immediately and shipped to a laboratory for analysis of NH_4^+, nitrate (NO_3^-), nitrite (NO_2^-), and pH (for a description of analysis procedures see Harper et al. 2000, 2006). All lagoons were sampled similarly on a monthly basis.

The precision of biogas emission measurement was evaluated using the absolute value of the coefficient of variation, or relative standard deviation (RSD), obtained by dividing the standard deviation by the mean. To evaluate precision of the individual carboy measurements, the daily carboy emissions of the six lagoon carboys were used to calculate the daily average for the lagoon along with its RSD. The daily RSDs of the lagoons were averaged to calculate the average daily RSD and standard deviation of the daily RSD. To evaluate the precision of the farm lagoon emissions, a similar procedure was followed. Individual farm lagoon emissions are the average of the six carboy measurements. As lagoon emissions from three identical farms were measured for each farm type, the average individual farm emissions per farm type could be determined as well as the standard deviation of the individual farm emissions to calculate daily, monthly and yearly RSDs for each farm type. The RSDs for each farm type were averaged and a standard deviation was subsequently determined for daily, monthly and yearly RSD of individual farm emissions.

Results and discussion

Precision of biogas emission measurements

Average annual gas emissions (total component and percent of total) increased as the amount of farm input-feed per size-of-lagoon increased (i.e. increased manure C with respect to lagoon processing size). Biogas production varied substantially among the six collectors on each lagoon site. The RSD between collectors on a single lagoon, on a daily basis, was 48 ± 13 %. While there was considerable variability between individual collector's measurements, the

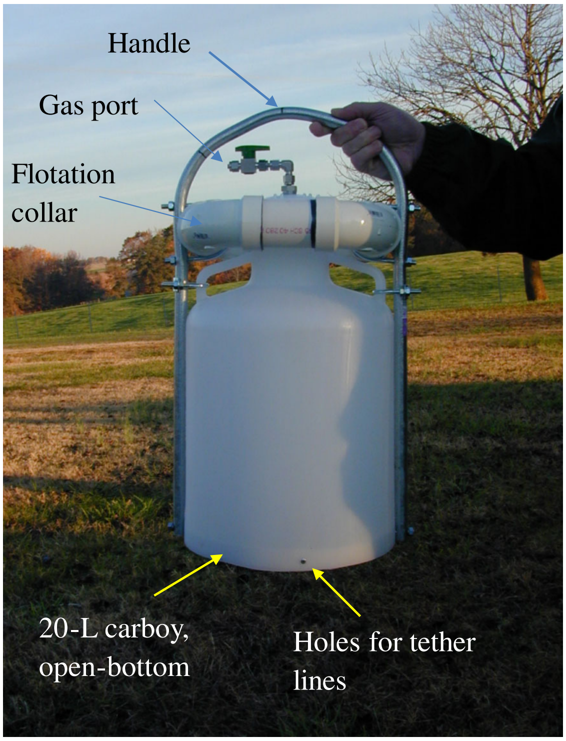

Fig. 1 Gas collectors constructed from open-bottom 20-L carboys. Graduations for evaluation of gas volumes located on the side of the carboy

variability of biogas emissions measurements was much less between lagoons when the six collectors were averaged. For example, the average RSDs of lagoon daily biogas emissions (average of six collectors) from lagoons of identical farms in the CGB were 23 ± 2 %. The variability of measurements between identical farms decreased even further when compared on a monthly (average RSD = 14 ± 6 %) or yearly basis (8.8 ± 6.0 %). We interpret this to mean that the 6 collectors are adequate to determine representative emission measurements on a yearly basis. Individual gas emissions showed regression relationships vs. feed input (R^2) greater than 0.67 for total component emissions (Fig. 2a) and greater than 0.86 for percent of all component gas emissions versus feed input (Fig. 2b).

Climate/temperature effects

When comparing biogas production between farms, temperature effects in the lagoon sludge must be considered. Farms from the CGB were included to test the robustness of the trends in biogas production and their relationship to feed input and temperature. Each system monitored in the CGB was comprised of three identical farms allowing for quantification of the variance in the data. Average monthly CH_4 production was directly related to sludge (where most of the processing occurs) temperature (Fig. 3a). On an annual basis, measured sludge temperatures were found to be within one degree (0.8 °C higher) of the average annual air temperature at 1.5 m height (Harper and Weaver, unpublished data), and we suggest that air temperature can be used as a surrogate temperature for the sludge. When CH_4 production was plotted versus average monthly air temperature (Fig. 3b), the gas production dependence upon monthly air temperature was almost as good as sludge temperature (R^2 values similar). Additionally, the dry climate of the CGB causes much higher evaporation rates and results in different management of swine lagoons.

Feed input effects

Data from the NC farms were used to test for the effects of feed input on biogas production. The NC data demonstrated that total biogas emissions (kg gas ha^{-1} d^{-1}) increased linearly (Fig. 2a) with daily feed input per lagoon size (kg feed d^{-1} ha^{-1}) ($R^2 = 0.67$). Component gas emissions all increased linearly with feed input but CH_4 had the largest increase with feed input per lagoon size ($R^2 = 0.78$). Carbon dioxide and N_2 gas emissions also increased linearly but at smaller rates than CH_4 (with correlations of $R^2 = 0.76$ and 0.32, respectively). Lower correlations for N_2 gas can be partially explained by a change in composition of biogas with feed input (Fig. 2b) where CH_4 and CO_2 emissions, as the percentage of total gas production, increased and N_2 emissions decreased with respect to increased daily feed input rates (kg feed $d^{-1}ha^{-1}$). The N_2 gas produced from the conversion of NH_4^+ to N_2, was not positively correlated with feed input (as was CH_4 and CO_2) since N_2 is produced via a different mechanism than methanogenesis (Weaver et al. 2012).

Feed input values from the CGB could not be used to predict emissions (using the linear relationships determined in Fig. 2a) in other areas due to large differences in lagoon temperatures and to very different animal and manure management. In the CGB lagoons, no effluent was discharged from the lagoon system to maintain water levels (evaporation was sufficient); thus, organic matter was diminished only by anaerobic decomposition and all lagoon N was removed either via NH_3 volatilization and/or conversion of NH_4^+ to N_2 gas (Harper et al. 2000, 2004b; Weaver et al. 2012). Harper et al. (2000) found no N_2O emissions from swine anaerobic processing lagoons (indeed, there was absorption of N_2O from the atmosphere by the lagoon). Additionally, because the feed input (kg feed $ha^{-1}d^{-1}$) was similar between lagoons in CGB (Table 1), the relationship between feed input and emissions could not be tested in the CGB.

The relationship between NH_4^+ concentration and gas emissions was evaluated in the NC lagoons. Similar to lagoon biogas emissions in a GA study (Harper et al. 2000), as NH_4^+ concentration increased across the six NC lagoons, total and individual gas emissions increased. However, the increased emissions effect was due to an increase in manure availability resulting in more biological decomposition from more feed input (Fig. 2a). Additionally, NH_4^+ concentrations also increased with more biological decomposition. Consequently, higher NH_4^+ concentrations and gas emissions are both correlated to feed input and not necessarily to each other.

Mechanisms for N_2 production

When Harper et al. (2000) could not balance the feed N input and all forms of N output (including meat, lagoon NH_3 volatilization, field application NH_3 losses, field denitrification losses of N_2 and N_2O emissions, lagoon N_2O emissions, etc.), they suggested the possibility that some of the NH_4^+ may have been converted to N_2 during manure-processing and that different reactions were involved, depending on the N form and concentration. With higher NH_4^+ concentration and biological activity (i.e. CH_4 production) their studies suggested that the N_2 production may have occurred via 'chemical denitrification' (Van Cleemput 1997). Thermodynamics and the Gibbs free energy of reaction for chemical denitrification (Van Cleemput 1972) suggest that spontaneous conversion of NH_4^+ to N_2 may occur in animal manure lagoons (Harper et al. 2004b, Table 7). It is possible that there is some biological denitrification in the lagoons, but we think it is small since we measured little NO_3^- (<0.1 mg NO_3^--N L^{-1}). Furthermore, dissolved

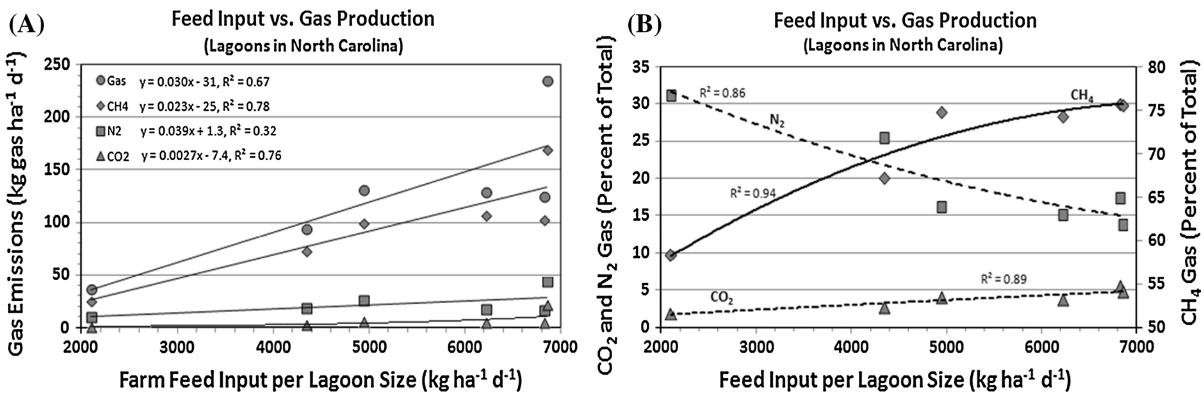

Fig. 2 Average annual lagoon methane, dinitrogen, and carbon dioxide emissions as emissions per unit area of lagoon surface (**a**) and percent of total gas emissions (**b**) with respect to feed input per lagoon size in North Carolina

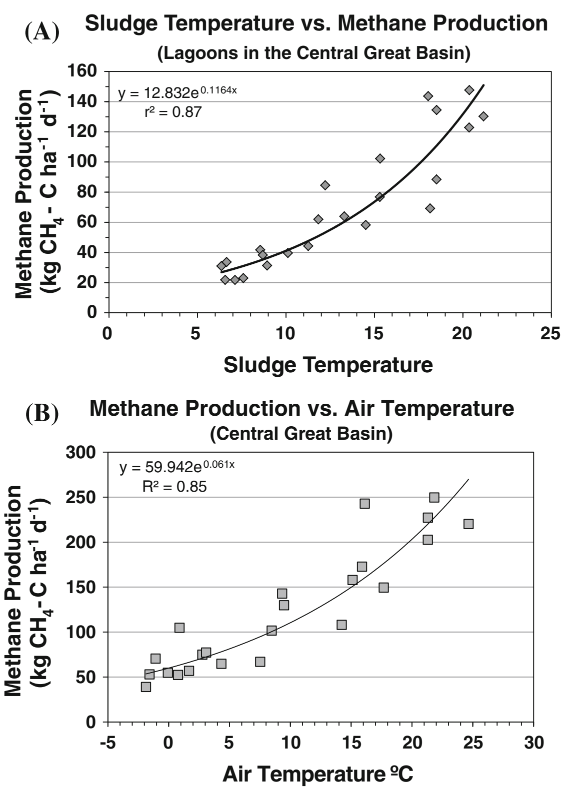

Fig. 3 **a** Average monthly methane production (of three farms) in relation to the sludge temperature at the bottom of the lagoons (where most of the decomposition occurs) over 2 years in the Central Great Basin (CGB). **b** Average monthly methane production (of three farms) in relation to the air temperature over 2 years in the CGB

oxygen (O_2) concentrations (mean of about 0.1 % dissolved O_2 across all the primary lagoons) can barely support autotrophic nitrification even under otherwise optimal conditions. We did not find NO_2^-, an intermediate step in biological nitrification/denitrification, in any of the primary lagoons. Zhang (2003) in studies of an anaerobic sludge reactor also found almost all nitrite removed (97–100 %) with gas contents of 89, 8, and 3 % of N_2, CH_4, and CO_2, respectively. These and other anaerobic laboratory studies (Harper et al. 2001, unpublished data) showed similar conversion of solution NH_4^+ to N_2 gas. Studies of swine lagoons by Hunt et al. (2010) found similar conclusions to Harper et al. (2000, 2004) finding little N_2O (produced from incomplete denitrification) being part of the system N balance. They also found there was a lack of sufficient denitrification enzyme activity (DEA) within the wastewater to support large N_2 losses via classical nitrification and denitrification.

There are other possible microbial processes to explain the N_2 production (Thamdrump 2012). Like classical denitrification and the anaerobic ammonia oxidation bacterial process (ANAMMOX), the full extent of conversion of NH_4^+ to N_2 remains unclear (Ettwig et al. 2009). Kartal et al. (2011) has recently presented strong evidence to explain the ANAMMOX mechanism for conversion of NH_4^+ to N_2 production; meanwhile, in this paper we demonstrate that the simple conversion of NH_4^+ to N_2 is thermodynamically favorable later in the manuscript.

Harper et al. (2000) showed that as lagoon NH_4^+ increased, NO_3^- and dissolved O_2 decreased, while N_2 and CH_4 emissions increased. Other studies have shown that when organic C is removed for biogas production, methanogenesis is reduced and the lagoon NH_4^+ content is increased (Amon et al. 2005) and measured whole-farm NH_3 emissions are increased (Harper et al. 2010). The above studies had treatments which reduce methanogenesis or lagoon NH_4^+ concentration but the studies in this research compare emissions from normal animal management and manure processing systems. The three lagoons in which organic matter was removed for biogas production were included to provide an additional comparison for the effect of reducing decomposition and methanogenesis.

Thermodynamic relationships

The net effect of all these studies suggests that as methanogenesis is decreased, conversion of NH_4^+ to N_2 is decreased. We think the causal relationship between methanogenesis and NH_4^+ to N_2 conversion is thermodynamically favored, while competing with other NH_4^+ oxidizing reactions. The following reactions are considered:

$$NH_4^+(aq) + 0.75\ O_2(g) \rightarrow 0.5\ N_2(g) + H^+(aq) + 1.5\ H_2O\,(l) \quad (1)$$

$$NH_4^+(aq) + 2\ O_2(g) \rightarrow NO_3^-(aq) + 2\ H^+(aq) + H_2O\,(l) \quad (2)$$

$$NH_4^+(aq) + 1.5\ O_2(g) \rightarrow NO_2^-(aq) + 2H^+(aq) + H_2O\,(l) \quad (3)$$

Reaction (1) could represent either a chemical denitrification step, or a microbial process. Without more direct evidence no distinction can be made between a chemical and a microbial process. Hence, we simply refer to reaction (1) as a "conversion" without specifying its nature. Reaction (2), or nitrification, is discussed below. Reaction (3) is significant as no appreciable concentrations of NO_2^- were determined in any of the lagoons. The nitrite ion is key for the anaerobic oxidation of NH_3 (ANAMMOX) as NO_2^- must be present [Eq. (4)]:

$$NH_4^+(aq) + NO_2^-(aq) \rightarrow N_2(g) + 2\ H_2O\,(l) \quad (4)$$

The Gibbs free reaction energy Δ_rG of the three reactions was calculated under the following conditions: NH_4^+ concentration 1500 mg L^{-1}, NO_3^--N concentration 0.1 mg l^{-1}, NO_2^--N concentration 0.1 mg L^{-1}, pH 8, N_2 partial pressure 81 kPa, O_2 partial pressure 0.1–10^{-15} bar. The calculation is similar to that of Harper et al. (2004b) except that the speciation between NH_3 and NH_4^+ was not considered (NH_4^+ is the dominant species and its concentration does not influence the relative Δ_rG between the three reactions).

This relationship is illustrated in Fig. 4. A negative value of Δ_rG indicates that the reaction is thermodynamically favorable. It is clear that the formation of NO_3^- from NH_4^+ is thermodynamically more favorable than N_2 or NO_2^- formation at O_2 partial pressures above 10^{-8} bar when other concentrations remain the same. At lower O_2 partial pressures N_2 formation is thermodynamically more favorable than the formation of NO_3^- and NO_2^- from NH_4^+. This might explain why N_2 production and CH_4 production are correlated. Methanogenesis is only possible at extremely low O_2 concentrations, and under these conditions N_2 production is thermodynamically more favorable than NO_3^- production. This should not be interpreted as conclusive evidence, as both reactions are thermodynamically favorable in all conditions considered; and, other factors like kinetics play a role as well. The presence of an electron donor (organic material) removes oxygen to the point where NO_3^- production becomes thermodynamically less favorable than N_2 production. Kinetically, nitrification has an estimated saturation constant of 0.5 mg L^{-1} according to the standard activated sludge model, ASM3 (Gujer et al. 1999). The ASM3 model predicts that nitrifiers cannot maintain their activity at oxygen concentrations below 0.026 mg L^{-1} (6.3×10^{-4} bar) under otherwise optimal conditions (i.e., in the absence of any other limiting factor).

The sensitivity of the thermodynamics of reactions (1) (2), and (3) to the variables that were kept constant in the above analysis was investigated. The sensitivity of Δ_rG to any of the reactants or products was determined to be less than 11.42 kJ mol^{-1} for any 100-fold change in concentration (or 2 pH units). It is concluded that the thermodynamics of NH_4^+ oxidation is only slightly sensitive to pH and concentrations of NO_3^-, NO_2^-, N_2, and NH_4^+, so a possible uncertainty of any of these variables will not invalidate the analysis.

Comparison of system N emissions

The relative N emissions (ratio of N emitted to feed N input) from the farms in a geographical area (in NC) are shown in Fig. 5 (volatile NH_3-N from housing and lagoons, NH_4^+-N conversion to N_2, protein-N, and unknown-N). Measured N_2 emissions were not consistent within farm types or across all farm locations. The smallest N_2 flux occurred in a farm (FW #3) which also had the highest estimated housing NH_3 emissions. Inversely, the largest N_2 flux was in a farm (FF) with the smallest housing NH_3 emission losses [housing NH_3 emissions were only slightly linearly correlated with N_2 emissions across all farms, R = 0.63 ($R^2 = 0.40$, n = 6)]. Although not conclusive, the inverse relationship suggests that increased N loss as NH_3 will reduce N_2 emissions.

Statistical models for gas emissions

The correlation of methanogenesis and N_2 emissions ($R^2 = 0.78$) was quite good across the lagoons studied in NC leading us to consider if the relationship ($y = 0.23x$) would be comparable across wide geographical regions of the U.S., as well as with management practices. Methane and N_2 emissions were combined (Fig. 6) with the studies in the CGB and from a previous study in GA (Harper et al. 2000). The relationship between CH_4 and N_2 emissions were surprisingly similar changing the overall correlation only slightly, $R^2 = 0.71$, and a linear relationship of $y = 0.20x$, suggesting a near-universal relationship between methanogenesis and conversion of NH_4 to N_2 in highly anaerobic conditions (comparing NC results to all results). The correlation of fluxes was significant at the 2 % level ($t = 3.76$). A linear relationship can be inferred from the data:

$$F_{N_2} = BF_{CH_4} \tag{5}$$

with F_i the flux of compounds i in kg ha^{-1} d^{-1} and B an empirical coefficient. Based on simple linear regression, the value of $B = 0.20$ is found because of the similar compositions of gas from individual systems.

The S farms were not included in the relationship (see X data point) since the animal size and management, feed input, and manure and urine management were very different.

Gas emissions will vary with respect to farm management (feed input, animal weight, etc.) and climatic conditions. As such, it is difficult to directly compare emissions from different locations. Farm management factors most correlated (and data most likely available) are feed input and size of animal. The climatic

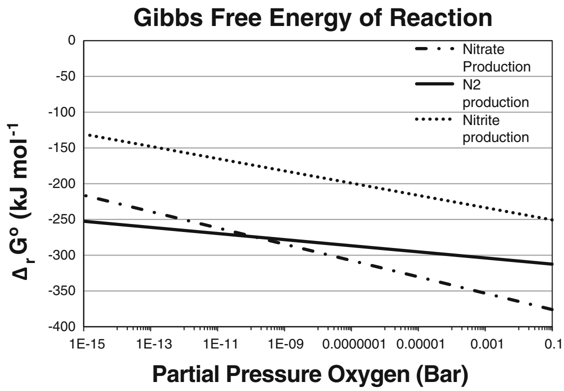

Fig. 4 Gibbs free energy of reaction for N_2 production and NO_3^- production from NH_4^+ versus O_2 partial pressure (NH_4^+ concentration 1,500 mg L^{-1}, NO_3^- concentration 0.1 mg L^{-1}, pH 8, N_2 partial pressure 81 kPa)

factor which most affects the biological decomposition of sludge is the temperature of the biological material in the lagoon anaerobic layer (i.e., sludge temperature, see Fig. 3a). Measurements were used from all the farm systems in the CGB to correct for temperature effects by correlating monthly air temperature with monthly gas emissions as discussed previously. The dependence upon feed input per surface area was estimated from NC data where there was no significant temperature difference between farms. Annual CH_4, N_2, and CO_2 emissions (kg gas component ha^{-1} d^{-1}) were estimated from lagoons by the following relationships:

$$CH_4 = (0.023 \times \text{FIS} - 25) \times (0.039 \times T_a + 0.26) \tag{6}$$

$$N_2 = (0.0039 \times \text{FIS} + 1.3) \times (0.033 \times T_a + 0.41) \tag{7}$$

$$CO_2 = (0.0027 \times \text{FIS} - 7.4) \times (0.040 \times T_a + 0.24) \tag{8}$$

where FIS is the annual average daily feed input per lagoon surface area (kg feed d^{-1} ha^{-1}) and T_a is the average annual air temperature (°C) at the site. Temperature corrections were standardized to the average annual air temperature in the NC studies (18.85 °C). When these relationships were used to estimate CGB gas emissions, estimated CH_4 emissions were 74 ± 24 % high, CO_2 emissions were 58 ± 13 % low, and N_2 emissions were 49 ± 42 % high compared to measured emissions.

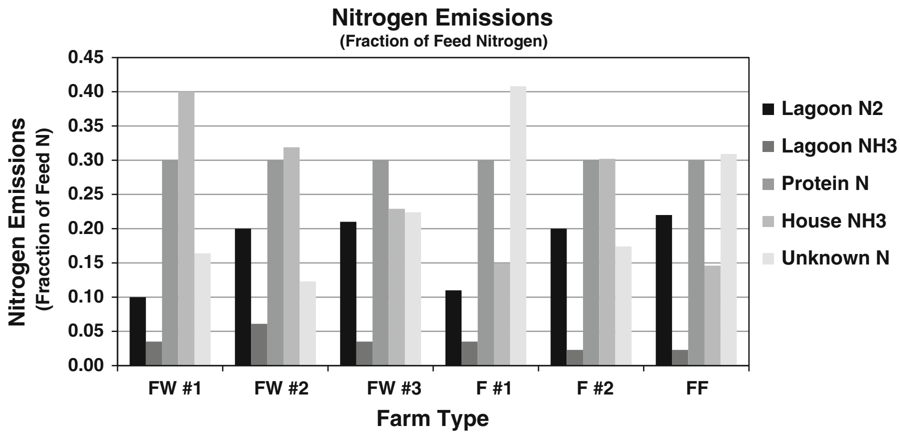

Fig. 5 Comparison of N emissions from farms due to housing NH_3 losses, lagoon NH_3 and N_2 losses, and protein N removal from the farms (*F* finisher, *FF* farrow to finisher, and *FW* farrow to wean)

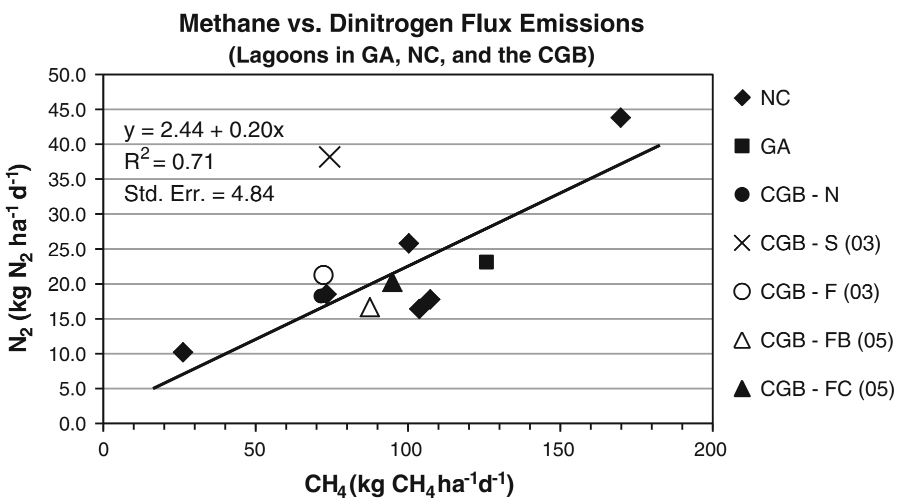

Fig. 6 Average annual dinitrogen production due to anaerobic decomposition in relation to methane production at six farms in North Carolina, 15 in the Central Great Basin (CGB, all data points from the CGB are the average of three identical farms), and one in Georgia (from Harper et al. 2000). The two data points from CGB –F(05) are from Weaver et al. 2012. The Sow farm in the CGB was not used in the relationship since the animal and waste management systems were very different (see X data point)

Using information on the variables measured, we analyzed the data to determine the variables most related (and possibly causal), not already mentioned, to the conversion of NH_4^+ to N_2 for the studies in NC. The amount of feed per average animal weight (and C input) had the highest correlation with N_2 emissions ($R^2 = 0.87$). This is not surprising as feed per animal correlates highly with C and N lagoon input (and consequent increased methanogenesis), along with NH_4 conversion to N_2, across studies over three states (Fig. 6).

Conclusions

In summary, gas emissions were measured in six anaerobic, manure-processing swine lagoons across NC, 15 in the CGB, and one in GA. Conversion of

NH_4^+ to N_2 was observed in all lagoons and a correlation was found between methanogenesis (CH_4 emissions) and conversion of ammoniacal N to benign N_2 gas. Anaerobic digestion not only decomposes organic C to CH_4, but also organic N to NH_4 conceptually leading to an increase in NH_4 concentration and, as a consequence, a potential increase in NH_3 emissions. However, we find in these studies that a reduction of C causes an *increase* in NH_3 emissions, rather than a decrease, since NH_4 is not converted to N_2. Dinitrogen emissions were seen to linearly increase with methanogenesis (CH_4 production), further explaining why removal of organic material from lagoons for biogas production would increase NH_3 emissions from lagoons, a phenomenon which has been seen in other studies (Harper et al. 2010; Weaver et al. 2012). A causal effect for the relationship between methanogenesis and the potential conversion of NH_4^+ to N_2 is explained based on thermodynamics. Dinitrogen emissions can be estimated across all regions utilizing CH_4 emissions (if available). The highest correlation between normally-obtained management variables and N_2 emissions was input-feed per animal-weight which provides the organic C for methanogenesis. Simple statistical regression models including average annual feed input and annual average air temperature were developed which explained most of the N_2 emissions variability and had an acceptable error when tested against other lagoons. These studies provide the capability to estimate farm lagoon CH_4, CO_2 and N_2 emissions from normally-available farm input and local climate data. Further investigations into the mechanisms of NH_4^+ to N_2 conversion and into the variability of CH_4 emissions are needed.

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ATTACHMENT 43

The Effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions

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Methane (CH_4) and ammonia (NH_3) are emitted to the atmosphere during anaerobic processing of organic matter, and both gases have detrimental environmental effects. Methane conversion to biofuel production has been suggested to reduce CH_4 emissions from animal manure processing systems. The purpose of this research is to evaluate the change in CH_4 and NH_3 emissions in an animal feeding operation due to biofuel production from the animal manure. Gas emissions were measured from swine farms differing only in their manure-management treatment systems (conventional vs. biofuel). By removing organic matter (i.e., carbon) from the biofuel farms' manure-processing lagoons, average annual CH_4 emissions were decreased by 47% compared with the conventional farm. This represents a net 44% decrease in global warming potential (CO_2 equivalent) by gases emitted from the biofuel farms compared with conventional farms. However, because of the reduction of methanogenesis and its reduced effect on the chemical conversion of ammonium (NH_4^+) to dinitrogen (N_2) gas, NH_3 emissions in the biofuel farms *increased* by 46% over the conventional farms. These studies show that what is considered an environmentally friendly technology had mixed results and that all components of a system should be studied when making changes to existing systems.

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Mitigation of trace gas emissions has become an important consideration in the design and management of animal feeding operations (AFOs). Although the trend toward increased size of AFOs may increase unit animal production efficiency, both in terms of energy consumption and emissions due to urine and manure management, the consolidation of large numbers of animals can result in a large, localized source of greenhouse and air-quality gases, including methane (CH_4), ammonia (NH_3), and odors. The conversion of animal manure to biofuels (e.g., methanol) is often promoted as an environmentally beneficial management system, with the potential to reduce greenhouse gas emissions (e.g., Wulf et al., 2006; Ghafoori et al., 2006; Brown et al., 2007). Biofuel production systems are designed to remove organic matter (carbon [C]) from the manure stream to produce hydrocarbon fuels. In traditional manure management, this C would ultimately be emitted to the atmosphere as carbon dioxide (CO_2) and CH_4 during decomposition. In theory, biofuel management leads to a direct reduction in greenhouse C gas emissions by diverting the products of manure decomposition, that is, CH_4 with a global warming potential (GWP) equivalent of approximately 25 times CO_2, to a fuel source that is consumed for its energy with its byproduct C emitted as CO_2 (with a GWP of 1).

During urine and manure management processing, however, complex decomposition interactions occur between C and nitrogen (N) compounds. Much of the N that enters into manure processing lagoons is converted to environmentally benign dinitrogen (N_2) gas (Harper and Sharpe, 1998; Harper et al., 2000, 2001, 2004) by microbial and/or chemical denitrification, which reduces the potential for NH_3 emissions. Harper et al. (2000, Table 2) found interesting correlations between emissions of NH_3, CH_4, nitrous oxide (N_2O), and CO_2 from the urine and manure-processing lagoons, suggesting that manipulation of the management system to reduce emissions of one constituent may affect the emissions of another. This study showed that in urine and manure lagoons with a high rate of methanogenesis, there was a significant

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Abbreviations: AFO, animal feeding operation; bLS, backward Lagrangian stochastic analysis; GWP, global warming potential.

amount of chemical conversion of ammonium (NH_4^+) to N_2; however, when CH_4 production (methanogenesis) decreased, smaller N_2 emission rates coincided with higher rates of N_2O production instead of N_2 production. Methanogenesis is only possible at extremely low O_2 concentrations, and under these conditions N_2 production is thermodynamically more favorable than nitrate (NO_3^-) production.

In a series of six geographically widely spaced swine operations in North Carolina, Harper and De Visscher (unpublished data) showed a relationship between CH_4 production and conversion of NH_4^+ to N_2. They found that as CH_4 emissions increased, N_2 emissions increased by a 4:1 CH_4/N_2 ratio (i.e., for a four CH_4 emissions unit increase, N_2 emissions increased by one unit). Amon et al. (2005) and Clemens et al. (2006) found higher NH_3 emissions from biogas-effluent manure slurry than from untreated manure, which they explained by higher NH_4–N content and pH of the effluent. These studies are consistent with the work of Strik et al. (2006), who found an increase in NH_4–N content of manure with time spent in a biogas reactor, and that of Loria et al. (2007), which suggested biogas production increases the NH_4–N content of manure slurry. These latter two studies did not evaluate N_2 production. Because of these relatively new insights, there was concern that removal of the organic matter (and decreasing lagoon methanogenesis) from lagoon urine and manure processing systems may reduce the NH_4^+-to-N_2 conversion and increase the amount of NH_3 emissions from the farms associated with biofuel production. Could the implementation of a biofuel system as a component of the urine and manure processing strategy for AFOs also have negative consequences? Accordingly, the purpose of this study is to evaluate CH_4 and NH_3 gas emissions from farms that are nominally identical, except in regard to manure management, and to determine if the reduction of CH_4 by a biofuel production system affects the total nutrient cycling in the animal production system.

Materials and Methods

Biofuel Facility

The focus of this study was a biofuels facility constructed to capture biogas from digested pig manure and for conversion of the biogas into biomethanol and biodiesel, a process whereby C in the animal manure is converted to a usable and less-polluting fuel for internal combustion engines. As a result, the manure C, which is aerobically or anaerobically decomposed in normal manure-management systems and generally lost to the atmosphere as CO_2 and CH_4, is converted to a usable product.

This biofuel facility is part of a multifarm swine complex in the semiarid Central Great Basin of the United States. Farms are spread over several kilometers along a broad valley (elevation 1500 m), separated from each other by several hundred meters. Liquid manure is collected from 12 12,000-animal finishing farms (144,000 animal total, Fig. 1A) and conveyed to a central treatment plant. The manure is first concentrated, by means of gravity thickeners, and then conveyed to two covered earthen digester tanks, where it is heated to 35°C for undergoing bacterial processing ("digestion"), resulting in the biogas production. The digester effluent, digester sludge, and gravity thickener supernatant are conveyed back to the farm manure

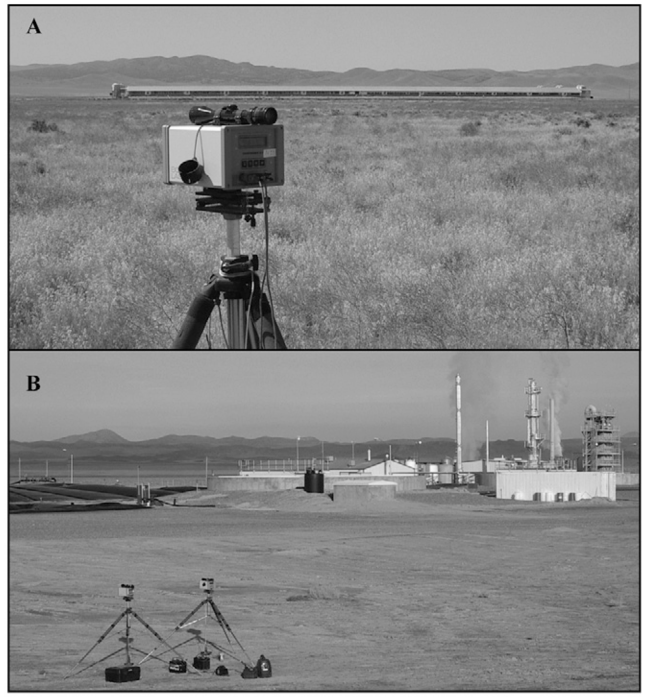

Fig. 1. (A) Open path laser unit located downwind of a farm. (B) Two laser units located at the biofuel facility (B). The laser reflectors are not visible in these pictures.

lagoons where the lagoons operate at the same hydraulic loading rate as they would without the biofuels plant, but with a much-decreased solids loading rate due to the extraction of manure C for methanol production. The biogas is collected and conveyed to a biomethanol conversion plant on the site (Fig. 1B). Biomethanol produced at the facility is trucked from the site as a liquid for conversion to biodiesel at a location remote from the swine production facility.

The Swine Farms

Methane and NH_3 gas emissions were measured from three farms at the production complex. These farms were nominally identical 12,000 animal "finisher" farms. Each consisted of three closely spaced and joined barns with adjacent primary and secondary open-air manure lagoons. The "conventional" farm (Control) uses traditional manure treatment: liquid manure is transferred from the barns into lagoons, where evaporation and decomposition maintain long-term manure equilibrium at the farm such that no manure removal (other than that occurring naturally, i.e., unimpeded venting of C and N compounds to the atmosphere) is performed. Water temperature data were collected using HOBO Water Temp Pro data loggers (Onset Computer Corporation, Bourne, MA). Lagoon pH was measured with pH probes on-site and on samples of the effluent collected twice a month, and frozen, for subsequent pH and NH_3/NH_4^+ analysis. The two "biofuel" study farms (BF1 and BF2) were converted from the traditional manure-management system to the biofuel system as described above.

Emissions measurements were also made at the biofuel plant site. Because the gravity thickeners expose manure to the air, there will be gas emissions from these locations. The rest of the biofuel site is enclosed (i.e., sealed off from the atmosphere), the intent being to eliminate all other emission points.

Emission measurements were conducted from 30 January to 23 February and 27 June to 14 July 2005.

Inverse-Dispersion Technique

The inverse-dispersion technique uses an atmospheric dispersion model to infer the emission rate that best explains the observed downwind gas concentration under existing meteorological conditions, namely, wind direction and speed, temperature stratification, and (consequent) degree of turbulent mixing (Flesch et al., 2004). Consider an area source emitting tracer gas at a uniform but unknown rate Q (g m^{-2} s^{-1}) and assume an average tracer concentration C (g m^{-3}) measured in the plume of dispersing gas. The dispersion model predicts the ratio of the concentration at the measurement location (or more specifically, the increase in atmospheric concentration above background levels attributable to the source) to the emission rate, $(C/Q)_{\mathrm{sim}}$. The emission rate may be computed as

$$Q = \frac{(C - C_{\mathrm{b}})}{(C/Q)_{\mathrm{sim}}} \qquad [1]$$

where C_{b} is the background tracer concentration. This technique is well suited for "ideal surface layer problems" (see Flesch et al., 2004), that is, horizontally uniform terrain where the wind and turbulence can be described by well-known functions of height. In these cases, the wind statistics needed to predict $(C/Q)_{\mathrm{sim}}$ can be inferred from the friction velocity, u_*, the Monin–Obukhov stability length, L, the surface roughness length, z_0, and the average wind direction, β. These primary meteorological properties can be measured with a three-dimensional sonic anemometer. The averaging interval for this type of measurement is ideally in the range of 10 to 60 min.

The terrain around the farm complex is ideal for application of idealized dispersion models: flat and uniform with a sparse coverage of low sagebrush and grass extending in all directions (Fig. 1A). But the farms can complicate an idealized dispersion calculation because farm structures can create wind complexity and the exact spatial distribution of emissions from the barns and lagoons are unknown. (For a compound emission source, e.g., a farm with barns and urine and manure lagoons, one must make assumptions about the relative distribution of the component emissions.) However, Flesch et al. (2005b), McGinn et al. (2006), and Gao et al. (2010) suggested that if the concentration and wind measurements are made far enough downwind of the farm, the emission calculations will be insensitive to these complications. With regard to estimating how far downwind to measure concentrations, Flesch et al. (2005b) suggested that the crucial distance scales are the height of the largest wind obstacle, h, and the maximum distance between the source components, x_{s} (e.g., the distance between the centers of the barn and lagoon). They recommended concentration observations be made more than $10h$ downwind of a farm and more than $2x_{\mathrm{s}}$. This technique has been used in a number of animal feeding operations (e.g., Flesch et al., 2005b; McGinn et al., 2006; Harper et al., 2009, 2010).

Concentration and Wind Observations

Methane and NH_3 concentrations were measured with open-path lasers (GasFinders, Boreal Laser Inc., Edmonton, Canada). The path-average gas concentration was measured between the laser and a distant retroreflector and processed to give 15-min averages (C). Lasers were calibrated on-site using calibration tubes flooded with gas standards. Three NH_3 lasers and a single CH_4 laser were used. Background concentrations of CH_4 and NH_3 were taken as C_{b} = 1.75 and 0 μL L^{-1}, respectively (corroborated when the wind brought air with no farm-gas sources upwind over the laser paths).

The lasers were placed so their paths would be downwind of the emission sites for the prevailing southwest winds. Figure 2 shows the configuration of the sites and laser locations for summer measurements. Configurations for winter measurements were slightly different due to a shift in predominant wind direction between seasons. At the three farms the lasers paths were approximately 200 m downwind of the barns and/or lagoons at the nearest point (Fig. 1A). This was more than $25h$ from the barns and, depending on the farm, was $1.3x_{\mathrm{s}}$ to $6x_{\mathrm{s}}$ downwind (where x_{s} is the distance separating barns from lagoons). Ammonia emissions were measured from the three farms concurrently using the three NH_3 lasers. With only one CH_4 laser during summer, we first measured CH_4 emissions from the control farm and then moved to one of the treatment farms. During winter, lasers were available to measure downwind concentrations for both farms simultaneously. All lasers were used at the biofuel site (Fig. 1B). The NH_3 lasers were positioned so their paths made up three sides of a square

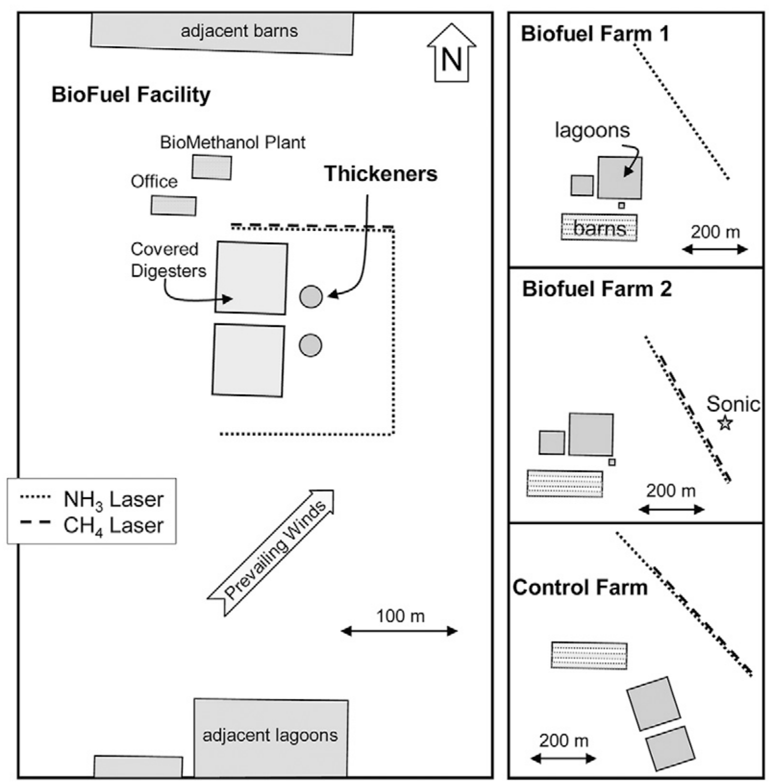

Fig. 2. Summer measurement locations at the four emission sites.

around the two open-air thickeners. In theory, this allowed us to make emission measurements for a wide range of wind directions. The single CH_4 laser during summer was placed northeast of the thickeners. At the closest location, the laser paths were 60 m from the thickeners.

A three-dimension sonic anemometer (CSAT-3, Campbell Scientific, Logan, UT) provided the wind information for the dispersion calculations. The anemometer was placed at height of 2 m. Wind velocity and temperature signals were sampled at a frequency of 16 Hz. The necessary wind statistics (u_*, L, z_0, β) were calculated for 15-min intervals to match the C observations, as described in Flesch et al. (2004). The sonic observations also provided velocity standard deviations (in each of three-dimensions) that were used in the dispersion model.

The sonic anemometer was placed to measure the ambient winds (unaffected by farm structures) at a location central to the study sites: approximately 400 m from the first treatment farm, 1.5 km from the second treatment farm, and 8 km from the control farm. Given the homogeneous character of the landscape (Fig. 1A, flat terrain, low shrub desert flora, no obstacles), we assumed the winds measured at the sonic location were representative of the conditions at the four sites. It should be noted that because of the distance between the farms (and the anemometer), the measured wind statistics for any one 15-min analysis period may not be applicable (concurrently) at all the sites. This adds uncertainty to our individual 15-min emission values. However, over many observations we assume the wind regime between the sites is nearly identical, and our average emission calculations are more accurate.

Backward Lagrangian stochastic Application Details

Following Flesch et al. (2005a), we used a backward Lagrangian stochastic (bLS) dispersion model to calculate $(C/Q)_{\mathrm{sim}}$ (WindTrax, Thunder Beach Scientific, Nanaimo, Canada). Thousands of trajectories are calculated upwind of the laser path for each 15-min observation period (e.g., 250,000). The important information is contained in the trajectory intersections with ground ("touchdowns"), and we compute

$$(C/Q)_{\mathrm{sim}} = \frac{1}{N}\sum\left|\frac{2}{w_0}\right| \qquad [2]$$

where N is the number of computed trajectories, w_0 is the vertical velocity at touchdown, and the summation covers only touchdowns within the source. (The units of Q are g m^{-2} s^{-1} in this equation. Hereafter, we multiply the areal emission rate by the source area and report Q as an area-integrated emission rate with units of kg h^{-1}.) The touchdowns map the concentration "footprint", that is, the ground area where emissions influence concentration.

Each farm is represented as three surface area sources: the two lagoons and the area outlined by the barns (Fig. 2). Each area is assumed to have the same areal emission rate so that touchdowns in any of these areas are counted equally in Eq. [2]. Equating the areal emissions rates in this manner represents an approximation. The two lagoons may have somewhat different emission rates, while the barn is not, in fact, an area source (emissions occur from vents on the walls) and, even if treated as such, may have an effective area source strength differing from that of the lagoons. However, following the arguments of Flesch et al. (2005a), we assume that with C measured sufficiently far downwind, the inferred mean emission rate from the complex is insensitive to the correctness of the approximation. Emissions at the biofuel site were assumed to originate exclusively from the two thickeners, which, having identical manure input rates, were assumed to have equal areal emission rates.

The bLS technique to estimate emission rate depends on a good description of atmospheric transport, which is known to be difficult in extreme conditions. Following Flesch et al. (2005a), we eliminated measurements during periods (i) where $u_* \leq 0.15$ m s^{-1} (low wind conditions), (ii) where $|L| \leq 10$ m (strongly stable/unstable atmosphere), and (iii) where $z_0 \geq 1$ m (associated with uncertainty regarding the proper wind profile). For some wind directions, the farm plumes only "glanced" the path of the lasers, giving more uncertain Q estimates. To avoid these problems we removed periods (iv) where the laser touchdowns do not cover some portion of all the source areas (e.g., at least 50% of the barn and lagoons, or both thickeners).

The bLS technique for calculating emissions has been tested in a number of tracer release studies, conducted in a variety of terrain settings and source configurations. Some of these verification studies are listed in the Appendix. One concludes that with careful use of the technique (e.g., proper equipment siting, data filtering, data averaging), the expected accuracy of the emission calculations should be approximately ±10%.

A problem in this study was a lack of nighttime data suitable for emission calculations, either because of laser alignment problems, unsuitable wind directions, or wind statistics that did not meet the analysis criteria (e.g., low winds). The summer control farm, in particular, had little data that conformed to the selection criteria during the periods from 2300 to 0800 h. To obtain sufficient data to create daily emission curves we relaxed the nighttime data criteria, for the summer control farm only, to include data with an atmospheric stability of $|L|$ from 5 to 10 m for nighttime periods (see Fig. 3C, open circles, for the relaxed data criteria periods). As explained in Flesch et al. (2005b), we thus anticipate greater errors than ±10% (Appendix) for these periods.

Results and Discussion

Biofuel Production Site

Manure slurry from the swine farms is transported and injected into sealed digesters at the biofuel production site. Before injection into the digesters, there was a potential for CH_4 and NH_3 emissions from the gravity thickeners (Fig. 1B and 2), which had an open surface.

Methane emissions should be small at the biofuel site. However, we found some periodic evidence of fugitive CH_4 emissions around the digester pits (either from leaking pipes, digester covers, or leakage from the soil around the pits). We omitted these emission periods on the principle that they are not representative of a well-designed and maintained system. Thus, in our study, we assume CH_4 emissions from the biofuel site are negligible (compared with the farm sites) and can be ignored.

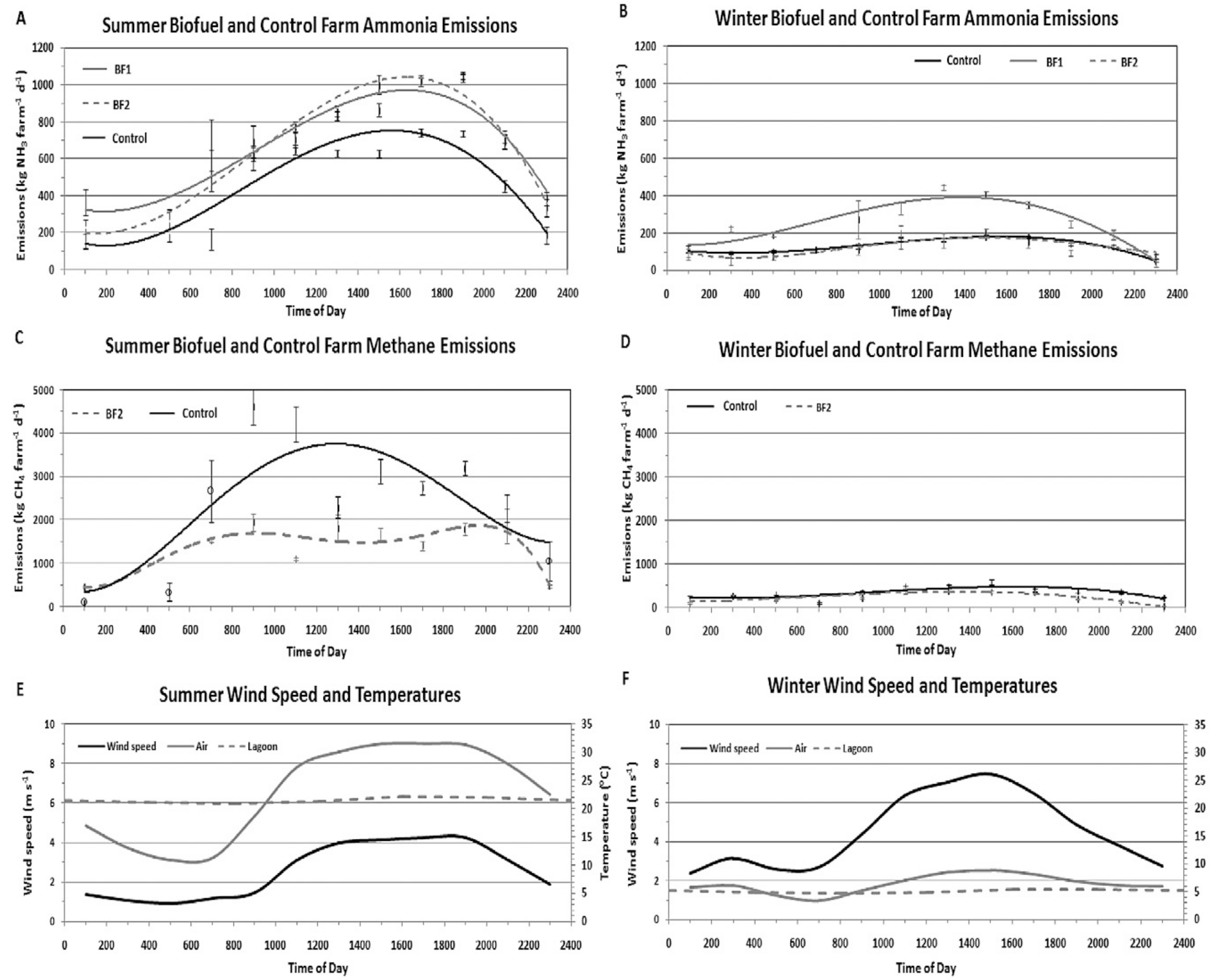

Fig. 3. (A, B) Average summer and winter NH_3 emissions, plotted versus time of day, for biofuel (BF1, BF2) and control farms in the Central Great Basin (error bars are one standard error for each 2-h period). (C, D) Average summer and winter CH_4 emissions for biofuel and control farms. (E, F) Average summer and winter wind speed plus air temperature (height = 2 m) and lagoon temperature (depth = –0.05 m) at the farm site.

On-site NH_3 measurements showed average summertime emissions were 2.6 kg NH_3 h^{-1}, equivalent to 0.4 kg NH_3 h^{-1} for each of the six farms providing organic matter to the site. This was only 2% of the farm emissions during this period. Thus, like methane emissions, these additional NH_3 emission rates were ignored when calculating representative emissions from the biofuel system.

Farm Sites

Figure 3 gives the average diurnal emissions relationship for NH_3 and CH_4 during summer and winter for two biofuel farms and one control farm, along with average wind speed plus air and lagoon temperatures. These emissions were calculated by grouping the available data (15-min observations) according to the time of day, and averaging the data in 2-h blocks throughout the 24-h day. The objective was to create a properly weighted daily average emission rate (i.e., our 15-min observations were not evenly distributed over the day). The curves in Fig. 3 were obtained from these 2-h block averages, with the error bars showing the standard error of the 15-min observations within each block.

The annual emissions were estimated by averaging the daily emission rates from the summer and winter periods. Previous studies (Harper et al., 2004, 2009; Zhu et al., 2010) showed that an annual estimate obtained from the winter and summer averages adequately estimates the annual average, and the addition of a transitional season (three total seasons) did not significantly change the average annual emissions.

Methane and Ammonia Emissions

Comparison between Biofuel Farms

We anticipated that the two biofuel farms, which are nominally identical in management, would have similar emission rates. This was observed to be the case for NH_3 during summer (Fig. 3A) as the two biofuel treatment farms showed no significant difference in NH_3 emissions ($p > 0.05$). However, there was a small difference during winter (Fig. 3B, $p < 0.01$). The difference between the two biofuel farms' emissions during winter was perhaps due to differences in lagoon solution temperature (average wintertime temperatures for the BF1 and BF2 farms were 6.4 and 5.4°C, respectively.). We believe the lagoon temperature for BF1 farm was higher since it is about 1 km closer to the biofuel production site (effluent thermal absorbtion by the soil during return transmission). Ammonia emissions showed diurnal variability during both measurement seasons, but, as anticipated, the winter emissions were a fraction of the summer emissions. Comparison of annual emissions showed no difference between the biofuel farms ($p > 0.05$).

Comparison between Biofuel and Control Farms

Comparisons between the average of the two biofuel farms and the control farm NH_3 emissions (Table 1) showed significantly higher emissions from the biofuel farms during summer (188 kg NH_3 farm^{-1} d^{-1}, +38%, $p < 0.02$) and winter (62 kg NH_3 farm^{-1} d^{-1},+48%, $p < 0.01$). Although there was no significant difference (Table 1) in annual lagoon NH_4^+ concentrations between the biofuel and control farms ($p > 0.05$), the annual lagoon pH was significantly higher in the biofuel farms ($p < 0.05$). This higher pH allows for a greater dissociation of NH_4^+ to NH_3 and therefore the potential for larger NH_3 emissions (Harper, 2005; Koelliker and Kissel, 1988). Comparison of CH_4 emissions between the biofuel and control farms showed significantly smaller CH_4 emissions from the biofuel farm during both summer (1552 kg CH_4 farm^{-1} d^{-1}, –49%, $p < 0.05$) and winter (108 kg CH_4 farm^{-1} d^{-1}, –32%, $p < 0.05$). Both winter NH_3 and CH_4 emissions were only a fraction of summer emissions. When we take the average of winter and summer emission rates to estimate annual emissions, we find that the biofuel farm had annual emissions of CH_4 that were 47% lower but NH_3 emissions that were 46% higher, with the differences being mostly due to the differences in summer emissions. This study showed that on an annual basis, the biofuel farms had CH_4 and NH_3 emissions differences (Table 1) greater than the 10% level of uncertainty we believe exists in the bLS technique (Appendix).

We found the expected result that CH_4 and NH_3 emissions were correlated with wind speed and air temperature (Fig. 4A–D). Studies (Harper et al., 2000, 2004) have shown that increasing wind speeds lead to increasing emissions, on a short-term or diurnal basis, from ponds and naturally ventilated sources. For example, at the control farm (Fig. 4A and 4C), emissions were weakly correlated with wind speed within seasons (r^2 ranging from 0.01 to 0.46). When comparing winter and summer emissions with air temperature, however, we see that long-term emissions are better related to air temperature ($r^2 \geq 0.80$ for both gases). Over long periods of time, the air temperature will be a surrogate of the solution temperature (Fig. 3E and 3F) in both the lagoons and barns. Studies have shown for NH_3 (Harper et al., 2000, 2004) that solution temperature affects the dissociation between NH_4^+ and NH_3 in solution and the diffusion of NH_3 in solution, both of which will alter the emission rate. (NH_3 is a diffusive gas influenced by the physical and chemical factors of solution concentration [NH_4^+], solution hydrogen ion concentration [pH], turbulence [wind speed], and solution temperature. For a discussion of the relationship of physical and chemical factors to NH_3 emissions, see De Visscher et al., 2002; Harper, 2005.) Temperature has a similar effect on emissions from the barns because of the amount of barn air exchanged to maintain comfort of the animals. The effect of temperature on CH_4 emissions has a greater effect on biological activity than on the physical chemistry of gas in solution and transport since CH_4 has five orders of magnitude less solubility in water than NH_3.

GWP Comparison between the Biofuel and Control Farms

There was a 47% decrease in average annual CH_4 emissions at the biofuel farm compared with the control farm; however, average annual NH_3 emissions increased by 46% compared with control farm emissions. These changes in emissions can be used to look at the effect of biofuel production on the GWP of a farm system.

The annual GWP decrease, when considering that methanol (from methane, GWP of 25, 100-yr time horizon) is consumed as biodiesel and emitted as CO_2 (GWP of 1), was 45% compared with control farm emissions. However, NH_3 emissions were increased on the biofuel farms, which is a precursor of indirect nitrous oxide (N_2O, GWP = 298, 100-yr time horizon) emissions. Because a fraction of the NH_3 will, on redeposition on the soil environment elsewhere, be transformed into N_2O (assuming an emissions factor of 1% of deposited NH_3/NH_4^+, Intergovernmental Panel on Climate Change default), an annual NH_3 increase of 144 kg NH_3 d^{-1} would result in a combined net reduction in GWP of 44% for this biofuel management system. By removing the C, and reducing methanogenesis and global-change gas emissions (Harper et al., 2000, 2004; Harper and De Visscher, unpublished data) from the manure-processing system of swine production, the

Table 1. NH_3 and CH_4 emissions from biofuel production and control farms.

Farm type	Season	Lagoon surface concentration	Lagoon surface pH	NH_3 emissions	CH_4 emissions
		mg NH_4^+-N L^{-1}	$[H^+] = 1 \times 10^{-pH}$	kg NH_3 farm^{-1} d^{-1}	kg CH_4 farm^{-1} d^{-1}
Biofuel farms	Summer	1818	8.21	692 ± 59†	1651
Control farm	Summer	1753	8.13	504	3203
% difference	Summer	–	–	+38%	–49%
Biofuel farms	Winter	1827	8.34	190 ± 24†	232
Control farm	Winter	1855	8.16	128	340
% difference	Winter	–	–	+48%	–32%
Biofuel farms	Annual	1823	8.27	460	942
Control farm	Annual	1804	8.14	316	1772
% difference	Annual	–	–	+46%	–47%

† Standard error.

processing of manure NH_3/NH_4^+ to N_2 gas was detrimentally causing an *increase* in NH_3 emissions (air-quality gas) by 46%. This increase in NH_3 emissions creates a potential for additional NH_4^+ particulates and haze production compared with conventional production practices. What had been intended to function as an environmentally friendly technology had (according to our observations) mixed results.

Conclusions

We have seen encouragement to develop "manure-to-fuel" technologies for farming systems, including removal of organic C from animal manure processing systems (e.g., lagoons) for fuel production. As we document in this study, manure-to-fuel production has the benefit of reducing greenhouse gas emissions from animal production systems. It appears, however, that it can also have unforeseen consequences. With the manure removed from the processing system (reduction of methanogenesis), NH_3 emissions were increased. Other studies also have shown that as methanogenesis decreases, the rate of conversion of NH_4^+ to (harmless) N_2 undergoes a parallel decrease. These studies show that we should be aware of the potential for countervailing interactions (in terms of release rates to the atmosphere of diverse species) when modifying animal manure management systems.

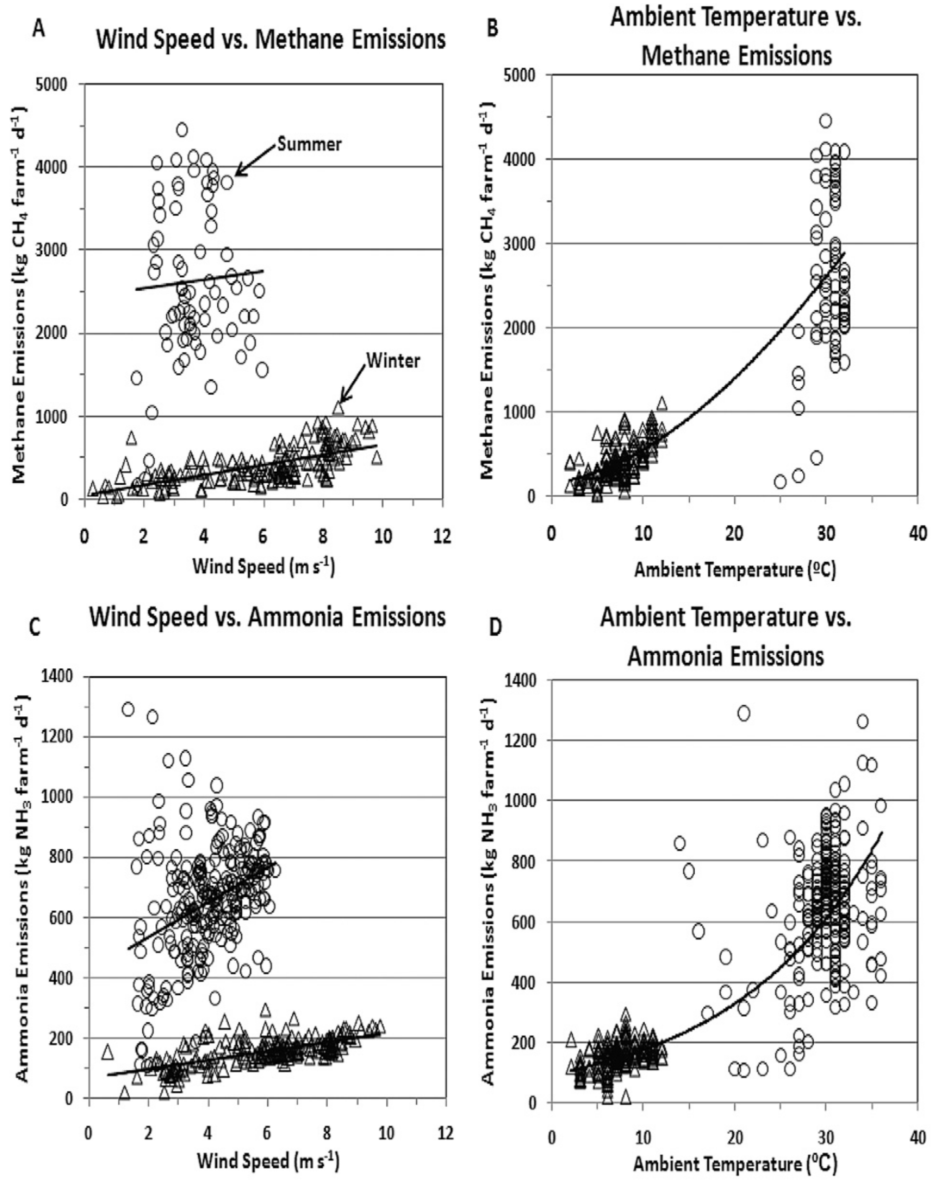

Fig. 4. Emission rates of CH_4 and NH_3 plotted versus ambient wind speed and air temperature (height = 2 m) from the control farm.

Appendix

Table A1 summarizes several tracer studies on the accuracy of the bLS analysis technique for calculating emissions (Q). Accuracy is indicated by the gas recovery, which is the percentage ratio of the bLS calculated emissions to actual emissions (i.e., $Q_{bLS}/Q_{release} \times 100$). These studies had an average recovery of 98% with a standard deviation of 5%. We conclude that for a good site with appropriate instrument placement and data filtering (as discussed in detail in these studies), one can expect a nominal bLS accuracy of 100 ± 10% (± two standard deviations—a span that includes 95% of a Gaussian distributed population). This would be the accuracy of an average of multiple measurement periods. For a single observation (e.g., one 15-min value), these results suggest a higher ±42% uncertainty (twice the average of the within-study standard deviation). This period-to-period uncertainty is due to uncertainties in the bLS model, uncertainty in the idealized representation of the wind, noisy C observations, and so on, uncertainties that are reduced by appropriate averaging.

Table A2 compares nine studies where trace-gas emissions were determined using the bLS analysis procedure and alternative techniques (true emissions were unknown). The alternative techniques include the integrated horizontal flux and flux-gradient micrometeorological techniques and a SF_6 ruminant tracer technique (SF_6).

Acknowledgments

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Table A1. Tracer studies using the backward Lagrangian Stochastic (bLS) analysis procedure. These studies evaluated emissions in both ideal wind conditions (homogeneous terrain) and in winds disturbed by obstacles, from either single or multiple sources, and with either point or line-average concentration (*C*) measurements. Each study was based on many measurement periods (typically 15 min each), and the average gas recovery and standard deviation of the recovery are presented. Well-described data-filtering criteria were used in each study to reduce bLS errors. For studies with wind obstacles, the downwind distance (*M*) between the *C* measurement and the upwind obstacle is given in terms of the obstacle height (*h*).

Gas	Recovery ± SD	Site characteristics	Reference
	%		
CH_4	102 ± 22	Grass, no obstructions, line-average *C*	Flesch et al. (2004)
CH_4	98 ± 20	Grass, obstructions ($M > 5h$), line-average *C*	Flesch et al. (2005a)
CH_4	93 ± 14	Grass, no obstructions, multi-source,† line-average *C*	Gao et al. (2008)
CH_4	98 ± 11	Grass, obstructions ($M > 10h$), multisource,‡ line-average *C*	Gao et al. (2008)
CH_4	106 ± 16	Grass, no obstructions, line-average *C*	McBain and Desjardins (2005)
CH_4	99 ± 20	Grass, obstructions ($M > 5h$), line-average *C*	McBain and Desjardins (2005)
SF_6	100 ± 29	Whole-farm, dairy ($M > 9h$), line-average *C*	McGinn et al. (2006)
CH_4	86 ± 17	Whole-farm, dairy ($M > 9h$), line-average *C*	McGinn et al. (2006)
CH_4	103 ± 16	Grass, no obstructions, line-average *C*	Gao et al. (2009a)
CH_4	99 ± 30	Grass, no obstructions,§ line-average *C*	Gao et al. (2009b)
CH_4/CO_2	99 ± 29	Wheat stubble, no obstructions, line-average *C*	Loh et al. (2009)
CH_4	98 ± 18	Barn source ($M \geq 10h$), line-average *C*	Gao et al. (2010)
Average	98 ± 5		
Average SD (within-studies)	21 ± 6		

† Results for trial C1 are given here—simplest of the multisource layouts studied. Recovery is calculated based on the summation of all emission sources.

‡ Results for trial C2 are given here—simplest of the layouts studied, but with obstacles placed around the sources. Another trial in this study used higher obstacles ($M \sim 5h$), and the recovery was reduced to 0.75. Recovery is calculated based on the summation of all emission sources.

§ Reported recovery in this study was 109 ± 45. The high standard deviation (45%) was due to the experimental design (4 laser lines, solving for C_b [background tracer concentration] and *Q* [uniform gas emission rate]) which led to ill-conditioned solutions. We report the recovery calculated when periods with very large uncertainties are removed (as discussed in Gao et al., 2009b).

Table A2. Studies comparing the backward Lagrangian stochastic (bLS) analysis procedure with alternative techniques.

Gas	Differences	Site characteristics	Reference
Metolachlor	bLS 6% higher than IHF†	Bare soil, no obstructions, point *C*‡	Flesch et al. (2002)
CH_4	bLS ≈ 10% higher than IHF	Cows in pasture (enteric emissions), line-average *C*	Laubach and Kelliher (2005a)
CH_4	bLS 14% higher than IHF	Cows in pasture (enteric emissions), point *C*	Laubach and Kelliher (2005b)
CH_4	bLS 3 and 18% higher than IHF (2 sensor configurations, all fetches)	Cows in paddock (enteric emissions), point and line-average *C*	Laubach et al. (2008)
Chemical	bLS 2% higher than IHF	Short grass, point *C*	Flesch et al. (1995)
NH_3	bLS 22 to 36% higher than FG§	Commercial beef feedyard, point *C*	Todd et al. (2007)
NH_3	two bLS configurations were +10% and –5% of IHF calculation	Field emissions of manure slurry, point *C*	Sanz et al. (2009)
SF_6	bLS 7% less than SF_6	Cows in small pens (enteric emissions), line-average *C*	McGinn et al. (2009)
NH_3	For all measurements bLS averaged 19% higher than IHF. For cumulative emissions, bLS was within 5% of IHF.	Urea volatilization from wheat crop, 25-m radius circular plot, point *C*.	Turner et al. (2010)

† IHF, integrated horizontal flux.

‡ *C*, concentration.

§ FG, flux-gradient.

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ATTACHMENT 44

United States Department of Agriculture

Natural Resources Conservation Service

CONSERVATION PRACTICE STANDARD

ANAEROBIC DIGESTER

Code 366

(No.)

DEFINITION

A component of a waste management system in which biological treatment breaks down animal manure and other organic materials in the absence of oxygen.

PURPOSE

This practice is applicable for one or more of the following purposes:

- manage odors
- reduce the net effect of greenhouse gas emissions
- reduce pathogens
- captures biogas to facilitate energy production

CONDITIONS WHERE PRACTICE APPLIES

This practice applies where:

- Biogas production and capture are components of a waste management system plan and Comprehensive Nutrient Management Plan (CNMP)
- Sufficient and suitable organic feedstocks are readily available.

CRITERIA

General Criteria Applicable to All Purposes

Laws and Regulations. Plan, design, and construct the anaerobic digester to meet all federal, state, local and tribal laws and regulations.

Location. Locate the anaerobic digester outside the 100-year floodplain unless site restrictions require locating it within the floodplain. If located in the floodplain, protect the facility from inundation or damage from a 25-year flood event. Additionally, follow the policy found in the NRCS General Manual (GM) 190, Part 410.25, Floodplain Management, which may require providing additional protection for structures located within the floodplain.

Feedstock Characteristics. Digester design must take into account the varying feedstock properties. Depending on the system design, extraneous material such as soil, sand, stones or fibrous bedding material (including clumps of straw), may need to be ground, removed, reduced, or otherwise handled. Ensure that the total solids of feedstock influent match the digester type and process design. Exclude excess water and foreign material from the digester. Food waste, wastewater from food processing operations, and other allowable organic substrates may be added as supplemental feedstock to a digester when the digester is designed to treat such wastes, as described in the operation and maintenance plan.

Connections. Ensure that all connections and fittings are properly sized and installed for design flows and vibrations.

Agricultural Waste Management System. Do not consider the volume of the digester in determining the storage requirement of the waste storage facility.

Safety. If the digester has the potential to create a safety hazard, install fence and post warning signs to prevent using it for purposes other than intended. Include appropriate safety features to minimize the hazards of the facility (refer to American Society of Agricultural and Biological Engineers (ASABE) Standard EP470, Manure Storage Safety for guidance, as needed).

Biogas is flammable, highly toxic, and potentially explosive. For the design of the digester and gas components, including the gas collection, control, and utilization system and understand the hazards associated with normal operation and maintenance, provide adequate safety measures including appropriate earthquake loads (as required)**,** and install components in accordance with standard engineering practice for handling a flammable gas and to prevent undue safety hazards. As a minimum:

- Post “Warning Flammable Gas” and “No Smoking” signs.
- Provide appropriate fire protection equipment and biogas leak detection sensors, especially in confined areas.
- Install a flare for the anaerobic digestion system unless another option is provided by the manufacturer which adequately addresses conditions to prevent biogas release into the atmosphere.
- Locate flares the appropriate distance from the digester and other buildings according to manufacturer’s specifications, and electrical code. Install flares a minimum of 10 feet above the ground. Locate open flares a minimum distance of 50 feet from the biogas source. Properly ground flares or protect to minimize potential damage caused by lightning strikes.
- Provide a flame trap device in the biogas line between the digester and sources of ignition to prevent flame migration from the flare to the gas source or as otherwise recommended by the flame arrester manufacturer.
- Use explosion-proof motors, switches and other spark producing devices on all biogas blowers or other equipment installed where biogas is present.
- Provide and maintain above ground permanent markers to indicate the location of underground gas lines to prevent accidental disturbance or rupture. Mark exposed pipe to indicate type of pipe whether gas or other.

Criteria for Plug Flow Digester

- Recommended total solids content of influent is 11 to 14 percent.
- Minimum digester retention time is 20 days.
- Operational temperature is mesophilic (ranging from 95 to 104 ºF).
- Minimum length to width ratio of digester flow path is 3.5:1
- Maximum ratio of flow path width to fluid depth is 2.5:1.
- Design the floor and wall shapes to facilitate the movement of all material through the digester to minimize short-circuiting flow.

Criteria for Complete Mix Digester

- Recommended total solids content of manure influent is less than 11 percent.
- Minimum digester retention time is 17 days.
- Operational temperature is mesophilic (ranging from 95 to 104 ºF).
- Provide appropriate devices, as necessary, to assure a continuous flowing and mixing process.

Criteria for Covered Lagoon

Meet the “General Criteria for All Lagoons” given in Practice Standard 359, Waste Treatment Lagoon, as appropriate. Additional requirements include:

- Minimum Design Operating Volume. Base the design operating volume either on the daily volatile solids (VS) loading rate per 1,000 ft^3 or the minimum hydraulic retention time (HRT) adequate for methane production, whichever volume is greater. Select and apply the maximum daily VS loading rate from the values listed on the map in Figure 1. Select and apply the minimum HRT from values indicated on the map in Figure 2.
- Required Total Volume. The required total volume of the digester is equal to the minimum design operating volume except where waste storage is also included in the design. For this exception, meet the additional criteria pertinent to volume requirements found in the Design Storage Volume in Practice Standard 359, Waste Treatment Lagoon, as appropriate.
- Provide a minimum of 2 feet of freeboard above the digester design water surface; if rainfall is included in determining the operating volume, only 1 foot of freeboard is required. The digester storage volume does not need to account for rainfall for completely covered digesters.
- Operating Depth. The minimum operating depth of the digester is 8 feet.
- Inlet and Outlet. Locate the inlet and outlet devices as far apart as practical to minimize “short circuiting.” Locate the inlet discharge a minimum of 12 inches below the digester liquid surface. Equip the digester with an outflow device that maintains the digester liquid surface at its design operating level.
- Digester Cover. Design the digester cover, materials, anchorage, and all appurtenances, such as weights and floats, to capture and convey biogas to the gas collection system. The digester cover and associated materials must meet the requirements of Practice Standard 367, Roofs and Covers.

Criteria for Alternative Type Digester

For digester types not meeting the above criteria or for digester types other than listed in this standard (such as fixed film, induced blanket, high solids (dry digesters) or thermophilic reactors) follow the documented design and performance requirements of the proposed anaerobic digester.

Alternative Type Digester Containment Characteristics. For the various alternative digester types, ensure that the following applicable criteria are applied:

- For earthen structures meet the “General Criteria for All Lagoons” given in Practice Standard 359, Waste Treatment Lagoon, as appropriate.
- Design tanks and internal components, including heat pipes to facilitate periodic removal of accumulated solids and for corrosion protection.
- For tanks meet the structural criteria for “Fabricated Structures” in Practice Standard 313, Waste Storage Facility, and the requirements of state and local seismic codes as applicable.
- The following additional criteria apply:
  - Design Operating Volume. Size the digester to retain the design requirements to meet the hydraulic and solids retention times (days).
  - Inlet and Outlet. Locate the inlet and outlet devices to facilitate process flow. Design inlets and outlets of any permanent material to resist corrosion, plugging, freeze damage, and prevent gas loss. To maintain the operating level, maintain a gas seal under the cover, prevent gas loss, and release effluent directly to separation, storage, or other treatment facility. Equip the digester with an outflow device, such as an underflow weir.
  - Cover. For covers meet the requirements of Practice Standard 367, Roofs and Covers. Equip tanks with suitable covers designed for accumulation and collection of biogas.

- o Heating System (if required). Design and install the heating system to maintain proper digester temperature and to minimize corrosive attack and scalding build-up on the heated surfaces.

Gas Collection, Transfer, and Control System. Design the biogas collection, transfer, and control system to convey captured gas from within the digester to gas utilization equipment or devices (flare, boiler, engine, etc.).

- Gas collection and transfer – Meet the following for pipe and/or appurtenances:
  - o Design the gas collection system within the digester to minimize plugging or install cleanout ports as needed.
  - o Securely anchor pipe and components within the digester to prevent displacement/damage from normal forces including loads associated with scum accumulated.
  - o Design the collection and transfer pipe for wet biogas. In colder climates, protect the pipe as necessary to prevent frost buildup. Use pipe sizes no smaller than 3-inch diameter, unless a detailed design is performed to account for frost buildup and pressure drop in a low-pressure system. Design pressurized systems as an Alternative Type Digester.
  - o For pipes used to transfer biogas include provisions for drainage of condensate, pressure and vacuum relief, and flame traps.
  - o For steel pipe meet the requirements of AWWA Specification C-200 or ASTM A53/A211 for stainless steel.
  - o For plastic pipe meet the requirements of AWWA Specification C-906 or ASTM D-3350 for HDPE.
  - o Install pipes to ensure all sections can be safely isolated and cleaned as part of routine maintenance.
- 2. Gas Control
  - Locate and shelter all equipment and components from the elements.
  - The minimum service life for all equipment and components is 2 years or more. Provide easy access for replacement or repair of components.
  - Base the size of equipment and connecting pipe on head loss, cost of energy, cost of components, and manufacturers' recommendations.
  - Where electrical service is required at the control facility, follow the National Electrical Code and local and state requirements for the installation and all electrical wire, fixtures, and equipment.

Gas Utilization. Design and install gas utilization equipment in accordance with standard engineering practice and the manufacturer's specifications.

- Equip flares with automatic ignition and powered by battery/solar or direct connection to electrical service. Ensure that the flare capacity is equal to or greater than the anticipated maximum biogas production. Install a windshield or other device to protect an open flare against wind.
- As needed, design appropriate facilities to store excess gas.
- Design gas-fired boilers, fuel cells, turbines, and internal combustion engines to burn biogas directly, in a mix with other fuel, or include equipment for removing hydrogen sulfide and other contaminants from the biogas.
- Install and maintain a gas meter, suitable for measuring biogas.

Monitoring for mesophilic and thermophilic digesters. Install equipment needed to properly monitor the digester and gas production as part of the system. As a minimum the following equipment is required:

- Temperature sensors and readout device to measure internal temperature of digester

- Temperature sensors and readout device to measure inflow and outflow temperature of digester heat exchanger

CONSIDERATIONS

Location. Locate the digester as near the source of manure and as far from neighboring dwellings or public areas as practicable. Consider slope, distance of manure transmission, vehicle access, prevailing wind direction, proximity to hydrologically sensitive areas, and visibility for proper location. Locate the digester near a suitable site for energy utilization equipment. Minimize distances for the transmission of biogas through buried pipe. Locate the waste storage facility, considering elevation and distance from the digester, to take advantage of gravity flow.

Manure Characteristics. Consider using only fresh manure which has the highest energy content. The biogas yield from aged manure (generally less than 6 months old) is dependent on the biodegradation that has taken place during the storage period. Little biodegradation occurs when frozen. Manure stored in a warm, moist state could be significantly degraded resulting in reduced biogas production.

Chemicals and Amendments. Consider potential inhibitory effects on gas production of any antimicrobial agents in the manure or waste stream.

Waste Separation. Consider waste separation to prepare the waste stream for introduction to the anaerobic digester or for post-digestion treatment.

Collection/Mix Tank. Consider using a collection/mix tank to accumulate manure, settle and separate foreign material, pre-heat, and/or pre-treat influent waste to the appropriate total solids concentration. A volume of 1 to 3 days of manure collection, depending on the planned system management, is often used.

Overflow Protection. In case of digester equipment failure, consider designing the transfer system with the capability to bypass the digester, going directly to storage or land application equipment.

Digester Type. The type of digester selected may be affected by geographical location (Figure 3), energy considerations, wastewater properties, and other design considerations (Figure 4).

Digester Design. A digester operating fluid depth of 8 feet or greater is usually more economical for tank design. Tank dividers or flow separators may be utilized to increase efficiency and prevent short-circuiting. Install interior slopes as steep as permitted by soil properties and construction techniques.

Grounding and Cathodic Protection. Stray voltage, electrolysis and galvanic corrosion can damage pipes inside digesters. Consider the design requirements for electrodes and anodes.

Electrical Component Protection. Very small concentrations of biogas can corrode electrical hardware. Consider locating electrical controls in a separate room or building away from the digester and generator.

Temperature Maintenance: For the design include a means of maintaining the digester within acceptable operating temperature limits, use insulation where appropriate.

Gas Transfer Pipe. Exposed pipe conveying flammable gas is generally painted yellow, per IAW ASME A13.1-2015.

Gas Collection Cover. In areas of extreme wind or excessive snow, consider installing structures to protect inflatable and floating digester covers from damage.

Air Quality. Recovering energy from the biogas may be a preferable alternative to flaring. This could reduce fossil fuel combustion and associated emissions, thereby reducing the net effect of greenhouse gases and improving air quality. Some energy recovery options, such as the use of internal combustion engines to convert biogas to energy, may also result in additional emissions of some air pollutants.

Gas Utilization. Investigate and select the most beneficial and economical use of the biogas energy. Sales of carbon credits may affect the manner of utilization. Depending on the design and climate, digesters may require more than 50 percent of the biogas heat value to maintain the design temperature in the winter. Digesters can be heated by hot water from boilers burning biogas or by heat recovery from internal combustion engines and micro turbines burning biogas for power generation.

Effluent Tank. Due to the potential use of digested separated solids for bedding or soil amendment consider utilizing an effluent tank to hold digester effluent for subsequent mechanical solid-liquid separation.

Siting and Vegetation. Analyze the visual impact of the digester within the overall landscape context and effects on aesthetics. Consider screening with vegetative plantings, landscaping, or other measures to alleviate a negative impact or enhance the view. In addition, vegetate disturbed areas as soon as possible.

Soil Properties. Consider soil properties such as texture, saturated hydraulic conductivity, flooding, slope, water table and depth, as well as limitations related to seepage, corrosivity, or compactability of soil material when designing and installing an anaerobic digester. Refer to local soil survey information and on-site soil investigations during planning.

Nutrient Availability. Consider the effects of digestion upon nutrient availability. Land application of digester effluent, compared with fresh manure, may have a higher risk for both ground and surface water quality problems. Compounds such as nitrogen, phosphorus and other elements become more soluble due to anaerobic digestion and therefore have higher potential to move with water.

PLANS AND SPECIFICATIONS

Prepare plans and specifications that describe the requirements for applying the practice according to this standard. As a minimum, include—

- Plan view of the system layout and location of livestock facilities, waste collection points, waste transfer pipe, digester, biogas utilization facilities, and digester effluent storage. Include utilities and structures on the site.
- Grading plan showing excavation, fill, and drainage, as appropriate.
- Materials and structural details of the digester, including all premixing tanks, inlets, outlets, pipes, concrete, pumps, valves, and appurtenances as appropriate for the complete system.
- Foundational requirements including preparation and treatment.
- Details of biogas collection, control, and utilization system including type of materials for pipe, valves, regulators, pressure gages, electrical power and interface as appropriate, flow meters, flare, utilization equipment, and associated appurtenances.
- Specify insulation, heat exchanger capacity, and energy requirements as appropriate for maintaining the digester operating temperature within acceptable limits.
- Provide a process flow diagram with the following design information:
    - Flow rates of influent, effluent, and biogas.
    - Design total and volatile solids content of influent and effluent.
    - Digester volume.
    - Hydraulic and solids retention times.
    - When applicable, heating system type and capacity, control, and monitoring.
    - Biogas production, including methane yield.
    - 12-month energy budget when applicable.
    - Safety features

OPERATION AND MAINTENANCE

Prepare an operation and maintenance plan for the operator.

As a minimum, include the following items in the operation and maintenance plan:

- Proper loading rates of the digester and total solids content of the influent.
- Accounting for the nutrient impact of all feedstock in the farm's nutrient management plan.
- Proper operating procedures for the digester.
- Estimates of biogas production, methane content, and potential energy recovery.
- Description of the planned startup procedures, normal operation, safety issues, and normal maintenance items.
- Alternative operation procedures in the event of equipment failure.
- Instructions for safe use and flaring of biogas.
- Digester and other component maintenance.
- Troubleshooting guide.
- Monitoring plan with frequency of measuring and recording digester inflow, operating temperatures,

biogas yield, and/or other information as appropriate.
- Maintain the internal temperatures for controlled temperature digesters as appropriate to the digester type and design. For mesophilic digesters maintain the temperature between 95 ºF and 104 ºF with an optimum of 100 ºF and daily fluctuation of digester temperature limited to less than 1 ºF.
- Design the digester with appropriate freeboard and overflow or automatic shutdown devices to prevent accidental spillage of effluent or discharge into the gas collection system.
- Establish and maintain emergency contact information for consultation with qualified experts.

REFERENCES

American Society of Agricultural and Biological Engineers (ASABE), Standard EP470, Manure Storage Safety, 2011.

Agricultural Waste Management Field Handbook, USDA-NRCS.

American Water Works Association (AWWA), Standards.

American Society for Testing and Materials (ASTM), Standards.

United States Department of Agriculture (USDA), NRCS, Agricultural Waste Management Field Handbook.

United States Department of Agriculture (USDA), NRCS, General Manual.

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ATTACHMENT 45

Summary of Expert Opinions
Dr. Shane W. Rogers
September 10, 2021

1. **The lagoon and spray swine waste manure management system used at swine concentrated animal feeding operations (CAFOs) in North Carolina leads to negative environmental and public health consequences.**[1]

   A. The U.S. Environmental Protection Agency defines an Animal Feeding Operation (AFO) as one in which animals have been, are, or will be stabled or confined and fed or maintained for a total of 45 days or more in any 12-month period, and crops, vegetation, forage growth, or post-harvest residues are not sustained in the normal growing season over any portion of the lot or facility."[2] Concentrated Animal Feeding Operations (CAFOs) are generally identified as the largest of the AFO operations owing their potential for significant pollution. Large swine CAFOs are those operations that confine at least 2,500 swine weighing over 55 pounds, or that confine more than 10,000 swine weighing less than 55 pounds.[3]

   B. It takes about 24 to 29 weeks for a hog to reach an average market weight of 283 pounds from birth, including about 3 weeks to wean, 6 weeks in a nursery stage, and then 16 to 20 weeks for the finishing stage.[4] A typical finishing operation for Smithfield may have 2.5 sellouts (rotation of hogs) per year. To get from birth to 283 pounds in 200 days means that the hogs must gain weight rapidly; industry reports reflect average daily weight gains of as much as 1.95 pounds as of the year 2001.[5] This requires significant food and water intake, and with that comes significant manure production. Based on reported manure production characteristics, hogs produce multiple times more waste than humans. As shown in Table 1, a 283-pound finishing hog can be expected to produce on average about 1.67 gallons (13.8 pounds) of manure each day.[6]

   C. Most swine CAFOs in North Carolina, including (at present) the subject swine CAFOs, use a lagoon and spray system of manure waste management. There are three primary

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[1] Swine CAFOs may also be referred to as industrial hog operations in some scientific literature.
[2] See https://www.epa.gov/npdes/animal-feeding-operations-afos
[3] According to the U.S. EPA regulatory definitions of Large CAFOs, Medium CAFOs, and Small CAFOs, swine AFOs that are smaller than this size threshold may also be designated a CAFO owing a manmade ditch or pipe discharge of manure or wastewater to surface water, if there is direct contact of animals with surface water that passes through the confinement area, or if the permitting authority finds an operation to be a significant contributor of pollutants. See https://www.epa.gov/npdes/npdes-afos-policy-documents-0 for more details.
[4] United States Department of Agriculture, Economic Research Service, web page, Hogs and Pork, available at http://www.ers.usda.gov/topics/animal-products/hogs-pork/background.aspx (chart showing that it may take 2 to 3 weeks to wean a pig, then nursery stage for 6 weeks, then 16 to 20 weeks for the finishing stage = 24 to 29 weeks total = 168 to 203 days).
[5] National Hog Farmer, Tracking Progress in Grow-Finish, article dated October 15, 2002, available at http://nationalhogfarmer.com/mag/farming_tracking_progress_growfinish.
[6] Midwest Plan Service, 2004. "Manure Characteristics", Manure Management Systems Series, MWPS-18 Section 1, second edition, Iowa State University, Ames, IA.

sources of concern regarding pollution in this system that include the swine housing facilities, anaerobic lagoons, and spray application of swine wastes.

Table 1. Manure production characteristics of swine[7]

Life Stage	Animal size	Daily Manure Production		
	Pounds	Pounds	Gallons	Liters
Nursery	25	1.9	0.23	0.87
	40	3.0	0.37	1.40
Finishing	150	7.4	0.89	3.37
	180	8.9	1.07	4.05
	220	10.9	1.31	4.96
	260	12.8	1.55	5.87
	300	14.8	1.79	6.78
Gestating	300	6.8	0.82	3.10
	400	9.1	1.10	4.16
	500	11.4	1.37	5.19
Lactating	375	17.5	2.08	7.87
	500	23.4	2.78	10.5
	600	28.1	3.33	12.6

Animals housed in swine CAFOs continually defecate. As swine manure waste is produced in the swine houses, it falls, or is pushed through (by the animals), slots in the concrete floor into pits or flush lanes under the floor. Removal of manure waste from underfloor of the swine houses occurs 4-8 times per day (typical). This is accomplished by flushing the manure that accumulates under the slotted portion of the floors out to the lagoon using wastewater drawn from near the top of the anaerobic lagoon.

D. An anaerobic lagoon is an in-ground manure holding structure, commonly left uncovered and open to the air. The subject swine CAFO sites have historically relied on open lagoons to store manure. Aside from rain that may fall over the anaerobic lagoons, no additional water aside from that contained in swine manure is typically added to the lagoon - by design. The high waste load rapidly depletes oxygen, and thus decomposition of the waste occurs under anaerobic conditions. Anaerobic bacteria in well-functioning anaerobic lagoons break down and stabilize the organic fraction of materials. When this process is upset due to a number of potential factors such as overloading with wastes, solids accumulation and reduction of treatment volume, pH changes or other upsets, the stabilization process is greatly reduced or eliminated.

E. Anaerobic decomposition in anaerobic lagoons generates significant quantities of methane gas, carbon dioxide, ammonia, nitrous oxide, and other noxious gases that are emitted to the atmosphere along with other pollutants. Methane, a greenhouse gas, is a

[7] Source: Midwest Plan Service, 2004 Manure Characteristics, MWPS-18 Section 1, Second Edition, Jeff Lorimor, Wendy Powers, and Al Sutton (eds.). Manure Management Systems Series, Iowa State university, Ames, IA.

significant component of the biogas produced during anaerobic decomposition.[8] The quantity of ammonia nitrogen emitted is also large and of concern.[9] Its uncontrolled emission reduces the nitrogen content of the waste and poses a risk to the environment and public health. It has been estimated that 80-90% of the total ammonia emitted from livestock operations is redeposited uncontrolled within 10 km of the source, while the remainder is dispersed into the atmosphere contributing to the haze, acid rain, acidification of terrestrial and aquatic ecosystems, and eutrophication of surface water bodies.[10]

F. Swine manure wastes are held, exposed to the atmosphere, in anaerobic lagoons until such time they can be applied to land. Land application of manures is permitted in North Carolina so long as it is done in accordance with a certified animal waste management plan (CAWMP). The purpose of the CAWMP is to assure that manure is applied at a rate to meet, but not exceed, the nutrient requirements of the crop to receive the manure as a nutrient source. In general, CAWMPs consider application history and nutrients contents of the soil, nutrient content of manure, infiltration rates and application equipment used (effect on nutrient delivery to the plants), and land area over which the crop(s) will be grown. Like for most swine CAFOs in North Carolina, the subject swine CAFO sites are permitted to apply their manure waste at an agronomic rate for nitrogen. This does not assure that overapplication of phosphorus, an important nutrient that affects water quality, does not occur. It also does not assure that nitrogen and other pollutants in swine manure do not affect nearby water quality.

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[8] *See* The Humane Society of the United States, An HSUS Report: The Impact of Animal Agriculture on Global Warming and Climate Change. Available at http://www.hsi.org/assets/pdfs/farming_climate_impact.pdf (page 7 – "Storing and disposing vast quantities of manure can produce anthropogenic methane and nitrous oxide emissions. According to the Pew Center on Global Climate Change, farm animal manure management currently accounts for 25% of agricultural methane emissions in the United States and 6% of agricultural nitrous oxide emissions. As noted above, methane has 23 times the GWP of carbon dioxide, and its concentrations have increased by approximately 150% since 1750. Globally, farm animals are the most significant source of anthropogenic methane, responsible for 35-40% of methane emissions worldwide."

[9] See for example Szögi, A.A. and M. B. Vanotti (2007) Abatement of Ammonia Emissions from Swine Lagoons Using Polymer-Enhanced Solid-Liquid Separation, *Applied Engineering in Agriculture*, 23(6): 837-845; these authors measured for one anaerobic lagoon ammonia emissions of 13,633 kg /ha/yr. See also Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; these authors report from a sample of 40 reports of ammonia emissions measurements from swine waste anaerobic lagoons an average emission of 13,000 kg-N/ha/yr. See also Aneja, V.P., S.P. Arya, D.-S. Kim, I.C. Rumsey, H.L. Arkinson, H. Semunegus, K.S. Bajwa, D.A. Dickey, L.A. Stefanski, L.Todd, K.Motlus, W.P. Robarge, C.M. Williams (2008) Characterizing Ammonia Emissions from Swine Farms in Eastern North Carolina: Part 1 – Conventional Lagoon and Spray Technology for Waste Treatment, Journal of the Air & Waste Management Association, 58:1130-1144 ; these authors report ammonia emissions from three anaerobic lagoons were temperature dependent, ranging from as low as 2.2 kg-N/ha/d in the winter to as great as 57.8 kg-N/ha/d in the summer. See also Nkoa. R. (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review, *Agronomy for Sustainable Developmen*t, Springer Verlag/EDP Sciences/INRA, 34 (2):473-492. 10.1007/s13593-013-0196-z hal-01234816; on page 480 - "NH_3 emission inventories from several countries have shown that agriculture produces approximately 90% of the total emission of NH_3 to the atmosphere."

[10] See Nkoa. R. (2014) Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review, *Agronomy for Sustainable Developmen*t, Springer Verlag/EDP Sciences/INRA, 34 (2):473-492. 10.1007/s13593-013-0196-z hal-01234816, page 481.

G. When anaerobic manure lagoon effluents and solids are land-applied, nutrients, oxygen-demanding materials, metals, odors, particulates, endotoxins, bacteria, ammonia, and other harmful pollutants are released into the environment. Ammonia and other pollutants emitted to the atmosphere can move downwind or deposit uncontrolled onto land, physical structures, or waterbodies nearby. Land-applied pollutants can run off from fields in surface drainages, move via groundwater, or through tile drainage flows to affect water quality.[11] In a recent study of the U.S. Geological Survey, an overall measurable effect of swine CAFO waste manures on stream water quality in watersheds containing swine CAFOs was reported. Land application of waste manure at swine CAFOs influenced ion and nutrient chemistry in many of the North Carolina Coastal Plain streams that were studied, and the effect was directly related to higher swine barn densities and (or) higher total acres available for applying waste manure at swine CAFOs.[12]

H. In a lagoon and spray system, including the system permitted for the subject swine CAFOs, waste application is typically completed using methods of application that spray waste through the air, for example, using a pump and reel or center pivot spray applicator. Less frequently, operators may choose to use alternative application techniques such as low height or drag hose spreading or injection. The operators at the subject swine CAFOs use a combination of spraying waste and drag hose spreading.

I. Spraying swine manure through the air results in significant ammonia emissions. Direct injection of manure, such as can be accomplished with Aerway spreaders, is one alternative to spray irrigation that can reduce potential for off-site transport of harmful pollutants in manures or anaerobic digester effluents. Reduction in hydrogen sulfide emissions is estimated to be between 50-75%, but there may be a slight increase in greenhouse gas emissions (up to 10%) owing a slight increase in nitrous oxide emissions from decomposition by soil microbes. Injection also results in greater preservation of nutrients (up to 90% reduction in ammonia volatilization compared to spray application), which reduces the amount of anaerobic lagoon effluent that must be applied to land to meet crop nutrient requirements. Lower mass of anaerobic lagoon effluent applied per land area means less pollutant loading and emissions.

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[11] See, for example, (i) Harden, S.L., Rogers, S.W., Jahne, M.A., Shaffer, C.E., and Smith, D.G. 2012. Characterization of nutrients and fecal indicator bacteria at a concentrated swine feeding operation in Wake County, North Carolina, 2009–2011: U.S. Geological Survey Open-File Report 2012–1047, 31 p.; (ii) Rogers, S. and J. Haines. "Detecting and mitigating the environmental impact of fecal pathogens originating from confined animal feeding operations: review", U.S. EPA, NRMRL, EPA/600/R-06/021, September 2005. (iii) Harden, S.L. (2008) Microbial and Nutrient Concentration and Load Data During Stormwater Runoff at a Swine Concentrated Animal Feeding Operation in the North Carolina Coastal Plain, 2006-2007: U.S. Geological Survey Open-File Report 2008-1156, 22p., https://doi.org/10.3133/ofr20081156.
[12] Harden, S.L. (2015) Surface-water quality in agricultural watersheds of the North Carolina Coastal Plain associated with concentrated animal feeding operations: U.S. Geological Survey Scientific Investigation Report 2015-5080, 55 p., 7 apps., http://dx.doi.org/10.3133/sir20155080.

2. **Well-designed anaerobic digestion technologies for swine manure treatment can reduce select environmental pollution and public health risks of concern relative to anaerobic lagoons, but may exacerbate others.**

   A. Anaerobic digestion is a rapidly growing technology for farm waste management in the United States. The process of anaerobic digestion uses microbes to produce biogas, which is a mixture of methane, carbon dioxide, and trace gases. While anaerobic digestion occurs in an anaerobic lagoon, an anaerobic digester differs in that it is covered with an impermeable material rather than left open to the atmosphere and it is not exposed to diluting rainfall. Because anaerobic digesters are covered, the methane, a powerful greenhouse gas with high emissions from anaerobic lagoons, is captured rather than emitted to the atmosphere, thus reducing emissions that contribute to global warming. Owing the absence of diluting rainwater and other efficiencies, anaerobic digesters are more efficient at waste decomposition than anaerobic lagoons. Gas capture in anaerobic digesters may also reduce the amount of pathogens in the waste.

   B. Nutrients in manure are conserved during anaerobic digestion, but are converted to more readily available and mobile forms with higher potential to move with water.[13] Organic nitrogen is converted to ammoniacal nitrogen, which is not only inhibitory of the anaerobic digestion process, but can also result in higher ammonia emissions during subsequent storage if left uncovered, and during land application if not incorporated when applied.[14] For example, relative to anaerobic lagoons, ammonia emissions from anaerobically digested swine manure stored in open lagoons in one study increased by 46%. This was hypothesized to be caused by reduced methanogenesis and its reduced effect on the chemical conversion of ammonium to dinitrogen gas.[15]

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[13] See United States Department of Agriculture - Natural Resources Conservation Service, Conservation Practice Standard: Anaerobic Digester, Code 366, October 2017, 10 p. On page 6: "Consider the effects of digestion upon nutrient availability. Land application of digester effluent, compared with fresh manure, may have a higher risk for both ground and surface water quality problems. Compounds such as nitrogen, phosphorus, and other elements become more soluble due to anaerobic digestion and therefore have higher potential to move with water."

[14] See Westerman, P., M. Veal, J. Cheng, and K. Zering "Biogas Anaerobic Digester Considerations for Swine Farms in North Carolina" North Carolina State University, A&T University Cooperative Extension, 8 p.; See also Aguirre-Villegas, H., R.A. Larson, M.D. Ruark (2016) Dairy Anaerobic Digestion Systems and their Impact on Greenhouse Gas and Ammonia Emissions, Sustainable Dairy Fact Sheet Series, University of Wisconsin Extension, 5 p. See also Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; p. 1 "Anaerobically digested slurry shows higher emissions during storage for NH_3 while losses tend to be lower for CH_4 and little changes occur for N_20 and CO_2 compared to untreated slurry. All cover types are found to be efficient for emission mitigation of NH_3 from stores."

[15] Harper, L. T.K. Flesch, K.H. Weaver, J.D. Wilson (2010) The effect of Biofuel Production on Swine Farm Methane and Ammonia Emissions, Journal of Environmental Quality, 39:1984-1992. In the abstract on pg 1984: "NH_3 emissions in the biofuel farms increased by 46% over the conventional farms. These studies show that what is considered an environmentally friendly technology had mixed results and that all components of a system should be studied when making changes to existing systems."

3. **Subject swine CAFO characteristics as currently operating and with modifications to the waste treatment system: minimal changes in the permits and certificates of coverage are not protective of air and water quality.**

A. Thousands of Smithfield hogs are being kept in each of the subject swine CAFOs as summarized in Table 2, along with the number and types of anaerobic lagoons at each site. Each of the subject swine CAFOs currently employ the lagoon and spray system of swine manure management. The subject swine CAFOs' irrigation records indicate that the predominant method of waste application is spraying using pump and reel and center pivots.[16] Murphy-Brown has indicated that there is also some limited use of drag hoses at these sites. Table 3 presents the basic characteristics of the proposed anaerobic digestion systems for each subject swine CAFO.

Table 2. Production characteristics and anaerobic lagoon characteristics of the subject swine CAFOs[17]

Facility	Operation type	Permitted hog counts	Number and type of anaerobic lagoons
Benson	Feeder to finish	6,120	One single stage
Goodson 2037/2038	Feeder to finish	20,992	One single stage and one 2-stage (primary and secondary)
Kilpatrick	Wean to Finish	13,336	One 2-stage (primary and secondary)
Merritt	Wean to Finish	5,083	One single stage
M&M	Wean to finish	12,308	Three single stage

Notes: Permitted hog count is the allowable annual average count calculated pursuant to the permit, not the number of hogs present at any particular time.

Table 3. Lagoon characteristics at each of the subject swine CAFOs[18]

[16] Data acquired from the most recent irrigation design plans provided including Benson: MB000788; Goodson 2037/2038: MB000127; Kilpatrick & Merritt: MB001143; M&M: MB001851. Document identification is by the first page number of the document.

[17] Data acquired from lagoon design information provided including Benson: MB000741; Goodson 2037/2038: MB000092 and MB000085; Kilpatrick & Merritt: MB001092 and MB001104; and M&M: MB001733, MB001741, and MB001748.

[18] Data acquired from the most recent nutrient utilization plans (NUPs), anaerobic digester permit applications, and permits or certificates of coverage issued by the DEQ provided for each of the subject swine CAFOs. These include (i) NUPs: Benson: MB000798; Goodson 2037/2038: MB000160 and MB000179; Kilpatrick: MB001154 and MB001184; M&M: MB001836; (ii) Permit / COC applications: Benson: MB000667; Goodson 2037/2038 MB000001; Kilpatrick: MB001016; M&M: MB001632; and (iii) DEQ Permits / COCs: Benson: MB000720; Goodson 2037/2038: MB000065; Kilpatrick: MB001090; M&M: MB001730 and MB001711. Note that the "MBXXXXXX" document identification is by the first page number of the document.

Facility	Proposed Anaerobic Digester (AD) Size *gallons*	New AD Construction or New anaerobic digester or Lagoon Cover	Proposed number of anaerobic digestate storage lagoons
Benson	1,693,156	New	One single stage, soil improved or clay lined
Goodson 2037/2038	5,044,114	New	One 2-stage, clay lined
Kilpatrick	9,121,716	Cover	One single stage, synthetically lined
Merritt	NA	NA	NA
M&M	2,672,790	New	One single stage synthetic, two single stage clay lined

NA = not applicable. Note that an anaerobic digester will not be installed to serve the Merritt farm, which will continue to be served by the existing single stage lagoon.

B. Reviewing the permit applications and permits issued by the North Carolina Department of Environmental Quality, it is evident that some discrepancies exist. For example, the DEQ issued permit for the Benson Farm indicates that one existing clay-lined lagoon will be used to store anaerobic digester effluent, whereas the permit application submitted by Cavanaugh & Associates on behalf of Smithfield Foods identifies the lagoon having a soil improved liner rather than a clay liner.[19] While investigation of historical satellite imagery catalogued on Google Earth could not reveal such detail, it does reveals an interesting feature of the lagoon, what looks to be a divide running through the center, the reduced volume for which is not reflected in the lagoon design information provided by Smithfield, and dated December 17, 2008.[20] Satellite imagery reported in Figure 1 dating from February 1998 through July 2018 clearly show a divide that is deep enough to affect water quality differences from one side of the lagoon to the other. Other discrepancies between permit applications and actual lagoon liner details for Goodson 2037/2038 and M&M Waters & Riverbank exist, but were corrected in the final permits.

[19] Compare the DEQ issued Permit no. AWI310039 dated March 31, 2021 (MB000720), page 3: "…consisting of a 1,693,156 gallon synthetically lined anaerobic digester with an 80 mil HDPE synthetic cover, one existing clay lined lagoon, one influent pump station…") to the permit application stamped MB000667 and dated 12-19-19, page 2, which identifies the existing lagoon having a soil improved liner.

[20] See the Benson Farm Anaerobic Waste Lagoon Design document dated 12/17/08 (starting on MB000741).

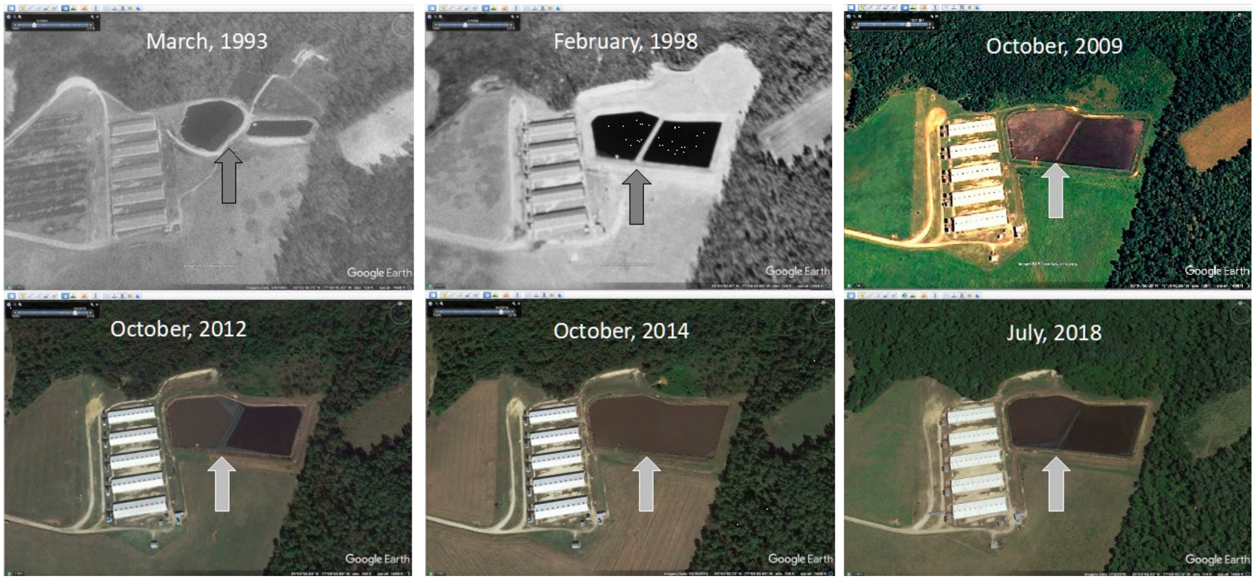

Figure 1. Google Earth historical satellite imagery of the Benson swine CAFO anaerobic lagoon showing a divide in the lagoon center (see February 1998) that roughly correlates to a historic drive (see March 1993), and which is large enough to also be visible and cause water quality differences between the two sides of the lagoon as seen in the subsequent images. This divide is not accounted or in the calculation of the lagoon design volume as reported in in documents provided by Smithfield (see Benson Lagoon Design, dated December 17, 2008 and identified MB000741 on page 1).

C. Permits in North Carolina prohibit siting of the swine CAFO housing units and waste treatment systems in the 100-year flood plain; the same is not true for swine manure application areas. These areas are subject to increased risk of leaching manure pollutants into the waters of North Carolina during and following rainfall. Figure 2 shows satellite imagery of the Goodson 2037 and 2038 swine CAFO and spray fields. As can be seen in the image to the left of this figure, the 100-year flood plain intersects portions of spray fields at this swine CAFO. Historical satellite images suggest that these spray irrigation fields are subject to flooding.

Figure 2. Satellite imagery of the Goodson 2037 / 2038 swine CAFO and manure spray fields. Left: overlay of the 100-year flood plain and the irrigation system map reveals spray irrigation of swine manure occurs in areas subject to flooding, increasing runoff potential of manure pollutants. Right: Historical satellite images from Google Earth reveal recurrent saturation of spray fields within the 100-year flood plain.

D. The permits issued by the North Carolina State Department of Environmental Quality authorize the waste treatment systems of the subject swine CAFOs to change; however, the permits included very few substantive changes to accommodate the new anaerobic digestion waste treatment systems, even though the new technology can alter the waste composition and nutrients as described above. The subject swine CAFOs must continue to monitor their manure wastes as usual and adjust application rates based upon either total nitrogen or total Kjeldahl nitrogen results. Notably, increased ammonia volatilization from lagoon storage of anaerobically digested manure may increase uncontrolled air emissions of ammonia nitrogen, reducing the nitrogen content of the waste to be applied, and allowing for increased concentration of waste application considering that crop nutrient requirements will not change. Owing to changes in the form of the nutrients, increased mobilization may occur. The permits issued for Goodson 2037 & 2038 and Benson swine CAFOs mandate two years of quarterly monitoring and reporting of influent and effluent total nitrogen, total Kjeldahl nitrogen, nitrate nitrogen, ammonium nitrogen, total phosphorus, copper, sulfur, zinc, and fecal coliform bacteria. The nutrient management plans must be modified as needed based on performance of the anaerobic digestion system, and if performance is not as predicted, immediate measures must be taken. Regardless, these measures will only monitor changes in composition at the source. Notably, permits for swine CAFOs in North Carolina, including the four at issue here, do not generally require monitoring of nearby groundwater, surface water, or air emissions to detect potentially increased pollutant mobilization, and thus associated air and water quality degradation by such changes will go undocumented and unaddressed.

E. Other changes in the permits or Certificates of Coverage are not clearly related to changes in the swine manure waste treatment technology. For example, changes in the Certificate of Coverage at the Kilpatrick & Merritt and the permit for Benson swine CAFOs include a directive to review the facilities' CAWMP with respect to land application areas in landscape positions that are in close proximity to public roads, dwellings, and wells, and provide within 180 days a report to the division to describe what, if any, additional BMPs are to be implemented in these areas to improve protections and further reduce risk of off-site impacts. Nothing more is required than a paper exercise. The permit for the Benson swine CAFO also includes a mandate to limit application on fields with a high phosphorus loss assessment rating to crop nutrient requirements for phosphorus, and restricts swine manure application on fields with very high phosphorus loss assessment rating.

4. Biogas-compatible technologies that could mitigate environmental impacts from the subject swine CAFOs by decreasing nutrients in land-applied waste and reducing ammonia emissions are available.

A. Considerable progress has been made in manure treatment technology development in the last 24 years. There are several technologies readily available and in development that can reduce significantly gaseous emissions from anaerobically digested swine CAFO manure. At a basic level, impermeable covers could be installed on the lagoons that will store anaerobic digestate and reduce ammonia volatile losses from the lagoon by nearly 90%.[21] Volatile losses of ammonia can be reduced 70-90% upon application by injection of the anaerobically digested swine manure waste into the ground rather than spray applying. Conservation of the nitrogen in the manure rather than emission into the atmosphere can reduce nearby effects of uncontrolled ammonia deposition as well as public health risks.

B. Other treatment technologies are also available and currently used at other swine CAFO operations that can remove or recover nutrients, significantly reducing environmental and public health risks. One high achieving set of developments stemmed from the Super Soils / Terra Blue technology developed in North Carolina under the Smithfield Agreement.[22] Since that time, advancements from the research group have included development of more efficient two-stage and then one-stage ANAMMOX bioreactors to replace nitrification/denitrification and gas permeable membranes to recover ammonia and phosphate minerals from swine wastewater.[23] An advanced waste treatment technology developed in a joint venture with Embrapa (Brazil) known as Sistrates has successfully implemented the Terra Blue technology with anaerobic digestion at a 9,500 head swine CAFO in Brazil.[24] Swine waste anaerobic digestion-compatible membrane separations technologies have also matured, such as that of Digested Organics LLC which can recover nutrients in higher value streams and simultaneously produce water of sufficient quality for livestock.[25]

5. Conclusion.

A. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that the anaerobic digestion systems to be implemented at the subject swine CAFOs are more likely than not to exacerbate ammonia emissions relative to the current anaerobic lagoon and spray system. Resulting unregulated emissions and

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[21] See Kupper. T., C. Häni, A. Neftel, C. Kincaid, M. Bühler, B. Amon, A. Vanderzaag (2020) Ammonia and greenhouse gas emissions from slurry storage – A review, Agriculture, Ecosystems and Environment, 300: 106963; Table 11, page 10.

[22] Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, F.J. Humenik (2007) Development of environmentally superior treatment system to replace anaerobic swine lagoons in the USA, *Bioresource Technology*, 98(17):3184-94. doi: 10.1016/j.biortech.2006.07.009.

[23] See Vanotti, M.B., K.S. Ro, A.A. Szogi, J.H. Loughrin, P.D. Millner (2018) High-rate solid-liquid separation coupled with nitrogen and phosphorus treatment of swine manure: Effect on water quality, *Frontiers in Sustainable Food Systems* 2:49. https://doi.org/10.3389/fsufs.2018.00049; see also Vanotti, M.B., P.J. Dube, A.A. Szogi, M.C. Garcia (2017) Recovery of ammonia and phosphate minerals from swine wastewater using gas-permeable membranes. *Water Research*, 112:137-146.; see also Magri, A., M.B. Vanotti, A.A. Szogi (2012) Anammox sludge immobilized in polyvinyl alcohol (PVA) cryogel carriers, *Bioresource Technology*, 114(2):231-240.

[24] See Tápparo, D.C., D. Cândido, R.L. Radis Steinmetz, C. Etzkorn, A. Cestonarodo Amaral, F. Goldschmidt Antes, A. Kunz (2021) Swine manure biogas production improvement using pre-treatment strategies: Lab-scale studies and full-scale application, Bioresource Technology Reports, 15:100716 (8p.)

[25] For more details, see https://digestedorganics.com/manure-and-digestate-management/.

increased nutrient mobilization are more likely than not to negatively affect environmental quality and public health.

B. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that the subject permits authorizing swine CAFO waste management systems known to change the nature of the waste material in ways that may exacerbate water quality impacts from land application, and increase atmospheric ammonia emissions, fail to prevent cumulative effects on water quality and allow adverse environmental impacts.

C. For all of the reasons stated above, it is my opinion, within a reasonable degree of scientific certainty, that technologies to abate these pollution sources are readily available.

D. All of my opinions are expressed to a reasonable degree of scientific certainty.

ATTACHMENT 46

frontiers in Bioengineering and Biotechnology

ORIGINAL RESEARCH
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Life Cycle Environmental Impacts of Electricity from Biogas Produced by Anaerobic Digestion

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The aim of this study was to evaluate life cycle environmental impacts associated with the generation of electricity from biogas produced by the anaerobic digestion (AD) of agricultural products and waste. Five real plants in Italy were considered, using maize silage, slurry, and tomato waste as feedstocks and cogenerating electricity and heat; the latter is not utilized. The results suggest that maize silage and the operation of anaerobic digesters, including open storage of digestate, are the main contributors to the impacts of biogas electricity. The system that uses animal slurry is the best option, except for the marine and terrestrial ecotoxicity. The results also suggest that it is environmentally better to have smaller plants using slurry and waste rather than bigger installations, which require maize silage to operate efficiently. Electricity from biogas is environmentally more sustainable than grid electricity for seven out of 11 impacts considered. However, in comparison with natural gas, biogas electricity is worse for seven out of 11 impacts. It also has mostly higher impacts than other renewables, with a few exceptions, notably solar photovoltaics. Thus, for the AD systems and mesophilic operating conditions considered in this study, biogas electricity can help reduce greenhouse gas (GHG) emissions relative to a fossil-intensive electricity mix; however, some other impacts increase. If mitigation of climate change is the main aim, other renewables have a greater potential to reduce GHG emissions. If, in addition to this, other impacts are considered, then hydro, wind, and geothermal power are better alternatives to biogas electricity. However, utilization of heat would improve significantly its environmental sustainability, particularly global warming potential, summer smog, and the depletion of abiotic resources and the ozone layer. Further improvements can be achieved by banning open digestate storage to prevent methane emissions and regulating digestate spreading onto land to minimize emissions of ammonia and related environmental impacts.

Keywords: agricultural waste, anaerobic digestion, biogas, electricity, life cycle assessment, renewable energy

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INTRODUCTION

The need to mitigate climate change and improve security of energy supply is driving a growing interest in renewable energy sources, with many world regions and countries setting ambitious targets. For example, the EU directive on the promotion of the use of energy from renewable sources (EC, 2009) sets the target of achieving a 20% share of energy from renewable resources by 2020, including biogas produced by anaerobic digestion (AD) of agricultural feedstocks.

Production of biogas is expanding rapidly in Europe. According to EurObserv'ER (2014), about 13.4 million ton oil equivalent (Mtoe) of biogas primary energy was produced in the EU during 2013, a 10% increase on the 2012 levels. Germany is the largest producer of biogas, not only in Europe but also in the world. In 2013, it had 7874 AD plants with a total installed electrical capacity of 3384 MW, which generated 27 TWh/year (EurObserv'ER, 2014; Fuchsz and Kohlheb, 2015). By comparison, the second largest world producer – China – generates just over one-quarter of that (7.6 TWh/year in 2009) (Chen et al., 2012). Italy follows closely in third place at 7.4 TWh of electricity per year produced by 1300 AD plants with a total installed capacity of 1000 MW (Brizzo, 2015). The plants are fed largely with maize grown specifically for this purpose, which in Italy occupies 10% of the total maize cultivation area (1,172,000 ha) (Casati, 2011). However, this is still only half the area in Germany (2,282,000 ha) where it covers one-third of the total maize land (Dressler et al., 2012).

The rapid expansion of biogas production in Europe is largely due to the feed-in-tariffs (FiT) schemes available in 29 countries (Whiting and Azapagic, 2014). For example, electricity generators in Italy using biogas produced in AD plants smaller than 1 MW are paid €280/MWh generated. In the UK, the subsidies are significantly lower, ranging from €130 to 210/MWh, depending on the plant size (Whiting and Azapagic, 2014). This perhaps explains why the deployment of AD was initially slower than in Italy, with only 180 AD plants installed so far, but with a further 500 projects currently under development (NNFCC, 2015). However, the FiT scheme in Italy has recently been changed, reducing the subsidy for electricity by 15–30% and introducing payments for utilization of heat and other coproducts (Ministero dello Sviluppo Economico, 2012). In the US, the growth of biogas production has also been slower than elsewhere, with only 244 AD plants currently in operation (Ebner et al., 2015); this is largely due to the absence of adequate subsidies.

Biogas produced by AD is considered to have a high saving potential with respect to greenhouse gas (GHG) emissions (EC, 2009). However, beyond that, other environmental implications of biogas production are still unclear despite quite a few life cycle assessment (LCA) studies having been carried out. This is due to several reasons. First, most previous studies of biogas have either focused on climate change or considered a limited number of impacts; for a summary, see **Table 1**. As far as the authors are aware, out of 26 studies found in the literature, only five have considered a full suite of impacts normally included in LCA studies, two of which are based in the UK (Mezzullo et al., 2013; Whiting and Azapagic, 2014), one in Argentina (Morero et al., 2015), one in Italy (Pacetti et al., 2015), and one in China (Xu et al., 2015). It is also apparent from **Table 1** that the goal, scope, life cycle impact assessment (LCIA) methodology, feedstocks, and geographical regions covered by the studies vary widely. Most studies are based in Europe with several in China and one each in Argentina, Canada, and the US. All plants have a capacity below 1 MW, with the majority being around 500 kW (where reported); some are electricity only and others combined heat and power (CHP) installations. Most studies have excluded the impacts of constructing and decommissioning the AD and power plants. Maize is the most commonly considered feedstock, followed by animal slurry. The functional unit is largely based either on a unit of feedstock used to generate biogas or a unit of energy (biogas, heat, or electricity). Most studies have relied on secondary foreground data to estimate the impacts or used only limited primary data. However, the greatest variation among the studies is found in the number of impacts considered and the methodologies used to estimate them. The former range from 1 to 18 and the latter cover almost all known LCIA methods, including EcoIndicator 99 (Goedkoop and Spriensma, 2001), CML 2001 (Guinée et al., 2002), Impact 2002+ (Olivier et al., 2003), and ReCiPe (Goedkoop et al., 2009). These and the other differences, including the credits for coproducts, have led to very different results among the studies, making it difficult to compare them, and draw any generic conclusions on the environmental sustainability of biogas.

This study aims to make further contributions to the discussion on the environmental sustainability of biogas. The paper considers life cycle environmental impacts of electricity generation in five real AD-CHP systems using biogas produced from differing mixes of four types of feedstock. The plants are situated in Italy. The novel aspects of the work compared to previous studies include:

- estimation of impacts associated with electricity generated from biogas using different feedstocks, including dedicated maize crops, their mixture with animal slurry, and agricultural waste as well as a mixture of slurry and waste;
- use of primary data for both the feedstock production and operation of the AD-CHP systems;
- consideration of the influence of different scales of the AD-CHP systems on the environmental impacts;
- inclusion of construction and decommissioning of AD and CHP plants;
- estimation of the avoided emissions from using the digestate instead of slurry as fertilizer; and
- comparison of impacts with grid electricity, natural gas, and renewable sources of electricity.

MATERIALS AND METHODS

The environmental impacts of biogas electricity were estimated using LCA as a tool. The study was carried out in accordance with the ISO 14040/44 methodology for LCA (ISO, 2006a,b). The systems were modeled using Gabi LCA software V6.11 (Thinkstep, 2015). The CML 2001 method (Guinée et al., 2002), April 2013 update, was followed to estimate the following 11 impacts considered in this method: abiotic depletion potential of elements (ADP elements), abiotic depletion potential of fossil fuels (ADP fossil), acidification potential (AP), eutrophication potential (EP), freshwater aquatic ecotoxicity potential (FAETP), global warming potential (GWP), human toxicity potential (HTP), marine aquatic ecotoxicity potential (MAETP), ozone layer depletion potential (ODP), photochemical oxidants creation potential (POCP), also known as summer smog, and terrestrial ecotoxicity potential (TETP). For further details on the estimation of the impacts, see Supplementary Material.

TABLE 1 | LCA biogas studies available in the literature.

Reference	Country	No. of AD plants	Plant size	Feedstocks[a]	Functional unit	Foreground LCI data[b]	Capital goods	Impacts (LCIA method)[c]	Best options[c]
Jury et al. (2010)	Luxemburg	Not reported	Not reported	• 4 winter cereals • 4 summer cereals	1 MJ supplied to the natural gas grid	Secondary	Excluded	GWP and CED (impact 2002+)	Not reported
De Vries et al. (2010)	Western Europe	Not reported	Not reported	• Cattle slurry • Maize silage • Codigestion of above	1 ton of feedstock (wet)	Secondary	Excluded	GWP, AP, EP, CED, and LU (not specified)	Codigestion for GWP, EP, AP, and CED; slurry for LU
Blengini et al. (2011)	Italy	Not reported	Not reported	• Maize • Sorghum • Triticale • Miscanthus • Slurry	1 MJ of net energy (heat or electricity) delivered	Secondary	Included	6 (CML 2001)	Miscanthus for GWP, EP, and AP; maize silage for photochemical smog
Dressler et al. (2012)	Germany	1	510 kW	• Maize silage	1 kWh of electricity	Secondary	Excluded	GWP, AP, EP (CML 2001)	Not reported
Lansche and Müller (2012)	Germany	1	186 kW	• Cattle slurry • Maize silage • Grass silage • Codigestion of above	1 MJ of electricity	Primary	Excluded	GWP, AP, EP (CML 2001)	Cattle slurry
Meyer-Aurich et al. (2012)	Germany	1	500 kW	• Cattle slurry • Maize silage • Codigestion of above	1 kWh of electricity	Secondary	Excluded	GWP (IPCC, 2007)	Cattle slurry
De Vries et al. (2012)	The Netherlands	1	500 kW	• Pig slurry • Maize silage • Glycerine • Beet tails • Roadside grass • Codigestion of above	1 ton of feedstock (wet)	Secondary	Excluded	7 (ReCiPe mid-point)	Pig slurry for GWP, AP, ME, and LU; codigestion for FFD, FE, and PMF
Bacenetti et al. (2013)	Italy	3	250–999 kW	• Maize silage • Pig slurry • Codigestion of above	1 kWh of electricity	Primary	Excluded	GWP and CED (IPCC, 2007)	Pig slurry for GWP; maize silage for CED
Mezzullo et al. (2013)	UK	1	Not reported	• Cattle slurry	1 m^3 of methane	Secondary	Included	11 (Ecoindicator 99)	Not reported
Zhang et al. (2013)	China	1	Not reported	• Household waste	Household biogas (digester volume 8 m^3)	Secondary	Included	CO_2 emissions (Not specified)	Not reported
Lijó et al. (2014a)	Italy	2	250 and 500 kW	• Animal slurry • Maize silage	1 ton of feedstock (wet)	Primary only for AD and CHP plant	Excluded	8 (ReCiPe mid-point)	Animal slurry
Lijó et al. (2014b)	Italy	1	500 kW	• Codigestion of maize and triticale silage	100 kWh of electricity	Primary only for AD and CHP plant	Excluded	8 (ReCiPe mid-point)	Maize silage

(Continued)

TABLE 1 | **Continued**

Reference	Country	No. of AD plants	Plant size	Feedstocks[a]	Functional unit	Foreground LCI data[b]	Capital goods	Impacts (LCIA method)[c]	Best options[c]
Rodriguez-Verde et al. (2014)	Spain	1	500 kW	• Pig slurry • Molasses • Fish • Biodiesel • Vinasse residues	110,000 ton/year of pig slurry	Primary and secondary	Excluded	6 (CML 2001)	Not reported
Styles et al. (2014)	UK	4	72–185 kW	• Food waste • Cattle slurry • Maize and grass silage • Miscanthus • Codigestion of above	1 year of farm operation	Secondary	Excluded	GWP, AP, EP, and RDP (CML 2010)	Slurry and food waste
Whiting and Azapagic (2014)	UK	1	170 kW	• Codigestion of slurry, cheese whey, fodder beet, and maize silage	Cogeneration of 1 MWh of heat and electricity	Primary and secondary	Included	11 (CML 2001)	Farm waste better than maize for 8 out of 11 impacts
Bacenetti and Fiala (2015)	Italy	5	100–999 kW	• Cattle slurry • Pig slurry • Cereal silage • Codigestion of above	1 kWh of electricity		Tractors and equipment included; AD and CHP plant excluded	GWP (IPCC, 2007)	Feedstocks
Ebner et al. (2015)	USA	1	Not reported	• Codigestion of cattle slurry and food waste	1 ton of feedstock (wet)	Secondary	Excluded	GWP (IPCC, 2007)	Not reported
Fuchsz and Kohlheb (2015)	Germany	3	600 kW	• Maize silage • Cow slurry • Codigestion of above	1 kWh of electricity	Primary only for AD plant construction	Included	GWP, AP, EP (not specified)	Maize silage for GWP; slurry for AP and EP
Ingrao et al. (2015)	Italy	1	999 kW	• Codigestion of by-products from wheat processing and maize silage	1 kWh of electricity	Primary	Excluded	GWP (IPCC, 2007)	Not reported
Jin et al. (2015)	China	1	Not reported	• Food waste	1 ton of food waste	Secondary	Excluded	5 (CML 2001)	Not reported
Lijó et al. (2015)	Italy	1	1000 kW	• Codigestion of pig slurry and maize silage	1 ton of feedstock (wet)	Primary only for AD and CHP plant	Excluded	8 (ReCiPe mid-point)	Not reported
Morero et al. (2015)	Argentina	2	531–573 kW	• Agroindustrial wastes	1 m^3 of biogas and 1 kWh of electricity	Primary and secondary	Excluded	11 (CML 2001)	Not reported
Pacetti et al. (2015)	Italy	1	Not reported	• Maize • Sorghum • Wheat silage	1 GJ of energy in the biogas	Secondary	Excluded	18 (ReCiPe mid-point)	Sorghum

(Continued)

TABLE 1 | Continued

Reference	Country	No. of AD plants	Plant size	Feedstocks[a]	Functional unit	Foreground LCI data[b]	Capital goods	Impacts (LCIA method)[c]	Best options[c]
Siduo et al. (2015)	Canada	Not reported	Not reported	• Dairy slurry	1100 ton of dairy slurry	Primary and secondary	Excluded	7 (CML 2001)	Not reported
Xu et al. (2015)	China	Not reported	Not reported	• Food waste	1 ton of volatile solids	Secondary	Excluded	18 (ReCiPe mid-point)	Not reported
This study	Italy	5	100–999 kW	• Maize silage • Cow slurry • Codigestion of pig slurry, tomato waste, and maize silage • Codigestion of pig slurry and maize silage • Codigestion of pig slurry, maize silage, and maize ear silage	1 kWh of electricity	Primary	Included	11 (CML 2001)	Slurry for 9 out of 11 impacts; codigestion of slurry, waste, and maize sludge for marine and terrestrial ecotoxicity

[a]*Each bullet point represents a feedstock stream fed to the anaerobic digesters one at a time.*
[b]*LCI, life cycle inventory. Foreground data refer to the AD and CHP plants. Primary data are directly measured and/or collected as part of the study. Secondary data are from databases and literature.*
[c]*LCIA, life cycle impact assessment; AP, acidification potential; CED, cumulative energy demand; EP, eutrophication potential; FE, fresh water eutrophication; FFD, fossil fuel depletion; GWP, global warming potential; LU, land use; ME, marine eutrophication; PMF, particulate matter formation; RDP, resource depletion potential.*

The next sections detail the goal of the study, the assumptions, and data used in the study.

Goal and Scope of the Study

The main goal of the study was to estimate the environmental impacts of electricity generated by different AD-CHP systems utilizing maize silage and agricultural waste. The results were compared with electricity from the grid, natural gas, and different renewables to help evaluate the environmental sustainability of biogas electricity relative to other available options.

Five real AD-CHP systems were considered using differing combinations of the following feedstocks: maize and maize ear silage; pig and cow slurry; and tomato peel and seeds (**Table 2**). The volume of the AD digesters ranged from 1650 to 2750 m^3 and the installed electrical capacity of the CHP plants from 100 to 999 kW. The plants are located at farms producing the feedstocks in Lombardy in Northern Italy, where the majority of the country's biogas plants are situated (Negri et al., 2014).

As indicated in **Figure 1**, the scope of the study was from "cradle to grave," including:

- production of maize silage (where used), comprising cultivation, transport from fields to the farm (1 km), and the ensiling;
- collection of slurry and tomato waste and delivery to the AD plants;
- construction and decommissioning of AD and CHP plants;
- production of biogas in the AD plants and its treatment (filtration, dehumidification, and desulfurization);
- cogeneration of electricity and heat in the CHP plants; the heat, except that used for heating the digesters, is considered as waste as it is not used;
- storage and subsequent use of digestate as fertilizer; note that all plants but no. 2 use open storage of digestate.

Electricity distribution and consumption were excluded from the system boundary.

The functional unit was defined as "generation of 1 MWh of electricity to be fed into the grid." Although heat is cogenerated with electricity, all the impacts were allocated to the latter as the excess heat not utilized in the system is discharged as waste.

Inventory Data

Feedstock Production

The inventory data for the production of maize silage are detailed in Tables S1 and S2 in Supplementary Material. As indicated in the tables, data for field operations were collected directly from the farms. The background data were sourced from Ecoinvent (Nemecek and Kägi, 2007) and modified to match the characteristics of the machinery used for maize cultivation in Lombardy, based on information in Bodria et al. (2006). No environmental impacts were considered for tomato waste and slurry as they are waste.

Ammonia and nitrous oxide emissions as well as nitrate leachates from the application of the digestate and urea as fertilizers were estimated according to Brentrup et al. (2000). Phosphate leachates and run-offs were calculated based on Nemecek and Kägi (2007). To estimate pesticide emissions to the environment, several factors need to be considered, such

TABLE 2 | Summary of the main characteristics of the AD-CHP plants considered in the study.[a]

Plant	Feedstock	Volume of AD digesters (m^3)	Dry matter content in digesters (%)	Organic loading in digesters (kg/day m^3)	Methane content in biogas (%)	Installed CHP power (kW)	Electricity generation (MWh/year)	Electricity consumption (MWh/year)	Heat generation (MWh/year)	Heat consumption by AD (MWh/year)
Plant 1	• Pig slurry • Tomato peel and seeds • Maize silage	1650	8.7	0.92	52.8	230	1945	173	2549	809
Plant 2	• Pig slurry • Maize silage	2250	10.6	1.07	52.6	300	2429	206	3184	814
Plant 3	• Pig slurry • Maize silage • Maize ear silage	2000	9.7	0.98	52.7	300	2505	276	3514	799
Plant 4	• Maize silage	2 × 2750	10.7	3.40	52.1	999	7972	717	8771	2505
Plant 5	• Cow slurry	1850	8.5	0.58	56.0	100	781	86	1095	547

[a]*All data sourced directly from the farm/plant owners.*

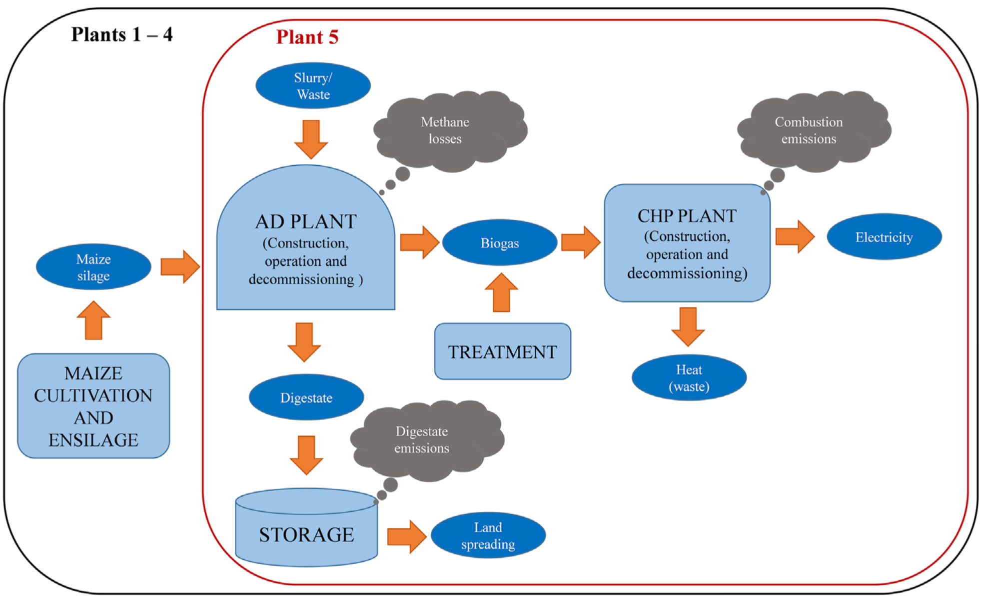

FIGURE 1 | System boundaries considered in the study. No environmental impacts are considered for the tomato waste, pig and cow slurry as they are waste. All the impacts are allocated to electricity as heat is not exported from the system.

as the way in which a pesticide is applied, the soil type, and the meteorological conditions during application (EMEP/EEA, 2013). However, considerations of these parameters is often impractical in LCA studies due to a lack of detailed data (Milà i Canals, 2007). Thus, pesticide emissions to air, water, and soil were determined in accordance with Margni et al. (2002) and Audsley (1997), assuming the following partitioning of the active pesticide components: 85% of the total amount applied remains in the soil, 5% in the plant, and 10% is emitted into the atmosphere; furthermore, 10% of the applied dose is lost as a run-off from the soil into the water. This method is also recommended for use by Curran (2012) and was applied in some other LCA studies [e.g., Boschiero et al. (2014), Falcone et al. (2015), and Fantin et al. (2015)].

Land use change was not considered as the maize feedstock is grown on land previously used to cultivate cereals.

The transport and packaging of pesticides and fertilizers were not included in the system boundaries because of a lack of data. This is not deemed a limitation as some other studies found that their contribution was insignificant [e.g., Cellura et al. (2012)].

AD and CHP Plants

In all the AD plants evaluated in this study, the digestion takes place in continuously stirred reactors under mesophilic conditions at a temperature of 40°C (±0.2°C), which is controlled and monitored continuously. Therefore, the digesters are operated at the top end of the temperature scale, which for mesophilic digestion ranges from 30 to 40°C (Weiland, 2010). The digesters are made from iron-reinforced concrete and have an expanded polyurethane external insulation. The biomass is fed into the digesters every 90 min in small amounts and heated using the heat generated by the adjacent CHP. As indicated in **Table 2**, the dry matter content in the digester varies from 8.5 to 10.6%, and the organic loading rate from 0.58 to 3.4 kg/day m^3. The biogas composition is similar across the plants with the methane content ranging from 52 to 56% of the biogas volume.

The biogas is stored on top of the digesters in a gasometer dome with a spherical cap. Before being fed into the CHP plant, the biogas is filtered through a sand filter, dehumidified in a chiller, and then desulfurized using sodium hydroxide (NaOH). NOx emissions are controlled by a catalytic converter. The digestate is pumped from the bottom of the digesters and stored in open tanks in all the plants except for Plant 2, where it is stored in a covered tank.

The biogas is fed into the CHP plant to generate electricity and heat. Electricity is sold to the national grid while the heat is used for heating the digesters and the excess is dissipated by fan-coolers. The electricity consumption for operating the AD plants is sourced from the national grid to ensure continuous operation during the CHP downtimes. The amount of electricity used by the system ranges from 8.5 to 11% of the total electricity generated (**Table 2**).

Detailed inventory data for the AD and CHP plants can be found in **Tables 2** and **3**. The operational data (feedstock production, consumption of electricity and heat, electricity generation) were obtained from the owners. Chemical characterization of different types of feedstock and their biogas production potentials were determined by laboratory tests (Fiala, 2012; Negri et al., 2014; Bacenetti et al., 2015) and used to calculate the biogas production by the AD plants. The emissions from the CHP plants were calculated based on NERI (2010). The useful lifetime of the AD plants was assumed to be 20 years (Nemecek and Kägi, 2007). For the CHP plants, the lifespan is shorter, between 8 and 10 years because of the high content of hydrogen sulfide (Fiala, 2012). At the end of a plant's useful lifetime, its construction materials were assumed to be landfilled, except for plastic materials, which were incinerated; the influence on the impacts of recycling is explored in a sensitivity analysis later in the paper.

The background data on the construction materials, their transport (120 km by rail and 35 km in 20–28 ton trucks) and landfilling were sourced from the Ecoinvent database v2.2 (Ecoinvent, 2010). Since the data for construction materials for the AD and CHP plants in Ecoinvent correspond to a different plant size (300 m^3 for the AD and 160 kW$_{el}$ for the CHP plants), the environmental impacts from their manufacture were estimated by scaling up or down their capacity to match the sizes of the AD and CHP plants considered in this study. This was carried out following the approach used for cost estimation in scaling up process plants (Coulson et al., 1993) but instead of costs, estimating environmental impacts as follows (Whiting and Azapagic, 2014):

$$E_2 = E_1 \cdot (C_2 / C_1)^{0.6} \tag{1}$$

where E_2 environmental impacts of the larger plant (AD or CHP); E_1 environmental impacts of the smaller plant (AD or CHP); C_2 capacity of the larger plant (volume for the AD plant and installed power for the CHP plant); C_1 capacity of the smaller plant (volume for the AD plant and installed power for the CHP plant); 0.6 scaling factor.

Digestate Use and Methane Emissions Credits

In all the plants except no. 4, the digestate is used as fertilizer on the farms, replacing pig or cow slurries applied previously as part of a traditional slurry management method (see **Figure 2**). Both digestate and the slurry from Plants 1, 3, and 5 are stored in open tanks before application, during which they emit methane. However, the emissions from digestate are lower than from slurry storage (Amon et al., 2006; Wang et al., 2014), and the AD systems were credited for the avoidance of the emissions. Note that in Plant 2, the digestate is stored in covered tanks, with no emissions of methane (IPCC, 2006); thus, the net emissions from this system are negative (**Table 3**).

At Plant 4, a closed maize cycle is practiced, whereby the digestate is used as fertilizer for the maize which is fed into the same plant (**Figure 3**). The digestate at this plant is stored in open tanks.

Alternative Electricity Sources

Grid electricity was considered here as the main alternative to electricity from biogas. This is due to the latter being fed into the national grid, displacing an equivalent amount of grid electricity. The Italian electricity mix is shown in Figure S1 in Supplementary Material. Given that the electricity mix is dominated by natural gas (53%) (IEA, 2011), biogas electricity was also compared to this

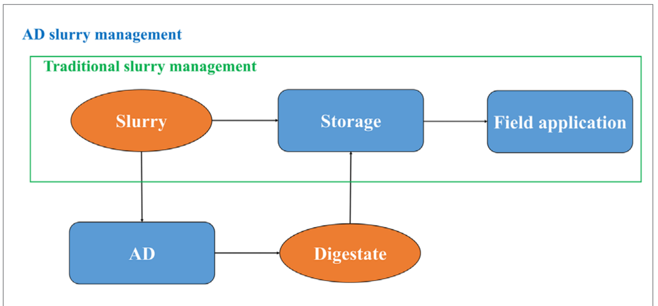

FIGURE 2 | **Traditional and AD slurry management**.

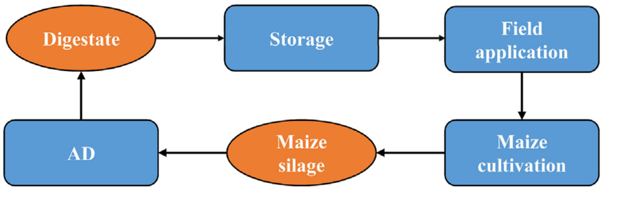

FIGURE 3 | **Maize silage cycle**.

TABLE 3 | Inventory data for the AD and CHP plants (expressed per megawatt hour of electricity).

	Unit	Plant 1	Plant 2	Plant 3	Plant 4	Plant 5	Data sources
AD							
Pig slurry	ton	8.4	6.0	7.3	–	–	Farm owner
Cow slurry	ton	–	–	–	–	21.0	-||-
Maize silage	ton	0.9	2.25	0.8	2.45	–	-||-
Tomato peel and seeds	ton	1.5	–	–	–	–	-||-
Ear maize silage	ton	–	–	0.66	–	–	-||-
Water	ton	0.94	0.75	–	0.23	–	-||-
Sodium hydroxide	g	28.3	29.6	29.6	29.9	30.0	-||-
Electricity from the grid	MWh	0.09	0.09	0.11	0.09	0.11	-||-
Heat from CHP	MWh	0.42	0.34	0.32	0.38	0.70	-||-
Net biogas production	Nm^3	280	278	289	252	285	Own calculations based on farm owner's data
CHP							
Electricity generated	MWh	1	1	1	1	1	-||-
Heat generated	MWh	1.3	1.3	1.4	1.1	1.4	Own calculations based on farm owner's data
Emissions associated with AD							
Methane emissions from AD plant	m^3	3.8	3.8	4.0	3.4	3.9	Bacenetti et al. (2013)
Methane emissions from digestate storage	kg	8.9	0	8.9	8.9	8.9	Edelmann et al. (2011)
Credit for avoiding methane emissions from slurry storage	kg	−6.9	−6.3	−6.0	0	−32.0	Amon et al. (2006) and Wang et al. (2014)
Net emissions of methane	kg	5.9	−2.5	6.9	12.3	−19.2	Own calculations
Ammonia emissions from digestate storage	kg	0.2	0.0	0.2	0.2	0.2	Edelmann et al. (2011)
Emissions from CHP							
NOx	g	56.1	56.1	56.1	56.1	56.1	NERI (2010)
NMVOC[a]	g	2.8	2.8	2.8	2.8	2.8	-||-
CH_4	g	120.6	120.6	120.6	120.6	120.6	-||-
CO	g	86.1	86.1	86.1	86.1	86.1	-||-
N_2O	mg	444	444	444	444	444	-||-
As	mg	11	11	11	11	11	-||-
Cd	mg	1	1	1	1	1	-||-
Co	mg	58	58	58	58	58	-||-
Cr	mg	50	50	50	50	50	-||-
Cu	mg	86	86	86	86	86	-||-
Hg	mg	33	33	33	33	33	-||-
Mn	mg	53	53	53	53	53	-||-
Ni	mg	64	64	64	64	64	-||-
Pb	mg	1	1	1	1	1	-||-
Sb	mg	33	33	33	33	33	-||-
Se	mg	58	58	58	58	58	-||-
Tl	mg	58	58	58	58	58	-||-
V	mg	11	11	11	11	11	-||-
Zn	mg	1097	1097	1097	1097	1097	-||-

[a]*Non-methane volatile organic compounds.*

option. Furthermore, as biogas is a renewable resource, it was also compared to the other renewables contributing to the Italian mix (see Figure S1 in Supplementary Material). The system boundary for all the alternatives was from "cradle to grave," and all the data were sourced from Ecoinvent (2010). As for the biogas electricity, distribution and consumption of electricity were not considered.

RESULTS

The results suggest that biogas electricity generated by Plant 5 is environmentally the best option among the five plants considered (**Figure 4**), largely because it does not use maize silage as a feedstock. The exceptions to this are the MAETP and TETP for which Plant 1 is slightly better because these impacts are not affected by maize silage (as discussed further below). Plant 1 is also the second best option for all other impacts apart from GWP and POCP, for which Plant 2 is better because of the lower methane emissions from digestate.

The differences in the impacts for Plants 2 and 4, which are fed with approximately the same amount of maize silage, are due to the differences in the digestate emissions and the capacities of the AD and CHP plants.

Despite the highest biogas production, Plant 3 is the worst option across all the impact categories because of the maize ear silage, which has impacts twice as high as maize silage owing to its lower yield (Table S2 in Supplementary Material). The exceptions to this are GWP and POCP, for which Plant 4 is worst because of the higher net methane emissions (**Table 3**).

The following sections discuss in more detail the impacts from the different plants (**Figure 4**) and the contributions of different life cycle stages (**Figures 5A–E**).

Abiotic Depletion Potential (ADP Elements and ADP Fossil)

Abiotic depletion of elements and fossil resources range from 142 to 243 mg Sb eq./MWh and from 1010 to 1570 MJ/MWh, respectively, with Plant 5 being the best and Plant 3 the worst option for both impacts.

As indicated in **Figures 5A–D**, the depletion of elements for Plants 1–4 is mainly due to the cultivation of maize and is associated with the materials used for agricultural machinery. For Plant 5, on the other hand, the major contributors are construction materials for the AD and CHP plants (**Figure 5E**); the latter is also a hotspot for Plant 1. This is due to economies of scale: they have smaller CHP plants and thus a higher consumption of resources per megawatt hour electricity generated.

As also shown in **Figures 5A–D**, the major contributors to fossil depletion for Plants 1–4 are the fuel used in the agricultural machinery for maize cultivation and the electricity for the AD plants. For Plant 5, the grid electricity used to operate the AD plant accounts for the majority of this impact (**Figure 5E**).

Acidification and Eutrophication Potentials

The estimated AP varies from 2.6 to 5.5 kg SO_2 eq./MWh and EP from 0.2 to 1.9 kg PO_4 eq./MWh. As for ADP, biogas electricity generated by Plant 5 is the best and by Plant 3 the worst option for these two impacts. For Plants 1–4, maize cultivation is responsible for the large majority of AP and EP (**Figure 5A–D**), whereas for Plant 5 (**Figure 5E**), it is the ammonia emitted during the digestate storage as well as the emissions of acid gases and nutrients in the life cycle of the grid electricity used for AD.

Global Warming Potential ($GWP_{100\ years}$)

The values for GWP range from −395 to 408 kg CO_2 eq./MWh, with electricity from Plant 5 being the best option and from Plant 4 the worst. The vast majority of GWP (64%) is due to methane emissions from the digestate during its storage. For Plant 2, GWP is mainly from the maize silage (**Figure 5B**). The negative contributions shown in the figure are due to the methane credits for the avoidance of the traditional slurry management, as described in Section "Digestate Use and Methane Emissions Credits." For Plant 5, the methane credits are higher than the methane emissions from the digestate, leading to a negative impact of −395 kg CO_2 eq./MWh (**Figure 5E**). Note that carbon dioxide emissions from biogas combustion in the CHP plant are not considered as they are biogenic in nature.

Human Toxicity Potential

This impact is lowest for electricity generated by Plants 1 and 5 [79 kg dichlorobenzene (DCB) eq./MWh] and highest for Plant 3 (114 kg DCB eq./MWh). For Plants 1–4, the main contributor is the production of maize silage and the emissions from biogas combustion, in particular chromium and thallium (see **Table 3**). For Plant 5, HTP is mainly affected by CHP operation, followed by AD operation and plant construction (**Figure 5E**).

Ecotoxicity Potentials (FAETP, MAETP, and TETP)

The lowest FAETP is estimated for Plant 1 (198 kg DCB eq./MWh) and the highest for Plant 3 (413 kg DCB eq./MWh). The

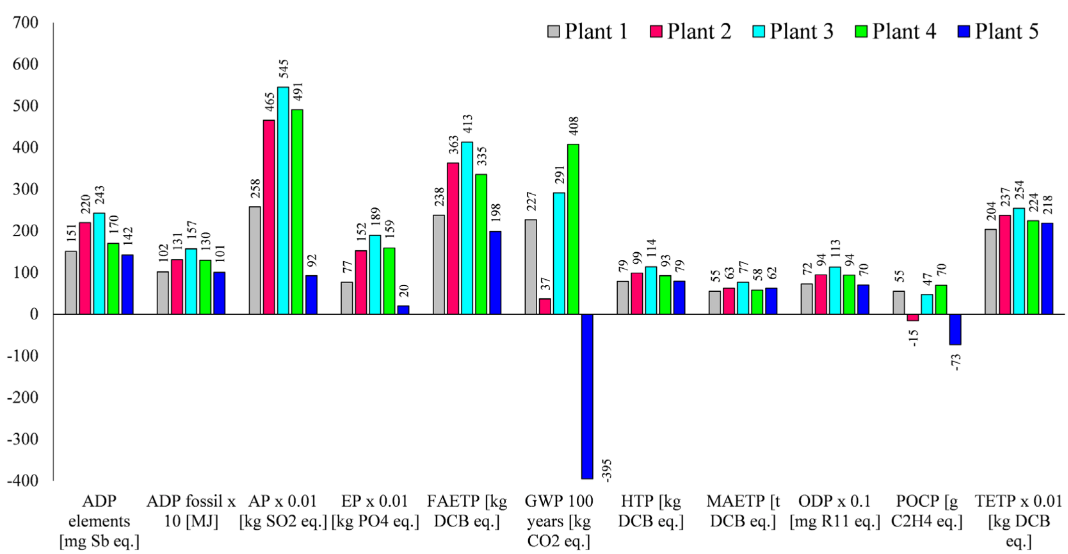

FIGURE 4 | **The environmental impacts associated with the generation of biogas electricity**. All impacts expressed per megawatt hour of electricity generated. Impacts nomenclature: ADP elements, abiotic depletion potential for elements; ADP fossil: abiotic depletion potential for fossil fuels; AP, acidification potential; EP, eutrophication potential; FAETP, freshwater aquatic ecotoxicity potential; GWP, global warming potential; HTP, human toxicity potential; MAETP, marine aquatic ecotoxicity potential; ODP, ozone depletion potential; POCP, photochemical oxidants creation potential; TETP, terrestrial ecotoxicity potential; DCB, dichlorobenzene.

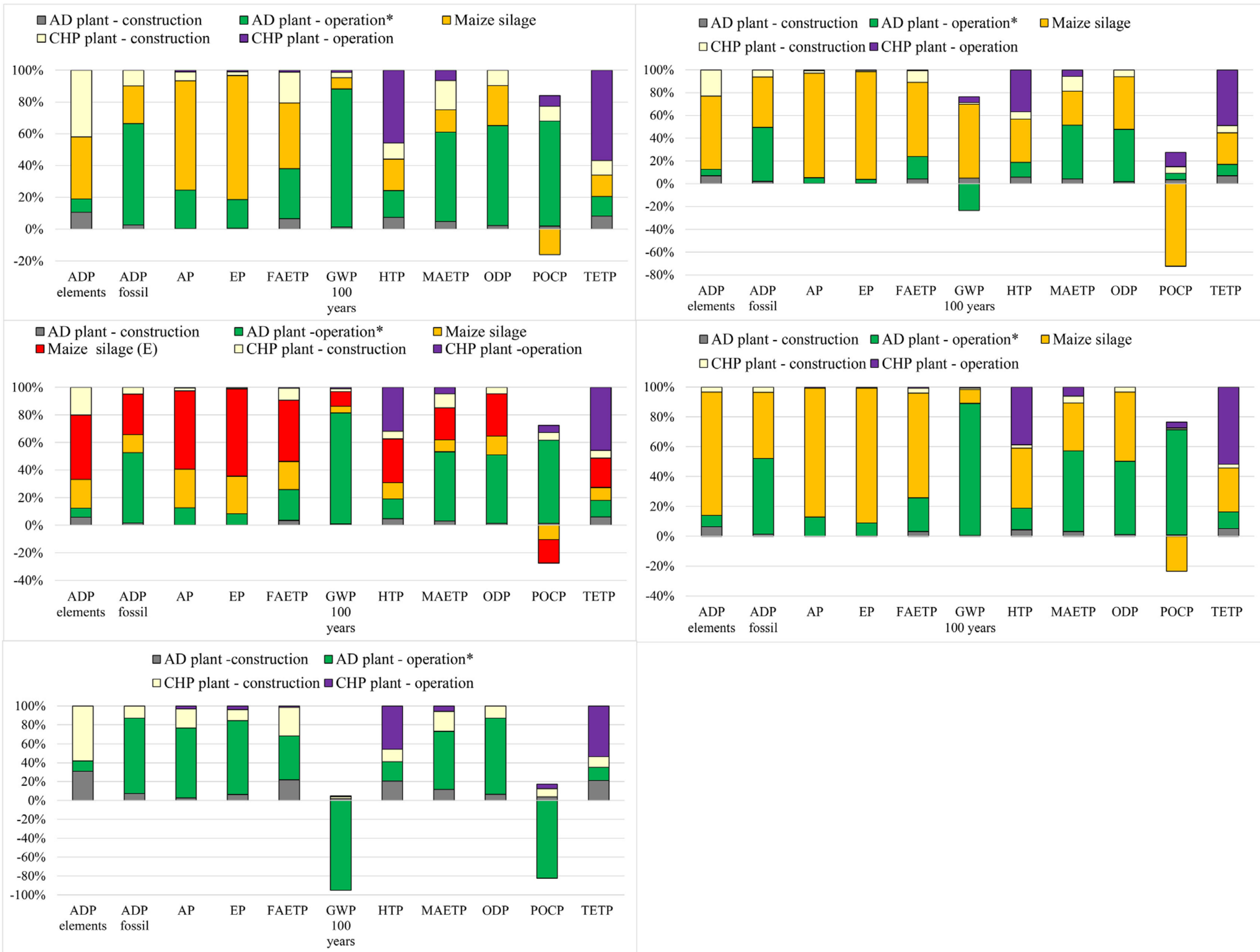

FIGURE 5 | **Contribution analysis for different AD-CHP plants**. **(A)** Plant 1 (top left); **(B)** Plant 2 (top right); **(C)** Plant 3 (middle left); **(D)** Plant 4 (middle right); **(E)** Plant 5 (bottom). AD plant – operation* includes grid electricity used for AD, methane losses during AD and emissions associated with digestate storage. Maize silage **(E)** maize ear silage. For impacts nomenclature, see figure. For the feedstocks, see **Table 2**. Negative values represent the credits for the avoidance of methane emissions by using digestate as fertilizer instead of animal slurry.

production of maize silage and the plant operation are the main contributors to this impact for Plants 1–4. This is mainly due to the emissions of pesticide used for maize cultivation (**Table 3**) and metals (nickel, beryllium, cobalt, and vanadium) emitted in the life cycle of the grid electricity. It can be noted that Plant 1 has lower MAETP and TETP, which is due to the efficiency associated with economies of scale as these impacts are mainly influenced by the plant operation (**Figures 5A,E**).

Unlike HTP, the best option for MAETP is Plant 5 at 55 ton DCB eq./MWh but, as for HTP, Plant 3 has the highest impact (77 ton DCB eq./MWh). The main hotspot is grid electricity used for AD because of the emissions of beryllium and hydrogen fluoride in the life cycle of electricity generation.

The same trend is found for TETP, with Plant 5 being the best option (2 kg DCB eq./MWh) and Plant 3 the worst (2.5 kg DCB eq./MWh). Maize silage and CHP operation are the main contributors to TETP for Plants 1–4. Like HTP, the latter is mainly due to the emissions of chromium and thallium from biogas combustion. For Plant 5, CHP operation is the main hotspot (biogas combustion), followed by AD operation and plant construction.

Ozone Layer Depletion Potential

At 7 mg R11 eq./MWh, Plant 5 has the lowest ODP and, as for most other impacts, Plant 3 the highest (11.3 mg R11 eq./MWh). The main contributors are halons emitted in the life cycle of grid electricity used in AD (related to natural gas transportation), followed by the emissions from diesel used in the machinery during maize cultivation (Plants 1–4).

Photochemical Oxidants Creation Potential

The POCP ranges from −73 g C_2H_4 eq./MWh for Plant 5 to 70 g C_2H_4 eq./MWh for Plant 3. For Plants 1, 3, and 4, the impact is largely due to the emissions of methane from the digestate and the methane losses from the AD plant. The negative contributions (**Figure 5**) are due to two reasons: first, according to the CML 2001 method, nitrogen oxides emitted during the cultivation of maize reduce POCP (Plants 1–4); and second, because of the methane credits (Plant 5).

Comparison with Alternative Electricity Sources

The biogas electricity is compared to electricity from the grid, natural gas, and renewables in **Figure 6** and the ranking of different options with respect to each impact is summarized in the heat map in **Figure 7**.

As can be seen in **Figure 6**, grid electricity has higher impacts than electricity from biogas for seven out of 11 categories: ADP fossil, FAETP, GWP, HTP, MAETP, ODP, and POCP. This is mainly due to the high contribution of fossil fuels in the Italian electricity mix. An exception to this is Plant 3 which has a higher HTP than the grid because of the toxic emissions in the life cycle of maize ear silage.

Electricity from the grid also has lower AP (by 10–57%) and EP (32–72%) than biogas electricity; this is due to maize cultivation which contributes significantly to these two impacts (see **Figure 5**). The exception to this is Plant 5 which has lower impacts than grid electricity (by ~60%) because it does not use maize silage.

Two further impacts are lower for grid electricity: depletion of elements and TETP. This could be explained by the greater economies of scale of the plants on the grid, which require a lower amount of resources and thus have lower toxic emissions on a life cycle basis per unit of electricity generated than the agricultural machinery and the AD-CHP plants.

Unlike grid electricity, electricity from natural gas is environmentally more sustainable than biogas for most categories, except ADP fossil, GWP, ODP, and POCP (**Figure 6**). In comparison to the renewables, biogas electricity has mostly higher impacts, with a few exceptions. For example, biogas has a lower AP than geothermal power across all the AD-CHP plants considered. Furthermore, Plant 5 has lower GWP and Plant 2 lower POCP than any other renewable option. Biogas is also better than solar PV in terms of ADP elements, HTP, FAETP, MAETP, ODP, and

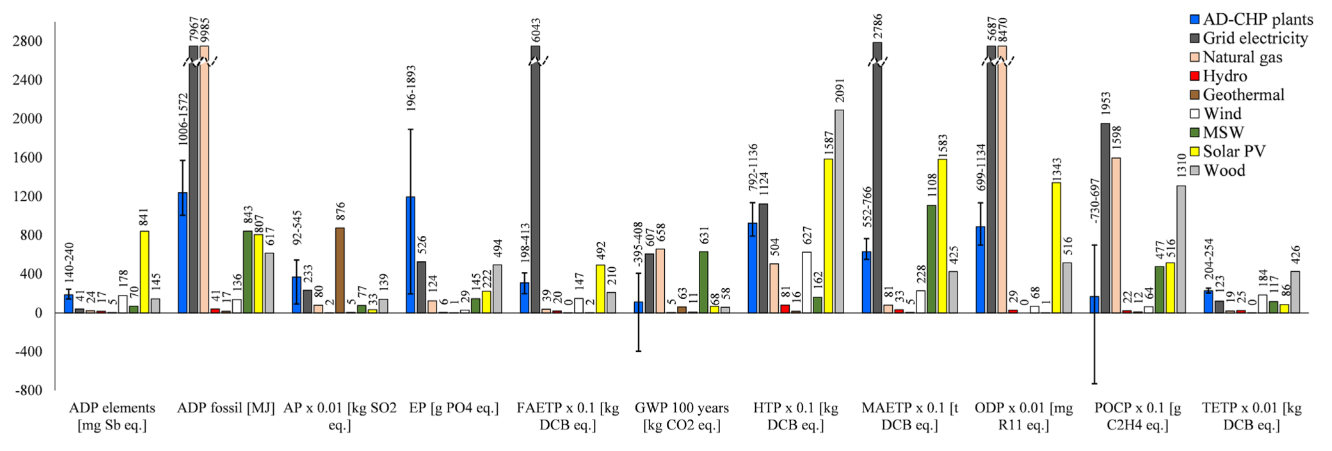

FIGURE 6 | **Comparison of biogas electricity with the alternatives**. All impacts expressed per megawatt hour of electricity. For the AD-CHP plants, the average results are shown, with the error bars representing the impacts ranges for different plants. For impacts nomenclature, see **Figure 5**. MSW, municipal solid waste; wood, wood chips in a CHP plant.

	Plant 1	Plant 2	Plant 3	Plant 4	Plant 5	Grid electricity	Natural gas	Hydro	Geothermal	Wind	MSW	Wood	Solar PV
ADP elements	2%	3%	2%	3%	2%	0.5%	0.3%	0.2%	0.1%	2%	1%	2%	100%
ADP fossil	10%	13%	13%	16%	10%	80%	100%	0.4%	0.2%	1%	8%	6%	8%
AP	29%	53%	56%	62%	11%	27%	9%	0.2%	100%	1%	9%	16%	4%
EP	41%	81%	84%	100%	10%	28%	7%	0.3%	0.1%	2%	8%	26%	12%
FAETP	4%	6%	6%	7%	3%	100%	1%	0.3%	0%	2%	0%	3%	8%
GWP 100 years	34%	6%	62%	44%	-60%	92%	100%	1%	10%	2%	96%	9%	10%
HTP	38%	47%	44%	54%	38%	54%	24%	4%	1%	30%	8%	100%	76%
MAETP	20%	23%	21%	27%	22%	100%	3%	1%	0.2%	8%	40%	15%	57%
ODP	9%	11%	11%	13%	8%	67%	100%	0.3%	0%	1%	0%	6%	16%
POCP	28%	-8%	36%	24%	-37%	100%	82%	1%	1%	3%	24%	67%	26%
TETP	48%	56%	53%	60%	51%	29%	5%	6%	0%	43%	27%	100%	20%

FIGURE 7 | **Heat map of environmental impacts from biogas electricity and the alternatives considered in this study**. The worst option is set at 100% and the others are expressed as a percentage of impact relative to the worst option. Waste, municipal solid waste; MSW, municipal solid waste; wood, wood chips in a CHP plant; solar PV, solar photovoltaics. For impacts nomenclature, see **Figure 5**.

POCP. It also has a lower MAETP than electricity from municipal solid waste and it outperforms wood for HTP, POCP, and TETP.

With a specific reference to GWP, the main driver for biogas production, Plant 5 is the best option overall, sequestering 395 kg CO_2 eq./MWh. All other plants generate higher GHG emissions than any of the renewable options considered here. The only other impact for which biogas electricity is a better option than any other is POCP, but again only for Plant 5; however, this plant has the highest TETP than any other alternative.

These results are summarized in **Figure 7**, which shows the percentage difference between the worst option and the rest of the alternatives for each impact. Overall, assuming equal importance of all the impacts, hydropower could be considered the best option and grid electricity the worst, with biogas being on average a middle-ranking option.

Comparison with Other Studies

As discussed in the Section "Introduction," comparison of the results from different studies is not easy for the reasons outlined there. The only studies for which comparison is possible are those by Blengini et al. (2011), Dressler et al. (2012), Meyer-Aurich et al. (2012), Bacenetti et al. (2013), Whiting and Azapagic (2014), and Ingrao et al. (2015); for a summary of these studies, see **Table 1**.

As can be inferred from **Figure 8**, the results from the current study compare favorably in terms of AP, EP, GWP, and POCP, given the different assumptions, system credits, and geographical locations across the studies. However, the average GWP estimated in this work appears to be lower than in the other studies, mainly because of Plant 5 which has a negative value for this impact. Nevertheless, the impact for the AD-CHP system using pig slurry reported by Bacenetti et al. (2013) compares well with Plant 5 which uses cow slurry (−368 and −395 kg CO_2 eq./MWh, respectively). The GWP in Blengini et al. (2011) is consistent with that estimated for Plant 4, while the values found by Dressler et al. (2012), Meyer-Aurich et al. (2012), Bacenetti et al. (2013), and Ingrao et al. (2015) agree well with the results for Plants 1 and 3. It should be noted that, unlike other studies, Meyer-Aurich et al. (2012) have considered land-use change (associated with maize cultivation), finding that it increases GWP by 20%; however, differences in other assumptions cancel out this effect and, consequently, the results still agree with those in the current study.

The comparison of the other impacts is only possible with the study by Whiting and Azapagic (2014), since the other authors did not consider them. As can be seen in **Figure 8**, the results agree for HTP but differ for ADP, FAETP, MAETP, ODP, and TETP. The reason for these differences could be due to the different updates of the CML method and Gabi software, as well as the different assumptions, credits for fertilizers, and geographical locations. On the other hand, both studies are in agreement that the contribution of the AD and CHP plants construction is significant for ADP elements and the toxicity-related impacts.

Sensitivity Analysis

Because of their significant contribution to the impacts, the following parameters are considered in the sensitivity analysis:

(i) maize yield;
(ii) heat utilization;
(iii) recycling of AD and CHP construction materials; and
(iv) covered storage of digestate in Plant 4.

The results are discussed in the following sections.

Maize Yield

To explore the effect of this parameter on the impacts, the maize yield was varied by ±15% against the baseline shown in Table S2 in Supplementary Material. The results in **Figure 9** suggest that the overall effect of maize yield on the environmental impacts is

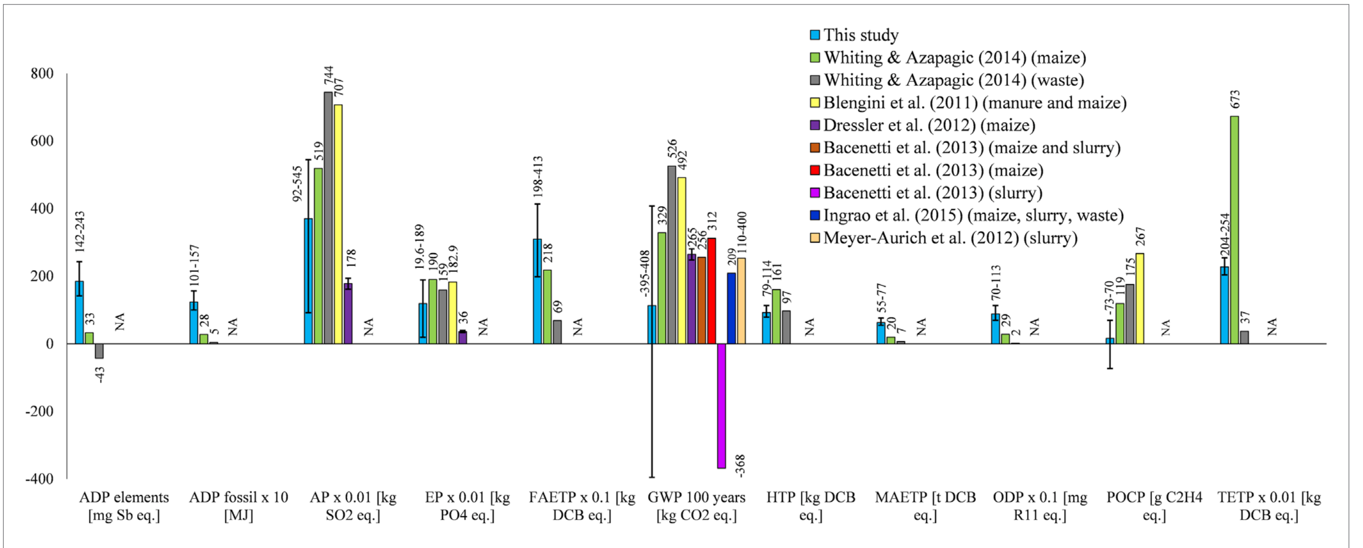

FIGURE 8 | **Comparison of the results with the literature**. All impacts expressed per megawatt hour of electricity. The error bars represent the range of results for the different plants. NA, not available. Waste, agricultural. For impacts nomenclature, see **Figure 5**.

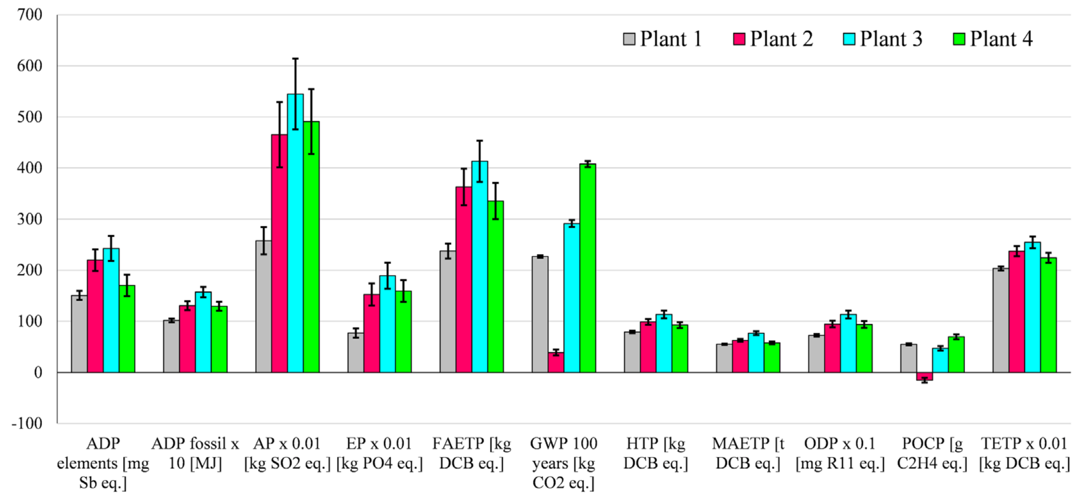

FIGURE 9 | **Sensitivity analysis assuming different maize yields for biogas produced in Plants 1–4**. All impacts expressed per megawatt hour of electricity. The height of the columns corresponds to the yield indicated in Table S2 in Supplementary Material. The error bars refer to the yield variation of ±15%. For impacts nomenclature, see **Figure 5**.

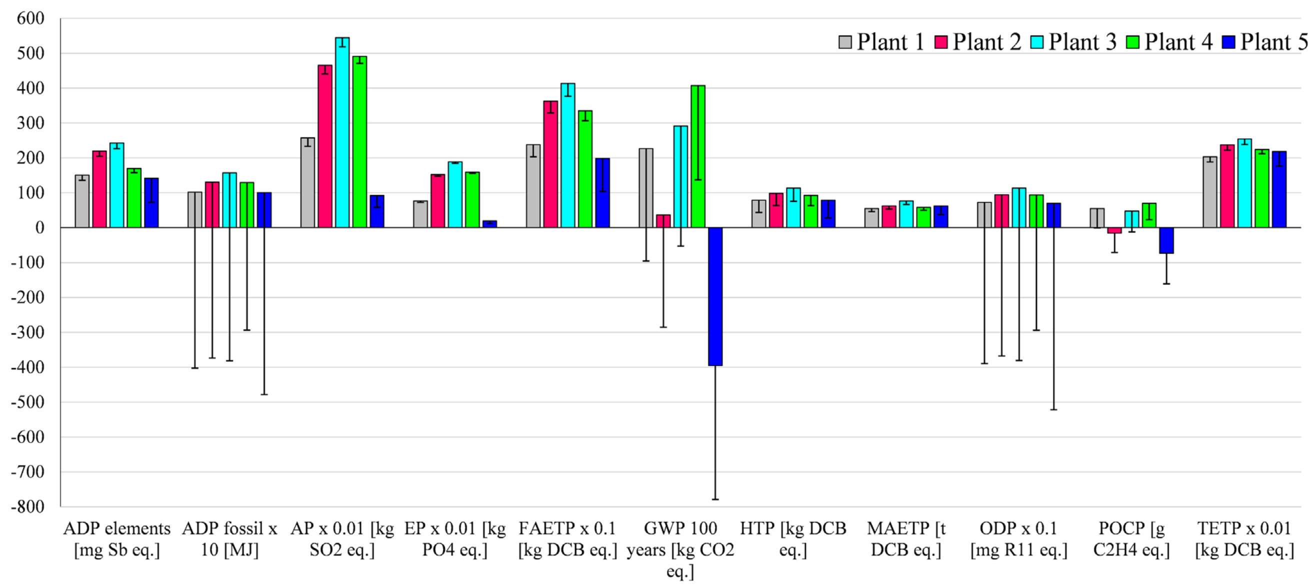

FIGURE 10 | **Sensitivity analysis assuming the net heat produced is used and substitutes a gas boiler**. All impacts expressed per megawatt hour of electricity. Capacity of boiler: >100 kW for Plants 1–4 and <100 kW for Plant 5. For impacts nomenclature, see **Figure 5**.

small for most impacts, except for AP and EP which change by up to 14%. This is to be expected given the high contribution of maize cultivation to these categories.

The ADP elements and FAETP results are also affected for Plant 4, varying by up to 12%, because of the change in the resource requirements for the agricultural machinery and the related toxicity of the construction materials. Despite these changes, the variation in the maize yield considered here does not affect the comparison of biogas with the alternative electricity sources discussed in Section "Comparison with Alternative Electricity Sources."

Heat Utilization

This part of the sensitivity analysis considers a scenario in which the net heat produced by the CHP plants is used instead of being wasted. This is motivated by the introduction of subsidies for heat (see Introduction), which aim to stimulate its utilization. It was assumed that the heat generated by the CHP substitutes a gas boiler for which the AD-CHP systems were credited. The LCA data for the boiler were sourced from Ecoinvent (2010).

As indicated in **Figure 10**, if the heat were utilized all of the impacts would be reduced, some of them significantly, across the different plants: ADP fossil would be lower by four to six times, GWP up to nine times, ODP by five to eight times, and POCP two to four times. This means that biogas electricity from all five plants would have lower impacts for these categories than any other renewable option considered here. However, there would be no change in ranking with respect to grid electricity because ADP elements, AP, EP, and TETP remain higher for biogas electricity.

Recycling of Construction Materials

As mentioned earlier, it was assumed that all the construction materials apart from plastics are landfilled after decommissioning of the plants. Since the construction of the plants has a significant contribution for some impacts, particularly for Plants 1 and 5 (**Figures 5A,E**), the sensitivity analysis considers if and how they would change if concrete, steel, iron, and platinum (in the CHP catalytic converter) were recycled. For these purposes, the recycling rates for the former three materials were assumed equal to current recycling rates in Italy: 60% for concrete (UNI, 2005) and 74% for steel and iron (Fondazione per lo sviluppo sostenibile, 2012). As there are no data for platinum recycling, a recovery rate of 90% was assumed. Plastic materials were not considered for recycling as their quantity is small.

The results are presented in **Figure 11** for the impacts that are affected by the recycling. The greatest reduction would be achieved for ADP elements (up to 39%) and POCP (up to 13.5%), followed by AP and FAETP (~8%); MAETP would also go down (~5%). The effect on the other impacts is small (<2%).

Covered Storage of Digestate

As discussed in Section "Results," biogas electricity from Plant 4, which uses maize silage as the AD feedstock, has higher GWP and POCP than any other plant. Given that much of that is due to methane emissions from the open storage of digestate

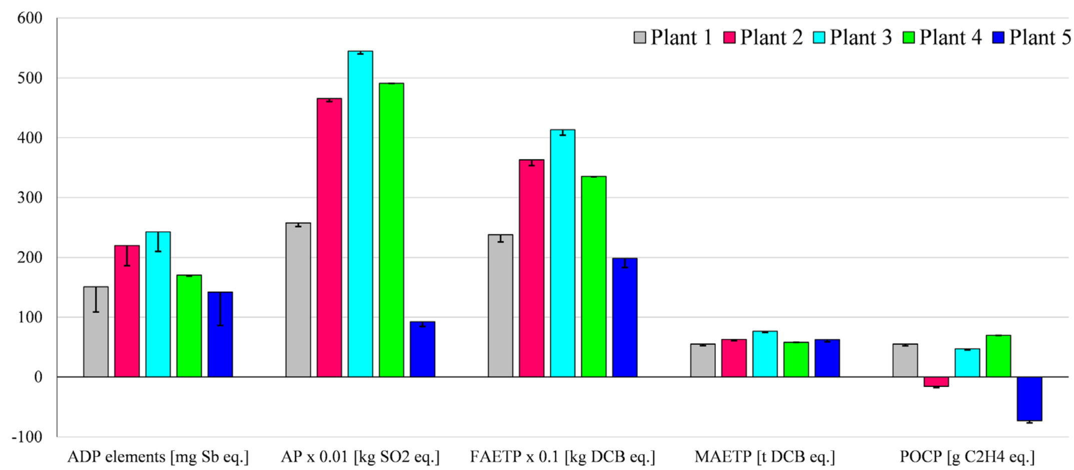

FIGURE 11 | Sensitivity analysis assuming recycling of construction materials. For impacts nomenclature, see **Figure 5**.

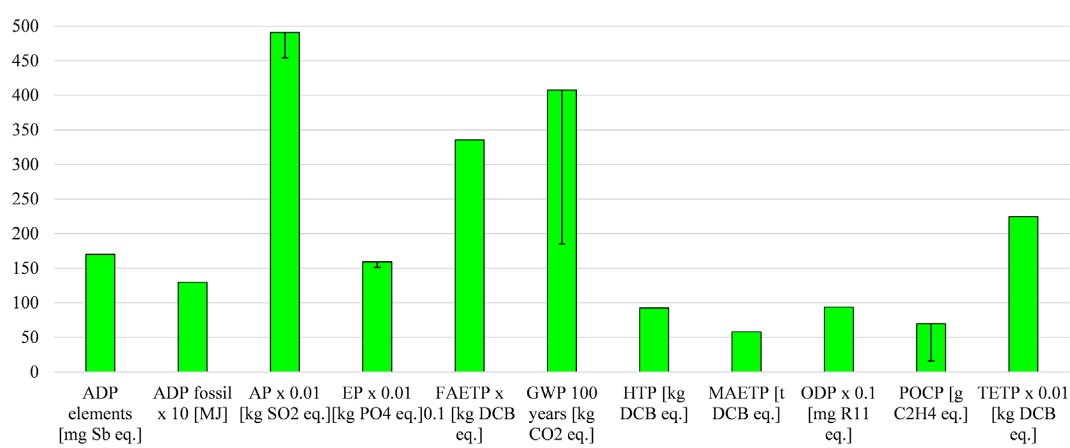

FIGURE 12 | Sensitivity analysis assuming the covered storage of digestate in Plant 4. All impacts expressed per megawatt hour of electricity. For impacts nomenclature, see **Figure 5**.

(**Figure 5D**), it is important to consider by how much the impacts would change if the digestate were stored in covered tanks, as in Plant 2.

The results in **Figure 12** suggest that both impacts would decrease significantly: GWP by two times and POCP threefold. In that case, Plant 4 would have lower impacts than Plant 1 and 3 but still higher than Plant 2. The AP and EP results would also be reduced, by 7 and 5%, respectively, because of the avoided ammonia emissions. This would make Plant 4 a better option than Plant 2 for these two impacts.

With respect to grid electricity, Plant 4 would have half the GWP. It would also be a better option for POCP with respect to solar PV and waste power plants.

CONCLUSION

The aim of this study was to evaluate the life cycle environmental impacts associated with generation of electricity from biogas produced by AD of agricultural products and waste. Five real AD-CHP plants situated in Italy were considered and compared to electricity from the national grid, natural gas, and different renewable technologies.

The results suggest that the main contributors to the impacts from biogas electricity are the production of the maize silage and the operation of the anaerobic digester, including open storage of digestate. Therefore, the system using animal slurry (Plant 5) is the best option among the five plants considered, except for marine and terrestrial ecotoxicity potentials for which the best system is the one utilizing slurry, agricultural waste, and a small amount of maize silage (Plant 1). The plant fed with maize ear silage (Plant 3) is the worst option because of the high impacts of the feedstock, which are almost double that of maize silage.

In reference to the size of AD-CHP plants, larger capacity does not appear to have a positive effect on environmental impacts despite the higher efficiencies typically associated with economies of scale. This is due to the larger plants requiring a high organic load to make them viable, which can only be achieved with cereal feedstocks as they have much higher biogas yield than slurry or agricultural waste. For example, a 1 MW CHP plant requires around 50 ton of maize silage per day but 400–800 ton of slurry. As this amount of slurry cannot be supplied by a single farm, it would have to be collected from different farms and transported to the plant which would not be economically and environmentally viable. Furthermore, the digester would be impractically large (20,000–40,000 m^3 assuming a hydraulic retention time of 50 days) and thus expensive. Therefore, as the results of this work suggest, it is better to have smaller plants using slurry and waste rather than bigger installations: the latter may be more efficient but require cereal silage, which in turn leads to higher environmental impacts. On the other hand, smaller plants require more resources for construction per unit of electricity generated, so there are some trade-offs.

The results also suggest that utilizing the heat generated by the CHP plant would reduce all the impacts, some of them significantly (specifically depletion of fossil fuels and the ozone layer, global warming, and summer smog), making biogas electricity a better option for these categories than any other renewable alternatives considered here. Recycling the AD and CHP construction materials would reduce the depletion of elements, acidification, freshwater, and marine toxicity as well as summer smog. The latter would also improve in addition to global warming if digestate was stored in covered tanks.

Biogas electricity is environmentally more sustainable than electricity from the grid for seven out of 11 impacts considered. This is due to the high contribution of fossil fuels in the Italian electricity mix. The remaining four impacts, for which grid electricity is a better option, are depletion of elements, acidification, eutrophication, and terrestrial ecotoxicity. Thus, biogas electricity reduces GHG emissions compared to the grid, as intended by government and the European Commission, but aggravates some other impacts.

However, in comparison with natural gas, seven out of 11 impacts are higher for electricity from biogas. It also has mostly higher impacts than the renewables, except for solar PV for which six out of 11 impacts are higher than biogas. Furthermore, biogas is a better option than geothermal power for acidification across all the feedstocks considered. If only slurry is used (Plant 5), it also has lower global warming and summer smog potentials than geothermal. Moreover, marine ecotoxicity is greater for electricity from municipal solid waste than that from biogas.

Focusing on global warming potential which drives biogas production, using slurry as a feedstock (Plant 5) is the best option across all the electricity options considered here, sequestering 395 kg CO_2 eq./MWh. All the other biogas systems generate higher greenhouse emissions than any of the renewable options considered here. The only other impact for which biogas electricity is a better option than any other is summer smog, but only for the slurry feedstock; however, it also has higher terrestrial ecotoxicity than any other electricity alternative.

In summary, biogas electricity can help reduce GHG emissions relative to fossil-intensive grid electricity such as that of Italy; however, some other impacts are increased. On the other hand, if mitigation of climate change is the main aim, then other renewables have a greater potential to reduce GHG emissions. If, in addition to this, other impacts are considered, then hydro, wind, and geothermal power are better alternatives to biogas. However, if the subsidies for heat utilization are successful, the environmental sustainability of biogas electricity would improve significantly, particularly for global warming, summer smog, and depletion of the ozone layer and abiotic resources. Further policy changes should include a ban on open digestate storage to prevent methane emissions and regulation on digestate spreading on land to minimize emissions of ammonia and related environmental impacts.

Finally, it should be noted that the results obtained in this study correspond to mesophilic digestion at 40°C and may differ from the results for other operating conditions. Furthermore, the analysis did not consider other environmental aspects, such as habitat destruction and biodiversity loss, as they are outside the scope of LCA. These and other impacts could be evaluated in future research alongside economic costs and social impacts as part of a broader sustainability assessment.

AUTHOR CONTRIBUTIONS

AA and MF conceived and supervised the work; JB collected the data; AF carried out the LCA study; AA, AF, and JB wrote the paper.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at http://journal.frontiersin.org/article/10.3389/fbioe.2016.00026

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ATTACHMENT 47

ENVIRONMENTAL RESEARCH
LETTERS

LETTER • OPEN ACCESS

At scale, renewable natural gas systems could be climate intensive: the influence of methane feedstock and leakage rates

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LETTER

At scale, renewable natural gas systems could be climate intensive: the influence of methane feedstock and leakage rates

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Abstract

Renewable natural gas (RNG) is a fuel comprised of essentially pure methane, usually derived from climate-neutral (e.g. biogenic or captured) carbon dioxide (CO_2). RNG is proposed as a climate friendly direct substitute for fossil natural gas (FNG), with the goal of enabling diverse natural gas users to continue operating without substantial infrastructure overhauls. The assumption that such substitution is climate friendly relies on a major condition that is unlikely to be met: namely, that RNG is manufactured from waste methane that would otherwise have been emitted to the atmosphere. In practice, capturable waste methane is extremely limited and is more likely to be diverted from a flare than from direct atmospheric release in a climate-conscious policy context, which means that RNG systems need to be more destructively efficient than a flare to provide climate benefits versus the likely alternative management strategy. Assuming demand levels consistent with the goal of using existing FNG infrastructure, RNG is likely to be derived from methane that is either intentionally produced or diverted from a flare, so essentially any methane leakage is climate additional. Further, in a decarbonizing system, RNG will likely compete with lower-emissions resources than FNG and thus provides fewer net emissions benefits over time. Anticipated leakage is climatically significant: literature estimates for methane leakage from biogas production and upgrading facilities suggest that leakage is in the 2%–4% range (mass basis), up to as much as 15%. Policy makers should consider that under reasonable leakage and demand assumptions, RNG could be climate intensive.

1. Introduction

Climate change motivates an urgent global transition away from the use of fossil fuels for energy (Intergovernmental Panel on Climate Change 2014, Geels *et al* 2017, Mccauley and Heffron 2018, Davis *et al* 2018). Fossil fuels account for 85% of global commercial energy consumption (2018) (BP 2019) and dominate global energy infrastructure. Given the scale, costs, and economic implications of abandoning infrastructure before the end of its useful life, and given the challenge of transitioning energy systems quickly, there is substantial interest in the idea of renewable drop-in fuels (Rye *et al* 2010, Horvath 2016, Lynd 2017) that can use existing infrastructure without creating the problems of fossil fuel use. This interest is particularly salient in the context of end uses that use specific fossil fuels directly. For example, transportation services currently rely primarily on refined oil, and many industrial and other heating applications directly burn natural gas. These direct users of fossil fuels are often unable to accommodate alternative fuels without abandoning functional infrastructure (e.g. internal combustion engine cars and natural gas-fired water heaters) in favor of infrastructure compatible with the new fuels (e.g. electric cars and electric water heaters).

This work assesses renewable natural gas (RNG) (Götz *et al* 2016, Gasper and Searchinger 2018), here referring both to biomethane (Parker *et al* 2017, Paolini *et al* 2018) and power-to-gas (Götz *et al* 2016, Collet *et al* 2017), as a direct substitute for fossil natural gas (FNG). Drop-in substitutes for FNG specifically are valued due to the diversity of uses, and thus infrastructure, for FNG. In the United States (US), FNG accounts for about 30% of primary energy consumption (EIA 2018), split relatively evenly among power generation (~40%), industrial uses (~30%), and commercial and residential uses (~30%) (Energy Information Administration 2020). RNG has been proposed as a way to decarbonize this system while leveraging existing fossil infrastructure, including pipelines and end use equipment like home and industrial heating devices (Washington State University Energy Program 2018, Bataille 2019). This substitution is particularly relevant for non-electricity uses because they are often more difficult to decarbonize (Davis *et al* 2018, Bataille 2019), though RNG is also valued as an electricity fuel because RNG plants could provide fully dispatchable electricity generation that could reduce the need for costly electricity storage or demand management (Tarroja *et al* 2020). Similarly, like hydrogen, RNG manufacturing has been proposed as a sink for excess variable electricity that can be stored for later use, in the form of power-to-gas schemas (Götz *et al* 2016).

Like FNG, RNG is primarily methane (Gasper and Searchinger 2018), a potent greenhouse gas (GHG) (Intergovernmental Panel on Climate Change 2014) second only to carbon dioxide (CO_2) in its overall contribution to climate change (Weyant *et al* 2006). When RNG is produced from waste methane, converting it to CO_2 by burning it has climate benefits because of methane's much higher climate forcing potential (Intergovernmental Panel on Climate Change 2014) relative to CO_2. If the waste methane were going to be emitted to the atmosphere anyway, any system leakage (i.e. methane emissions) is a lost opportunity but not a climate stressor; otherwise, it contributes to climate change. This analysis shows that 1) RNG from intentionally produced methane, even from climate-neutral CO_2 sources, has substantial climate impacts at methane leakage levels observed in the existing, mature biogas industry (Pertl *et al* 2010, Flesch *et al* 2011, Whiting and Azapagic 2014, Ravina and Genon 2015, Hijazi *et al* 2016, Liebetrau *et al* 2017, Paolini *et al* 2018, Vo *et al* 2018, Ramírez-Islas *et al* 2020); (2) for any meaningful system scale, RNG is likely to be derived from intentionally produced methane; and (3) even RNG from waste methane can have negative climate impacts relative to the most likely alternative of flaring, not venting, the methane when leakage from RNG production and use exceeds flaring loss rates.

2. Methods

This analysis evaluates the GHG intensity of RNG, focused on three methane feedstock pathways for RNG production: (1) from waste methane that would have otherwise been emitted to the atmosphere; (2) from waste methane that would have otherwise been flared; and (3) from intentionally created methane that would otherwise not have existed. The carbon for RNG is assumed to be climate neutral, for example, biogenic or sourced from a carbon capture activity. Reported GHG intensities use IPCC's Fifth Assessment Report (AR5) 20- and 100-year GWPs with climate-carbon feedback, distinguishing between fossil and nonfossil methane GWPs (see Working Group 1, chapter 8, Tables 8.7 and 8.A.1). For comparison, the GHG intensity of FNG, generic resources with life cycle 2050 GHG intensity consistent with 2 °C warming (Pehl *et al* 2017), and zero carbon resources are also included. Full details and calculations are available in the Supplementary Data File.

The absolute GHG intensity of RNG is assumed to derive from methane leakage only (because combustion GHG emissions for RNG are climate neutral by assumption), drawing the system boundary at the point when the methane is diverted from the alternative management strategy (venting, flaring, or not existing) and excluding embodied GHGs in infrastructure or any of the production feedstocks. For example, power-to-gas pathways are implicitly assumed to use GHG-neutral power in facilities with zero embodied GHGs. GHG intensity is given as kilograms (kg) of carbon dioxide equivalent (CO_2e) per gigajoule (GJ) of methane consumed—that is, the denominator is the amount of methane that is ultimately delivered to the entity that combusts it, which is less than the amount of methane that is produced or withdrawn if system leakage exceeds 0%. Emissions associated with leakage are thus calculated as follows:

$$\begin{aligned}&\text{absolute leakage} - \text{related GHG intensity} = \\ &(\text{mass CH}_4\text{ produced} \times \text{system leakage} \times \text{GWP}_{\text{CH}_4}) \\ &= \frac{\text{mass CH}_4\text{ delivered}}{(1 - \text{system leakage})} \times \text{system leakage} \times \text{GWP}_{\text{CH}_4}\end{aligned} \tag{1}$$

where system leakage is mass methane emitted/mass methane produced. Net emissions relative to the alternative fate for methane are calculated by subtracting the counterfactual methane emissions. For Path 1 (waste methane would have otherwise been emitted to the atmosphere), counterfactual emissions are that system leakage = 100%. For Path 2 (waste methane would have otherwise been flared), counterfactual emissions are that system leakage = (1-flare efficiency). For Path 3 (methane would not otherwise have existed), counterfactual emissions are 0. Thus, net emissions are given as:

$$\begin{aligned}&\text{net leakage emissions} = (\text{mass CH}_4\text{ produced} \\ &\quad \times \text{GWP}_{\text{CH}_4}) \times (\text{system leakage} - \text{counterfactual leakage})\end{aligned} \tag{2}$$

For the three paths, Equation 2 becomes:

$$\text{Path 1, net leakage emissions} = (\text{mass } CH_4 \text{ produced} \times GWP_{CH_4}) \times (\textit{system leakage} - 1) \quad (3)$$

$$\text{Path 2, net leakage emissions} = (\text{mass } CH_4 \text{ produced} \times GWP_{CH_4}) \times (\text{system leakage} - \text{flare leakage}) \quad (4)$$

$$\text{Path 3, net leakage emissions} = (\text{mass } CH_4 \text{ produced} \times GWP_{CH_4}) \times (\text{system leakage} - 0) \quad (5)$$

Note that because of the presentation of results per unit of methane combusted by a user and the fact that methane production = methane delivered/(1—system leakage) (equation (1)), equation (3) reduces to (mass CH_4 delivered $\times -GWP_{CH_4}$) and is independent of leakage rate. For Path 2, emissions upstream of the flare are not considered because waste methane that is vented prior to diversion to the flare is not diverted to the flare, thus falling under Path 1.

Leakage from the RNG system is evaluated as a range because of substantial uncertainty about what leakage levels would be under future conditions, particularly if newer RNG pathways (e.g. power-to-gas) became widespread. This work considers the implications of RNG system leakage between 0%–15% mass leakage/mass produced in order to inform consideration of potential RNG futures. The range is most proximately based on Scheutz and Fredenslund's (2019) evaluation of 23 biogas plants, including seven facilities encompassing production through biogas upgrading to biomethane, where facility leakages from 0.4 to 14.9% of production were observed (see Supplementary Data File for details). Specific leakage sources are not always evident but might be correlated with plant complexity (e.g. number of units), maintenance regimes, and the status of biogas production as a core or non-core function (Scheutz and Fredenslund 2019). Published values in other studies and GHG protocols reflect ranges generally narrower than but consistent with Scheutz and Fredenslund's findings (Flesch *et al* 2011, Liebetrau *et al* 2013, 2017, Hrad *et al* 2015, Vo *et al* 2018, Bartoli *et al* 2019).

Leakage downstream of production and processing (i.e. during transportation, storage, and end use) is assumed to be identical for FNG and RNG. Although it is challenging to assign a value for these processes due to the diversity of end uses, lack of information about leakage during end uses, differential use of transmission, storage, and distribution by end users, and the dependence of transportation leakage on distance, a value of 0.8% (mass leaked per mass withdrawn or produced) was chosen in service of estimating absolute GHG intensities for comparison with zero-GHG systems. This value is based on assumptions and data from the literature (Liebetrau *et al* 2013, Lavoie *et al* 2017, Alvarez *et al* 2018) (see Supplementary Data File for details). For FNG, this value is added to an estimate of production and processing leakage of 2.1% mass leakage/mass withdrawn (Alvarez *et al* 2018), and for RNG, it is assumed to be included in the assessed 0%–15% range for full system potential leakage rates.

3. Results

3.1. GHG intensity of renewable natural gas

Table 1 shows the estimated GHG intensity of RNG for three production pathways as a function of system methane leakage: (1) RNG produced from waste methane that would have otherwise been emitted to the atmosphere (Path 1); (2) RNG produced from waste methane that would have otherwise been flared with 99% destructive efficiency (Path 2); and (3) RNG produced from intentionally created methane (Path 3).

In all cases, this analysis assumes that the CO_2 emitted from burning RNG is climate neutral for the RNG user, e.g. because it is sourced from biogenic or captured carbon that was taken up from and returned to the atmosphere over a period of time that is short from a climate perspective. Further, to emphasize the particular challenge posed by methane leakage, this analysis assumes RNG has no GHG intensity other than that associated with net impacts from methane leakage or destruction—that is, inputs to RNG production like electricity, hydrogen, and support infrastructure are assumed to be climate neutral. This assumption is consistent with the notion that zero-GHG electricity or hydrogen are potential alternatives to RNG. Note that because conversion processes are never 100% efficient, any GHG intensity for RNG associated with electricity or hydrogen inputs would exceed that of the electricity or hydrogen available for use.

As table 1 shows, the GHG intensity of RNG is driven by the counterfactual—that is, what would have otherwise happened to the source methane. Path 1, waste methane diversion from the atmosphere, is highly GHG negative because the counterfactual is that all utilized methane would have been emitted as methane. Leakage is irrelevant to GHG impact per unit of utilized methane because any leaks are methane that would have escaped anyway. Although Path 2 also uses waste methane, the counterfactual is that the waste methane would have been nonproductively burned in a flare, so RNG is GHG negative in this case only if the RNG system's total leakage is lower than leakage from the flare (1%), which is unlikely given that a best-guess estimate of downstream emissions alone is 0.8%. Path 3 uses intentionally produced methane. Here, the counterfactual is that no methane would have been released to the atmosphere, so any system leakage is GHG positive.

Table 1. Renewable natural gas carbon dioxide equivalent intensity by pathway, assuming climate-neutral combustion emissions of carbon dioxide.

(a) GWP-100, kg CO2e/GJ methane productively consumed							
system leakage (mass CH4 emitted/mass CH4 produced)	0	0.025	0.05	0.075	0.1	0.125	0.15
Path 1: Waste methane diverted from emission to atmosphere	−680	−680	−680	−680	−680	−680	−680
Path 2: Waste methane diverted from a 99% efficient flare	−7	10	29	48	68	89	112
Path 3: Intentionally produced methane used	0	17	36	55	76	97	120
(b) GWP-20. kg CO2e/GJ methane productively consumed							
system leakage (mass CH4 emitted/mass CH4 produced)	0	0.025	0.05	0.075	0.1	0.125	0.15
Path 1: Waste methane diverted from emission to atmosphere	−1720	−1720	−1720	−1720	−1720	−1720	−1720
Path 2: Waste methane diverted from a 99% efficient flare	−17	26	72	121	172	226	283
Path 3: Intentionally produced methane used	0	44	91	139	191	246	304

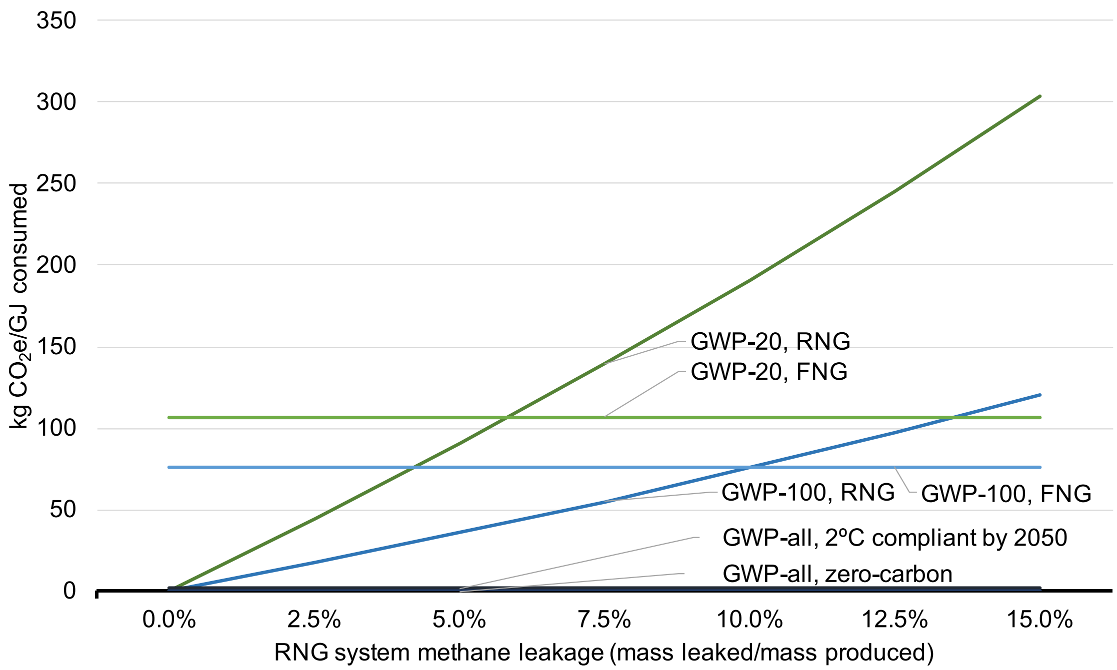

Figure 1. Greenhouse gas intensity of renewable natural gas from intentionally produced methane as a function of system methane leakage. GWP-20 and GWP-100 are shown for intentionally produced RNG for RNG system methane leakage of 0–15%, with GWP-20 and GWP-100 of FNG shown for reference.

3.2. Intentionally produced methane for RNG

A major finding of this analysis is that, as with FNG (Brandt *et al* 2014, Alvarez *et al* 2018, Grubert and Brandt 2019, Zhou *et al* 2019), RNG can have significant climate impacts associated with system methane leakage if the methane is intentionally produced (Path 3; table 1). Figure 1 shows the GHG intensity (kg CO_2e/GJ CH_4 productively consumed, e.g. for heat or electricity generation) of RNG from intentionally produced methane as a function of RNG system methane leakage.

As figure 1 shows, RNG from intentionally produced methane is always GHG positive unless total system leakage is 0. Given demonstrated transportation and end use leakage values on the order of 0.4–0.8% (Liebetrau *et al* 2013, Lavoie *et al* 2017, Alvarez *et al* 2018), RNG from intentionally produced methane cannot outperform zero-GHG hydrogen or electricity systems on GHG intensity. Although this analysis does not consider non-operational, non-methane GHGs, note that both hydrogen and electricity are likely inputs to intentionally produced methane for RNG, which therefore inherits and amplifies embodied emissions. The estimated methane-only GHG footprint of such RNG exceeds the combustion plus methane leakage GHG footprint of FNG when RNG system leakage is higher than about 10% (GWP-100) or 6% (GWP-20) on a mass leaked per mass produced basis. Accounting for IPCC stated uncertainty in the GWP of methane (Intergovernmental Panel on Climate Change 2014), the estimated leakage range within which RNG becomes more GHG intensive than FNG is about 9.1–11.1% (GWP-100) or 5.0–6.6% (GWP-20). Although power-to-gas systems and evaluations remain rare enough that data on leakage are not widely available (though leakage has been discussed (Vo *et al* 2018)), such leakage rates—particularly for a full system—are not uncommon for biogas to RNG systems (Liebetrau *et al* 2013, 2017, Scheutz and Fredenslund 2019). For electricity, assuming the heat rate of a US natural gas combined cycle power plant and GWP-100, RNG's operational methane GHG intensity surpasses the 15 kg CO_2e/MWh total life cycle 2050 GHG intensity consistent with a 2 °C warming limit (Pehl *et al* 2017) for system leakage of 0.3%, which is less than some observed leakage from power plants alone (Lavoie *et al* 2017). Calculations can be found in the Supplementary Data File.

3.3. At scale, most methane feedstocks for RNG would likely be intentionally produced

How much RNG is likely to come from intentionally produced methane, which includes all power-to-gas RNG and RNG produced from feedstocks that would not degrade anaerobically (i.e. to methane rather than CO_2) absent intentional intervention (Meyer-Aurich *et al* 2012, Börjesson *et al* 2015, Agostini *et al* 2015)? The answer depends on assumptions about total demand and the availability of waste methane for diversion to RNG production. If the goal is to maintain the usefulness of FNG infrastructure, one potential assumption for RNG demand is that it would match current FNG demand. In 2017, US consumer consumption was 27.2 exajoules (EJ) of FNG, including 10.1 EJ for electric power and 8.7 EJ for often difficult-to-decarbonize industrial uses (see Supplementary Data File). The energy content of 2017 US uncontrolled methane emissions was about 1.6 EJ/year, about 0.3 EJ of which were emitted from biogenic sources (as opposed to, say, the FNG system) that could reasonably be captured (wastewater treatment plants and landfills, not enteric fermentation) (US EPA 2019, Grubert 2020), not all of which would become consumable RNG (i.e. due to parasitic energy requirements, conversion losses, etc). Thus, although some capturable waste methane (Paths 1 and 2) clearly exists, the degree to which RNG systems can depend on such resources at scale is low (<1%) relative to current natural gas demand.

One important observation for contextualizing these values is that not all methane from waste is waste methane. For example, the National Renewable Energy Laboratory (NREL) estimates that the energy content potential from methanogenic US wastes is about 5% the size of the US natural gas system (National Renewable Energy Laboratory 2013), including a methane potential estimate from wastewater (2.3 million tonnes/year) (National Renewable Energy Laboratory 2013) that is four times the US Greenhouse Gas Inventory (GHGI) estimate for methane emissions from wastewater treatment plants as of 2017 (0.57 million tonnes/year, Table 2–2) (US EPA 2019). Why? Unintentionally produced waste methane typically results from natural anaerobic digestion of wet organic wastes, like animal manure, sewage, and landfilled wastes, but this digestion process does not completely convert carbon wastes to methane. Rather, digestion produces biogas, a blend of methane and CO_2 that can then be upgraded into near-pure biomethane, a form of RNG. Crucially, in part because biogas and biomethane can generate revenue, it is not only possible but expected to intervene in biological systems to increase methane production beyond what would have happened anyway when there is an incentive to do so (Hijazi *et al* 2016, Ferreira *et al* 2019, Garcia *et al* 2019). Thus, a single facility might produce both Path 1 (GHG-negative) and Path 3 (GHG-positive) methane from the same wastes.

Despite its limited availability, Path 1 methane is so GHG negative (table 1) that it is reasonable to investigate whether climate benefits can be retained if small amounts of very climate-negative RNG are blended with RNG from intentionally produced methane. Figure 2 shows GHG intensity of RNG for blends of 0%–100% Path 3 methane (intentionally produced) with Path 1 methane (waste methane diverted from release).

Figure 2 suggests that blending very GHG-negative RNG with GHG-positive RNG can enable a fairly large RNG system that is overall at least somewhat GHG-negative, assuming leakage levels within a typically observed range (see Supplementary Data File for detailed calculations and values). Assuming all 0.3 EJ of uncontrolled methane emissions from landfills and wastewater treatment plants could be captured and converted to consumer-ready RNG, either current industrial demand or electric power demand for FNG could be fulfilled by an RNG system with up to about 3% system leakage and theoretically remain GHG-negative (GWP-100; see Supplementary Data File). As the next section shows, however, such an outcome is unlikely because of the actual nature of waste methane management.

3.4. Capturable waste methane would be flared, not vented

The possible conclusion that sufficient highly GHG negative methane exists to support a large (e.g. FNG electricity system-sized) GHG negative RNG system is based on the assumption that waste methane is diverted from emission to the atmosphere (Path 1). This assumption is flawed if one also assumes that GHG emissions reductions are a policy priority, as existing practice is not the appropriate baseline for determining the counterfactual management practice for waste methane that could be available for RNG production (Haya *et al* 2019). Specifically, if the methane can be captured for RNG production, it can be captured for diversion to a flare, and it is unrealistic to assume that capturable methane would be vented under a GHG conscious policy regime. Even without federal climate regulation, the US regulates methane emissions from new landfills (US EPA 2016),

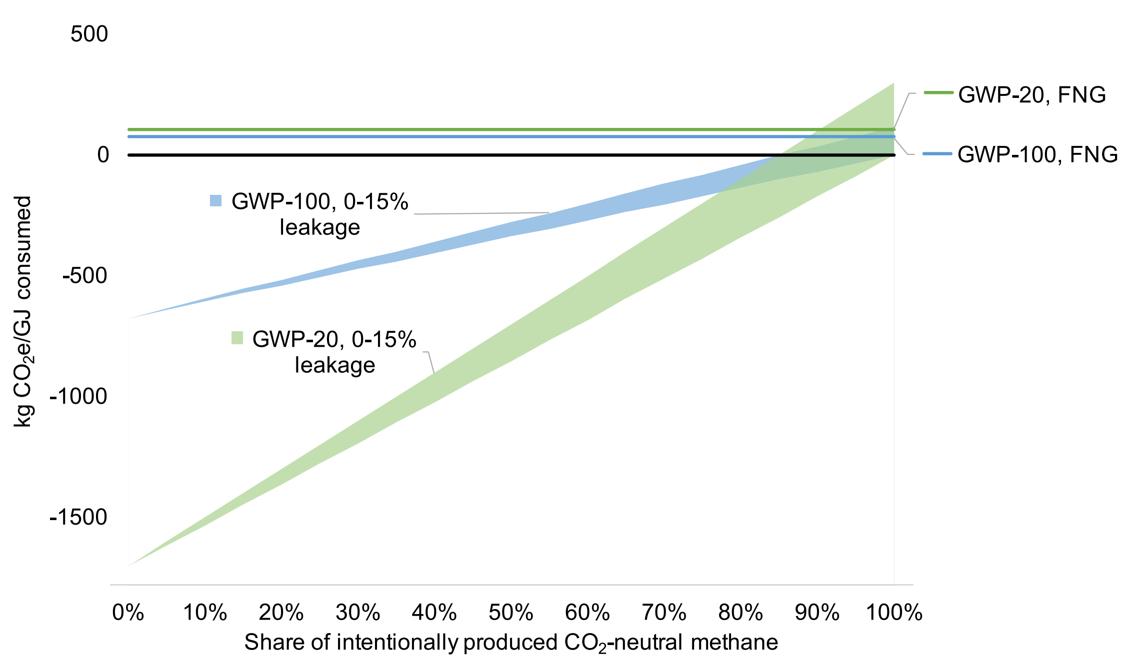

Figure 2. Greenhouse gas intensity of RNG systems that blend waste methane that would otherwise have been released to the atmosphere with intentionally produced methane, for 0%–15% system leakage. Lower bound of shaded areas = 0% system methane leakage for RNG; upper bound of shaded areas = 15% system methane leakage for RNG. GWP-20 and GWP-100 for RNG and FNG are shown for blends of 0% to 100% RNG from intentionally produced methane.

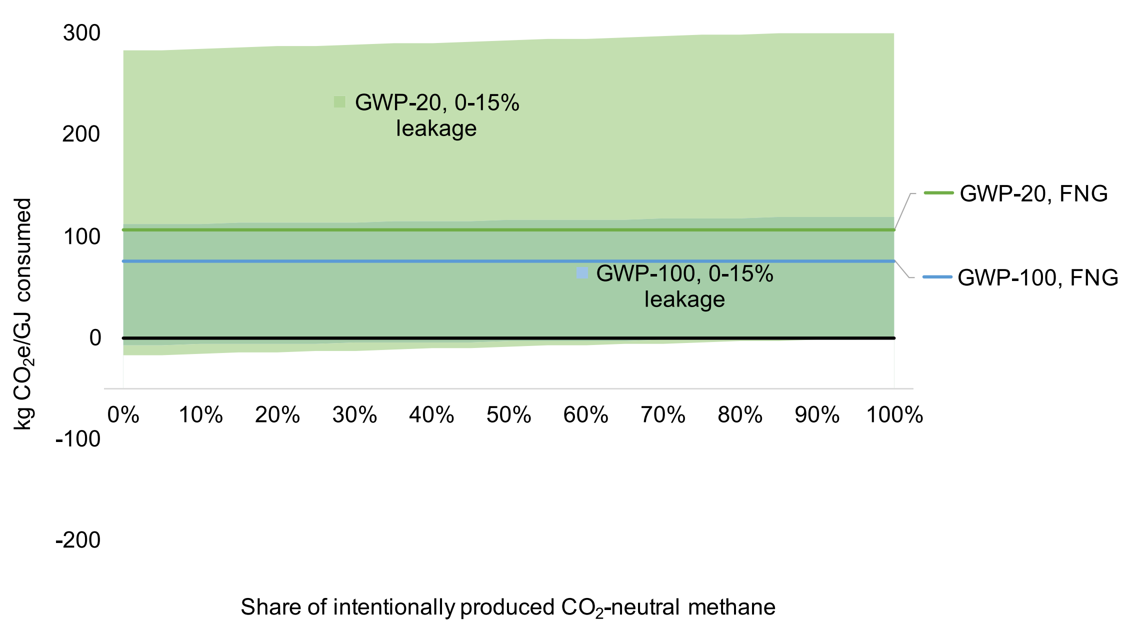

Figure 3. Greenhouse gas intensity of RNG systems that blend waste methane that would otherwise have been flared with intentionally produced methane, for 0%–15% system leakage. Panel A: Flare with 99% destruction efficiency. Panel B: Flare with 86% destruction efficiency. Lower bound of shaded areas = 0% system methane leakage for RNG; upper bound of shaded areas = 15% system methane leakage for RNG. GWP-20 and GWP-100 for RNG and FNG are shown for blends of 0% to 100% RNG from intentionally produced methane.

and many methanogenic facilities use methane capture with flares for safety reasons. Flaring destroys the methane with the same destructive benefit as combusting the methane productively. Figure 3 updates the assumptions used in figure 2 to show the same results, but assuming that RNG using Path 3 (intentionally produced) methane is blended with RNG using Path 2 (waste diverted from flare) rather than Path 1 (waste diverted from release) methane. Figure 3 assumes 99% flare efficiency, described in the GHGI as a median value (US EPA 2019). Figure S1 (available at stacks.iop.org/ERL/15/084041/mmedia) shows results assuming the GHGI's lower flare efficiency bound of 86 (US EPA 2019).

As figure 3 shows, conclusions about the viability of a large, GHG-negative RNG system change radically when the more realistic counterfactual of methane destruction rather than methane venting is applied. RNG system leakage would need to be essentially 0 (that is, lower than the flare's leakage) to be GHG-negative versus typical flare performance. Based on literature values for leakage, including the estimate of 0.8% leakage for processes downstream of production and processing, productive use of waste methane is unlikely to be more destructively efficient than a flare. Although waste methane being diverted for productive use arguably would not have been captured without the financial incentive of energy sales, given that capture infrastructure is not free, flaring is most likely the less GHG intensive alternative for waste methane once it has been captured. In a decarbonized energy system where RNG would be less likely to be replacing GHG-intensive fuels (and thus offsetting their emissions), and when a policy regime requiring or incentivizing destruction of GHG-intensive wastes might reasonably be expected to be in place, expected levels of methane leakage suggest that RNG is unlikely to be a low GHG energy resource relative to alternatives.

4. Discussion

RNG is not inherently climate friendly. Based on consideration of both the source of methane used to produce RNG and the likely alternative fate of that methane, and using reasonable assumptions about likely system methane leakage, it is unlikely that an RNG system could deliver GHG-negative, or even zero GHG, energy at scale. Substantial GHG benefits can be attained when waste methane is genuinely diverted from emission to the atmosphere, but the availability of such methane is low (Liu and Rajagopal 2019, US EPA 2019) relative to potential demand for climate friendly RNG, especially when considering that the alternative fate of capturable methane is more likely flaring than venting in a GHG-conscious setting. Under some system leakage rates that have been observed for biogas systems (Liebetrau *et al* 2017, Scheutz and Fredenslund 2019), RNG might not even meet the less stringent threshold of outperforming FNG from a GHG perspective.

Designing a system that depends on RNG, or delaying transition to a system that does not depend on natural gas because of the promise of RNG, could delay climate mitigation because of induced demand for intentionally produced methane. Particularly given that past experience demonstrates that policy can rapidly drive resource allocation to RNG (Bartoli *et al* 2019), RNG's environmental performance should be carefully compared with that of its likely long-term competitors—not just FNG—before resources are allocated. Current literature on RNG often assumes the context of a fossil-based system (see e.g. the reference systems for papers included in a review of LCA of biogas production (Hijazi *et al* 2016)), which leads to the crediting of lower environmental burden relative to this context (e.g. when RNG is given credit for avoided GHG impacts from FNG consumption (Scheutz and Fredenslund 2019, Ramírez-Islas *et al* 2020)). Such fossil-linked benefits disappear in a context where RNG could be substituting for zero-GHG alternatives like zero-GHG electricity or hydrogen rather than FNG, petroleum fuels, and GHG-intensive electricity.

Even beyond GHG emissions, environmental burdens associated with RNG that are acceptable relative to FNG merit deeper investigation when the alternative is, e.g. zero-GHG electricity. RNG is designed to be effectively indistinguishable from FNG at the point of use, so local combustion impacts are likely to be similar for clean RNG, and potentially worse for less pure RNG (Paolini *et al* 2018). Upstream of use, RNG would likely have different socioenvironmental impacts than FNG. Although RNG can use existing pipeline and user infrastructure, for example, it would obviate the need for FNG's production infrastructures, which have substantial socioenvironmental impacts (Jacquet *et al* 2018). RNG production facilities using primarily waste products (e.g. agricultural wastes, landfill gas, wastewater treatment gas, excess electricity generation) would likely not qualitatively change socioenvironmental impacts from those activities, though making certain practices financially viable could extend their life and extent (Haya *et al* 2019). Relatedly, having access to RNG could extend the life of existing fossil infrastructure, with mixed socioenvironmental outcomes.

To the extent that RNG facilitates lower impact energy systems, e.g. by avoiding the need for mineral (Sovacool *et al* 2020)- and cost-intensive electricity storage to help match supply and demand (Tarroja *et al* 2020), some of the marginal impacts of RNG could be offset by system benefits. These benefits are not guaranteed, however. As demonstrated by experience with renewable drop-in transportation fuels, the potential for drop-in renewable fuel use might not actually lead to renewable fuel use (Pouliot and Babcock 2017), and the renewable fuels themselves might have undesirable environmental characteristics (Liu and Rajagopal 2019). This work shows that RNG needs to be carefully evaluated in the context of expected long-run system conditions before it is adopted as a component of a zero GHG energy system, particularly given its potential for methane leakage-related climate pollution.

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Data Availability Statement

Any data that support the findings of this study are included within the article.

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ATTACHMENT 48

Analysis of greenhouse gas emissions from 10 biogas plants within the agricultural sector

J. Liebetrau, T. Reinelt, J. Clemens, C. Hafermann, J. Friehe and P. Weiland

ABSTRACT

With the increasing number of biogas plants in Germany the necessity for an exact determination of the actual effect on the greenhouse gas emissions related to the energy production gains importance. Hitherto the life cycle assessments have been based on estimations of emissions of biogas plants. The lack of actual emission evaluations has been addressed within a project from which the selected results are presented here. The data presented here have been obtained during a survey in which 10 biogas plants were analysed within two measurement periods each. As the major methane emission sources the open storage of digestates ranging from 0.22 to 11.2% of the methane utilized and the exhaust of the co-generation units ranging from 0.40 to 3.28% have been identified. Relevant ammonia emissions have been detected from the open digestate storage. The main source of nitrous oxide emissions was the co-generation unit. Regarding the potential of measures to reduce emissions it is highly recommended to focus on the digestate storage and the exhaust of the co-generation.

Key words | biogas, greenhouse gas emissions, quantification of emission sources

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INTRODUCTION

The production of renewable energy from biogas plants plays an increasingly important role in Germany with close to 6,000 plants and an installed capacity greater than 2,000 MW at the end of 2010. One of the main targets of the implementation of the subsidies for energy production from renewable sources is the reduction of greenhouse gases originating from the energy sector. The evaluation of the efficiency regarding the reduction of the effects on the climate is usually done by means of a life cycle assessment (LCA). However, the assessment can only produce reliable data if it is based on a precise determination of the process related greenhouse gas emissions.

In particular methane emissions have a large influence on the LCA results due to the CO_2 equivalent of methane of 25 (IPCC 2007). With the background of current discussions about sustainability criteria for energy from biomass, the identification of greenhouse gas emission along the whole production process gains more and more importance.

So far, the overall emissions of biogas facilities have been estimated by theoretical assumptions, but actually only by means of direct measurement of emissions does a veritable analysis become possible. There are in general two methods to determine the emissions of a large industrial facility or areas with diffuse emission sources. One way is to attempt to capture the overall emissions of the facilities by means of concentration measurements in the surroundings and the application of inverse dispersion models (Flesch *et al.* 2011) or radial plume mapping (Hashmonay *et al.* 2008). These methods allow the determination of the overall emissions of a large area with uncertain sources of emission. These methods do not allow the location of single sources and allocation of a certain quantity to them. However, for further efficient measures to reduce emissions it is very important to identify and quantify the emission sources on site. For this reason the methods used in this study focus on the identification and quantification of single sources. The results of the investigation presented were obtained within a project in which the state of the art of biogas plants were investigated, representative plants were selected and the occurring emissions were determined. The aim of the investigation was the determination of emission quantity from the most relevant technical components (substrate storage, feed in devices, digesters, digestate storage, gas transportation and utilization units) within the production process.

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This paper presents data obtained from a measurement program investigating the greenhouse gas emissions of 10 biogas plants. The program included two measurement periods in each plant, in which all plant components, from substrate storage to digestate storage were investigated. Further analysis will include a LCA and the development of proper measures for emission reduction. Based on the measured data and knowledge of the state of the art of the biogas industry it will be possible to calculate emission factors with a greater precision.

METHODS

Table 1 shows the features of the 10 investigated plants. All plants are continuously operated except plant no. 5 which is operated in batch mode with a garage style dry fermentation system. Furthermore eight of the 10 plants convert the biogas into electricity. Plant nos. 8 and 9 inject the biogas to the natural gas grid after an upgrading process.

The method adopted for the emission measurements of different technical components from a biogas plant is based on a survey of the plant targeting leakages or sections of increased concentration. Typical diffuse leakages appear on gas pipe adapters, the connection of the digester foil cover with the digester wall or on inspection windows. The leakage detection is carried out by means of two different gas detection methods, tunable diode laser absorption spectroscopy (TDLAS) and flame ionization detection (FID).

After the identification of a spot with a potential leakage, the area is encapsulated to form an aerated chamber. This measurement method is often used to investigate the emissions of diffuse area sources such as manure storages or lagoons (e.g. Husted 1994; Aneja *et al*. 2001). The application of this method on technical components requires a flexible enclosure made of a gas tight foil. The chamber has an input and output pipe and a connected blower to produce a constant air flow through the chamber. The gas from the emission source (leakage) and the fresh air are mixed in

Table 1 | Features of the 10 investigated biogas plants

	Fermentation system		Operation temperature		Installed capacity (CHP)			
Plant no.	Wet	Dry	Mesophilic	Thermophilic	kW	Main substrates	HRT[a] Process/Total	Digestate storage tank
1		x	x		526, 185	Energy crops	135	Sealed
2	x		x		350	Energy crops, pig manure (25%)	68/178	Open
3	x		x		537	Energy crops, water (12%)	64/135	Open
4	x		x	Sec. step	526	Energy crops, pig manure (30%)	23/69	Covered[b]
5[c]		x	x		526	Energy crops, manure (26%)	28	Open
6		x	x		2 × 249	Energy crops	33	Sealed
7	x		x		2 × 160	Energy crops, cattle manure (86%), chicken manure (8%)	41	Open
8[d]	x				Equal to 2,200	Energy crops	102	Open
9[d]	x		x		Equal to 2,600	Energy crops, pig and cattle manure (46%)	58	Covered[b]
10	x			x	1,000, 526	Energy crops, chicken manure (26%)	90	Open

[a]Hydraulic retention time.
[b]Not gas tight covered.
[c]Plant operates with a garage style fermentation system.
[d]Plant has an gas-upgrading facility.

the space of the chamber and the concentration of the target gas is analysed by sampling the gas in the in- and output stream of the chamber. Methane is detected by gas chromatography with a flame ionization detector (FID), nitrous oxides by gas chromatography and with an electron capture detector (ECD) and ammonia by absorption in an acid solution. The flow rate is measured by a vane anemometer or a Pitot tube. Then the quantity of the emission source is calculated from the concentration difference and the flow rate of the blower by using the following equation.

$$F = Q \cdot \rho \cdot (c_{\mathrm{out}} - c_{\mathrm{in}}) \tag{1}$$

F, emission flow rate ($\mathrm{mg} \times \mathrm{h}^{-1}$); Q, air flow rate ($\mathrm{m}^3 \times \mathrm{h}^{-1}$); ρ, density of the target gas ($\mathrm{kg} \times \mathrm{m}^{-3}$); c_{out} exhaust gas concentration ($\mathrm{mg} \times \mathrm{kg}^{-1}$); c_{in}, background gas concentration ($\mathrm{mg} \times \mathrm{kg}^{-1}$).

The surface of an open digester storage tank is investigated optionally by a closed (for CH_4, N_2O) or an aerated chamber (for NH_3). The aerated chamber (Figure 1) uses the same measuring principle as described above. It is a floatable unit with a surface area of 0.25 m^2. Due to the defined surface area of the chamber a specific emission flow rate ($\mathrm{mg} \times \mathrm{m}^{-2} \times \mathrm{h}^{-1}$) is calculated. In order to measure the ammonia emissions the air flow rate within the aerated chamber is adjusted to 18–36 m^3/h by a blower. An aliquot from the exhaust of the chamber is directed through a wash bottle with an acid solution of 0.05 mol/L H_2SO_4. The ammonia concentration within the solution is determined by a photometric kit. The quantity of the emissions is calculated according to German technical guideline VDI 3496 (VDI 1982) – the emissions from the tank are calculated according to the area ratio between chamber and tank. In case of partial swimming layer, both surfaces (swimming layer and liquid surface) are investigated.

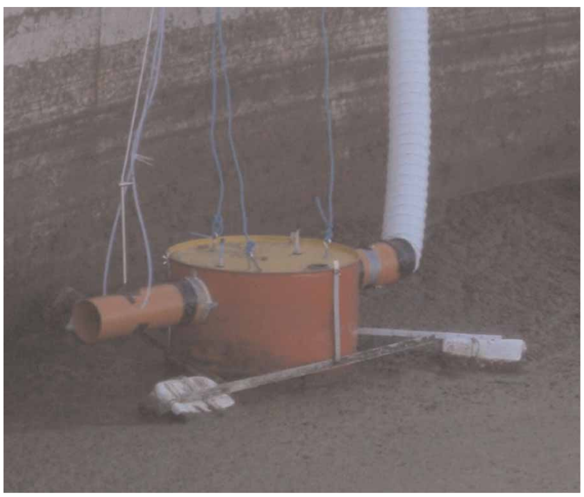

Figure 1 | A floating chamber on an open digestate storage tank.

The closed chamber system is based on a different measuring principle. Once the closed chamber is placed on the emission surface, the gradually increasing concentrations of CH_4 and N_2O under the chamber can be used for calculation of the flow of the gases into the chamber and consequently for the emissions occurring under the defined area of the chamber. Samples are taken after defined time intervals (0, 10, 20, 30 min). The emission rate can be calculated from the slope of the gas concentration, the chamber volume and the encapsulated surface area (0.25 m^2) with Equation (2).

$$F = \frac{\partial c}{\partial t} \cdot \frac{V}{A} \tag{2}$$

F, emission flow rate ($\mathrm{mg} \times \mathrm{m}^{-2} \times \mathrm{h}^{-1}$); $\partial c/\partial t$, slope of the gas concentration ($\mathrm{mg} \times \mathrm{m}^{-3} \times \mathrm{h}^{-1}$); V, chamber volume (m^3); A, encapsulated surface area (m^2).

Similar measurements to investigate diffuse CH_4 emissions from area sources such as soils, landfills or manure storages by using the closed chamber method are described, for example, in Jäkel & Mau (1999), Cardellini *et al.* (2003), Rodhe *et al.* (2009) or Rochette & McGinn (2005).

For the covered, but not gas tight covered digestate storage tanks the measurement of the emissions is carried out differently. Since the surface is not accessible for the chamber measurement, the whole headspace of the tank is aerated by a blower with a defined flow rate (400–500 m^3/h). After the gas under the cover is replaced by fresh air, the exhaust of the tank is sampled for methane and nitrous oxide and a distinct volume is sampled for ammonia concentration as described above.

In the case of gas utilization, in particular, the combined heat and power unit (CHP), the flow and the concentration were evaluated directly in the exhaust pipe according to German technical guideline VDI 4200 (VDI 2000).

On every plant, there were two measurement periods. Each of the periods usually lasted one week per plant. Within this week the several plant components were investigated consecutively. If given as average, the results of the two periods were averaged.

Due to a limited number of possible measurements it was decided to focus on the identification of major sources. The evaluation of plants for a longer period of time would give a better insight into the driving factors of emission, but require a longer period of analysis on site. In order to get more results from different plants it was decided to analyse more plants in short periods of time. In order to ensure comparable results, the plants were only investigated under full load operation. In doing so, no malfunction situation and resulting emissions such as from flaring or overpressure valve releases were evaluated. Any evaluation of malfunction situations should be linked with an investigation regarding the frequency of such occurrences, otherwise the representativeness of the results will be difficult to estimate. The emissions of each component have been set in relation to the installed capacity in order to get an emission value according to the energy output (g CH_4/kWh_{el}). In the case of the investigated plants, the installed capacity matched the actual output at the moment of the measurement.

For reasons of clarity the numbers of methane emissions have been converted into percentage of the utilized methane. The amount of utilized methane is estimated by assuming an electric efficiency of 40% of the co-generation unit and an energy yield of methane of 10 $kWh \times m^{-3}$.

RESULTS AND DISCUSSION

The measurements of the emissions on the plants have been sorted according to the investigated plant components. The following plant components were separately analysed and the results are presented in the following sections:

- Silage storage
- Feeding systems
- Digester
- Digestate storage tanks (sealed, not gas tight covered, open)
- Gas utilization (co-generation units or gas-upgrading facilities).

Silage storage

Methane emissions from the silage were proven to be negligibly low. The average of eight plants where measurements were carried out accounts for 1.25×10^{-3} g CH_4/kWh and the averaged loss of methane was about 0.69×10^{-3}% (eight plants). Emissions of ammonia were also very low, at 2.52×10^{-3} g NH_3/kWh (seven plants evaluated). There were no detectable N_2O emissions.

Feeding systems

There are several types of input systems used within the biogas industry. The following were investigated:

- Screw conveyor (feeds via an open pit, where the solid material is stored and a screw system directly into the digester, under the liquid surface).
- Substrate storage tank (tank, mostly open, where manure is stored prior to feeding).
- Mixing tank (tank, mostly open, where manure, silage and digestate is mixed prior to feeding).

Due to the low pH the silage was quite stable, therefore relevant emissions only occurred in case of mixing the fresh substrate before feeding with manure or digestate in an open tank. The results of the measurements of the feeding systems are shown in Table 2.

The emissions from one screw conveyor, two mixing tanks and one substrate storage tank appeared to be

Table 2 | Methane emissions from feeding system

Plant no.	Screw conveyor		Substrate storage tank			Mixing tank		
(*n*)	Average g CH_4/kWh	Average % CH_{4total}	Plant no. (*n*)	Average g CH_4/kWh	Average % CH_{4total}	Plant no. (*n*)	Average g CH_4/kWh	Average % CH_{4total}
1 (2)	28.6×10^{-2}	0.160	2 (2)	9.29×10^{-3}	0.005	4 (2)	28.4×10^{-2}	0.158
2 (2)	7.14×10^{-3}	0.004	4 (1)	55.9×10^{-2}	0.311	9 (2)	2.95×10^{-2}	0.016
6 (2)	0.52×10^{-3}	2.9×10^{-4}	8 (2)	3.06×10^{-2}	0.017	9 (2)	2.95×10^{-2}	0.016
			9 (2)	5.62×10^{-2}	0.031	9 (2)	2.38×10^{-2}	0.013
						9 (2)	2.91×10^{-2}	0.016
						9_{total}	11.2×10^{-2}	0.062
						10 (2)	33.8×10^{-2}	0.189

n, number of measurements.

elevated. In case of the screw conveyer, it was suspected that the end of the feeding pipe was not under the liquid surface. The substrate storage tank of plant 4 is quite large (500 m^3) and contains swine manure. On the site of plant number 10, digestate was mixed with silage in a 20 m^3 open tank prior to feeding.

All feeding components emit on average 15.4×10^{-2} g CH_4/kWh which equals 8.6×10^{-2}% of the utilized methane (11 feeding units). Ammonia emissions account for 7.58×10^{-3} g NH_3/kWh (10 feeding units). Emissions of nitrous oxide have been detected, but they were also low. The mean of N_2O-emissions at the feeding units adds up to a negligible 2.5×10^{-4} g N_2O/kWh (nine feeding units).

Apparently the feeding system doesn't produce great amounts of greenhouse gas emissions. However, the highest emissions occur as to be expected in the case of using open tanks for substrate storage or mixing. In regard to the emissions these systems are to be considered as inferior to closed systems.

Digester

Since the main target of a biogas plant is gas utilization, a gas tight cover of the digester is a crucial part of the whole concept. Besides two cases of detected leakages (see Table 3), which are classified as malfunction, the emissions released from the digesters have been quite low. The leakages are a poorly maintained service opening (major leakage) and a badly manufactured lead through of a pipe.

Table 3 | Methane emissions from digesters with a foil cover

Plant no. (n)	Period 1	Period 2	Average	Average
	g CH_4/kWh			% CH_{4total}
1 (6)	0	0	0	0
2 (2)	1.2×10^{-2}	0.94×10^{-2}	1.07×10^{-2}	0.006
2 (2)	3.66×10^{-2}	0.86×10^{-2}	2.26×10^{-2}	0.013
2_{total}	4.86×10^{-2}	1.8×10^{-2}	3.33×10^{-2}	0.019
3 (2)	0	1.1×10^{-2}	0.55×10^{-2}	0.003
3 (2)	8.75×10^{-2}	0	4.38×10^{-2}	0.024
3_{total}	8.75×10^{-2}	1.1×10^{-2}	4.93×10^{-2}	0.028
4 (1)	0	nm	0	0
7 (1)	nm	0	0	0
8 (4)	0	0	0	0
9 (4)	nm	0	0	0
10 (2)	0	1.83×10^{-2}	0.91×10^{-2}	0.005

n, number of measurements.
nm, not measured.

After taking care of the leakages, no emissions have been detected in the second measurement period. Besides this, digester tanks with a concrete roof don't show detectable emissions at all.

Digesters with foil cover emissions have been reported occasionally. The greatest emissions released from a digester were 4.38×10^{-2} g CH_4/kWh which translates into 2.44×10^{-2}% of the utilized methane. At five plants no emissions at all were detected.

The connection of the foil and the digester were identified as emission sources. The air in the double layer foil cover systems showed occasionally high methane concentrations. The sources could not be identified due to limited access to the covers. The membrane material itself has been checked but no emissions have been found there.

The gas tight constructed digestate storage tanks were analysed like the digesters, since the construction features are very similar. No emissions were detected.

Emissions of ammonia were very low at 0.06×10^{-3} g NH_3/kWh (eight plants) and no emissions of nitrous oxide occur (eight plants).

Digestate storage tanks

The investigation of the digestate storage tanks (not gas tight covered) gives interesting, but not very consistent results (see Table 4). The values obtained in the different programs differ in some cases substantially. Since the emissions from digestate tanks are dependent on the temperature, wind, atmospheric pressure, filling level of the tank and the process parameter of the plant, it is quite difficult to determine an average emission by means of measurement, which can only represent a very short period of time. However the results show that some of the storage tanks can produce substantial emissions, whereas others show very little emissions. Further investigations will be needed to identify the driving factors for the emissions. On plant 5 – a garage style digestion system – the highest emissions (as average) occurred, the digested material having been stored outside the digester, waiting to be used as inoculum for the next filling or to be distributed on arable land. The emission factor of the digestate tank on plant 4 has a high uncertainty due to the missing measurement in period 2. All storage tanks except plants 4 and 5 emitted on average 6.25 g CH_4/kWh (six plants) which equals 3.5% of the utilized methane.

For comparison, some literature values have been evaluated. Woesch-Gallasch *et al.* (2007) found a gas production in the digestate storage which equaled 1.9% of the energy production of the plant. Berghold (2008) showed that the

Table 4 | Emissions from not gas tight digestate storage tanks and nitrogen characteristics of the digestate

Plant no.	Emissions of methane Period 1 (g CH_4/kWh)	Period 2 (g CH_4/kWh)	Average (g CH_4/kWh)	Average % CH_{4total}	Substrate characteristics NH_4-N g/kg	N^a_{total} g/kg	Emissions of ammonia Average g NH_3-N/kWh
3	2.36	5.52	3.94	2.20	1.6	4.3	8.03×10^{-1}
4	18.49	nm	18.49	10.30	2.4	5.4	1.49×10^{-1}
5	32.04	8.04	20.14	11.22	2.9	7.5	nm
7	10.11	26.47	18.29	10.19	3.3	5.8	3.68
8	nm	0.40	0.40	0.22	3.0	6.5	2.05×10^{-1}
9	0.77	1.52	1.14	0.64	2.5	5.5	1.55×10^{-2}
9	0.44	1.48	0.96	0.53	2.4	5.5	5.00×10^{-3}
9	0.48	1.60	1.04	0.58	2.5	5.6	2.32×10^{-3}
9	nm	1.50	1.50	0.83	2.5	5.6	1.12×10^{-2}
9_{total}	1.69	6.10	4.64	2.58			3.40×10^{-2}
10	3.87	4.17	4.02	2.24	4.3	8.4	2.36

nm, not measured.
[a]Digestate samples were taken at the outlet of the last (heated) step of the fermentation process.

gas tight cover of the digestate storage had increased the energy output of two facilities by 1.9 and 3.5% respectively. Balsari *et al.* (2010) evaluated a storage tank for separated liquids of digestate. The methane production from the tank added up to approximately 3% of the methane converted by the 1 MW co-generation unit installed at the facility.

Weiland *et al.* (2009) investigated the gas potential of digestates of 60 biogas plants at 20–22 °C and found an average 3.5% of the methane production for one stage processes and 1.5% for two stage processes. These values represent the maximum possible emissions at this temperature and are yet much lower than some results obtained here for real emissions.

Looking at the gas potential of the digestate (see Table 5) it is obvious that by comparing the plants with elevated methane emissions from the digestate storage (no. 4,5,7,10) plants 4 and in particular 5 show elevated gas potentials also in the digestate – but 7 and 10 do not. The order of magnitude of the emissions from plants 4 and 7 seems high in comparison to the gas potential of the digestate. The reasons for that could not be identified. One explanation could be particular weather conditions such as falling atmospheric pressure on the day of measurement.

Very often the retention time is named as one of the major factors influencing the remaining gas potential of the digestate and consequently potential emissions from the open storage. Weiland *et al.* (2009) and Reinhold & Gödeke (2011) concluded from their investigation of digestates that a significant reduction of the gas potential of the digestates can be achieved by retention times in the fermentation system greater than 100 days. However, due to a great variability of other process parameters, this estimation does not give a clear answer to the options within the process design which might lead to a similar reduction. Both investigations – as does this one too – displayed data in which low gas potentials in digestates can also be achieved under comparable low retention times.

Table 5 | Digestate characteristic gas potentials (methane) over 60 days from the investigated plants

Plant no.	Digester output[a] TS %	VS % wm	Gas potential at 39 (20) °C l_N CH_4/ kg wm
1	8.1	6.3	4.7 (2.6)
2	6.4	4.8	2.6 (0.7)
3	6.5	5.2	4.5 (1.3)
4	7.6	6.1	8.2 (3.6)
5	19.1	13.5	26.9
6	13.6	11.7	18.3 (6.2)
7	7.0	5.2	5.0 (2.0)
8	9.7	7.6	6.5 (2.7)
9	6.8	5.2	6.2 (3.1)
10	10.5	7.9	5.4 (1.3)

[a]Samples taken at the outlet of the last step of the fermentation process; wm = wet mass; l_N = litres (dry and standard conditions 273.15 K, 1013.25 mbar).

The temperature of the stored digestate has a great influence on the emissions as well. Laboratory tests within the studies of Weiland *et al.* (2009) and Reinhold & Gödeke (2011) showed that depending on the temperature of the digestate during storage, the emission potential can be significantly reduced. In this study, the average methane potentials obtained at 20 °C represent 39% of the methane potential obtained at 39 °C.

Reinhold & Gödeke (2011) found that at 25 °C the methane production is reduced to 40–50% of the value obtained at 37 °C and at 10 °C the methane production goes down to even 1%.

The temperature profile of digestate storage tanks can vary quite a bit. In Gioelli *et al.* (2011) the average ambient temperature was 16.6 °C, the digestate temperature in the storage tank was 29.2 °C. In our case, the investigations at the lagoon of plant number 7 showed that the average digestate temperature was 18.5 °C over a period of 38 days, with an average ambient temperature of 7.4 °C. However, these data do not allow a clear estimation of the emission to be expected. But it can be concluded that the digestate temperatures can be quite high as can the occurring emissions subsequently.

The highest ammonia emissions (plants 7 and 10) were caused by digestates with the highest ammonia concentration in the liquid phase. Both plants use chicken manure as substrates.

The results from the open digestate storage tanks clarify that the open tanks have the potential to represent a substantial emission source. The variability of the results and the contradictions in the gas potential analysis in some cases show that a precise determination is rather difficult due to the many factors influencing the emissions from open tanks. For further investigations the method needs to be evaluated and additional information needs to be included in the analysis. Additionally, longer measurement periods are necessary in order to determine the fluctuation of the emissions over time.

Gas utilization (co-generation units or gas-upgrading facilities)

Following the open digestate storage the gas utilization is the second major emission source. Due to incomplete combustion the engines emit on average from 0.40 to 3.28% of the utilized methane. The co-generation units emit on average 3.11 g CH_4/kWh (11 co-generation units), which equals 1.74% of the utilized methane.

In the literature, similar results can be found. Woesch-Gallsch *et al.* (2007) found methane concentrations of 1,100 mg/m³ methane from a co-generation unit which correlates to 1.79% CH_4 of the utilized methane. Aschmann *et al.* (2006) measured averaged 290 mg/m³ C_nH_m (0.5% methane slip) from gas engines and averaged 560 mg/m³ C_nH_m (0.9% methane slip) from pilot injection gas engine. Aschmann *et al.* (2009) reported 1–2% for gas engines and 2–3% for pilot injection engines. For a single occasion they also reported a high emission of 6.7 $g_{total-C}/kWh_{el,}$ which is close to the maximum value in Table 6. The literature values fit to the results obtained in this study. The slip cannot be avoided and only a secondary oxidation step in the exhaust gas can reduce these methane emissions. The two gas upgrading units displayed quite high emissions. For plant number 8 the post-treatment of the exhaust gas was supposed to be not functioning during the measurements.

The concentrations of nitrous oxide from the co-generation units of plant 7 showed during all four measurements very high values. This could be linked with the high ammonia concentration in the liquid which can lead to an increased ammonia concentration in the biogas. In case this is oxidized within the co-generation unit, an elevated nitrous oxide production could be the result.

Miscellaneous

The emissions occurring during the separation process were low. The stored solids also emitted only negligible amounts of methane. On two occasions major leakages were detected (Table 7). One of them emitted approximately 5% of the converted gas from a not properly closed service opening. The matter points to the fact that a frequent leakage detection might help to avoid losses of methane and even dangerous release of greater amounts of methane. In another case the leakage could be reduced by a factor of two by taking immediate measures, in the second period, no emissions occurred at this digester. The biofilter was used for the exhaust gases from the garage style system. The value is an average since the opening of the garages produced for a short period of time quite high emissions.

Biogas plants in comparison

Figure 2 displays the methane emissions according to the sources. It shows in summary the methane emission factors from three sources of typical biogas plants in Germany. The storage of digestate in open or not gas

Table 6 | Methane and nitrous oxide emissions from the gas utilization (combustion or upgrading)

	Co-generation unit						Gas upgrading facilities	
Plant-no.	Period 1 CH_4 g/kWh Period 1	Period 2 Period 2	Average Average	Average % CH_{4total}	Average g N_2O/kWh Average	Plant-no.	Average g CH_4/kWh	Average % CH_{4total}
1	6.47	5.30	5.89	3.28	0.008			
1	6.63	3.63	5.13	2.86	0.117[b]	Pressure swing adsorption		
$1_{P_{total}}$	4.79	3.92	4.36	2.43	0.006	8 (aeration)	0.42	0.23
$1_{P_{total}}$	1.73	0.95	1.34	0.74	0.030[b]	8 (exhaust)	9.58	5.34
2	1.89	1.81	1.85	1.03	0.015			
3[c]	3.40	nm	3.40	1.89	0.006			
4	1.77	6.63	4.20	2.34	0.014			
5	4.17	4.13	4.15	2.31	0.025	Pressurized water scrubbing		
6[a]	1.78	2.00	1.89	1.05	0.006	9 (aeration I)	0.45	0.25
6[a]	2.97	3.19	3.08	1.72	0.005	9 (aeration II)	0.03	0.017
$6^a_{P_{total}}$	0.89	1.00	0.95	0.53	0.003	9 (exhaust)	2.19	1.49
$6^a_{P_{total}}$	1.49	1.59	1.54	0.86	0.002			
7	2.23	1.76	2.00	1.11	0.208			
7	1.62	1.52	1.57	0.87	0.345			
$7_{P_{total}}$	1.11	0.88	1.00	0.56	0.104			
$7_{P_{total}}$	0.81	0.76	0.78	0.44	0.173			
10	nm	nm	nm	nm	nm			
10	1.17	1.02	1.09	0.61	0.022			
$10_{P_{total}}$	0.76	0.67	0.72	0.40	0.014			

P_{total} – the emissions of the co-generation unit are related to the total power of the whole facility and in the other cases to the power of the single unit.
[a]Pilot injection engine, diesel fraction not corrected.
[b]This high value result from a single measurement compared to the one in period 2.
[c]Measurement only in one of the periods.

Table 7 | Miscellaneous emission sources

Plant no.	Separation g CH_4/kWh	% CH_{4total}	Plant no.	Miscellaneous g CH_4/kWh	% CH_{4total}
1 storage of solids	0.56×10^{-3}	3.1×10^{-4}	3 concrete roof		
6 separator	22.9×10^{-2}[a]	0.128	Leakage	0.56[a]	0.314
6 storage of solids	0.38×10^{-3}	0.2×10^{-3}	After taking measures	0.27[a]	0.152
8 separator	2.00×10^{-2}	1.11×10^{-2}	Service opening	9.05[a]	5.04
8 storage of solids	0.23×10^{-3}	0.13×10^{-3}	5 biofilter	0.24	0.135

[a]Measurement only in period 1.

tight covered tanks generates the highest measured methane emissions. Plants equipped (e.g. 1 and 6) with gas tight storage tanks show much lower emissions, which shows the high effectiveness of a gas tight cover for emission reduction. The methane slips from the CHP or gas upgrading units are the second important source, creating high methane emissions. Those emissions could only be avoided by means of a post treatment of the exhaust gas. Miscellaneous sources generate negligible methane emissions in comparison to the digestate storage and the methane slip of the gas utilization.

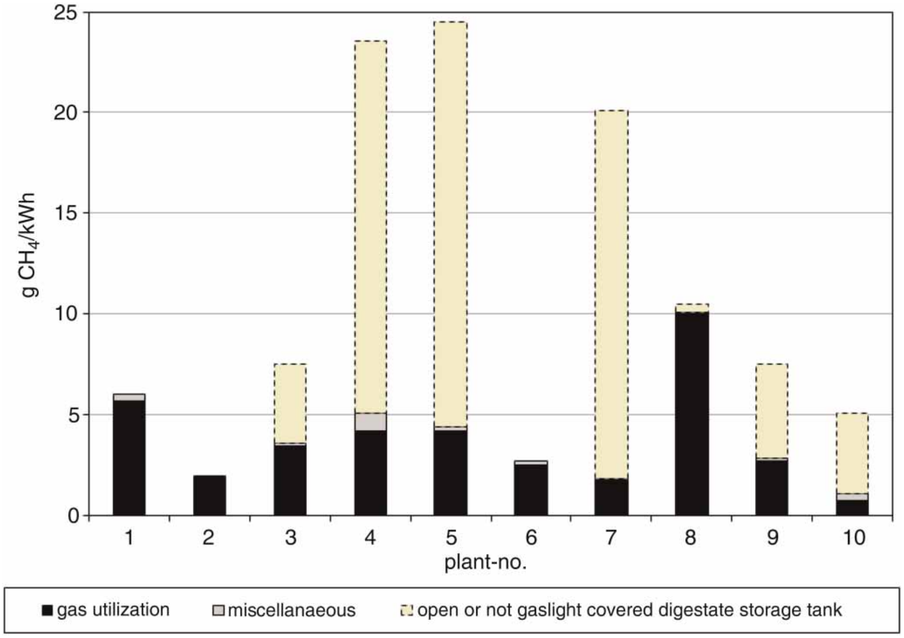

Figure 2 | Main emission sources for methane of the investigated biogas plants.

CONCLUSIONS

The measurement program focused on an evaluation of the constructively conditioned emissions and emissions due to leakages within the biogas production and utilization process. The program was set up in a way to obtain results for a variety of plants, consequently long term analyses could not be realized within this project. For every plant, two measurement programs were carried out.

Looking at the overall methane emissions of the plants, it appears to be clear that the main emission sources are the open digestate tanks and the gas utilization system (Figure 2). However, the results for the open digestate tanks need to be interpreted carefully, since they cannot represent an average emission over a longer period of time. Nevertheless, for the purpose of emission reduction a gas tight cover of any open digestate storage will have a great effect on the emissions. The reduction of the methane emissions of the co-generation unit can only be achieved by means of a secondary oxidation process. There are units available for this purpose, but the investment costs are quite high preventing a widespread application. Emphasizing on these two components any measures to reduce the emissions will show the most pronounced effect.

Nitrous oxide has been found only in the exhaust gas of the co-generation unit.

The results from some components display varying emissions (in particular open digestate tanks), but these variations could not be explained by the available data. A long term investigation at several plants combined with a full mass balance and weather data would help to identify the factors influencing the emissions in detail.

It has been shown that leakages can cause major emissions without recognition by the operators. Therefore a frequent check for leakage identification is recommended.

The ammonia emissions are relevant if the digestate is stored in open tanks and the ammonia content within the digestate is elevated.

The investigation focused on plants under full load conditions. Further investigation might also look into the emission situation in case of disturbances or partial load conditions.

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Total methane emission rates and losses from 23 biogas plants

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ABSTRACT

Methane losses from biogas plants are problematic, since they contribute to global warming and thus reduce the environmental benefits of biogas production. Total losses of methane from 23 biogas plants were measured by applying a tracer gas dispersion method to assess the magnitude of these emissions. The investigated biogas plants varied in terms of size, substrates used and biogas utilisation. Methane emission rates varied between 2.3 and 33.5 kg CH_4 h^{-1}, and losses expressed in percentages of production varied between 0.4 and 14.9%. The average emission rate was 10.4 kg CH_4 h^{-1}, and the average loss was 4.6%. Methane losses from the larger biogas plants were generally lower compared to those from the smaller facilities. In general, methane losses were higher from wastewater treatment biogas plants (7.5% in average) in comparison to agricultural biogas plants (2.4% in average). In essence, methane loss may constitute the largest negative environmental impact on the carbon footprint of biogas production.

1. Introduction

Biogas from anaerobic digestion, using various substrates such as manure, food waste, organic industrial waste and sludge from wastewater treatment, may result in several greenhouse gas (GHG) mitigation effects, including fossil fuel substitution, the possible balancing of energy sources in a supply system with a high proportion of wind and solar power and a reduction in methane (CH_4) emissions from manure management (Clemens et al., 2006; IPCC, 2011; Sommer et al., 2004). Fugitive CH_4 emissions from biogas plants, however, will reduce the environmental benefits of biogas production, mainly because of the relatively high global warming potential of CH_4, in that releasing just 1 kg of CH_4 into the atmosphere has the same effect with regards to global warming as the release of 28 kg of carbon dioxide (CO_2) integrated over a 100-year period (not including climate feedback) (Myhre et al., 2013). Data on the magnitude of these emissions are sparse, which in turn causes uncertainty with regards to the environmental assessment of biogas production concerning global warming (Meyer-Aurich et al., 2012; Møller et al., 2009). Recent studies suggest that the extent of CH_4 emissions expressed as a fraction of production lost to the atmosphere (also referred to as "CH_4 loss") may vary between facilities. Liebetrau et al. (2013), for instance, monitored CH_4 emissions from ten German biogas plants, using an on-site approach where individual leaks were identified and emission rates were subsequently measured. It was found that CH_4 emissions relative to the energy output of the biogas plants varied by approximately one order of magnitude between plants, and that open digestate storage tanks in many cases were the most significant emission source. Other sources of CH_4 emission from biogas plants may include unburnt CH_4 from gas engine exhausts, pressure relief valves, biogas upgrading units, ventilation from buildings, leaks in pipes, tanks, etc. (Kvist and Aryal, 2019; Angelidaki et al., 2018; Fredenslund et al., 2018; Liebetrau et al., 2013; Reinelt et al., 2017, 2016; Samuelsson et al., 2018).

An important step in understanding and subsequently reducing CH_4 emissions in the biogas sector is the reliable identification and quantification of single emission sources and the quantification of overall plant emissions. In general, two main approaches can be used for gas emission quantification: on-site and ground-based remote sensing approaches. The on-site approach measures emissions from various single sources at the plant, and it is the method most commonly used (Reinelt et al., 2016; Daniel-Gromke et al., 2015; Westerkamp et al., 2014; Liebetrau et al., 2013). Often a two-step procedure is followed where the first step includes a leakage search performed by using infrared cameras or handheld methane analysers. The second step includes quantification of each identified leakage or emission source often using the stationary or dynamic flux chamber technique. The ground-based remote sensing approach includes different methodologies and measures emissions from a good distance (for example one kilometre) away from the plant, thus providing plant-integrated emission numbers (Fredenslund et al., 2018; Delre et al., 2017; Groth et al., 2015; Yoshida et al., 2014; Hrad et al., 2014; Westerkamp et al., 2014; Flesch et al., 2011). Ground-based remote sensing techniques

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encounter inverse dispersion techniques using for instance open-path lasers and tracer gas dispersion methods. Recent measurement comparison studies have found that methods measuring the plant's total CH_4 emission often result in a higher emission rate in comparison to on-site measurements, where the total emission is obtained by summing up those measured from single sources (Fredenslund et al., 2018; Reinelt et al., 2017). The reason for the discrepancy between on-site and ground-based remote sensing approaches is most likely that single sources are overlooked and/or not identified, or they are technically not quantifiable when using a particular measurement technique (e.g. open tanks). For GHG emission reporting or environmental assessment, the plant's total emissions are important; however, if the purpose of measuring CH_4 emissions at a biogas plant is to identify mitigation options, and thereby provide options to improve the environmental benefits of biogas production, on-site methods are needed.

The objective of this study was to quantify CH_4 emission rates and losses from full-scale biogas plants. The study focused primarily on large, centralised, manure-based biogas plants, which produce the bulk of biogas in Denmark. Production capacity at this type of facility was in the expansion phase nationally at the time of this study. In addition, CH_4 emission rates and losses were measured at biogas plants located at wastewater treatment plants (WWTPs). Landfill gas extraction and utilisation sites were not included. The paper compiles the results taken from several biogas plants, in order to provide an estimate of CH_4 losses from biogas production and to assess the importance of minimising this issue. CH_4 emissions were measured using the tracer gas dispersion method, which measures plant-integrated emission rates. The environmental importance of fugitive CH_4 emissions from biogas plants was evaluated by performing CO_2 footprint calculations for a generic, manure-based agricultural biogas plant.

2. Methodology

2.1. Site descriptions

The biogas plants included in this study all utilise continuously stirred anaerobic digesters to produce biogas, and they all are commercially operated facilities. They varied in terms of feedstocks, size, rate of gas production, type of gas utilisation and other factors. Table 1 provides an overview of the main characteristics of the plants.

Thirteen of the biogas plants (plants 1–13) are categorised herein as "agricultural", which means that the feedstocks consist mainly of manure, energy crops and agricultural waste, though they can also receive other feedstocks such as slaughterhouse waste or food waste. Out of the 13 agricultural plants, nine receive manure as the main feedstock (>75% of dry matter input is manure), whereby organic waste (organic industrial waste and/or food waste) is used as a supplement to increase gas production. Two biogas plants (plants 8 and 12) rely on energy crops (grass, maize silage and forage rye), one (plant 5) receives mainly organic waste (~80% of dry matter input) but also receives manure and one plant (plant 13) mainly uses food waste. Plant 13 (and possibly also plant 5) could depending on definition be termed a waste treatment biogas plant as it mainly treated slaughterhouse waste, food industry waste and household food waste. However, as this plant was the only one of this type and also as the generated digestate is spread on farmland, the plant was included in the agricultural biogas plant category.

Five of the agricultural biogas plants (plants 1, 2, 5, 6 and 8) were recently constructed (constructed in 2013 or later), whereas the remaining agricultural plants generally were constructed in the 1980s or 1990s. For a number of reasons, the 2000s saw very low levels of investment in Danish biogas production (Raven and Gregersen, 2007), whereas increases and diversification in subsidies in recent years have led to a "second wave," with most new production capacity emanating from large facilities that upgrade and inject the biogas into the Danish natural gas distribution grid. At the smaller and older plants, it is more common that the biogas is utilised on-site in a combined heat and power (CHP) unit.

Biogas plants 14–23 are categorised herein as "wastewater treatment biogas plants". These plants utilise sludge from wastewater treatment to produce biogas, and they are all located on the grounds of the WWTP from where the sludge originates. They can thus be considered part of a larger plant, the primary function of which is to remove pollutants from wastewater before discharge to a recipient, with energy production as secondary function. The biogas plants categorised as "agricultural biogas plants" all rely on gas production for revenue. Although the wastewater treatment biogas plants receive revenue from their gas production, their primary function is to stabilise and reduce the volume of sludge, and thereby the costs of further sludge treatment. The WWTPs may thus arguably have less incentive to minimize loss of methane compared to the agricultural biogas plants.

The size of the plants varied in size in terms of treated feedstocks, from 30,000 to 600,000 tonnes (wet weight) per year for agricultural biogas plants, while the WWTPs treated between 60,000 and 805,000 PE, which corresponded to a load to the on-site biogas plants of between 3,000 and 112,000 tonnes (wet weight) per year.

The biogas plants differ with regards to gas utilisation (Table 1). At 12 plants, all or some of the produced biogas is utilised on site in a CHP unit. At plants 3, 4 and 7 some of the gas (~20–30%) is used on site in a CHP unit providing process heat for the biogas reactors. The generated electricity is sold to the grid. The remaining part of the gas is routed off site to a nearby power plant (where it is used in a CHP unit). At eight plants, all or some of the biogas is upgraded to biomethane, using technologies such as water scrubbers or chemical scrubbers. At these facilities, the gas is either compressed and transported off site or is injected into a natural gas distribution network. At four plants (5, 11, 14 and 21), all gas utilisation occurs off site. An example of this type is plant 5, where the biogas is led to a nearby power plant (and used in a CHP unit) to generate electricity to the grid and heat to a district heating network.

Open digestate storage units may be significant emitters of CH_4 from biogas plants (Samuelsson et al., 2018; Reinelt et al., 2017; Baldé et al., 2016; Liebetrau et al., 2013). Table 1 lists those facilities, which store digestate in open tanks on site. All biogas plants were equipped with gas storage units with capacities typically corresponding to ~1 to 2 days gas production.

In all, the 23 biogas plants included in this study represent a variety of continuously stirred reactor biogas plant types with regards to amounts of feedstock utilised, feedstock types, gas production rates and gas utilisation.

2.2. Tracer gas dispersion method

CH_4 emission rates from each biogas plant were quantified using a tracer gas dispersion method, whereby a gaseous tracer (here acetylene gas – C_2H_2) is continuously released at the biogas plant, and concentrations of CH_4 and C_2H_2 are then measured while traversing the CH_4/C_2H_2 plume at distances up to ~2 km away, using a vehicle-mounted, high-precision gas analyser. The method has been applied to quantify fugitive emissions from various facilities such as landfills, composting facilities, WWTPs and biogas plants (Andersen et al., 2010; Fredenslund et al., 2018; Mønster et al., 2014; Scheutz et al., 2011; Yoshida et al., 2014). An advantage of this method compared to on-site methods, where emission sources are quantified individually, is the measurement

Table 1
Overview of the main characteristics of the investigated biogas plants.

Agricultural biogas plants	Type of feedstock and annual total amount treated at the plant (in tonnes wet weight per year)	On site gas utilisation (CHP[1]/biogas upgrade)	Digestate storage (open/closed)
1*	Manure, maize silage, organic waste (600,000)	Biogas upgrade: chemical scrubber, gas grid injection	Closed
2*	Manure, slaughterhouse waste (240,000)	Biogas upgrade: water scrubber, gas grid injection	Closed
3	Manure, organic waste (300,000)	CHP (partly off site)	Closed
4	Manure, slaughterhouse waste, other organic waste (235,000)	CHP (partly off site)	Closed
5*	Industrial waste, manure (200,000)	None – routed for off-site use in a CHP	Closed
6*	Manure, maize silage (118,000)	Biogas upgrade: chemical scrubber, gas grid injection	Closed
7	Manure, slaughterhouse waste, other organic wastes (225,000)	CHP (partly off site)	Closed
8*	Maize silage, forage rye	CHP and biogas upgrade: chemical scrubber, gas grid injection	Closed
9	Manure, organic waste, maize silage (170,000)	CHP	Closed
10	Manure, organic waste (37,000)	CHP	Closed
11	Manure, maize and grass silage, glycerol (30,000)	None – routed for off-site use in a CHP	Closed
12	Grass and maize silage, manure	CHP	Open
13	Organic waste (slaughterhouse waste, industrial food waste and household food waste) (104,000)	Biogas upgrade: chemical and water scrubber, gas to vehicle fuel	Open
Wastewater treatment biogas plants	**Feedstock (amount given in person equivalent)**	**On site gas utilisation (CHP[1]/biogas upgrade)**	**Digestate storage (open/closed)**
14	Sludge from wastewater treatment (750,000 PE[2])	None – routed for off-site biogas upgrading	Closed
15	Sludge from wastewater treatment (265,000 PE)	CHP	Open
16	Sludge from wastewater treatment (150,000 PE)	CHP	Closed
17	Sludge from wastewater treatment (420,000 PE)	Biogas upgrade: chemical scrubber, gas grid injection	Closed
18	Sludge from wastewater treatment (95,000 PE)	CHP	Closed
19	Sludge from wastewater treatment (60,000 PE)	CHP	Open
20	Sludge from wastewater treatment (125,000 PE)	CHP	Closed
21	Sludge from wastewater treatment (805,000 PE), industrial food waste and sewage sludge from small WWTPs	None – routed for off-site biogas upgrading, gas to vehicle fuel	Open
22	Sludge from wastewater treatment (95,000 PE), food waste	Biogas upgrade: chemical scrubber, gas to vehicle fuel	Open
23	Sludge from wastewater treatment (120,000 PE)	Biogas upgrade: chemical scrubber, gas to vehicle fuel	Open

[1] CHP: Combined heat and power.
[2] PE: Person equivalent.
* Constructed in 2013 or later.

of the biogas plant's total CH_4 emission, with little risk of underestimating them due to undetected emission sources (Fredenslund et al., 2018).

The method and instrumentation are described in detail in Mønster et al. (2014) and Yoshida et al. (2014). The overall error of the method has been the subject of a recent validation study, and it was found very likely to be less than 20% (Fredenslund et al., 2019). The potential error of the tracer gas dispersion measurement technique was determined to 15% by establishment of an error budget including the analytical error, error in the tracer gas release rate, data processing, and error in tracer gas placement and source simulation. The error of a measurement is the combined error of the method and the variability of the quantification, which was found to be about 20% in a controlled release test and comparable to the error obtained by comparison of the measured emission rate and the known controlled release rate (Fredenslund et al., 2019).

Emission rates are calculated using Eq. (1):

$$E_{target} = Q_{tracer} \times \frac{\int_{plume\ start}^{plume\ end} C_{target} - C_{target,backgruound})dx}{\int_{plume\ start}^{plume\ end} C_{tracer} - C_{tracer,backgruound})dx} \times \frac{MW_{target}}{MW_{tracer}} \tag{1}$$

where E_{target} is the emission rate of CH_4 in kg h^{-1}; Q_{tracer} is the release rate of the acetylene tracer gas in kg h^{-1}; C_{target} and C_{tracer} are the measured downwind concentrations in parts per billion (ppb); $C_{target,\ background}$ and $C_{tracer,\ background}$ are the measured background concentrations in parts per billion (ppb) and MW_{target} and MW_{tracer} are the molar weights of the two gases.

The measurements were taken by driving through the downwind plumes several times (typically 10 to 20 traverses per measurement campaign). Each plume traverse resulted in one CH_4 emission measurement, calculated using Eq. (1). The CH_4 emission rate (in kg h^{-1}) was calculated as the average value of the individual plume traverses, and any uncertainty was estimated as the standard error of the mean of the measurements (Fredenslund et al., 2019).

CH_4 loss (%) was determined as the ratio of the measured CH_4 emission to the CH_4 production of the biogas plants, logged the day the measurement was performed.

2.3. Measurement campaigns

The measurements were performed July 2013 through June 2018. At six biogas plants, this happened on a single day, whereas for the remaining 17 plants, measurements were repeated up to a maximum of six days (Table 2).

All measurements were performed using the same analytical equipment and the method described in Section 2.2. The measurements were performed during normal operation of the biogas plants. No malfunctions were reported by the plants for the periods of measurement. As implementation of the method required certain adjustments in each case, some variability with regards to tracer gas release rates and number of release points exists.

Table 2
Overview of measurements performed.

Plant number	Days of measurement campaigns	Number of plume traverses	Tracer gas release rate (kg C_2H_2 h^{-1})
Agricultural biogas plants			
1	1	20	2.24
2	2	66	1.07
3	1	14	2.29
4	3	54	1.90
5	2	32	1.44
6	2	42	0.97
7	3	54	0.83
8	5	166	1.32
9	2	39	1.24
10	1	17	0.91
11	2	29	1.50
12	4	138	1.51
13	2	21	0.44
Wastewater treatment biogas plants			
14	6	82	0.57
15	1	21	0.92
16	4	63	0.51
17	2	37	1.68
18	2	40	0.93
19	4	89	0.48
20	1	16	0.90
21	1	16	–
22	3	81	0.91
23	3	82	0.78

The average tracer gas release rate varied between 0.11 and 2.29 kg C_2H_2 h^{-1}, and the number of tracer gas release points varied between one and three. The measurement distance varied from a few hundred metres up to more than 1 km, according to the availability of drivable roads downwind and the detectability of elevated concentrations of CH_4 and C_2H_2 in the plume – low emission rates and high wind speeds increase dilution, and so it may be necessary to traverse the plume closer to the source of emission.

2.4. Impact of methane emissions on the overall CO_2 footprint of biogas plants

The impact of CH_4 loss on the overall CO_2 footprint of biogas plants was evaluated by using a calculation model provided by the Danish Ministry of Environment for environmental impact assessments of biogas projects (Danish Nature Agency, 2014). The model considers the following factors in determining the overall CO_2 footprint of biogas plants:

Substitution of fossil fuels
Substitution of chemical fertiliser
Transportation of feedstock and digestate
Change in manure management compared to conventional storage and use of manure in agriculture (fewer GHG emissions from manure storage at farms when manure is digested before storage)
Energy use of the biogas plant
Direct GHG emissions from biogas production and utilisation

Emissions and savings were determined by considering five levels of direct CH_4 loss from an agricultural biogas plant: 1%, 2%, 5%, 10% and 20%. Losses of produced biogas contribute directly (CH_4 emitted into the atmosphere) and indirectly (less substitution of fossil fuel as a result of lost biogas production), so both losses were included in the model.

In this assessment, we considered a generic agricultural biogas plant receiving 50,000 tonnes yr^{-1} of cattle manure, 60,000 tonnes yr^{-1} pig manure and 5000 tonnes yr^{-1} organic waste, which in combination produced 2.2 million m^3 CH_4 yr^{-1}. This calculation example is similar to one described by The Danish Nature Agency (2014). Two biogas utilisation options were considered: CHP and biogas upgrade and injection into the natural gas grid.

Table 3 provides an overview of emission factors as well as energy use and CHP energy conversion efficiencies. Two emission factors regarding the use and production of electricity were considered in terms of CHP gas utilisation, namely average and marginal. The average emission factor corresponds to the average emissions associated with the provision of electricity in Denmark, whereas the marginal factor is derived from an estimate of which electricity sources are reduced when production from (for example) biogas plants is increased – also in Denmark. The provision of electricity, on average, consisted of 17% coal, 6% natural gas, 55% wind, hydro and solar, 18% waste incineration, biomass and biogas, 1% oil and 3% nuclear – as reported by the Danish national authority on electricity production (Energinet.dk, 2018), whereas the provision of marginal electricity consisted of 80% coal, 15% natural gas and 5% renewables from a recent study on CO_2 emissions caused by

Table 3
Overview of parameters used in carbon footprint calculations.

Parameter	Value	Reference
Emission factors		
Provision of electricity (average)[a]	0.053 kg CO_2-eq. MJ^{-1}	Energinet.dk (2018)
Provision of electricity (marginal)[b]	0.24 kg CO_2-eq. MJ^{-1}	Ea Energianalyse (2016)
Provision and consumption of natural gas	0.057 kg CO_2-eq. MJ^{-1}	Danish Energy Agency (2018)
Provision of heat (district heating, Danish average value)[c]	0.056 kg CO_2-eq. MJ^{-1}	Danish Nature Agency (2014)
Production of N fertiliser	7.0 kg CO_2-eq. kg N^{-1}	Danish Nature Agency (2014) and Wood and Cowie (2004)
Production of P fertiliser	0.5 kg CO_2-eq. kg P^{-1}	Danish Nature Agency (2014) and Wood and Cowie (2004)
Transportation of digestate, manure, etc.	0.09 kg CO_2-eq. $tonne^{-1}km^{-1}$	(Danish Nature Agency, 2014)
Emission of CH_4	28 kg CO_2-eq. kg CH_4^{-1}	Myhre et al. (2013)
Manure management, cattle	−15 kg CO_2-eq. tonne $manure^{-1}$	Danish Nature Agency (2014)
Manure management, pigs	−23 kg CO_2-eq. tonne $manure^{-1}$	Danish Nature Agency (2014)
Other factors		
Process heat	8.4% of energy output	Danish Energy Agency (2017a)
Electricity use, biogas plant	3.7% of energy output	Danish Energy Agency (2017a)
Electricity use, biogas upgrade and compression	5.3% of energy output	Danish Energy Agency (2017a)
Electrical efficiency, CHP unit	44%	Danish Energy Agency and Energinet.dk (2014)
Total efficiency, CHP unit	92%	Danish Energy Agency and Energinet.dk (2014)

[a] Provision of electricity, average: 17% coal, 6% natural gas, 55% wind, hydro and solar, 18% waste incineration, biomass and biogas, 1% oil and 3% nuclear.
[b] Provision of electricity, marginal: 80% coal, 15% natural gas and 5% renewables.
[c] Average value of Danish district heating networks utilising various energy sources (waste incineration, solar, surplus heat from coal and biomass electricity production and more).

increasing electricity demand in Denmark (Ea Energianalyse, 2016). The differences in energy mix in the provision of average electricity, and provision of marginal electricity cause a relatively large difference in emission factors at 0.053 kg CO_2-eq MJ^{-1} (average) and 0.24 kg CO_2-eq MJ^{-1} (marginal). In the scenario considering the biogas upgrade, for simplicity we only considered the average emission factor for electricity use. In all scenarios, the consumption of heat by the biogas plant was presumed to be in the form of natural gas. In both CHP scenarios, the same emission factor for heat substitution was used, namely an average value of district heating networks in Denmark.

Both feedstock and digestate were assumed to be transported 5 km to and from the biogas plant. The anaerobic digestion of organic waste and the land application of digestates, and thereby recycling of the contained nutrients, was assumed to result in the reduced use of chemical fertiliser at 10 tonnes N yr^{-1} and 5 tonnes P yr^{-1}. The nutrient content of the manure was not considered to contribute to the reduced use of chemical fertiliser, since these nutrients would be applied to agricultural land anyway as raw manure without digestion.

3. Results and discussion

3.1. Measured CH_4 emission rates

Table 4 lists the measured CH_4 emission rates and losses for the 23 biogas plants in this study. The table also lists the biogas production rate of each plant, which was reported by individual plant operator in each case in the form of average daily production at the time of measurement. In those cases where CH_4 emission rates were measured over several campaigns, the listed CH_4 emission rates, gas production rates and CH_4 losses are average values.

Overall, the average CH_4 emission rates varied between 2.3 and 33.5 kg CH_4 h^{-1}. CH_4 losses (CH_4 emission relative to CH_4 production) varied between 0.4 and 14.9%, with the average being 4.6%. These results are comparable to Liebetrau et al. (2013), who found CH_4 losses from single, dominant sources (CHP units and open digestate storage) equating to between 0.22 and 11.2% of the utilised gas at 10 biogas plants. They are also comparable to the results of a study of a Canadian biodigester, where losses under normal operating conditions corresponded to 3.1% of CH_4 production (Flesch et al., 2011).

In general, CH_4 losses were higher from wastewater treatment biogas plants (average 7.5%) than from agricultural plants (2.4%) (Table 4). At seven of the 23 biogas plants, the average measured CH_4 loss was higher than the overall average (4.6%) (Table 4). Of these seven plants, six were WWTPs. The agricultural biogas plant that emitted more than 4.6% (plant 11, Table 4) actually had the lowest level of biogas production (Table 4). Of the agricultural plants, the highest CH_4 loss was 8.4% (biogas plant 11). The reported loss was based on two measurement campaigns, which both showed high CH_4 emissions. There was no on-site gas utilisation and no open mixing tanks, digestate storage tanks or similar. A specific reason as to why the biogas plant had a higher loss than

Table 4
Overview of measured average CH_4 emission rates and losses.

Plant number	On-site sources included in the measured emission (CHP or biogas upgrade unit)	Average biogas production kg CH_4 h^{-1}	Average CH_4 emission rate kg CH_4 h^{-1}	Average CH_4 loss %	Estimated revenue loss[e] k€ y^{-1}	Off-site sources not included in the measured emission (CHP or biogas upgrade unit
1	Biogas upgrade unit	1469	6.5 ± 0.6	0.4 ± 0.04	27.4 ± 2.3	–
2[a]	Biogas upgrade unit	1083	19.1 ± 2.5	1.8 ± 0.23	80.9 ± 10.6	–
3	CHP[f]	888	23.2 ± 1.7	2.6 ± 0.19	98.5 ± 7.2	CHP
4	CHP[f]	858	6.4 ± 0.5	0.7 ± 0.06	27.0 ± 2.0	CHP
5	–	498	3.0 ± 0.3	0.6 ± 0.06	12.9 ± 1.4	CHP
6[a]	Biogas upgrade unit	411	10.7 ± 0.5	2.6 ± 0.12	45.2 ± 2.1	–
7[a]	CHP[f]	404	6.4 ± 0.2	1.6 ± 0.06	27.1 ± 1.0	CHP
8	CHP and biogas upgrade unit	400	2.3 ± 0.4	0.6 ± 0.10	9.9 ± 1.6	–
9	CHP	333	14.9 ± 0.9	4.5 ± 0.26	63.1 ± 3.6	–
10	CHP	234	6.1 ± 0.8	2.6 ± 0.35	26.1 ± 3.5	–
11	–	74	6.4 ± 0.4	8.6 ± 0.50	27.2 ± 1.6	CHP
12	CHP	127	2.6 ± 0.4	2.1 ± 0.35	11.0 ± 1.9	–
13[b]	Biogas upgrade unit	815	21.2 ± 3.3	2.6 ± 0.40	90.0 ± 13.8	–
Plant average CH_4 loss, **agricultural**: 2.4%						
Production weighted average CH_4 loss, **agricultural**: 1.7%						
14[c]	–	440	9.8 ± 0.7	2.2 ± 0.15	41.7 ± 2.8	Biogas upgrade unit
15[a]	CHP	162	13.5 ± 0.5	8.3 ± 0.33	57.3 ± 2.3	–
16[c]	CHP	100	2.6 ± 0.4	2.6 ± 0.39	11.1 ± 1.6	–
17	Biogas upgrade unit	96	12.3 ± 1.2	12.8 ± 1.29	52.0 ± 5.2	–
18	CHP	88	8.1 ± 0.5	9.1 ± 0.60	34.2 ± 2.3	–
19[c]	CHP	85	2.6 ± 0.1	3.0 ± 0.16	11.0 ± 0.6	–
20	CHP	262	10.0 ± 1.0	3.8 ± 0.38	42.5 ± 4.2	–
21[d]	–	525	33.5 ± 0.6	6.4 ± 0.12	142.3 ± 2.6	Biogas upgrade unit
22[c]	Biogas upgrade unit	83	10.0 ± 0.6	12.0 ± 0.78	42.3 ± 2.7	–
23[c]	Biogas upgrade unit	58	8.6 ± 0.4	14.9 ± 0.72	36.5 ± 1.7	–
Plant average CH_4 loss, **WWTP**: 7.5%						
Production weighted average CH_4 loss, **WWTP**: 5.8%						
All biogas plants						
Plant average CH_4 loss, all: 4.6%						
Production weighted average CH_4 loss, all: 2.5%						

[a] Results were partly (first measurement) reported in Fredenslund et al. (2018).
[b] Results were reported in Reinelt et al. (2017).
[c] Results were reported in Delre et al. (2017).
[d] Results were reported in Samuelsson et al. (2018).
[e] Considering an estimated revenue of 0.7 €/Nm^3 CH_4.
[f] About 20–30% of the gas is used in a CHP unit, while the remaining is transported off site.

the average agricultural biogas plant was thus not identified. In general, the CH_4 emission rate relative to production seemed to correlate with the size of the biogas plant (Fig. 1), in that units with the highest gas production emitted proportionally less CH_4 compared to plants with relatively low output. One reason for this finding may be that the larger facilities have more economical resources for maintenance, re-investment and employment of highly proficient plant operators. Another reason may be that the number of potential emission sources (number of process units, pipes, joints, valves, etc.) is not necessarily proportional to the rate of biogas production. There was also the tendency that the larger agricultural plants were built more recently and thus may better represent the most up-to-date technology. CH_4 emission from biogas plants is not regulated directly in Denmark, so no regulatory explanation for the difference in methane loss for small biogas plants compared to larger plants was found. CH_4 losses from plants built within approximately the last 5 years (plants 1, 2, 5, 6 and 8) were relatively low (0.4, 1.8, 0.6, 2.6 and 0.6%, respectively). Two of the 13 agricultural plants were solely energy plants, where the input was mainly crops grown specifically for energy production, and both had CH_4 losses lower than the average for all plants.

As mentioned previously, agricultural biogas plants rely mostly on revenue from energy production for their existence, whereas energy production is a secondary activity in the case of wastewater treatment biogas plants. The economic incentive to maximise energy production, and therefore minimise leaks, may therefore be stronger for agricultural biogas plants. Finally, it should be noted that in this study total CH_4 emissions from the plants were measured. Wastewater treatment plants are more complex in structure than agricultural biogas plants, as they also have a water treatment operation in addition to sludge management and biogas production. Therefore, CH_4 emission rates measured at wastewater treatment biogas plants could also encounter CH_4 emissions from the water treatment line and from the open storage of sludge, which is more common at WWTPs in comparison to agricultural plants. However, at WWTPs, the main CH_4-emitting source will be biogas activities, even though CH_4 emissions can also occur from the plant inlet and from aeration tanks. Samuelsson et al. (2018) quantified CH_4 emissions from various unit processes at a WWTP and found that overall, about 81% of the CH_4 emissions quantified on site were released from the sludge treatment line. Delre et al. (2017) came to a similar conclusion based on on-site screenings of atmospheric CH_4 concentrations, where the highest elevated intensities were seen in the vicinity of sludge treatment activities. Sludge (un-digested or digested) storage in open tanks or basins can be a potential source of CH_4, which is challenging to quantify due to the large open surface area. At some of the WWTPs, open digestate storage of sludge could explain (but only partly) the higher emission rates. As an example, the average CH_4 loss at WWTPs with open storage was 9.2% in comparison to plants without on-site open storage (6.1%).

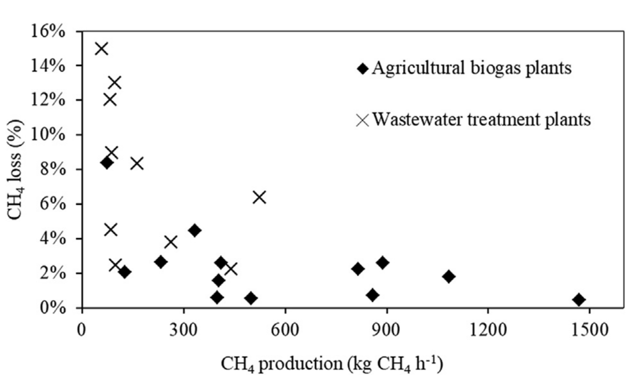

Fig. 1. Average CH_4 loss as a function of the average gas production at biogas plants.

Finally, it should be noted that at four of the plants (5, 11, 14 and 21) all gas utilisation occurs off site and at three of the plants part of the gas utilisation (~70–80%) occurs off site (plants 3, 4 and 7). For these plants, any CH_4 emission from the off site utilisation was therefore not included in the measured total CH_4 emission and thus the total CH_4 emission from the combined production and utilisation could be higher than the values reported in Table 4. At two of the plants (WWTPs 14 and 21) the generated biogas is routed off site for biogas upgrading. CH_4 emission factors from biogas upgrading units vary depending on technology applied. An average CH_4 slip of 0.81% was recently reported based on measurements of nine biogas upgrading units located in Denmark (Kvist and Aryal, 2019). The highest (1.97%) CH_4 slip was detected in the water scrubber methane upgrading technology, while the lowest (0.04%) CH_4 loss was detected in an amine based chemical scrubber (Kvist and Aryal, 2019). At five of the plants (agricultural plants 3, 4, 5, 7 and 11) the generated biogas is used (or partly used) in a CHP located off site. Liebetrau et al. (2013) found biogas co-generation units to emit on average 1.74% of the utilized methane with losses ranging from 0.40 to 3.28% (based on measurements at 10 biogas plants).

3.2. Contribution of methane emissions to the overall CO_2 footprint

Applying the methodology described in Section 2.4, the importance of various levels of CH_4 loss on the environmental performance of an agricultural biogas plant was assessed (Fig. 2). The impact in terms of GHG emissions (reported in CO_2-eqvivalents) of the different levels of CH_4 loss was assessed for three scenarios. In scenario A, biogas is upgraded to biomethane and substitutes for natural gas. In scenario B, biogas is utilised in a CHP unit, whereby electricity is supplied to the grid, and heat is used for district heating. In scenario B, the average emission factor regarding the production and consumption of electricity was used, as described in Section 2.4. In scenario C, biogas is also used in a CHP unit, but here the marginal emission factor for the production and consumption of electricity (Section 2.4) was used, meaning in this case that electricity production replaces more fossil fuel.

In all scenarios, CH_4 losses from biogas plants had a significant effect on the overall CO_2 footprint (Fig. 2). At 5% loss, CH_4 emissions make a greater contribution to the CO_2 footprint burden (positive CO_2 emission) compared to the other individual positively contributing emissions, namely energy consumption and the transportation of feedstock and digestate in all scenarios.

In scenarios A and B, a CH_4 loss of 20% caused the net GHG emissions to be positive, meaning that the biogas plant can be considered a net emitter of GHG, despite the substitution of fossil fuels, the reduction of GHG from manure storage and the substitution of chemical fertiliser. This is seen similarly in Table 5, where emission factors are listed for the three scenarios and five levels of CH_4 loss. These emission factors are the calculated net GHG emissions of the biogas production per one tonne of feedstock (wet weight) derived from the calculation example described in Section 2.4. The emission factors vary significantly in cases where CH_4 loss is relatively low (1–2%), to cases where the loss is relatively high (10–20%). The results also show that the emission factors in scenario B (CHP, average) vary highly in comparison to scenario C (CHP, marginal). The cause of this difference is the much lower electricity emission factor in the average mix of electricity sources (0.053 kg CO_2-eq. MJ^{-1}) compared to the marginal emission factor (0.24 kg CO_2-eq. MJ^{-1}) (Table 3).

The average CH_4 emission from the 13 agricultural biogas plants equated to 2.4% of the daily plant production (Table 4). Comparing this average CH_4 emission to implications on the total CO_2 foot-

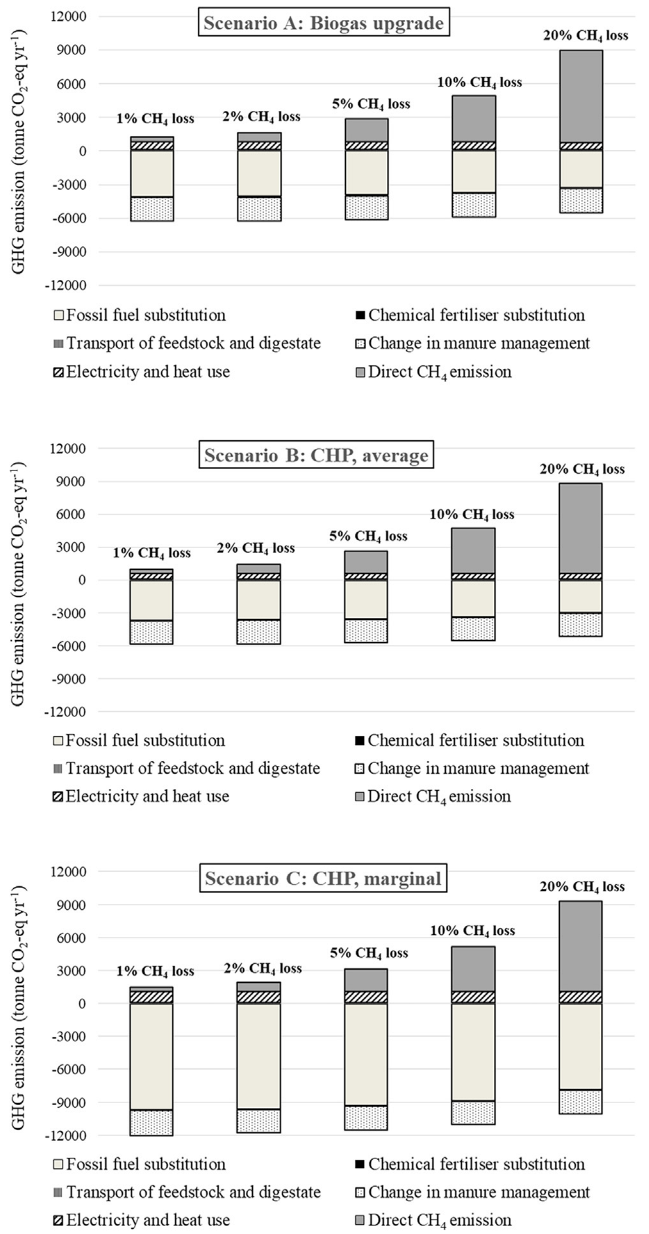

Fig. 2. Greenhouse gas (GHG) emissions calculated for an agricultural biogas plant, considering different biogas utilisation scenarios and five levels of methane (CH_4) loss. CHP: Combined heat and power.

print shown in Fig. 2, this relatively low loss indicates that the production of biogas is a net benefit with regards to GHG emissions. Since CH_4 emission rates compared to production varied greatly between biogas plants (Table 4), it is also likely that the CO_2 footprint of each individual plant will do so, too. Biogas plants, where the loss is particularly high (more than ~15%), may be net emitters of GHG, which underlines the importance of minimising CH_4 emissions from these facilities.

In this study, the carbon footprint of WWTPs was not determined, mainly because the primary purpose of a WWTP is not biogas generation but wastewater treatment, which implies that the services provided by the two types of plants are not comparable. Furthermore, not only CH_4 but also N_2O (another potent GHG) is emitted from WWTP,s primarily from the water treatment line, which needs to be included in footprint calculations. For an evaluation of the carbon footprint for biogas plants located at WWTPs, we instead refer to a recent study by Delre et al. (2019), which assessed carbon footprints for seven Scandinavian WWTPs, including some of the plants in this study. The study showed net carbon footprint values between 0.15 and 0.66 kg CO_2 eq. (Mg of input material)$^{-1}$, depending on the treatment facility. Direct CH_4 and N_2O emissions were the main contributors to the carbon footprint, accounting for between 44 and 71% of the total emission burden (Delre et al., 2019).

3.3. Fugitive methane emissions from Danish biogas production

Danish biogas producing facilities can be divided into four categories: agricultural (centralised and farm-scale) biogas plants (mainly treating manure), industrial biogas plants, wastewater treatment biogas plants (treating sewage sludge) and landfill gas. In total, 165 biogas-producing plants exist in the form of 82 agricultural (28 centralised and 54 farm-based), 51 wastewater treatment biogas, five industrial biogas and 27 landfill gas facilities (Danish Energy Agency, 2017b). The production of biogas has increased from 266 TJ (~5328 tonnes of CH_4) in 1990, to 7899 TJ (~157,985 tonnes of CH_4) in 2016 (Nielsen et al., 2018). In 2016, 86% of the generated biogas was based on manure/organic waste, 12% on sludge from wastewater treatment and only 2% came from landfills (Nielsen et al., 2018). Biogas production at the plants from which emissions were measured in this study represented about between 41% (agricultural) and 45% (WWTPs) of the annual Danish total (in 2016). National CH_4 emissions from Danish biogas production were estimated by applying the measured CH_4 emission factors to nationally generated CH_4 production, distinguishing between emission factors from agricultural and WWTP biogas plants, respectively. Two sets of CH_4 emission factors were used: a plant average and a weighted production average. The plant average was an average of CH_4 losses measured at the plants (sum of CH_4 losses divided by the number of plants), whereas the weighted production average was the sum of all CH_4 emission rates divided by the sum of all plants' CH_4 production rates (cf. Table 4). The plant average represents the biogas technology, whereas the weighted production average represents the combined biogas production in Denmark. Table 6 shows the estimated national CH_4 emissions (tonnes CH_4) for agricultural and WWTP biogas plants, and the total. The total estimated CH_4 emissions are between 3409 and 4683 tonnes, with emissions from agricultural biogas plants making up 6870%, while 30–32% originate from WWTPs (Table 6).

The 2006 IPCC Guidelines consider emissions from biogas plants (anaerobic digestion) as part of the waste sector. According to the IPCC Guidelines, emissions of CH_4 from biogas facilities, due to unintentional leakages during process disturbances or other unexpected events, will generally be between 0 and 10% of the

Table 5
Greenhouse gas emission factors (kg CO_2-eq tonne feedstock^{-1}) calculated for different biogas utilisation scenarios and five levels of CH_4 loss. A negative value implies an overall benefit to the environment, while a positive value implies an overall burden to the environment.

Scenario	1% loss	2% loss	5% loss	10% loss	20% loss
	(kg CO_2-eq tonne feedstock^{-1})				
Scenario A: Biogas upgrade	−44.6	−40.7	−29.0	−9.4	29.7
Scenario B: CHP, average	−42.1	−38.2	−26.5	−7.0	32.0
Scenario C: CHP, marginal	−89.7	−85.3	−72.0	−49.9	−5.7

Table 6
Estimated national CH_4 emissions from the anaerobic digestion of organic waste in agricultural biogas plants and biogas plants at wastewater treatment plants (WWTPs) in 2016 (excluding landfill gas). Numbers in brackets give the percentage out of total CH_4 emissions (excluding landfill gas).

Biogas plant type	Agricultural biogas plants	WWTP biogas plants	Total
CH_4 production, tonnes	135,867	18,958	154,825
CH_4 emission, tonnes (Plant average; $EF_{Agricultural}$ = 2.4% and EF_{WWTP} = 7.5%)	3261 (70%)	1422 (30%)	4683
CH_4 emission, tonnes (Production average; $EF_{Agricultural}$ = 1.7% and EF_{WWTP} = 5.8%)	2310 (68%)	1100 (32%)	3409

amount of CH_4 generated. In the absence of further information, a default value of 5% for the CH_4 emissions should be used (Eggleston et al., 2006).

CH_4 emissions from biogas production are reported in the Danish national greenhouse gas inventory as being a part of the waste sector's GHG emissions (Nielsen et al., 2018). CH_4 emissions were reported at 6635 tonnes in 2016, using an average adopted emission factor (EF) set equal to 4.2% for all types of biogas plants. This emission factor was based on a Danish project where CH_4 leakages were measured at nine biogas plants in Denmark, using on-site point measurement methods (Danish Energy Agency, 2015). Five of the plants were small, single-farm plants, while the other four were larger, centralised agricultural plants. The results were that the CH_4 losses varied from nil to 10% of production, resulting in a weighted average of 4.2%, which was adopted in the national inventory reporting for biogas production independently of the type of biogas plant. Our study shows a lower emission factor from agricultural plants, whereas the emission factor from biogas plants at WWTPs is higher than 4.2%. However, as the share of biogas generated at WWTPs is lower (12%) in comparison to agricultural plants (86%), the combined CH_4 emissions from these two types of facilities are almost comparable, resulting in a national emission of 3409 to 4683 tonnes of CH_4, which is close to the nationally reported figure.

The Danish Biogas Association is a trade organisation representing the Danish biogas sector, with members including plant owners, suppliers, agriculture and energy companies. Within the last few years, this organisation has initiated a voluntary measurement programme with the aim of keeping CH_4 loss at a minimum via a target of 1% loss for the sector. Our results indicate that some improvements are needed to reach this goal. However, the production weighed average loss was just 1.7% for the agricultural biogas plants, where most gas is produced and where production capacity is expanding, and thus the 1% target for the sector as a whole seems to be within reach. However, at plant level, emission rates are higher.

4. Conclusions

Methane losses were measured at 23 biogas plants and found to vary between 0.4 and 15.0% of the production total. Comparing those measured losses to an evaluation of the impact of methane loss on the overall carbon footprint of biogas production, it may be the case that methane loss is the largest positive contributor to greenhouse gas emissions for many biogas plants compared to other factors, such as energy use and the transportation of biomass.

Acknowledgements

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ATTACHMENT 50

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The False Promises of Biogas: Why Biogas Is an Environmental Justice Issue

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ABSTRACT

Years of community-driven research and participatory action have shed an important light on the copious negative health issues burdening communities adjacent to industrial agriculture. Rural communities in Wisconsin and Delaware have helped us in establishing an emerging source of pollution toward environmental justice communities—biogas. Biogas is being falsely marketed as a renewable energy solution to solve the problems of an already polluting industry, Concentrated Animal Feeding Operations (''CAFOs''). This greenwashing is problematic for many reasons and is in itself an environmental justice issue. The production of biomethane from manure-to-energy projects, such as manure digesters, is hazardous to local communities, locks farmers into more debt, and perpetuates the expansion of our current harmful agriculture practices, while increasing fossil fuel infrastructure by entrenching CAFOs with pipelines for the gas that is produced. In this article, we breakdown why biogas is not sustainable, how manure-to-energy projects perpetuate environmental injustices, examine current state policies on manure-to-energy projects, and how policy can be improved to protect frontline communities and farmers.

Keywords: CAFO, biogas, manure digesters, biomethane, environmental justice, factory farm

INTRODUCTION

There is a suite of social science and public health studies that have documented environmental injustices in rural areas stemming from industrialized agriculture and other extractive industries.[1] A pattern of negative pollution and public health consequences of industrial animal agriculture facilities threatening environmental justice communities in rural areas has been established.[2]

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[1]Kendall Thu and E. Paul Durrenberger. *Pigs, Profit, and Rural Communities.* (State Albany, NY: University of New York Press, 1998); John Gaventa. *Power and Powerlessness: Quiescence and Rebellion in an Appalachian Valley.* (Urbana, IL: University of Illinois Press, 1980); Pew Commission on Industrial Farm Animal Production. *Putting Meat on the Table: Industrial Farm Animal Production.* (A Project of the Pew Charitable Trusts and Johns Hopkins Bloomberg School of Public Health, 2008).

[2]Kelley J. Donham, Steven Wing, David Osterberg, Jan L. Flora, Carol Hodne, Kendall M. Thu, and Peter S. Thorne. ''Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations.'' *Environmental Health Perspectives* 15 (2007): 317–320; Sacoby M. Wilson, Frank Howell, Steve Wing, and Mark Sobsey. ''Environmental Injustice and the Mississippi Hog Industry.'' *Environmental Health Perspectives* 110 (suppl 2) (2002): 195–201; Steve Wing. ''Social Responsibility and Research Ethics in Community-Driven Studies of Industrialized Hog Production.'' *Environmental Health Perspectives* 110 (2002): 437–444; S. Wing and S. Wolf. ''Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents.'' *Environmental Health Perspectives* 108 (2000): 233–238; S. Wing, D. Cole, and G. Grant. ''Environmental Injustice in North Carolina's Hog Industry.'' *Environmental Health Perspectives* 108 (2000): 225–231.

Research has shown that improving sustainable healthy rural communities is found to be dependent on integrating socioeconomic development and environmental protection.[3] The concentration and industrialization of agriculture are associated with economic and community decline locally and regionally,[4] and one of the most significant social impacts of Concentrated Animal Feeding Operations (''CAFOs'') is found to be the disruption of quality of life for neighboring residents.[5] Furthermore, it has been established that CAFOs are disproportionally located in communities of color and, or low income communities—a form of environmental injustice that has negative impacts on community health.[6]

We identify a growing problem in this realm that is disguised as a solution to the waste problems caused by CAFOs—biogas or manure-to-energy projects. Although the agribusiness industry, government, and even public interest environmental organizations have touted this technology as beneficial to the environment, farmers, and rural communities, we find the opposite to be true. We bring to light government regulatory failures, failures in environmental justice initiatives, and the reality of environmental harms that are exacerbated by biogas systems.

Our research seeks to bridge the divide between social science and public health research with laws and policies that perpetuate the problem, as opposed to address it. We offer policy solutions to address some of the problems rural communities are experiencing as a result of the false promises of biogas technology.

BACKGROUND

Dr. Sacoby Wilson's groundbreaking research in ''Environmental Injustice and the Mississippi Hog Industry'' revealed that living near industrial hog operations is a major public health concern for disproportionately burdened communities.[7] This study and others also referenced herein indicate that emissions from swine confinement houses are associated with adverse respiratory problems and a decline in quality of life for communities in their proximity.[8] The high density of hogs grown in confinement houses produce vast amounts of waste,[9] and community members who live close to these operations may have adverse health effects such as irritation to their eyes, noses, and throats; decline in quality of life; and possible mental health disorders. There are also water quality problems associated with leakage from the manure lagoons, and runoff from the spray fields that can contaminate surface and groundwater.[10] Some of the environmental contaminants emitted into the atmosphere include ammonia, hydrogen sulfide, volatile organic compounds, particulates, and other pollutants.[11] Wilson's research reveals the disproportionate amount of CAFOs in Black communities and how these hazardous operations adversely impact the physical, mental, and economic health of rural communities.

Dr. Wilson continued to expand on this research in ''An Ecologic Framework to Study and Address Environmental Justice and Community Health Issues'' (2009).[12] In this study, Wilson discusses the history of environmental justice and expands the vocabulary needed to accurately describe the many layers of intersecting structural oppression. He states as follows:

> I introduce the terms ''environmental slavery'' and ''environmental servitude'' as interchangeable conceptualizations that capture the experience of disadvantaged and vulnerable communities who are differentially exposed to unhealthy environmental conditions and resource-poor settings. Vulnerable communities are used (directly or indirectly) to host social and environmental disamenities and externalities through planning, zoning, industrial siting, infrastructure and development inequities; while communities consisting of dominant racial and class populations benefit from the inequities, access to more amenities, and the ecological goods and services of host communities. There is an underdevelopment and/or destabilization in the growth, health, and quality of life of host communities overburdened by environmental and social externalities and spatially and socially bounded by limited access to environmental amenities. Moreover, the footprints (ecological, economic, and social) of dominant racial and class populations lead not only to the use of host communities as sinks, but also the use of individual community members as sinks for environmental and psychosocial stressors.[13]

In Paul Mohai and Robin Saha's ''Reassessing Racial and Socioeconomic Disparities in Environmental Justice

[3]Kelley J. Donham, Steven Wing, David Osterberg, Jan L. Flora, Carol Hodne, Kendall M. Thu, and Peter S. Thorne. ''Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations.'' *Environmental Health Perspectives* 115 (2007): 318.

[4]Ibid at 317.

[5]Ibid at 318.

[6]Ibid. See also S. Wing and S. Wolf. ''Intensive Livestock Operations, Health, and Quality of Life Among Eastern North Carolina Residents.'' *Environmental Health Perspectives* 108 (2000): 233–238; S. Wing, D. Cole, and G. Grant. ''Environmental Injustice in North Carolina's Hog Industry.'' *Environmental Health Perspectives* 108 (2000): 225–231.

[7]Sacoby M. Wilson, Frank Howell, Steve Wing, and Mark Sobsey. ''Environmental Injustice and the Mississippi Hog Industry.'' *Environmental Health Perspectives* 110 Supplement 2 (2002): 199.

[8]Ibid.

[9]The new trend of large-scale production involves a high density of hogs grown in confinement houses and producing vast amounts of waste. The hog waste is collected and stored through different systems, including below-floor slurry storage (deep pit), underground slurry storage, anaerobic lagoons, and oxidation pits. One of the most popular methods is the storage of the waste in anaerobic cesspools, commonly called ''lagoons,'' where it undergoes microbial digestion. The hog waste effluent is later sprayed onto fields. Ibid at 195.

[10]Ibid at 195–196.

[11]Ibid.

[12]Sacoby M. Wilson. ''An Ecologic Framework to Study and Address Environmental Justice and Community Health Issues.'' *Environmental Justice* 2 (2009): 16.

[13]Ibid.

Research'' (2006), the authors examine the significant racial and socioeconomic disparities associated with hazardous sites, and the variation within the disparities found.[14] Their study addresses the failures of current methods used to assess environmental disparities adequately in accounting for the proximity between the hazard under investigation and nearby residential populations.[15]

Christopher W. Tessum expands on these inequities in ''Inequity in Consumption of Goods and Services Adds to Racial–Ethnic Disparities in Air Pollution Exposure'' (2019). This seminal study found that Black and Hispanic communities on average bear a ''pollution burden'' of 56% and 63% excess exposure, respectively, relative to the exposure caused by their consumption.[16] $PM_{2.5}$ air pollution is disproportionately induced by the racial–ethnic majority and disproportionately inhaled by racial–ethnic minorities.[17]

Dr. Wilson lays out a holistic framework to address environmental justice and health issues by reiterating that we must take an ''ecological systems approach to community health, [which] incorporates spatial and temporal concepts on the social organization of our living environments, considers ecologic features of the built and social environments that influence health, and utilizes contextual expertise to address environmental justice and health issues at the community level.''[18]

Although social and public health scientists have identified rural areas as a geo-special dimension of environmental justice research,[19] rural environmental injustices have lacked proper attention by the environmental justice movement as a whole. Notably there is a void of adequate legal and policy solutions available to rural people.[20]

Correspondingly, research is revealing how government-driven agricultural policies legalize pollution and the differential treatment of rural people. The greenwashing of biogas as a solution to the environmental hazards associated with CAFOs is an example of this kind of legalized pollution.

METHODS

Our research is community driven in nature—in that questions about whether farmers should invest in manure-to-energy projects, and how and why governments are supporting them, were questions that needed immediate answers in communities already burdened by manure-to-energy projects. In collaboration with our community partners significant data was collected through publicly available government records, or records obtained from various governmental agencies through the Freedom of Information Act. We also researched government laws and regulations; scientific and other kinds of peer-reviewed journals, biogas industry trade magazines and other types of publications. In addition, we incorporate both participant observation and participatory action research conducted via community engagement with the Socially Responsible Agriculture Project (''SRAP''), and the rural communities in which SRAP works. We participated with our community partners in the engagement of public officials and regulatory entities to try to address problems or anticipated problems from CAFO biogas facilities. For example, we explored how California's cap-and-trade program generates carbon-offset credits for factory farms with biogas digesters in Wisconsin. From this background research,[21] we were able to better understand the motivation for the expansion of biogas infrastructure to factory farms. Through these methods we observed how harmful biogas projects were being perpetuated in part through government action, despite the sunlight being shown on their false promises.

FINDINGS

One of our major findings that became apparent throughout our engagement was the government's lack of understanding and lack of transparency regarding how industrial agriculture facilities, and their waste streams, directly harm rural communities that are fenceline to CAFOs. In this article, we shine a light on the budding issue of manure-to-energy projects because rural communities are alarmed by the transformation of CAFOs into combined factory farms and biogas facilities under the guise of ''green energy'' or ''compost projects.'' In this study, we explain how a lack of oversight, regulation, and transparency perpetuates the expansion of industrial agriculture in already burdened environmental justice communities. We argue that biogas is not a solution and

[14]P. Mohai and R. Saha. ''Reassessing Racial and Socioeconomic Disparities in Environmental Justice Research.'' *Demography* 43 (2006): 383–399.

[15]Ibid.

[16]Christopher W. Tessum, Joshua S. Apte, Andrew L. Goodkind, Nicholas Z. Muller, Kimberley A. Mullins, David A. Paolella, Stephen Polaskyf, Nathaniel P. Springer, Sumil K. Thakrar, Julian D. Marshall, and Jason D. Hill. ''Inequity in Consumption of Goods and Services Adds to Racial–Ethnic Disparities in Air Pollution Exposure.'' *PNAS* 116 (2019): 6001–6006.

[17]Ibid at 6003.

[18]Sacoby M. Wilson. ''An Ecologic Framework to Study and Address Environmental Justice and Community Health Issues.'' *Environmental Justice* 2 (2009): 18.

[19]David Pellow. ''Environmental Justice and Rural Studies: A Critical Conversation and Invitation to Collaboration.'' *Journal of Rural Studies* 47 (2016): 381–386; Loka Ashwood and Kate MacTavish. ''Introduction: Tyranny of the Majority and Rural Environmental Injustice.'' *Journal of Rural Studies* 47(A) (2016): 271–277.

[20]Lisa R. Pruitt. ''The Rural Lawscape: Space Tames Law Tames Space.'' In: I. Braverman, N. Blomley, D. Delaney, and A. Kedar (eds.) *The Expanding Spaces of Law: A Timely Legal Geography*. (Stanford University Press, 2013); Lisa R. Pruitt, Amanda L. Kool, Lauren Sudeall, Michele Statz, Danielle M. Conway, and Hannah Haksgaard. ''Legal Deserts: A Multi-State Perspective on Rural Access to Justice.'' *Harvard Law & Policy Review* (2018): 15–156; Ann M. Eisenberg. *Distributive Justice and Rural America*. 61 B.C. L. Rev. 189 (2020): 223. <https://lawdigitalcommons.bc.edu/bclr/vol61/iss1/5> (Last accessed on March 10, 2021).

[21]Socially Responsible Agriculture Project. ''Trading Pollution: Wisconsin Industrial Dairies with Documented Regulatory Compliance Problems Benefit from California Greenhouse Gas Cap-and-Trade Program.'' August 2020. <https://sraproject.org/2020/08/2016/> (Last accessed on March 10, 2021).

examine current policy while also suggesting policy recommendations that shift away from this polluting industry and instead invest back into communities.

Biogas is an environmental justice issue

Biogas is not sustainable. Although biogas comes from organic materials such as animal waste or food waste, it is hardly ''clean'' or ''green'' in the way most people understand those concepts.[22] Biogas is flammable, highly toxic, and potentially explosive.[23] Harmful compounds and air contaminants are introduced into the environment during biogas production and use through both combustion processes and diffusive emissions.[24] Burning manure-produced gas emits the same air contaminants as the combustion of fossil fuels. To make matters worse, the factory farms that produce the biomethane can emit harmful pollutants into the air and discharge nitrates into groundwater.[25]

Manure-to-energy projects, specifically manure digesters,[26] are sold as a solution to farmers to help them mitigate the costs of production by turning excess animal waste into energy through biogas. The installation of a manure digester on a factory farm is the first step for farmers in the process of turning their manure into a revenue stream, but it is also the first step in entrenching factory farms in more fossil fuel infrastructure, as producing and transporting biogas requires pipelines, fleets of trucks, and interconnection with the local power grid.[27] Furthermore, gas pipelines and other infrastructure leak tremendous volumes of methane that contribute to climate change, negating any alleged ''renewable natural gas'' savings.[28] Annual methane emissions have increased by about 50 million tonnes from the 2000–2006 average, mainly driven by agriculture and the natural gas industry,[29] and atmospheric concentrations of methane are now >2.5 times above preindustrial levels.[30] Although methane is only one component of total factory farm greenhouse gas emissions, these also include enteric methane, nitrous oxide (NOx) from fertilizer and manure application, and carbon dioxide from fuel combustion and input manufacture.[31]

Studies show that even if manure digesters were installed on every single dairy farm across the country and worked at optimal efficiency, this would still fall short of the industry's goal of reducing its total greenhouse gas emissions by 25%.[32] Similar to biogas, natural gas has been falsely marketed as a renewable and clean energy source, whereas in reality it destroys communities and has been proven to be a radioactive and hazardous energy source.[33] Biogas is the industry's next attempt at greenwashing another polluting fuel to save their industry. Fossil fuels, including natural gas fields and leaking pipelines, contributed 108 million tonnes of methane emissions in 2017, a rise of 17%.[34] Ultimately, biomethane is a false solution that perpetuates the expansion of big ag monopolies, the toxic, hazardous, and destructive practices of CAFOs and fossil fuel infrastructure.

Manure-to-energy projects perpetuate environmental injustices. Factory farms are inherently polluting entities that poison adjacent rural communities with toxic chemicals that eventually cause local public health disasters, economic hardship, and generational trauma.

[22]Jessica McKenzie. ''The Misbegotten Promise of Anaerobic Digesters.'' *The Counter*, 3 December 2019. <https://thecounter.org/misbegotten-promise-anaerobic-digesters-cafo/> (Last accessed on March 10, 2021).

[23]U.S. Department of Agriculture. ''Conservation Practice Overview: CPS Anaerobic Digester (Code 366).'' October 2017. <https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_026500.pdf> (Last accessed on March 10, 2021).

[24]Valerio Paolini, Francesco Petracchini, Marco Segreto, Laura Tomassetti, Nour Naja, and Angelo Cecinato. ''Environmental Impact of Biogas: A Short Review of Current Knowledge.'' *Journal of Environmental Science and Health Part A* 53 (2018): 899–906.

[25]Center for Food Safety. ''Stop the Dairy Digester Scam.'' 23 April 2019. <https://www.centerforfoodsafety.org/issues/305/food-and-climate/blog/5580/take-action-stop-the-dairy-digester-scam> (Last accessed on March 10, 2021).

[26]Manure digesters use anaerobic digestion to convert organic material into biogas, which can then be refined into biomethane and used to produce electricity. Three main substances come out of the process of manure digestion, methane gas, also known as biomethane, that can be used as an energy source; liquid manure that can be used for fertilizer; and solid manure that can be used for composting and animal bedding. (Scott Gordon. ''What Manure Digesters Can and Can't Do.'' *WisContext*, 30 November 2016. <https://www.wiscontext.org/what-manure-digesters-can-and-cant-do> (Last accessed on March 10, 2021).

[27]Daniel P. Duffy. ''The Costs and Benefits of Anaerobic Digesters.'' *MSW Management*, 4 June 2017. <https://www.mswmanagement.com/landfills/article/13030153/the-costs-and-benefits-of-anaerobic-digesters> (Last accessed on March 10, 2021) (to be referred to as ''*The Costs and Benefits of Anaerobic Digesters*'' throughout the rest of the document).

[28]Robert W. Howarth, *et al.* ''Methane and the Greenhouse-Gas Footprint of Natural Gas from Shale Formations.'' *Climatic Change* (April 2011): 679, 687, 688; Robert W. Howarth. ''A Bridge to Nowhere: Methane Emissions and the Greenhouse Gas Footprint of Natural Gas.'' *Energy Science & Engineering* 2 (2014): 1, 2; Robert B. Jackson, *et al.* ''Natural Gas Pipeline Leaks Across Washington, DC.'' *Environmental Science & Technology* 48 (2014): 2051; Lavoie. *Environmental Science & Technology* 52 (2017): 3373.

[29]Quirin Schiermeier. ''Global Methane Levels Soar to Record High.'' *Nature*, July 14, 2020. <https://www.nature.com/articles/d41586-020-02116-8?utm_source=Nature±Briefing&utm_campaign=8a93e2b69c-briefing-dy-20200715&utm_medium=email&utm_term=0_c9dfd39373-8a93e2b69c-44035969> (Last accessed on March 10, 2021).

[30]Ibid.

[31]Jude L. Capper, Roger A. Cady, and Dale E. Bauman. ''The Relationship Between Cow Production and Environmental Impact.'' 2011: 10. <https://wcds.ualberta.ca/wcds/wp-content/uploads/sites/57/wcds_archive/Archive/2011/Manuscripts/Capper.pdf> (Last accessed on March 10, 2021).

[32]Ibid.

[33]Justin Nobel. ''America's Radioactive Secret.'' *Rolling Stone*, 21 January 2020. <https://www.rollingstone.com/politics/politics-features/oil-gas-fracking-radioactive-investigation-937389/> (Last accessed on March 10, 2021).

[34]Quirin Schiermeier. ''Global Methane Levels Soar to Record High.'' *Nature*, 14 July 2020. <https://www.nature.com/articles/d41586-020-02116-8?utm_source=Nature±Briefing&utm_campaign=8a93e2b69c-briefing-dy-20200715&utm_medium=email&utm_term=0_c9dfd39373-8a93e2b69c-44035969> (Last accessed on March 10, 2021).

Several studies have shown that a disproportionate number of CAFOs are located in low-income and nonwhite areas and near low-income and nonwhite schools.[35] These facilities and the hazardous agents associated with them are generally unwanted in local communities and are often thrust upon those sectors with the lowest levels of political influence.[36]

CAFOs can house anywhere from hundreds to millions of animals—the quantity of urine and feces from even the smallest CAFO is equivalent to the urine and feces produced by 16,000 humans.[37] The waste produced at factory farms contains antibiotics, hormones, pathogens, heavy metals, and other animal drugs and chemicals that contaminate significant ground and surface water across the country. Noxious gases are also released through ventilation systems from the CAFO confinement houses, and environmental contaminants are also released through volatilization from the waste decomposing in lagoons, spray fields, and other waste collection sites.[38] Furthermore, studies show that manure management activities are the third major category of U.S. agricultural emissions, releasing NOx and methane in quantities that total 16% of total U.S. agricultural emissions.[39]

Manure-to-energy projects have a direct negative impact on frontline communities. In a recent study, ''the Composition and Toxicity of Biogas Produced from Different Feedstocks in California,'' scientists found that the concentrations of minor chemical and biological components in biogas have ''the potential to be toxic to human health and the environment, to form toxic substances during the combustion process, or to form toxic substances after photochemical aging in the atmosphere.''[40] Furthermore, The California Air Resources Board (CA-ARB) and the Office of Environmental Health Hazzard Assessment compiled a list of 12 trace components potentially present in biogas at levels significantly above traditional fossil natural gas, including carcinogens (arsenic, p-dichlorobenzene, ethylbenzene, n-nitroso-di-n-propylamine, and vinyl chloride) and noncarcinogens (antimony, copper, hydrogen sulfide, lead, methacrolein, mercaptans, and toluene).[41] Because the composition of biogas varies so greatly between feedstocks, and being that there are so few studies on the differences in trace contaminates, it is irresponsible to invest in anaerobic digestion until the public health consequences are determined.

Although manure digesters might have the potential to reduce methane emissions, emissions of other air pollutants, such as NOx may increase to unacceptable levels.[42] Breathing air with a high concentration of NO_x can cause breathing problems, headaches, chronically reduced lung function, eye irritation, loss of appetite, and corroded teeth.[43] Community environmental air quality assessments have shown concentrations of hydrogen sulfide and gaseous ammonia that exceed U.S. Environmental Protection Agency (''EPA'') and Agency for Toxic Substances and Disease Registry recommendations.[44] Studies have reported that neighbors of confinement facilities experienced increased levels of mood disorders, including anxiety, depression, and sleep disturbances attributable to exposures to malodorous compounds.[45] Research has also found that lower concentration and secretion of salivary immunoglobulin among swine CAFO neighbors during times of moderate to high odor compared with times of low or no odor, suggesting a stress-mediated physiological response to malodor.[46] Such stressors, coupled with inadequate health-promoting infrastructure (e.g., supermarkets, parks, open spaces, and medical facilities), reduce the community's ability to defend against the adverse health consequences of their differential burden and exposure.[47]

One recent study, ''Mortality and Health Outcomes in North Carolina Communities Located in Close Proximity to Hog Concentrated Animal Feeding Operations'' (2018), explains that residents living in proximity to hog CAFOs are chronically exposed to contaminants from land-applied wastes and their overland flows, leaking lagoons, and pit-buried carcasses, as well as airborne

[35]Kelley J. Donham, Steven Wing, David Osterberg, Jan L. Flora, Carol Hodne, Kendall M. Thu, and Peter S. Thorne. ''Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations.'' *Environmental Health Perspectives* 115 (2007): 318.

[36]Ibid.

[37]Sierra Club. ''Why Are CAFOs Bad.'' <https://www.sierraclub.org/michigan/why-are-cafos-bad> (Last accessed on March 10, 2021).

[38]Sacoby M. Wilson, Frank Howell, Steve Wing, and Mark Sobsey. ''Environmental Injustice and the Mississippi Hog Industry.'' *Environmental Health Perspectives* 110, Supplement 2 (2002): 195.

[39]Peter Lehner and Nathan A. Rosenberg. *Legal Pathways To Carbon-Neutral Agriculture* 47 Envtl. L. Rep. News & Analysis 10845, 10847. <https://earthjustice.org/sites/default/files/files/Legal-Pathways-Carbon-Neutral-Agriculture.pdf>. (Last accessed on March 10, 2021).

[40]Yin Li, Christopher P. Alaimo, Minji Kim, Norman Y. Kado, Joshua Peppers, Jian Xue, Chao Wan, Peter G. Green, Ruihong Zhang, Bryan M. Jenkins, Christoph F.A. Vogel, Stefan Wuertz, Thomas M. Young, and Michael J. Kleeman. ''Composition and Toxicity of Biogas Produced from Different Feedstocks in California.'' *Environmental Science & Technology* 53 (2019): 11569–11579.

[41]Ibid at 11569.

[42]Jude L. Capper, Roger A. Cady, and Dale E. Bauman. ''The Relationship Between Cow Production and Environmental Impact.'' 2011: 10. <https://wcds.ualberta.ca/wcds/wpcontent/uploads/sites/57/wcds_archive/Archive/2011/Manuscripts/Capper.pdf> (Last accessed on March 10, 2021).

[43]''NOx Gases in Diesel Car Fumes: Why Are They so Dangerous?'' *Phys.org*, September 2015. <https://phys.org/news/2015-09-nox-gases-diesel-carfumes.html#:~:text=NOx%20has%20direct%20and%20indirect,land%E2%80%94harming%20animals%20and%20plants> (Last accessed on March 10, 2021).

[44]Kelley J. Donham, Steven Wing, David Osterberg, Jan L. Flora, Carol Hodne, Kendall M. Thu, and Peter S. Thorne. ''Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations.'' *Environmental Health Perspectives* 115 (2007): 318.

[45]Ibid.

[46]Ibid.

[47]Sacoby M. Wilson, Herb Fraser-Rahim, Edith Williams, Hongmei Zhang, LaShanta Rice, Erik Svendsen, and Winston Abara. ''Assessment of the Distribution of Toxic Release Inventory Facilities in Metropolitan Charleston: An Environmental Justice Case Study.'' *American Journal of Public Health* 102 (2012): 1974.

emissions, resulting in higher risks of certain diseases.[48] In fact, a previous survey based on studies of residential communities reported significant health risks for residents, including higher risks of bacterial infections, higher frequencies of symptoms of respiratory and neurological disorders, and depression.[49] This exposed that people living in southeastern North Carolina communities located near hog CAFOs had poorer outcomes for a variety of health conditions in different age groups than the residents of North Carolina communities located in zip codes without hog CAFOs; they had higher mortality due to infections, anemia,[50] kidney disease, and perinatal conditions, and higher rates of hospital admissions and emergency department visits for low birth weight infants.[51] The authors conclude that people who live in these types of rural fenceline communities may simultaneously be affected by multiple risk factors, including low income and education, higher smoking prevalence, and lower access to medical care.

In addition, manure digesters installed at CAFOs require supplementary fossil fuel infrastructure, such as miles of pipelines stretching from the CAFO to the refinement facility. This funnels more pollution into and through already burdened local communities. Odor abatement, noise mitigation, truck queuing, effluent discharge, gas pipeline usage, and interconnection with the local power grid requires both physical hookups, and net metering agreements that can impact the health and wellness of neighboring families.[52]

Incentivizing farmers to install manure-to-energy projects instead of encouraging farmers to shift to sustainable farming practices solely profits developers, while locking communities into a cycle of sickness, loss, injury, and destruction. For example, from 2010 through 2019 fifteen Wisconsin dairy CAFOs received 1,317,236 carbon credits for their manure digesters,[53] despite the fact that in May of 2017 the CA-ARB's Environmental Justice Advisory Committee (EJAC)[54] made a priority recommendation to the Board to: ''[S]top investing in dirty energy [and] [e]liminate subsidies and financing for fossil fuels and in technologies such as corn-based biofuels, agricultural methane, biomass burning, waste-to-energy, or other unsustainable technologies that result in negative impacts on environmental justice communities.''[55] In addition, the EJAC advised against committing California ''Cap-and-Trade through the Clean Power Plan,'' since ''carbon trading cannot be verified…''[56] Furthermore, these funds are supposed to be aimed at ''Improving public health, quality of life and economic opportunity in California's most burdened communities at the same time they're reducing pollution that causes climate change.[57]'' Expanding any polluting industry in areas already burdened by factory farms perpetuates the systemic oppression of environmental justice communities.

Manure-to-energy projects are rarely beneficial for farmers. Manure-to-energy projects are expensive, temperamental, and require farmers to produce more waste to

[48]Julia Kravchenko, Sung Han Rhew, Igor Akushevich, Pankaj Agarwal, and H. Kim Lyerly. ''Mortality and Health Outcomes in North Carolina Communities Located in Close Proximity to Hog Concentrated Animal Feeding Operations.'' *NCMJ* 79 (2018): 277–288.

[49]Ibid.

[50]Studies have suggested that exposure to ammonia, hydrogen sulfide, methane, and particulate matters near the CAFOs, contamination of water and soil with zinc, exposure to the antibiotic chloramphenicol previously widely used to treat infections in hogs, and inappropriate human use of veterinary medications (certain NSAIDs or antibiotics) cause anemia. Ibid at 284.

[51]Ibid at 284.

[52]''The Costs and Benefits of Anaerobic Digesters.'' <https://www.mswmanagement.com/landfills/article/13030153/the-costs-and-benefits-of-anaerobic-digesters> (Last accessed on March 10, 2021).

[53]California Air Resources Board [hereinafter CA-ARB], Offset Credit Issuance Table. <https://ww3.arb.ca.gov/cc/capandtrade/offsets/issuance/issuance.htm>. (Last accessed on May 22, 2020).

[54]The California Environmental Protection Agency (Cal/EPA) Advisory Committee on Environmental Justice was formed in 2001 to help Cal/EPA incorporate environmental justice into all of its programs and policies. Three key recommendations called for Cal/EPA to recognize the significant burden of toxics and pollution on impacted communities. The advisory committee recommended that Cal/EPA: (1) Use a precautionary approach: A precautionary approach to decision making means that regulations should prevent harm when there is credible evidence that harm is occurring, or is likely to occur—even when complete scientific evidence or proof is not available—in drafting and enforcing regulations. (2) Prioritize pollution prevention over pollution control: All too often communities of color have been left feeling sorry by pollution control—sorry for their lost health and quality of life. (3) Evaluate the cumulative impacts of toxics in an impacted community when making regulatory decisions. This process requires that the health effects of all sources of pollution be taken into consideration when determining the impact of pollution in individuals, communities, and the environment. The landmark environmental justice policies were adopted by Cal/EPA. <https://www.environmentalhealth.org/index.php/en/where-we-work/state-of-california/california-environmental-justice> (Last accessed on March 10, 2021).

[55]CA-ARB, 2017 Scoping Plan, Appendix A, AB 32 Environmental Justice Advisory Committee Recommendations. November 2017: 14. <https://ww3.arb.ca.gov/cc/scopingplan/2030sp_appa_ejac_final.pdf>. (Last accessed on May 22, 2020).

[56]CA-ARB, 2017 Scoping Plan, Appendix A, AB 32 Environmental Justice Advisory Committee Recommendations. November 2017: 6. <https://ww3.arb.ca.gov/cc/scopingplan/2030sp_appa_ejac_final.pdf>. (Last accessed on May 22, 2020).

[57]California Environmental Protection Agency. ''California Climate Investments to Benefit Disadvantaged Communities.'' <https://calepa.ca.gov/envjustice/ghginvest/#:~:text=Known%20as%20California%20Climate%20Investments,pollution%20that%20causes%20climate%20change>. (Last accessed on January 21, 2021).

meet the needs of the digester.[58] The costs to build and run a manure digester are rarely recovered, especially when taking into consideration both the construction and operating costs. One study revealed that the economic concentration of agricultural operations tend to remove a higher percentage of money from rural communities than when the industry is dominated by smaller farm operations, which tend to circulate money within the community.[59]

Without outside funding from a designee, operator, or developer, it simply does not make financial sense for most farms to build or operate a digester.[60] The capital costs for a digester include lift station pumps, mixing tanks, the digester tank itself, piping for gas and hot water, gas pumps, flow meters, safety features, generators, electrical wiring and controls as well as power transmission lines, design engineering, and on-site buildings for generators, maintenance, and operations.[61] In addition to direct financial considerations, there is considerable overhead generated by legal and management issues, such as insurance premiums, building permits, design and consulting fees, licensing and zoning, sales agreements with utilities to buy back electricity, and more.[62] With capital costs often exceeding $1 million, anaerobic or manure digesters are beyond the price range of most farmers in the United States.[63] The payback period (capital costs divided by annual net benefits) of this capital investment can be between 5 and 6 years.[64]

Grants and cost-share agreements from states, federal programs, and utility companies help fund the cost of manure digesters.[65] For example, in just under a 4-year period beginning in 2010, 12 Wisconsin dairy CAFOs received >13 million dollars in grants from the U.S. Department of Agriculture's Rural Energy for America Program.[66] Additional financial incentives have also been provided to these projects through the U.S. Department of Treasury's administrative of Section 1603 of the American Recovery and Reinvestment Tax Act of 2009,[67] among other programs.[68] California alone has funded >100 digester projects, spending nearly $200 million of its ambitious California Climate Investments dollars on digesters instead of using the money to help fund farmers through its Smart Agriculture Programs.[69] According to the California Climate Investments Annual Report (2020), $69.1 million dollars has been assigned for future dairy digester development and research.[70] Tax payer dollars should be used to fund sustainable farming practices that go directly to farmers instead of to major developers. Unfortunately, these digester projects are also helping to fund gas infrastructure and development, rather than sustainable farming practices, which would create jobs,

[58]The size of a conventional digester is equal to 15–20 times the daily waste volume produced, or more if the waste is diluted before digestion (Don D. Jones, John C. Nye, and Alvin C. Dale. ''Methane Generation from Livestock Waste.'' *Energy Management in Agriculture*. Department of Agricultural Engineering, Purdue University. <https://www.extension.purdue.edu/extmedia/AE/AE-105.html>. (Last accessed on March 10, 2021). The volume of waste that must be disposed of increases accordingly if dilution water is used. The EPA's ''minimum'' requirements reveal what large investment manure digesters are and how they do not reduce the waste on CAFOs, but instead incentivize the farmer to produce even more to *potentially* be successful in making any profit or breaking even from the digester. The U.S. EPA states that for farms to be potentially successful with anaerobic digestion, a minimum of 500 head of cattle, 2000 hogs with anaerobic lagoons or liquid slurry manure management systems, or 5000 hogs with deep pit manure management systems are suggested. (U.S. Environmental Protection Agency. ''Is Anaerobic Digestion Right for Your Farm?'' <https://www.epa.gov/agstar/anaerobic-digestion-right-your-farm>). This significantly limits their use, as more than 90% of dairy farms in the United States have fewer than 500 cows, accounting for 40% of all dairy cows in the country (Peter Lehner and Nathan A. Rosenberg. *Legal Pathways to Carbon-Neutral Agriculture*, 47 Envtl. L. Rep. News & Analysis 10845: 10865. <https://earthjustice.org/sites/default/files/files/Legal-Pathways-Carbon-Neutral-Agriculture.pdf>. (Last accessed on March 10, 2021).

[59]Kelley J. Donham, Steven Wing, David Osterberg, Jan L. Flora, Carol Hodne, Kendall M. Thu, and Peter S. Thorne. ''Community Health and Socioeconomic Issues Surrounding Concentrated Animal Feeding Operations.'' *Environmental Health Perspectives* 115 (2007): 317.

[60]''The Misbegotten Promise of Anaerobic Digesters.'' <https://thecounter.org/misbegotten-promise-anaerobic-digesters-cafo/>. (Last accessed on March 10, 2021).

[61]''The Costs and Benefits of Anaerobic Digesters.'' <https://www.mswmanagement.com/landfills/article/13030153/the-costs-and-benefits-of-anaerobic-digesters>. (Last accessed on March 10, 2021).

[62]Ibid.

[63]''Legal Pathways To Carbon-Neutral Agriculture.'' 10865. <https://earthjustice.org/sites/default/files/files/Legal-Pathways-Carbon-Neutral-Agriculture.pdf>. (Last accessed on March 10, 2021).

[64]''The Costs and Benefits of Anaerobic Digesters.'' <https://www.mswmanagement.com/landfills/article/13030153/the-costs-and-benefits-of-anaerobic-digesters>. (Last accessed on March 10, 2021).

[65]A reference document by the U.S. EPA and Aster regarding digester funding showed case studies in which operators paid between 0% and 30% of the capital cost of the manure digester, with the remaining costs subsidized through grants and cost-share agreements from states, federal programs, and utility companies. See Aster, U.S. EPA. *Funding On- Farm Anaerobic Digestion* (September 2012).

[66]USDA, Rural Development, Rural Business-Cooperative Service, Wisconsin Recipients Renewable Energy Systems/Energy Efficiency Loan and Grant Program, see ''List of REAP Recipients in Wisconsin.'' <https://www.rd.usda.gov/programs-services/rural-energy-america- program-renewable-energy-systems-energy-efficiency/wi>/. (Last accessed on May 22, 2020); see also List of REAP Recipients in Wisconsin. <https://www.rd.usda.gov/files/WI_REAP_Awards.pdf>. (Last accessed on May 22, 2020).

[67]U.S. Department of the Treasury, American Recovery and Reinvestment Act of 2009 1603 Program (providing payments for specified energy property in lieu of tax credits). <https://home.treasury.gov/policy-issues/financial-markets-financial-institutions-and-fiscal-service/1603- program-payments-for> (see List of Awards). (Last accessed on May 22, 2020).

[68]See Good Jobs First, Tracking Subsidies, Promoting Accountability in Economic Development, Subsidy Tracker. <https://www.goodjobsfirst.org/subsidy-tracker>. (Last accessed on May 22, 2020).

[69]<https://civileats.com/2020/04/24/are-dairy-digesters-the-renewable-energy-answer-or-a-false-solution- to-climate-change/>. See also State of California, California Climate Investments website. <www.caclimateinvestments.ca.gov>. (Last accessed on July 31, 2020).

[70]California Climate Investments 2020 Annual Report. <https://ww2.arb.ca.gov/sites/default/files/classic/cc/capandtrade/auctionproceeds/2020_cci_annual_report.pdf>. (Last accessed on March 10, 2021).

facilitate carbon sequestering, and empower and rehabilitate rural and frontline communities.

The gas industry has an inherent incentive to convince farmers to invest in manure digesters. From their perspective, it is easier to transform an existing factory to produce and process biogas than to build a new one. Developers and energy executives have been selling the production of biogas as a solution to problems already existing on factory farms (too much manure, open lagoons, etc.), falsely motivating farmers to invest in ''composting[71]'' infrastructure that in actuality is much more complex and extensive, and is built to produce, process, and transport biogas, allowing the gas industry to profit from the dividends of the farmers' practically unpaid labor.

Unfortunately, some states are buying the industry's sales pitch about renewable natural gas and are allowing factory farms to receive carbon offset credits or compliance credits for installing manure-to-energy projects on farms. Carbon cap-and-trade programs allow industrial polluters to pay other pollution sources for their claimed pollution reductions.[72] For example, manure-to-energy projects that meet California's carbon offset protocols can be traded through its cap-and-trade program by approved national registries or through private contractors, or designees, who are registered with the state.[73] The polluting entity can then use the credits obtained from the trade to meet states' air pollution control standards. Negotiated agreements for the exchange of carbon credits are generally done in a private market, so it is difficult to know exactly how much money or other benefits manure digester projects receive.[74] Also, many factory farms that are receiving these credits are not in compliance with environmental and health regulations.[75] Critics such as the Institute for Agriculture and Trade Policy and the National Family Farm Coalition have argued that carbon markets are inherently inequitable, lock out most farmers, and could lead to more pollution, particularly in disadvantaged communities.[76] In short, the public is footing the bill for manure digesters,[77] whereas the CAFO industry profits by offsetting operation costs, selling the energy to utilities, and selling their claimed air pollution reductions to other polluters. Taxpayer dollars should be used to fund sustainable farming practices that go directly to farmers and environmental justice communities instead of to the pockets of major developers.[78]

[71]Glenn Rolfe. ''Environmental Groups: Seaford Poultry Digester Project Needs Public Input.'' *Delaware State News*, December 17, 2020. <https://delawarestatenews.net/business/environmental-groups-seaford-poultry-digester-project-needs-public-input/>. (Last accessed on March 10, 2021).

[72]''California Cap-and-Trade Program Summary.'' *Socially Responsible Agriculture Project*. August 2020. <https://sraproject.org/2020/09/california-cap-and-trade-program-summary/>. (Last accessed on March 10, 2021).

[73]Ibid.

[74]Ibid, see also SRAP. ''Trading Pollution: Wisconsin Industrial Dairies with Documented Regulatory Compliance Problems Benefit from California Greenhouse Gas Cap-and-Trade Program.'' Press Release, August 2020. <https://sraproject.org/2020/08/2016/>. (Last accessed on January 31, 2021).

[75]SRAP. ''Trading Pollution: Wisconsin Industrial Dairies with Documented Regulatory Compliance Problems Benefit from California Greenhouse Gas Cap-and-Trade Program.'' Press Release, August 2020. <https://sraproject.org/2020/08/2016/>. (Last accessed on January 31, 2021). Even if compliance issues at these CAFOs result in an invalidation of credits, the CA-ARB's regulations only call for invalidations for the number of days a facility is out of compliance. Seventy-five percent of the examples of invalidation on CA-ARB's website are of dairy operations (California Air Resources Board. ''California Air Resources Board Offset Credit Regulatory Conformance and Invalidation Guidance.'' February 2015. <https://ww2.arb.ca.gov/sites/default/files/classic//cc/capandtrade/offsets/arboc_guide_regul_conform_invalidation.pdf>; California Air Resources Board. ''Offset Credit Invalidation.'' <https://ww2.arb.ca.gov/our-work/programs/compliance-offset-program/offset-credit-invalidation>).

[76]Gosia Wozniacka. ''Are Carbon Markets for Farmers Worth the Hype?'' *Civil Eats*, 24 September 2020. <https://civileats.com/2020/09/24/are-carbon-markets-for-farmers-worth-the-hype/>. (Last accessed on March 10, 2021).

[77]Gosia Wozniacka. ''Are Dairy Digesters the Renewable Energy Answer or a 'False Solution' to Climate Change?'' *Civil Eats*, 24 April 2020. <https://civileats.com/2020/04/24/are-dairy-digesters-the-renewable-energy-answer-or-a-false-solution-to-climate-change/>. (Last accessed on March 10, 2021).

[78]Manure to energy projects that meet California's carbon offset protocols can be traded by approved registries or through private contractors, or designees, that are registered in the state's Compliance Instrument Tracking System Service before these manure-to-energy projects are submitted. The California-based polluting entity can then use the credits obtained from the trade to meet the state's air pollution control standards. Negotiated agreements for the exchange of carbon credits are generally done in a private market. Thus, it is difficult to know exactly how much money or other benefits manure digester projects receive. However, the private company 3Degrees Group explains that digester projects are very expensive and carbon credits provide a very important stream of revenue to help make the projects economically viable <https://3degreesinc.com/resources/sunny-knoll-farm-digester/>. (Last accessed on March 10, 2021).What's important to understand, is that this program only benefits those that are able to comply with the rigorous guidelines of the California Air Resource Board's (''ARB'') Offset Protocols. Those that abide by these standards can generate offset credits that are facilitated by Offset Project Registries (''Registries'') <https://ww2.arb.ca.gov/our-work/programs/compliance-offset-program/offset-project-registries>. These Registries help facilitate the listing, reporting, and verification of offset projects developed, and issue Registry offset credits (''ROCs''). Id. But these ROCs cannot be used for compliance with the Cap-and-Trade Program, they need to be converted to ARB offset credits to be eligible for use in the Cap-and-Trade Program. After the issuance of ROCs, the ARB determines whether ARB offset credits should be issued for each offset project. If a project wants to deliver voluntary offsets, they may monitor, report, and verify greenhouse gas emission reductions under the Livestock Offset Protocol, but elect not to transition the resulting ROCs issued by the Registries to ARB offset credits, and are free to deliver the ROCs on the voluntary market. <https://ww2.arb.ca.gov/sites/default/files/classic/cc/capandtrade/protocols/livestock/livestock.2014.faq.pdf>. (Last accessed on March 10, 2021): 5. This complex system of compliance to even be eligible to receive carbon credits or participate in the carbon market is just another example of the powerful agriculture industry consolidating and dominating against those that are not part of the system. Funding and developing manure-to-energy projects does not benefit your average farmer because to actually profit from the project, you need to spend copious amounts of time and money toward being recognized by the ARB as an offset project, and even so, might not be granted the credits by the Registries to participate in the market. Those that opt in to allowing ARB recognized developers to front and fund the projects do not benefit from the cap-and-trade revenue and are stuck with operating costs.

Current national and state policies perpetuate environmental injustices and fail to support rural communities and sustainable farming practices

As of now, our national agriculture policies do not protect or incentivize sustainable farming practices that are necessary to build back national soil health and sequester carbon. Instead of subsidizing false solutions such as manure-to-energy projects, we must organize to reform our national and local-level farm bills and other government programs to invest and insure renewable, just, and equitable farming practices.

Some states are already taking the initiative to fight against the expansion of biogas. New York State just recently passed one of the country's most progressive climate-forward policies in its Climate Leadership and Community Protection Act (''CLCPA'').[79] The CLCPA ''prohibits waste-to-energy projects'' and ''biofuels used for energy or transportation purposes.[80]'' In essence, the state has banned the future development of manure-to-energy projects from its renewable energy platform. This kind of innovative lawmaking protects and prioritizes environmental justice communities, ensuring disadvantaged communities are not disproportionally burdened with more polluting infrastructure.

As we explained earlier, unlike New York, other states are propping up manure-to-energy projects as a gateway to regulatory ''compliance'' and profit through carbon cap-and-trade markets. For example, SRAP found Wisconsin CAFOs, which already struggle with pollution and other compliance issues, participate in and, therefore, benefit from the CA-ARB carbon trading program.[81] Regardless of the claimed air pollution reductions by CAFO manure digesters, we found numerous types of pollution events and regulatory problems caused by CAFOs participating in the program.[82] The CA-ARB carbon trading program does not take into account the severity of the environmental or health consequences of the regulatory violation.[83] CA-ARB invalidation verifiers should be checking for all violations of local, state, and federal laws that occur between initial waste collection and final disposal—not just specific dates of air-related noncompliance issues.[84] Strengthening regulations for factory farm operations and manure-to-energy projects is essential in protecting local communities from unnecessary health and economic burdens.

There are many ways that we can expand the programs currently in place to benefit struggling farmers and fenceline communities. Ensuring funding from existing federal conservation programs such as the Conservation Stewardship Program and Environmental Quality Incentives Program are directed toward truly sustainable farming practices, as well as expanding and creating local programs such as California's Healthy Soils Program Incentives, will help farmers and communities in numerous ways.[85] It is also imperative that we create new agriculture programs that lead with Indigenous knowledge, while creating equity and economic opportunities for Tribes and Indigenous communities. Shifting away from factory farming will create more jobs while bringing integrity, joy, and community back to farming.

The agribusiness industry's position that manure-to-energy projects help address climate change and CAFO waste problems needs to be more thoroughly studied and scrutinized. The reality experienced by those living in surrounding communities is that CAFOs rarely, if ever, actually deal with the pollution problems they create.[86] If nothing else, manure-to-energy projects have served as a facade to further a failed system that benefits the few at the expense of the public and the environment. Policymakers should be effectively enforcing existing environmental and public health regulations, as well as supporting and promoting more sustainable forms of livestock production and legitimate renewable energy programs.

RECOMMENDATIONS

Other policy suggestions include the following:

- There needs to be more openness and government transparency in permitting and other regulatory issues related to manure-to-energy projects.
- Governmental bodies need to stop providing financial incentives for the development of CAFO manure-to-energy technologies and instead focus funding on the promotion of sustainable agricultural practices.
- Subsidies that go to CAFOs must be eliminated so that traditional pasture-based animal agriculture and regenerative agriculture has a fair opportunity to compete and succeed in a free market.
- The federal government and states should establish, strengthen, and effectively enforce existing air and water pollution laws against CAFOs and stop subsidizing these facilities, particularly those that have caused pollution.
- States and local governments should provide citizens with more robust citizen suit provisions for enforcing environment regulations (similar to those provided under the federal Clean Water Act and Michigan's Environmental Protection Act). This way, the public would have greater ability to enforce environmental regulations when responsible agencies fail to act.

[79]Senate Bill S6599 (New York state climate leadership and community protection act). *The New York State Senate* 2019–2020. <https://www.nysenate.gov/legislation/bills/2019/s6599>.
[80]Ibid at page 13, line 6.
[81]SRAP. ''Trading Pollution: Wisconsin Industrial Dairies with Documented Regulatory Compliance Problems Benefit from California Greenhouse Gas Cap-and-Trade Program.'' Press Release, August 2020. <https://sraproject.org/2020/08/2016/>. (Last accessed on January 31, 2021).
[82]Ibid.
[83]Ibid.
[84]Ibid.
[85]''Are Carbon Markets for Farmers Worth the Hype?.'' <https://civileats.com/2020/09/24/are-carbon-markets-for-farmers-worth-the-hype/>. (Last accessed on March 10, 2021).
[86]SRAP. ''Trading Pollution: Wisconsin Industrial Dairies with Documented Regulatory Compliance Problems Benefit from California Greenhouse Gas Cap-and-Trade program,'' Press Release, August 2020. <https://sraproject.org/2020/08/2016/>. (Last accessed on January 31, 2021).

- Environmental protection and enforcement measures should be strengthened to reduce pollution, as opposed to using pay-to-pollute schemes.
- Congress could condition the receipt of federal subsidy funds in the agricultural sector on the implementation of truly sustainable farming and feedstock practices, and federal government should rescind support for manure biogas projects. Companies participating in the carbon market should have to demonstrate certification of sustainable operations before receiving any federal permits or other government approvals.[87]
- States should adopt or expand programs aimed at greenhouse gas emissions mitigation or sequestration from the forestry or agricultural sectors, such as[88] the following:
  - Create more local subsidy programs, such as California's Smart Agriculture Programs, that create value in land's carbon sink potential and shape cultivation techniques consistent with decarbonization objectives.
  - State governments should consider requiring farm owners to comply with basic climate-friendly practices, such as installing buffer strips next to streams, to receive tax benefits for agricultural activities or easements.[89]
    - The CA-ARB's program should take into consideration the type and magnitude of regulatory violations in addition to total periods of noncompliance when invalidating carbon credits. The current program that invalidates credits only for the time period in which the violation occurred is an inadequate deterrence for poor management.
    - The CA-ARB should meaningfully address input provided by its EJAC to ''[s]top investing in dirty energy'' and to eliminate subsidies and financing for ''waste-to-energy and other unsustainable technologies'' that impose negative impacts on environmental justice communities.
- The carbon credit market should be nationalized, instead of privatized, and credits should only be traded locally.
  - The CA-ARB should place geographic restrictions on trading and limit the amount of pollution ''offset'' credits that companies can use to comply with the program. This will help incentivize local emissions reductions and the annual statewide reduction of GHGs.
- Congress should require the Farm Service Agency and the Farm Credit System lending institutions to offer programs providing favorable credit to farmers and ranchers using truly sustainable and climate-friendly practices relating to all loans.
- States should look to New York State and its climate-forward policy in its Climate Leadership and Community Protection Act (''CLCPA'') and create more innovative laws that protect and prioritize environmental justice communities, ensuring disadvantaged communities are not disproportionately burdened with more polluting infrastructure.
- All states should look to New York's policy that disallows manure-to-energy projects in its renewable energy portfolio and follow suit.

[87]Blake Hudson and Uma Outka. ''Chapter 25: Bioenergy Feedstocks,'' *Legal Pathways to Deep Carbonization in the United States Summary and Key Recommendations,* November 2018: 65. <https://biotech.law.lsu.edu/blog/deep_decarb_summary_booklet_online.pdf>. (Last accessed on March 10, 2021).

[88]Ibid.

[89]Peter H. Lehner and Nathan A. Rosenberg. ''Chapter 30: Agriculture.'' *Legal Pathways to Deep Carbonization in the United States Summary and Key Recommendations*, November 2018: 77–78. <https://biotech.law.lsu.edu/blog/deep_decarb_summary_booklet_online.pdf>. (Last accessed on March 10, 2021).

CONCLUSION

Overall, manure-to-energy projects have the potential to be helpful for small farms (not CAFOs) if the biogas produced on the farm is reused only at that farm. When manure-to-energy projects are installed on factory farms, the processing of the methane produced for the power grid or for the transportation sector releases CO_2 and hazardous air pollutants, and requires the installation of gas pipelines and other infrastructure that leak tremendous volumes of methane. Biomethane production burdens and poisons local communities while degrading our planet's health and sustainability. Manure-to-energy projects are not a sustainable solution to the problems caused by CAFOs because they entrench an already polluting facility with more contamination mechanisms. Local, state, and national governmental entities need to stop promoting and incentivizing CAFO manure-to-energy projects at the expense of the environment and rural communities.

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Environmental impact of biogas: A short review of current knowledge

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ABSTRACT
The social acceptance of biogas is often hampered by environmental and health concerns. In this study, the current knowledge about the impact of biogas technology is presented and discussed. The survey reports the emission rate estimates of the main greenhouse gases (GHG), namely CO_2, CH_4 and N_2O, according to several case studies conducted over the world. Direct emissions of gaseous pollutants are then discussed, with a focus on nitrogen oxides (NO_x); evidences of the importance of suitable biomass and digestate storages are also reported. The current knowledge on the environmental impact induced by final use of digestate is critically discussed, considering both soil fertility and nitrogen release into atmosphere and groundwater; several case studies are reported, showing the importance of NH_3 emissions with regards to secondary aerosol formation. The biogas upgrading to biomethane is also included in the study: with this regard, the methane slip in the off-gas can significantly reduce the environmental benefits.

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Air quality; anaerobic digestion; biogas; digestate; renewable energy; secondary aerosol; waste management

Introduction

The environmental benefits of biogas technology are often highlighted, as a valid and sustainable alternative to fossil fuels.[1] Together with the reduction of greenhouse gas (GHG) emissions, biogas can enhance energy security, thanks to its high energetic potential.[2–4] As a renewable energy source, it allows exploiting agricultural and zootechnical byproducts and municipal wastes, with a lower impact on air quality when compared to combustion-based strategies for these biomasses.[5–7] Furthermore, while ashes from combustion find scarce agronomic applications,[8,9] the by-product of anaerobic digestion, i.e. digestate, looks as a reliable material for agricultural uses.[10] Another important advantage of biogas technology is its easy scalability, allowing exploiting the energetic potential of decentralized biomass sources.[11,12] Finally, biogas can be upgraded to biomethane, suitably used as a vehicle fuel, or injected into national natural gas grids,[13,14]

The energy potential of biogas is reported in Figure 1, based on data from the World Bioenergy Association.[15] For Europe, China and USA, data are detailed in terms of the following sources: manure, agriculture residues, energy crops, organic fraction of municipal solid waste (MSW), agro-industry waste and sewage sludge. For the total world biogas potential, data are only divided into waste (i.e. organic fraction of MSW, agro-industry waste and sewage sludge) and agricultural byproducts (i.e. manure, agriculture residues and energy crops).

In spite of the above cited advantages, social opposition is often observed towards biogas plants, generally based on concerns about environmental and health issues.[16] The frequency on which these opposition phenomena are observed depends on different factors, including the inclusion strategies and the considered country.[17,18] In order to overcome social and cultural barriers hampering a wider diffusion of biogas, the accurate and complete evaluation of the environmental impact of these processes remains an issue of high scientific and technical relevance. The aim of this work is to report an updated state of the art of current knowledge about the environmental impact of biogas and biomethane.

Greenhouse gas emissions

A main objective of biogas industry is the reduction of fossil fuel consumption, with the final goal of mitigating global warming. However, anaerobic digestion is associated to the production of several greenhouse gases, namely carbon dioxide, methane and nitrous oxide. As a consequence, dedicated measures should be taken in order to reduce these emissions. According to Hijazi,[19] the main measures to improve the global warming reduction potential of biogas plants are: to use a flare avoiding methane discharge, to cover tanks, to enhance the efficiency of combined heat and power (CHP) units, to improve the electric power utilisation strategy, to exploit as much thermal energy as possible, to avoid leakages. Similar conclusions were obtained by Buratti and co-workers[20] for the specific case study of cereal crops in Umbria, Italy. Biomethane chain exceeds the minimum value of GHG saving (35%) mainly due to the open storage of digestate; usual practices to improve GHG reduction (up to 68.9%) include using heat and electricity produced by the biogas CHP plant, and covering digestate storage tanks.

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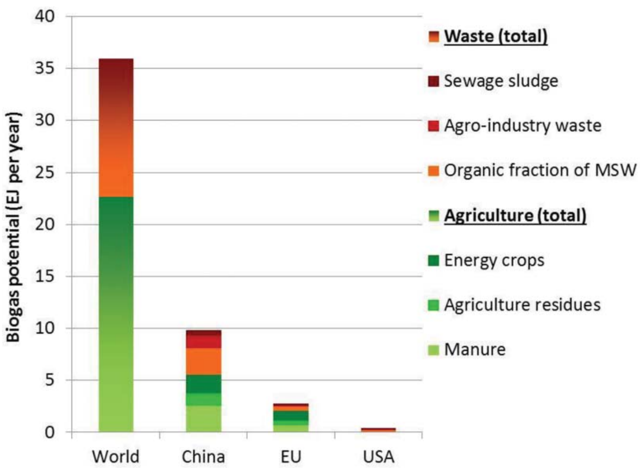

Figure 1. Energy potential of biogas.

The impact induced by biogas plants on global warming needs to be studied case by case. Bachmaier and co-workers[21] calculated the GHG impact of ten agricultural biogas plants. GHG emissions coming from electricity production in the investigated biogas plants ranged from −85 to 251 g CO_2-eq/kWh_{el}, and the GHG saving was 2.31 – 3.16 kWh_{fossil}/kWh_{el}. The results obtained also highlighted that reliable estimates of GHG emissions in the case of electricity production from biogas can be only made on the basis of individual monitoring data, for instance: reduction of direct methane emission and leakage, exploiting of heat obtained from cogeneration, amount and nature of input material, nitrous oxide emission (e.g. from energy crop cultivation) and digestate management. Battini and co-workers,[22] in a case study of an intensive dairy farm situated in the Po valley (Italy), calculated a GHG emission reduction due to anaerobic digestion ranging between −23.7% and −36.5%, depending on digestate management. In a Finnish case study,[23] the GHG release reduction was estimated equal to 177.0, 87.7 and 125.6 Mg of CO_2 eq. yr^{-1} for dairy cow, sow and pig farms, respectively. Optimizing all process parameters looks important with regard to final environmental impact: for instance, a specific case study on wastewater treatment showed that the process optimization could result into the emission abatement equal to 1,103 kg CO_2 eq/d for N_2O, 256 kg eq/d for CO_2 and 87 kg CO_2 eq/d for CH_4.[24]

Carbon dioxide emissions

Harmful compounds and air contaminants are introduced into the environment during biogas production and use through both combustion processes and diffusive emissions. Considering carbon dioxide, combustion of biogas leads to efficient methane oxidation and conversion to CO_2, with a rate of 83.6 kg per GJ (based on a biogas with 65% CH_4 and 35% CO_2[25]). Other releases of this contaminant are related to transport and storage of biomass, as well as digestate use. In the case of both biogas combustion and biomass/digestate emission, CO_2 is considered as biogenic and calculated neutral with regards to the impact on climate. Taking into account the reduction of fossil fuel, it can be demonstrated that biogas production leads globally to mitigation of anthropogenic greenhouse impact of the environment. Poeschl and co-workers[26] have investigated the CO_2 emissions associated to biogas production from several feedstocks, and the relative contribution of feedstock supply, biogas plant operation and infrastructure, biogas utilization and digestate management. According to this study, biogas use gives rise to a negative CO_2 balance because CO_2 caption results every time higher, in absolute values, than positive emissions from feedstock supply and biogas plant operation. As expected, biogas production from byproducts (e.g. from food residues, pomace, slaughter waste, cattle manure, etc.) is a more sustainable approach than energy crops utilization such as whole-wheat plant silage. Besides, digestate management provides significant contributions to total emission reduction in the case of specific feedstock such as municipal solid waste. A dedicated section of this study will below discuss the impact of digestate in full details, in paragraph 5.

Methane emissions

Methane released by biogas processes is not considered relevant for health issues: though exposure to hydrocarbon mixtures can have some adverse effects on humans,[27] no evidence exists of relevant interactions between methane and biologic systems.[28] However, methane is a greenhouse gas whose global warming power is estimated to be 28–36 times higher than CO_2 over 100 years: as such, it is the second major component among anthropogenic greenhouse chemicals.[29] Hence, in evaluating the impact of biogas industry on climate change, methane emissions are a point of primary importance. Methane can be released during biogas incomplete combustion; however a strong contribution to this contaminant comes out from diffusive emission related to biomass storage and digestate management. On the other hand, other biomass management strategies must be taken into account to abate emissions related to biogenic methane. In the above mentioned study of Poeschl and co-workers,[26] methane emissions were also discussed; in all investigated cases, the emission rates were below 5 g kg^{-1}. Considering cattle manure, important reductions in methane emission are related to digestate processing and handling, since this kind of biomass is characterized by high methane emission rate when spread in the field without any pre-treatment.

Nitrous oxide

Besides CO_2 and CH_4, nitrous oxide (N_2O) is another important GHG: Due to its high greenhouse effect potential, N_2O emissions from biogas production processes can result into a significant contribution to global warming budget.[30,31] The relative impact of nitrous oxide mostly depends on the chosen climate metrics: indeed, N_2O impact can even exceed those of CO_2 and CH_4, when the considered metric is Global Temperature change Potential with a time horizon of 100 years (namely GTP-100).[32]

Total GHG emission for energy production from biogas are generally calculated in a range between 0.10 and 0.40 kg CO2-eq/kWh_{el}, which is for instance 22–75% less than GHG emissions caused by the present energy mix in Germany.[33] The wide uncertainty about the estimates of global warming mitigation potential depends on N_2O emission rate assessment as well as on storage and use as a fertilizer of digestate, as discussed in paragraphs below.

Gaseous pollutants from biogas combustion

Along GHG reduction benefits, it must be considered that biogas combustion is associated to release of pollutants in the atmosphere; therefore, the correct assessment of these emissions is a key point in social acceptance of this technology. A summary of emission factors for the main gaseous pollutants are reported in Table 1.

Carbon monoxide (CO) is produced in all oxidation processes of carbon containing materials, and is an important by-product of incomplete combustion of biogas. Methane emission rates are 0.74 and 8.46 and g CO per Nm^{-3} CH_4 for flaring and CHP, respectively.[34] CO emissions related to energy production are estimated in a range between 80 and 265 mg CO MJ^{-1}, depending on the plant efficiency.[35]

Sulphur dioxide (SO_2) emissions from biogas plants manly depend on the desulphurization degree of the introduced biogas. The SO_2 emission rate of a CHP biogas plant is estimated to lie in the range 19.2–25 mg MJ^{-1}.[25] The UK National Society for Clean Air (NSCA) estimates an emission factor of 80 and 100 $g_{SO2}/tonn_{waste}$ for flaring and CHP, respectively.[36] The relatively high SO_2 concentrations in the proximity of biogas plants can depend on different reasons, e.g.: direct emission from biogas combustion, H_2S oxidation from diffusive emissions, and diesel truck exhausts.[37]

Emissions of NO_x are one of the most critical point with regard to environmental impact of biogas plants.[38] According to Kristensen and co-workers,[35] the NO_x emission level of biogas is, in general, higher than for natural gas engines: the averaged aggregated emission factor is 540 g NO_x GJ^{-1}, which is more than three times the rate from natural gas engines. When emission factor is reported to methane consumption, an emission factor of 0.63 and 11.6 g_{NOx}/Nm^3_{CH4} can be assumed for flaring and CHP, respectively.[34] The importance of controlling this pollutant is demonstrated by several case studies. For instance, Battini and co-workers[22] in the above mentioned case study of an intensive dairy farm situated in the Po valley (Italy) reported a low enhancement in acidification (5.5–6.1%), particulate matter emissions (0.7–1.4%) and eutrophication (+0.8%), while on the other hand a significant enhancement in photochemical ozone formation potential (41.6–42.3%) was calculated. In another case study, Carreras-Sospedra and co-workers[39] estimated a potential enhancement of up to 10% of NO_x emission in 2020 in California (US); nevertheless, their study included both biogas and biomass burning. Indeed, the lower emissions of methane from storage and the credits from substituted electricity are not enough to compensate the increase in NO_x emissions from the biogas combustion.

Biogas is a gaseous fuel rich in volatile organic compounds (VOCs), compared to natural gas: indeed, VOCs concentration normally ranges between 5 and 500 mg/Nm^3, and in some cases up to 1700 mg/Nm^3 were observed.[40,41] Generally, only non-methane volatile organic compounds (NMVOC) are considered in these studies. If combustion is assumed to reduce VOCs concentration of 99%,[42] VOCs emission from biogas combustion are in general lower, compared to liquid and solid biofuels. However, a specific critical issue can be highlighted for formaldehyde. In a case study conducted on anaerobic waste treatment plants in Barcelona (Spain), VOC emission factors was in the range 0.9 ± 0.3 g s^{-1}, contributing for 0.3–0.9% of total VOCs in the area. On the other hand, formaldehyde emission factors from biogas engines were found between 0.2 and 3.0 mg s^{-1}, resulting in a ~2% contribution to the total.[43] It is important to remark that a similar emission pattern is observed for natural gas: indeed, formaldehyde is a by-product of methane oxidation. Compared to natural gas, emissions of VOCs are 40% lower in biogas engines, while formaldehyde emissions are slightly lower and higher aldehydes (present in natural gas due to the presence of higher hydrocarbons) are almost absent.[35]

Noticeably, fuel-cycle emissions can be strongly influenced by the raw materials. For instance, CO_2, CO, NO_x, hydrocarbons and particles may differ by a factor of 3–4 between ley crops, straw, sugar beet byproducts, liquid manure, food industry waste and municipal solid waste. On the other hand, differences by a factor of up to 11 can be observed in SO_2 emissions, due to the high variability of H_2S and organic sulphur compounds in the produced biogas.[44]

Table 1. Emission factors of biogas plants operating direct biogas combustion.

Pollutant	Emission factor (g GJ^{-1})	Source
Carbon monoxide (CO)	310	Nielsen et al.,[25]
	256	Kristensen et al.,[35]
Sulphur dioxide (SO_2)	25	Nielsen et al.,[25]
Nitrogen oxides (NO_x)	202	Nielsen et al.,[25]
	540	Kristensen et al.,[35]
Non-methane volatile organic compounds (NMVOC)	10	Nielsen et al.,[25]
	21.15	Kristensen et al.,[35]
Formaldehyde (CH_2O)	8.7	Nielsen et al.,[25]
	14	Kristensen et al.,[35]

Impact of feedstock and digestate storage and treatment

In the biogas combustion management, feedstock and digestate storage and treatments can be the most important processes to achieve the global warming benefits of biogas production processes. Indeed, the impact of a biogas plant on GHG emission is heavily influenced by feedstock storage: most of N_2O can be abated when a closed storage is used for manure and co-digestion feeding.[45]

Emissions from uncovered biomass storage have also been identified as the main ammonia source along the whole biogas production chain,[46] and closed storage is strongly advised.

In a specific French case study of anaerobic digestion and composting plant for municipal solid waste, Beylot and co-workers[38] have identified four conditions for process operation, which highly influence the impact of the whole plant; they are: (i) the features of degradation of the fermentable fraction; (ii) the collection efficiency of gas streams released by biological operations; (iii) the abatement effectiveness of collected pollutants; and (iv) NO_x emission rate from biogas combustion. The importance of digestate storage step has been highlighted by Battini and co-workers,[22] in the above mentioned case study of intensive dairy farm situated in the Po valley (Italy): GHG emission reduction due to AD, calculated as equal to −23.7%, can reach −36.5% when a gas-tight tank is used for digestate storage.

A proper design and management of feedstock and digestate storage units looks also important in order to mitigate the odour impact of the plant. Indeed, the two major sources of the olfactory annoyance are biomass storage production of biogas and digestate composting units.[47] Closed-operated hydrothermal hydrolysis has positive effects on overall fugitive odour control in plants; on the other hand, eventual fugitive emissions during high-temperature and seemingly open pre-treatments can be the principal source of odours.[48]

In conclusion, gas tight storage should always be advised, since the corresponding GHG and ammonia fugitive emissions are even more important those coming from fertilizers.[49] As mentioned above, avoiding leakages and using closed tanks are among the most important ways to reduce the global warming impact of biogas plants.[19]

Impact of digestate final use

The use of agricultural and zootechnical byproducts and MSW as soil improver and fertilizer is a sustainable approach, allowing to reduce the production, transport and use of synthetic chemicals: however, spreading untreated biomass on soils sometimes implies the release into the atmosphere of huge amounts of chemicals such as methane, nitrous oxide, ammonia, volatile hydrocarbons, etc. Anaerobic digestion of biomass followed by the use of digestate as biofertilizer is a common practice related to biogas production. In this paragraph, the current knowledge concerning the environmental impact of this practice is briefly discussed.

A recent study on this topic[50] concluded that direct effects of anaerobic digestion on long-term sustainability in terms of soil fertility and environmental impact at the field level are of minor relevance; indeed, the most relevant issue (with regard to both emissions to atmosphere and in soil fertility) is related to possible changes in cropping systems. According to this study, the main direct aftermaths of anaerobic digestion are short-term effects on soil microbial activity and changes in the soil microbial community. Considering soil quality, digestate is significantly more inert vs. atmospheric and biological agents than the biomass itself: this property results into a lower degradation rate of the organic matter. In fact, labile fractions of original biomass such as carbohydrates are rapidly degraded, causing the enrichment of more persistent molecules such as lignin and non-hydrolysable lipids.[51] In a specific case study on pig slurry anaerobic digestion, a high biological stability of biomasses was achieved, with a Potential Dynamic Respiration Index (PDRI) close to 1,000 mg O_2 kg VS^{-1} h^{-1}.[10]

With regard to nitrate leaching and release into the atmosphere of ammonia and nitrous oxide, the current state of knowledges needs to be improved: however, the impact is considered “negligible or at least ambiguous”.[50] The “ambiguity” of previous studies, as highlighted by this Author, is probably due to the different impact of digestate depending on the type of considered soil. For instance, Eickenscheidt and co-workers[52] investigated the emission of methane, nitrous oxide and ammonia from untreated manure and digestate applied on several soils: while methane emissions did not significantly change, high N_2O emissions were observed in the correspondence of high carbon loadings. A significative impact of soil moisture-soil mineral-N interactions on N_2O emissions was also observed by Senbayram and co-workers.[31]

Considering N_2O and CH_4, digestate can give rise to significant emission rates into the atmosphere: however, these emissions are generally lower than untreated biomass.[53] As for nitrous oxide, digested products are more recalcitrant than fresh slurry; thus, microbial degradation is slower, in which leads to relatively few anoxic microsites and poor N_2O emission compared to fresh slurry application.[54–56] Conversely, methane emissions from digestate are generally lower than those of original biomass, since the methanogenic potential is reduced: this is particularly relevant in the presence of reduced methane coming from manure[26,45] (Poeschl et al., 2012; Boulamanti et al., 2013). As for methane emission, an exception is known in the specific case of rice cultivation: indeed, adding digestate to paddy results into the methane emission rate enhancement from 16.9 to 29.9 g m^{-2},[57] whilst no significant effects are observed for N_2O.[57,58]

Based on the above-cited literature, N_2O and CH_4 emissions from digestate are not critical, while ammonia release and nitrate leaching are still a critical point. For instance, ammonia emissions from digestate higher than from original manure have been observed in several studies.[56,59,60] It was also reported that up to 30% of nitrogen can be lost by ammonia volatilization, due to the enhancement of soil pH.[59,60] Specifically, Matsunaka and co-workers[61] reported a 13% nitrogen volatilization as ammonia, when anaerobically digested cattle slurry was used as soil fertilizer for grassland. The practice of fertilizing soil with anaerobically digested materials increases soil concentration of NO_3^- (+30/40% compared to raw cattle slurry): this is associated to the four times more readily degradable organic C increased microbial biomass, depleting nitrogen and oxygen concentration in soil and resulting in the 10 times increase of CO_2 and N_2O emissions.[62] A proper management of digestate can mitigate its environmental impact: ammonia emission rates ranging from 1.6 to 30.4 were reported, depending on the adopted practice.[63]

With regards to pesticides, heavy metals and harmful microorganisms, the risk of food chain contamination is generally considered low,[64] but the soil burden of persistent organic pollutants (POPs) caused by the use of digestate as biofertilizer still needs to be fully assessed.[65] On the other hand, anaerobic digestion can have relevant effects on phytotoxicity of specific biomass: for instance, the phyto-toxic character of olive mill effluent is reduced after anaerobic digestion,[66] and the degradation of aflatoxin B1 from corn grain can be reached.[67] Finally, an odour reduction up to 82–88% can be obtained.[63]

In conclusion, the main critical issue in final use of digestate is nitrogen release into the environment, which can be reduced by applying the best practices for preserving soil quality. The management of nitrogen dosage is sometimes difficult because of the feedstock variability. It is also important to remark that fugitive emissions from digestate storage are generally more important than those released by its use into soil, as indicated above.[20,49]

Impact on particulate matter

With regards to particulate matter (PM), biogas combustion is not a significant emission source when compared to other fuels: emission factors of 0.238 and 0.232 g/Nm^3_{CH4} have been estimated for flaring and CHP, respectively.[34] However, secondary PM formation can occur, due to NO_x emissions from CHP and NH_3 volatilization from storage and digestate final use. Indeed, during secondary PM formation, the prominent roles of ammonia[68] and NO_x[69] are ascertained. As reported by Boulamanti and co-worker,[45] NO_x emissions are in general the principal source of secondary PM from biogas. As discussed above, closed storage can significantly abate ammonia emissions, resulting also into the global reduction of PM formation from this contaminant.

Impact of biogas upgrading to biomethane

Biomethane production is an efficient approach to increase the market share of biogas, resulting in a further reduction of fossil fuels. The equivalent CO_2 saving raises considerably if methane slip is limited to 0.05%,[70] while the process results no longer sustainable when methane losses reach 4%. Biomethane use as an alternative to gasoil is expected to improve local air quality, with regards to NO_x and particulate matter. As a consequence, biogas upgrading for vehicle fuelling purposes produces optimum benefits with respect to photochemical oxidant formation, marine eutrophication and ecotoxicity; on the other hand, scarce benefits are observed in terms of climate change compared to biogas combustion in CHP.[71]

Depending on several factors such as energy consumption, production and transport of materials used, produced waste and methane slip, the environmental impact of biomethane production depends on the upgrading technology adopted. In PSA, the eventual recovery of the off-gas plays a key role.[72,73] Starr and co-workers[74] reported that the most CO_2-efficient upgrading technology for MSW biogas is the BABIU (bottom ash upgrading) based on ash produced by municipal waste

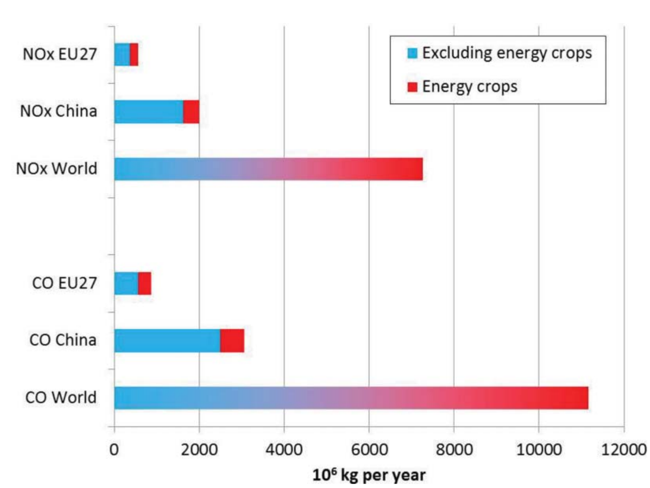

Figure 2. Emission potential of biogas plants for NO_x and CO.

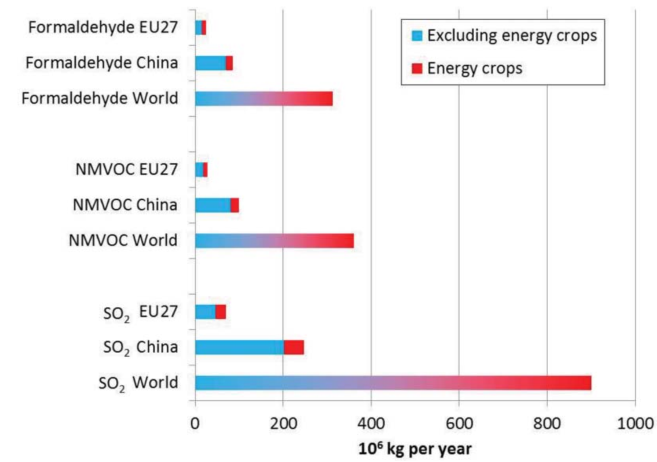

Figure 3. Emission potential of biogas plants for formaldehyde, NMVOC and SO_2.

incinerators. The condition required is that the incinerator lies within 125 km from the biogas upgrading plant. Considering water scrubbing in basic solutions, a lower impact can be achieved by replacing KOH with NaOH. Water from biogas upgrading plants can be recycled in the process or treated as wastewater, depending on chemical composition: the most common VOC in the wastewater of biogas upgrading plants are p-cymene, d-limonene and 2-butanone[75]; the maximum VOC content is observed in MSW treatment plants, reaching up to 238 mg/L, but no inhibition is observed when wastewaters are recycled in the plant.

Along its impact on climate, biomethane use as gasoil substitute of is expected to improve urban air quality, because emission factors of methane are up to 10 times lower than those of liquid fuels, considering PM, VOCs and polycyclic aromatic hydrocarbons.[76] Biomethane injection in the national grid may also reduce residential solid fuels consumption in some specific regions, with relevant benefits on indoor air quality and human health.[77]

Global emission potential

The potential emission associated to biogas plants is reported in Figure 2 (NO_x and CO) and in Figure 3 (for formaldehyde, NMVOC and SO_2). Data are obtained combining emission factors reported in Table 1[25] and energy potential reported in Figure 1. For Europe and China, the contribution of energy crops is reported separately, since their use is often disregarded due to its negative impact on land availability for food. In the case of the global potential, the relative contribution of energy crops is not available.

Conclusions

Biogas can significantly contribute to abate greenhouse gas emissions. However, attention must be payed towards undesired emissions of methane and nitrous oxide (N_2O). The emission budgets of the two compounds are scarcely related to direct release from biogas/biomethane combustion, whilst biomass storage and digestate management are the critical steps. Similar considerations apply to ammonia: to reduce its impact on secondary aerosol formation, efficient biomass and digestate storage should always be recommended. Among all the gaseous pollutants considered in direct emission from biogas combustion, nitrogen oxides (NO_x) level were worth of some concern in several case studies. On the other hand, volatile organic compounds do not seem to constitute a critical issue. Considering the aftermaths of digestate spreading on soil quality, further studies are needed in order to fully assess the long-term impact. In the medium-short term, digestate seems to be preferable compared to untreated biomass. The upgrading to biomethane can generally improve air quality and reduce GHG emissions; however methane losses in the off-gas can affect the sustainability of the whole process.

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ATTACHMENT 52

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Assessment of Methane Emissions from the U.S. Oil and Gas Supply Chain

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Data and materials availability: All data and methods needed to reproduce the results in the paper are provided in the paper or as supplementary material.

Supplementary Materials:
Materials and methods
Figs. S1 to S11
Tables S1 to S12
References (37–77)
Database S1 (supporting datasets to understand and assess the paper's conclusions)
Database S2 (shapefiles of top-down area boundaries used in this work)

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Abstract

Methane emissions from the U.S. oil and natural gas supply chain were estimated using ground-based, facility-scale measurements and validated with aircraft observations in areas accounting for ~30% of U.S. gas production. When scaled up nationally, our facility-based estimate of 2015 supply chain emissions is 13±2 Tg/y, equivalent to 2.3% of gross U.S. gas production. This value is ~60% higher than the U.S. EPA inventory estimate, likely because existing inventory methods miss emissions released during abnormal operating conditions. Methane emissions of this magnitude, per unit of natural gas consumed, produce radiative forcing over a 20-year time horizon comparable to the CO_2 from natural gas combustion. Significant emission reductions are feasible through rapid detection of the root causes of high emissions and deployment of less failure-prone systems.

One Sentence Summary:

A synthesis of recent measurements shows that methane emissions from the U.S. oil and natural gas supply chain exceed U.S. EPA estimates by ~60%.

Methane (CH_4) is a potent greenhouse gas, and CH_4 emissions from human activities since pre-industrial times are responsible for 0.97 W m^{-2} of radiative forcing, as compared to 1.7 W m^{-2} for carbon dioxide (CO_2) (1). CH_4 is removed from the atmosphere much more rapidly than CO_2, thus reducing CH_4 emissions can effectively reduce the near-term rate of warming (2). Sharp growth in U.S. oil and natural gas (O/NG) production beginning around 2005 (3) raised concerns about the climate impacts of increased natural gas use (4, 5). By 2012, disagreement among published estimates of CH_4 emissions from U.S. natural gas operations led to a broad consensus that additional data were needed to better characterize emission rates (4–7). A large body of field measurements made between 2012 and 2016 (Table S1) has dramatically improved understanding of the sources and magnitude of CH_4 emissions from the industry's operations. Brandt *et al.* summarized the early literature (8); other assessments incorporated elements of recent data (9–11). This work synthesizes recent studies to provide an improved overall assessment of emissions from the O/NG supply chain, which we define to include all operations associated with oil and natural gas production, processing and transport (Section S1.0) (12).

Measurements of O/NG CH_4 emissions can be classified as either top-down (TD) or bottom-up (BU). TD studies quantify ambient methane enhancements using aircraft, satellites or tower networks and infer aggregate emissions from all contributing sources across large geographies. TD estimates for nine O/NG production areas have been reported to date (Table S2). These areas are distributed across the U.S. (Fig. S1) and account for ~33% of natural gas, ~24% of oil production, and ~14% of all wells (13). Areas sampled in TD studies also span the range of hydrocarbon characteristics (predominantly gas, predominantly oil, or mixed), as well as a range of production characteristics such as well productivity and maturity. In contrast, BU studies generate regional, state, or national emission estimates by aggregating and extrapolating measured emissions from individual pieces of equipment, operations, or facilities, using measurements made directly at the emission point or, in the case of facilities, directly downwind.

Recent BU studies have been performed on equipment or facilities that are expected to represent the vast majority of emissions from the O/NG supply chain (Table S1). In this work we integrate the results of recent facility-scale BU studies to estimate CH_4 emissions from the U.S. O/NG supply chain, and then we validate the results using TD studies (Section S1). The probability distributions of our BU methodology are based on observed facility-level emissions, in contrast to the component-by-component approach used for conventional inventories. We thus capture enhancements produced by all sources within a facility, including the heavy tail of the distribution. When the BU estimate is developed in this manner, direct comparison of BU and TD estimates of CH_4 emissions in the nine basins for which TD measurements have been reported indicates agreement between methods, within estimated uncertainty ranges (Fig. 1).

Our national BU estimate of total CH_4 emissions in 2015 from the U.S. O/NG supply chain is 13 (+2.1/−1.6, 95% confidence interval) Tg CH_4/y (Table 1). This estimate of O/NG CH_4 emissions can also be expressed as a production-normalized emission rate of 2.3% (+0.4%/−0.3%) by normalizing by gross natural gas production (33 tcf/y, (13) with average CH_4 content of 90 vol%). Roughly 85% of national BU emissions are from production, gathering, and processing sources, which are concentrated in active O/NG production areas.

Our assessment does not update emissions from local distribution and end use of natural gas, due to insufficient information addressing this portion of the supply chain. However, recent studies suggest that local distribution emissions are significant, exceeding the current inventory estimate (14–16), and that end-user emissions might also be important. If these findings prove to be representative, overall emissions from the natural gas supply chain would increase relative to the value in Table 1 (Section S1.5).

Our BU method and TD measurements yield similar estimates of U.S. O/NG CH_4 emissions in 2015, and both are significantly higher than the corresponding estimate in the U.S. Environmental Protection Agency's Greenhouse Gas Inventory (EPA GHGI) (Table 1, Section S1.3) (17). Discrepancies between TD estimates and the EPA GHGI have been reported previously (8, 18). Our BU estimate is 63% higher than the EPA GHGI, largely due to a more than two-fold difference in the production segment (Table 1). The discrepancy in production sector emissions alone is ~4 Tg CH_4/y, an amount larger than the emissions from any other O/NG supply chain segment. Such a large difference cannot be attributed to expected uncertainty in either estimate: the extremal ends of the 95% confidence intervals for each estimate differ by 20% (i.e., ~12 Tg/y for the lower bound of our BU estimate can be compared to ~10 Tg/y for the upper bound of the EPA GHGI estimate).

We believe the reason for such large divergence is that sampling methods underlying conventional inventories systematically underestimate total emissions because they miss high emissions caused by abnormal operating conditions (e.g., malfunctions). Distributions of measured emissions from production sites in BU studies are invariably "tail-heavy", with

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large emission rates measured at a small subset of sites at any single point in time (19–22). The difference between the EPA GHGI and BU estimates derived from facility-level measurements is explained if measurements used to develop GHGI emission factors under-sampled abnormal operating conditions encountered during the BU work. Component-based inventory estimates like the GHGI have been shown to underestimate facility-level emissions (23), probably because of the technical difficulty and safety and liability risks associated with measuring large emissions from, for example, venting tanks such as those observed in aerial surveys (24).

Abnormal conditions causing high CH_4 emissions have been observed in studies across the O/NG supply chain. An analysis of site-scale emission measurements in the Barnett Shale concluded that equipment behaving as designed could not explain the number of high-emitting production sites in the region (23). An extensive aerial infrared camera survey of ~8,000 production sites in seven U.S. O/NG basins found that ~4% of surveyed sites had one or more observable high emission-rate plumes (24) (detection threshold of ~3–10 kg CH_4/h was 2–7 times higher than mean production site emissions estimated in this work). Emissions released from liquid storage tank hatches and vents represented 90% of these sightings. It appears that abnormal operating conditions must be largely responsible, because the observation frequency was too high to be attributed to routine operations like condensate flashing or liquid unloadings alone (24). All other observations were due to anomalous venting from dehydrators, separators and flares. Notably, the two largest sources of aggregate emissions in the EPA GHGI – pneumatic controllers and equipment leaks – were never observed from these aerial surveys. Similarly, a national survey of gathering facilities found that emission rates were four times higher at the 20% of facilities where substantial tank venting emissions were observed, as compared to the 80% of facilities without such venting (25). In addition, very large emissions from leaking isolation valves at transmission and storage facilities were quantified using downwind measurement but could not be accurately (or safely) measured using on-site methods (26). There is an urgent need to complete equipment-based measurement campaigns that capture these large emission events, so that their causes are better understood.

Outdated component emission factors and temporal bias are unlikely to explain the difference between our facility-based BU estimate and the GHGI. First, an equipment-level inventory analogous to the EPA GHGI but updated with recent direct measurements of component emissions (Section S1.4) predicts total production emissions that are within ~10% of the EPA GHGI, although the contributions of individual source categories differ significantly (Table S3). Second, we consider unlikely an alternative hypothesis that systematically higher emissions during daytime sampling cause a high bias in TD methods (Section S1.6). Two other factors may lead to low bias in EPA GHGI and similar inventory estimates. Operator cooperation is required to obtain site access for emission measurements (8). Operators with lower-emitting sites are plausibly more likely to cooperate in such studies, and workers are likely to be more careful to avoid errors or fix problems when measurement teams are on site or about to arrive. The potential bias due to this “opt-in” study design is very challenging to determine. We therefore rely primarily on site-level, downwind measurement methods with limited or no operator forewarning to construct our BU estimate. Another possible source of bias is measurement error. It has been suggested

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that malfunction of a measurement instrument widely used in the O/NG industry contributes to underestimated emissions in inventories (27); however, this cannot explain the >2× difference in production emissions (28).

The tail-heavy distribution for many O/NG CH_4 emission sources has important implications for mitigation since it suggests that most sources – whether they represent whole facilities or individual pieces of equipment – can have lower emissions when they operate as designed. We anticipate that significant emissions reductions could be achieved by deploying well-designed emission detection and repair systems that are capable of identifying abnormally operating facilities or equipment. For example, pneumatic controllers and equipment leaks are the largest emission sources in the O/NG production segment exclusive of missing emission sources (38% and 21%, respectively; Table S3) with malfunctioning controllers contributing 66% of total pneumatic controller emissions (Section S1.4) and equipment leaks 60% higher than the GHGI estimate.

Gathering operations, which transport unprocessed natural gas from production sites to processing plants or transmission pipelines, produce ~20% of total O/NG supply chain CH_4 emissions. Until the publication of recent measurements (29), these emissions were largely unaccounted by the EPA GHGI. Gas processing, transmission and storage together contribute another ~20% of total O/NG supply chain emissions, most of which come from ~2,500 processing and compression facilities.

Our estimate of emissions from the U.S. O/NG supply chain (13 Tg CH_4/y) compares to the EPA estimate of 18 Tg CH_4/y for all other anthropogenic CH_4 sources (17). Natural gas losses are a waste of a limited natural resource (~$2 billion/y), increase global levels of surface ozone pollution (30), and significantly erode the potential climate benefits of natural gas use. Indeed, our estimate of CH_4 emissions across the supply chain, per unit of gas consumed, results in roughly the same radiative forcing as does the CO_2 from combustion of natural gas over a 20-year time horizon (31% over 100 years). Moreover, the climate impact of 13 Tg CH_4/y over a 20-year time horizon roughly equals that from the annual CO_2 emissions from all U.S. coal-fired power plants operating in 2015 (31% of the impact over a 100-year time horizon) (Section S1.7).

We suggest that inventory methods would be improved by including the substantial volume of missing O/NG CH_4 emissions evident from the large body of scientific work now available and synthesized here. Such empirical adjustments based on observed data have been previously used in air quality management (31).

The large spatial and temporal variability in CH_4 emissions for similar equipment and facilities (due to equipment malfunction and other abnormal operating conditions) reinforces the conclusion that significant emission reductions are feasible. Key aspects of effective mitigation include pairing well-established technologies and best practices for routine emission sources with economically viable systems to rapidly detect the root causes of high emissions arising from abnormal conditions. The latter could involve combinations of current technologies such as on-site leak surveys by company personnel using optical gas imaging (32), deployment of passive sensors at individual facilities (33, 34) or mounted on ground-based work trucks (35), and *in situ* remote sensing approaches using tower networks, aircraft or satellites (36). Over time, the development of less failure-prone systems would be expected through repeated observation of and further research into common causes of abnormal emissions, followed by re-engineered design of individual components and processes.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

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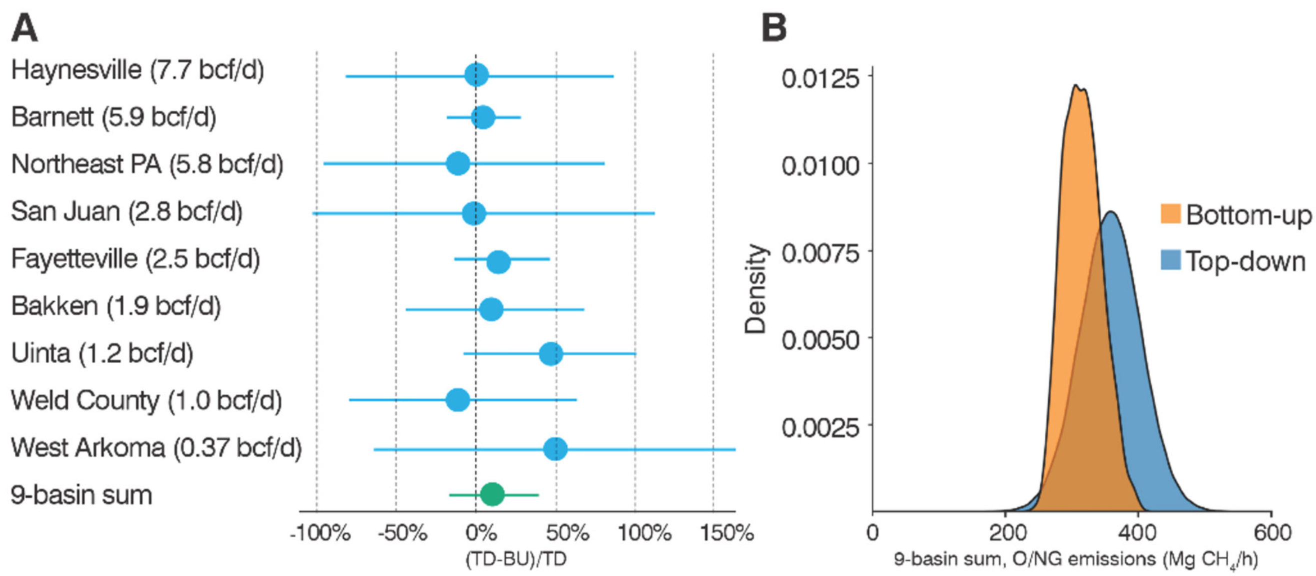

Figure 1.

Comparison of this work's bottom-up (BU) estimates of methane emissions from oil and natural gas (O/NG) sources to top-down (TD) estimates in nine U.S. O/NG production areas. (O/NG) sources to top-down (TD) estimates in nine U.S. O/NG production areas. A: relative differences of the TD and BU mean emissions, normalized by the TD value, rank ordered by natural gas production in billion cubic feet per day (bcf/d, where 1 bcf = 2.8×10^7 m^3). Error bars represent 95% confidence intervals. B: distributions of the 9-basin sum of TD and BU mean estimates (blue and orange probability density, respectively). Neither the ensemble of TD-BU pairs (A) nor the 9-basin sum of means (B) are statistically different (p=0.13 by a randomization test, and mean difference of 11% [95% confidence interval of −17% to 41%]).

Table 1.

Summary of this work's bottom-up estimates of CH4 emissions from the U.S. oil and natural gas (O/NG) supply chain (95% confidence interval) and comparison to the EPA Greenhouse Gas Inventory (GHGI).

Industry segment	2015 CH_4 Emissions (Tg/y)	
	This work (bottom-up)	**EPA GHGI (17)**
Production	7.6 (+1.9/−1.6)	3.5
Gathering	2.6 (+0.59/−0.18)	2.3
Processing	0.72 (+0.20/−0.071)	0.44
Transmission and Storage	1.8 (+0.35/−0.22)	1.4
Local Distribution *	0.44 (+0.51/−0.22)	0.44
Oil Refining and Transportation *	0.034 (+0.050/−0.008)	0.034
U.S. O/NG total	13 (+2.1/−1.7)	8.1 (+2.1/−1.4) †

* This work's emission estimates for these sources are taken directly from the GHGI. The local distribution estimate is expected to be a lower bound on actual emissions and does not include losses downstream of customer meters due to leaks or incomplete combustion (Section S1.5).

† The GHGI only reports industry-wide uncertainties.

ATTACHMENT 53

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Effects on Carbon and Nitrogen Emissions due to Swine Manure Removal for Biofuel Production

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Methane (CH_4) and ammonia (NH_3) are emitted from swine-manure processing lagoons, contributing to global climate change and reducing air quality. Manure diverted to biofuel production is proposed as a means to reduce CH_4 emissions. At a swine confined animal feeding operation in the U.S. Central Great Basin, animal manure was diverted from 12 farms to a biofuel facility and converted to methanol. Ammonia emissions were determined using the De Visscher Model from measured data of dissolved lagoon ammoniacal N concentrations, pH, temperature, and wind speed at the lagoon sites. Other lagoon gas emissions were measured with subsurface gas collection devices and gas chromatography analysis. During 2 yr of study, CO_2 and CH_4 emissions from the primary lagoons decreased 11 and 12%, respectfully, as a result of the biofuel process, compared with concurrently measured control lagoon emissions. Ammonia emissions increased 47% compared with control lagoons. The reduction of CH_4 and increase in NH_3 emissions agrees with a short-term study measured at this location by Lagrangian inverse dispersion analysis. The increase in NH_3 emissions was primarily due to an increase in lagoon solution pH attributable to decreased methanogenesis. Also observed due to biofuel production was a 20% decrease in conversion of total ammoniacal N to N_2, a secondary process for the removal of N in anaerobic waste lagoons. The increase in NH_3 emissions can be partially attributed to the decrease in N_2 production by a proposed NH_4^+ conversion to N_2 mechanism. This mechanism predicts that a decrease in NH_4^+ conversion to N_2 increases ammoniacal N pH. Both effects increase NH_3 emissions. It is unknown whether the decrease in NH_4^+ conversion to N_2 is a direct or physical result of the decrease in methanogenesis. Procedures and practices intended to reduce emissions of one pollutant can have an unintended consequence on the emissions of another pollutant.

Confined animal feeding operations are economical from a production standpoint; however, concentrating a large number of animals in a relatively small geographical area increases the amount of feces and urine (manure) generated that must be managed by the producer. Manure is generally managed by anaerobic digestion and/or land application. In the digestion process, animal manure is pumped into large open retention ponds (lagoons) for microbial digestion. As the manure is digested, various gases, including methane (CH_4), carbon dioxide (CO_2), ammonia (NH_3), and dinitrogen (N_2), are released into the atmosphere. Methane, CO_2, and NH_3 have environmental consequences. Methane is a global climate change gas. Although CO_2 is normally considered a global climate change gas, CO_2 from agricultural sources is simply returning carbon (C) sequestered by agriculture to the environment. Nevertheless, agricultural CO_2 emissions are of interest because of their role in C cycling. Ammonia reacts with acidic gases in the atmosphere (mainly from fossil fuel combustion) to form ammonium (NH_4^+) aerosols. These aerosols affect air quality and are involved in wet and dry deposition, enriching N availability to cropping and natural ecosystems. Nitrogen may be beneficial in cropping systems, but in natural ecosystems deposited N may provide competitive advantage to nitrophyllic over nitrophobic plants.

Ammonia emissions were initially reported by the USDA to be about 60% of feed input (Hatfield et al., 1993). The USEPA estimates NH_3–N emissions to be 71% of the N discharged into the lagoon or about 53% of feed input (USEPA, 2004). The state of North Carolina estimates that NH_3–N emissions are 36% of feed input (Doorne et al., 2002). More recently, the USDA in the North Carolina and Georgia Coastal Plains (humid East) and in the U.S. Central Great Basin (arid West) (Harper and Sharpe, 1998; Harper et al., 2000, 2004, 2010) have shown that manure processing lagoons emit significantly less NH_3 than previously thought. Harper et al. (2004) found, in a highly measured swine production operation, that about 5.2% of the N going into the swine production operation as feed left the lagoon as NH_3, along with another 7.4% from housing and 2% from field application of waste effluent to nearby crops

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Abbreviations: bLS, inverse Lagrangian stochastic analysis (an inverse dispersion analysis technology); GC, gas chromatography.

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(total of 14.6%). These emissions and other N components from a total N system's analysis (protein N output, crop N use, soil N denitrification to N_2 and nitrous oxide [N_2O] emissions, and lagoon NH_4^+ conversion to N_2 gas) accounted for 95% of the total feed N input to the system. Studies in the Central Great Basin (semiarid Utah) showed similar total NH_3 emissions of 15.2% of feed N but with a breakdown of 3.5 and 11.7% for housing and lagoon emissions, respectively (Harper, Weaver, and Flesch, unpublished data, 2002–2003).

Methane gas is particularly interesting not only because it has a radiative forcing about 25 times larger than CO_2 but also because it can be used as an energy source. Thus, capture of the energy-rich CH_4 gas greatly reduces the net radiative forcing of the lagoon gases emitted. To capture this escaping energy, the CH_4 can be collected from the manure-management lagoons or the manure may be collected from the production system and transported to a biofuel production facility where the organic materials are allowed to be microbially digested at an elevated temperature. The gases from this thermally accelerated digestion process are collected and scrubbed to remove CO_2 and other minority gases produced during the process. The purified CH_4 gas is converted onsite to methanol to be used in biodiesel for energy recovery.

In 2004, a biofuel treatment system was constructed at an existing facility in the semiarid U.S. Central Great Basin. The emissions from this farm system had previously been measured during 2000–2001 (Harper et al., 2006), so this was an ideal location to compare the effects of treatment for conversion of manure to biofuels (on gas emissions) with the previously measured gas emissions. In addition to this study, a concurrent short-term study was conducted onsite using different measurement technology (Harper et al., 2010). Thus, not only did we have an excellent location for the study of the effect of biofuel implementation, but we also were able to measure the effect on emissions by more than one technology with a short-duration and a relatively long-duration study.

There are complex interactions between C and N compounds during manure processing (Harper et al., 2000, 2004), and the conversion from a lagoon manure-management system to a biofuel system was anticipated to affect C and N emissions. Much of the N entering into manure management lagoons is converted to N_2 gas (Harper and Sharpe, 1998; Harper et al., 2000, 2001, 2004) by microbial and/or chemical conversion of NH_4^+ to N_2 (Van Cleemput, 1972, 1997, 1998). Although the exact mechanism of this process is not known, Van Cleemput (1998) suggested the term *chemical conversion of* NH_4^+ *to* N_2 be used to describe the process. Others have suggested the terms *chemical denitrification* and *chemo-denitrification*. We choose to describe this process as NH_4^+ conversion to N_2. The term *denitrification* is not used to avoid confusion with the denitrification process involving nitrate (NO_3^-) consumption, and *chemical* is also omitted because the exact process of this conversion is not known. Although little emissions research on the relationship between CH_4 and CO_2 has been published (Sharpe et al., 2001; McGinn et al., 2006) in confined animal feeding operations, Harper et al. (2000) found correlations between emissions of NH_3, CH_4, N_2O, and CO_2 from manure-processing lagoons, suggesting that manipulation of the manure-management system to reduce emissions of one constituent may affect the emissions of another. Harper et al. (2000) showed that, in a series of manure lagoons with a high rate of methanogenesis, there was a significant amount of NH_4^+ conversion to benign N_2 (molecular N) gas; however, when methanogenesis decreased, smaller emissions of N_2 occurred, and higher rates of N_2O were produced. They found that as the amount of NO_3^- and O_2 increased in the latter lagoon stages, even though smaller amounts of total gas flux were produced, the percentage of N_2 in the collected samples increased. The higher NO_3^- and O_2, along with lower NH_4^+, may have provided more suitable conditions for biological denitrification. Furthermore, there is no intermediate step producing N_2O in the NH_4^+ conversion to N_2, whereas in biological denitrification, N_2O is produced as an intermediate step (e.g., increasing concentrations of N_2O as O_2 and NO_3^- are evident). Harper et al. (unpublished data) found a similar relationship between methanogenesis and conversion of NH_4^+ to N_2 in a series of six geographically widely spaced swine operations in North Carolina. The purpose of our study was to evaluate the effects on C and N emissions from converting these farms to a biofuel production system.

Materials and Methods

From 2000 to 2003, CH_4 and NH_3 emissions were quantified from three 12,000-head swine finishing farms' manure processing lagoons in the U.S. Central Great Basin. Each farm consisted of three closely spaced and joined barns with adjacent primary and secondary manure processing lagoons (open-air). Each farm in the system had a 1.69-ha primary lagoon (8 m deep) where effluent from the control farm houses was transferred for processing. Each farm had a secondary lagoon for overflow from the primary lagoon if needed. The control farms' primary lagoons were organic rich with surface dissolved oxygen concentration less than 1% saturation. Due to the darkness of the liquid, no algae or surface bacterial layer was present. The secondary lagoons (0.59 ha) were similar in appearance, but the liquid was thicker due to evaporation. Evaporation and decomposition in both the biofuel and control farms maintained long-term manure equilibrium at the farm, and no manure removal was required.

In 2004, three of the previously measured farms, along with an additional nine finishing farms, were converted to a biofuel production system. This operation provided an excellent opportunity to test the effect of the biofuel production system on trace-gas emissions because the emissions from the biofuel farms could be compared with historical data from these same farms and with emissions from similar farms. At each of the biofuel farms, manure from the swine houses was first pumped to a small excavated collecting basin where organic material from the individual facilities is gravitationally concentrated. The organic-rich bottom layer was then pumped to the central biofuel production facility for further organic matter concentration. The final concentrated organic matter was then pumped to digesters for thermally accelerated anaerobic bacterial digestion resulting in biogas production. The CH_4–rich biogas produced in the digesters was then conveyed to a biomethanol conversion plant on site. The biomethanol produced at the site was transported to a refinery for conversion to biodiesel. All effluent from the concentrators and effluent from the digesters was then returned to the farms for further digestion and to maintain water levels at the manure-management effluent lagoons. Previous studies (2000–2003) from three farms at the production site (Farms 1, 2, and 3) were used as control data because they were converted to biofuel (treatment) production farms. Only a portion of the similar 12,000-head farms was converted to the biofuel manure treatment system within the swine farm complex. Three unconverted farms (Farms 4, 5, and 6) were studied intensively during our study and served as concurrent control farms.

The De Visscher model (De Visscher et al., 2002) was used to calculate NH_3 emissions from the lagoons. Ammonia emission from the De Visscher model (De Visscher et al., 2002) is described as a two-film model where NH_3 diffuses from the bulk liquid to the air–water interface through a thin boundary layer, each layer being characterized by a transfer coefficient. The model combines Henry's Law with the diffusion relationships and the mass-transfer coefficient in the boundary layer (derived from the relationship of the wind speed measured at a specified height). Because diffusion is concentration dependent, the higher the concentration of dissolved NH_3 in solution, the greater the diffusion rate. Dissolved NH_3 is dependent on the ammoniacal N concentration ($[NH_4^+ + NH_3]$) and the pH of the solution. The pH affects the concentration of dissolved NH_3 in a nonlinear relationship, where $[NH_3(aq)] = [NH_4^+ + NH_3] \times K_a/(10^{(-pH)} + K_a)$; K_a is the acid dissociation constant for NH_4^+. The model explained 70% of the variability compared with noninterference micrometeorological techniques using average daily NH_3 emissions. Average daily NH_3 emissions were calculated from average daily wind speed, lagoon temperature, pH, and dissolved ammoniacal N concentrations.

Water temperature, wind speed, pH, and $[NH_4^+ + NH_3]$ at the lagoons were measured for 13 mo. Wind speed data were collected using an anemometer (R.M. Young 03001-5, Campbell Scientific) whose output was recorded as 15-min averages. Water temperature data for emissions calculations were collected using one data logger per lagoon (HOBO Water Temp Pro, Onset Computer Corp.) attached to a buoy maintaining the logger at a depth of 5 to 7 cm. An additional data logger was placed at the bottom of each primary lagoon. One lagoon water sample was collected from the surface (0–4 cm) of each lagoon twice a month and then frozen for subsequent pH and $[NH_3 + NH_4^+]$ analysis. The pH and $NH_3 + NH_4^+$ measurements were performed on the thawed samples using an appropriate ion selective electrode (Accumet, Fisher Scientific). Water temperature and wind speed data were averaged to calculate daily average values. Although the actual measured pH and $[NH_3 + NH_4^+]$ values were used for emissions calculations on the days they were measured, days between consecutive measurements were represented by the average value of the measurements. For example, if the pH was 8.15 on 5 July and 8.25 on 19 July, a pH value of 8.15 would be used on 5 July, a pH value of 8.20 would be used from 6 July until 18 July, and a pH value of 8.25 would be used on 19 July for emissions calculations. The amount of feed N was determined using feed input data provided by the producer. Percent N in the feed was determined by Ward Laboratories.

The emissions of CH_4 and CO_2 gases were measured for approximately 24 mo. Methane and CO_2 gas emissions were determined by first measuring total biological gas emissions using gas collection devices (carboy method) to capture the evolved gases below the lagoon surface, with subsequent analysis by gas chromatography (GC). The gas collection devices were constructed from 5-gal plastic carboys with the bottoms removed (Harper et al., 2000, 2004). The carboys were tethered to boat anchors with 8 to 9 m of rope to the bottom of the lagoon, and each remained upright by ballast secured to the bottom of the carboy and a buoy secured to the top. The carboys were filled with water and allowed to sink below the water surface. The gases produced from decomposition of the animal manure displaced the water in the carboy, causing it to rise to the surface. Each carboy was graduated with liter lines to measure how much gas was in the carboy. Carboy volume was measured as frequently as twice a week in the summer and as infrequently as twice a month in the winter when gas production was low. Six carboys were placed uniformly in each primary manure treatment lagoon. The volumetric gas emission rates (adjusted for vapor pressure) were converted to kmol gas to compensate for the variation in gas composition along with change in lagoon temperature throughout the year. The kmol of gas were calculated using the monthly average surface temperatures of the lagoon (to estimate gas temperature), the average atmospheric pressure (83.6 kPa) corresponding to the elevation of the farms, and the ideal gas law.

The composition of the gases was determined by GC. Valves were placed in three carboys in each lagoon to allow for the transfer of the biological gases to SUMMA canisters (B.R.C. Rasmussen) for GC analysis. Gas samples were collected in the SUMMA canisters as frequently as every 2 to 3 wk in the summer to every 4 to 8 wk in the winter. Earlier tests using an inert gas (i.e., helium) in the containers provided a measure of potential atmospheric N_2 contamination in the collection and transfer processes and showed contamination to be less than 2% (Harper et al., 2004). Collected gases were analyzed using an Agilent 5390 gas chromatograph with a Supelco PLOT-5 column and a thermal conductivity detector. The configuration of the gas chromatograph allows for the determination of the composition of the major components present in the gas samples. The composition of the collected gases and the total gas emission rates were then used to calculate the emission of each gas.

Statistical Analysis

Most of the results of the emission measurements were analyzed using the independent two-sample *t* test assuming equal variances. All *p* values reported in the paper correspond to this *t* test. Because some interesting trends in average monthly emissions were seen, we examined the monthly means as paired data. The traditional paired *t* test and repeated measures ANOVA cannot be used because the monthly means are not normally distributed. Instead, we used a nonparametric test, the one-tailed Wilcoxon signed-rank test. Results of this analysis are simply reported as significant or not significant at the 95% confidence level.

Results and Discussion

Ammonia Emissions

Table 1 details the monthly average NH_3 emissions from each primary lagoon of the study. Ammonia emission parameters were measured from 1 July 2004 to 31 July 2005. Thus, July emissions were determined twice. The results from July 2004 and July 2005 were used to calculate an average July emissions rate, which, with all the month's emission averages, was used to calculate an annual emission rate. Annual average NH_3–N emissions from the biofuel farms primary lagoons were 18.8 kg NH_3–N d^{-1} $lagoon^{-1}$ larger ($p < 0.001$) than emissions from the control farms. We recognize that many units are used in expressing emissions and emission factors. For clarity, emissions expressed in other units have been omitted and are summarized in Table 2. All primary lagoons are equal (1.69 ha) and all secondary farms are equal (0.59 ha) in this study. Emissions per lagoon are used to evaluate the effect of the biofuel treatment system. In all but 1 mo, the control farm emissions were less than the biofuel farm NH_3 emissions. In 5 out of the 13 months, NH_3 emissions were significantly larger ($p < 0.05$) from the biofuel farms than from the control farms. In the month when emissions from the control farms were higher, the difference was very small (<1%).

The increase in NH_3–N emissions can be explained by a change in chemistry in the lagoons. Although $[NH_4^+ + NH_3]$ was not found to be higher in the biofuel lagoons (Fig. 1A), the average yearly pH of the biofuel farms was higher ($p < 0.01$). The pH (Fig. 1B) of the lagoons determines the form of ammoniacal N. Ammonia emissions are directly dependent on the concentration of dissolved NH_3 ($[NH_3(aq)]$), not on $[NH_4^+ + NH_3]$. Because NH_3 is a soluble gas, it exists in significant concentrations dissolved in the liquid and as a gas over the liquid. At equilibrium, the vapor pressure of this gas is directly proportional to ($[NH_3(aq)]$). The difference in $[NH_3(aq)]$ is shown in Fig. 1C, where the average monthly $[NH_4^+ + NH_3]$ for each lagoon and the average monthly pH for each lagoon are used to calculate the monthly $[NH_3(aq)]$ for each farm, which is used to calculate the average dissolved $[NH_3]$ for each farm system. On a year-to-year basis, the $[NH_3(aq)]$ was 31% larger ($p < 0.001$) in the biofuel farms' primary lagoons compared with the control lagoons.

The NH_3–N emissions were determined in three different studies at this production site. The farms were studied in 2000–2001 (Harper et al., 2006), in 2002–2003, and in the present study. The average yearly NH_3–N emissions from the primary lagoons of each farm are summarized in Table 3. The control farm results from 2000–2001 were from an earlier study (Harper et al., 2006) in which the published yearly emissions were divided by 365, and the results from 2002–2003 are from a previously unpublished study. The 2004–2005 control and the biofuel results are from the present study. The average control farm NH_3 emission of the three studies was 52.7 ± 1.8 kg NH_3–N d^{-1} $lagoon^{-1}$, which is significantly lower than emissions from the biofuel-treated lagoons (71.5 ± 1.6 kg NH_3–N d^{-1} $lagoon^{-1}$). Thus, the biofuel farms' NH_3–N primary lagoon emissions were 36% ($p < 0.001$) larger than the control lagoons. The close agreement of control NH_3 emissions between the three studies demonstrates a consistency between annual emissions, which is due not only to similar annual effluent characteristics ($[NH_4^+ + NH_3]$ and pH) but also to similar effluent temperature and wind speed. For example, the average annual wind speeds (July–July) for 2002–2003 and 2004–2005 were 2.76 and 2.79 m s^{-1}, respectively.

Emissions from secondary lagoons for 2004–2005 are summarized in Table 4. In this treatment system, the secondary lagoons are used as overflow reservoirs to capture any extra liquid from primary lagoons and to provide a larger surface area for evaporation. There is little microbial digestion (no bubbles observed at surface of lagoons) in these lagoons; thus, there would be little direct effect on NH_3 emissions anticipated due to reduced methanogenesis and conversion of ammoniacal N to N_2 gas (Harper et al., 2000, 2004). However, any changes in pH and $[NH_4^+ + NH_3]$ seen in the primary lagoons would also be carried over to the secondary lagoons. In the initial months of the study, some of the biofuel farms' secondary lagoons were empty due to water being diverted to the biofuel digester basin. Thus, emissions in the biofuel farms' secondary lagoons could not be measured at the biofuel farms during July, August, and September 2004. Between October and July, the biofuel farm secondary lagoons emitted 9.9 kg NH_3–N d^{-1} $lagoon^{-1}$ more (115%; $p < 0.01$) than those of the control farms. Omission of the first 3 mo is valid because the biofuel data are incomplete for this time period and the biofuel system had not completely affected the secondary farms until after this time. The 10-mo averages for the biofuel

Table 1. Monthly summary of monthly ammonia emissions for biofuel and control farm primary lagoons from July 2004 to July 2005.

Month	Ammonia emissions				Statistically significant
	Biofuel		Control		
	Monthly average	SD	Monthly average	SD	
	kg NH_3–N d^{-1} $lagoon^{-1}$†				
July	111.2	8.2	96.1	6.9	
Aug.	131.8	20.8	111.1	8.8	
Sept.	126.1	40.7	75.2	11.3	
Oct.	68.4	10.0	43.9	6.5	*
Nov.	20.5	2.4	20.8	1.4	
Dec.	17.0	0.7	11.6	0.4	*
Jan.	15.7	4.3	9.8	1.3	
Feb.	24.7	2.4	15.6	1.5	*
Mar.	42.4	3.0	31.6	3.2	*
Apr.	58.1	15.6	46.5	9.6	
May	109.6	19.3	75.4	15.6	
June	111.9	5.4	97.6	21.0	
July	145.0	4.2	100.3	5.6	*
Yearly average	71.5	1.6	53.3	2.7	***
Percent of feed	14.1		10.5		N/A

* Significant at the 0.05 probability level.

*** Significant at the 0.001 probability level.

† Lagoon dimensions are 1.69 ha each. Emissions were calculated from the De Visscher Model (De Visscher et al., 2002) using pH and ammoniacal N concentration, lagoon temperature, and wind speed.

and control secondary lagoons (18.5 and 8.6 kg NH_3–N d^{-1} $lagoon^{-1}$, respectively) are reasonable estimates of annual NH_3 emissions and are used for further comparisons. Average annual [NH_4^++NH_3] was higher in the biofuel secondary lagoons (240%; $p < 0.01$), but the pH was lower (0.18 pH units; $p > 0.05$). When these two factors are combined, the [NH_3 (aq)] was calculated to be 74% larger ($p < 0.001$) in the biofuel farm secondary lagoons. When NH_3 emissions from both secondary and primary lagoons are added, total lagoon emissions from the biofuel farms are 90.0 ± 1.8 kg NH_3–N d^{-1} farm-$lagoon^{-1}$, which is 47% larger than the 61.3 ± 3.0 kg NH_3–N d^{-1} farm-$lagoons^{-1}$ from the control farms.

Methane and Carbon Dioxide Emissions

Although NH_3 emissions increased at the biofuel farms relative to the control farms, CH_4 and CO_2 emissions decreased. Because the purposes of the biofuel facility were energy capture (manure conversion to methanol) and the subsequent reduction of CH_4 emissions, the reduction of C emissions was expected. Figure 2 summarizes average total biological gas emissions (total gas collected in the carboys) per farm type. Total biological gas emissions are expressed in kmol to compensate for changes in gas composition and temperature throughout the year. There was a slight difference between farm systems because the biofuel farm biogas emissions were 7% less the first year (2004–2005) and 13% less the second year (2005–2006) ($p > 0.05$). Although these differences were not statistically significant on an annual basis using standard deviations of the average yearly emissions from each farm ($p > 0.05$), a definite trend exists. The question is whether this trend is significant. During the first year of the study when the biofuel system had just been initiated, the biofuel farm biological gas emissions were larger or equal to control

Table 2. Summary of emissions/emission factors.

Lagoon type/gas	Farm type	Per lagoon(s)	Per unit area	Percent feed N	Per kg animal	Percent difference
		Ammonia emissions				
		kg NH_3–N d^{-1} $lagoon^{-1}$	kg NH_3–N d^{-1} ha^{-1}	% NH_3–N vs. feed N	kg NH_3–N yr^{-1} kg $animal^{-1}$	
Primary lagoon	biofuel	71.5	42.3	14.1	0.0267	
Primary lagoon	control†	52.7	31.2	10.4	0.0197	
Primary lagoon	difference	18.8	11.1	3.7	0.0070	35.7
Secondary lagoon	biofuel	18.5	31.4	3.6	0.0069	
Secondary lagoon	control	8.6	14.6	1.7	0.0032	
Secondary lagoon	difference	9.9	16.8	2.0	0.0037	115
		kg NH_3–N d^{-1} farm-$lagoons^{-1}$			kg NH_3–N yr^{-1} kg $animal^{-1}$	
Total lagoon	biofuel	90.0		17.8	0.0259	
Total lagoon	control	61.3		12.1	0.0229	
Total lagoon	difference	28.7		5.7	0.0107	46.8
		Primary lagoon methane emissions				
		kg CH_4–C d^{-1} $lagoon^{-1}$	kg CH_4–C d^{-1} ha^{-1}		kg CH_4–C kg $animal^{-1}$	
First year‡	biofuel	118.6	70.2		0.0443	
First year	control	123.5	73.1		0.0461	
First year	difference	−4.9	−2.9		−0.0018	−4.0
Second year§	biofuel	103.1	61.0		0.0385	
Second year	control	117.1	69.3		0.0437	
Second year	difference	−14.0	−8.3		−0.0052	−12.0
		Carbon dioxide primary lagoon emissions				
		kg CO_2–C d^{-1} $lagoon^{-1}$	kg CO_2–C d^{-1} ha^{-1}		kg CO_2–C yr^{-1} kg $animal^{-1}$	
First year	biofuel	16.6	9.8		0.00620	
First year	control	16.7	9.9		0.00623	
First year	difference	−0.1	−0.1		−0.00004	−0. 6
Second year	biofuel	15.2	9.0		0.00567	
Second year	control	17.2	10.2		0.00638	
Second year	difference	−2.0	−1.2		−0.00075	−12.0
		Dinitrogen primary lagoon emissions				
		kg N_2–N d^{-1} $lagoon^{-1}$	kg N_2–N d^{-1} ha^{-1}	% N_2–N vs. feed N	kg N_2–N yr^{-1} kg $animal^{-1}$	
First year	biofuel	27.5	16.3	5.4	0.0103	
First year	control	32.7	19.3	6.4	0.0122	
First year	difference	−5.2	−3.0	−1.0	−0.0019	−15.9
Second year	biofuel	28.8	17.0	5.7	0.0107	
Second year	control	35.9	21.2	7.1	0.0134	
Second year	difference	−7.1	−4.2	−1.4	−0.0027	−19.8

† Average value of all three control studies.

‡ First year: April 2004 to March 2005.

§ Second year: April 2005 to March 2006.

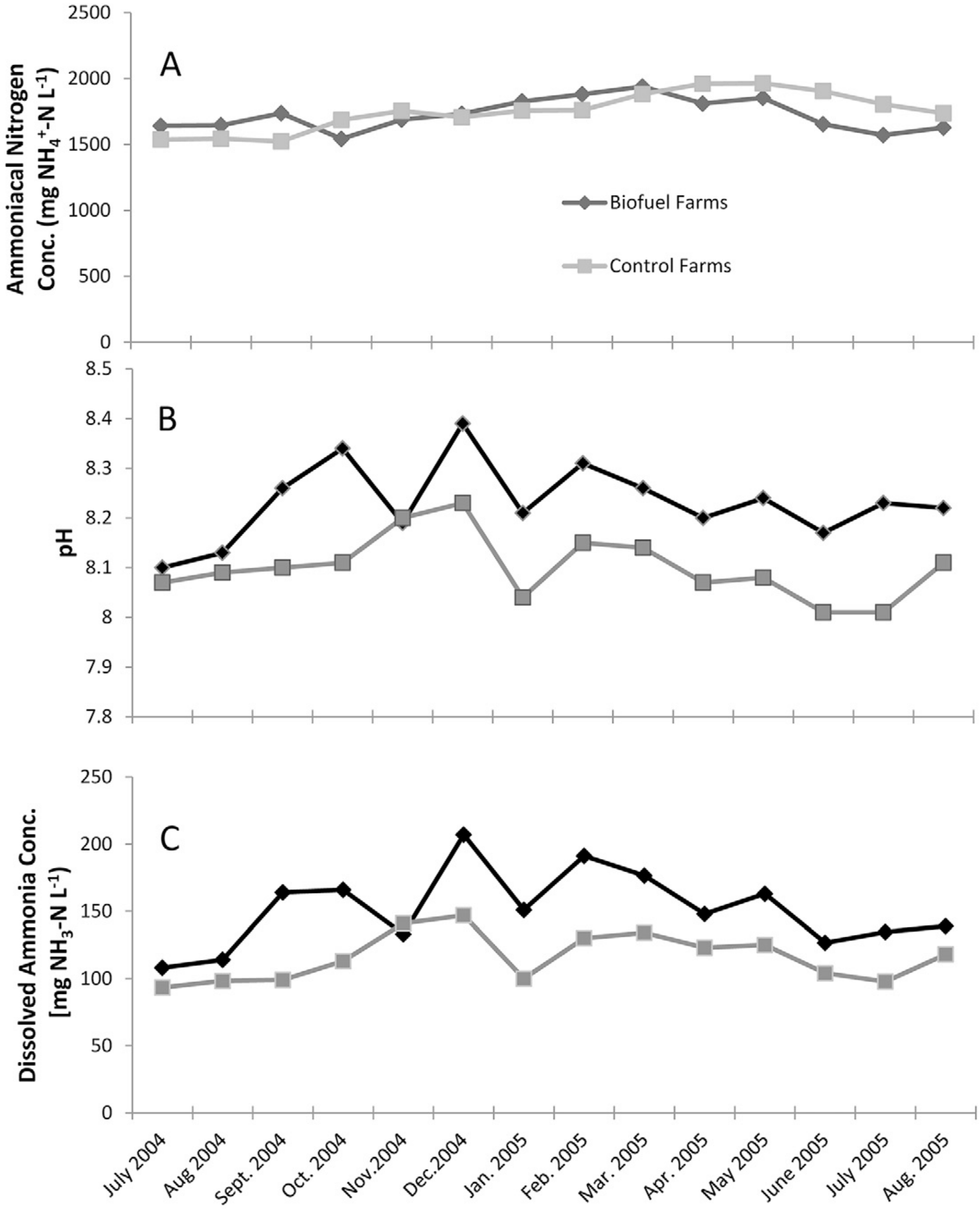

Fig. 1. The effect of concentration (A) and pH (B) of dissolved ammoniacal N on dissolved ammonia of effluent of primary lagoons at depth of 5 to 7 cm. The dissolved ammonia concentration (C) was calculated using the pH and ammoniacal N concentration and the acid dissociation constant for NH_4^+.

emissions during the first 5 mo. During the remaining 7 mo, the biofuel farms emissions were larger only during October. Again, during the second year, biofuel farm biogas emissions were larger only during the months of September and October. When these monthly means are evaluated by the Wilcoxon signed-rank test, the biofuel farms produced less biogas than the control farms after October of 2004 at the 95% confidence level. When the months before October 2004 were included in the analysis, no statistical significance was seen at the same confidence level.

Larger biofuel farm biological gas emissions in September and October may be explained by the following hypothesis where the biofuel facility is constantly returning warmer water, consequently affecting the gas production in the fall. This is important because microbes tend to operate in a very narrow temperature range and bacteria that operate in the lagoons would be a different thermal group of bacteria than those of the biofuel facility. The anaerobic bacteria are divided into three thermal groups: psychrophiles (<20°C), mesophiles (25–40°C), and thermophiles (>40°C) (Conrad et al., 2009; Donoso-Bravo et al., 2009; van Lier et al., 1997). Microbes operating in the biofuel facility have very little activity in colder temperatures where only psychrophilic methanogens would grow. During the summer months temperatures exceed 20°C, and the mesophiles would be expected to have diminished growth rates (van Lier et al., 1997). As the summer ends, both lagoons would cool, and at first mesophilic microbes from the biofuel facility would enhance methanogenesis by increasing the population and diversity of microbes. As the lagoon temperatures further cool, these mesophilic microbes would become dormant and would not affect biogas production. van Lier et al. (1997) found mesophiles to remain active at low metabolic rates even after 8 mo of psychrophilic conditions. It is speculated that there would be a time when the additional C loading of the control lagoons would be offset by the effect of the increased microbe population. The temperature readings at the bottom of the primary lagoons of the biofuel lagoons tended to be warmer in the winter, but the temperatures were not statistically different (p > 0.05). A warming effect from the return of effluent to the biofuel lagoons is not likely due to heat removed in the surface pipes during transmission. Instead, we think an introduction of the mesophilic microbes may have had an effect on gas production.

Gas chromatography was used to measure the percent composition of gases collected from the lagoon sampling carboys to determine the daily composition of each gas emitted. Only CH_4, CO_2, and N_2 were detected. No N_2O was seen in the GC chromatograms; therefore, there were no N_2O emissions. This phenomenon has been seen in earlier work (Harper et al., 2000, 2004) in anaerobic lagoons where little or no N_2O was seen in similar gas collection devices or from emissions observed by tunable diode lasers over the lagoons. Concentrations of N_2O over the lagoons were found (Harper et al., 2000) to be lower than background N_2O levels, indicating that N_2O was consumed from the atmosphere. The data from GC analysis of all of gas samples collected in any 2-wk period was averaged, and this average composition was mathematically fit to a sixth-order polynomial curve. The polynomial fit was used to estimate daily percent composition of each gas for the sampling time period. The percent composition of each gas was calculated from the average GC data of all of the farms because the large variance in the data and the occasionally limited number of samples during some periods prevented calculation of statistically significant individual control and biofuel averages. The validity of pooling the data was tested by curve fitting the gas composition data from the control and biofuel farms independently. The standard error of each fit was used as the 95% confidence interval. Because the confidence intervals overlapped almost completely, we could not say that the gas compositions of the biofuel and control farms were significantly different, and it was proper to combine the data. A sine curve was also fit to the combined data. The polynomial fit could not be used to extrapolate outside of the data points. However, the periodic sine curve takes into account the cyclic nature of the data due to the seasonal variation and

Table 3. Summary of primary biofuel and control farm's lagoon ammonia emissions.

Lagoon	Ammonia emissions				
	Farm #1	Farm #2	Farm #3	Average	SD
	kg NH_3–N d^{-1} lagoon^{-1}†				
2000–2001‡ (control)	51.78	43.05	57.16	50.7	7.1
2002–2003 (control)	54.30	53.37	54.46	54.1	0. 7
	Farm #4	**Farm #5**	**Farm #6**		
2004–2005 (control)	50.73	56.04	53.17	53.3	2.7
2000–2005 (control average)§				52. 7	1.8
Biofuel farms¶	**Farm #1**	**Farm #2**	**Farm #3**		
2004–2005 (biofuel)	72.72	72.07	69.74	71.5	1.6

† Lagoon dimensions are 1.69 ha each primary lagoon. Emissions were calculated from the De Visscher Model (De Visscher et al., 2002) using pH and ammoniacal N concentration, lagoon temperature, and wind speed.

‡ Harper et al. (2006).

§ The average of the three control studies. The SD is calculated as the deviation from this mean.

¶ In 2004 the manure treatment system of Farms 1, 2, and 3 was converted to the biofuel treatment system.

Table 4. Secondary biofuel and control farm's lagoon ammonia emissions from October 2004 to July 2005.

Month	Biofuel lagoons			Control lagoons		
	Farm #1	Farm #2	Farm #3	Farm #4	Farm #5	Farm #6
	kg NH_3–N d^{-1} lagoon^{-1}†					
Oct.	24.4	17.3	20.2	6.3	8.5	7.1
Nov.	5.7	4.4	4.4	3.5	2.3	2.4
Dec.	3.9	3.5	3.9	2.3	1.4	0.8
Jan.	5.6	3.9	5.4	2.5	2.9	2.1
Feb.	8.2	9	11.1	11.2	4.6	2.9
Mar.	9.9	11.5	14.3	10.2	5.8	3.2
Apr.	12.7	17.9	15.3	11.3	9.6	2.4
May	31.7	40.3	39.6	18.5	16.6	4.3
June	32.2	38	31.3	23.1	16.9	5.1
July	41.9	47.1	39.1	22	14.8	32.7
Yearly average	17.6	19.3	18.5	11.1	8.3	6.3
Average		18.46			8.6	
SD		0.84			2.4	

† Lagoon dimensions are 0.59 ha each secondary lagoon. Emissions were calculated from the De Visscher Model (De Visscher et al., 2002) using pH and ammoniacal N concentration, lagoon temperature, and wind speed.

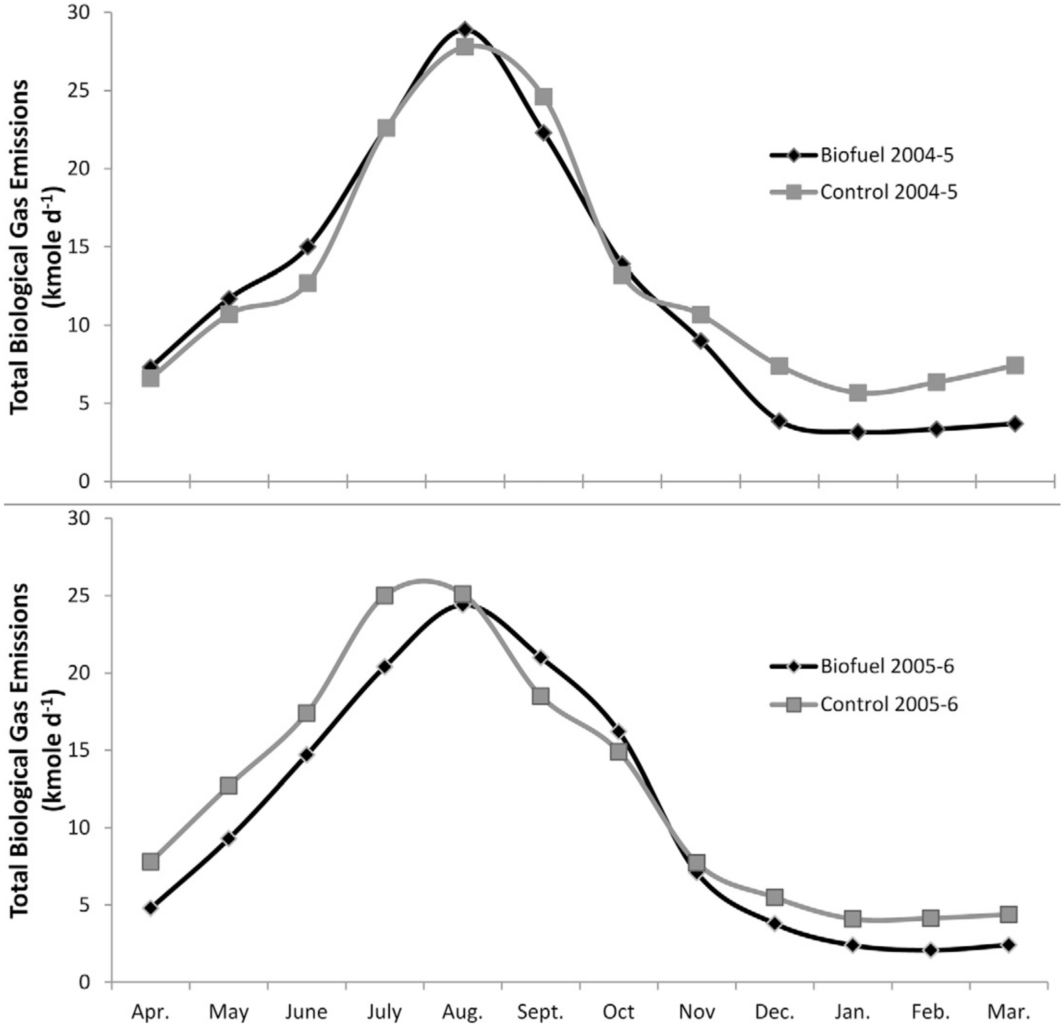

Fig. 2. Total biological gas production of the primary lagoons for each month.

extrapolates the data very well. Therefore, the sine curve was used to estimate gas compositions for the final 40 d of the study. The percent composition for each gas from the fits of the data from all farms was used to represent the daily composition of each gas for subsequent calculations. Figure 3 shows the fit of the average data along with the standard deviation demonstrating the uncertainty of the measurements.

Figure 3 shows a seasonal trend in the gas composition. Maximum CO_2 and CH_4 compositions (as a percent of total gas emissions) occurred in the summer months, whereas maximum N_2 composition occurred in the late winter. There are several possibilities that explain this variation in gas composition. Because the sampling interval of the carboys was much longer in the winter before collection for GC analysis, several processes must be considered. First, there is the possibility that the CH_4 and CO_2 in the carboys and not N_2 are being dissolved back into the effluent. This is highly unlikely because these lagoon waters are saturated with the gases from the constant emission of gases from the bottom. Second, there is a possibility that the gases that are saturated in the effluent from the atmospheric partial pressure, namely N_2, will transfer into the carboys. This is also not likely because carboys filled with helium did not show appreciable contamination with atmospheric N_2 in earlier experiments (Harper et al., 2004) and gas-solution calculations showed insignificant N_2 transport from the gas in solution to gas in the carboys (Harper, 2005; A. De Visscher and L.A. Harper, unpublished data). The individual steps of anaerobic microbial digestion involve different microbes that have been shown to be affected differently by temperature (Conrad et al., 2009; Fey and Conrad, 2000). We infer that if NH_4^+ conversion to N_2 occurs by another chemical or biological process that is affected by temperature differently, the percent composition of gases would not remain the same throughout the year. For example, if the NH_4^+ conversion to N_2 takes place via pychrophilic organisms, the production of N_2 would not coincide with the mesophilic methanogenesis that occurs in the summer. Instead, methanogenesis would tend to produce more CO_2 and CH_4 via mesophilic methanogens relative to N_2 in the summer and less in the winter.

The monthly gas emissions (Table 5) were calculated for each farm using the total biological gas emission of each farm type multiplied by the average fractional composition of each gas of all farms. The monthly average of each farm was then used to calculate the monthly biofuel and control average emissions. Monthly standard deviations ranged from 7 to 52% of the monthly means (data not shown).

Examination of CH_4 emissions shows important trends (Table 5), where CH_4 was generally lower. Methane emissions were lower the second year for both types of farms. Average CH_4

emissions from the biofuel farms were 118.6 ± 1.9 and 103.1 ± 19.7 kg CH_4–C d^{-1} $lagoon^{-1}$ for the first and second years, respectively. Average emissions from the control farms were 123.5 ± 11.4 for the first year and 117.1 ± 11.1 kg CH_4–C d^{-1} $lagoon^{-1}$ for the second year. During both years, the control emissions were higher than the biofuel farms, but in each case, the differences were not significant ($p > 0.05$). Although the difference in CH_4 emissions was not significant on an annual basis, the average monthly biofuel farms emissions were consistently lower after September of the first year. Monthly biofuel average emissions were generally lower after October of the first year (September and October of the second year were exceptions to this trend). Evaluation by the Wilcoxon signed-rank test showed that the biofuel farms produced less CH_4 than the control farms after October 2004 at the 95% confidence level.

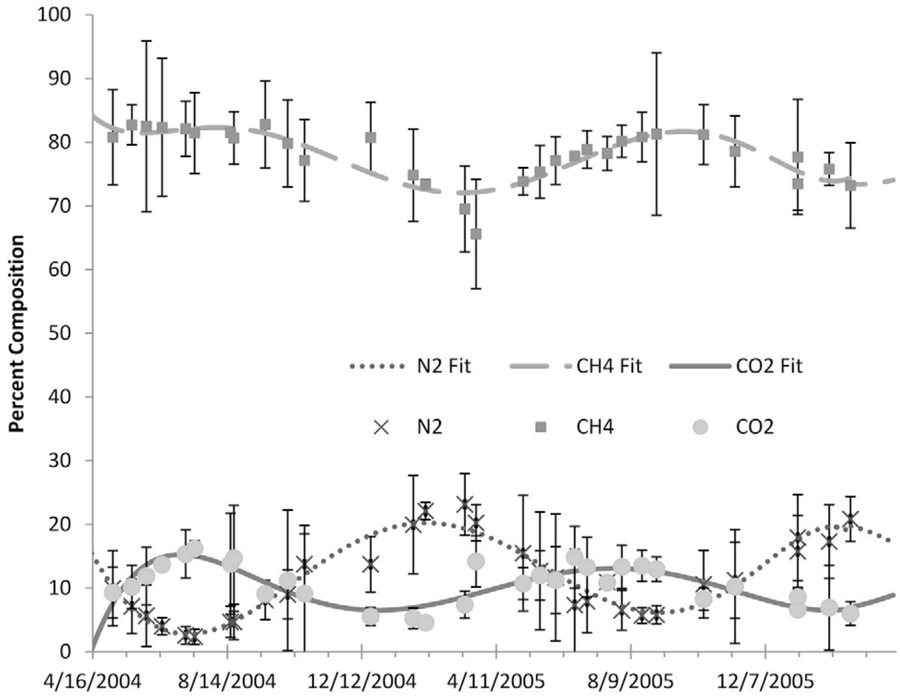

Fig. 3. Comparison of biological gas composition. Markers indicate average composition of gas samples collected by SUMMA canister and analyzed by gas chromatography for a given 2-wk period. Error bars are SDs of these averages. The curves are the best fit of the data.

This difference in CH_4 emissions between farm systems is best explained by an overall decrease in biological gas production (less decomposition and methanogenesis), which has been previously discussed. As total biogas production decreases, each individual gas within the biological gas decreases. The biofuel farm CO_2 emissions (Table 5) were also found to be lower than the control farms, but, similar to CH_4, there was no difference between farm systems or between yearly averages for the first and second years of the study ($p > 0.05$). Average CO_2 emissions from the biofuel farms were 16.6 ± 0.1 the first year and 15.2 ± 2.9 kg CO_2–C d^{-1} $lagoon^{-1}$ the second year, and for the control farms the averages were 16.7 ± 1.9 and 17.2 ± 1.5 kg CO_2–C d^{-1} $lagoon^{-1}$ for the first and second years of the study, respectively. The same month-to-month statistically significant trend was seen for CO_2 and CH_4 emissions: Monthly biofuel CO_2 emissions were lower after October of first year with the exception of September and October of the second year. As was the case with biogas, when

Table 5. Carbon dioxide and methane emissions of primary lagoons.

Month	Methane emissions				Carbon dioxide emissions			
	Biofuel		Control		Biofuel		Control	
	Year 1†	Year 2‡	Year 1	Year 2	Year 1	Year 2	Year 1	Year 2
	kg CH_4–C d^{-1} $lagoon^{-1}$§				kg CO_2–C d^{-1} $lagoon^{-1}$			
Apr.	73.6	42.6	66.7	69.3	3.1	6.1	2.0	9.8
May	115.2	83.5	102.8	113.5	15.4	13.0	12.4	17.4
June	148.7	136.5	123.2	160.7	26.5	22.4	21.7	26.2
July	224.6	194.1	215.4	237.7	40.7	32.0	39.3	39.3
Aug.	288.5	239.3	255.3	245.2	46.8	38.2	41.9	39.4
Sept.	220.3	208.1	243.2	183.0	29.9	30.8	33.7	27.4
Oct.	134.7	160.5	128.3	148.0	14.9	21.4	14.6	20.0
Nov.	85.0	68.9	101.7	75.1	8.1	8.1	9.8	9.0
Dec.	35.5	35.2	68.0	51.5	3.1	3.5	6.0	5.2
Jan.	28.2	21.5	50.8	37.7	2.6	1.9	4.6	3.4
Feb.	29.4	18.6	55.9	37.2	3.1	1.7	5.7	3.4
Mar.	32.3	21.6	64.9	39.2	3.9	2.4	7.8	4.2
Average	118.6	103.1	123.5	117.1	16.6	15.2	16.7	17.2
SD¶	1.9	19.7	11.4	11.1	0.1	2.9	1.9	1.5

† Year 1: April 2004 to March 2005.

‡ Year 2: April 2005 to March 2006.

§ The lagoon size of each lagoon in the study is 1.69 ha. The emission of each gas was calculated from the measured daily average of biogas collected from SUMMA canisters and the daily percent composition of each gas measured from gas chromatography.

¶ The SD from the mean of the annual emissions of the three farms from each farm type.

the trend in the monthly means was evaluated by the Wilcoxon signed-rank test, the biofuel farms produced less CO_2 than the control farms after October of the first year at the 95% confidence level. When the months before October of the first year were included in the analysis, no statistical significance was seen at the same confidence level. Biofuel CO_2 emissions decreased in the biofuel system as the study progressed.

The trend in CH_4 and CO_2 leads us to ask whether the biofuel conversion process stabilized such that the study characterized the emissions of a normally operating biofuel/methanol facility. We believe that the change in the production of biogas beginning in September or October of the first year is an excellent indication that the system was ending in its startup phase. The fact that the biofuel farms produced more gas the first 5 mo and then produced less thereafter leads us to believe that the emission trends seen in the 2 yr of the study would not have reversed but instead would have maintained or increased with time.

Reduction of the individual biogas emissions is attributed to the reduction of C loading in the primary lagoons. A general annual decrease in available organic matter (sludge layer) from that accumulated at the bottom of the lagoons during winter months was observed (Weaver and Harper, unpublished data) as methanogenesis increased each summer due to higher lagoon temperatures. Measurements of organic matter between systems may be difficult because these differences are expected to be less than those observed between winter and summer sludge levels. Because earlier attempts to quantify C loading and the sludge layer were very difficult due to the large variability in the data, these measurements were not made in this study. However, it is not unreasonable to expect a similar phenomenon to occur in the biofuel facility. In this facility, the operating temperature (35°C) is at least 11 to 18°C more than measured temperatures in the bottom of the lagoons. Therefore, because methanogenesis increases with temperature (Hill et al., 2001), the net (biofuel facility + lagoon) methanogenesis should be higher than that observed in the control farm lagoons. We hypothesize that this would result in a decrease in the amount of organic matter entering the lagoons, thus creating a new baseline concentration of organic matter in the lagoon. During the first year, only a small decrease in CH_4 production was observed due to the residual manure decomposition. This process predicts the amount of manure stored in the bottom of the lagoons to decrease and lower CH_4 production. Carbon dioxide emissions result from the same microbial processes, and, likewise, CO_2 emissions decreased due to depletion of residual C in the lagoons. The decrease in NH_4^+ conversion to N_2 agrees with earlier studies where a decrease in methanogenesis resulted in less NH_4^+ conversion to N_2 (Harper et al., 2000).

Dinitrogen Gas Emissions

Another gas found in the carboys was N_2 gas (Table 6). Dinitrogen gas emissions from the biofuel farms were 27.5 ± 0.9 and 28.8 ± 6.8 kg N_2–N d^{-1} $lagoon^{-1}$ the first and second years and for the control farms were 32.7 ± 2.2 and 35.0 ± 2.8 kg N_2–N d^{-1} $lagoon^{-1}$ for the first and second years, respectively. Dinitrogen emissions were significantly lower at the biofuel farms for the first (7.1 kg N_2–N d^{-1} $lagoon^{-1}$; 16%; $p < 0.05$) and second years (6.0 kg N_2–N d^{-1} $lagoon^{-1}$; 20%; $p < 0.01$). The trend that had been seen for CH_4, CO_2, and total biogas production was seen where monthly N_2 production was lower in the biofuel lagoons after October of the first year (except for September and October of the second year). When the monthly means are evaluated by the Wilcoxon signed-rank test, the biofuel farms produced less N_2 than the control farms after October of the first year at the 95% confidence level. When the months before October of the first year (when CO_2 production was greater) were included in the analysis, no statistical significance was seen at the same confidence level.

Comparison to a Concurrent Study

Individual biological gas species' emissions (CH_4 and CO_2) decreased throughout the course of the study. Methane emissions were reduced 4.0% ($p > 0.05$) the first year and 12.0% ($p > 0.05$) the second year due to diminishing available organic matter in the lagoons for decomposition (removed for biofuel production). Likewise, CO_2 emissions were reduced 0.6% ($p > 0.05$) and 11.6% ($p < 0.05$) the first and second years of the study, respectively. The reduction in N_2 gas emissions in the biofuel lagoons was associated with reduced methanogenesis. It is possible that the thermodynamics of the system and Gibbs free energy for NH_4^+ conversion to N_2 (Van Cleemput, 1972) suggest that spontaneous conversion of NH_4^+ to N_2 may occur. With reduced NH_4^+ conversion to N_2, NH_3 emissions would be expected to increase. During the same period, primary lagoon NH_3 emissions increased 36% ($p < 0.001$), whereas N_2 production during the first and second years of the study decreased 16% ($p < 0.05$) and 20% ($p > 0.05$).

The decrease in CH_4 emissions and the increase in NH_3 emissions agrees with a concurrent study at the same location when NH_3 and CH_4 emissions were measured intensely during brief periods of high emissions (summer) and low emissions (winter). In this concurrent study (Harper et al., 2010), whole-farm (including barns and lagoons) emissions were measured

Table 6. Monthly primary lagoon dinitrogen biological gas emissions.

Month	Biofuel		Control	
	Year 1†	Year 2‡	Year 1	Year 2
	kg N_2–N d^{-1} $lagoon^{-1}$§			
Apr.	27.3	22.3	25.9	37.1
May	24.4	36.5	24.2	51.3
June	16.3	45.8	14.2	56.1
July	19.0	47.2	18.0	60.1
Aug.	36.7	45.5	31.2	48.1
Sept.	47.4	37.9	49.9	33.2
Oct.	44.6	36.1	40.5	32.5
Nov.	37.7	21.2	43.9	22.2
Dec.	19.6	15.9	36.6	22.7
Jan.	17.7	12.5	31.5	21.5
Feb.	19.1	11.4	36.2	22.8
Mar.	20.0	12.5	40.5	22.7
Average	27.5	28.8	32.7	35.9
SD¶	0.9	6.8	2.2	2.8

† Year 1: April 2004 to March 2005.

‡ Year 2: April 2005 to March 2006.

§ The lagoon size of each lagoon in the study is 1.69 ha. The emission of N_2 gas was calculated from the measured daily average of biogas collected from Summa canisters and the daily percent composition of each gas measured from gas chromatography.

¶ The SD from the mean of the annual emissions of the three farms from each farm type.

by the inverse Lagrangian stochastic analysis (bLS) technique (Flesch et al., 2004, 2005, 2007; McGinn et al., 2006; Harper et al., 2009). In the study using the bLS technique, the whole-farm CH_4 emissions decreased 47% (830 kg CH_4–C farm^{-1} d^{-1}) compared with control farms, whereas NH_3 emissions increased by 46% (144 kg NH_3–N d^{-1} farm^{-1}, 28.4% of feed N). In our study, CH_4 primary lagoon emissions decreased 12%, and NH_3 primary plus secondary lagoon emissions increased 47%.

To better compare differences in emission measurements between the two studies, emissions were calculated using the De Visscher model for the same brief time periods (winter and summer) of the bLS study for the secondary and primary lagoons. The winter and summer emissions were then averaged to calculate a yearly emissions rate for the lagoons as was done in the bLS study. Using this calculation procedure, the biofuel farms' total NH_3 emissions from the primary and secondary lagoons were 58% larger than those from the control farms (97.7 vs. 62.1 kg N-NH_3 d^{-1} farm-lagoons^{-1}). Similarly, larger emissions' differences were also seen for CH_4. Biofuel CH_4 emissions measured by the carboy method during the bLS time period were 23.3% less (95.8 vs. 124.9 kg CH_4–C lagoon^{-1} d^{-1}) than control lagoon emissions. This is a considerably larger difference compared with 12% smaller CH_4 emissions determined for the second year of the study. As differences in emissions were larger during the bLS study period due to the limited measuring time period, care should be used when representing annual emissions with short-term studies.

Although the NH_3 emissions' relative difference between farm systems is similar between bLS and the De Visscher methods during the bLS study period (46 and 58%, respectively), carboy-measured CH_4 emissions' relative decrease (23%) was less than half of that (47%) measured by the bLS method. The comparison of differences is compounded by the fact that the bLS determines emissions from both the lagoons plus barns combined (whole-farm measurement). Although bLS emission measurements for CH_4 and NH_3 are considerably larger than corresponding values measured by the carboy and De Visscher methods, both studies demonstrate an observed decrease in CH_4 emissions corresponding with an increase in NH_3 emissions.

The Effect of pH on Lagoon Emissions

Along with producing energy from manure, normally a waste commodity, a decrease in global warming gas emissions was one of the original intentions for biofuel production. There was a decrease in total CH_4 emissions, which resulted from reduction in lagoon C loading in the farm lagoons due to diverted organic matter being converted to biofuel. The relationship between NH_4^+ conversion to N_2 and methanogenesis predicted that a decrease in methanogenesis (smaller CH_4 and CO_2 emissions) would result in a decrease in the amount of [NH_4^++NH_3] converted to N_2 and an increase in [NH_4^++NH_3], causing a potential for NH_3 emissions. However, the decrease of 6.2 kg N_2–N d^{-1} lagoon^{-1} emissions from [NH_4^++NH_3] conversion is much smaller than the increase of 18.8 kg NH_3–N d^{-1} lagoon^{-1} emissions and cannot be explained simply by a decrease in denitrification.

The difference between the relative increase in denitrification (emission of N_2) and increase in NH_3 emissions can be explained by the change in pH observed in the lagoons. Ammonia emissions are controlled by four factors: effluent temperature, wind speed (turbulence at the solution/air interface), effluent pH, and [NH_4^++NH_3] (Harper, 2005). Any process that affects these four factors will affect NH_3 emissions. Dissolved ammoniacal N exists in two forms: the nonionic form (NH_3), which is volatile, and the ionic form (NH_4^+), which is not volatile. The ratio of these two forms is controlled by the relationship [NH_3]/[NH_4^+] = $10^{(pH-pKa)}$. Therefore, a small increase in pH greatly increases the NH_3/NH_4^+ratio at the pH range observed in the lagoons. As the concentration of dissolved NH_3 increases, the partial pressure of NH_3 increases according to Henry's Law, thus increasing NH_3 emissions.

The importance of pH is demonstrated in Fig. 1B, with lagoon pH being higher in the biofuel lagoons. Examination of the data in Fig. 1A shows that the N concentrations are very similar, suggesting no difference in NH_3 emissions; however, slight differences in pH affect NH_3 emissions considerably more than differences in [NH_4^++NH_3]. For example, a 0.3 pH unit difference in two lagoons with the same ammoniacal N concentration would result in a nearly a twofold difference in aqueous [NH_3] and subsequently a twofold difference in NH_3 emissions from the more alkaline lagoon. To have the equivalent effect, the [NH_4^++NH_3] would have to double. Thus, a small change in pH can have a large effect on emissions.

Although methanogenesis rates control CO_2 and CH_4 emissions and influence conversion to N_2, methanogenesis only indirectly affects NH_3 emissions by affecting the lagoon solution pH. Methanogenic microbial communities consist of various guilds of microorganisms that perform various functions (Conrad et al., 2009; McInerney and Bryant, 1981; Zehnder, 1978; Zinder, 1993). Many products of methanogenesis are acidic. The first step in the microbial degradation of manure involves conversion of complex organic matter to fatty acids, alcohols, and CO_2 (also acidic). The fatty acids and alcohols are converted to acetic acid, which is converted to CO_2 and CH_4 (Abbanat et al., 1989). These products would interact with the buffering system of the lagoons and lower the pH. Conversely, a decrease in methanogenesis would therefore result in increased solution pH. Because all the lagoons in this study were similar due to identical feeding operations, the buffer capacities, which were not measured, should be equal, and pH effects would be due to changes in methanogenesis rather than differences in buffer capacity between treatment systems. The effect of methanogenesis on pH has already been seen in another study, where pH and temperature depth profile measurements of the lagoons were conducted (Weaver and Harper, unpublished data). The pH was considerably lower (>0.3 pH units) at the bottom (deepest 1–2 m) of the lagoons where the manure solids accumulate as sludge and methanogenesis is largest. However, all of the bioreactors' methanogenesis acidic products, except for CO_2, are returned to the primary lagoons. Furthermore, the primary lagoons are saturated with CO_2, so, whether the bioreactors are present or not, most CO_2 produced in the lagoons is released into the atmosphere. Therefore, whether the methanogenesis occurs in the bioreactor or the primary lagoons, the pH effect should be same. Could another process also be affecting pH?

Although other factors may exist in the biofuel process that increase pH, a decrease in NH_4^+ conversion to N_2 could also affect pH as well as increase [NH_4^++NH_3]. Harper et al. (2004) theoretically demonstrated that NH_4^+ conversion to N_2, under low dissolved oxygen conditions (partial pressure O_2 = 10 kPa to 1 × 10^{-13} kPa), is thermodynamically ($\Delta G \approx -1249$ to

−992 kJ mol^{-1}) favorable for the following reaction under typical lagoon conditions where dissolved oxygen concentrations are too low to be measured by a dissolved oxygen meter:

$$4(1-a)NH_3\,(aq) + 4aNH_3(aq) + 3O_2(g) \rightarrow 2N_2(g) + 6H_2O(l) + 4(1-a)\,H^+\,(aq) \qquad [1]$$

According to this proposed reaction, if NH_4^+ is oxidized, H^+ is produced, thereby lowering the pH. If NH_3 is oxidized, the acid base equilibrium of NH_4^+ is shifted to produce more NH_3, which produces more H^+.

$$NH_4^+(aq) \leftrightarrow NH_3(aq) + H^+\,(aq) \qquad [2]$$

Consequently, a decrease in conversion to N_2 by the proposed mechanism would tend to increase the pH. Additionally, because N_2 production only occurs in the presence of methanogenesis (Harper et al., 2000), a decrease in methanogenesis would increase the [NH_4^++NH_3]. Therefore, a decrease in NH_4^+ conversion to N_2 would not only increase NH_3 emissions by producing more available NH_4^+(aq) and NH_3(aq), but, more importantly, it would raise the pH, increasing the [NH_3(aq)] and thus the NH_3 emissions (Fig. 1).

The implementation of the biofuel system demonstrated immediate small decreases in CO_2 and CH_4 lagoon emissions after the first year of the study. These C emissions were decreased even further (11–12%) after the second year of the study due to the diversion of manure for conversion to biofuel. The implementation of the biofuel system also resulted in a decrease in N_2 gas emissions, which we attribute to a decrease in the conversion of NH_4^+ to N_2. The lagoon solution pH also increased corresponding to the decrease in conversion to N_2 and the decrease in methanogenesis. This pH increase resulted in an increase (47%) in lagoon NH_3 emissions. Further studies are necessary to determine if the effect is seen at other biofuel facilities at different locations and using different systems. This study shows the unintended consequences of how an effort to reduce one emissions' process (greenhouse gas emissions) affects the amount of another process (air quality gas emissions). This should be considered when making changes to any management system because the benefits of any biofuel system should be weighed against any detriment that may occur.

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ATTACHMENT 54

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Quarterly Progress Report
April to June 2019

Submitted via email on July 30, 2019

In accordance with Appendix H of the Consent Decree, Premium Standard Farms, LLC, Murphy-Brown of Missouri LLC and KC2 Realty, hereafter collectively referred to as "PSFMBM" or "the Companies", provide this Quarterly Progress Report for the Second Quarter of 2019. Please be made aware that Premium Standard Farms has changed its name to Murphy-Brown of Missouri LLC.

Under the Consent Decree, each quarterly progress report shall contain the following information: (1) a brief description of the actions taken by the Defendants towards achieving compliance with this Consent Decree during the reporting period, including actions related to compliance with the requirements set forth in Appendices A through G; (2) a summary of the results of any testing conducted pursuant to Appendices B, C, and F during the reporting period; (3) an identification of all instances of noncompliance with the performance standards set forth in Appendix D and any other failures known to the Defendants to comply with the requirements of this Consent Decree during the reporting period, the reasons for such failures to comply, and actions already taken or planned to be taken to correct such failures to comply; and (4) a brief description of the actions that the Defendants anticipate taking towards achieving compliance with this Consent Decree during the next quarter, including any possible delays or other problems that may affect compliance with the Consent Decree and the Defendants' anticipated actions to resolve such delays or problems.

I. Actions Toward Achieving Compliance with Consent Decree Including Appendices

This section will provide a brief description of the actions taken by the Companies towards achieving compliance with this Consent Decree during the reporting period, including actions related to compliance with the requirements set forth in Appendices A through G. This section will also provide a summary of the results of any testing conducted pursuant to Appendices B, C, and F during the reporting period.

A. Appendix A – Technology Alternatives

Somerset, Locust Ridge and Hedgewood Projects

All required technologies have been installed and are operational at the Locust Ridge and Somerset farms, including the AND system and scraper systems at all required barn sites. Both farms have been repopulated and are currently operating as Class IA farms on a daily basis.

Quarterly Progress Report
April to June 2019

The AND system has not been installed on the Hedgewood farm and it remains a Class IB operation. This farm has and continues to maintain compliance with the 50 percent nitrogen reduction standard.

The total storage volume available at Hedgewood on June 28, 2019 was 10,969,889 gallons.

During this quarter, we land applied 3,073,651 gallons of effluent at the Hedgewood farm.

During this quarter, the AND systems at Somerset and Locust Ridge returned to normal operation.

The total storage volume available at Somerset on June 28, 2019 was 36,210,176 gallons. During this quarter, we land applied 22,719,260 gallons of effluent at the Somerset farm.

The total storage volume available at Locust Ridge on June 28, 2019 was 26,820,828 gallons. During this quarter, we land applied 1,557,256 gallons of effluent at the Locust Ridge farm.

Green Hills and South Meadows Projects

During this quarter, the AND systems returned to normal operation.

The total storage volume available at Green Hills on June 28, 2019 was 29,540,978 gallons. During this quarter, we land applied 7,245,900 gallons of effluent at the Green Hills farm.

The scraper systems at Green Hills were installed and operational on the following dates: Green Hills 10, October 10, 2011; Green Hills 11, November 1, 2011.

The total storage volume available at South Meadows on June 28, 2019 was 20,778,203 gallons. During this quarter, we land applied 7,823,727 gallons of effluent at the South Meadows farm.

All primary lagoons at South Meadows have impermeable covers installed.

The scraper systems at South Meadows were installed and operational at all sites in March, 2011.

During this quarter, the lagoon cover at South Meadows 3 experienced damage. A majority of the cover is still in working order, but we plan replace the cover in the upcoming months.

Homan and Ruckman Projects

During this quarter, the AND systems returned to normal operation.

The total storage volume available at Homan on June 28, 2019 was 67,628,126 gallons. During this quarter, we land applied 20,311,486 gallons of effluent at the Homan farm.

Quarterly Progress Report
April to June 2019

The scraper systems at Homan were installed and operational on the following dates: Homan 17, February, 11, 2011; Homan 18, February 2, 2011; Homan 19, June 28, 2011; Homan 20, January 28, 2011; Homan 21, May 20, 2011; Homan 22, June 3, 2011 and Homan 23, February 18, 2011.

The total storage volume available at Ruckman on June 28, 2019 was 60,916,781 gallons. During this quarter, we land applied 20,414,231 gallons of effluent at the Ruckman farm.

During the second quarter of 2017, a storm destroyed the lagoon cover at site 11. Installation of the new cover at site 11 was completed during the second quarter of 2018.

The scraper systems at Ruckman were installed and operational in all 70 barns in April, 2011.

Badger/Wolf / Brantley Project

During this quarter, the AND systems returned to normal operation.

The total storage volume available at Badger/Wolf/Brantley on June 28, 2019 was 43,650,154 gallons. During this quarter, we land applied 36,567,076 gallons of effluent at the Badger/Wolf/Brantley farm.

Terre Haute Project

During this quarter, the AND systems returned to normal operation.

The total storage volume available at Terre Haute on June 28, 2019 was 37,090,262 gallons. During this quarter, we land applied 49,263,999 gallons of effluent at the Terre Haute farm.

Scraper installation at Terre Haute was completed on December 24, 2011. This consisted of scrapers being installed in all farrowing, breeding and gestation barns for sites 1, 2, 5, 6, 7, 11, 13, 14 and all barns at Cooley Mega and Terre Haute Mega.

Valley View Project

During this quarter, the AND systems returned to normal operation.

All primary lagoons have impermeable covers installed.

The total storage volume available at Valley View on June 28, 2019 was 58,436,447 gallons. During this quarter, we land applied 16,385,545 gallons of effluent at the Valley View farm.

Whitetail Project

During this quarter, the AND systems returned to normal operation.

The total storage volume available at Whitetail on June 28, 2019 was 23,216,329 gallons. During this quarter, we land applied 15,772,471 gallons of effluent at the Whitetail farm.

All the Whitetail sites (72 barns) were installed with scraper systems by December 31, 2010.

In an effort to minimize the impact of storm water on our irrigation basins and the AND system at Whitetail, PSFMBM began the installation of a high density polyethylene cover on site 1 lagoon in late 2010. The cover installation process was completed during the first quarter of 2011. This system has a storm water collection system in place as well as a gas collection system. During the fall of 2011, PSFMBM began the installation of an additional impermeable cover on the Whitetail site 5 lagoon. The cover installation was completed during the third quarter of 2012.

B. Appendix B – Lagoon Integrity Testing

On March 5, 2007, EPA sent a letter to PSFMBM requesting quarterly monitoring for a period of one year at monitoring wells near the Terre Haute 11 and Ruckman 15 lagoons. A report summarizing data from the first two quarters of sampling was submitted to EPA on October 19, 2007. The fourth quarter of sampling was completed during December 2007. A summary report on all four quarters has been sent to EPA. PSFMBM received a letter from EPA on January 19, 2010 requesting additional testing of monitoring wells. The test wells were drilled at our Scott/Colby and Whitetail farms. Sampling at the farms has been completed and we submitted the final report for the Scott/Colby farm to EPA and CLEAN during the first quarter of 2012. EPA responded with an approval letter for the Scott/Colby report in the second quarter of 2012. The report for the Whitetail lagoon integrity testing results was submitted during the third quarter of 2012. In late 2013, PSFMBM received a letter from EPA that approved the Whitetail lagoon integrity test results.

C. Appendix C – Testing Criteria for Technology Alternatives

All data indicates the Whitetail, Terre Haute, Badger/Wolf/Brantley, Homan, Ruckman, Green Hills, South Meadows, Valley View, Somerset and Locust Ridge AND systems are operating as designed and meeting the treatment goals.

Appendix D – Performance Standards

During this quarter, the Badger/Wolf, Whitetail, Terre Haute, Homan, Ruckman, Green Hills, South Meadows, Valley View, Somerset and Locust Ridge AND systems returned to normal operation.

D. Appendix E – Best Management Practices

During this quarter, PSFMBM had no releases that required reporting to MDNR.

E. Appendix F – Air Emissions Monitoring

On November 17, 2004, PSFMBM mailed to EPA the *Air Emissions Monitoring Completion Report* pursuant to Appendix H, paragraph II.A.(8) of the Consent Decree. PSFMBM revised the Report in response to EPA's September 2, 2011 comments and submitted the revised report to EPA on September 26, 2011. EPA subsequently approved the report.

During the fourth quarter of 2012, PSFMBM submitted the post-technology monitoring report dated October 18, 2012 for the Whitetail Farm to EPA for review; that report was entitled *Post-technology Emissions Measures of a Covered Anaerobic Lagoon and Nitrification/Denitrification Treatment Cells at the Whitetail Farm* (Whitetail Report). EPA disapproved the report by letter dated December 16, 2013. PSFMBM's January 31, 2014 response explained that the disapproval was based on incomplete information (which was provided in the response) and expressed a desire to resolve the disapproval without invoking dispute resolution. Since then, PSFMBM, EPA and CLEAN have been engaged in discussions to resolve this and other issues through amendments to the Consent Decree. In October, 2016, Murphy-Brown of Missouri, LLC, as successor-in-interest, filed a notice of dispute with the federal court (WD Mo.) pursuant to Section XV (Dispute Resolution) of the Consent Decree. That notice of dispute concerns the requirement in Subsection II.A of Appendix F to the Consent Decree and EPA's December 16, 2013 disapproval of the Whitetail Report. At the request of the United States, MBM agreed to extend the dispute resolution informal negotiation period to February 21, 2017. PSFMBM received EPA's response letter on February 21, 2017.

Although not required by the Consent Decree, PSFMBM submitted to EPA a Quality Assurance Project Plan (QAPP) for calculating emissions from the Hedgewood, Locust Ridge and Somerset Farms during the fourth quarter of 2012 in the mistaken belief that EPA wished to see emissions calculations from these down-sized farms. After EPA disapproved the QAPP by letter dated December 16, 2013, however, PSFMBM withdrew the QAPP by letter dated January 31, 2014. PSFMBM has decided to replace the CPF system at the Valley View Farm with the AND system, and therefore, will not be conducting post-technology emissions monitoring on the CPF system.

F. Appendix G – Supplemental Environmental Projects

On December 15, 2003, PSFMBM and ContiGroup Companies submitted via electronic mail to EPA the final *Report on Evaluation of Oil Sprinkling SEP* pursuant to Appendix H, paragraph II.A.(5) of the Consent Decree. There were no Supplemental Environmental Project-related activities during this reporting period.

II. Noncompliance with Performance Standards and Other Requirements of Decree

This section will identify all instances of noncompliance with the performance standards set forth in Appendix D and any other failures known to the Defendants to comply with the requirements of this Consent Decree during the reporting period, the reasons for such failures to comply, and actions already taken or planned to be taken to correct such failures to comply.

PSFMBM is not aware of any failures to comply with the requirements of the Consent Decree during this reporting period.

III. Planned Actions to Comply With Decree and Potential Delays/Problems

This section provides a brief description of the actions that PSFMBM anticipates taking towards achieving compliance with this Consent Decree during the next quarter, including any possible delays or other problems that may affect compliance with the Consent Decree and PSFMBM's anticipated actions to resolve such delays or problems.

At this time, PSFMBM does not anticipate any delays or other issues that may affect compliance with the Consent Decree.

CERTIFICATION STATEMENT

To the best of my knowledge, after thorough investigation, I certify that the information contained in or accompanying this submission is true, accurate and complete. I am aware that there are significant penalties for submitting false information, including the possibility of fine and imprisonment for knowing violations.

Blake Boxley
Director Environment, Health and Safety
Project Coordinator

Dated: 7/30/19

ATTACHMENT 55

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Coupling Nitrogen Removal and Anaerobic Digestion for Energy Recovery from Swine Waste Through Nitrification/Denitrification

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Abstract

As a major swine-producing state, North Carolina (United States) has adopted incentives for energy recovery from swine waste and environmental performance standards for new swine facilities. However, there are no treatment performance requirements for existing farms; therefore, management of swine waste in open lagoons with spray irrigation for disposal is nearly universal in North Carolina. Emissions of nitrogen to the atmosphere and the landscape from large industrial swine farms have led to concerns over the impact on environmental quality and human health. Accordingly, there has been increasing interest in developing alternate treatment methods for swine waste, including methods that allow for energy recovery. To evaluate the technical feasibility and limitations of coupling biological nitrogen removal with anaerobic digestion of swine waste for energy recovery, we operated a pilot nitrification/denitrification system at an 8,000-head finishing farm already practicing full-scale anaerobic digestion in covered lagoons with methane capture. Of primary interest was the extent to which alkalinity and biodegradable chemical oxygen demand (COD) in the digested waste could meet the stoichiometric requirements for oxidation of ammonium-N (AN) and denitrification, respectively. The system removed 98% of the influent AN and 83% of influent total nitrogen. Approximately 75% of influent total COD was oxidized, mostly as electron equivalents for denitrification. Alkalinity in digested waste may not meet the alkalinity demand from nitrification, depending on the extent of denitrification. Stripping of ammonia into the gas phase was negligible, but 8.2% of the ammonium-N removed was converted to nitrous oxide-N.

Key words: ammonia; nitrification; nitrogen removal; swine waste

Introduction

Anthropogenic inputs of fixed nitrogen to the environment, particularly from intensive agriculture, have led to substantial pollution of both air and water globally (Schlesinger, 2009). In the United States, North Carolina is currently home to 8.5 million swine (NCDACS, 2013), most of which are raised in industrial-scale facilities. Waste from swine farms in North Carolina is typically stored in large uncovered lagoons and periodically applied to sprayfields to fertilize crops (NCDENR). The scale of waste generation results in more nitrogen than can be assimilated at agronomic rates in the entire region of swine production (NCDENR).

High concentrations of ammonium in swine waste lead to emissions of ammonia to the atmosphere (Aneja *et al.*, 2009), which can cause significant concentrations of ammonia in air at nearby communities (Wilson and Serre, 2007) as well as transport and subsequent deposition over longer distances (Dennis *et al.*, 2010). Volatilized ammonia has been linked to respiratory problems among exposed populations, including swine farm workers (Dosman *et al.*, 2004) and those living or attending school near the farms (Merchant *et al.*, 2005; Mirabelli *et al.*, 2006). Furthermore, ammonia reacts in the atmosphere to form fine particles that can cause respiratory disease (McCulloch *et al.*, 1998). In addition to ammonia, uncovered lagoons are sources of methane emission (Sharpe and Harper, 1999), with corresponding implications for climate change (Aneja *et al.*, 2009; Miller *et al.*, 2013).

In an effort to mitigate the environmental and human health impacts of emissions from industrial-scale swine farms, the North Carolina legislature enacted the 2007 Swine Farm Environmental Performance Standards Act (NC General Assembly, 2007b). This legislation banned the construction of new swine farms that employ open anaerobic lagoons and sprayfields as the primary methods of waste treatment and disposal (as crop fertilizer), respectively;

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instead, new farms must employ technologies that meet environmental performance standards (NC General Assembly, 2007b; NCDENR, 2009). The standards require substantial reductions in emissions of various pollutants to soil, groundwater, surface water, and air, including emissions of ammonia. The 2007 NC Renewable and Energy Efficiency Portfolio Standard Act also established a target for statewide energy production from swine waste, and it created a pilot program that authorized higher rates of payment for electricity generated from anaerobic digestion and methane capture systems on swine farms (NC General Assembly, 2007a). However, the incentives for energy production are not coupled to requirements to meet environmental performance standards. In addition, there are no environmental performance standards for existing swine farms, so that the state of practice for waste management continues to be storage and treatment in open lagoons with spray irrigation on nearby cropland for disposal.

We conducted a pilot study to evaluate the technical and economic feasibility of coupling conventional biological nitrogen removal (nitrification and denitrification) with anaerobic digestion for methane capture and energy recovery at a swine farm. The study was conducted at one of the few swine farms in North Carolina practicing full-scale anaerobic digestion of waste in covered lagoons with a methane capture system. Of particular interest was an analysis of stoichiometric issues relevant to nitrogen conversions in waste from which a substantial fraction of organic matter would have been removed by anaerobic digestion, including the availability of electron donors for denitrification, oxygen consumption, and net alkalinity demand of combined nitrification and denitrification. Production of nitrous oxide (N_2O) was also quantified. Details of the economic analysis are available elsewhere (Bunk, 2012).

Several previous studies have been conducted on combined anaerobic digestion and nitrification/denitrification of swine waste at much smaller scales than used in this study (Obaja *et al.*; Poo *et al.*; Deng *et al.*; Dosta *et al.*; Anceno *et al.*; Rajagopal *et al.*). To our knowledge, this is the first study to incorporate the modified Ludzack-Ettinger (MLE) concept for nitrogen removal (Grady *et al.*, 2011), although without external solids separation or recycle of settled biomass. In the MLE process, influent wastewater enters a denitrification reactor, which is followed by a nitrification reactor; internal recirculation between the reactors allows the oxidized nitrogen produced in the nitrification reactor to be denitrified in the denitrification reactor at the expense of electron donors in the influent. We also note that because there are no effluent concentration standards for waste treatment at existing swine farms in North Carolina (nor federal-level standards in the United States), the objectives of treatment and performance characterization are not the same as for treated wastewaters that are discharged to surface waters.

Materials and Methods

Study design and general characteristics

The pilot-scale nitrogen removal system was installed in a trailer located at the edge of a covered lagoon at Butler Farms in Lillington, NC (Supplementary Fig. S1 in Supplementary Data). Butler Farms is an 8,000-head grow/finishing farm with 10 barns, each housing an approximately equal number of animals. The lagoon that served as the source of influent for this study was the larger of two at the farm, receiving waste from six of the barns. The lagoon is not mixed, has a maximum volume of 2.5×10^7 L, and a maximum depth of 3.3 m. As is typical for swine farms in North Carolina, waste is flushed from the barns with liquid from the lagoons. Periodically, the lagoon liquid is sprayed onto on-site fields in accordance with State agronomic regulations. The lagoon cover and methane collection system were installed ~2 years before this study was initiated.

Influent to the pilot system was pumped continuously from the lagoon through an opening in the lagoon cover at the opposite end of the lagoon from the barn discharge, from a depth of 1 m below the liquid surface. The lagoon liquid was pumped to a flow-through sealed tank in the trailer, from which the influent to the pilot system was pumped through a peristaltic pump. The remainder of the lagoon liquid flow was recirculated to the lagoon through a second opening in the cover, ~4.5 m from the intake.

The on-site trailer accommodated the reactors (Supplementary Fig. S2), pumping equipment, and associated instrumentation as described below, computer for system monitoring and control, and an area for analytical equipment (analytical balance, drying oven, chemical oxygen demand [COD] digester, filtration apparatus, titration apparatus, and benchtop pH meter; Supplementary Fig. S2). It was heated and air-conditioned to maintain a consistent inside temperature; over the course of the study, the temperature in the trailer was 23.2°C ± 3.2°C (n = 288).

Chemical characteristics of the lagoon liquid for the entire duration of continuous system operation are summarized in Table 1.

Pilot system overview

As noted above, the nitrogen removal system incorporated the MLE concept in two bioreactors in series without external solids separation. Influent (lagoon liquid) was pumped to the denitrification reactor. Denitrified liquid was pumped to the nitrification reactor, to which pure oxygen was supplied as the oxygen source. Mixed liquor was internally recycled by

Table 1. Lagoon Liquid Characteristics Over Project Duration (September 10, 2010–May 27, 2011)

Parameter	*Mean ± S.D.* (n)	*Range*
NH_4^+-N (mg/L)	2,310 ± 280 (101)	1,770–2,670
Total N (mg/L)	2,750 ± 230 (85)	2,120–3,740
TDN (mg/L)	2,610 ± 180 (81)	2,060–2,980
Total COD (mg/L)	7,550 ± 2,240 (38)	1,770–10,200
Soluble COD (mg/L)	5,370 ± 1,820 (37)	1,390–7,740
TSS (mg/L)	1,580 ± 260 (27)	1,180–2,180
VSS/TSS (—)	0.40 ± 0.04 (15)	0.34–0.51
pH	7.71 ± 0.23 (27)	7.28–8.12
Total alkalinity (mg $CaCO_3$/L)	12,200 ± 420 (27)	11,200–13,100
Bicarbonate alkalinity (mg $CaCO_3$/L)	10,600 ± 460 (27)	9,840–11,300

COD, chemical oxygen demand; TDN, total dissolved nitrogen; TSS, total suspended solids.

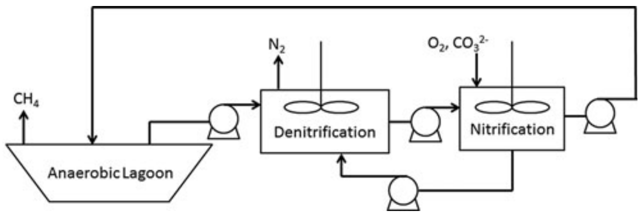

FIG. 1. Schematic of pilot-scale nitrogen removal system; not to scale.

pumping between the nitrification and denitrification reactors at a recycle ratio of 2.5 (recycle flow rate/system influent flow rate). Effluent from the nitrification reactor was pumped to the lagoon for discharge. A simplified process flow diagram is provided in Figure 1.

Hydraulic retention times (HRT) were $\sim$33 days for the nitrification reactor and $\sim$9 days for the denitrification reactor, based on the system influent flow and mean volume of each reactor. The pilot system was operated continuously from September 10, 2010, through May 27, 2011. Over the first 5 months, there were several issues with equipment failure and other operational problems that led to reactor upsets, usually manifested as nitrite accumulation in the nitrification reactor. Therefore, reactor operating and performance data are reported only for the period after which the final operating conditions were established (final 107 days; referred to below as the performance reporting period). Since the influent characteristics are important for stoichiometric analysis and did not depend on reactor performance, as noted above, the lagoon liquid properties are reported for the entire period of continuous operation in Table 1.

Reactor design

Reactors were 5,000 L (nitrification) and 1,000 L (denitrification) high-density polyethylene tanks. Operating volumes were $\sim$2,000 L and 500 L, respectively. The reactors were totally enclosed except for ports for pump tubing, and off-gas; each reactor was operated with a gauge headspace pressure $\sim$0.5 kPa. To prevent reactor short circuiting, liquid pumped into a reactor was discharged above the liquid surface and the intakes for liquid pumped from the reactor were located near the bottom of the tank. Mixing in the denitrification reactor was provided by an internal recirculation line with intake and discharge located at opposite ends of the tank. Mixing in the nitrification reactor was provided by a submersible pump located at the bottom near one end of the tank, whose discharge was directed to the opposite end of the tank. Each reactor was assumed to be completely mixed, so that the effluent composition from each reactor was the same as its contents.

Pure oxygen was provided to the nitrification reactor through a fine-bubble diffuser (Western Outdoor Aquatics, Inc., Frederick, CO) and through a mass flow controller (Omega, Stamford, CT); pure oxygen was used rather than air because of logistical constraints on the frequency of exchange of gas cylinders in the field. The pH in the nitrification reactor was adjusted by pumping a concentrated sodium carbonate solution into the reactor in response to continuous pH measurement using a proportional digital controller (Hannah Instruments, Woonsocket, RI), with pH 6.8 as the minimum set point. The majority of sodium carbonate required for pH control was consumed over the first few months of system operation, with relatively little consumed over the performance reporting period. Liquid volume in each reactor was measured by comparing the liquid level to a calibrated scale on the exterior of the tank; the volume in each reactor was recorded daily.

All pumping into and out of each reactor was with dedicated peristaltic pumps (MasterFlex computerized drive with Easy-Load II head and high-performance precision Norprene L/S 36 tubing; Cole-Parmer, Vernon Hill, IL). Each pump was operated continuously with adjustment as needed to maintain the desired volume in each reactor. The pump drives were controlled with software (WinLIN; Cole-Parmer) installed on a personal computer. Pump flow rates were periodically calibrated by timed delivery into a graduated cylinder.

Startup

The nitrification reactor was filled to the desired volume with tap water and $\sim$200 L of return activated sludge from the Orange Water and Sewer Authority wastewater treatment plant (Chapel Hill, NC), which performs nitrification and biological phosphorus removal. The denitrification reactor was filled to the desired volume with lagoon liquid. The system was operated in batch mode with internal recycle between the nitrification and denitrification reactors for 12 days, and then the internal recycle between reactors was turned off to allow strictly batch operation in the nitrification reactor for 4 weeks. Continuous operation was initiated after this period of batch operation.

Instrumented measurements

The temperature of the lagoon liquid was measured continuously with a probe submerged in the lagoon near the intake for the pilot system influent. The temperature and pH of mixed liquor in the denitrification reactor were monitored continuously with probes mounted in the internal recirculation line used for mixing. Temperature, pH, and dissolved oxygen (DO) of mixed liquor in the nitrification reactor were monitored continuously with probes similarly mounted in an internal recirculation line. Temperature probes were calibrated against an electronic thermometer (Cole-Parmer) that had been calibrated against a mercury thermometer whose calibration was traceable to the US National Institute of Standards and Technology; room-temperature deionized water was the calibration medium. The mass flow controller for oxygen delivery was calibrated using a Gilibrator automated bubble meter (Sensidyne, St. Petersburg, FL). Calibration of the DO and pH probes was checked weekly according to the manufacturer's instructions. For reporting purposes, temperature, pH, and DO were recorded manually daily.

Sample collection

Lagoon liquid was collected from the sealed tank inside the trailer used as the source of reactor influent. Samples from each reactor were obtained from a port located on the respective internal recirculation line. Grab samples of lagoon liquid, denitrification effluent, and nitrification effluent were collected at least twice weekly, but not all samples were analyzed for every parameter. Samples were immediately filtered through glass fiber filters (Whatman GF/B or GF/C) on-site using a filtration apparatus dedicated to each sampling location. COD, total suspended solids (TSS), and alkalinity were measured on-site. Otherwise, filtered and unfiltered samples were frozen in an on-site freezer and transported weekly to the laboratory at the University of North Carolina, Chapel Hill campus, for further analysis. Off-gas from the nitrification reactor and headspace gas from the denitrification reactor were collected weekly over the performance reporting period and stored in air-tight syringes for transport to the campus laboratory.

Analytical methods

Filtered samples were analyzed for ammonium, nitrite, and nitrate according to standard methods 4500-NH_3F, 4500-NO_2^-B, and 4500-NO_3^-F, respectively (Eaton *et al.*, 1999). Each of the duplicate dilutions (100×–5,000× as needed) was measured in duplicate. Ammonium chloride, sodium nitrite, and potassium nitrate were used to prepare standard curves. Concentrations are reported in mg N/L. In preliminary analyses of lagoon liquid, neither nitrite nor nitrate was detected (data not shown).

Total nitrogen (TN) and total dissolved nitrogen (TDN) were analyzed on unfiltered and filtered samples, respectively, using a Shimadzu (Columbia, MD) total organic carbon analyzer with TN attachment. Duplicate dilutions (500×–2,000× as needed) were each measured in duplicate or triplicate; triplicates were analyzed when the difference between duplicate measurements was >2%. Disodium ethylenediaminetetraacetic acid was used to prepare standard curves for TN and TDN analyses.

Total COD (tCOD) and soluble COD (sCOD) were determined using CHEMetrics (Midland, VA) COD digestion vials (20–1,500 ppm range) on unfiltered and filtered samples, respectively, in duplicate. Samples were diluted 10× directly in the COD vials. Sodium acetate was used as the standard, on the assumption that the majority of sCOD in the anaerobically digested lagoon liquid comprised volatile fatty acids.

Alkalinity was measured using standard method 2320B (Eaton *et al.*, 1999). Fresh sulfuric acid solution was prepared as needed, and samples were titrated to pH 4.3 as a measure of total alkalinity. The volume of acid required to reach pH 5.8 was recorded and used to calculate bicarbonate alkalinity. Acid equivalents required to reach the respective pH endpoints were converted to alkalinity in mg $CaCO_3$/L. TSS were measured using standard method 2540D (Eaton *et al.*, 1999).

Nitrous oxide was measured on a Shimadzu 14A gas chromatograph (GC) with an electron capture detector and 90% argon/10% methane as carrier gas. Methane (CH_4) was measured on a Shimadzu 8AIF GC with a flame ionization detector and ultra-high-purity (UHP) N_2 as carrier gas. Carbon dioxide (CO_2) was measured with a Shimadzu 8AIT GC with a thermal conductivity detector and UHP He as carrier gas. Gas-phase ammonia was captured in 0.2% (w/v) boric acid and measured as aqueous ammonium.

Data analysis

Cumulative mass loading and mass discharge were calculated by multiplying the concentration of a constituent by the net pumping rate into or out of a reactor, respectively. Pumping rates were recorded daily; concentration data were linearly interpolated between measured values. Cumulative mass loading and discharge across the system were used to calculate the removal efficiency over the performance reporting period.

Oxygen consumption in the nitrification reactor was assumed to equal the oxygen required to oxidize ammonium to nitrite and nitrate plus the net change in total COD across the reactor. The proportion of oxygen required for oxidation of ammonium to nitrite versus nitrate was based on the ratio of nitrite and nitrate in the nitrification reactor effluent and the known stoichiometry of nitrification reactions (3.43 g O_2/g NH_4^+-N for ammonium oxidation to nitrite and 4.57 g O_2/g NH_4^+-N for ammonium oxidation to nitrate).

Cumulative off-gas flow from each reactor was combined with the mean gas-phase concentration of N_2O to estimate the yield of N_2O relative to ammonium-N removal (cumulative mass of N_2O-N production/cumulative mass of AN consumption) over the performance reporting period. The off-gas flow from the nitrification reactor was estimated based on the known mass of O_2 addition (converted to molar units based on the mean temperature in the trailer and the ideal gas law), mass of oxygen consumed, and the volumetric composition of measured gases (CO_2, CH_4, N_2O, and NH_3), with the balance assumed to be O_2. The off-gas flow from the denitrification reactor was based on the mass of nitrogenous gases (N_2, N_2O, and NH_3) produced and the measured gas composition, assuming the unmeasured balance to be N_2; the cumulative mass production of nitrogenous gases was estimated from the net change of TN across the system minus the N_2O-N released from the nitrification reactor.

Results

Characteristics of the lagoon liquid over the entire project period are summarized in Table 1. The most variable influent parameter was COD, which resulted from the variability of temperature in the lagoon (Fig. 2). Since the lagoon was not heated, the temperature of the lagoon liquid varied seasonally in accordance with the ambient temperature. Gas production

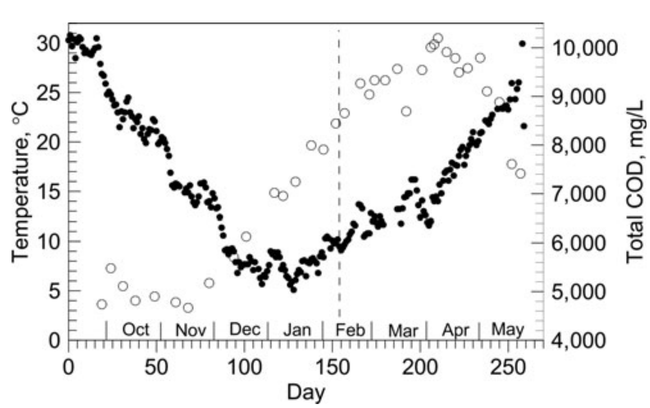

FIG. 2. Lagoon liquid temperature (*filled symbols*) and total chemical oxygen demand (COD) concentration (*open symbols*) over the duration of the project (day 0 was September 10, 2010). The *dashed vertical line* indicates the beginning of the performance reporting period as defined in the text (February 10–May 27, 2011).

in the lagoon declines with decreasing temperature in the winter months (Supplementary Fig. S3), with a corresponding decrease in anaerobic consumption of COD and increasing COD concentration in the lagoon (Fig. 2). COD concentrations in the lagoon liquid did not begin to decline from the peak until the lagoon temperature reached ~15°C (Fig. 2).

The majority (95%) of TN in the lagoon liquid was dissolved, with ammonium-N accounting for 89% of the TDN (Table 1). The majority of TSS appear to be inert, as the mean VSS/TSS ratio was only 0.4.

System performance

Data on nitrogen species, COD, and other characteristics of the influent and each reactor over the performance reporting period are summarized in Table 2. Based on cumulative mass loading and discharge over this period (Table 3), the system achieved 98% removal of NH_4^+-N, 83% removal of total N, and 75% removal of total COD. Cumulative mass loading and discharge of TN and NH_4^+-N are plotted in Supplementary Figure S4. Nearly all of the effluent nitrogen other than N_2O could be accounted for as nitrite and/or nitrate (Tables 2 and 3).

It is not possible to quantify organic N in the reactors because the data for TN and TDN in the nitrification reactor (or system effluent; Tables 2 and 3) are less than the sum of the inorganic N species. In addition, the apparent total N removal across the system (Table 3) is equivalent to the NH_4^+-N removal alone. We believe these discrepancies are likely a result of the instrumental method used for TN and TDN analysis, which relies on catalytic thermal conversion of N species; at the high dilution required for our samples, even small inefficiencies in conversion would be magnified and manifested as lower-than-expected concentrations.

Consumption of COD

There was no measurable removal of total COD across the nitrification reactor (Fig. 3), suggesting that virtually all the total COD removal across the system occurred in the denitrification reactor. Accordingly, the system effluent COD appears to comprise mostly nonbiodegradable or very slowly biodegradable COD (i.e., ~25% of the influent tCOD was nonbiodegradable over the retention times used in this study). Based on the cumulative mass data in Table 3, assuming that the TN removed was oxidized to nitrite and nitrate in proportion to the masses discharged (65.7% NO_2^--N), and assuming COD consumption by NO_2^--N equal to 60% of that of NO_3^--N, the total COD removed was 3.8 g COD/g NO_3^--N equivalent.

Table 3. Cumulative Mass Loading and Discharge Across the System Over Performance Reporting Period (February 10–May 27, 2011)

Parameter	*Loading (kg)*	*Discharge (kg)*	*Removal (%)*
NH_4^+-N	14.9	0.4	97.5
Total N	17.7	3.1	82.7
TDN	16.9	3.2	81.4
NO_2^--N	NA	2.3	NA
NO_3^--N	NA	1.2	NA
N_2O-N	NA	1.2	NA
Total COD	55.2	13.9	74.7
Soluble COD	38.8	10.6	72.7

Data are rounded to one decimal place.

Gas-phase measurements

Data on the composition of the gas phase in each reactor over the performance reporting period are summarized in Table 4. For stoichiometric analysis, the most important of these gases was N_2O. Based on the cumulative mass production of N_2O and the mass removal of ammonium across the system (Table 3), 8.2% of the oxidized NH_4^+-N was converted to N_2O-N. Since the concentration of N_2O in the off-gas from the denitrification reactor was much lower than that from the nitrification reactor (Table 4), as well as the fact that there was a much greater flow of gas (primarily oxygen) through the nitrification reactor, the contribution of N_2O from denitrification was negligible (3.3% of the total N_2O produced).

With concentrations of off-gas ammonia in the ppmv range (Table 4), volatilization of ammonia was negligible (~0.01% of the ammonium removal across the system).

Table 2. Influent and Reactor Characteristics Over Performance Reporting Period (February 10–May 27, 2011)

Parameter	*Influent*	*Denitrification*	*Nitrification*[a]
NH_4^+-N (mg/L)	2,370 ± 140 (2,070–2,670; 41)	651 ± 86 (499–858; 42)	78 ± 53 (1–199; 42)
NO_2^--N (mg/L)	NA	32 ± 43 (ND–140; 42)	484 ± 217 (3–779; 42)
NO_3^--N (mg/L)	NA	71 ± 75 (ND–249; 42)	288 ± 234 (66–1,190; 42)
Total N (mg/L)	2,820 ± 135 (2,510–3,200; 41)	836 ± 158 (609–1,250; 41)	681 ± 191 (439–1,080; 41)
TDN (mg/L)	2,690 ± 150 (2,340–2,980; 37)	794 ± 165 (565–1,180; 37)	682 ± 191 (450–1,090; 40)
Total COD (mg/L)	9,270 ± 742 (7,410–10,200; 20)	3,080 ± 210 (2,600–3,580; 31)	3,180 ± 460 (2,620–4,680; 32)
Soluble COD (mg/L)	6,660 ± 1,190 (3,900–7,740; 19)	2,240 ± 170 (1,890–2,470; 31)	2,390 ± 330 (1,610–2,940; 32)
Volume (L)	NA	533 ± 59 (450–750; 86)	2,000 ± 83 (1,500–2,200; 99)
Temperature (°C)	16.3 ± 4.9 (9.1–30.0; 93)	22.3 ± 2.2 (16.5–26.3; 100)	24.5 ± 2.7 (16.4–30.5; 100)
DO (mg/L)	NA	NA	10.9 ± 7.6 (0.8–27.8; 100)
pH	7.53 ± 0.17 (7.28–7.86; 13)	7.92 ± 0.34 (7.01–8.56; 99)	6.97 ± 0.11 (6.84–7.43; 100)

Data are mean ± S.D. (range; *n*).
[a]Nitrification reactor characteristics = system effluent characteristics.
DO, dissolved oxygen; NA, not applicable; ND, not detected.

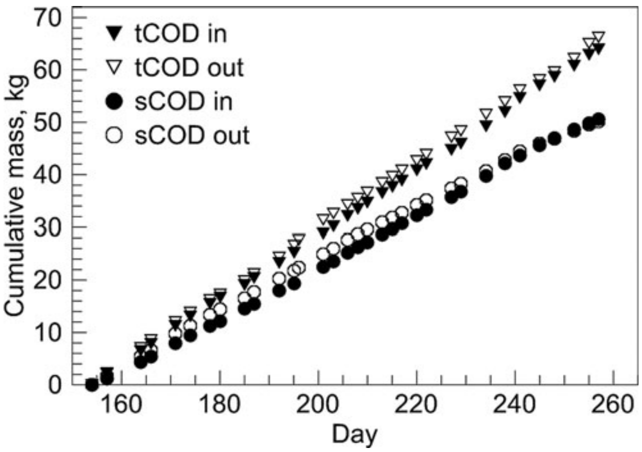

FIG. 3. Cumulative COD loading and discharge across nitrification reactor over performance reporting period (February 10–May 27, 2011); day numbering is relative to the start of the overall project.

Methane in the headspace of the denitrification reactor is assumed to represent volatilization of dissolved methane present in the influent (lagoon liquid).

Discussion

Swine waste is high strength with respect to both biodegradable organic matter and ammonium-N nitrogen. It is, therefore, a candidate for anaerobic digestion with energy recovery as well as a significant source of nitrogen pollution to the environment. Most earlier work on coupling anaerobic digestion with biological nitrogen removal has included some combination of at least partial oxidation of ammonium with nitrogen removal by denitrification or anaerobic ammonium oxidation (ANAMMOX), although various schemes have been proposed. These include bypassing a fraction of the raw waste around the anaerobic digester to provide more electron donors for denitrification (Obaja *et al.*, 2004; Deng *et al.*, 2008; Anceno *et al.*, 2009; Rajagopal *et al.*, 2011); recycling nitrified effluent to the anaerobic digester, which therefore would be responsible for both denitrification and methanogenesis in the same reactor (Bernet *et al.*, 2000; Bortone, 2009; Rajagopal *et al.*, 2011); nitritation and ANAMMOX for nitrogen removal from anaerobically digested waste (Hwang *et al.*, 2006; Karakashev *et al.*, 2008; Yamamoto *et al.*, 2008; Zhang *et al.*, 2012); and the concept evaluated in the present study, anaerobic digestion of the complete waste stream followed by nitrification/denitrification (Poo *et al.*, 2004; Deng *et al.*, 2008; Dosta *et al.*, 2008; Rajagopal *et al.*, 2011).

The previous studies, in which either nitritation/ANAMMOX or nitrification/denitrification were evaluated on anaerobically digested swine waste, have been conducted at a small laboratory scale. In the study by Rajagopal *et al.* (2011), both the anaerobic digestion and nitrogen removal processes were operated at a small pilot scale ($\sim$120 L). None of the studies on nitrification/denitrification utilized the MLE configuration and, in some cases, a supplemental carbon source was added to maximize denitrification (Poo *et al.*, 2004; Dosta *et al.*, 2008; Zhang *et al.*, 2012). Therefore, none of the previous studies on nitrogen removal from anaerobically digested swine waste is comparable to the present study.

Denitrification

Employing nitrification/denitrification on waste that has already been subjected to anaerobic digestion maximizes the amount of organic matter available for conversion to methane, while minimizing the amount of oxygen required for subsequent aerobic treatment. A drawback to this approach, however, is that the residual organic matter from anaerobic digestion might be insufficient to meet the electron donor demand for denitrification, thereby limiting the extent of nitrogen removal. Nitrate has the capacity to remove 2.86 g COD per g NO_3^--N (more if biomass growth is accounted for). Using mean data for the entire project (Table 1), the total COD in the lagoon liquid would not have met the demand for denitrification if all the TN was converted to nitrate. As noted above, the residual COD in the lagoon varies seasonally in response to the extent of gas production in the lagoon, so that the extent of denitrification in a biological nitrogen removal system can be expected to vary seasonally as well.

In principle, the total COD in the lagoon liquid over the performance reporting period (Tables 2 and 3) should have been sufficient to completely remove nitrate if all the TN in the lagoon liquid was converted to NO_3^--N. However, only 88% TN removal was achieved. The COD concentration in the denitrification reactor was high (Table 2), suggesting that residual COD not used for denitrification may not have been readily biodegradable. This is supported by the observations that there was little difference in the COD concentrations between the denitrification and nitrification reactors (Table 2) and that there was no removal of sCOD across the nitrification reactor (Fig. 3), which suggests that most of the residual COD was not aerobically degradable either. The bioavailability of residual COD in anaerobically digested swine waste has been demonstrated to depend on the solids retention time of the digester (Kinyua *et al.*, 2014).

As noted above, previous studies have explored either bypassing a fraction of the raw waste around the anaerobic digester or recycling nitrified effluent to the anaerobic digester to provide the necessary electron equivalents for

TABLE 4. REACTOR OFF-GAS COMPOSITION OVER PERFORMANCE REPORTING PERIOD (FEBRUARY 10–MAY 27, 2011)

Gas	*Denitrification reactor*	*Nitrification reactor*
CO_2 (%)	8.9 ± 2.7 (5.2–15.1; 13)	37.2 ± 8.9 (16.9–49.8; 13)
CH_4 (%)	2.2 ± 0.8 (1.5–4.1; 14)	0.02 ± 0.05 (0.01–0.20; 14)
N_2O (%)	0.13 ± 0.20 (0.01–0.67; 14)	0.78 ± 0.19 (0.43–1.11; 14)
NH_3 (ppmv)	38 ± 21 (2–82; 11)	5.0 ± 4.3 (<1–15; 11)

Data are mean ± S.D. (range; *n*).

denitrification. Both approaches would reduce the amount of methane that could be generated from the waste. In addition, adding raw waste directly to the nitrogen removal process can substantially increase the oxygen consumption associated with the aerobic component of the process (Deng *et al.*, 2008). Another proposed strategy, which simultaneously would decrease oxygen consumption, is to partially oxidize ammonium to nitrite (nitritation), relying on the produced nitrite for denitrification (Dosta *et al.*, 2008; Rajagopal *et al.*, 2011); the electron donor demand from denitrification with nitrite (1.71 g COD/g NO_2^--N) is less than that with nitrate. Although not intentional, in this study, nearly two-thirds of the oxidized nitrogen in the effluent from the nitrification reactor (system effluent) was in the form of nitrite (Table 2).

Unlike municipal wastewater treatment, in which there may be upper limits on TN discharged to receiving water, there are no such limitations on nonpoint nitrogen sources from agricultural waste management in the United States. Nevertheless, extensive removal of ammonium-N by nitrification and even partial denitrification of the oxidized nitrogen would still have a substantial impact on human health and on reducing nitrogen loads to the environment, respectively. These factors should be taken into account when evaluating the impact and limitations of coupling anaerobic digestion with nitrogen removal from swine waste.

Ammonium oxidation

Throughout the study, nitrite tended to accumulate in the nitrification reactor in lieu of complete oxidation of ammonium to nitrate, even during most of the performance reporting period (Table 2). Accumulation of nitrite typically is observed under oxygen-limiting conditions, which is the principal means of promoting nitritation over complete ammonium oxidation to nitrate (Lackner *et al.*, 2014). However, DO was not limiting in the present study (Table 2), suggesting that other mechanisms were responsible for limiting nitrite oxidation. It is likely that populations of nitrite-oxidizing bacteria (NOB) were slower to recover from process upsets that occurred before the performance reporting period; as seen in Supplementary Figure S5, effluent nitrite concentrations declined steadily over time, with a substantial decrease over the last few weeks of the study. Inhibition of NOB by nitrite or free nitrous acid (Anthonisen *et al.*, 1976; Vadivelu *et al.*, 2006; Park *et al.*, 2010; Zhou *et al.*, 2011; Wang *et al.*, 2014) or by free ammonia (Anthonisen *et al.*, 1976; Park *et al.*, 2010) could explain such slow recovery. Free nitrous acid can be inhibitory in the sub-mg/L range, and free ammonia can be inhibitory in the low mg/L or even sub-mg/L range (Anthonisen *et al.*, 1976; Park *et al.*, 2010). At the mean pH and temperature of the nitrification reactor (Table 2), free nitrous acid concentrations would have ranged from about 0.02 to 0.17 mg/L (0.11 mg/L at the mean NO_2^--N concentration) and free ammonia from about 0.05 to 0.81 mg/L (0.43 mg/L at the mean NH_4^+-N concentration); the higher concentrations in these ranges could have inhibited growth of NOB (Park *et al.*, 2010). It is also possible that characteristics of the anaerobically digested swine waste adversely affected NOB.

Although ammonium removal was excellent, the mean effluent concentration over the performance reporting period (Table 2) was higher than might be expected at steady state for the retention time of the nitrification reactor. Like nitrite, ammonium concentrations in the effluent declined steadily over time (Supplementary Fig. S5); it is possible that, therefore, relatively slow recovery of AOB during the performance reporting period also could have been a result of inhibition from free nitrous acid and/or free ammonia (Anthonisen *et al.*, 1976; Park *et al.*, 2010).

It was necessary to add supplemental alkalinity (as carbonate) to the anaerobically digested lagoon effluent to maintain pH > 6.8 in the nitrification reactor. Most of the added alkalinity was required earlier in the project before stable reactor operation was achieved, so that the total carbonate requirement during the performance reporting period was not quantifiable.

Oxidation of ammonium to either nitrite or nitrate consumes 7.1 g alkalinity as $CaCO_3$ per g NH_4^+-N oxidized. Based on the lagoon liquid characteristics over the duration of the project (Table 1), the total alkalinity would be far less than required for oxidation of all the TN. However, denitrification from nitrate produces alkalinity to an extent that is nearly half of the alkalinity consumed from ammonium oxidation per unit nitrogen (Grady *et al.*, 2011), so that there would be sufficient alkalinity in the lagoon liquid for complete oxidation of TN followed by denitrification of the nitrate produced. However, as noted above, the extent of denitrification would vary seasonally and also depends on the biodegradability of the residual COD after anaerobic digestion. Therefore, the need for additional alkalinity to maintain pH for nitrification can be expected to vary seasonally. Depending on how much alkalinity is required, this can represent a significant operating cost in a full-scale system (Bunk, 2012). Providing alkalinity is one rationale for enhancing denitrification by adding undigested raw waste to the nitrification/denitrification process (Deng *et al.*, 2008).

A strategy that would simultaneously reduce the alkalinity demand, oxygen demand, and electron donor demand for nitrogen from anaerobically digested swine waste is to combine nitritation of a fraction of the ammonium with ANAMMOX to remove the remainder of the ammonium (Hwang *et al.*, 2006; Karakashev *et al.*, 2008; Yamamoto *et al.*, 2008; Zhang *et al.*, 2012). Nitritation/ANAMMOX treatment of the anaerobically digested swine waste evaluated in this study is described in a companion article (Staunton and Aitken, 2015).

Nitrous oxide production

The yield of N_2O-N per unit NH_4^+-N oxidized (8.2%) obtained in this study is within the range reported in the literature for biological nitrogen removal systems (Ni *et al.*, 2011), although it is higher than the range observed in a survey of municipal wastewater treatment plants in the United States (Ahn *et al.*, 2010). Production of N_2O tends to be higher in systems operated for nitritation rather than complete nitrification of ammonium to nitrate (Yang *et al.*, 2009; Ahn *et al.*, 2011) and has generally been associated with low DO conditions (Aboobakar *et al.*, 2013; Peng *et al.*, 2014; Pijuan *et al.*, 2014). Ammonium oxidation to nitrite exceeded the complete oxidation to nitrate in this study, but the mean DO concentration in the nitrification reactor was generally quite high (Table 2). Ahn *et al.* (2010) have suggested that a combination of high nitrite and high DO can also lead to increased rates of N_2O production in nitrifying systems. As noted above, it is also possible that the characteristics of swine waste may inherently influence the activity of NOB and/or N_2O production during ammonium oxidation.

Nearly all the N_2O produced in this study was attributable to the nitrification process, rather than denitrification. A similar observation was made in a previous study on nitrification/denitrification of anaerobically digested swine waste (Rajagopal and Béline, 2011), although the yield of N_2O per unit TN removed was much lower than in the present study.

Scale-up issues

There are currently no U.S. federal or North Carolina concentration standards for on-farm disposal of waste (treated or otherwise) from existing industrial-scale animal operations. Accordingly, issues relevant to scale up of any contemplated treatment scheme cannot be extrapolated from standard practice for discharge of municipal or industrial wastewater to receiving waters. Treated effluent from the coupling of nitrogen removal to anaerobic digestion for energy recovery, as considered in this study, would simply be used to flush barns to remove fresh waste and/or to irrigate fields at the farm. Under such a scenario, solids removal from the effluent would not be a priority. Furthermore, operation of a nitrification/denitrification system in an activated sludge format (with solids separation and biomass recycle) would add a level of operational complexity that could be a disincentive for the typical farmer to adopt the technology. At existing industrial-scale farms, there would be plenty of capacity to modify existing uncovered lagoons to accommodate both anaerobic digestion and the two-reactor nitrogen removal system, without biomass recycle as evaluated in this study. For example, at Butler Farms, the combined capacity of its two lagoons is over 40 million liters; for the volume of waste generated at Butler Farms, the combined HRT of the nitrification/denitrification system that we evaluated would require less than 2% of the existing lagoon capacity.

Summary

Nitrification/denitrification is capable of achieving high extents of ammonium removal from anaerobically digested swine waste. The extent to which TN can be removed depends on the biodegradable organic matter in the digested waste that is available for denitrification; the available organic matter, in turn, can depend on the extent of gas production during the anaerobic digestion process, which can vary seasonally. In the pilot system operated in this study, the majority of biodegradable influent organic matter was consumed in the denitrification step. The extent of denitrification will also influence the stoichiometry of net alkalinity consumption across the system; the lower the extent of denitrification, the more likely that supplemental alkalinity would be required to maintain neutral pH in the nitrification reactor. Production of N_2O from the nitrification/denitrification process can be significant, offsetting some of the greenhouse-gas benefit from anaerobic digestion with methane capture for energy recovery.

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Author Disclosure Statement

No competing financial interests exist.

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ATTACHMENT 56

Loyd Ray Farms, Inc.
Innovative Animal Waste Management System
Permit No. AWI990031
Permit Compliance Semi-Annual Report

July 1, 2018 – December 31, 2018 Semi-Annual Reporting Period

Submitted January 31, 2019

Submitted on Behalf of:
Loyd Ray Farms, Inc.
2049 Center Rd.
Boonville, NC 27011

This Semi-Annual Compliance Report provides an overview of the manner in which the subject facility, Loyd Ray Farms, has maintained compliance with the conditions of the Innovative Animal Waste Management System permit for the reporting period from July 1, 2018 through December 31, 2018. During this reporting period, the system was operated in accordance with the Innovative Swine Waste Treatment System and subject to the requirements thereof.

Overview of System

The animal waste treatment system installed at Loyd Ray Farms is designed to meet the Environmental Performance Standards set forth by North Carolina law for new and expanded swine facilities through the use of nitrification/denitrification and further treatment. This report confirms on a semi-annual basis that the innovative waste management system is in compliance with NC Department of Environmental Quality and its divisions, to insure that the utilization of the anaerobic digester technology to turn raw animal waste into biogas for the purpose of reducing greenhouse gas emissions minimizes the overall environmental impact of the swine farm, and explains the occurrences of operations, and testing requirements over the six month period, to monitor the

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system, as it continues to produce renewable energy, generate carbon offsets, and reduce odor on the farm. The report is designed to not only show a synopsis of the maintenance activities on the farm, but also to supply the analysis of the system's performance and further describe the results of the monitoring and testing activities.

In addition to addressing compliance with the conditions of the permit, the following summaries provide an overview of the system operations including graphs of systems performance, the Microturbine performance, and biogas levels (pages 2-5), and lists all sampling and reporting requirements per the Innovative Animal Waste Management System Permit No. AW1990031 (pages 16-18). For each requirement, this report records on-site monitoring that occurred, with a brief explanation for each farm site visit (pages 6-16) for this reporting period. Additionally, detailed site visits recording maintenance and repairs completed during the second half of 2018, from July 1 through December 31, 2018 are also included in this report. In summary, From July 1, 2018 through December 31, 2018, all processes that comprise the innovative swine waste treatment system were fundamentally operational, and electricity generation was capable for the greater percentage of the reporting period, except for several weeks in September and October, which required the system to undergo repairs which are specified in more detail later in this report. Overall, except for a few disruptions for minor repairs, the system performed well from operational, power generation, greenhouse gas emission reduction and environmental performance perspectives. The maintenance activities were a little more accelerated as the system is getting older, and as the farm staff was unable to maintain a regular flushing schedule. Additional observations of system performance are exhibited in the operator log attached to this report. (Appendix A)

During this compliance period, ambient air analyses were accomplished on September 25th, 2018, and December 19th, 2018, details of the monitoring events have been added to this report (pages 20-22). The air emissions from water surfaces were found to be in compliance and much lower than the permit allows and show that the system is performing according to expectations.

This report was completed on behalf of Loyd Ray Farms, Inc., by Cavanaugh & Associates, P.A., under the direction of the Duke Carbon Offsets Initiative (DCOI). Please contact Matthew Arsenault at 919-613-7466 (Matthew.Arsenault@duke.edu) with any questions. A copy of this report will be provided to Loyd Ray Farms, Inc., and will be maintained on-site with the other permit compliance documentation.

Environmental Treatment System

Figure 1. below, **Environmental Treatment System Uptime**, depicts the operation of the aeration system that performs the nitrification function for the monitoring period. The environmental treatment system performed well and remained consistent for most of the year with the exception of a few weeks in September 2018. A few required Operations and Maintenance activities over the months of September and October will be explained in more detail in the Site Observation logs. Once corrected, the environmental treatment system returned to normal operations, and continues to generate electricity derived from hog manure and wastes.

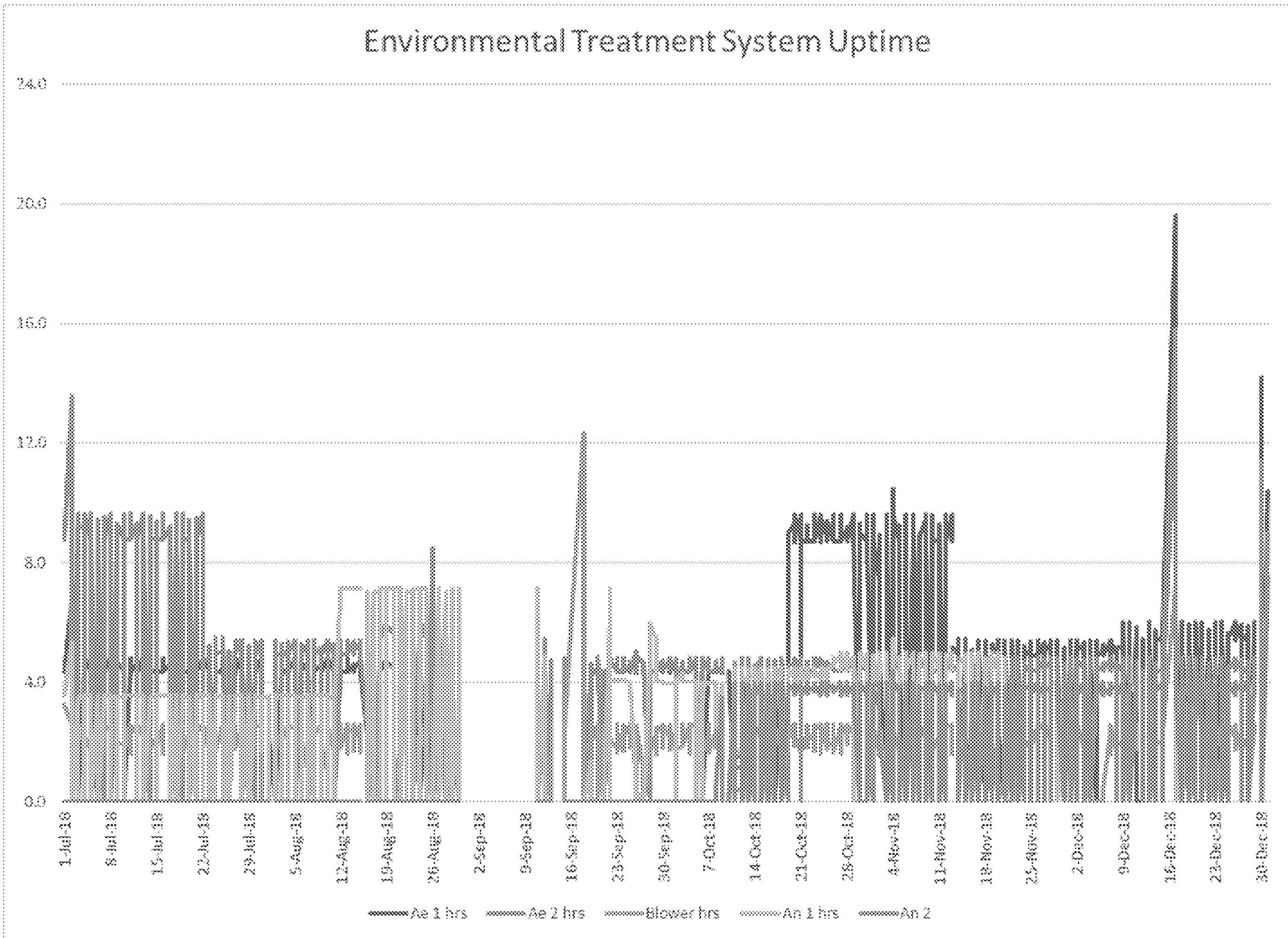

Figure 1. Environmental Treatment System Generator Uptime 7-1-2018 through 12-31-2018

The information in the daily log sheets correlates directly with the graph above, an electrical shutdown occurred during the first two weeks of September, which required replacing some parts to get the environmental system and the SCADA system back operational. Another complication during that time was a clogged digester mixing pump. After troubleshooting the problem, the operations staff was able to open the valves, back flush the system, rectify the bilge pump, and get the system operational again. Many areas in North Carolina received historical flooding in September, and Loyd Ray Farms received an unusually heavy amount of rain. However, the digester and operations remained intact and were not particularly affected by the storms nor rainfall. During the electrical outage that occurred around September 16th the flare was kept burning via gravity feed of the biogas continuously until we could get the system back online, as evidenced in Figure 1. Another atypical occurrence was October 8th, when the system required a repair of the digester pump. A spike in blower hours also occurred in December, when installation of like replacement fans and like replacement circuit boards in the Phase converter were required to properly cool down the system. Once corrected, the environmental treatment system returned to normal operations. Both aerobic and anaerobic performance during the compliance months are recorded in the **Environmental System Uptime** graph above. The anaerobic mixing system uptime was 71% for the reporting period, while the aeration system uptime was reported as 70%.

Figure 2. below depicts the **Microturbine Output in kilowatt hours** (kWh) during the compliance period. Biogas flow is also monitored and recorded for the system. The biogas may only be disposed of through use by the microturbine and flare, controlled release through venting, or leaks from the system, which cannot be measured.

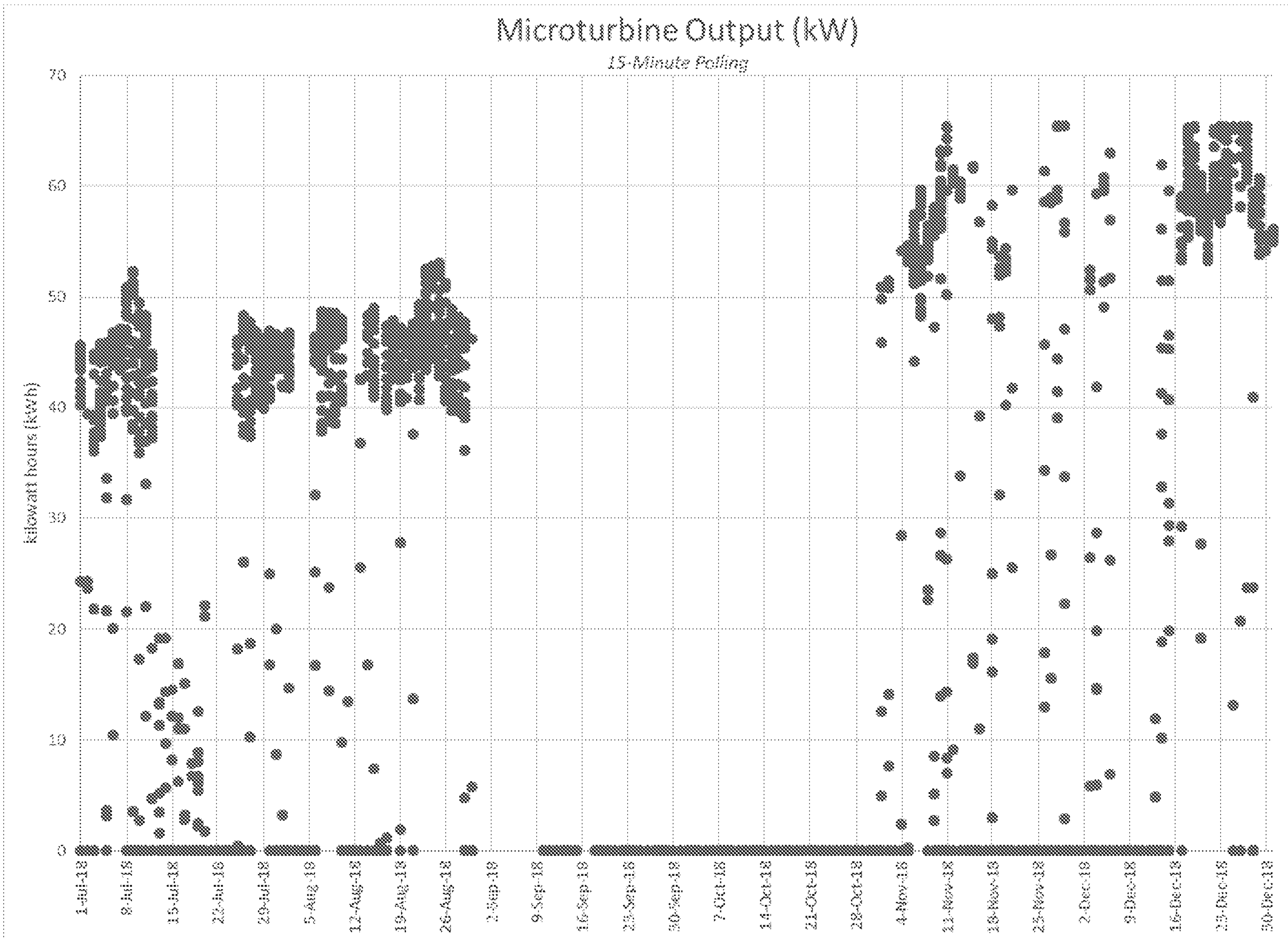

Figure 2. Microturbine Output per Kilowatt Hour (KWh) July 1, 2018- December 31, 2018

Figure 3., **Measured Biogas Flow and Flare Use,** which follows, depicts the dataset relative to the measured biogas flow and flare usage, which utilizes the same dataset for the duration of the compliance period. The measured gas flow directly correlates to Figure 2, the microturbine output above, both reflect the system consistency, except for the downtime in the months of September and October of the reporting period. Once the required maintenance activities were accomplished, and the system returned to operational, the performance was normalized.

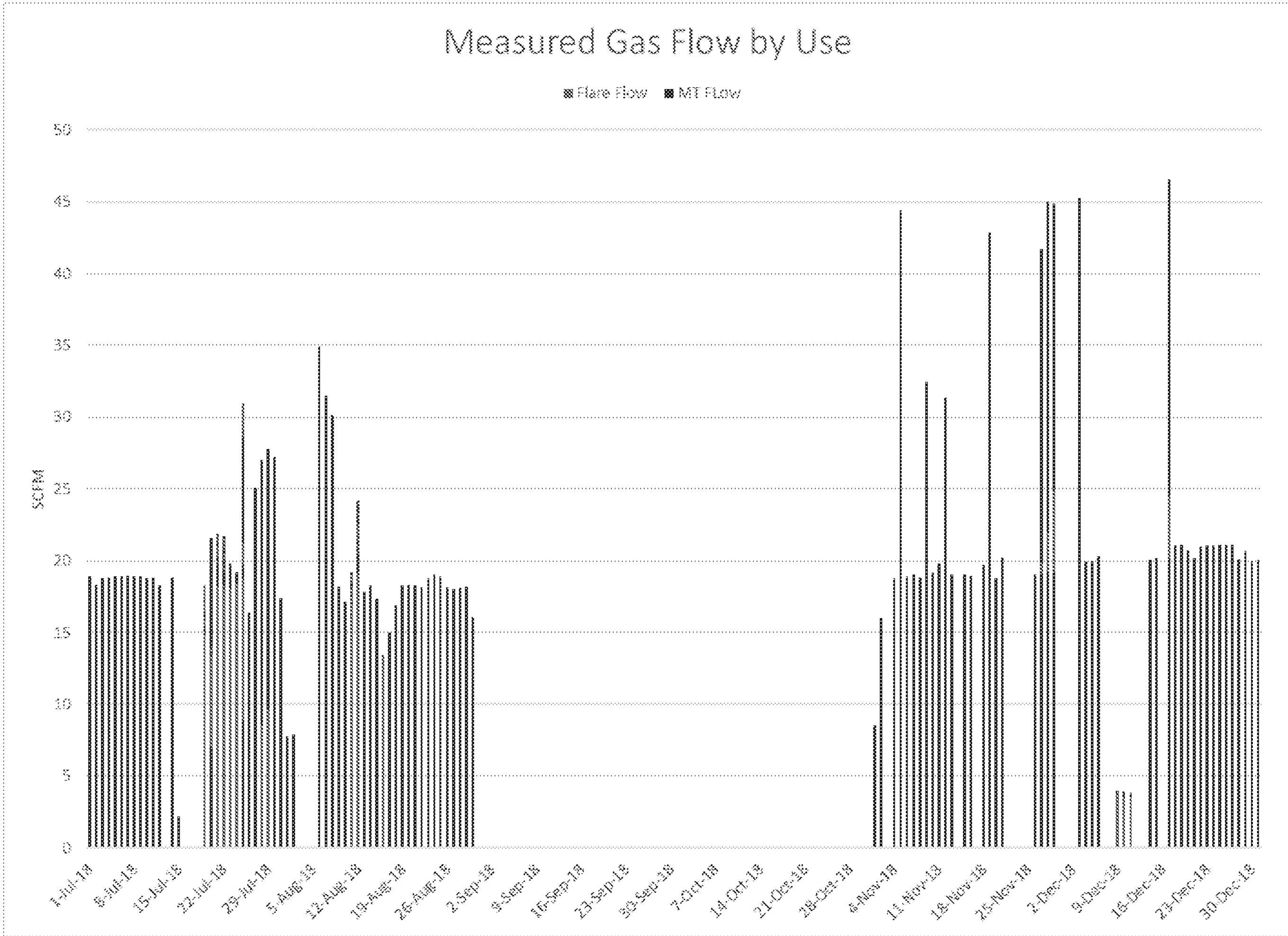

Figure 3. Measured Biogas Flow and Flare Use (July 1, 2018-December 31, 2018)

The above graph illustrates the measured biogas usage for the system. Flare usage, as indicated by measured flow to the flare meter, for the reporting period may also be surmised from the graph. It should be noted that days that indicate zero flow may sometimes indicate a disruption with the data acquisition system. As described previously, system operations had periodic downtime(s) in September and October, which correlates directly with the microturbine output above. After system repairs replacing like parts during these months, the microturbine output was improved and remained high for the months of November and December. The hog barns were also full during those months, compared with only half full in the early months of this report, which may help to explain the increase in electricity generation (kWh's).

The volume of gas is measured in Standard cubic feet per minute (SCFM). During normal operations, July through the beginning of September, the system was fairly consistent in the 18-19 SCFM range, with a little better flow in the months of November through December months averaging between 19 and 21 SCFM, with a few intermittent spikes on days of very high gas flow reaching up to 45 SCFM.

Overview of System Maintenance and Repairs

Overall, the operating biogas system and the environmental treatment system performed well, and remained under compliance. All maintenance exceptions appear in the log below, as maintained and recorded physically in the **Loyd Ray Farms Inspection and Operation Log Sheets**. While remote monitoring occurs on a daily basis, those activities are not normally captured in the report. We will note here only occurrences which required a site visit to resolve, or how the technicians would troubleshoot any problems that arose. If a system alert precipitated a site visit, we have indicated how the monitoring team went about troubleshooting the problem, and logged the experience required to make the corrections. Oftentimes, Cavanaugh's team was able to resolve the issue, but if a representative from either Unison, the biogas skid provider, or another service technician, such as an electrician, was required for further assessment or repairs, we have also noted the dates of their presence, how they troubleshooted the problem, and if replacement, new or rebuilt parts were required. Please note that the system required a little more frequent servicing as some of the parts in this pioneering system commissioned in 2011 are approaching their expected service life, but most of the service activities are viewed as normal Operations and Maintenance (O&M), and in all instances, no system parts were added to normal operations.

In summary, most of the maintenance activities during this period were gas conditioning skid shutdowns, which required on-site inspections or manual restarts. Please note that at the beginning of this semi-annual report only five of the nine barns were filled with hogs, which affected the biogas produced. Some severe weather occurrences, such as heavy rains required surface water pumping, oftentimes necessitating a reset of the system. While one occurrence, at first was diagnosed to be a bad transformer, but after further investigation, it turned out to be a converter issue. The electrical service technician determined the converter could be rebuilt. Considering the significant rainfall in the state following Hurricanes Michael and Florence, the storms themselves did not particularly disrupt the routine flow of biogas, but sometimes required more frequent pumping of surface water. On occasion, back-flushing of the system was required, along with other remedies to unclog the digester pump, if the bilge pump failed. During the months of November and December, when all the barns were completely full, gas was flowing steadily, and the rate at which the biogas was accumulated and stored beneath the HDPE cover exceeded the capacity of the microturbine, and the flare was used to augment biogas use. A necessary improvement was the installation of a new camera to replace the older model, to enhance images, and provide a clearer picture for remote observances and monitoring. It should also be noted that since this system's installation in 2011, many improvements in technology and instrumentation have developed.

The summary of the detailed operations log of on-site activities and monitoring for the period of July 1, 2018 through December 31, 2019 is presented as follows:

Date	Observation
7-2-2018	Monitored system remotely
7-25-2018	Site Visit to meet with Alex Gusnes of E-finity. We serviced the Microturbine (MT) and replaced air filter and the faulty fan we had been running. We found a faulty Rosemount meter which was registering incorrectly going to the MT. Also pumped surface water and did a site inspection.
7-27-2018	After remote monitoring, did a site visit. Pumped surface water and did a walk around site check. Turned flare on with 10 CMF going to flare.
7-30-2018	After remote monitoring, did site visit. Pumped surface water and performed site check. Flare off the balloon is getting low, as 5 of the 9 hog farms are empty. I installed a temporary

	cover for gas MH and dug a small ditch to help divert water away from the MH. Most operations automatic, with the exception of flush pumps by hand.
7-31-2018	Remote system monitoring during storms and heavy rainfall.
8-1-2018	Remote system monitoring, then site visit to inspect and review storm damage. After 6 inches of rain, worked all day with all entities to try and restore system back to normal operations. Monitored operations after storm damage trying to keep it running
8-2-2018	Remote system monitoring, then site visit to review storm damage. After 6 inches of rain, worked all day with all entities to try and restore system back to normal operations. Monitored operations after storm damage trying to keep operations running in stable mode.
8-3-2018	Monitored system remotely, checking to make sure post storm operations are normal.
8-6-2018	Monitored system remotely, then site visit to try and start the system. The gas balloon is growing, I started the flare, and pumped surface water and nursed the system to run. It failed twice due to heat, the outside temperature was in the 90's today.
8-7-2018	Monitored system remotely, then site visit to restart and monitor the operations. I shut the Flare off, as the balloon reach the level needed to shed rain. Pumped surface water and nursed the system to run. Communicated with reps at Unison and E-finity. A technician from E-finity is scheduled to be here on Thursday.
8-11-2018	Monitored system remotely, then site visit to do a restart after skid shut down because the MT would not start remotely. I ran a test and found that the MT failed to restart automatically after a skid shut down, so I started the flare. I will do a follow-up email with Nick at Unison.
8-13-2018	Site visit to do a restart of system with E-finity. We had a good start up, but now a skid warning for 33/342 reheat temp at 2:45 p.m. Will do a follow up email to Nick at Unison.
8-14-2018	Site visit to do a restart of system with E-finity. After the heat up at 2:45 p.m. yesterday, the skid was restarted, but the MT would not start, so I shut it down. I have restarted the skid today and will get E-finity to unblock the MT, and re-start it. Did a walk around inspected the system and started the auto pump for surface water.
8-16-2018	Monitored system remotely, Site inspection. Started the surface water pump. During the night, the Flare would not start. I turned on the mail at the MT, and it ran fine. I took influent, digester and effluent samples.
8-19-2018	Monitored system remotely, we had a shut-down today (Sunday). This was due to the outside temperature, after it cooled, I accomplished a successful restart of both the skid and the MT.
8-20-2018	Walk around inspection and site visit, operations are working normally.
8-21-2018	Monitoring system remotely, experienced a couple of shut downs, but was able to restart remotely.
8-22-2018	Monitoring system remotely, experienced a couple of shut downs, but was able to get back on line remotely.

8-23-2018	Site visit to meet with Matt Arsenault, Alex Gusner of Duke University, and Sarah Lanier (a student there). We took samples from the Lagoon Basin and Digester and also gas samples. I performed a site inspection and restarted the automatic pump and another non-automatic pump to handle surface water on the cover. System operations are normal.
8-27-2018	Site visit to do an On-site inspection, system and ground check. I worked on the camera with little success, system operating normally.
8-29-2018	After remote monitoring, did a site visit, tried to adjust the camera, system operations are running normally.
8-30-2018	System monitored remotely, then Site visit to do a system and ground check. Found the Unison system down. I tried to hard boot it, with no success. Unison is scheduled to be here on 9-10-2018, and I will call Unison to discuss.
9-5-2018	Remote monitoring this week.
9-10-2018	Site Visit. Met Marty Kass of Unison there to do service work. Flare is burning gravity gas. We found out we had no power. I called Salem Electric to do an emergency visit. They think the transformer is bad, and are checking on a source for a replacement one.
9-11-2018	We are still without power. Marty Kass of Unison, and Keith and Bryan from ProPump were on site, and I asked them to assess the no-electricity situation. They found the phase converter was bad, which showed like a bad transformer. They took down the two-phase converters and will ship them off to be rebuilt. They are also troubleshooting to change the flush pump from 3-phase to single phase, and are working to rebuild the IT. Marty Kass of Unison could not finish his service, and went to another job close by.
9-12-2018	ProPump returned with the converter rig for the flush pump and wired it inside the building to the pump with Kevin's help. We were able to get it back online and were able to flush.
9-13-2018	Flare is burning gravity gas. Kevin and Marvin worked to unclog the digester pump, but it is still clogged.
9-14-2018	Flare is burning gravity gas. Site visit to monitor operations and to prep for upcoming Tropical Storm Florence, which may be a hurricane.
9-15-2018	Monitored system remotely, flare is burning gravity gas.
9-15-2018	Monitored system remotely, then site visit to check system and water levels, flare is burning gravity gas.
9-16-2018	Remote monitoring, Flare is burning gravity gas, Site visit to check system and water levels.
9-17-2018	Remote monitoring, Flare is burning gravity gas
9-18-2018	After remote monitoring, went to Site to do a system and ground check. Found digester pump still clogged, tried to back flush system, but was not able to get valves open. The balloon is growing so I vented for one hour. The auto bilge pump failed, so I pumped surface water with two pumps for two hours.
9-22-2018	Remote monitoring, then site visit to check gas levels. Still flaring, but only had to vent once.
9-23-2018	Monitored system remotely, No Site visit today. System is still flaring, but not venting, only once.

9-24-2028	Monitored system remotely, No Site visit today. System is still flaring, but not venting, only once.
9-25-2018	Site visit to do a system and ground check, and found the digester pump still clogged, tried to backflush to see if I could unclog. I am still flaring but vented only once. The auto bilge pump failed so I pumped surface water with two pumps for the entire visit.
9-26-2018	Site visit to do a system and ground check, and found the digester pump still clogged, tried to backflush to see if I could unclog. I am still flaring but vented only once. The auto bilge pump failed so I pumped surface water with two pumps for the entire visit.
9-27-2018	Site visit to do a system and ground check, and found the digester pump still clogged, tried to backflush to see if I could break it free. I finally got the Digester pump to work, and plan to let it run all night to get it cleaned out. Still flaring the gas, vented only once. The auto bilge pump still failing, so I pumped surface water with two pumps for the entire visit.
9-28-2018	Site visit to do a system and ground check. Found the digester pump still working, so I moved it to the auto cycle. Pumped surface water during the site visit, vented at 2 ports for two hours.
10-1-2018	Site visit for system and ground check. Found digester pump still working, so I kept it on the auto cycle. Pumped surface water during the site visit.
10-2-2018	Site visit for a system and ground check. Operations are normal and the digester pump, remains on auto cycle. Pumped surface water. Conducted a tour of Duke University students and professors who came to observe operations.
10-3-2018	Site visit for a system and ground check. Digester pump still operating correctly, remains on auto cycle. Pumped excess surface water during the site visit.
10-4-2018	Monitored system remotely, no Site visit today.
10-5-2018	Site Visit to do a system and ground check and found our digester pump still working so I kept it on the auto cycle. Pumped surface water during site visit. Vented at two ports for 2 hours
10-6 & 10-7	Monitored system remotely without incidence.
10-8-2018	Site Visit to do a system and ground check and found digester pump still working so I kept it on the auto cycle. Pumped surface water during site visit. Met with Josh Amon to get the repaired Digester Pump installed. We worked on getting pumps unclogged, we are going to try to run as long as possible but not leave them unattended for a while, as sometimes they clog up and no fluid is being pumped.
10-9-2018	Site Visit for system and ground check, digester pump still working. Pumped surface water during site visit. I changed the timers and after the Digester pump restarted with a prime, I am going to try it through the evening.
10-10-2018	Site Visit to do a system and ground check and found our digester pump still working. Pumped surface water during site visit. I changed the timers and after the Digester pump restarted with a prime I am going to try it through another evening.
10-11-2018	Pumped surface water during site visit. I changed the timers and after the Digester pump restarted with a prime I am going to try it through another evening. Heavy rains from

	Michael with some flooding in the ditch. Mr. Bryant not happy with the ditch. Lost power for an hour or so all back running and seeing breaks in the clouds
10-12-2018	Pumped surface water during site visit. I changed the timers and after the Digester pump restarted with a prime I am going to try it through another evening. The Flare continues to run on gravity gas flow of 8-10 CFM I needed to vent today at two ports for 2.5 hours
10-14-2018	Monitored system remotely, particularly the flare.
10-15-2018	Site Visit to do a system and ground check and found our digester pump still working. Pumped surface water during site visit. The timers are working well with the restart of the Digester pumps. I am going to leave them as they are for now. The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018.
10-17-2018	Pumped surface water during site visit. The timers are working well with the restart of the Digester pumps. I am going to leave them as they are for now. The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018.
10-18-2018	Remote monitoring
10-19-2018	Pumped surface water during site visit. The timers are working well with the restart of the Digester pumps. I am going to leave them as they are for now. I worked on Drainage some. The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018.
10-21-2018	The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018.
10-22-2018	Site Visit to do a system and ground check and found our digester pump still working. Timers continue to work well, worked on drainage more. No venting since Friday 10-12-2018.
10-23-2018	Site Visit, no changes since yesterday.
10-24-2018	Site Visit, Pumped surface water during site visit. The timers are working well with the restart of the Digester pumps. The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018.
10-25-2018	Site Visit, no change since yesterday.
10-26-2018	Site Visit needed to check system the team viewer was not working dependably we need to install cameras ASAP to save on visits. Timers still operating correctly to restart pumps. No venting since 10-12-2018.
10-28-2018	Monitored system remotely.
10-29-2018	The Flare continues to run on gravity gas flow of 8-10 CFM. No venting since Friday 10-12-2018. Technicians from ProPump were on site installing the Phase converters and setting up for the wiring changeover of Flush Pump from 1 Phase back to 3 Phase
10-30-2018	ProPump was on site installing the Phase converters and setting up for the wiring changeover of Flush Pump from 1 Phase back to 3 Phase, Kevin Harward joined us to assist. We put boat in Basin for the wiring change on the Flush Pump.
10-31-2018	Site visit, Kevin Harward came to the farm to install a power part on the Unison system. We attempted to start the system, but the chiller had a failure and would not start. We are trying to get help resolving the problem. ProPump will probably need to come back and help with

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	the SCADA. Kevin and I moved the hose to push some of the digester sludge water to the Lagoon
11-1-2018	Site visit to accomplish a manual restart after an overnight failure, then working with folks from ProPump, trying to get SCADA to communicate with the Unison skid. The MT and skid were running again and no flare. Talked to ProPump via phone to set up for the SCADA repair.
11-3-2018	Steve Cavanaugh made a Site visit to start the conditioner. The MT failed after several tries it shut down.
11-4-2018	Site visit to start conditioner and MT. I found conditioner running but the MT not running and SCADA not recording properly. The MT failed after several tries it shut down. I then did a hard boot and it has been running since 11:05 AM. The MT is producing 59.6 output 54.9 on 18.3 CFM. I will monitor and keep records of output until SCADA can be fixed.
11-5-2018	Bryan from ProPump came to farm to work on SCADA, I was able to talk him through a restart. He had to drain the water from the gas pump on the south end of the skid and then the skid would start. I made a site visit to meet with Bryan. We were able to get the Skid and SCADA communicating and we are now running full bore. We have a lot of gas and I plan to stay as long as possible running both flare wide open; and the MT wide open burning about 50 CFM. I shut the Flare off at 4:30 PM
11-7-2018	I turned the flare off before I left on Monday and monitored remotely all-day Tuesday. Site visit today, the gas is still up, and the MT has been running since Monday.
11-8-2018	Site visit today, had a power blip that shut off team viewer, when I got to the site there were no alarms and the computer was back up the skid was just sitting there and not running, and the MT was in standby mode. Started the skid and when it was ready and sending to the MT; the MT would not start it was on, but not starting. I had to shut-off the breaker as before and when I turned it back on the MT started automatically. The gas volume is still up, and I will return tomorrow, and we may need to flare. Eight of the 9 hog houses are full of animals, loaded one out just now leaving the 8. The MT has been running since Monday,
11-9-2018	Site visit today, we have been running up until around 12:33PM we had a skid fault of high condensate at 741. I reset and restarted skid and the MT came on as it is supposed to at 2:00 PM. I started the flare to run while I am on site as the gas volume is still up. I did a walk around and up on the cover all is well. I installed a replacement fridge today. I received a new camera and will try and install it next week.
11-10-2018	Remote monitoring all day; we had three shutdowns and then late we had a MT fault. No site visit

11-11-2018	Site visit to restart the MT, as remote monitoring indicated a fault on the screen and the breaker had tripped. I had cut the skid off when the MT was in fault, so I restarted it, reset the breaker and opened the Flare valve. Everything restarted as it should have.
11-12-2018	Site visit to restart the MT I found a fault showing on the screen and the breaker had tripped. I had cut the skid off when the MT was in fault, so I restarted it, reset the breaker and opened the Flare valve. Everything restarted as it should have. This is the same as yesterday. I started the Gravity-flow Flare and it is running 10+ CFM, even though it does not seem to register on SCADA
11-13-2018	Remote monitoring, shutdown today, will do a site visit tomorrow.
11-14-2018	On Site Visit, I started the Gravity Flow Flare and it is running 10+ CFM even though it does not seem to register on SCADA and I am glad because we had a shut down on Tuesday.
11-15-2018	Site visit to restart system it refused to restart, but after 3 tries, I finally got the system restarted.
11-16-2018	Site Visit, due to shut-downs this week. I started the Gravity Flow Flare and it is running 10+ CFM even though it does not seem to register on SCADA and I am glad because we had a shut down on Tuesday and Wednesday and another during the evening on Thursday. Site visit to restart system and I found that the skid is not communicating with the SCADA and I am unable to start and stop or monitor skid data. I was able to re-start the skid and the flare continues to burn on gravity as above. The MT started as it should, and is running fine. I will monitor but if we shut down then it will be Sunday before I can manually restart. We need to burn all the gas that we can, the volume is high.
11-18-2018	During the evening on Thursday and again after site visit on Friday we had shut-downs. Monitored off and on Saturday, Flare burned at 10+CFM all the time. We need to burn all the gas that we can the volume is high. Site visit to try and restart system I had to do a hard boot of the Skid and the MT before I could get the System to run properly. When I did the hard boot on the Skid the communication with SCADA came back?? We are up and running again.
11-19-2018	Site visit to try and restart system Kevin restarted and was able to re-establish the communication Skid to SCADA by resetting at the panel several times. The MT started as it should at 12:52 PM. We started the flare through the conditioner and opened 2 vents at 1:45 PM. We had a shutdown at 3:16 PM and a quick restart. We shut the vents off at 3:45 venting for 2 hours. I cut the flare off coming through the Skid and restarted the Gravity Flow Flare and it is running 10+ CFM.

11-20-2018	Kevin and Marvin did a Site visit and took quarterly water samples. Had to reset the communications on the Unison panel as it shut down yesterday, acting like power is lost on the panel or something is going bad. Able to restart Skid and MT at 10:30a.m. Contacted Unison to let them know the issues hopefully get it fixed and or schedule a site visit soon. Think we have 2 bad level switches on the skid, they keep tripping-off and on, for 5- 30 seconds, once they are on for 30 seconds, the alarm is tripped, one is a high-level switch and is causing a shut down, again will let Unison know. We installed the new camera and set it up on team viewer. We did a walk around and up on top to check for leaks
11-21-2018	We did a site visit to restart same problem shutdown for condensate that is not there and faults out, so we cannot restart remotely but have to go to site to manually restart. Gravity Flare is burning at 10+ CFM
11-22-2018	Monitored system remotely most of week, trying to burn all the gas we can while the volume is high.
11-26-2018	Had to reset the communications on the Unison panel which shut down yesterday, acting like power is lost on the panel or something is going bad, was able to restart Skid and MT at 4:00PM. I emailed Unison to try to troubleshoot site issues and requested a site visit hopefully to get it fixed as soon as possible. Think we have 2 bad level switches on the skid, they keep tripping off. We did a site visit to restart, without success. The same problem occurred; shutdown for condensate that is not there, and the system short circuits, or faults out, disrupting the normal flow of the system, so we cannot restart remotely but must visit the site manually to restart. Gravity Flare is burning at 10+ CFM Started venting at two vents at 4:05 PM and closed them at 5:05 PM. By the time I got home at 8 it shut down.
11-27-2018	Monitored all during the night to see if flare was continuing to burn at 10 CFM. Site visit today to restart the system. I shut off the Gravity Flare at 11:00 AM and opened the valve and flared with gas through the skid at feed=28.4 and flow = 21.4 CFM. The skid is running with the fault light showing on SCADA, but the MT and skid are running full. Every shut down, or system failure, is requiring an on-site visit. At 2:00 PM I went back to Gravity Flare at 10+ CFM. The skid and MT have been running 4 hours. The red fault light is still showing on SCADA, but the system is running, and it will continue to fault out. We need to burn gas and make KWs. System shutdown at 3:10p.m. Flare continued to burn at 10+ CFM.
11-28-2018	Monitored all during the night to see if flare was continuing to burn at 10 CFM. Site visit today to restart the system. I shut off the Gravity Flare at 2:38p.m. and opened the valve and flared with gas through the skid at feed=30.4 and flow = 25.4 CFM. The skid is running with the fault light showing on SCADA, but the MT and skid are running full. At 2:00p.m., I shut

MB018704

	the system down and did a hard boot and this time the fault light on SCADA picture of the skid went off and the Unison screen started registering data. At 2:08 p.m., we are running full bore. Back to Gravity Flare at 10+ CFM. The skid and MT have been running 4 hours. The red fault light is still showing on SCADA, but the system is running, and after troubleshooting I discovered we need to burn gas and make KWs. System shutdown at 3:10p.m., re-fired at 4:10p.m., shutdown@5:27p.m., restart at 8:57-shutdown@11:57p.m. Flare continued to burn at 10+ CFM.
11-30 through 12-2-2018	Monitored all during the day and night to see if flare was continuing to burn at 10 CFM
12-3-2018	Monitored all during the night to see if flare was continuing to burn at 10 CFM. Site visit to restart system started at 11:45a.m. --Flared using skid at 24CFM 12:45p.m. until 3:50p.m. Reset gravity Flare at 10+ CFM for the night.
12-4-2018	Monitored during the night to see if flare was continuing to burn at 10 CFM. Site visit to meet with tech from Unison. Worked with Curt Schiesl of Unison to try to resolve the problem with the skid. I shut the flare off at 9:00a.m. He changed out switches and tried all kinds of things to keep it running. He had to order parts shipped overnight, and will continue troubleshooting tomorrow.
12-5-2018	Monitored system with Curt Schiesl, Field Service and Start-up Technician for Unison by computer and phone as he continued to try and fix the problem with the skid. He left for his home stating that he thought the problem was that the Phase converters were overheating. We had a shut down and panel fault as before.
12-6-2018	Site visit to restart the system and found we had a shutdown but no loss of power to panel, it just faulted as before. All I had to do to start the skid and MT running was to press the start button. I still do not have any data on skid panel screen, but we are running. We had a shutdown and showing no power to Unison panel. I did a hard boot to PC and after a short pause the Unison panel lit up with information, it ran for about 30 minutes and shutdown still showing power to the Unison panel. I restarted without any numbers and it is running; if and when we have a shutdown, it will have to be restarted by onsite visit. I started the Gravity Flare burning at 10+ CFM and plan for it to run until Monday regardless of what the Skid and/or the MT does.
12-11-2018	Site visit to restart the system and found we had a shutdown, but no loss of power to panel. The Gravity Flare has been burning at 10+ CFM continuously since I left on 12-06. I met with Norman and Bryan of ProPump and plan for it to run until Monday regardless of what the Skid and or the MT does.

12-12-2018	Site visit, met with ProPump and we continued to troubleshoot along with Doug from Unison.
12-13-2018	Site visit to restart the system and found we had a shutdown, but no loss of power to panel. The Gravity Flare has been burning at 10+ CFM continuously since I left on 12-06. Met with Bryan from ProPump and we continued to troubleshoot along with Doug from Unison. We added some new parts and it seemed to be fixed. Then in the evening we continued to have shut downs, Flare still running.
12-14-2018	The Gravity Flare has been burning at 10+ CFM continuously since I left on 12-06. Bryan from ProPump came to site and he installed a part and we were running. I monitored and sent text to Norman and Bryan of ProPump, and Doug from Unison. We added some new parts and it seemed to be fixed. Then in the evening we continued to have shut downs. Flare still running. I monitored all weekend during that time I lost communication due to a power Failure by Surry-Yadkin, Flare continued to burn.
12-17-2018	Site visit to restart the system and found we had a shutdown but no loss of power to panel. The Gravity Flare continued to burn all weekend. At 10.0+ CFM. I met with Bryan of ProPump and we spent the day troubleshooting system with concentration on Phase converter. With the help of a conventional fan we were able to cool Phase converter enough to run until we could get parts to repair. Started running at 10:50a.m. We shut the gravity flare off on the restart of the Skid and MT and ran the flare hard until 3:15 PM.
12-18-2018	Monitored system remotely by SCADA and Camera The system has been running from 11:00a.m. Monday without a shut down. Gravity Flare is off.
12-19-2018	Site visit to do a system check, the parts did not arrive, so after the inspection and repair of a small leak, I traveled home to return tomorrow. Gravity Flare is off.
12-20-2018	Site visit to do a system check. I met with Bryan of ProPump and he installed fans and circuit boards in Phase converter. We restarted system and we are up and running. The Gravity Flare is off.
12-21-2018	Site visit to do a system check. I met with Matt Arsenault of Duke U. We have been running solid since we replaced PC Fans yesterday. The Gravity Flare is off.
12-30-2018	Site visit to do a system inspection. The Gravity Flare is off. System was working but computer was down.
12-31-2018	Site visit to do a system check The Gravity Flare is off. System was working but computer was down again. Rebooted it again Checked Team Viewer with Nancy at home.

The following table lists the compliance requirements as per the permit for the subject system, and the performance / compliance relative to each requirement:

	Description of Monitoring Requirement	Status	Result
1	Maintenance of adequate records by Permittee to track the amount of sludge/separated solids disposed.	N/A	No solids or sludge disposal occurred during the reporting period; some sludge returned to the anaerobic digester for further breakdown in accordance with the Division approved Operations & Maintenance Plan.
2	Inspection of entire Innovative System waste collection, treatment, and storage structures and runoff control measures at a frequency to insure proper operation but at least monthly and after all storm events of greater than one (1) inch in 24 hours; Permittee maintenance of inspection log or summary including at least the date and time of inspection, observations made, and any maintenance, repairs, or corrective actions taken by Permittee.	☑	Inspections and observations conducted by representatives of Loyd Ray Farms, Inc., Cavanaugh & Associates, P.A., and DCOI. Observations recorded, and actions taken to adjust the operation of the System are recorded in log book kept onsite, and emailed in.
3	Maintenance of a log of all operational changes made to the Innovative System including at least the process parameter that was changed, date and time of the change, reason for the change, and all observations made both at the time of the change and subsequently as a result of the change by Permittee/ Permittee's designee.	☑	Log book entries, as described in item #2, above, maintained on site; copies attached to report (Appendix A).
4	Representative Standard Soil Fertility Analysis to be conducted annually on each application site receiving animal waste.	☑	NCDA&CS Agronomic Division Report No. FY19-SL009269, shows the results of the Predictive Home & Garden Soil Report for Loyd Ray Farms. The samples were compiled on 10/22/2018, and were completed on 11/01/2018, which are added to this report, they can also be accessed here: http://www.ncagr.gov/agronomi/
	Wastewater Analysis		
5	Quarterly tests shall be conducted once within each of the following windows w/ at least sixty (60) days between any 2 sampling events. Water quality samples include analysis of copper, zinc, total suspended solids, pH, total nitrogen, TKN, NO2 + NO3, phosphorus, ammonia, and fecal coliform.		
	Quarter 3 (July 1 – September 30)	☑	Sample Collected: 8/17/2018 Sample Analyzed: 9/18/2018 Results Reported: 9/18/2018

			Results included in the attached report from Research & Analytical Laboratories, Inc. (Appendix B)
	Quarter 4 (October 1 – December 31)	☑	Sample Collected: 11/20/2018 Sample Analyzed: 12/20/2018 Results Reported: 12/20/2018 Retest of Fecal Coliform: Sample Collected: 1/9/2019 Sample Analyzed: 1/11/2019 Sample Reported: 1/11/2019 Results included in the attached report from Research & Analytical Laboratories, Inc. (Appendix B)
Ambient Air Sampling			
	Fall Season Ambient Air Sampling	☑	A fall season ambient air sample taken on September 25, 2018. Results included in the Explanation of Results and Sampling Methods.
	Waste Treatment and Storage System	☑	
	Barns	☑	
	Sprayfields	☐	
	Winter Season Ambient Air Sampling	☑	A second ambient air sample (winter analysis) was completed on December 19th, by Duke University, and has been added to the report. Results included in the attached Explanation of Results and Sampling Methods.
	Waste Treatment System	☑	
	Barn Exhaust	☑	
	Sprayfields	☐	
Odor Sampling			
6	Permittee shall monitor for odor compliance quarterly at both upwind and downwind locations on the property boundary. Permittee shall document monitoring locations on a site map, indicating prevailing wind direction, for each monitoring event.		
	Quarter 3 (July 1 – September 30)	☑	Odor sampled by Duke University Explanation of Results and Sampling Methods. September 25, 2018.

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7	Quarter 4 (October 1 -December 31)	☑	Odor sampled by Duke University Explanation of Results and Sampling Methods. December 19, 2018.
		Record Keeping	
8	All records, including operation, maintenance, and repair records, shall be maintained on site and in chronological and legible form for a minimum of five (5) years by the Permittee; records shall be maintained on forms provided by or approved by the Division and shall be readily available for inspection.	☑	A copy of the report and all monitoring records are maintained in a binder in the System Control Building; the electronic form combines inspection and operations records on a single form, entitled "Loyd Ray Farms Inspection, Operations & Maintenance Log Sheet" which are being collected electronically, and submitted to the Regional Office via email.

EXPLANATION OF RESULTS AND SAMPLING METHODS

1. Amount of Sludge or Separated Solids Disposed

N/A. No disposal of sludge or separated solids was required from the Innovative System during the 7/1/2018-12/31/2018 reporting period. Some sludge was returned from the aeration basin to the anaerobic digester for further breakdown, as per usual and typical operations, in accordance with the design and Operation and Maintenance Manual.8

2. Log of System Inspections

See Operator Log Book, Appendix A. (digitally attached)

3. Log of Operational Changes to the Innovative System

See Operator Log Book, Appendix A. (digitally attached)

4. Results of Standard Soil Fertility Analysis

Two separate reports, by NCDA & CS Agronomic Division, analyze independent soil samples which were taken at Loyd Ray Farms on October 22, 2018, as stated in the reports completed on November 1, 2018. The actual test results and recommendations for each sample can be found in Appendix C. The following tables are compiled to easily view the aggregated results.

Loyd Ray Farms Report No. FY19-SL009268

	Sample #	1A 01	1B 02	IC03	3A	3B	3C	9B
HM	Percent humic matter	0.41	0.41	0.41	0.41	0.36	0.42	0.51
W/V	Weight per volume	1.12	1.15	1.16	1.08	1.10	1.09	1.08
1	Cation exchange capacity	7.2	7.4	7.6	9.8	9.4	10.2	9.4
9	Manganese Index	165	167	172	119	120	140	232
Zn-1	Zinc Index	782	486	443	542	492	629	499
Cu-I	Copper Index	135	143	137	102	106	128	110
128	Sulfur Index	32	32	32	31	32	34	33

P-1	**Phosphorus Index**	**53**	**57**	**56**	**84**	**91**	**92**	**86**
K-1	**Potassium Index**	41	39	41	**474**	50	**363**	**345**
pH	**Acidity**	**6.4**	**7.2**	**7.2**	**7.2**	**7.2**	**7.0**	**6.7**

Loyd Ray Farms Report No. **FY19-SL009269**

	Sample #	2	4	5	6	7	8	9A	9B
HM	Percent humic matter	0.41	0.41	0.46	0.41	0.51	0.36	0.51	0.51
W/V	Weight per volume	1.07	1.08	1.10	1.04	1.07	1.02	1.07	1.10
CEC	Cation exchange capacity	10.7	10.7	12.1	11.2	9.9	11.4	9.0	8.8
Mn-I	Manganese Index	153	150	131	162	198	166	222	224
Zn-1	Zinc Index	616	608	1601	735	569	696	475	445
Cu-I	Copper Index	147	141	128	121	105	114	108	108
S-I	Sulfur Index	36	35	36	35	32	33	32	29
P-1	**Phosphorus Index**	**87**	**89**	**116**	**90**	**88**	**91**	**85**	**83**
K-1	**Potassium Index**	**535**	**539**	**643**	**574**	**595**	**618**	**334**	**351**
pH	**Acidity**	**7.1**	**7.1**	**7.2**	**7.2**	**7.3**	**7.3**	**6.8**	**6.8**

In almost all samples, the Phosphorus Index (P-I) and Potassium Index (K-I), were found to be *Above Optimum*. The range for *Optimum* is between 50 and 70, Sample 1B 02, and 1C 03 were a little below the *Optimum* range, but all others were very desirable. All of the samples in the 0269 group, in bold lettering above, exceeded those limits, and were *Above Optimum.* All of the samples in the 0268 group were at least at *Optimum* level, and many of which were in the *Above Optimum* range. The pH test for acidity results were higher than the 5.8 to 6.5 *Optimum* range, averaging about 7.1 on Report #FY19-SL009269. Similarly, the Sample # FY19-SL009268, was also in the *Optimum* range, averaging about 6.9.
The exact agronomist's comments and recommendations for fertilizer application can be found in the Actual Soil Reports See **Appendix C.**

5. Results of Water and Air Quality Sampling

a. Results of Waste Water Analysis
Water quality samples were taken in each quarter, a synopsis of the results is found below. Samples were analyzed by Research Analytical Laboratories, Inc. in Kernersville. The 4th quarter sample of 2018, as well as 1st and 2nd quarters of 2018, resulted in higher fecal coliform counts than expected, and thus, additional samples were taken. While the re-sampling was done in January 2019, we have added it to the report for clarity. The following table compares the results of the water quality analysis of the final effluent from the Innovative System:

Parameter	8/17/2018	11/21/2018
TOT N	1080	972
TKN	1080	972
NO_2+NO_3	0.27	<.05

TP	62.2	215
NH_3-N	689	702
COPPER	0.088	0.334
ZINC	0.489	2.32
TS	848	1300
FECAL	1400000	33000000
pH	8.23	8.17

The fecal coliform count for most of the sampling events exceeded the permit limit, and this has not been resolved. Almost all other constituent parameters as recorded above are decreasing since the beginning of the year, as indicated in the final effluent recordings in the chart above. The chart above describes the waste water analyses that is required to be conducted on a quarterly basis. These parameters are: total N, NH_3-N, NO_3-N/NO_2-N, total P, % solids, copper, zinc, pH and pathogens. Samples are to be taken from the digester and the effluent (leaving the aeration basin). All sampling was conducted:

1. Given the resampling produced fecal coliform counts that were quite high compared to the permit limit, an additional resampling event was conducted on January 9, 2019. Again, a composite sample was obtained of the effluent, split into three (3) sample bottles, then sent to the laboratory for analysis. The results are as follows:

    1. Sample ID: Effluent #1; Fecal Coliform – MPN = 400000 MPN/100mL
    2. Sample ID: Effluent #2; Fecal Coliform – MPN = 1700000 MPN/100mL
    3. Sample ID: Effluent #3; Fecal Coliform – MPN = 330000 MPN/100mL

The results were much improved over the previous fecal coliform samples.

b. **The Results of Air Sampling**

Duke University's Dr. Marc Deshusses took Fall and Winter ambient Air Samples on September 25, 2018, and December 19th, 2018, respectively, the results of which are described below.

Odor Sampling #1

Odor was monitored to comply with Section I.6.b.ii of the Swine Animal Waste Management Permit. Two monitoring events occurred over this compliance period. The first monitoring event was conducted on September 25, 2018.

Sampling took place at about 11:30 am. It was an overcast day (70° F) with relatively steady wind (1.3-1.8 m/s). Several measurements for wind speed and direction were taken. The predominant wind direction and sampling points for odor were selected as shown in Figure 4.

Odor was monitored by Marc Deshusses, whose findings are reported here. Odor panelist rules were listed in the previous report and are not repeated here. Odor was monitored using a Nasal Ranger (http://www.nasalranger.com/) field olfactometer, following the manufacturer recommended instructions.

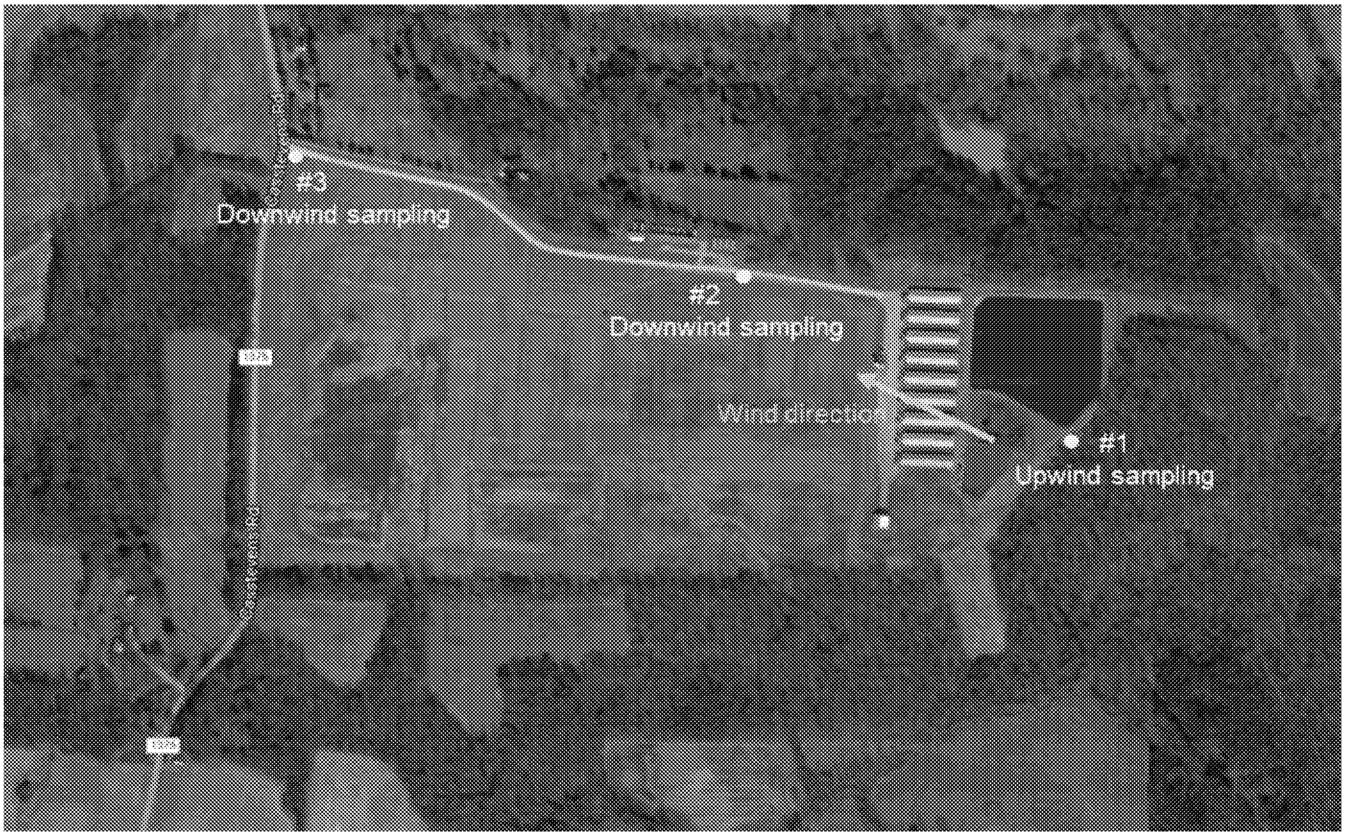


Figure 4. Aerial view of the facility and location of the monitoring points for odor for the June 26, 2018 sampling. The arrows indicate the prevailing wind direction the day of the sampling.

<u>Sampling upwind</u>
Odor could not be detected at the 2 D/T level. This indicates that the odor level was lower than 2 D/T. Then the Nasal Ranger was taken off the nose and ambient air was sniffed and compared to odorless air from the Nasal Ranger. This was to determine whether a difference could be detected between ambient air and odorless air from the Nasal Ranger. No significant difference could be detected.

<u>Sampling downwind</u>
Odor was measured at first downwind odor monitoring point (Downwind #2, see Figure 1). Odor could be detected at the 7 D/T level, but this location is still on the property. A second monitoring point was chosen at the property line, as specified in the permit (Downwind #3), no odor could be detected at the 2 D/T level at this location. This indicates that the odor level was lower than 2 D/T. Then the Nasal Ranger was taken off the nose and ambient air was sniffed and compared to odorless air from the Nasal Ranger. This was to determine whether a difference could be detected between ambient air and odorless air from the Nasal Ranger. No difference could be noticed.

These results indicate that odor levels complied with Section I.6.b.ii of the Swine Animal Waste Management Permit.

Odor Sampling #2

Odor was monitored to comply with Section I.6.b.ii of the Swine Animal Waste Management Permit. A second ambient air sampling was conducted on December 19, 2018, and monitored by Marc Deshusses.

Sampling took place at about 9:00 am. It was a partly overcast cold day (29° F) with quasi no wind (barely measurable to 0.3 m/s). Several measurements for wind speed and direction were taken. Although wind was quasi nil, the predominant wind direction and sampling points for odor were selected as shown in Figure 5.

Odor panelist rules were listed in the previous report and are not repeated here. Odor was monitored using a Nasal Ranger (http://www.nasalranger.com/) field olfactometer, following the manufacturer recommended instructions.

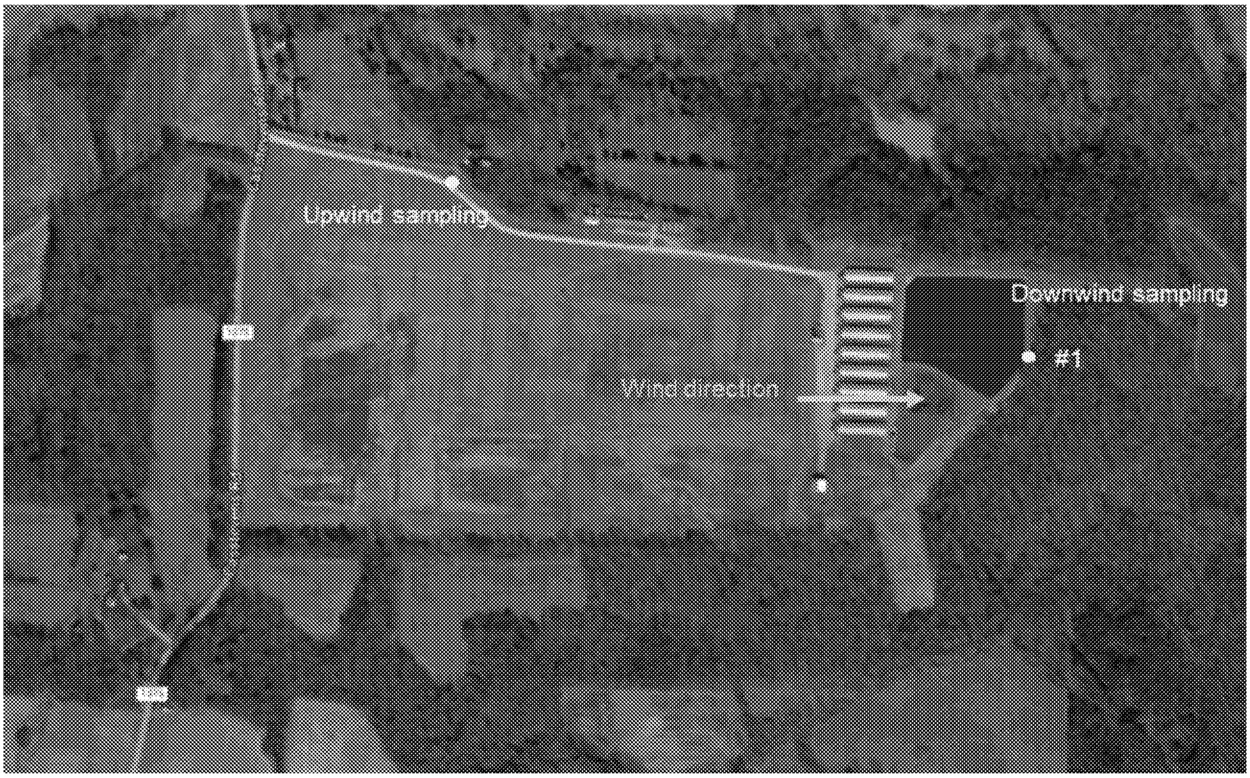


Figure 5. Aerial view of the facility and location of the monitoring points for odor for the June 26, 2018 sampling. The arrows indicate the prevailing wind direction the day of the sampling.

<u>Sampling upwind</u>
Odor could not be detected at the 2 D/T level. This indicates that the odor level was lower than 2 D/T. Then the Nasal Ranger was taken off the nose and ambient air was sniffed and compared to odorless air from the Nasal Ranger. This was to determine whether a difference could be detected between ambient air and odorless air from the Nasal Ranger. No significant difference could be detected.

Sampling downwind
No odor could be detected at the 2 D/T level at location #1. This indicates that the odor level was lower than 2 D/T. Then the Nasal Ranger was taken off the nose and ambient air was sniffed and compared to odorless air from the Nasal Ranger. This was to determine whether a difference could be detected between ambient air and odorless air from the Nasal Ranger. There was a faint piggery/barn odor with notes of ammonia, but as mentioned before these odors were below the 2 D/T level.

These results indicate that odor levels complied with Section I.6.b.ii of the Swine Animal Waste Management Permit. Ammonia nitrogen emissions from the aeration basin and lagoon were quantified to determine if significant volatilization of NH_3-N occurred from this part of the waste management system. Emissions from the water surfaces were determined using a buoyant convective flux chamber (BCFC) which method was described in detail and illustrated with pictures in the February 15, 2012 report.

Emissions from Animal Waste Treatment and Storage System – September 25, 2018

Ammonia nitrogen emissions from the aeration basin and lagoon were quantified to determine if significant volatilization of NH_3-N occurred from this part of the waste management system. Emissions from the water surfaces were determined using a buoyant convective flux chamber (BCFC) which method was described in detail and illustrated with pictures in the February 15, 2012 report. Sampling took place at about 12 pm. It was an overcast day (70° F) with relatively steady wind (1.3-1.8 m/s).

Results were as follows:

- Size of the chamber: 52.1 cm wide by 52.1 cm long and 2.5 cm in headspace height.
- Air sampling flow rate: 0.30 L/min
- **Average ammonia concentrations** in sweep air from the aeration basin while aeration was off: **114 ppm** (3 samples) or on average in mass concentration 0.064 g-N/m^3
- Ammonia concentrations in sweep air while aeration was on was not measured, earlier monitoring indicated that ammonia concentration in sweep air during aeration was slightly lower.

The total emission from the aeration basin can be calculated from the air sampling flow rate, the surface of the chamber and the surface area of the aeration basin. The latter surface is nominally 24,500 ft^2 (or 2277 m^2). Emission rate is calculated as follows:

NH_3 emission rate = NH_3 concentration × Sampling flow rate × Aeration basin area / Buoyant chamber area

After unit conversion, one obtains values of 9.7 g/h. This corresponds to a NH_3 emission rate of **1.63 kg NH_3-N/week.** This is a very low value compared to the **allowable emissions of 106 kg NH_3-N/week** from the swine waste treatment and storage structures as specified in Section I.6.a.i of the Swine Animal Waste Management Permit.

Surface emission rate of NH_3 from the **lagoon** was determined following the same method. Average concentration of ammonia in the sweep air (with the same chamber and at the same flowrate of 0.3 L/min) was 120 ppm (2 samples). With the surface area of the lagoon (19,425 m^2), emission of NH_3 from the lagoon are estimated to be **14.7 kg NH_3-N/week**.

Results for the emissions from the aeration basin and the lagoon are summarized in the table below. Total ammonia (TAN) in the aeration basin and lagoon at the time of sampling is also reported for information. These numbers show the system is performing as expected.

	Aeration basin	Lagoon
Surface area	2277 m^2	4.8 acres = 19,425 m^2
TAN	955 mg-N/L	215 mg-N/L
Emission rate	1.63 kg NH_3-N/week	14.7 kg NH_3-N/week
Total emission (lagoon + aeration basin)	**16.4 kg NH_3-N/week**	

Thus, together lagoon and aeration basin contribute to the emission of **16.4 kg NH_3-N/week**. This is well below the allowable 106 kg NH_3-N/week.

Emissions from the Barns

Ammonia emissions from the barns were also determined on September 25, 2018. It should be noted that accurate determination of emissions from animal houses is a difficult exercise. This is because of the variable nature of the emission, the difficulty in accurately measuring air flow from the fans on the animal houses, and the fact that fan operation is automated, i.e., they are turned on and off automatically triggered by a thermostat. Thus, uncertainties on the numbers reported below exist and can be important.

Ammonia in the exhaust air from the barns was determined using Draeger tubes. Details on the concentrations and number of fans on at the time of sampling are shown in the table below.

Barn	NH_3 Concentration (ppm)	Small Fans working	Large Fans working
1	4	1	1
2	1	0	2
3	1.5	1	1
4	0.9	1	0
5	2.7	0	2
6	1.5	1	2
7	5	1	1
8	4	1	2
9	3.5	1	1

The total emission of ammonia can be estimated by multiplying the ammonia concentration in each of the barn's exhausts by the exhaust flowrate of that barn (33,000 cfm for large fans and 13,000 cfm for the small fans). At the time of sampling, total exhaust flow was 487,000 cfm and concentrations ranged from 0.9 to 5 ppm (see Table above). The calculated total weekly ammonia emissions from the barns was **216 kg NH_3-N/week.**

Adding the emission from the treatment system and the lagoon (**16.4 kg NH_3-N/week**) to the emissions from the barns (371 kg NH_3-N/week) amounts to a **total of 232.1 kg NH_3-N/week** from the swine farm**.** This is below the allowable value of 476 kg NH_3-N/week specified in Section I.6.a.iii of the Swine Animal Waste Management Permit.

Emissions from Animal Waste Treatment and Storage System – December 19, 2018

Ammonia nitrogen emissions from the aeration basin and lagoon were quantified to determine if significant volatilization of NH_3-N occurred from this part of the waste management system. Emissions from the water surfaces were determined using a buoyant convective flux chamber (BCFC) which method was described in detail and illustrated with pictures in the February 15, 2012 report. Sampling took place on December 19, 2018 between 9:30 am and 11:00 am. It was a partly overcast cold day (29° F) with quasi no wind (barely measurable to 0.3 m/s). Temperature was 29° F.

Results were as follows:

- Size of the chamber: 52.1 cm wide by 52.1 cm long and 2.5 cm in headspace height.
- Air sampling flow rate: 0.30 L/min
- **Average ammonia concentrations** in sweep air from the aeration basin while aeration was off: **90 ppm** (3 samples) or on average in mass concentration 0.051 g-N/m^3
- Ammonia concentrations in sweep air while aeration was on was not measured, earlier monitoring indicated that ammonia concentration in sweep air during aeration was slightly lower.

The total emission from the aeration basin can be calculated from the air sampling flow rate, the surface of the chamber and the surface area of the aeration basin. The latter surface is nominally 24,500 ft^2 (or 2277 m^2). Emission rate is calculated as follows:

NH_3 emission rate = NH_3 concentration × Sampling flow rate × Aeration basin area / Buoyant chamber area

After unit conversion, one obtains values of 7.7 g/h. This corresponds to a NH_3 emission rate of **1.29 kg NH_3-N/week.** This is a very low value compared to the **allowable emissions of 106 kg NH_3-N/week** from the swine waste treatment and storage structures as specified in Section I.6.a.i of the Swine Animal Waste Management Permit.

Surface emission rate of NH_3 from the **lagoon** was determined following the same method. Average concentration of ammonia in the sweep air (with the same chamber and at the same flowrate of 0.3 L/min) was 65 ppm (2 samples). With the surface area of the lagoon (19,425 m^2), emission of NH_3 from the lagoon are estimated to be **7.97 kg NH_3-N/week**.

Results for the emissions from the aeration basin and the lagoon are summarized in the table below. Total ammonia (TAN) in the aeration basin and lagoon at the time of sampling is also reported for information. These numbers show the system is performing as expected.

	Aeration basin	**Lagoon**
Surface area	2277 m^2	4.8 acres = 19,425 m^2
TAN	1048 mg-N/L	277 mg-N/L
Emission rate	1.29 kg NH_3-N/week	7.97 kg NH_3-N/week
Total emission (lagoon + aeration basin)	**9.27 kg NH_3-N/week**	

Thus, together lagoon and aeration basin contribute to the emission of **9.27 kg NH_3-N/week**. This is well below the allowable 106 kg NH_3-N/week.

Emissions from the Barns

Ammonia emissions from the barns were also determined on December 19, 2018. It should be noted that accurate determination of emissions from animal houses is a difficult exercise. This is because of the variable nature of the emission, the difficulty in accurately measuring air flow from the fans on the animal houses, and the fact that fan operation is automated, i.e., they are turned on and off automatically triggered by a thermostat. Thus, uncertainties on the numbers reported below exist and can be important.

Ammonia in the exhaust air from the barns was determined using Draeger tubes. Details on the concentrations and number of fans on at the time of sampling are shown in the table below. It should be noted that at least one barn was empty, and that since it was a relatively cold day, several barns only had one fan operating, some fans where toggling on and off. Still a value of 1 fan on was used for these barns and thus emissions reported may be overestimated.

Barn	NH_3 Concentration (ppm)	Small Fans working	Large Fans working
1	5	1	0
2	18	1	0
3	not determined	0	0
4	8	1	0
5	8	1	0
6	12	1	0
7	11	1	0
8	9	1	0
9	15	1	0

The total emission of ammonia can be estimated by multiplying the ammonia concentration in each of the barn's exhausts by the exhaust flowrate of that barn (33,000 cfm for large fans and 13,000 cfm for the small fans). At the time of sampling, total exhaust flow was 104,000 cfm and concentrations ranged from 5 to 18 ppm (see Table above). The calculated total weekly ammonia emissions from the barns was **181 kg NH_3-N/week.**

Adding the emission from the treatment system and the lagoon (**9.27 kg NH_3-N/week**) to the emissions from the barns (371 kg NH_3-N/week) amounts to a **total of 190.0 kg NH_3-N/week** from the swine farm**.** This is below the allowable value of 476 kg NH_3-N/week specified in Section I.6.a.iii of the Swine Animal Waste Management Permit.

Additional Observations

The previous DWR Inspection mentioned that interior portion and aeration basin was eroding, and that vegetation should be established to control briars and other broadleaf weeds on embankments. The monitoring staff has been working with the Loyd Ray Farm staff to correct this situation, and has been actively seeding this area, with plans to re-seed again in Spring of 2019. Any undergrowth, including sprouting small trees or broadleaf weeds on the embankments are designated to be cut as soon as the ground can hold the equipment. The application fields appear to be well vegetated.

This Semi-annual Compliance Report is compiled and respectfully submitted by:

William G. "Gus" Simmons, Jr., P.E.
Cavanaugh & Associates, P.A.
1-877-557-8923

Attachments:

Appendix A – PDF of Actual log sheets
Appendix B – Sample Collection Dataset
Appendix C - Soil Report

APPENDIX A Operation and Log Sheets – Digitally Attached

APPENDIX B – Sample Collection Dataset (Digitally Attached)

APPENDIX C - NCDA&CS Agronomic Division Soils Report (Source: Website: www.ncagr.gov/agronomi/)

The NCDA&CS Agronomic Division Soils Report for two soil samples taken on 10/16/2018 follow.

Report No. FY19-SL009269 and **Report No.** FY19-SL009268, analyzed on 11/1/2018. We are sharing it here, as some of the data from this analysis was included in this report.

Report No. FY19-SL009269

NCDA&CS Agronomic Division   Phone: (919) 733-2655   Website: www.ncagr.gov/agronomi/   Report No. FY19-SL009269

Predictive Home & Garden

Soil Report

Mehlich-3 Extraction

Links to Helpful Information

Sampled: Received: 10/22/2018   Completed: 11/01/2018   Farm: 2094

Client: Loyd Bryant
Loyd Ray Farms Inc
2049 Center Rd.
Boonville, NC 27011

Sampled County : Yadkin
Client ID: 205223

Advisor:

Advisor ID:

Agronomist's Comments:

This report provides Test Results and Recommendations for each sample submitted for testing. Look for Lime Recommendations and N-P-K Fertilizer Recommendations. The lime recommendation is always listed next to the first crop and will be based on the higher target pH if the pH targets for crop 1 and crop 2 differ. Application at the indicated rate will raise soil pH to the optimal level for the plant you specified and should be sufficient for 2 to 3 years, depending on soil type. Common target pH values are as follows: 5.0 for azalea, camellia, rhododendron and mt. laurel; 5.5 for centipedegrass; 6.0 for other lawn grasses, shrubbery, and; flowering plants; and 6.5 for vegetable gardens. N-P-K Recommendations are based on the nitrogen (N) needs of the plants being grown and the soil test results for phosphorus (P-I) and potassium (K-I); a 50 to 70 index for either is optimum. If the exact fertilizer cannot be found, find the closest match and adjust the rate accordingly. Refer to "Understanding the Soil Report" (last page of this report) for additional explanation and links to helpful information.

Sample ID: 2

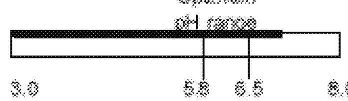
Lime History:

Loyd Bryant

	Lime Recommendations	N-P-K Fertilizer Recommendations *
Crop 1- Lawn (not centip.)	0.0 lb per 1,000 sq ft	5 lbs per 1000 sq ft 21-0-0 Group D
Crop 2-	0.0 lb per 1,000 sq ft	

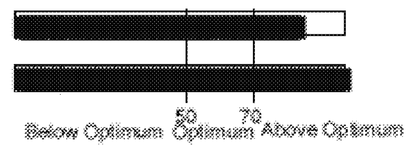
Test Results:

pH = 7.1



Phosphorus Index (P-I) =87

Potassium Index (K-I) =535



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.07 g/cm^3	10.7 meq/100 cm ^{3}	153	616	147	36

*If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.*

North Carolina
Tobacco Trust Fund Commission

Reprogramming of the laboratory-information-management system that makes this report possible is being funded through a grant from the North Carolina Tobacco Trust Fund Commission.

Thank you for using agronomic services to manage nutrients and safeguard environmental quality.

- *Steve Troxler, Commissioner of Agriculture*

NCDA&CS Agronomic Division   Phone: (919) 733-2655   Website: www.ncagr.gov/agronomi/   Report No. FY19-SL009269

Loyd Bryant   Page 2 of 6

Sample ID: 4

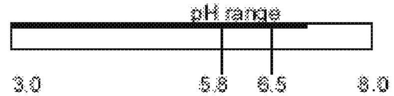
Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 7.1



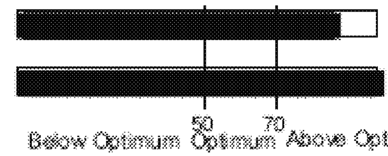
Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Phosphorus Index (P-I) =89
Potassium Index (K-I) =539



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.08	10.7	150	608	141	35
			g/cm^3	$meq/100 cm^3$				

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

Sample ID: 5

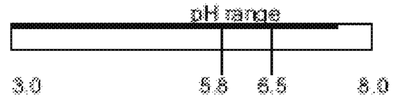
Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 7.5



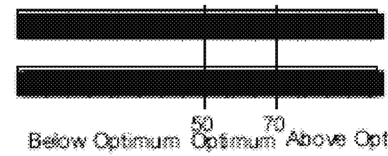
Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Phosphorus Index (P-I) =116
Potassium Index (K-I) =643



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.46	1.10	12.1	131	1601	128	36
			g/cm^3	$meq/100 cm^3$				

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

NCDA&CS Agronomic Division **Phone: (919) 733-2655** **Website: www.ncagr.gov/agronomi/** **Report No.** FY19-SL009269

Loyd Bryant Page 3 of 6

Sample ID: 6

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 7.2

Optimum pH range

3.0 5.8 6.5 8.0

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Phosphorus Index (P-I) =90

Potassium Index (K-I) =574

50 70
Below Optimum Optimum Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.04	11.2	162	735	121	35
			g/cm^3	meq/100 cm^3				

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

Sample ID: 7

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 7.3

Optimum pH range

3.0 5.8 6.5 8.0

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Phosphorus Index (P-I) =88

Potassium Index (K-I) =595

50 70
Below Optimum Optimum Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.51	1.07	9.9	198	569	105	32
			g/cm^3	meq/100 cm^3				

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

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Loyd Bryant

Page 4 of 6

Sample ID: 8

Lime History:

Loyd Bryant

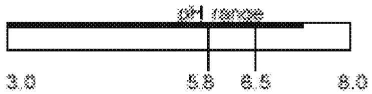
Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations
0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *
5 lbs per 1000 sq ft 21-0-0 Group D

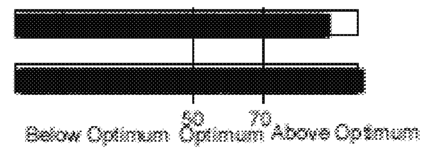
Test Results:

pH = 7.3



Phosphorus Index (P-I) =91

Potassium Index (K-I) =618



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.36	1.02 g/cm^3	11.4 meq/100 cm^3	166	696	114	33

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

Sample ID: 8A

Lime History:

Loyd Bryant

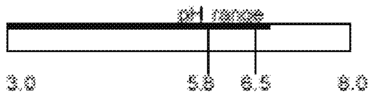
Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations
0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *
5 lbs per 1000 sq ft 21-0-0 Group D

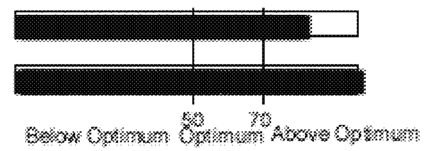
Test Results:

pH = 6.8



Phosphorus Index (P-I) =85

Potassium Index (K-I) =334



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.51	1.07 g/cm^3	9.0 meq/100 cm^3	222	475	108	32

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

MB018725

NCDA&CS Agronomic Division **Phone: (919) 733-2655** **Website: www.ncagr.gov/agronomi/** **Report No.** FY19-SL008269

Loyd Bryant

Page 5 of 6

Sample ID: 9C

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations

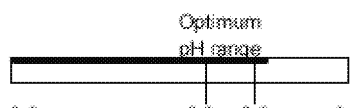
0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

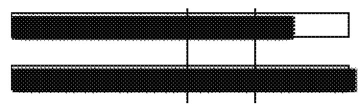
Test Results:

pH = 6.8



Phosphorus Index (P-I) =83

Potassium Index (K-I) =351



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.51	1.10 g/cm^3	8.8 $meq/100\ cm^3$	224	445	108	29

If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

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Loyd Bryant Page 6 of 6

Understanding the Soil Report

Lime

Application of lime at the recommended rate will raise soil pH to the optimum range. Do not apply too much lime. When soil pH becomes too high, lowering it is very difficult. Often, the best solution then is to choose plants that can tolerate a high pH.

Choosing dolomitic lime can be advantageous because it contains the nutrients calcium and magnesium. Pelleted lime is easier to spread uniformly than powdered lime.

Lime can be applied at any time of year, but because it reacts slowly, it is best to apply it several months before a new planting. Mixing it into the soil will speed the reaction time. Lime applied to the soil surface takes much longer to correct soil pH.

A surface application should not exceed 60 lb per 1,000 sq ft. If a soil report recommends more than this, apply 60 lb per 1,000 sq ft initially and the rest in similar increments every 6-9 months until the full rate is applied.

Fertilizer

Soil tests do not measure nitrogen (N) since it is very unstable in soils; the N recommendations provided on the soil report are based on plant needs. If soil-test P-I and K-I values are adequate (>50), only nitrogen is recommended- Group D below. A mixed (N-P-K) fertilizer is recommended if P-I and
K-I values are less than optimum- Groups A - C below. Although a specific fertilizer grade may be recommended (e.g., 5-10-10), other equivalent options are likely to be available (e.g., any fertilizer in Group A from Table 1).

Tips on Fertilizer Application

- To determine how much fertilizer to buy, estimate (in feet) the length (L) and width (W) of the area to be treated: L × W = sq ft. Square off curves to make estimates easier. If the recommendation is 20 lb per 1,000 sq ft and your area is 5,000 sq ft, then you need 100 lb (20 × 5) for your 5,000-sq-ft area.
- Calibrate your spreader according to manufacturer settings. Apply half the total rate in one direction; apply the rest at a 90° angle. This cross-hair pattern provides a more uniform application.
- After application, sweep up any fertilizer on hard surfaces and apply to fertilized areas so rainfall does not carry fertilizer to a storm drain.

Table 1. Groups of equivalent fertilizers that supply 1 lb of N per 1,000 sq ft.*

Group A: low P-I + low K-I		Group B: low P-I + high K-I		Group C: high P-I + low K-I		Group D: N only	
5-10-10	@ 20 lb	5-10-5	@ 20 lb	8-0-24	@ 12 lb	15-0-0	@ 7 lb
3-9-9	@ 30 lb	18-46-0	@ 6 lb	15-0-14	@ 7 lb	21-0-0	@ 5 lb
10-10-10	@ 10 lb	18-24-10	@ 6 lb	6-6-18	@ 18 lb	16-0-0	@ 6 lb
11-15-11	@ 10 lb	9-13-7	@ 11 lb	5-5-15	@ 20 lb	28-0-4	@ 4 lb
8-10-8	@ 12 lb	9-17-8	@ 11 lb	10-0-14	@ 10 lb	12-6-6	@ 8 lb

* Since these rates supply 1 lb N per 1,000 sq ft, use half the rate if centipede is the grass type.

Report Abbreviations

CEC	cation exchange capacity
Cu-I	copper index
HM%	percent humic matter
Mn-I	manganese index
pH	soil pH
S-I	sulfur index
SS-I	soluble salt index
W/V	weight per volume
Zn-I	zinc index

Time Fertilizer Application to Coincide with Plant Growth Cycle:

Bermudagrass: May, July, Sept
Centipedegrass: May
St. Augustine grass: May, August
Tall fescue: Sept, Nov, Feb
Zoysia: May, July
Flowers/shrubs: prior to planting or during the growing season
Vegetables: prior to planting

A Homeowner's Guide to Fertilizer

Note 4: Fertilization of Lawns, Gardens & Ornamentals

Caring for Your Lawn & Environment

Carolina Lawns

Soil Acidity and Liming: Basic Information for Farmers & Gardeners

Report No. FY19-SL009268

NCDA&CS Agronomic Division Phone: (919) 733-2655 Website: www.ncagr.gov/agronomi/ Report No. FY19-SL009268

Predictive Home & Garden

Soil Report

Mehlich-3 Extraction

Links to Helpful Information

Client: Loyd Bryant
Loyd Ray Farms Inc
2049 Center Rd.
Boonville, NC 27011

Advisor:

Sampled County : Yadkin
Client ID: 205223

Advisor ID:

Sampled: Received: 10/22/2018 Completed: 11/01/2018 Farm: 2094

Agronomist's Comments:

This report provides Test Results and Recommendations for each sample submitted for testing. Look for Lime Recommendations and N-P-K Fertilizer Recommendations. The lime recommendation is always listed next to the first crop and will be based on the higher target pH if the pH targets for crop 1 and crop 2 differ. Application at the indicated rate will raise soil pH to the optimal level for the plant you specified and should be sufficient for 2 to 3 years, depending on soil type. Common target pH values are as follows: 5.0 for azalea, camellia, rhododendron and mt. laurel; 5.5 for centipedegrass; 6.0 for other lawn grasses, shrubbery, and; flowering plants; and 6.5 for vegetable gardens. N-P-K Recommendations are based on the nitrogen (N) needs of the plants being grown and the soil test results for phosphorus (P-I) and potassium (K-I); a 50 to 70 index for either is optimum. If the exact fertilizer cannot be found, find the closest match and adjust the rate accordingly. Refer to "Understanding the Soil Report" (last page of this report) for additional explanation and links to helpful information.

Sample ID: 1A 01

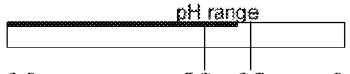
		Lime Recommendations	N-P-K Fertilizer Recommendations *
Crop 1-	Lawn (not centip.)	0.0 lb per 1,000 sq ft	7 lbs per 1,000 sq ft 15-0-14 Group C
Crop 2-		0.0 lb per 1,000 sq ft	

Lime History:

Test Results:

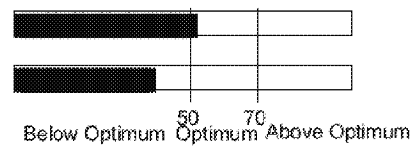
Optimum pH range

pH = 6.4



Phosphorus Index (P-I) =53

Potassium Index (K-I) =41



Loyd Bryant

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.12	7.2	165	782	135	32
			g/cm^3	meq/100 cm^3				

*If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

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Loyd Bryant | Page 2 of 5

Sample ID: 1B 02

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 6.4

Optimum pH range: 3.0 | 5.8 | 6.5 | 8.0

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

7 lbs per 1,000 sq ft 15-0-14 Group C

Phosphorus Index (P-I) =57

Potassium Index (K-I) =39

50 | 70
Below Optimum | Optimum | Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.15 g/cm^3	7.4 meq/100 cm ^{3}	167	486	143	32

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

Sample ID: 1C 03

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Test Results:

pH = 6.4

Optimum pH range: 3.0 | 5.8 | 6.5 | 8.0

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

7 lbs per 1,000 sq ft 15-0-14 Group C

Phosphorus Index (P-I) =56

Potassium Index (K-I) =41

50 | 70
Below Optimum | Optimum | Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.16 g/cm^3	7.6 meq/100 cm ^{3}	172	443	137	32

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

NCDA&CS Agronomic Division	Phone: (919) 733-2655	Website: www.ncagr.gov/agronomi/	Report No. FY19-SL009256

Loyd Bryant — Page 3 of 5

Sample ID: 3A

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations

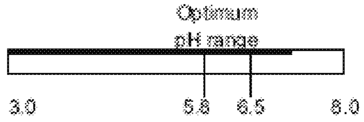
0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

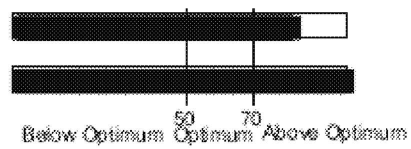
Test Results:

pH = 7.2



Phosphorus Index (P-I) =84

Potassium Index (K-I) =474



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.08 g/cm^3	9.8 meq/100 cm^3	119	542	102	31

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

Sample ID: 3B

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations

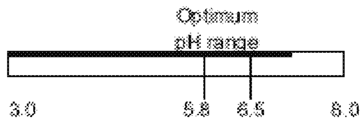
0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

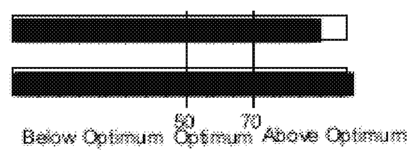
Test Results:

pH = 7.2



Phosphorus Index (P-I) =91

Potassium Index (K-I) =501



Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.36	1.10 g/cm^3	9.4 meq/100 cm^3	120	492	106	32

**If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.*
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.

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Loyd Bryant Page 4 of 5

Sample ID: 3C

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Test Results:

pH = 7.0

Optimum pH range

3.0 5.8 6.5 8.0

Phosphorus Index (P-I) =92

Potassium Index (K-I) =363

50 70
Below Optimum Optimum Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.41	1.09 g/cm^3	10.2 meq/100 cm^3	140	629	128	34

*If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.*

Sample ID: 9B

Lime History:

Loyd Bryant

Crop 1- Lawn (not centip.)
Crop 2-

Lime Recommendations

0.0 lb per 1,000 sq ft
0.0 lb per 1,000 sq ft

N-P-K Fertilizer Recommendations *

5 lbs per 1000 sq ft 21-0-0 Group D

Test Results:

pH = 6.7

Optimum pH range

3.0 5.8 6.5 8.0

Phosphorus Index (P-I) =86

Potassium Index (K-I) =345

50 70
Below Optimum Optimum Above Optimum

Additional Test Results:	Soil Class	HM%	W/V	CEC	Mn-I	Zn-I	Cu-I	S-I
	Mineral	0.51	1.06 g/cm^3	9.4 meq/100 cm^3	232	499	110	33

*If you cannot find the fertilizer recommended here, choose one from the same Group (A, B, C or D) listed on the last page of this report.
Note: This soil test does not measure nitrogen (N) levels. N fertilizer recommendations are based only on needs of the designated crop.*

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Loyd Bryant Page 5 of 5

Understanding the Soil Report

Lime

Application of lime at the recommended rate will raise soil pH to the optimum range. Do not apply too much lime. When soil pH becomes too high, lowering it is very difficult. Often, the best solution then is to choose plants that can tolerate a high pH.

Choosing dolomitic lime can be advantageous because it contains the nutrients calcium and magnesium. Pelleted lime is easier to spread uniformly than powdered lime.

Lime can be applied at any time of year, but because it reacts slowly, it is best to apply it several months before a new planting. Mixing it into the soil will speed the reaction time. Lime applied to the soil surface takes much longer to correct soil pH.

A surface application should not exceed 60 lb per 1,000 sq ft. If a soil report recommends more than this, apply 60 lb per 1,000 sq ft initially and the rest in similar increments every 6-9 months until the full rate is applied.

Fertilizer

Soil tests do not measure nitrogen (N) since it is very unstable in soils; the N recommendations provided on the soil report are based on plant needs. If soil-test P-I and K-I values are adequate (>50), only nitrogen is recommended- Group D below. A mixed (N-P-K) fertilizer is recommended if P-I and
K-I values are less than optimum- Groups A - C below. Although a specific fertilizer grade may be recommended (e.g., 5-10-10), other equivalent options are likely to be available (e.g., any fertilizer in Group A from Table 1).

Tips on Fertilizer Application

- To determine how much fertilizer to buy, estimate (in feet) the length (L) and width (W) of the area to be treated: L × W = sq ft. Square off curves to make estimates easier. If the recommendation is 20 lb per 1,000 sq ft and your area is 5,000 sq ft, then you need 100 lb (20 × 5) for your 5,000-sq-ft area.
- Calibrate your spreader according to manufacturer settings. Apply half the total rate in one direction; apply the rest at a 90° angle. This cross-hair pattern provides a more uniform application.
- After application, sweep up any fertilizer on hard surfaces and apply to fertilized areas so rainfall does not carry fertilizer to a storm drain.

Table 1. Groups of equivalent fertilizers that supply 1 lb of N per 1,000 sq ft*

Group A: low P-I + low K-I		Group B: low P-I + high K-I		Group C: high P-I + low K-I		Group D: N only	
5-10-10	@ 20 lb	5-10-5	@ 20 lb	6-0-24	@ 12 lb	15-0-0	@ 7 lb
3-9-9	@ 30 lb	18-46-0	@ 6 lb	15-0-14	@ 7 lb	21-0-0	@ 5 lb
10-10-10	@ 10 lb	18-24-10	@ 6 lb	6-6-18	@ 16 lb	16-0-0	@ 6 lb
11-15-11	@ 10 lb	9-13-7	@ 11 lb	5-5-15	@ 20 lb	28-0-4	@ 4 lb
8-10-8	@ 12 lb	9-17-8	@ 11 lb	10-0-14	@ 10 lb	12-6-6	@ 8 lb

* Since these rates supply 1 lb N per 1,000 sq ft, use half the rate if centipede is the grass type.

Report Abbreviations

CEC	cation exchange capacity
Cu-I	copper index
HM%	percent humic matter
Mn-I	manganese index
pH	soil pH
S-I	sulfur index
SS-I	soluble salt index
W/V	weight per volume
Zn-I	zinc index

Time Fertilizer Application to Coincide with Plant Growth Cycle:

Bermudagrass: May, July, Sept
Centipedegrass: May
St. Augustine grass: May, August
Tall fescue: Sept, Nov, Feb
Zoysia: May, July
Flowers/shrubs: prior to planting or during the growing season
Vegetables: prior to planting

A Homeowner's Guide to Fertilizer

Note 4: Fertilization of Lawns, Gardens & Ornamentals

Caring for Your Lawn & Environment

Carolina Lawns

Soil Acidity and Liming: Basic Information for Farmers & Gardeners

ATTACHMENT 57

Deposition of:

Kraig Westerbeek

September 1, 2021

In the Matter of:

Environmental Justice v NC Dept of Env. Quality

information together?
A. Yes.
Q. So let's start by, can you briefly explain what a nitrification/denitrification system is here?
A. In Missouri, Class 1A farms, which are the larger farms, much larger than the farms we are discussing here, as part of an agreement agreed to put in nitrification/denitrification systems in Northern Missouri which took the effluent from primary lagoons to convert ammonia to dinitrogen gas, with the goal of reducing nitrogen minimum by 50 percent.
Q. Okay.
A. So on these very large farms would have a central nitrification/denitrification system on the farms listed here.
Q. And that process is all of their waste?
A. Yes. The majority of their waste is processed through that. I think it is very important to understand that in Northern Missouri this only occurs -- it only operates during the summer months because it is too cold in the winter for it to operate properly.
So it only processes -- or processes the

effluent from say -- and I may not be exactly right on this but somewhere between April and, say, October.

Q. You mentioned that the farms involved in that are larger than the four farms that we're talking about here in North Carolina; is that right?

A. Yes. For the most part, yes.

Q. Does that make a difference? Why is that significant?

A. I don't know that it is significant or not. I just thought I'd point it out, Class 1A farms.

Q. Thanks. Making sure I wasn't missing something.

Does it work? In other words, the goal of 60 percent reduction of nitrogen levels?

A. Yes. In Missouri, it does reduce nitrogen levels by 50 percent or more. We can debate whether that is good or not, but it does what it is intended to do.

Q. What does Smithfield do with the removed nitrogen compounds, or I guess -- that's a bad question because it is going off of dinitrogen gas?

A. Yes, sir.
Q. Forget I said that.
A. I didn't hear it.
Q. Another question about the way these facilities are set up, are these facilities also pretty significant in collecting biogas?
A. They are today, yes. The first one to produce biogas was the Ruckman Farm in about 2014 or '15, I believe.
Since then, all of these listed with the exception of Terre Haute and Badger/Wolf have biogas collection and conversion of renewal natural gas.
Q. Okay. And I assume the biogas collection is happening prior to the nitrification/denitrification process?
A. That is correct.
Q. So the biogas collection and the nitrification/denitrification systems are compatible with each other; is that right?
A. They are compatible.
Q. At those facilities in Missouri, do they also employ any kind of solid liquid separation technology?
A. Yes. The digesters themselves are akin

to solid/liquid separation; nothing beyond that.
Q. Are there other forms of treatment technology being used at those facilities separate and apart from digesters and the nitrification/denitrification? Are there other add-on processes going on there?
A. Treatment technologies, no.
Q. Let's see, are those -- let me back up.
Have you been involved with the biogas project at those Missouri facilities that have installed that, you personally?
A. Yes, very involved.
Q. So are they successful? Are they -- from a business standpoint are they successful?
A. Yes. They're doing what we expected them to do, yes.
Q. Are they profitable?
MR. JOHNSON: I'll object to the relevance of this. You can go ahead and answer.
THE WITNESS: We're hoping this year they're profitable. They have not been profitable to date.
BY MR. TORREY:
Q. Does Murphy-Brown have any reason to be dispute doing nitrification/denitrification

process at the four North Carolina operations that we're talking about would reduce nitrogen levels in wastes at those North Carolina operations?

A. So your question is would nitrification/denitrification reduce nitrogen levels if that was installed in North Carolina in the effluent, that's your question?

Q. Yes, in the effluent from biogas digesters.

A. Yes, it would. That technology would reduce nitrogen in the effluent.

Q. Would Murphy-Brown have any reason to think it would perform differently in terms of the amount of reduction that you see in Missouri in a North Carolina setting at these facilities?

A. Yes. Because in Missouri, as stated before, there is only -- the biology that supports both biogas production and denitrification and nitrification only occurs part of the year, at most eight months -- seven to eight months out of the year. The rest is too cold for biological processes.

In Eastern North Carolina, that biological activity occurs year-round so you see much lower levels of nitrogen as a starting point then to apply to crops which is part of the neutral management plan.

You are not only spending money to do that, you are spending electricity, energy, greenhouse gases, et cetera, to destroy nitrogen that has value.

Q. Let me go back to my question because I'll try to ask it a better way.

The nitrogen that's reduced through the nitrification/denitrification process, does that include ammonia?

A. Yes. Ammonia would be one component.

Q. So would nitrification/denitrification apply to digester waste in North Carolina at these facilities, if you were to do that, reduce the ammonia levels in the waste, in the digester waste at those facilities in North Carolina?

A. Yes. Nitrification/denitrification could reduce ammonia levels, yes -- nitrogen levels including ammonia.

Q. Would changing the amount of cover crop that's grown reduce the need for nitrogen through land applications?

A. What's grown is as much a function to the entirety of the farming enterprise, and what's

ATTACHMENT 58

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BIORESOURCE TECHNOLOGY

Development of environmentally superior treatment system to replace anaerobic swine lagoons in the USA

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Available online 21 August 2006

Abstract

A full-scale treatment system for swine manure was developed to eliminate discharge to surface and ground waters and contamination of soil and groundwater by nutrients and heavy metals, along with related release of ammonia, odor, and pathogens. The system greatly increased the efficiency of liquid–solid separation by polymer injection to increase solids flocculation. Nitrogen management to reduce ammonia emissions was accomplished by passing the liquid through a module where bacteria transformed ammonia into harmless nitrogen gas. Subsequent alkaline treatment of the wastewater in a phosphorus module precipitated phosphorus and killed pathogens. Treated wastewater was recycled to clean swine houses and for crop irrigation. The system was tested during one year in a 4400-head finishing farm as part of the Agreement between the Attorney General of North Carolina and swine producers Smithfield Foods, Premium Standard Farms and Frontline Farmers to replace traditional waste treatment anaerobic lagoons with environmentally superior technology. The on-farm system removed 97.6% of the suspended solids, 99.7% of BOD, 98.5% of TKN, 98.7% of soluble ammonia (NH_4^+–N), 95.0% of total P, 98.7% of copper and 99.0% of zinc. It also removed 97.9% of odor compounds in the liquid and reduced pathogen indicators to non-detectable levels. Based on performance obtained, it was determined that the treatment system met the Agreement's technical performance standards that define an environmentally superior technology. These findings overall showed that cleaner alternative technologies are technically and operationally feasible and that they can have significant positive impacts on the environment and the livestock industry.

Keywords: Manure treatment; Pathogen inactivation; Nitrification–denitrification; Phosphorus and ammonia removal; Confined swine production; Piggery

1. Introduction

Minimizing livestock manure's impact on the environment is one of USA agriculture's major challenges. When properly managed, manure can be used to provide nutrients to crops and to improve soil properties through accretion of soil organic matter. However, improperly managed manure can pose a threat to soil, water, and air quality in addition to human and animal health. Anaerobic lagoons are widely used to treat and store liquid manure from confined swine production facilities (Barker, 1996). Environmental and health concerns with the lagoon technology include emissions of ammonia (Aneja et al., 2000; Szogi et al., 2005), odors (Loughrin et al., 2006; Schiffman et al., 2001), pathogens (Sobsey et al., 2001; Vanotti et al., 2005a), and water quality deterioration (Mallin, 2000). Thus, there is a major interest in developing alternative swine manure treatment systems that can also address these environmental and health problems.

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Widespread objection to the use of anaerobic lagoons for swine manure treatment in North Carolina prompted a state government-industry framework to give preference to alternative technologies that directly eliminate anaerobic lagoons as a method of treatment. The full-scale treatment demonstration described in this paper was conducted within this framework. In July 2000, the Attorney General of North Carolina reached an Agreement with Smithfield Foods, Inc., and its subsidiaries, the largest hog producing companies in the world, to develop and demonstrate environmentally superior waste management technologies for implementation on farms located in North Carolina that are owned by these companies. In October 2000, the Attorney General reached a similar agreement with Premium Standard Farms, the second largest pork producer in the USA. The agreement defines an environmentally superior technology (EST) as any technology, or combination of technologies, that (1) is permittable by the appropriate governmental authority; (2) is determined to be technically, operationally, and economically feasible; and (3) meets the following five environmental performance standards (Williams, 2001):

1. Eliminate the discharge of animal waste to surface waters and groundwater through direct discharge, seepage, or runoff.
2. Substantially eliminate atmospheric emissions of ammonia.
3. Substantially eliminate the emission of odor that is detectable beyond the boundaries of the swine farm.
4. Substantially eliminate the release of disease-transmitting vectors and airborne pathogens.
5. Substantially eliminate nutrient and heavy metal contamination of soil and groundwater.

Selection of EST candidates to undergo performance verification involved a request of proposals and competitive review by the Agreement's Designee and a Panel representing government, environmental and community interests, the companies, and individuals with expertise in animal waste management, environmental science and public health, and economics and business management. This process yielded 18 technologies candidates among about 100 submitted projects. Subsequently, the selected technologies completed design, permitting, construction, startup, and performance verification under steady-state operational conditions. In July 2005, five of the 18 technologies tested were shown to be capable of meeting the environmental performance criteria necessary for the technologies to be considered environmentally superior (Williams, 2004, 2005). Four of the five technologies selected treated separated manure solids using composting, high-solids anaerobic digestion, or gasification processes, and only one of the technologies selected treated the entire swine waste stream on-farm. This on-farm technology used liquid–solid separation, nitrification/denitrification, and soluble phosphorus removal processes linked together into a practical system. It was developed to replace anaerobic lagoon technology commonly used in the USA to treat swine waste (Vanotti et al., 2005b).

In this new manure treatment system, solids and liquid are first separated with polyacrylamide (PAM) polymer and filtration process, followed by treatment of the liquid stream using biological nitrogen (N) removal process, and then by phosphorus (P) extraction using a lime precipitation process. Flocculation treatment using PAM increases separation of suspended solids and carbon compounds from liquid swine manure (Vanotti and Hunt, 1999). Along with the solids, there is a significant separation of organic nutrient elements contained in small suspended particles typical of these wastes. For example, Vanotti et al. (2002) analyzed the fractions in liquid swine manure that are potentially removable by phase separation and found that 80% of the total suspended solids (TSS), 78% of the N and 93% of the P were contained in particles less than 0.3 mm in size. Soluble ammonia (NH_4^+–N) and soluble P (PO_4), which usually constitute 35–65% of total N and 15–30% of total P, are mostly unaffected by polymer separation. Biological removal of N by combined nitrification and denitrification processes (NDN) is regarded as the most efficient and economically feasible method available for removal of N from wastewaters (Focht and Chang, 1975; Tchobanoglous and Burton, 1991; Furukawa et al., 1993). Once NH_4^+–N and carbonate alkalinity concentrations are substantially reduced with nitrification treatment, the subsequent addition of hydrated lime [$Ca(OH)_2$] rapidly increases the pH of the liquid above 9, thereby promoting formation of calcium phosphate precipitate with small amounts of chemical added (Vanotti et al., 2003b).

The treatment system was first pilot tested for two years at the North Carolina State University's Lake Wheeler Road Swine Unit (Vanotti et al., 2003a). A full-scale version of the system was subsequently constructed in a swine farm in North Carolina for demonstration and performance verification of environmentally superior technology. In this paper, we report the water quality improvements by the treatment system operating at full scale. In addition, we report on characteristics of the separated solid fractions, energy balance of the system, and operational considerations. Performance verification was done during a one year period and included cold and warm weather conditions.

2. Methods

2.1. Site description

The full-scale demonstration facility was installed on Goshen Ridge farm (Unit 1) near Mount Olive, Duplin Co., North Carolina, and evaluated intensively during one year under steady-state conditions. The production unit contained six swine barns with 4360-head finishing pigs total, and a traditional anaerobic lagoon (0.9 ha) for treatment and storage of manure. Manure was collected

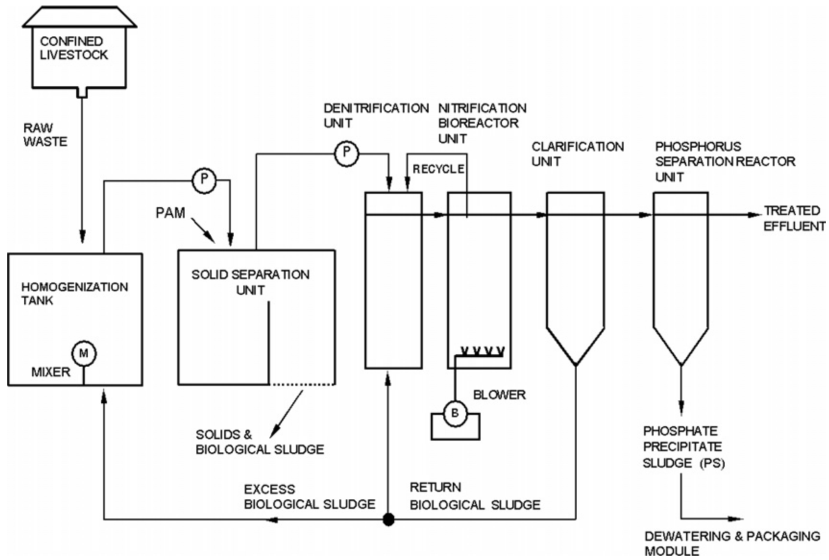

Fig. 1. Schematic drawing of the swine waste treatment system without lagoon.

under the barns using slatted floors and a pit-recharge system typical of many farms in North Carolina (Barker, 1996). The production unit with its traditional lagoon system was operational for about four years before the new waste treatment plant started operation. During traditional management, every week the liquid manure contained in the pits was completely drained by gravity into the anaerobic lagoon. After treatment in the lagoon (retention time = 180 days), the liquid was sprayed onto nearby fields growing small grains and forages. Lagoon liquid was also recycled (in a closed loop) to recharge the pits under the barns and facilitate flushing of the newly accumulated manure.

Once the treatment plant was operational, flow of raw manure into the lagoon was discontinued. Barn pits were flushed once a week as it was done before, but liquid manure was diverted into a 388-m^3 homogenization tank. Transfer rate was rather quick using a high capacity pump (1.9 m^3/min). Typically, half of the six barns were emptied on Monday and the other half on Thursday. The manure collected in the homogenization tank was kept well mixed using a submergible mixer (3.5 kW, 12.1 m^3/min flow, ABS Pumps Inc., Meriden, CT[1]). From there, the liquid manure received continuous treatment. The treatment system consisted of three process units in series: polymer-enhanced solid–liquid separation, biological N removal, and alkaline phosphorus extraction (Fig. 1).

2.2. On-farm treatment system

The treatment system used was a system without lagoon (Vanotti et al., 2005b, Fig. 1) comprised of (a) a solid separation unit, wherein flocculants are used to clump suspended solids and increase separation efficiency, (b) a denitrification unit in direct fluid communication with a clarified effluent from the solid separation unit, (c) a nitrification unit in fluid communication with the denitrification unit, (d) a phosphorus separation reactor unit in fluid communication with the liquid effluent from the nitrification unit, and (e) a clarification unit between the nitrification unit and phosphorus unit. Homogenization and storage tanks were added to the system to integrate discontinuous operations, such as flushing and barn pit recharge, with continuous operation of the treatment system (Fig. 2).

The on-farm system was constructed and operated by a private firm, Super Soil Systems USA of Clinton, North Carolina. It was implemented using three process units or modules (Fig. 3). The first process unit in the system – the Ecopurin solid–liquid separation module, developed by the Spain-based firm Selco MC of Castellon – quickly separated solids and liquid using polymer flocculation and dewatering equipment. The solid–liquid separation module was housed in a building of its own. It was automated through the use of a programmable logic controller (PLC) for a 24-h/day operation (Square D, Schneider Electric, North Andover, MA). Treatment parameters such as polymer rate, wastewater flow, and mixing intensity were set by the operator using a tactile screen in a control panel. Well mixed raw manure was continuously pumped from the homogenization tank to the separation module. Flow rate was uniform at 2 m^3/h during the year-long demonstration. The liquid manure was first reacted in a mixing chamber with a polymer solution (cationic polyacrylamide) that flocculated the suspended solids, and then it was passed through a rotating screen (0.2 mm opening size) that separated the flocs. Subsequently, a dissolved air flotation unit (DAF) polished the liquid effluent while a small belt filter press (Monobelt, Teknofanghi S.R.L., Italy) further dewatered the screened solids. The solid–liquid

[1] Mention of trademark, proprietary product, or vendor does not constitute a guarantee or warranty of the product by the US Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that also may be suitable.

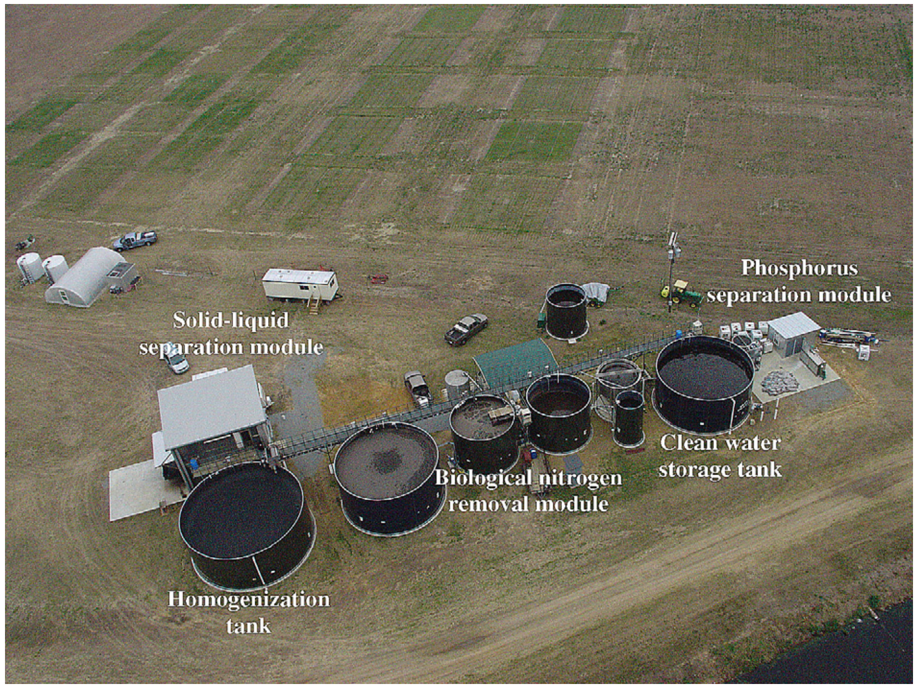

Fig. 2. Aerial view of the full-scale swine wastewater treatment system that replaced the anaerobic lagoon.

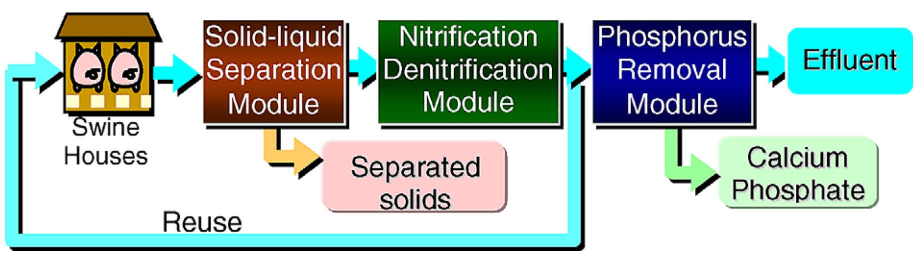

Fig. 3. Diagram of the swine manure treatment system with individual modules implemented at Goshen Ridge farm, North Carolina.

separation module produced a solids stream and a liquid stream. The solids were removed daily from the farm and transported in trailers to a centralized solids processing plant where they received aerobic composting. The liquid was lifted into the nitrogen removal module.

The second process unit in the system used a Biogreen nitrogen removal module (Hitachi Plant Engineering & Construction Co., Tokyo, Japan) that used nitrification/denitrification (NDN) to biologically convert NH_4^+–N into N_2 gas. The Biogreen process has a pre-denitrification configuration where nitrified wastewater is continuously recycled to an anoxic denitrification tank (Fig. 1). In this tank, suspended denitrifying bacteria uses soluble manure carbon contained in the liquid after separation to remove the nitrate and nitrite. The nitrification tank uses nitrifying bacteria immobilized in polymer gel pellets to increase the concentration and effectiveness of bacterial biomass (Vanotti and Hunt, 2000). Nitrifying 3-mm bio-cube pellets are kept inside the nitrification tank by means of a wedge-wire screen structure (1.5 mm opening). The full-scale Biogreen unit contained a 263-m^3 anoxic denitrification tank to remove soluble manure carbon and nitrate-N (NO_3–N), a 110-m^3 nitrification tank for conversion of NH_4^+ to NO_3^-, and a 33-m^3 tank for settling and recycling of suspended biomass solids to the denitrification tank or wasting excess biomass to the separation module (Fig. 1). The height of the liquid in these tanks was 4 m. The denitrification tank contained a submergible mixer (1.7 kW, 9.8 m^3/min flow, ABS Pumps Inc., Meriden, CT) and a concentration of 3–6 g/l mixed liquor suspended solids (MLSS). The nitrification tank contained 125 fine-bubble air diffusers (22.9-cm diameter) and 12 m^3 of polyethylene glycol (PEG) immobilized pellets. Air was provided with a 11.2 kW, rotary lobe blower (Kaeser Omega DB 165, Kaeser Compressors, Fredericksburg, VA). Nitrification activity of the pellets after 5 weeks of initial acclimation was 850 g N/100 l pellets/day. Corresponding nitrification activity of the 110-m^3 reactor tank (containing 12-m^3 of pellet media) was 102 kg N/day, or 0.93 kg N/m^3 reactor/day. Hydraulic retention time (HRT) of nitrification varied from 2.6 to 3.6 days (average = 2.8 day). Nitrified liquid and settled sludge were recirculated to the first denitrification tank at a rate average of 4.4 and 1.8 times the inflow rate, respectively.

After biological N treatment, the effluent was discharged into a 299-m^3 tank that stored water needed to recharge pits under the barns after barns were flushed. Excess water flowed by gravity from this storage tank into the phosphorus separation module developed by USDA-ARS (Vanotti et al., 2003b). This was the third and final process unit in the system. It was designed to recover soluble P (as calcium phosphate) and destroy pathogens by alkaline pH. In this module, liquid was first mixed with hydrated lime slurry in a reaction chamber. The lime slurry was a 30% $Ca(OH)_2$ suspension supplied in standard tote containers and ready to use (Chemical Lime Company, Charlotte, NC). A pH probe and controller linked to the lime injection pump kept the process pH at 10.5–11.0. The liquid and precipitate were subsequently separated in a 9-m^3 settling tank. The precipitated calcium phosphate was removed from the bottom of the tank with a pump and it was further dewatered using a 12-filter bag Draimad unit that also bagged the sludge (Teknobag-Draimad, Aero-Mod, Inc., Manhattan, KS). Anionic polymer was added in-line to the P precipitate to enhance separation by filter bags (Szogi et al., 2006). Bags containing the wet calcium phosphate were left to dry on a drying concrete pad and removed from the farm on a monthly basis. Process automation was provided by sensors integrated to another PLC for 24-h/day operation. Treatment parameters such as process pH or frequency of sludge transfer were set by the operator using a tactile screen located in the plant control panel. Clarified effluent from the P module was stored in the existing lagoon before use in crop irrigation. Cylindrical tanks used in the system were standard structures made of glass-fused to steel (Slurrystore, Engineered Storage Products Company, Dekalb, IL), while settling tanks were custom-made of stainless steel.

2.3. Wastewater sampling and monitoring

Liquid samples were collected twice per week using four refrigerated automated samplers (Sigma 900max, American Sigma, Inc., Medina, NY) placed before and after each of the treatment modules in the system as follows: (1) the untreated liquid manure in the mixing tank before solids separation, (2) the effluent from the solid–liquid separation treatment (post-separation), (3) the effluent after the nitrification–denitrification treatment (post-N removal), and (4) the effluent after the phosphorus and pathogen elimination treatment (post-P removal). Each sample was the composite of four sub-samples taken over a 3.5-day period. Samples were transported on ice to the ARS Coastal Plains Research Center in Florence, SC, for water quality analyses, or overnight shipped with cold packs to the ARS Sustainable Agricultural Systems Laboratory and Environmental Microbial Safety Laboratory in Beltsville, MD, for microbiological analyses.

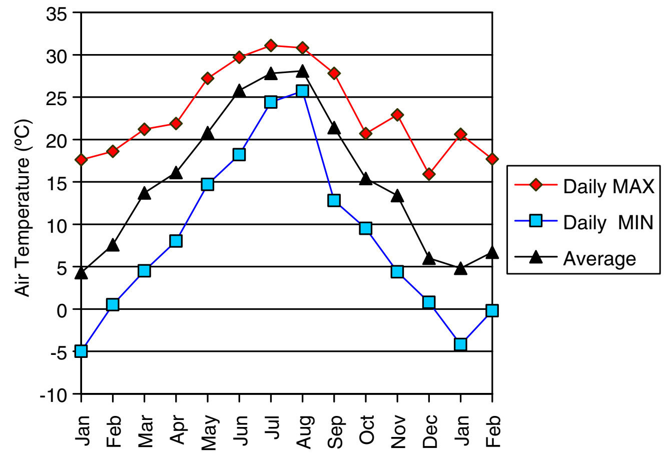

Fig. 4. Air temperature during January 2003–February 2004. Data are max and min of average daily temperatures and monthly average of average daily temperatures.

Wastewater flows throughout the system were measured with several calibrated flowmeters adapted to the characteristics of the liquid: raw manure transported from the barns into the homogenization tank was measured with a doppler flowmeter, liquid and sludge flows in the N and P modules were measured with magnetic flowmeters, and treated effluent was measured with a paddle-wheel flowmeter. Monitoring and process data were obtained every 5 min using a SCADA network (Monitor Pro v7, Schneider Automation, Inc., North Andover, MA) connected to the programmable logic controller (PLC) that provided plant automation. The data were temporarily stored in an industrial computer (IPC-6806, Advantech Co., Cincinnati, OH) at the farm and transmitted weekly to the Florence laboratory for analysis and summarization using SAS software (SAS, 2003).

To calculate electrical power use, we measured run-time (hours/day) of all electrical devices (35) installed in the plant that contributed to the power consumption by the system. This was done with the SCADA monitoring system that counted total hours of use during a 275-day period (April 2003–January 2004). Average run-time was multiplied by power use of each electrical device (kW) to calculate daily power requirements (kW h/day).

Performance evaluation included cold and warm weather conditions with average daily air temperatures ranging from −4.2 to 31.1 °C (Fig. 4).

2.4. Analytical methods

Wastewater analyses were performed according to Standard Methods for the Examination of Water and Wastewater (APHA, AWWA and WEF, 1998). Total solids (TS), total suspended solids (TSS), and volatile suspended solids (VSS) were determined with Standard Method 2540 B, D, and E, respectively. Total solids are the solids remaining after evaporation of a sample to constant weight at 105 °C and include TSS and dissolved solids (DS). Total suspended solids are the solids retained on a 1.5-μm glass microfiber filter (Whatman grade 934-AH, Whatman, Inc., Clifton, NJ) after filtration and drying to constant weight at 105 °C, while VSS is the fraction of the TSS lost on ignition in a muffle furnace at 500 °C for 15 min.

Chemical analyses consisted of pH, electrical conductivity (EC), chemical oxygen demand (COD), soluble COD, 5-d biochemical oxygen demand (BOD_5), soluble BOD_5, ammonia (NH_4^+–N), nitrate plus nitrite ($NO_3^- + NO_2^-$), total Kjeldahl N (TKN), orthophosphate-P (PO_4), total P (TP), copper (Cu), and zinc (Zn). For COD, we used the closed reflux, colorimetric method (Standard Method 5220 D). The orthophosphate (PO_4–P or soluble P) fraction was determined by the automated ascorbic acid method (Standard Method 4500-P F) after filtration through a 0.45-μm membrane filter (Gelman type Supor-450, Pall Corp., Ann Arbor, MI). The same filtrate was used to measure NH_4–N by the automated phenate method (Standard Method 4500-NH_3 G), and $NO_3^- + NO_2^-$ by the automated cadmium reduction method (Standard Method 4500-NO_3^- F). Total P and TKN were determined using acid digestion (Gallaher et al., 1976) and the automated ascorbic acid and phenate methods adapted to digested extracts (Technicon Instruments Corp., 1977). The organic P fraction is the difference between total P and PO_4 analyses and includes condensed and organically bound phosphates. The organic N fraction is the difference between Kjeldahl N and NH_4^+–N determinations. Alkalinity was determined by acid titration to the bromocresol green endpoint (pH = 4.5) and expressed as mg $CaCO_3$/l. Cu and Zn were measured in acid digestion extracts using inductively coupled plasma (ICP) analysis (Standard Method 3125A).

Solids samples were analyzed for moisture content using a microwave moisture analyzer. Dry solids samples were digested with concentrated acid, and the extracts were analyzed for TKN and TP with the automated methods described before. Carbon content was determined using a dry combustion analyzer.

Reduction in odor was characterized by measuring concentration of six odor compounds characteristic of swine manure (phenol, *p*-cresol, *p*-ethylphenol, *p*-propylphenol, indole, and skatole) directly in the liquid using gas chromatography and the method of Loughrin et al. (2006). Microbiological analyses of liquid samples were done using the standard protocols for pathogens and indicator microbes for the examination of wastewater (Vanotti et al., 2005a).

3. Results and discussion

3.1. Livestock and manure inventory

Pig inventory, live weight, and manure production data are summarized in Table 1. New batches of pigs were received January–February 2003, June–July 2003, November–December 2003, and March 2004. The pigs did not receive antibiotics, and the meat was marketed with a different label indicating this change. Total live animal weight (LAW) in the production unit averaged 237,000 kg but varied greatly within a growing cycle from a low of about 90,000 to 150,000 kg to a high of about 350,000 to 365,000 kg.

Manure production varied from 30.7 to 43.2 m^3 per day (Flushed manure, Table 1). Volume production was generally higher in warmer months. The system treated an average of 39 m^3 per day of raw manure flushed from the barns (Table 2). On the average, the flushed manure contained 33% recycled treated water (used to refill and flush the pits) and 67% manure and wasted water (urine, feces, water wasted by pigs). The manure and wasted water production (raw flushed manure – effluent recycled to barns, Table 2), which constitutes the newly generated manure, averaged 26.3 m^3 per day or 110 l/1000 kg LAW/day. This is consistent with expected table values of 101 l/1000 kg LAW/day (1.62 ft^3/1000 lb/day or 6.2 l/pig/day) for manure and wasted water production in feeder-to-finish operations in the USA (average pig weight = 135 lb or 61.2 kg) (Chastain et al., 1999). On the other hand, the total amount of flushed manure treated by the plant (manure/wasted water plus recycled water) was much lower than what is considered typical in feeder-to-finish operations in the USA. For example, the average 39 m^3 per day of raw manure flushed from the barns (Table 2) was equivalent to 165 l/1000 kg LAW/day. This is 2.6 times lower than the volume of 424 l/1000 kg LAW/day (6.80 ft^3/1000 lb/day or 25.9 l/pig/day) considered typical for pit-recharge systems in the USA (Chastain et al., 1999). This lower volume was obtained by a change in pit management incorporated with the new system that reduced the amount of recycle liquid into the barns to a minimum needed for effective

Table 1
Inventory of pigs and manure volume generation at Goshen Ridge farm (Unit 1) during demonstration of the new wastewater treatment system

Pigs and manure information	March	April	May	June	July	August	September	October	November	December	January
Number of pigs	3978	3975	3441	978	2787	4115	4015	3749	2831	4120	3814
Weight/pig (kg)	51.7	79.4	101.6	84.4	20.9	48.1	75.8	98.0	65.8	45.4	85.7
Total weight (Mg)	206	316	347	122	87	198	304	365	149	186	326
Flushed manure (m^3/day)[a]	30.7	32.6	36.3	36.0	43.2	45.0	55.3	48.1	33.3	36.0	34.1
Pit recharge (m^3/day)[b]	–	19.3	17.8	17.8	16.7	7.9	8.7	15.1	6.8	10.6	8.7

[a] Flushed manure is the average daily volume received in the homogenization tank.
[b] Pit recharge is the average daily volume treated liquid recycled from the clean water storage tank to the barns.

Table 2
Wastewater flows through the swine wastewater treatment system

Flow path	Total volume[a] (m^3)	Average flow rate (m^3/day)
Raw flushed manure to homogenization tank	12,050	39.0
Separated effluent to nitrogen module	12,070	39.1
N-treated effluent recycled to refill barns	3934	12.7
N-treated effluent to phosphorus module	8179	26.5
P-treated effluent to storage pond (former lagoon)	7975	25.8

[a] Monitoring values for period April 15, 2003, to March 1, 2004 (10.5 months).

cleanup of the barn pit. In turn, this change in management resulted in a lower volume (38%) of total flushed manure compared with management in traditional lagoon systems, which increased efficiency in terms of equipment (tanks, pumps, pipes, mixers), footprint, etc.

3.2. Water quality improvement by treatment system

System performance data were obtained during 10.5 months from April 15, 2003, to March 1, 2004, when all three modules were in-line. The on-farm system removed 97.6% of TSS, 98.9% of VSS, 97.4% of COD, 99.7% of BOD, 98.5% of TKN, 98.7% of NH_4^+–N, 95.0% of TP, 94.1% of soluble P, 98.7% of Cu, and 99.0% of Zn (Table 3).

Data in Table 3 and Fig. 5 show the unique contributions of each technology component to the efficiency of the total treatment system. Solid–liquid separation was effective separating suspended solids and organic nutrients. By capturing the suspended particles early in the process, most of the volatile and oxygen-demanding organic compounds were removed from the liquid stream. This early removal of suspended solids in the treatment train is a significant departure from wastewater treatment processes typically used in municipal systems because (1) it recovers the organic carbon and nutrient compounds contained in liquid manure, therefore enabling conservation and generation of organic value-added products, and (2) instead of breaking down organic compounds, the oxygen in subsequent biological aerobic treatment is used efficiently to convert NH_4^+–N. This is particularly important in animal treatment systems because as shown in Table 3, the effluent after solid–liquid separation contained significant amounts of N (953 mg/l), mostly soluble forms (NH_4^+). The NH_4^+–N was treated effectively in the biological N removal module. This module also consumed remaining carbon (BOD, COD) during denitrification, and alkalinity during nitrification. Soluble P contained in the liquid was not significantly changed by either liquid–solid separation or N treatment, but it was reduced significantly after treatment in the P-module (Table 3), and recovered as a solid calcium phosphate material.

Table 3
Removal of suspended solids, COD, BOD, nutrients, and heavy metals by on-farm wastewater treatment system at Goshen Ridge farm, North Carolina[a]

Water quality parameter	Raw liquid swine manure, mg/l (±s.d.)	After solid–liquid separation treatment, mg/l (±s.d.)	After biological N treatment, mg/l (±s.d.)	After phosphorus treatment, mg/l (±s.d.)	Removal efficiency with system (%)
TSS	11,051 (5914)	823 (637)	122 (68)	264 (154)	97.6
VSS	8035 (5016)	591 (456)	77 (54)	85 (50)	98.9
TS[b]	13,216 (5394)	4452 (1475)	3710 (694)	3339 (586)	74.7
COD	16,138 (8997)	3570 (2104)	617 (192)	445 (178)	97.4
Soluble COD	3129 (2017)	2289 (1499)	525 (164)	393 (166)	87.4
BOD_5	3132 (2430)	1078 (1041)	33 (25)	10 (16)	99.7
Soluble BOD_5	909 (935)	624 (656)	9 (16)	7 (8)	99.2
TKN	1584 (566)	953 (305)	34 (30)	23 (24)	98.5
NH_4^+–N	872 (329)	835 (292)	23 (34)	11 (19)	98.7
Organic N	712 (325)	111 (96)	12 (11)	11 (12)	98.5
Oxidized N[c]	1 (3)	1 (3)	224 (100)	224 (105)	–
Total N[d]	1584	954	258	247	89.4
Total P	576 (224)	174 (53)	147 (30)	29 (16)	95.0
Soluble P	135 (40)	121 (33)	134 (24)	8 (7)	94.1
Organic P	440 (197)	49 (41)	13 (19)	19 (16)	95.7
Copper	26.8 (12.2)	1.54 (1.82)	0.53 (0.28)	0.36 (0.26)	98.7
Zinc	26.3 (11.9)	1.47 (1.85)	0.40 (0.28)	0.25 (0.30)	99.0
Alkalinity	5065 (1791)	4345 (1555)	529 (323)	735 (263)	85.5
pH	7.60 (0.19)	7.91 (0.15)	7.24 (0.74)	10.49 (0.57)	–
EC (mS/cm)	10.44 (3.09)	10.39 (2.87)	5.13 (0.79)	4.86 (0.87)	–

[a] Values are mean (standard deviation) for 121 sampling dates (April 15, 2003–March 1, 2004).
[b] Total solids (TS) = Total suspended solids (TSS) + Dissolved solids.
[c] Oxidized-N = NO_3–N + NO_2–N (nitrate plus nitrite).
[d] Total N = TKN + Oxidized-N. System efficiency for total N = 89.4% on a mass balance basis. This considers that 33% of the N-treated effluent was recycled in a closed loop to refill barns where oxidized N was eliminated (Table 2).

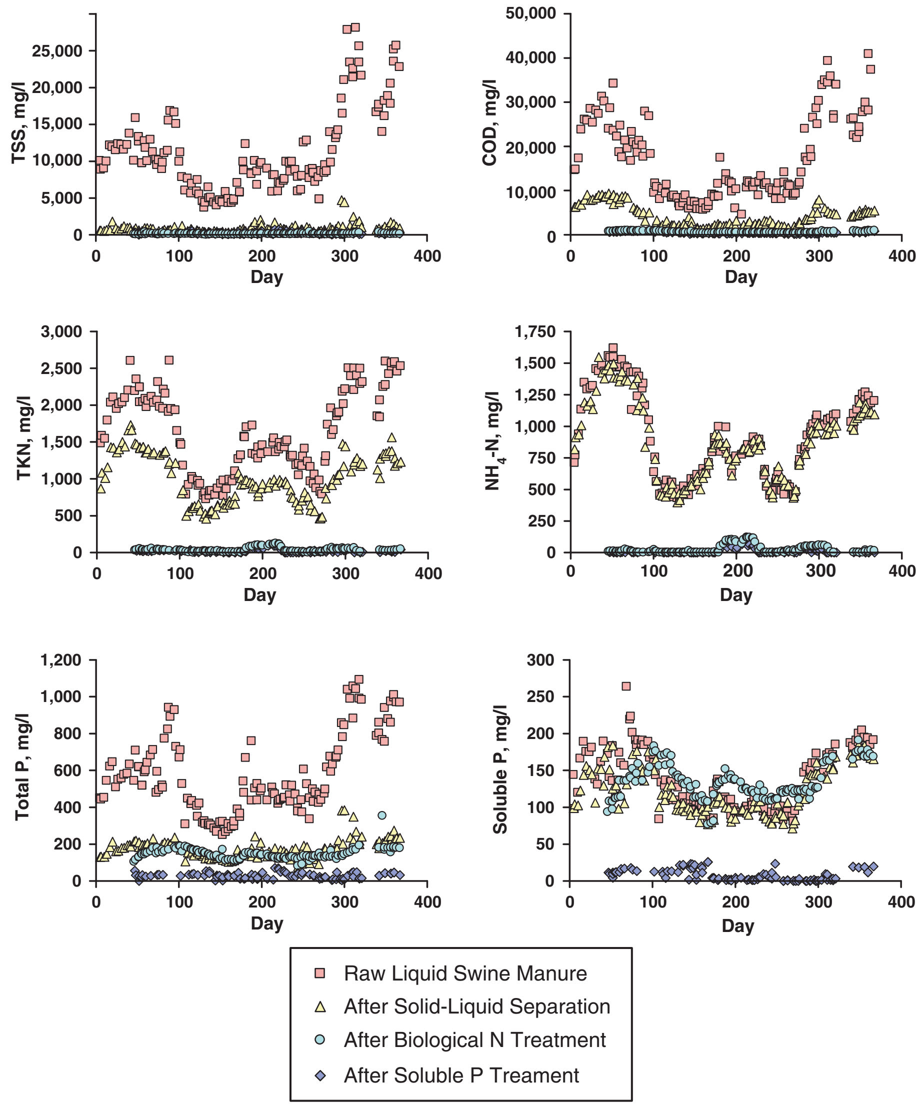

Fig. 5. Water quality improvements (TSS, COD, TKN, NH_4^+–N, TP and soluble P) in the on-farm wastewater treatment system at Goshen Ridge farm, North Carolina, as liquid swine manure passes through solid–liquid separation, biological N removal, and soluble P removal modules. Data show performance verification at steady-state conditions from March 1, 2003 (day = 1) to March 1, 2004 (day = 367).

The treatment system was also effective in reducing odor-generating compounds and pathogen indicator microorganisms contained in the liquid (Table 4). By measuring directly in the liquid the concentration of compounds typically associated with bad smell in animal wastes, we were able to quantify the potential of the effluent to produce offensive odors and the effect of each treatment step on odor reduction. The largest odor reduction

Table 4
Removal of odor compounds and pathogen indicator microorganisms by on-farm wastewater treatment system at Goshen Ridge farm, North Carolina

	Raw liquid swine manure	After solid–liquid separation treatment	After biological N treatment	After phosphorus treatment	Removal efficiency with system (%)
Odor compounds, ng/ml (±s.e.)[a]	206.78 (52.62)	181.69 (77.98)	4.61 (2.00)	4.29 (2.44)	97.6
Total fecal coliforms, $\log_{10}$/ml (±s.e.)[b]	3.74 (0.36)	3.09 (0.29)	1.01 (0.23)	BDL	>99.9

[a] Values are means (standard error) of five determinations (September–October 2003). Odor compounds are the sum of concentrations of six malodorous compounds contained in the liquid (phenol, *p*-cresol, *p*-ethylphenol, *p*-propylphenol, indole, and skatole) that are characteristic of swine manure.
[b] Values are means (standard error) of $\log_{10}$ colony forming units (cfu) per ml for duplicate samples of four determinations (July–December, 2003). BDL = below detectable limit, indicates there were no colonies to count.

was observed after the liquid passed through aeration in the nitrogen treatment. Overall, the treatment system eliminated 97.9% of the odor compounds. Microbiological analyses showed a consistent trend in reduction of fecal coliforms as a result of each step in the treatment system. It confirmed pilot studies (Vanotti et al., 2005a) that the phosphorus removal step via alkaline calcium precipitation produces a sanitized effluent.

3.3. Solid–liquid separation module

Efficiency of solid–liquid separation using polymer flocculation was consistently high with an average of 93% TSS separation. This high-separation efficiency was obtained with liquid manure TSS concentrations that varied from about 4000 mg/l to 28,000 mg/l (Fig. 5). Application rate of PAM varied from 106 to 178 g/m^3 (average = 136 g/m^3) corresponding to the changes in wastewater strength. The solids separation module also removed 93% of the volatile suspended solids, 78% of COD, 40% of TKN, 94% of zinc and copper, and 70% of TP from the wastewater (Table 3). As mentioned before, this reduction of organic compounds such as COD is an important system consideration for the efficiency of subsequent nitrification treatment. Soluble NH_4^+ and P concentrations changed little (4.2% and 10.4% reduction, respectively) with solids separation treatment. In contrast, organic N and P were effectively captured in the solids, resulting in average concentration reductions of 84.4% and 88.9%, respectively.

A total of 748 m^3 of solids were separated and left the farm in a 10.5-month period. This amount of manure weighed 596,200 kg and contained 18.2% (±1.3%) solids (81.8% moisture), 40,805 kg of carbon, 5379 kg of N, 3805 kg of P, 280 kg of Cu, and 281 kg of Zn. The separated solid waste was composted in a centralized solids processing facility and converted into organic plant fertilizer, soil amendments, and plant growth media (Vanotti, 2005).

3.4. Biological N removal module

Ammonia (NH_4^+–N) removal efficiencies of the Biogreen process were consistently high (average = 97%, Table 3). These high process efficiencies were obtained with influent NH_4^+–N concentrations varying from about 400 to 1500 mg/l (Fig. 5) and loading rates varying from about 18 to 45 kg N/day (average = 32 kg/day). After solids separation, most of the TKN was made of NH_4^+–N; therefore, removal efficiencies for TKN were also high (96%). Influent TKN concentration varied from 460 to 1730 mg/l, and loading rates varied from 20 to 50 kg N/day (average = 37 kg/day). Nitrogen loading rates into the N removal module fluctuated greatly (150%) within production cycles. These N loading fluctuations were well correlated ($r = 0.83$) with changes in total pig weight in the barns [N load (kg TKN/day) = 17.4 + 0.0820 live weight (Mg)]. The biological N removal process responded well to these highly changing N loading conditions as well as cold temperatures experienced during evaluation. Water temperatures during cold weather (December 2003–February 2004) were 11.9–13.0 °C for the monthly averages and >4.2 °C for the daily average. Corresponding air temperatures were 4.8–6.7 °C for monthly averages and >−4.2 °C for the daily average (Fig. 4).

Due to additional denitrification in the pits under the barns, a mass balance was required to understand system removal of total N. Mass balance utilized nutrient concentration (Table 3) as well as corresponding water flows (Table 2). Oxidized N contained in the recycled water was reduced from 224 mg/l to 1 mg/l after 7-day retention in the pits under the barns. We calculated that an additional 870 kg of oxidized N was removed by denitrification in this closed loop during the 10.5-month period summarized in Tables 2 and 3. The amounts of total N (TKN + oxidized N) contained in the flushed manure and the treated effluent were 19,100 kg and 2020 kg, respectively. Thus, total N removal on a mass basis ($TN_{in} - TN_{out}$) was 89.4%. A significant amount of N was further removed by denitrification in the former lagoon that stored the final effluent produced by the treatment plant. For example, oxidized N in the system effluent was reduced from 241 to 11 mg/l after storage in the former lagoon (average June 2003–May 2004), with lower final concentration (average 2 mg/l) during warmer months and higher final concentration (average 20 mg/l) during coldest months, thus indicating a biological process. This additional N removal by denitrification in the former lagoon increased total N removal efficiency of the system from 89.4% to 97.9%. Thus, when the new treatment system is retrofitted into a typical North Carolina facility and the old lagoon is used for water storage, removal of N by de-nitrification during final storage is an important consideration for total N removal design of the entire system.

The biological N removal system generated very little amount of waste sludge. This is because most of the organic compounds were separated by the liquid–solids separation before NDN treatment. All the separated biological sludge solids left the farm mixed in the manure solids, and the separated liquid was returned to the biological N system. Biological sludge was wasted every day by diverting <1 m^3 of the return sludge from the settling tank into the homogenization tank for dewatering in the solid–liquid separation module (Fig. 1). A total 24.54 m^3 of sludge was wasted per month with an average TSS concentration of 6346 mg/l that contributed 145 kg of dry solids per month to the separated manure solids (93% separation efficiency). Thus, the waste sludge from NDN process contributed only 1.4% to the total amount of separated waste (596,200 kg containing 18.2% solids in 10.5 months, Section 3.3).

3.5. Soluble phosphorus separation module

Removal efficiencies of the soluble phosphate using the P-removal module averaged 94% for wastewater containing 77–191 mg/l PO_4–P (Table 3). The process is based on the distinct chemical equilibrium between phosphorus and calcium ions when natural buffers are substantially eliminated (Vanotti et al., 2003b). It was discovered that reduction of carbonate and ammonium buffers during nitrification substantially reduces the $Ca(OH)_2$ demand needed for optimum P precipitation and removal at high pH (Vanotti et al., 2005b). For example, the biological N removal step eliminated 97% of the NH_4^+–N and substantially reduced bicarbonate alkalinity (from 4345 to 529 mg/l) which, in turn, affected the succeeding P separation step by promoting formation of calcium phosphate with smaller amounts of lime added. The average lime consumption to reach the set point pH of 10.5 was 567 g/m^3.

The high pH (10.5) in the phosphorus removal process is necessary to produce calcium phosphate and kill pathogens (Vanotti et al., 2005a). The liquid is poorly buffered, and the high pH in the effluent decreases readily once in contact with the air. For example, treatment of 1 l liquid effluent using 2 l/min aeration in bench studies reduced the pH from 10.5 to <9 in about 2 h (Vanotti et al., 2003a). However, natural aeration during storage may be equally effective to lower pH.

A total of 285 bags of calcium phosphate product containing 526 kg of P was produced and left the farm during a 9-month period. The concentration grade was $24.4 \pm 4.5\%$ P_2O_5. Each bag weighed an average of 34.8 kg and contained 8.1 kg of dry matter (23.3% solids and 76.7% moisture). The phosphorus was >90% plant available based on standard citrate P analysis used by the fertilizer industry.

3.6. Electrical power use

Data in Table 5 show the electrical power use by each process unit and the entire system in both kW h/day (first column) and kW h/m^3 to compare with other processes. A total of 404 kW h/day was needed to operate the treatment system on the 4360-pig farm. The separation portion of the treatment consumed 37% of the total power used by the system; 36% of this (54.17 kW h/day) was used to mix manure in the homogenization tank, while the remainder (94.6 kW h/day) was used to operate the separation equipment (pumps, polymer mixer, rotating screen, DAF, and filter press). The biological N removal module consumed 57% of the total power (230.27 kW h/day); 59% of this (136.62 kW h/day) was used to power the air blower, and the remainder was consumed by mixers and pumps. The phosphorus separation module consumed <6% of the total power, and <1% was used to flush the barns and recycle the water to the barns.

Table 5
Electrical power use by the wastewater treatment system

Unit process	Power consumption per process unit and system (kW h/day)	Power consumption per m^3 of wastewater treated[a] (kW h/m^3)
Barn flush (lift station) and recycle to barns	2.60	0.050
Homogenization tank	54.17	1.389
Solids separation	94.60	2.426
Biological N treatment	230.27	5.889
Phosphorus treatment	22.30	0.842
Total system	403.94	10.357

[a] Volumes treated are shown in Table 2. Total system calculation uses total raw flushed manure volume (30 m^3/day).

3.7. Operator requirements

A manual of operation and maintenance was developed as part of the demonstration. The system requires an operator with a high-school education. The operator needs to receive 2 weeks training by the company that includes detailed information on plant equipment, operation and maintenance, safety and health aspects, identification and reporting of malfunction, and simple troubleshooting. Our observations indicate that a trained operator can safely operate two farms within a 20-mile radius, each farm providing treatment to 4500–9000 pigs. In addition to the plant operator, successful operation of the technology also requires support from an engineer technician having a 2–4-year engineer technology degree and mechanical/electrical skills. This person can provide support to about 10 farms so that each plant is visited about twice a month to work on specialized maintenance issues such as system checks, software, electronics, or parts replacement.

4. Conclusions

Treatment technologies are needed that can replace lagoons, capture nutrients, reduce emissions of ammonia and nuisance odors, kill harmful pathogens, and generate value-added products from manure. A system of swine wastewater treatment technologies was developed to accomplish all of these tasks. The system was tested at full scale in a 4400-head finishing farm as part of an Agreement between the Attorney General of North Carolina and swine producers Smithfield Foods and Premium Standard Farms to replace current anaerobic lagoons with Environmentally Superior Technology.

Major goals in the demonstration and verification of a new wastewater treatment system for swine manure were achieved including replacement of anaerobic lagoon treatment and consistent treatment performance under cold and warm weather conditions, with varying solid and nutrient loads typical in animal production. The on-farm system greatly increased the efficiency of liquid–solid separation by polymer injection to increase solids flocculation. Nitrogen management to reduce NH_3 emissions was accomplished by passing the liquid through a module where bacteria transformed NH_4^+ into harmless nitrogen gas. Subsequent alkaline treatment of the wastewater in a P module precipitated P and produced a disinfected liquid effluent.

It was verified that the treatment system was technically and operationally feasible. Based on performance results obtained, it was determined that the treatment system met the Agreement's technical performance standards that define an Environmentally Superior Technology (Williams, 2004). These findings overall showed that cleaner alternative technologies can have significant positive impacts on the environment and the livestock industry. This project was considered an important milestone in the search of alternative treatment technologies in the USA and justified moving ahead with innovation and evaluation of second-generation systems.

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ATTACHMENT 59

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ORIGINAL RESEARCH
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High-Rate Solid-Liquid Separation Coupled With Nitrogen and Phosphorus Treatment of Swine Manure: Effect on Water Quality

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This study determined the water quality improvements in swine lagoons by an innovative swine manure treatment system operating at full-scale during five pig production cycles. The system performed high-rate solid-liquid separation, biological ammonia treatment and phosphorus treatment. The treatment system met the environmental performance standards for swine waste management systems in new or expanding operations in North Carolina. The system substantially reduced odor by 99.9%; pathogens by 99.99%, nutrients (phosphorus and nitrogen) by $>$ 90%, and heavy metals by 99%. As the treated effluent and/or rainwater renovated the liquid in the anaerobic lagoons, they became aerobic (Eh $>$ 300 millivolts). By the end of the second year, the NH_4-N concentration in the lagoons liquid declined from the initial 370–485 mg L^{-1} to lower than 15. After conversion, the sludge accumulation in the former lagoons was halted. This was a significant outcome because one converted lagoon served twice the number of animals than before implementation of the innovative manure treatment system, which is similar to a situation of herd expansion. These findings showed that environmentally superior waste management technologies can have substantial positive impacts on water quality in intensive swine production.

Keywords: water quality, solid-liquid separation, flocculants, nutrient recovery, swine lagoons, pig manure, nitrification, recovered calcium phosphate

INTRODUCTION

Typically, waste from confined swine production operations in the southeastern U.S. is stored and treated in large, open anaerobic lagoons prior to application on cropland (Barker, 1996a; Westerman et al., 2010). After year 2000, there was great public interest in developing new swine manure treatment systems in the region to address multiple environmental and health concerns associated with the anaerobic lagoon system. These concerns included emissions of ammonia (Aneja et al., 2008), pathogens (Sobsey et al., 2001; Vanotti et al., 2005), odors (Schiffman and Williams, 2005; Loughrin et al., 2006), and deterioration of water quality (Mallin, 2000). Consequently, demonstrations of new treatment systems were conducted on-farm to demonstrate feasibility of environmental superior waste management technologies (EST) that could address five environmental standards: "1. Eliminate the discharge of animal waste to surface waters and groundwater through direct discharge, seepage or runoff; 2. Substantially eliminate atmospheric emissions of ammonia; 3. Substantially eliminate the emission of odor that is detectable beyond the boundaries of the parcel or tract of land on which the swine farm is located; 4. Substantially eliminates the release of disease-transmitting vectors and airborne pathogens; and 5. Substantially eliminates nutrient and heavy metal contamination of soil and groundwater" (Williams, 2009). Nutrients of concern were nitrogen (N) and phosphorus (P), and heavy metals of concern were copper (Cu) and zinc (Zn). As a result of this process, new legislation in North Carolina was enacted enforcing the environmental performance standards of EST for the construction of new swine farms or expansion of existing swine farms (NC Legislature, 2007; 15A NCAC 02T, 2010; Sommer et al., 2013).

Typically, the separation efficiencies of mechanical solid–liquid separators are less than 68% (Chastain et al., 2001) and typically less than 34% (Riaño and García-González, 2014). Organic nutrients (N and P) are contained mostly in fine manure particles < pore size 0.3 mm (Vanotti et al., 2002) that are not separated with normal screening (Riaño and García-González, 2014). However, new advances in both equipment and flocculant polymer applications developed for high-rate separation treatment have improved removal efficiency of solids and plant nutrients (N and P) (Vanotti and Hunt, 1999; Hjorth et al., 2010; Chastain, 2013). More and more often, new treatment systems for manure encompass three or four process units in tandem to meet various environmental standards and nutrient recovery targets. The swine waste management system described in this work is a manure treatment system developed to meet the EST environmental standards referenced above (Vanotti et al., 2010). The system consisted of high-rate solid-liquid separation followed by ammonia treatment and phosphorus recovery. A detailed description of this system as well as system drawing, first year performance of the treatment plant, and economic considerations, are found in Vanotti et al. (2009). While treatment performance of the system *per se* can be correctly assessed at steady state over relatively short periods during cold and warm weather conditions, its effect on water quality needs longer periods of monitoring due the large volume and slow hydraulic retention time of existing lagoons. In a previous study, we were able to evaluate side by side the water quality of a swine lagoon (total volume 24,145 m^3) being cleaned with the treated effluent from a multiple-stage treatment system (4,360-head swine unit) compared with an identical control anaerobic swine lagoon receiving raw effluent from another 4,360-head swine unit (Vanotti and Szogi, 2008). Results of that evaluation showed the converted lagoon was transformed into an aerobic reservoir (dissolved oxygen, DO, 6.95 mg L^{-1}) within a year, and by the second year, the following concentration reductions in the lagoon liquid were realized: 73% of total suspended solids (TSS), 77% of biochemical oxygen demand (BOD), and 92% of ammonium (NH_4-N) (Vanotti and Szogi, 2008). In the present study, the multi-stage treatment system (performing the same environmental functions as before) was retrofitted into a 5,145-swine farm that for the previous 11–12 years used two anaerobic lagoons (16,552 and 13,120 m^3) to treat the manure (**Figure 1**). With the implementation of the new system, one lagoon was discontinued, and the other lagoon was used as effluent storage and received the effluent from all the barns after treatment in the new plant (**Figure 2**). This lagoon served twice the animals as before, similar to a situation of herd expansion. Thus, it presented unique conditions that have not been experimented before or could be predicted without experimental data on water quality.

FIGURE 1 | Aerial picture of waste treatment system and barns. It treated all the manure flushed from seven barns with 735 pigs each. Existing lagoons are shown in the foreground.

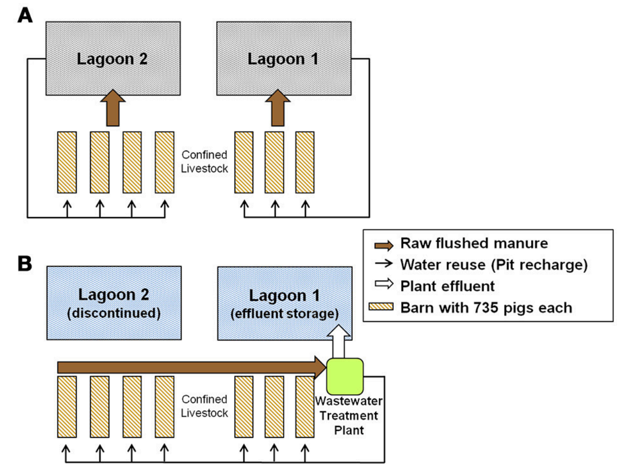

FIGURE 2 | Wastewater treatment system retrofitted into a 5,145-head swine finishing farm. **(A)** The previous system that used two anaerobic lagoons (0.6 ha each) for manure treatment and storage. **(B)** After installation of the new system, all the flushed manure was treated in the new plant and the treated effluent was stored in lagoon 1 while lagoon 2 was discontinued.

The objectives of this study were to: (1) Report the overall treatment efficiency consisting of high-rate solid-liquid separation followed by ammonia and phosphorus treatment evaluated intensively at steady state over a 2 years period and five swine production cycles. (2) Report the corresponding water quality improvements in the converted lagoons. (3) Report the changes in the sludge depth of the lagoons during a 6 years period of the new system operating at full-scale.

MATERIALS AND METHODS

Farm Description

The full-scale manure treatment system was installed on a swine farm near Clinton, Sampson Co., North Carolina and evaluated intensively with regards to water quality during 2 years under steady-state conditions that included five complete swine production cycles. The evaluation monitored the treatment plant and lagoons. Changes in the sludge depth in the lagoons were not clear in the first 2 years. For this reason, measurements of sludge depth were collected and reported for an additional 4 years of operation of the new treatment system.

The farm had 12.9 ha (32 acres) with a permitted capacity of 5,145-head feeder-to-finish swine placed in seven barns (735 heads/barn). The traditional anaerobic lagoon system (Barker, 1996a; NRCS, 2004), which is typical in North Carolina, was used for about 11 years before the new treatment system started operation in Dec. 2006. Production records during the three growing cycles before the start of the new plant showed the farm produced an average of 584,000 kg total live weight (487,000 kg net gain production) per growing cycle (5,296 pigs/cycle). The manure was collected under the barns using slatted floors and a pit-recharge system (Barker, 1996b). The liquid manure contained in the pits was emptied weekly by gravity into the anaerobic lagoons. There were two anaerobic lagoons for treatment and storage of the manure flushed from the barns (**Figure 2A**). Lagoon 2 received the flushes from four barns (1–4); it had a surface of 0.62 ha, a depth of 3.66 m and a design volume of 16,552 m^3. This volume included (1) a minimum treatment volume of 11,240 m^3 based on Steady State Live Weight (SSLW) (2,940 head × 61.24 kg/head =180,045 kg) and anaerobic treatment volume guidelines of 6.243 m^3/100 kg SSLW (1 ft^3/lb SSLW), and (2) a temporary storage volume of 5,016 m^3 based on waste volume generated stored for 180 days (0.00849 m^3/100 kg SSLW/d or 0.00136 ft3/lb SSLW/d), a positive balance of rain – evaporation (17.78 cm) and an additional 17.78 cm storage for a "25-year, 24 h" storm event. Lagoon 1 received the flushed raw manure from three barns (5–7); it had a surface of 0.60 ha, a depth of 2.74 m and a design volume of 13,120 m^3. This volume included (1) a minimum treatment volume of 8,433 m^3 based on SSLW (2,205 heads × 61.24 kg/head = 135,034 kg) and same anaerobic treatment volume guidelines, and (2) a temporary storage volume of 4,312 m^3 also based on waste volume generated stored for 180 days, rain – evaporation of 17.78 cm, and a "25-year, 24-h" storm storage of 17.78 cm. After treatment in the lagoons, the liquid was sprayed onto the farms's fields growing small grains and forages with a permitted capacity to utilize a total 5,390 kg of plant available N per year (average N application rate of 417.8 kg N/(ha.year). The lagoon supernatant liquid was recycled into the subfloor pits to facilitate waste flushing (**Figure 2A**).

With the new treatment system, the flow of raw wastewater into the lagoons was discontinued; instead, all the raw wastewater was sent to the treatment plant (**Figure 2B**). The barn pits were flushed once a week as before, but the flushed manure (barns 1–7) was diverted into a homogenization tank that mixed the manure before the solid-liquid separation step. A portion of the water after ammonia treatment was used to recharge the barn pits for the flushing (**Figure 3**). Water in excess of that needed for barn pit recharge was treated in the phosphorus treatment + disinfection module and stored in lagoon 1 for use in crop irrigation. As mentioned before, the new treatment system was evaluated intensively during five growing cycles of pigs (**Figure 4**). Within production cycles, the total pig weight in the seven barns varied greatly, from a low monthly average of 71.8 Mg to a high of 519.3 Mg (**Figure 4**). The average live animal weight (LAW) in the seven barns during the 5-cycle evaluation period was 335.8 Mg. This value is also referred to as steady-state live weight (SSLW, dotted line **Figure 4**). During the 5-cycle evaluation period, the farm sold an average of 624,345 ± 21,867 kg total live weight (516,239 ± 13,791 kg net gain produced) in each of the five growing cycles (5,265 ± 132 pigs/cycle).

The water quality of the two lagoons was monitored intensively during a 3 years period: the year before the project started when both lagoons performed anaerobic treatment, and the following 2 years when lagoon 1 received effluent from the alternative treatment system and lagoon 2 received only rain water. To help comparison of results of water quality monitoring, the same 36 months scale was used in the figures in this paper. After the intensive 3 years evaluation by the ARS

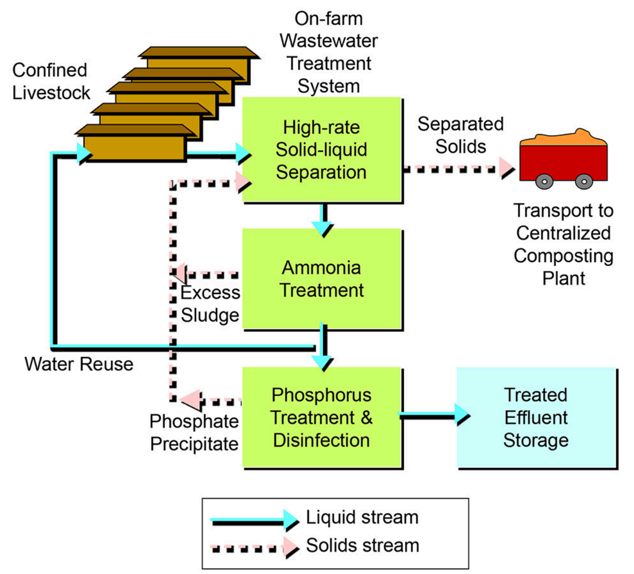

FIGURE 3 | Schematic diagram of wastewater treatment system using high-rate solid-liquid separation, ammonia treatment and phosphorus treatment.

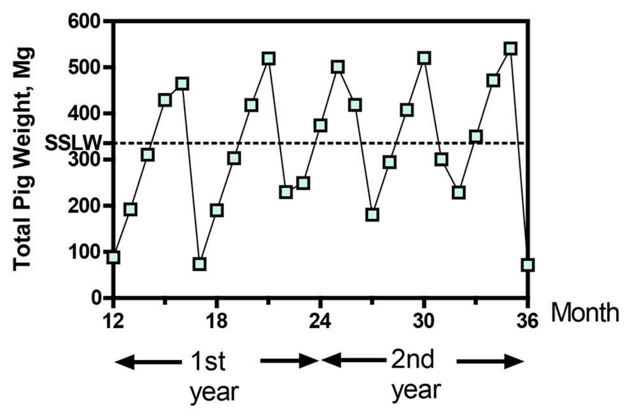

FIGURE 4 | Average monthly total live weight of the pigs during five production cycles in seven barns during demonstration of new wastewater treatment process. Dotted line is the steady state live weight (SSLW).

team (2006–2008), the treatment system kept operating full-scale without changes in swine inventory for an additional 4 years. As part of the permit No AWI820164 for using the innovative animal waste management system, the operator had to report to the State Permitting Authority the following parameters: daily volume of separated solids, the quarterly total volumes of the system wastewater influent and effluent, and quarterly chemical analyses of Total N, NH_4-N, NO_3+NO_2-N, BOD_5, Total P, TS, Cu, Zn, pH, and fecal coliforms in the separated solids (except BOD_5), the influent into the homogenization tank and the plant effluent from the phosphorus settling tank. After 3 years of successful compliance, the frequency of sampling and chemical analyses was reduced to two times per year to demonstrate compliance in both summer and winter seasons. During this extended period, the analyses were also done at ARS laboratory.

Wastewater Treatment System Description

The multistage system (Vanotti et al., 2010) consisted of three steps or process units in tandem: high-rate solid-liquid separation, biological ammonia treatment, and phosphorus treatment/disinfection (**Figure 3**). For a completed description and the schematic drawing of this system, see Vanotti et al. (2009). Before the first step, subfloor wastewater was emptied weekly by gravity into a receiving pit and pumped by a 946 L min^{-1} pump into a 379 m^3 capacity homogenization tank. The manure was kept well mixed using a 3.5 kW, 12.1 m^3 min^{-1} submersible mixer. The homogenized liquid manure was conveyed into the first step in the system—the liquid/solid separation process—at a process flow of 9.1 m^3 h^{-1}. The process used polymer flocculation to enhance the separation of fine suspended particles (Vanotti and Hunt, 1999; Garcia et al., 2007). Solids were separated by a rotary press separator (Fournier Industries Inc., Quebec, Canada) with a dual 1.2 m rotary press, two polymer preparation tanks, a polymer metering pump, manure feed pump and in-line flocculator. The polymer was dry cationic, linear polyacrylamide (PAM) with 35% mole charge (SNF Floerger, Riceboro, GA). The prepared polymer solution contained 2.14 g PAM L^{-1} (0.2%) and was mixed with the liquid manure at a rate of 6%. This resulted in a final polymer dosage of 128 mg L^{-1}. Separated manure solids were transported offsite to a solids processing facility and composed with cotton gin waste to produce value-added organic fertilizers, soil amendments and plant growth media (Vanotti et al., 2006). The separated wastewater was stored in another 379 m^3 tank and then pumped continuously into the second step of the system – the biological ammonia treatment process. This process used nitrification-denitrification (NDN) with a Modified Ludzack-Ettinger (MLE) configuration (Tchobanoglous et al., 2003). There were three tanks in the second step: the first tank was an anoxic tank (277 m^3) for denitrification (DN), the second tank was an aeration tank (227 m^3) for nitrification, and the third tank was a settling tank for clarification. Nitrification converted NH_4-N into NO_2-N and NO_3-N. The nitrified wastewater was continually recycled into the DN tank using the pre-denitrification configuration (MLE). In the DN tank, suspended denitrifying bacteria used soluble manure carbon in the separated manure to transform NO_2 and NO_3 into N_2 gas. The nitrification process used high-performance nitrifying bacterial sludge (HPNS), which was developed for high-ammonia wastewater and cold temperatures (Vanotti et al., 2013). To start the nitrification process, the aeration tank was inoculated with one liter of HPNS. Then a multiplication step was conducted in the same tank during 40 days to achieve full-scale nitrification activity prior to starting the complete system (Vanotti et al., 2009). Air was supplied continuously with a 10 HP rotary lobe blower and 98 fine-air diffusers. The DN tank contained a 1.7 kW, 9.8 m^3 min^{-1} submersible mixer. A settling tank (14.3 m^3) with conical bottom clarified the effluent after nitrification. The settled sludge was returned into the DN tank. The rates of sludge and nitrified liquid recycling into the DN tank were 0.5 and 3.5 times the inflow rate, respectively. The clarified effluent was stored in a 277 m^3 tank and used to refill the barn pits (**Figure 2B**). The average mixed liquor suspended solids (MLSS) and mixed liquor volatile suspended solids (MLVSS) in the nitrification tank during evaluation were 2,450 $\pm$ 1,680 mg L^{-1} and 1,980 $\pm$ 1,440 mg L^{-1}, respectively. Target MLSS concentrations were $>$ 2,000 and $<$ 4,000 mg L^{-1}. Once a week, the operator used a settling test (15 min, 1 L graduated cylinder) to estimate the MLSS in both denitrification and nitrification tanks based on an empirical relationship obtained at the site: [settled solids vol. (mL L^{-1}) = $-66.7 + 0.1132$ MLSS (mg L^{-1}); $r^2 = 0.759$] so that the settled solids volume in the 15 min test stayed between 160 and 390 mL L^{-1} (corresponding to the 2,000–4,000 mg MLSS L^{-1} target). This information was used by the operator to divert more or less sludge from the settling tank into the solids separator up-front to meet the MLSS target range. Considering a specific nitrification activity of 20.76 mg N/g MLVSS/ h and a nitrification tank volume of 227 m^3, the nitrification capacity of the unit was 223 kg N/day. In the third step of the system—the phosphorus treatment/disinfection process - the soluble P was recovered as a calcium phosphate solid (Vanotti et al., 2003), and pathogens were substantially reduced by the alkaline environment (Vanotti et al., 2005). The effluent was treated with hydrated lime slurry [12% $Ca(OH)_2$] in a 0.3 m^3 reaction chamber. The pH of the process was maintained at 9.5 by a pH probe and pH controller linked to the lime injection pump. The average lime consumption rate was 1.18 kg m^{-3}. The P precipitate (calcium phosphate) was separated in a settling tank with conical bottom (8.8 m^3) and further dewatered using the solid/liquid separation unit in the first unit of the system (Garcia et al., 2007). Phosphorus and manure solids as well as excess NDN sludge were combined in one solids stream for off-farm transport (**Figure 3**).

The average wastewater flows though the new treatment system (2 years averages) were the following: 36.3 m^3 d^{-1} of raw manure were flushed from the barns and treated (plant influent); 6.2 m^3 d^{-1} after N treatment were recycled to refill the barn's pit recharge system; 31.6 m^3 d^{-1} after P treatment (plant effluent) were stored in lagoon 1 for use in crop irrigation. On average, the flushed manure volume from the barns contained 17.1% recycled effluent from the treatment system and 82.9% of newly generated manure, urine, and water wasted by pigs. The newly generated wastewater stream (flushed manure—water reuse) averaged 30.1 m^3 day^{-1} or 40.8 L per 455 kg live animal weight (LAW) per day. For comparison, the industry average in feeder to finish operations using pit-recharge systems is 45.8 L per 455 kg LAW per day (1.62 ft^3 per 1,000 lbs. LAW per day) (Chastain et al., 1999).

Water Sampling and Monitoring

For the treatment system, composite liquid samples were collected twice per week during a 2 years period from four locations: (i) the homogenization tank containing raw flushed manure (plant influent), (ii) after solid-liquid separation treatment, (iii) after N treatment, and (iv) after P treatment (plant effluent). Samples were composited of four sub-samples taken over 3.5 days periods using refrigerated automated samplers (Sigma 900max, American Sigma, Inc., Medina, NY). Wastewater flows throughout the system were measured with five liquid-level ultrasonic probes and data logger (SR50 Sonic Ranging Sensor and CR800 data logger, Campbell Scientific Inc., Logan, UT). The ultrasonic probes measured liquid levels in the homogenization tank, separated water tank, clean water tank, and settling tank. The measurements of liquid height and area of the tanks were used to calculate actual volume dynamics and flows. The data logger also monitored air and water temperatures, precipitation, DO, ORP, and process pH. Average monthly maximum, average and minimum air temperatures and total monthly rain are shown in **Figure 5**. Average monthly minimum and maximum of daily air temperatures ranged from −8.4 to 38.8°C, average monthly air temperature was 16.5°C, and it ranged from 5.5 to 27.2°C, and precipitation averaged 1,048 mm per year.

Lagoon liquid samples were collected monthly during a 3 years period to monitor water quality characteristics at least 1 year before and 2 years after the treatment system was implemented. Sub-samples were collected from the lagoon supernatant within a 0.30 m depth using a 500 mL polyethylene dipper with a 3.6 m handle. From each lagoon, two composite

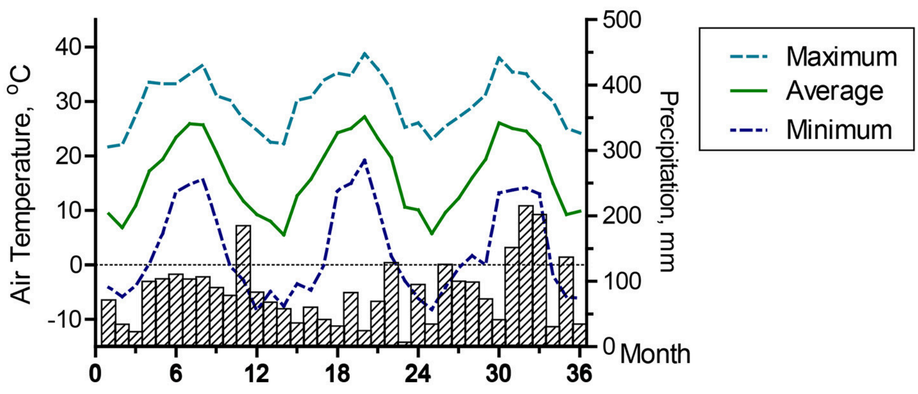

FIGURE 5 | Air temperature and precipitation during the 3-yr water quality monitoring period. Data are monthly maximum, average, and minimum of daily air temperatures, and monthly precipitation.

samples were obtained by mixing in a bucket eight sub-samples collected around the lagoon.

Collected samples were: (1) transported on ice to the ARS Coastal Plains Research Center in Florence, SC, for water quality analyses, or (2) overnight shipped with cold packs to the ARS Sustainable Agricultural Systems Laboratory and Environmental Microbial Safety Laboratory in Beltsville, MD, for microbiological analyses, and to the ARS Animal Waste Management Research Unit in Bowling Green, KY, for odor analyses.

The sludge depth in the lagoons was monitored yearly during 9 years: 3 years before and 6 years after the new system was implemented. The distance from the liquid surface level to the top of the sludge layer was measured with a sonar and the distance from the liquid surface to the lagoon bottom (soil) was measured with a pole. From 2004 to 2006, the sonar measurements were made from a boat at 8 or 10 sampling points per lagoon. Afterwards, the sonar measurements were done with a remote control boat that collected $1,150 \pm 170$ points per lagoon. Sludges were sampled five times to measure chemical characteristics using Sludge Judge probes 4.5 m long $\times$ 3.2 cm outside diameter (OD). Volume of sludge was calculated based on height using the formula volume of a trapezoid and design dimensions of the lagoons.

Analytical Methods

Water quality analyses were performed according to Standard Methods for the Examination of Water and Wastewater (APHA, 1998). Total solids (TS), total suspended solids (TSS), and volatile suspended solids (VSS) used Standard Method 2540 B, D, and E, respectively. Chemical analyses consisted of chemical oxygen demand (COD) using Method 5220 D, 5 days biochemical oxygen demand (BOD_5) using Method 5210 B, ammonia (NH_4-N) using Method 4500-NH_3 G, nitrate plus nitrite ($NO_3 + NO_2$ -N) using Method 4500-NO_3^- F, pH using Method 4500-H^+ B, electrical conductivity (EC) using Method 2510 B, alkalinity using Method 2320 B and endpoint pH of 4.5, soluble P (SP or PO_4) using Method 4500-P F after filtration through a 0.45-μm membrane filter. Total P (TP) and total Kjeldahl N (TKN) were determined using acid digestion (Gallaher et al., 1976) and colorimetric phosphorus and nitrogen methods adapted to acid digests (Technicon Instruments Corp, 1977). Organic P was the difference between total P and PO_4 analyses. Organic N was the difference between Kjeldahl N and NH_4-N analyses, and Total N was the sum of TKN and nitrate + nitrite. The potassium (K), calcium (Ca), magnesium (Mg), sodium (Na), copper (Cu), and zinc (Zn), were determined using nitric acid/peroxide block digestion (Peters, 2003) and inductively coupled plasma (ICP) analysis (Method 3125 A). Oxidation-reduction potentials (ORP) were measured at the time of sampling using a Ag/AgCl reference electrode and corrected to standard hydrogen electrode (Eh) values (Standard Method 2580 B). Reduction in odor was characterized as described by Loughrin et al. (2009) that measured in the liquid the concentration of five odor compounds characteristic of swine manure (phenol, *p*-cresol, *p*-ethylphenol, indole, and skatole) using extraction with Twister stir bars (Gerstel, Baltimore, MD) coated with polydimethylsiloxane followed by thermal desorption and gas chromatography-mass spectrometry. Total aromatic malodors were the sum of the five odor compounds. Microbiological analyses of liquid samples were done using standard protocols for pathogens and indicator microbes for the examination of wastewater (Vanotti et al., 2005).

Statistical Analysis

Data management, descriptive statistics (PROC MEANS), regression (PROC REG), and mean comparison for repeated measurements (PROC MIXED) analyses were performed with SAS (SAS Institute, 2008).

RESULTS AND DISCUSSION

Water Quality Improvements by Treatment System

The wastewater treatment performance of the plant are presented in **Table 1**; the various columns show changes in water quality indicators as the liquid manure passed through the three treatment steps as well as the overall system efficiency. The intensive evaluation period encompassed five cycles of pig production; this allowed assessment of the performance of the system through varying environmental conditions and waste loadings. High treatment efficiencies were obtained consistently despite air temperatures varying from −8.4 to 38.8°C (**Figure 5**) and large fluctuations in the strength of the manure. Taken on average through the evaluation period, flushed manure had high strength (TS $3.0 \pm 1.2\%$) according to the manure strength scale of Garcia-González and Vanotti (2015). The variation in concentrations because of changes in pig weight during production cycles was big: volatile solids, for example, averaged 17,800 mg L^{-1}, but ranged from 2,850 mg L^{-1} up to about 41,800 mg L^{-1} while BOD_5 averaged 7,360 mg L^{-1} and ranged from to 730 to over 25,000 mg L^{-1} (**Table 1**). Other quality parameters such as TKN (average 2,050 mg L^{-1}, range 810–4,220 mg L^{-1}) and NH_4-N (average 1,290 mg L^{-1}, range 310–3,445 mg L^{-1}) were also distinctive of a high-strength swine wastewater. The variations in wastewater volumes were also big because of the pig production cycles: average monthly volume of flushed wastewater was 1,095 m^3, but ranged from 396 up to 2,233 m^3. Similarly, the clean treated effluent volumes averaged 963 m^3 per month, and ranged from 290 to 1,666 m^3. In terms of mass loadings, the total nitrogen load into the treatment system (flushed manure) during the five pig cycles averaged 80.6 kg N d^{-1} (29,510 kg N yr^{-1}) and the monthly averages ranged from 20.9 to 211.5 kg N d^{-1}. The average NH_4-N load was 51.9 kg d^{-1} (range 13–150 kg d^{-1}), and the average TP load was 21.4 kg d^{-1}. The on-farm system removed 67.75% of TS, 90.2% of VS, 97.2% of TSS, 98.4% of VSS, 96.3% of COD, 99.4% of BOD_5, 95.7% of TKN, 96.5% of NH_4-N, 93.3% of TP, 98.8% of Zn, 98.8% of Cu, 56.0% of EC, and 77.5% of alkalinity (**Table 1**).

High-Rate Solid-Liquid Separation

The first step of the system was a high-rate solid-liquid separation via polymer flocculants (Chastain, 2013). The separation up-front allowed recovery of the organic materials in the manure, which can be utilized for the manufacture of composts, biochars,

TABLE 1 | Wastewater treatment plant performance by treatment step and overall system efficiency[a].

Water quality parameter	Raw flushed swine manure (system influent)	Treatment step: After solid-liquid separation treatment	After ammonia treatment	After phosphorus treatment (system effluent)	System efficiency
	mg L^{-1} [b]				%
TSS	11,754 ± 6,417	1,254 ± 1,015	227 ± 199	325 ± 215	97.2
VSS	8,926 ± 5,103	891 ± 756	154 ± 129	142 ± 105	98.4
TS	30,065 ± 12,475	14,244 ± 5,104	9,824 ± 2,312	10,008 ± 2,495	67.7
VS	17,799 ± 8,725	5,322 ± 2,893	1,818 ± 827	1,738 ± 1,046	90.2
COD	22,204 ± 14,363	8,196 ± 5,286	1,058 ± 541	821 ± 405	96.3
Soluble COD	7,338 ± 6,012	6,073 ± 4,098	862 ± 393	684 ± 308	90.6
BOD_5	7,364 ± 6,313	3,185 ± 2,692	62 ± 88	41 ± 61	99.4
TKN	2,054 ± 778	1,466 ± 600	138 ± 166	87 ± 130	95.7
NH_4-N	1,290 ± 615	1,213 ± 451	124 ± 171	45 ± 92	96.5
$NO_2 + NO_3$-N	1.4 ± 4.6	0.2 ± 1.5	221 ± 179	162 ± 144	–
Organic N	739 ± 447	230 ± 290	33 ± 38	36 ± 51	95.1
Total N	2,055	1,466	359	249	87.9
TP	492 ± 272	151 ± 79	83 ± 30	33 ± 23	93.3
Soluble P	94 ± 63	82 ± 42	76 ± 29	19 ± 17	79.8
Organic P	380 ± 259	62 ± 63	11 ± 12	12 ± 14	96.8
K	1,648 ± 562	1,551 ± 475	1,420 ± 371	1,443 ± 370	12.4
Ca	417 ± 196	106 ± 58	39 ± 18	90 ± 95	78.4
Mg	219 ± 110	44 ± 30	16 ± 7	12 ± 5	94.5
Zn	25.4 ± 12.6	2.9 ± 2.8	0.4 ± 0.4	0.3 ± 0.3	98.8
Cu	16.8 ± 11.1	2.0 ± 2.4	0.2 ± 0.1	0.2 ± 0.1	98.8
Fe	39.9 ± 21.3	4.81 ± 4.55	0.49 ± 0.40	0.39 ± 0.35	99.0
S	128 ± 60	49 ± 19	34 ± 8	31 ± 8	75.8
Na	512 ± 215	487 ± 188	434 ± 152	441 ± 157	13.9
ORP, mV	−64 ± 72	6 ± 135	202 ± 177	ND[c]	–
Alkalinity, mg $CaCO_3$ L^{-1}	7,027 ± 2,175	5,469 ± 1,505	1,422 ± 1,013	1,580 ± 835	77.5
pH	7.80 ± 0.35	7.78 ± 0.23	7.98 ± 0.50	9.72 ± 0.69	–
EC, mS cm^{-1}	14.97 ± 4.36	14.09 ± 4.08	7.25 ± 1.91	6.58 ± 1.57	56.0

[a] *Data are means ± standard deviations for 122 sampling dates (2 years of continuous operation).*

BOD_5, 5 days biochemical oxygen demand; COD, chemical oxygen demand; EC, electrical conductivity; ORP, oxidation reduction potential; TKN, total Kjeldahl nitrogen; TP, total phosphorus; TS, total solids; TSS, total suspended solids; VSS, volatile suspended solids.

[b] *Except for ORP (mV), EC (mS cm^{-1}), and pH. ORP values are standard hydrogen electrode (Eh); measurements were done weekly in grab samples (n = 56).*

[c] *ND, Not Determined.*

and other value-added products. It also allowed treatment of the liquid through biological nitrogen steps and phosphorus recovery/disinfection in an economical way to meet specific environmental standards. Compared to the flushed manure, the separation process concentrated the suspended solids > 25 times. It produced a relatively dry manure cake with 24.9% solids. The capture of the fine suspended solids through flocculation resulted in large decreases in TSS (90%) and COD (63%) concentrations (**Table 1**). TKN and total P were reduced by approximately 30% and 70%, respectively. Organic N and P were reduced 69 and 84%, respectively. In contrast, the soluble ammoniacal nitrogen (NH_4-N) and soluble P were unaffected by the solids separation process. The high-rate solid-liquid separation was also effective reducing heavy metals Cu and Zn concentrations; this was one of the five environmental treatment objective of EST. Initial Cu and Zn concentrations (16.8 and 25.4 mg L^{-1}) were both reduced > 88% just with the high-rate solid-liquid separation.

Biological Ammonia Treatment

The NDN step using the MLE process configuration treated NH_4-N effectively. Nitrification was accomplished using high performance nitrifying sludge (HPNS) adapted to high-ammonia and low temperatures (Vanotti et al., 2013). The pre-denitrification configuration of the MLE process allowed suspended denitrifying bacteria to consume most of the COD and BOD_5 remaining in the wastewater after solid-liquid separation. The average ratio COD/TN of the manure liquid after solid-liquid separation was 5.6 and appeared a good balance for N removal in this system without external carbon addition. On average, the NDN step reduced COD by 87% and BOD_5 by 98% relative to their concentration after solid-liquid separation (**Table 1**). The average ammonia (NH_4-N) removal efficiency was high (average = 90%) in spite of large variations of influent NH_4-N concentrations (310–3,445 mg L^{-1}) and monthly NH_4-N loading rates (14.7–117.3 kg NH_4-N d^{-1}; average load = 47.7 kg d^{-1}). Average TKN removal efficiency was 91%. Influent TKN concentration varied from 810 to 4,220 mg L^{-1}, and the N mass loading rates varied monthly from 16.8 to 166.1 kg TKN d^{-1} (average TKN load = 58.7 kg d^{-1}). The process responded well to cold temperatures experienced during evaluation. Monthly average water temperatures during cold weather (Dec–Feb) were 9.1–11.3°C, and corresponding daily minimum average water temperatures were 8.0–9.1°C. The N removal performance in this study was consistent with the performance obtained by Riaño and García-González in a full-scale, on-farm treatment plant in Castilla y Leon region, Spain, that also used a combination of high-rate solid-liquid separation with flocculants and nitrification-denitrification to treat raw swine manure: nitrification-denitrification step alone removed 84.5% of COD, 95.9% of TKN and 98.0% of NH_4-N, while the combined system removed 97% of COD, 97% of TKN and 89% of TP. In France, a combination of solid-liquid separation using centrifuges and biological NDN treatment have been successfully established at large scale in approximately 300 units treating about 15% of the swine effluents produced in Brittany region to remove N surplus and also the P excess (Bernet and Béline, 2009).

The N removal unit produced a cleaner and oxidized effluent with 124 mg L^{-1} of NH_4-N, 221 mg L^{-1} of NO_3-N + NO_2-N, 227 mg L^{-1} of TSS, 62 mg L^{-1} of BOD_5, and ORP 202 mV (**Table 1**). Part of the N treated effluent was recycled on the farm to refill the pits under the barns and facilitate flushing (under the previous anaerobic lagoon system, the anaerobic lagoon liquid supernatant was used for the flushing). This recycling of clean water with low ammonia into the barns improved the environment in the barns that benefited health and productivity of the animals. Production records for the five growth cycles before and the five cycles after conversion of waste management technology showed improvements in several animal productivity and health indicators. With the new manure treatment system, the animal mortality was decreased 47%, daily weight gain was increased 6.1%, and feed conversion was improved 5.1%. As a result, the farmer sold 28,100 kg more hogs (a 5.8% increase) per growth cycle using the new system compared to the previous anaerobic lagoon management.

Phosphorus Recovery Treatment

The clarified effluent from the biological N removal step was treated with hydrated lime that precipitated the phosphorus at process pH of 9.5. Removal efficiencies of the soluble phosphate with the treatment system averaged 80% for wastewater containing an average of 94 ± 63 mg PO_4-P L^{-1} (**Table 1**). The overall treatment system (three steps) recovered 93.3% of the TP, with two steps contributing significantly: the high-rate solid-liquid separation (first step) removed the organic P efficiently (from 380 to 62 mg P L^{-1}), and phosphorus module (third step) removed the soluble P efficiently (from 76 to 19 mg PO_4-P L^{-1}).

Odor and Pathogen Reductions

The substantial elimination of malodorous compounds was an important environmental standard to meet. A complete odor evaluation in this system have been reported by Loughrin et al. (2009). Five characteristic aromatic malodor compounds (phenol, p-cresol, p-ethylphenol, p-propylenphenol, indole, and skatole) were measured in the liquid at the successive stages of the treatment system (**Table 2**). Results obtained showed a 99.9% reduction of total odors (the sum of concentration of the five malodor compounds) in the treated effluent compared to the untreated swine manure. The solid-liquid separation step was not efficient to separate the malodorous compounds in the flushed manure and 89% of these compounds remained in the liquid fraction. However, they were effectively destroyed during the

TABLE 2 | Removal of odor compounds and pathogen indicator microorganisms by on-farm wastewater treatment system using high-rate solids separation coupled with ammonia and phosphorus treatment.

	Raw flushed swine manure (system influent)	After solid-liquid separation treatment	After ammonia treatment	After phosphorus treatment (system effluent)	Removal efficiency with system
Odor Compounds[a]			ng mL^{-1}		%
Total	71,269 ± 14,733	63,642 ± 12,366	40 ± 17	44 ± 11	99.9
Skatole	2,943 ± 496	2,540 ± 420	0 ± 0	0 ± 0	100.0
Pathogen/pathogen indicators[b]			log10 cfu mL^{-1}		%
Total fecal coliforms (Mac+ 44.5)	4.11 ± 0.19	3.47 ± 0.16	0.84 ± 0.23	0.17 ± 0.18	99.99
Enterococci (mEnt)	5.11 ± 0.13	3.62 ± 0.18	1.53 ± 0.34	1.14 ± 0.35	99.99
Salmonella (XLT4)	1.79 ± 0.11	1.14 ± 0.30	0.00 ± 0.00	0.00 ± 0.00	100.00

[a] *Data are means ± standard error of 15 monthly determinations that included cold and warm weather months. Total odor compounds are the sum of concentrations of five malodorous compounds contained in the liquid (phenol, p-cresol, p-ethylphenol, indole, and skatole) that are characteristic of swine manure.*
[b] *Data are means ± standard error of log_{10} colony forming units (cfu) per mL for duplicate samples of six determinations that included cold and warm weather conditions.*

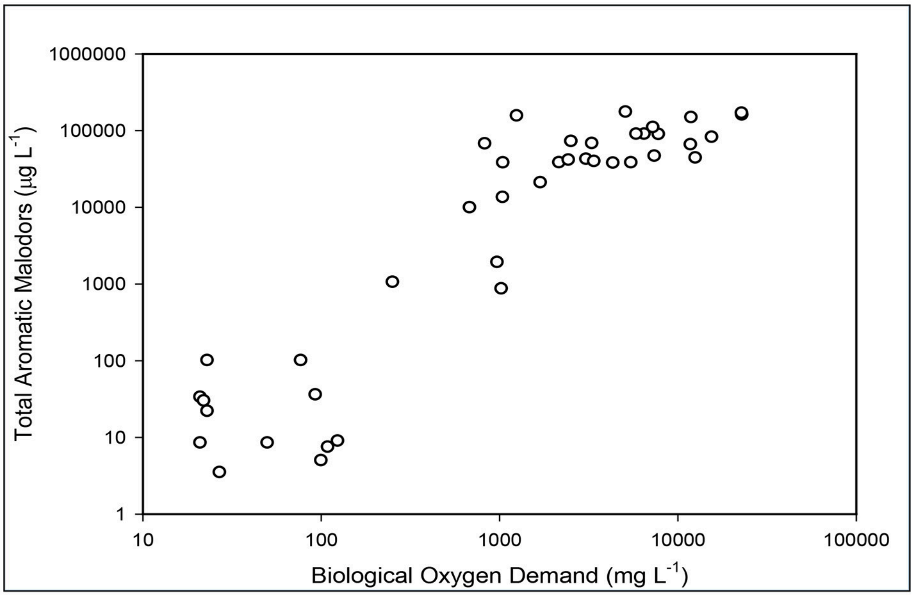

FIGURE 6 | Relationship between total odor compounds in the liquid and BOD_5 concentration as the liquid is being treated in the new plant.

subsequent biological ammonia treatment step. One important finding was that the concentration of total odor compounds in the liquid was related to BOD_5 concentration (**Figure 6**). This relationship was used later by the State Permitting Authority to determine the level of odor acceptable using this innovative animal waste management system, as a replacement of measuring odor intensity levels at the property level, which was more complicated to measure. It was stablished that, to demonstrate odor compliance, the BOD_5 concentration in the effluent samples shall not exceed 150 mg L^{-1}.

The substantial elimination of pathogens was another important environmental standard to meet. The multistep treatment system was efficient reducing pathogens in the liquid swine manure (**Table 2**). Results showed a steady reduction of microbial indicators and pathogens by each step in the treatment system. The largest reduction was obtained in the biological ammonia removal step (2.63 and 2.09 log10 reductions for total fecal coliforms and enterococci, respectively). The phosphorus treatment with its high pH provided a level of disinfection needed to meet the EST criteria of 4-log pathogen indicator reduction (99.99%). Salmonella, which was present in the raw manure at 1.79 log 10 cfu/mL, was eliminated by the second step in treatment system.

Water Quality Improvements in Lagoons

Initial Lagoon Conditions

Table 3 and **Figure 7** show the water quality changes in the two study lagoons during the 36 months monitoring period. **Table 3** show yearly changes of all the water quality parameters measured, and **Figure 7** show monthly changes of selected parameters. This monitoring period includes a common year before the project started (0–12 months) when both lagoons received raw manure directly from the barns (anaerobic lagoon management, **Figure 3A**) and the subsequent 2 years (12–36 months) when lagoon 1 received all the effluent from the new treatment plant, while lagoon 2 stopped receiving wastewater (raw or treated) (new manure management, **Figure 3B**). During initial conditions (0–12 months), the liquid characteristics in the two lagoons were similar as determined by water quality indicators shown in **Table 3**. The average TKN and NH_4-N concentrations (539–671 mg L^{-1} and 371–485 mg L^{-1}, respectively) were consistent with range values of 340–650 mg TKN L^{-1} and 280–570 mg NH_4-N L^{-1} reported for liquid in 10 swine lagoons in North Carolina (Bicudo et al., 1999). Under traditional management, the monthly average TKN concentrations varied significantly within a year, from a low of about 325 to a high 829 mg L^{-1} in lagoon 1 and 487–819 in lagoon 2 (**Figure 7**). The NH_4-N, which comprised 71% of the TKN, followed the same cyclic variation within a year. These N concentration cycles in the traditional lagoon followed seasonal temperature variations (**Figure 5**) with the lowest NH_4-N concentrations at the end of summer and highest at the end of winter. This is consistent with the previous study that monitored NH_4-N in traditional lagoon during 3 years (Vanotti and Szogi, 2008).

Lagoon Liquid Cleanup

In month 12 of the 3 years water quality monitoring period, manure flushes to both lagoons were halted and the conventional anaerobic lagoon treatment was discontinued. At that point, lagoon 1 received all the treated effluent generated by the new wastewater treatment plant. It went from receiving raw waste from 3 barns (permitted for 2,205-head feeder-to-finish swine) to receiving treated waste from 7 barns (5,145-heads). Lagoon 2 did not receive any effluent (treated or untreated), only rainwater,

TABLE 3 | Lagoon liquid analyses of two swine lagoons before and after implementation of new treatment system using high-rate solids separation coupled with ammonia and phosphorus treatment[a].

Water quality parameter[b]	Sampling period (3 years)							
	Year before the project started (traditional lagoon management)		1st year of new treatment operation		2nd year of new treatment operation		Lagoon effect	Time (year) effect
	Lagoon 1	Lagoon 2	Lagoon 1	Lagoon 2	Lagoon 1	Lagoon 2		
	mg L^{-1b}						Prob > t	
TSS	532 ± 189	435 ± 183	417 ± 150	424 ± 109	207 ± 94	290 ± 64	0.9412	<0.0001
VSS	417 ± 138	321 ± 117	320 ± 111	307 ± 86	140 ± 78	221 ± 77	0.7159	<0.0001
TS	11,709 ± 846	12,164 ± 2,750	9,728 ± 1,419	10,520 ± 1,346	8,174 ± 712	8,332 ± 1,335	0.2859	<0.0001
VS	2,968 ± 1,181	2,747 ± 1,050	1,960 ± 762	2,118 ± 742	1,420 ± 307	1,625 ± 326	0.8013	<0.0001
COD	2,298 ± 799	2,126 ± 301	1,548 ± 443	1,794 ± 206	907 ± 373	1,113 ± 404	0.3796	<0.0001
Soluble COD	1,390 ± 192	1,628 ± 311	1,068 ± 296	1,255 ± 102	668 ± 264	761 ± 233	0.0032	<0.0001
BOD_5	190 ± 150	219 ± 135	165 ± 130	195 ± 118	81 ± 80	71 ± 44	0.5518	<0.0001
TKN	539 ± 163	671 ± 108	291 ± 89	359 ± 103	140 ± 96	144 ± 76	0.0088	<0.0001
NH_4-N	371 ± 167	485 ± 128	181 ± 75	251 ± 117	65 ± 66	60 ± 53	0.0215	<0.0001
$NO_2 + NO_3$-N	0.0 ± 0.0	0.0 ± 0.0	1.1 ± 1.9	0.1 ± 0.4	11.5 ± 23.5	0.0 ± 0.0	0.0550	0.0697
TP	76 ± 15	83 ± 15	65 ± 15	73 ± 23	77 ± 14	86 ± 14	0.0556	0.0168
Soluble P	50 ± 6	54 ± 7	47 ± 10	52 ± 14	64 ± 13	67 ± 15	0.1244	<0.0001
K	1,391 ± 132	1,537 ± 132	1,327 ± 150	1,436 ± 192	1,388 ± 147	1,332 ± 177	0.1089	0.1099
Ca	46.6 ± 15.3	53.4 ± 19.0	51.1 ± 13.8	57.5 ± 15.5	32.6 ± 6.4	35.3 ± 6.5	0.1090	<0.0001
Mg	13.1 ± 2.01	13.4 ± 3.0	16.1 ± 5.3	17.7 ± 6.4	23.7 ± 6.4	28.7 ± 11.2	0.1291	<0.0001
Zn	0.71 ± 0.42	1.04 ± 0.58	0.69 ± 0.50	1.21 ± 0.56	0.37 ± 0.14	0.61 ± 0.14	0.0009	0.0011
Cu	0.49 ± 0.20	0.67 ± 0.32	0.17 ± 0.06	0.49 ± 0.12	0.09 ± 0.04	0.22 ± 0.08	<0.0001	<0.0001
Fe	2.29 ± 0.48	2.71 ± 0.40	1.43 ± 0.52	1.86 ± 0.34	0.95 ± 0.43	1.35 ± 0.34	<0.0001	<0.0001
S	38.6 ± 9.0	33.6 ± 14.1	46.5 ± 6.4	42.8 ± 15.1	27.5 ± 10.2	39.1 ± 13.1	0.7181	0.0021
Na	391 ± 45	442 ± 58	389 ± 54	447 ± 87	424 ± 70	398 ± 67	0.0822	0.9379
ORP, mV	60 ± 127	−4.8 ± 102	181 ± 168	179 ± 132	287 ± 126	287 ± 119	0.4901	<0.0001
Alkalinity, mg $CaCO_3$ L^{-1}	3,438 ± 1273	3,621 ± 470	2,360 ± 253	2,858 ± 356	1,817 ± 318	1,863 ± 371	0.0830	<0.0001
pH	8.19 ± 0.15	8.11 ± 0.20	8.20 ± 0.32	8.09 ± 0.29	8.28 ± 0.18	8.22 ± 0.19	0.1542	0.2469
EC, mS cm^{-1}	8.19 ± 1.38	9.37 ± 0.86	6.74 ± 0.40	7.58 ± 0.38	6.02 ± 0.68	5.66 ± 0.90	0.0046	<0.0001

[a]*Data are means ± standard deviations of monthly samples. During the previous year, both lagoons were managed as typical anaerobic swine lagoons receiving raw swine manure. Afterwards, lagoon 1 received all the swine effluent after being treated in the new plant, and lagoon 2 did not receive any swine influent (treated or untreated). Monthly changes for selected parameters are shown in* ***Figure 7****.*

[b]*Units in mg L^{-1} except for ORP, EC, and pH. ORP values are standard hydrogen electrode (Eh).*

and its situation resembles that of an inactive lagoon after depopulation of pigs (Sheffield, 2000). Excess water over storage capacity of the lagoons was applied onto crops and forages on the farm. Rainfall averaged 1,026 mm per year and contributed 7,500 m^3 of rain water annually to each lagoon (drainage area of each lagoon was 0.634 ha) or 13,000 m^3 of rainwater per lagoon in the 2 years period. Actual lagoon volumes were 9,565 ± 750 m^3 (lagoon 1) and 13,057 ± 1,180 m^3 (lagoon 2). Thus, rain alone renovated 70 and 50% of the total lagoon volumes per year (lagoon 1 and 2, respectively). Lagoon 1 received also the clean plant effluent, 11,552 m^3 per year (31.6 m^3 d^{-1}) with a renovation capacity of 1.2 lagoon volumes per year (HRT = 0.8 years).

Statistical tests showed significant reduction with time on most water quality parameters measured in the lagoons ($P <$ 001), an effect that was consistent across lagoons (**Table 3**). The exceptions were K, Na, TP, and pH, which were not significantly changed with time. In three instances when statistical differences ($P < 0.01$) in water quality parameters between lagoons occurred (Cu, Zn, and Fe, **Table 3**), the concentration decrease was quicker in lagoon 1 that received the treated water (**Table 3** and **Figure 7**). By the second year of new treatment operation, the following average reductions in water constituents were realized in lagoon 1 (**Table 3**): 61% of TSS, 66% of VSS, 57% of BOD_5, 74% of TKN, 82% of NH_4-N, 48% of Zn, 82% of Cu, 47% alkalinity, and 26% of EC. Corresponding reductions in lagoon 2 (inactive) were: 33% of TSS, 31% of VSS, 68% of BOD_5, 79% of TKN, 88% of NH_4-N, 41% of Zn, 67% of Cu, 49% alkalinity, and 40% of EC. Therefore, when multistep EST treatment technology is implemented in a swine operation with anaerobic lagoons, an additional environmental benefit is obtained: the progressive cleanup of the lagoon liquid without having to stop production.

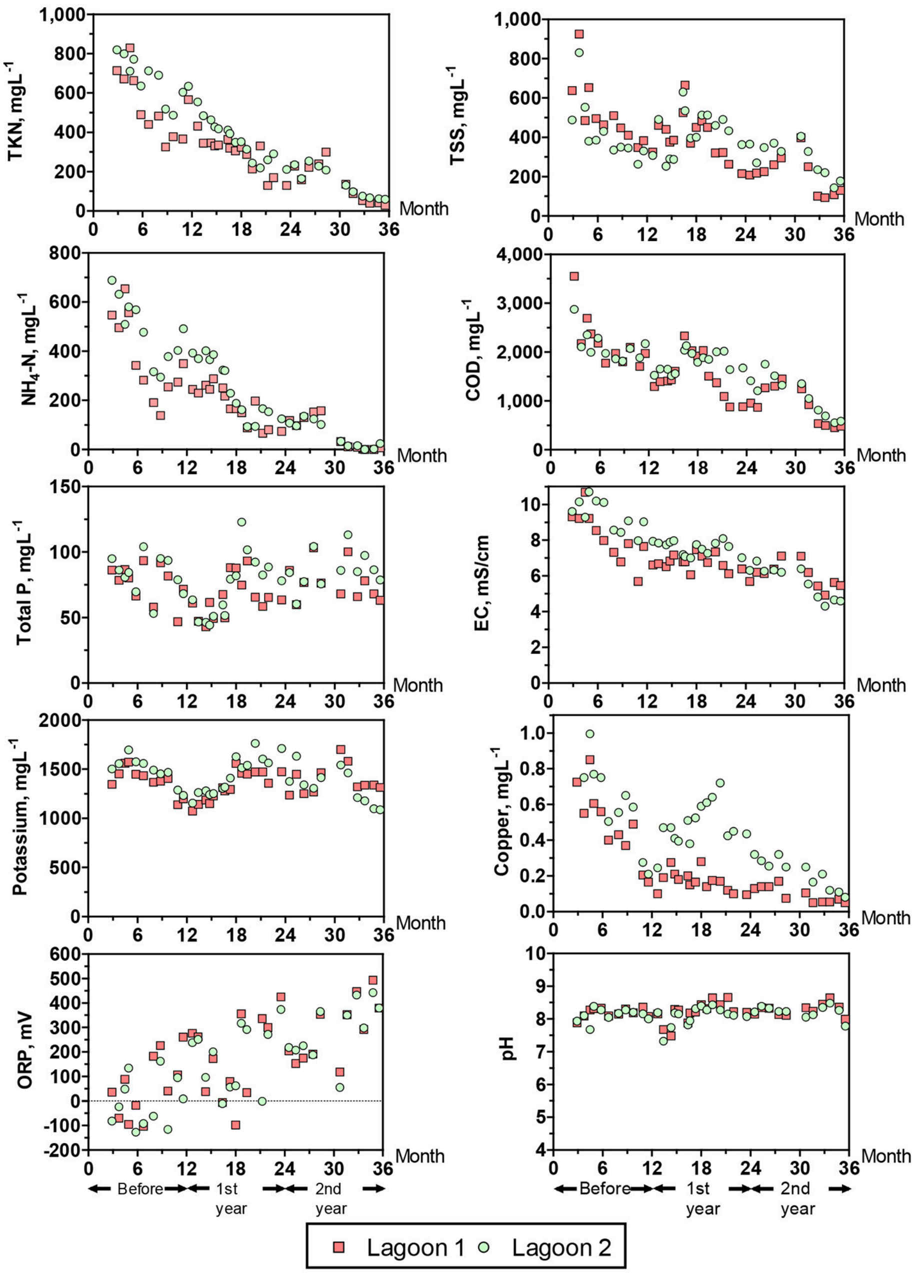

FIGURE 7 | Water quality changes in the two study lagoons during a 36 months period. 0–12 months = lagoons being managed as typical anaerobic swine lagoons receiving raw swine manure. Thereafter, lagoon 1 received all the swine effluent after treatment in the new plant, and lagoon 2 did not receive any swine influent (treated or untreated). Yearly averages are shown in **Table 3**.

Even though lagoon 1 served the production of more than twice the number of animals than it did before with the traditional lagoon system (average LAW increased from 144 to 336 Mg), remarkably, the overall cleaning performance of the new plant effluent on lagoon 1 liquid was similar to the cleaning performance by rainwater alone under lagoon inactivation and abandonment of production (lagoon 2). Indeed, the results of this study were used by the State Permitting Authority to issue Permit No AWI820164 using the innovative animal waste management system that would allow the expansion of total swine animal capacity in this farm from 5,145 to 11,015 feeder-to-finish using the same acreage (12.9 ha).

The NH_4-N concentration in the lagoons before the project started were 371 ± 167 mg L^{-1} in lagoon 1 and 485 ± 128 mg L^{-1} in lagoon 2 (**Table 3**). During the last 6 months the NH_4-N concentrations were very low: 10.0 ± 11.9 mg L^{-1} in lagoon 1 and

FIGURE 8 | Swine lagoon conversion into aerobic pond. Picture on the left shows Lagoon 1 under traditional management before start of the project, and picture on the right shows the same lagoon after the wastewater treatment plant (background) was in operation for about 10 months.

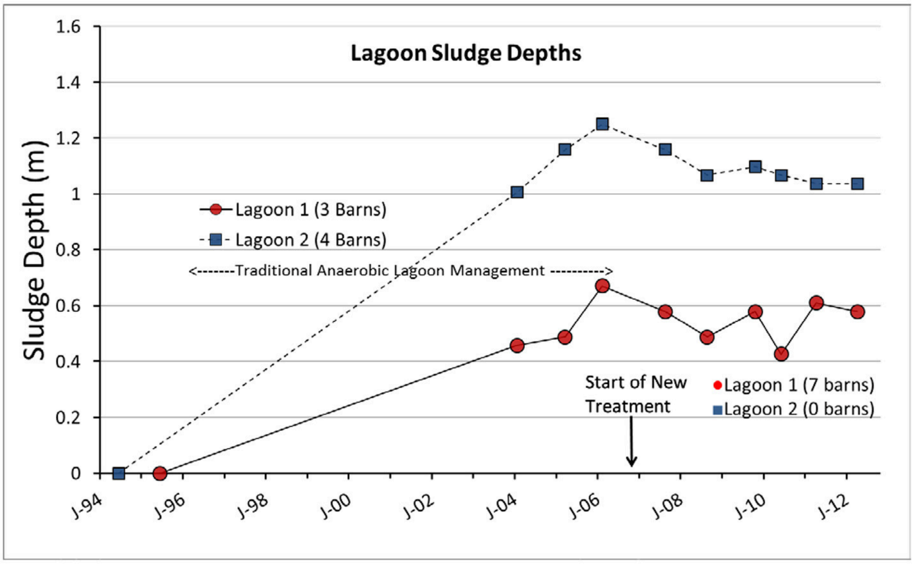

FIGURE 9 | Sludge depth dynamics of the two swine lagoons. The new treatment plant was installed after 11–12 years of conventional anaerobic lagoon treatment.

14.9 ± 12.5 mg L^{-1} in lagoon 2 (months 30–36, **Figure 7**); they approached average concentration of 4 mg NH_4-N L^{-1} reported for 30 lagoons in swine operations in North Carolina that were depopulated and inactive for 6 ± 4 years (Sheffield, 2000). In a companion paper, Ro et al. (2018) measured the ammonia emissions from this project including lagoons using open-path tunable diode laser and found that the ammonia emissions were reduced to below detectable levels. Another important parameter is the effect on EC (water salinity) that is an important water quality guideline on crop productivity, for example FAO (1994) guidelines indicate that the yield potential of irrigated cotton is reduced from 100 to 90% to 75 and 50% with irrigation water EC of 5.1, 6.4, 8.4 and 12 mS cm^{-1}, respectively. In the study, the average EC of the lagoon liquid before the project started was 8.19–9.37 mS cm^{-1} in lagoon 1 and 2, respectively. The EC was lowered to 5.33 ± 0.37 and 4.51 ± 0.19 mS cm^{-1} during the last quarter (**Figure 7**), which is optimal for cotton irrigation. As clean plant effluent and/or rain water replaced the liquid in the two lagoons, they become aerobic. From the point of view of microbial metabolism, a redox potential (ORP) > 300 millivolts is associated with aerobic, oxidized conditions (Reddy et al., 2000). The transition from anaerobic to aerobic, oxidized conditions took about 1.5 years. Average ORP were −4.8 to 60 millivolts during traditional management before the project started (**Table 3**); they increased steadily with time to reach consistent levels > 300 millivolts in the second half of second year (months 30–36, **Figure 7**). In addition to these chemical indicators of aerobic conditions, in 10 months of the new manure management the lagoon 1 changed color from brown to blue (**Figure 8**).

Changes in Sludge Accumulation in the Lagoons

Before the conversion and under traditional anaerobic lagoon management, the sludge in lagoon 1 accumulated to a depth 0.67 m (or 2,620 m^3) in 11 years of continuous swine production (serving 2,205-head feeder-to-finish) and sludge

TABLE 4 | Chemical composition of the lagoon sludges[a].

Component	Sludge lagoon 1	Sludge lagoon 2
	$g\ L^{-1b}$	
TS	208.3 ± 111.4	177.3 ± 60.9
VS	140.3 ± 102.4	105.8 ± 36.9
TSS	88.9 ± 25.0	91.9 ± 12.0
COD	105.9 ± 39.0	105.1 ± 20.4
Soluble COD	1.5 ± 0.7	1.7 ± 0.8
BOD_5	4.44 ± 2.98	5.36 ± 3.10
TKN	6.23 ± 2.43	6.41 ± 2.45
NH_4-N	0.51 ± 0.17	0.60 ± 0.15
$NO_2 + NO_3$-N	0 ± 0	0 ± 0
TP	6.59 ± 0.92	6.58 ± 1.10
Soluble P	0.26 ± 0.14	0.18 ± 0.04
K	1.65 ± 0.19	1.53 ± 0.15
Ca	3.50 ± 1.37	4.13 ± 1.18
Mg	3.18 ± 1.82	3.35 ± 1.71
Zn	0.37 ± 0.22	0.45 ± 0.14
Cu	0.40 ± 0.13	0.42 ± 0.12
Fe	0.72 ± 0.41	0.85 ± 0.22
S	1.34 ± 0.23	1.34 ± 0.20
Na	0.45 ± 0.05	0.42 ± 0.03
Alkalinity	26.2 ± 10.7	25.28 ± 7.05
pH	7.89 ± 0.27	7.80 ± 0.21

[a]*Data are means ± standard deviations for 5 sampling dates (months 0–24).*
[b]*Units in g L^{-1} except for pH.*

in lagoon 2 accumulated to a depth of 1.25 m (volume 4,440 m^3) in 12 years of continuous swine production (serving 2,940-head feeder-to-finish) (**Figure 9**). Therefore, the average rate of sludge accumulation in the two lagoons was 0.1170 ± 0.0127 m^3 sludge/feeder-to-finish head/year. It was consistent with the sludge generation standard for NC anaerobic swine lagoons of 0.1249 m^3 sludge/feeder-to-finish head/year (33 gal/animal of 135 lb/year) (AG-604, 2000).

After conversion, the sludge accumulation on both lagoons was halted (**Figure 9**). During the 6 years of new treatment, the sludge depth in lagoon 1 (that received all the plant effluent) did not increase; it was stabilized at a depth of about 0.55 ±0.07 m (volume = 2,100 ± 290 m^3). Similarly, lagoon 2 (discontinued lagoon) did not accumulate more sludge after discontinuation; the sludge depth remained about constant at 1.08 ±0.05 m (volume = 3,750±180 m^3).

Table 4 shows the composition of the sludges in the two lagoons determined five times at the beginning of the study (months 0–24). The sludges were of mineral nature, thick, black, with tar like smell, with similar chemical composition in the two lagoons (**Table 4**). A salient characteristic is the large amount of P contained in the lagoon sludges. Considering sludge volume and P concentration, there were 17.2 and 29.2 metric tons of P (39.5 and 66.9 metric tons P_2O_5) in lagoon 1 and 2, respectively. Therefore, new technologies that could harvest the P contained in lagoon sludges could have a great impact on global P cycling. One such technology is the Quick Wash process presented in this special issue (Szogi et al., 2018). It recovered 80% of the P from swine lagoon sludges.

CONCLUSION

More and more often, new treatment systems for manure combine three or four process units to meet various environmental standards and recovery targets. In North Carolina, USA, construction of new swine farms or expansion of existing swine farms are required new waste management systems that can replace anaerobic lagoon treatment for the waste and meet new environmental standards of ammonia and odor emissions, pathogens release, and the substantial elimination of soil and groundwater contamination by nutrients (phosphorus and nitrogen) and heavy metals. A treatment system that met these multiple standards was implemented at full-scale in a swine farm and operated for 6 years. It combined high-rate solid-liquid separation with N and P removal processes. The treatment plant removed from the manure: 97% of TSS, 90% of VS, 99% of BOD_5, 96% of TKN and NH_4-N, 93% TP, 99% of Zn and Cu, 99.9% odors and 99.99% pathogens. This study determined the water quality improvements in lagoons by an innovative swine manure treatment system operating at full-scale during five pig production cycles. After conversion, the sludge accumulation in the lagoons was halted. As plant effluent or rainwater replaced the liquid in the old lagoons, they became aerobic (Eh > 300 millivolts). In 2 years, the NH_4-N concentration in the lagoons liquid was reduced from the 370 to 485 to lower than 15 mg L^{-1}. While clean water is more valuable for both environmental quality and crop production, it is significant that the treatment process transformed the lagoon's water from a constituent-laden legacy condition to relatively cleaner water. Moreover, the transformation was accomplished while doubling the number of animals.

AUTHOR CONTRIBUTIONS

The author MV has designed and conducted the full-scale project, performed data summarization, and written the manuscript. KR measured ammonia emissions. AS assisted with water quality work. JL did odor quantification. PM did the pathogen assessment.

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ATTACHMENT 60

frontiers in Sustainable Food Systems

ORIGINAL RESEARCH
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High-Rate Solid-Liquid Separation Coupled With Nitrogen and Phosphorous Treatment of Swine Manure: Effect on Ammonia Emission

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A new treatment system was developed to meet multiple environmental performance standards including to substantially reduce ammonia emissions. It was tested full-scale for 2-years in a 5,145-head finishing swine farm with two anaerobic lagoons. The system combined high-rate solid-liquid separation with nitrogen and phosphorus removal processes. Both vertical radial plum mapping (VRPM) and floating static chamber techniques were used to measure NH_3 emission fluxes from anaerobic storage lagoons and the total farm-level NH_3 emission rates. The VRPM used an open-path tunable diode laser absorption spectroscopy (TDL) and the flux chamber used a photoacoustic gas analyzer to accurately measure NH_3 concentration. After the treatment system started, one of the two lagoons became inactive without receiving anymore flushed manure. The ammonia emission flux from the other lagoon with the treated effluent decreased from 43.9 to 6.8 kg-N ha^{-1} d^{-1} 1.5 years after implementation of the new treatment system. The NH_3 emission flux from the inactive lagoon also decreased similarly because the already stored old manure of the lagoon prior to inactivation was diluted with rainfalls and lost some NH_3 via volatilization. The total farm-level NH_3 emission rates decreased from 1.72 g s^{-1} to below detection level of the VRPM technique. Using the minimum detection level of the TDL with $R^2 > 90\%$ (i.e., 8.1 8.1 μL L^{-1}-m), the total farm-level NH_3 emission rates in the second year were less than 0.04–0.15 g s^{-1}. These results suggested that the impact of the new treatment system on NH_3 emission reduction was equivalent to closing conventional swine lagoons while actively growing 5,145 pigs with minimal ammonia emissions from the farm.

Keywords: total farm ammonia emission rate, swine lagoon, manure treatment, vertical radial plume mapping method, lagoon emission flux

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INTRODUCTION

Ammonia (NH_3) is an important fugitive gas mostly emitted from livestock operations in the United States (Doorn et al., 2002; Ro et al., 2017). Ammonia, a precursor of nitrate due to microbial nitrification/denitrification, causes acidification of both soil and surface water, and eutrophication in water bodies. It is also a principal source of atmospheric aerosols. Anaerobic lagoons are being widely utilized in the southeastern U.S. for storage and treatment of manure from confined swine production operations. The anaerobic treatment reduces the organic load of liquid manures but releases inorganic nitrogen as ammonia. Thus, ammonia emission from the lagoons is of environmental and health concerns in geographic areas with very intense confined swine production (Barker, 1996; Aneja et al., 2000; McCubbin et al., 2002; Szogi et al., 2006; Blunden and Aneja, 2008; Westerman et al., 2010). Addressing the environmental and health issues caused by ammonia emissions and potential contamination of water bodies with swine manure effluents Vanotti et al. (2010), developed an on-farm wastewater treatment system and demonstrated its high efficacy to meet multiple environmental performance standards (Vanotti et al., 2009) on a 5,145-head swine finishing farm. The multi-stage treatment system used solid separation, nitrification-denitrification, and phosphorous removal/disinfection processes. The new treatment system replaced the conventional anaerobic treatment lagoons of a swine farm and improved lagoon water quality dramatically. The treatment system removed 97% of total suspended solids (TSS), 99% of biochemical oxygen demand (BOD), 93% of total phosphorous (TP), 96% of total Kjeldahl nitrogen (TKN), and ammoniacal nitrogen (NH4-N) from the manure (Vanotti et al., 2018).

Although the new treatment system significantly improved the lagoon water quality (Vanotti et al., 2018) and reduced odors (Loughrin et al., 2009), its impact on ammonia (NH_3) emission had not been reported. The improved lagoon water quality would directly impact the NH_3 emissions from the lagoon and the animal houses using the cleaner plant-treated water for flushing manure (Szogi et al., 2006; Ro et al., 2008). Therefore, it is of great interest to assess the NH_3 emission reduction from the whole swine farm utilizing the new second-generation manure treatment system.

Micrometeorological techniques such as the integrated horizontal flux (IHF) and the backward Lagrangian stochastic (bLS) dispersion techniques can be used to measure the total ammonia emission from a swine farm. Although the bLS technique has shown its high accuracy in measuring gas emissions from point and distributed emission sources (Ro et al., 2009, 2011, 2013, 2014), one of the concerns for the use of the bLS technique in measuring emissions from a real farm with a variety of structures (buildings, stand pipes, etc.) is its underlying assumption of idealized wind flow over flat and homogenous terrain. The wind disturbance caused by farm building structures can be minimized by measuring downwind concentrations further downwind distance from the farm such as 10 times the height of the buildings (Flesch et al., 2005). However, measuring downwind concentration at longer distances from an emission source may not be feasible for certain conditions due to limited open space or dilution of emitted gas concentration below the detection limit of an analytical technique.

In contrast, the IHF technique, a mass-balance based method, is not very sensitive to changes in wind turbulence, which can used to measure an average gas emission rate of a large area (Wilson et al., 1983). The U.S. Environmental Protection Agency published the vertical radial plum mapping (VRPM) technique based on the IHF principles as Other Test Method 10 (OTM−10) in measuring fugitive gas emission rates from closed landfills (EPA, 2005). The VRPM technique estimates the horizontal flux of gas passing downwind of the emission source based on measured wind speed profiles and path integrated concentrations (PICs) (Ro et al., 2009, 2011; Viguria et al., 2015). The VRPM utilizes a bivariate Gaussian smooth basis function minimization (SBFM) to reconstruct a crosswind-smoothed mass-equivalent concentration map in a vertical plane from the downwind PIC data. Once all parameters for the bivariate Gaussian function are measured for a specific run, the VRPM procedure calculates the mass-equivalent concentration values for every square elementary unit (4×4 m) in a vertical plane. Then, the VRPM procedure computes and integrates the elementary unit flux over the entire vertical plane with corresponding wind speed data. Arcadis Inc. (Research Triangle Park, NC) developed the computer-based VRPM software which automatically calculates the emission rates based on the downwind PIC data, wind speed and direction information using the VRPM algorithm.

The objective of this study was to measure the ammonia emissions from the 5,145-head swine finishing farm with the new on-farm manure treatment system during a 2-year evaluation using the VRPM technique.

MATERIALS AND METHODS

Farm Description

The total NH_3 emissions from a finishing swine farm in North Carolina with a full-scale manure treatment system were measured from December 2006 to September 2008. The farm had seven swine houses with a permitted capacity of 5,145 head feeder to finish (735 head/barn). Two traditional anaerobic lagoons were used for about 11 years before the new manure treatment plant started operation in December 2006. With the new treatment system, the flushing of raw wastewater into the lagoons was discontinued. The treatment system consisted of three process units in series: polymer-enhanced solid-liquid separation, biological N removal, and alkaline phosphorus extraction and disinfection. More detailed description of the new manure treatment system and the farm operation can be found elsewhere (Vanotti et al., 2018).

Open-Path Tunable Diode Laser Absorption Spectroscopy

An open-path tunable diode laser absorption spectrometer (TDL; GasFinder2.0 for NH_3, Boreal Laser Inc., Edmonton, AB, Canada) was used to measure path-integrated concentrations (PICs) of NH_3. The TDL mounted on an automatic positioning device (APD; Model 20 Servo, Sagebrush Technology, Inc., Albuquerque, NM) emits a collimated beam (1,512 nm) aimed at distant three mirrors (retroreflectors) from which it is reflected to the TDL's detector. The TDL was designed for a sampling rate of about 1 Hz and had continuous internal calibration updates every 40 samples using an internal reference cell. The signal from the measurement path is compared with the signal from the internal reference cell and calculate the average gas concentration in the path as μL L^{-1}-m (i.e., μL L^{-1} multiplied by path length in m). The TDL also calculates the coefficient of determination (R^2) for each measurement to indicate the similarity between the waveform of the sample gas to that of the reference cell gas. A perfect match of the two waveforms would give R^2 of 1.0, and total mismatch would give R^2 of 0.0. Although the company recommends 5 μL L^{-1}-m as the minimum detection limit (MDL) for NH_3, we decided to determine the MDL for our TDL that would yield $R^2 > 90\%$. Determining the reliable value of MDL was important because the new manure treatment system dramatically improved the lagoon water quality and the NH_3 emission was so low that the path-integrated NH_3 concentrations measured with our TDL became at or below the MDL during the second year of the manure treatment operation in 2008.

The MDL of our TDL was determined by measuring PICs through a 4.5-m PVC pipe (5.1 cm diameter) in which 3 L min^{-1} of a mixture of pure N_2 and calibrated NH_3 gas (5.0 ppm, National Welders Supply Co., Inc., NC) flew through the PVC pipe. The two ends of the PVC pipe were sealed with transparent plastic film. Initially the PVC was filled with ultra-pure nitrogen gas (National Welders Supply Co., Inc.). Using a gas divider (SGD-710C, Horiba STEC, Sunnyvale, CA), gas mixtures of 10% NH_3 + 90% N_2 (0.5 μL L^{-1}-m NH_3), 30% NH_3 + 70% N_2 (1.5 μL L^{-1}-m NH_3), 50% NH_3 + 50% N_2 (2.5 μL L^{-1}-m NH_3), 70% NH_3 + 30% N_2 (3.5 μL L^{-1}-m), and 100% NH_3 (5.0 μL L^{-1}-m) were introduced into the PVC pipe. The measured PICs and the corresponding R^2-values were plotted as shown in **Figure 1**. We also plotted the average NH_3 PICs at different R^2-values (**Figure 2**). The average PIC was 5.3 μL L^{-1}-m at R^2 of 80% (**Figure 2**), which was similar to the NH_3 MDL (5.32 μL L^{-1}-m) reported by the U.S. Environmental Protection Agency's (EPA's) Environmental Technology Verification (ETV) statement (Myers et al., 2000). The MDL with R^2 of 90% was 8.1 μL L^{-1}-m. We used the MDL of 8.1 μL L^{-1}-m, when we observed the NH_3 PICs were below MDL during the second year of the new manure treatment system in 2008, to calculate the MDL for the VRPM NH_3 emission rate determination.

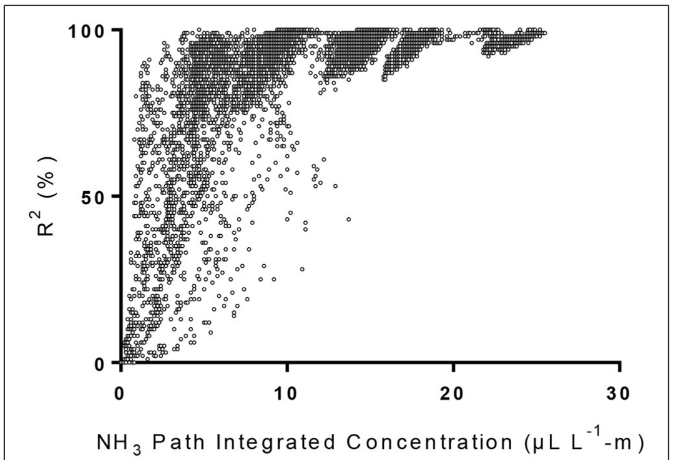

FIGURE 1 | Individual NH_3 path integrated concentrations (PICs) with associated R^2.

Total NH_3 Emission Rate Measurements Using the VRPM Technique

Total NH_3 emission rates from the swine farm (animal houses, lagoons, and the wastewater treatment system) were measured using the VRPM technique. The VRPM technique utilized the TDL mounted on the APD to measure downwind NH_3 PICs with 3 distant retroreflectors, one positioned at ground-level and the other two mounted on a weather station mast (10 or 20 m height). The upwind NH_3 concentrations were measured with a photoacoustic gas analyzer (INOVA, California Analytical, Orange, CA). The ranges of path lengths and the heights of the three retroreflectors are shown in **Table 1**. The path length between the TDL and the retroreflectors ranged from 129 to 266 m. The VRPM system was positioned downwind from the farm on a vertical plane, approximately perpendicular to the mean wind direction. The wind directions were mostly NNW or NNE. **Figure 3** shows the VRPM setup with the vertical plane approximately perpendicular to the NNE wind. The APD sequentially directed the infrared collimated beam of the TDL to each retroreflector. At each retroreflector, the TDL collected about 12–15 downwind PIC datasets before moving to the next position. Two cup anemometers (CS800-L Climatronics Wind Speed and Direction Sensor, Campbell Scientific, Logan, UT) mounted on the weather station mast at 2 and 10 m heights continuously measured wind speed and direction information during the emission monitoring campaigns.

The post-data filtering criteria recommended by the USEPA OTM−10 were used to remove error-prone data: concordance factor (CCF) > 0.8 (except when estimating the below minimum detection level emission rates with all PICs assuming to have the MDL of 8.1 μL L^{-1}-m) and the mean wind direction between -10^o and $+25^o$ from perpendicular to the vertical

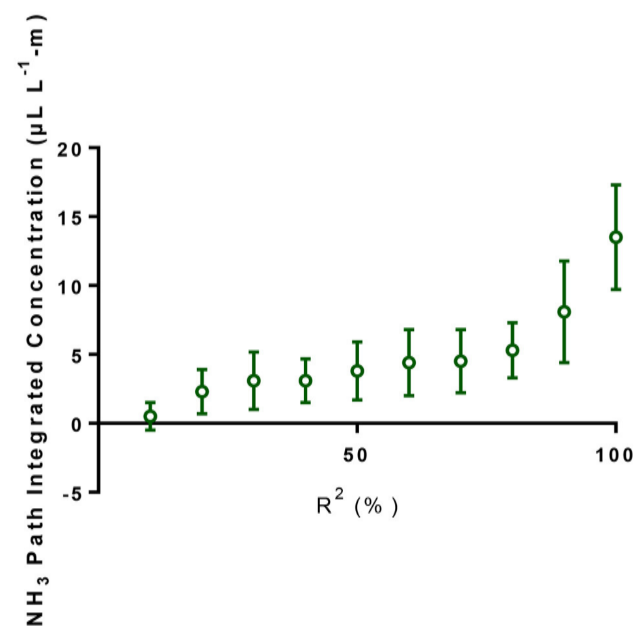

FIGURE 2 | Average NH_3 path integrated concentrations at desired R^2.

optical plane. After filtering data with the above criteria, the relative accuracies (calculated emission rate/actual emission rate) of the VRPM technique were 1.23 ± 0.17 (i.e., 23% over the actual emission rate) and 0.97 ± 0.44 (i.e., 3% less than the actual emission rate) for multi land emission sources and lagoon emission, respectively (Ro et al., 2011; Viguria et al., 2015).

Lagoon Emission

In addition to the VRPM technique for measuring whole farm NH_3 emission rates, the lagoon emissions were further verified using floating chamber techniques to compare the emission fluxes before and after the new treatment system reaching its steady state condition. The lagoon NH_3 emission fluxed were measured on June 18, 2007 while the treatment system was still under a startup period and on July 24, 2008 when the new treatment system was operating steady state.

Floating Static Chamber System

The static flux chamber was made of polyvinyl chloride (PVC) pipe (15.2 cm diameter and 29.8 cm length) one end sealed with an endcap. It was placed at the center of a 50.8 cm square foam sheet (thickness of 5.1 cm) to float on the lagoon water surface. The system was designed to submerge 11 cm of the PVC pipe under water to achieve a complete seal of the chamber headspace. The headspace gas was pulled from the chamber through Teflon tubing to the photoacoustic gas analyzer positioned at the bank of the lagoon and the analyzed gas was recirculated back to the chamber, thereby achieving well mixed condition inside the chamber. The floating flux chamber was launched into the lagoon at about 10 m away from the bank. The lagoon NH_3 emission was measured from 6 (3 points in each lagoon in 2007) to 10 (4 points in Lagoon 1 and 6 points in Lagoon 2 in 2008) different locations of the lagoons. After the treatment system started, Lagoon 2 became inactive without receiving anymore flushed manure from the pig houses. Instead, all manure flushed from the houses was treated first by the new treatment system and the treated effluent was stored in Lagoon 1 before use in crop irrigation. The increase in headspace NH_3 concentration was measured every 0.5–1 min with the photoacoustic gas analyzer for 10 min. The time-series headspace NH_3 concentration data were then used to calculate emission fluxes.

The NH_3 emission flux was calculated by fitting the time-series headspace NH_3 concentration data to a theoretical equation derived from performing a mass balance around the chamber. Assuming that the chamber is completely mixed and sealed, NH_3 mass balance around the chamber gives

$$\frac{dC}{dt} = k\,(C_S - C)\,d \tag{1}$$

TABLE 1 | Retroflector positions.

Retroreflectors	Height above ground (m)	Pathlength (m)
Retroreflector 1 (M1)	1.0	199.2–266.2
Retroreflector 2 (M2)	5.0 or10.0	128.6–228.5
Retroreflector 3 (M3)	10.0 or18.4	128.6–228.5

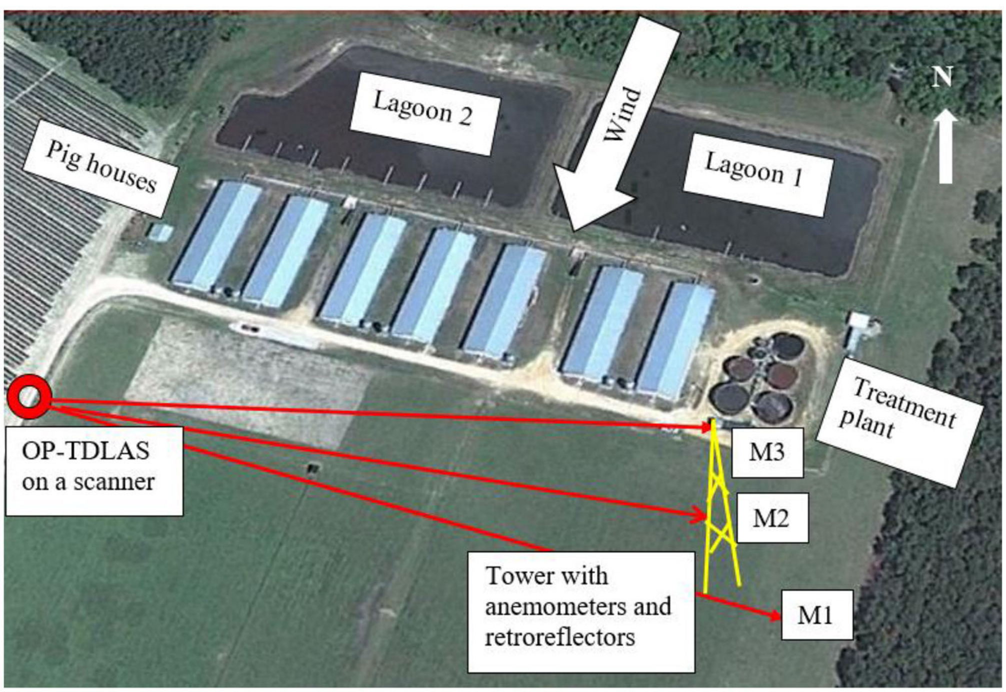

FIGURE 3 | VRPM beam geometry for NNE wind (M1, retroreflector 1 m above ground; M2, retroreflector at 5–10 m above ground; M3, retroreflector at 10–18.4 m above ground).

Where

C $=$ bulk headspace NH_3 concentration (mg m^{-3}),
C_S $=$ NH_3 concentration at the water surface (mg m^{-3}),
d $=$ mixing height of the flux chamber (m^2)
$= A_d/V$
A_d $=$ water surface area (m^2),
V $=$ headspace volume (m^3),
k $=$ mass transfer coefficient (m min^{-1}),
t $=$ time (min),

Integrating Equation (1) gives

$$C = C_S - (C_S - C_O)\exp\left(\frac{-kd}{t}\right) \quad (2)$$

Where CO $=$ initial headspace concentration (mg m^{-3}).

Instead of assuming initial concentration of ambient concentration, all three parameters (i.e., C_S, C_O, and k) were estimated simultaneously via non-linear regression analysis of the time-series headspace NH_3 concentration data using GraphPad Prism 7.03 (GraphPad Software, Inc., La Jolla, CA). This simultaneous regression of multi-parameters were recommended by the American Society of Civil Engineers (ASCE) in estimating oxygen transfer efficiency with the similar equation (ASCE, 1992). These three parameters estimated from fitting the time-series NH_3 concentration data to Equation (2). were then used to calculate the instantaneous flux (J) as time approached zero:

$$\left.\frac{dC}{dt}\right|_{t\to 0}\left(\frac{V}{A_d}\right) = J = k\,(C_S - C_O) \quad (3)$$

Statistical Analysis

The central tendency and precision of measurements were presented with arithmetic averages and standard deviations (given as $\pm$ values). All statistical parameters, analyses of variance (ANOVA) tests, and multi-regression analyses were obtained/performed using GraphPad Prism 7.03.

RESULTS AND DISCUSSION

Ammonia Emission Flux From Lagoons Using Floating Static Flux Chamber Technique

The increases in static flux chamber headspace NH_3 concentration with time at 6 (6/18/2007) and 10 (7/24/2008) different locations of the two lagoons and average NH_3 fluxes from each lagoon are shown in **Figure 4** and **Table 2**. Ammonia fluxes from the two lagoons ranged from 14.4 to 78.8 kg-N ha^{-1} d^{-1} with an average flux of 32.8 $\pm$ 24.2 kg-N ha^{-1} d^{-1} in 2007 (**Table 2**). These NH_3 emission fluxes were comparable to that from conventional swine lagoons without any manure treatment during summer months as reported in the literature (Arogo et al., 2003). It also indicated that the new treatment system had not yet been effective in reducing NH_3 emission from the lagoon 1 receiving treated effluent. Therefore, we decided to discontinue the VRPM monitoring of whole farm NH_3 emission rates until

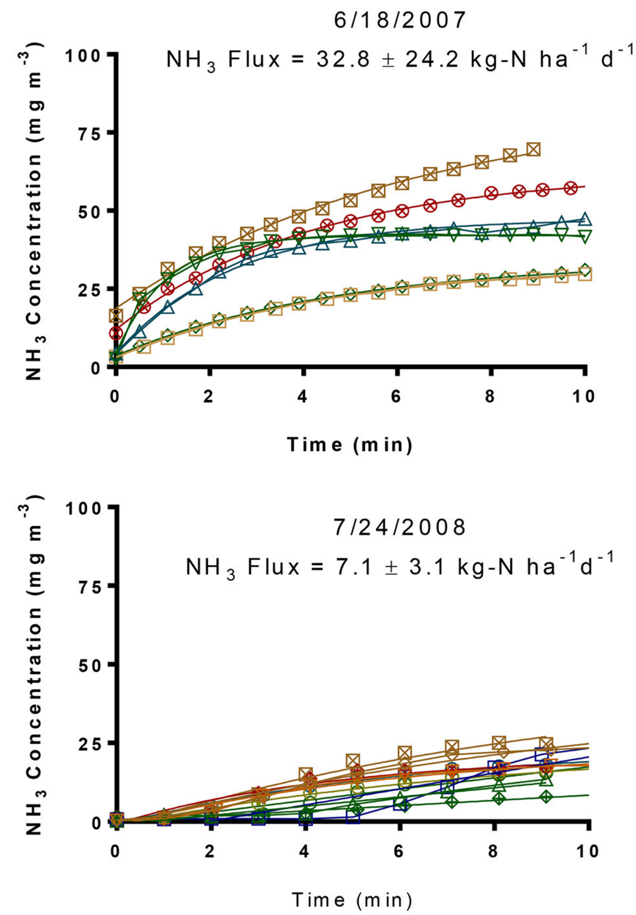

FIGURE 4 | Floating flux chamber headspace NH_3 concentrations from two treatment lagoons in summer 2007 and 2008.

TABLE 2 | NH_3 emission fluxes from the two lagoons.

	Emission Flux in June 2007 (kg-N ha^{-1} d^{-1})	Emission Flux in July 2008 (kg-N ha^{-1} d^{-1})
Lagoon 1	43.9 $\pm$ 32.4[a†]	6.8 $\pm$ 2.8[b]
Lagoon 2	21.7 $\pm$ 6.5[a]	7.3 $\pm$ 3.6[b]
Average for both lagoons	32.8 $\pm$ 24.2	7.1 $\pm$ 3.1

†*Emission flux values followed by the same superscript letter (a or b) were not significantly different at $P < 0.05$ (ANOVA).*

the new treatment system reached steady state and impacted NH_3 emission from the farm.

Figure 4 (bottom) shows that the flux-chamber time-series headspace ammonia concentrations increased to less than 25 mg L^{-1} after the chamber was deployed. In contrast, the NH_3 concentrations increased to 75 mg L^{-1} 10 min after deployment, indicating much higher emission fluxes in 2007 than in 2008. The NH_3 emission fluxes from the two lagoons ranged from 1.9 to 11.0 kg-N ha^{-1} d^{-1} with an average flux of 7.1 kg-N $\pm$ 3.1 kg-N ha^{-1} d^{-1}. The average NH_3 emission fluxes were 6.8 $\pm$ 2.8 kg-N ha^{-1} d^{-1} from Lagoon 1 and 7.3 $\pm$ 3.6 kg-N ha^{-1} d^{-1} from the inactive Lagoon 2. The significant decrease in NH_3 emission fluxes resulted from improving lagoon water quality by the new treatment system for Lagoon 1. The NH_3 emission flux from Lagoon 2 also decreased because the already stored old manure of the Lagoon 2 prior to inactivation was diluted with rainfalls and lost some NH_3 via volatilization. The decrease in NH_3 emission flux was further validated by the fact that the total ammoniacal N (TAN) of the lagoon liquid in June 2007 was 197 mg L^{-1} [or 20.0 mg L^{-1} $NH_{3\ (aq)}$-N] while that in July 2008 was only 32.7 mg L^{-1} {32.5 and 33.0 mg L^{-1} TAN [or 5.1 and 2.8 mg L^{-1} $NH_{3\ (aq)}$-N] for Lagoons 1 and 2, respectively} (Vanotti et al., 2018). These results suggested that the impact of the new treatment system on NH_3 emission reduction was equivalent to closing conventional swine lagoons while actively growing 5,145 pigs.

Whole Farm Ammonia Emission Reduction With the VRPM Technique

The NH_3 emission rates, measured with the VRPM technique, were the whole-farm emission rate that included three main NH_3 sources: (1) the NH_3 emissions from seven animal houses, (2) the new wastewater treatment system, and (3) the two lagoons (**Figure 3**). These NH_3 emissions rates were measured from 12/7/2006 to 9/17/2008 (**Table 3** and **Figure 5**). The start date (12/7/2006) was a day before the start date of the new wastewater treatment system. The NH_3 emission rate on 12/7/2006 was 0.80 g/s. However, 3 months after starting the new treatment system, the NH_3 emission rate increased to 2.64 g/s, because the air temperature along with the wind speed were much higher. In addition, it was too short period of time to observe the impact of the new treatment system on lagoon water quality (the lagoon renovation capacity with treated effluent was 1.2 lagoon volumes per year or hydraulic residence time, HRT = 0.8 years). After observing the NH_3 emission flux from the lagoons was comparable to that from conventional swine lagoons without any treatment (**Figure 4** top), NH_3 emission rate was not measured for the rest of 2007 to give more time for the new manure treatment system fully impacting the total farm emission. In 2008, NH_3 emission rates were monitored in relatively cold (4/4/2008), mild (9/16–9/17/2008), and hot days (7/24/2008 and 8/20/2008). Regardless the the wide variation of air temperatures and wind speeds, in the second year, downwind NH_3 PICs were all below the MDL (i.e., 8.1 μL L^{-1}-m) of TDL. Below-detection-limit emission rates of these days were estimated by assuming all PICs had the MDL concentration of 8.1 μL L^{-1}-m (**Table 3**). These below-detection emission rate ranges from 0.04 to 0.15 g/s, more than an order of magnitude lower the initial emission rates. These data clearly demonstrated that the new treatment system substantially reduced the NH_3 emission from the swine farm.

The NH_3 emissions rates obtained prior and during initial operation of the new wastewater treatment in this full-scale study were compared to emissions rates from other full-scale studies with conventional and converted lagoons in North Carolina. Initially with lower air and lagoon water temperatures (December 2006), the estimated live weight (LW) based emission rate of 0.80 g s^{-1} [1.19 kg N wk^{-1} (1,000 kg $LW)^{-1}$] was similar to the winter combined house-lagoon NH_3 emission rate of 1.06 kg N wk^{-1} (1,000 kg $LW)^{-1}$ from a conventional swine farm with similar number of pigs for a 5,784-head finisher farm using anaerobic lagoon technology (Aneja et al., 2008a). The emission rate obtained immediately after beginning the new treatment system in February 2007 was 2.64 g s^{-1} [3.92 kg N wk^{-1} (1,000 kg $LW)^{-1}$], due to very high wind speed at that day (8.33 m/s compared to 2.67 m/s in 12/7/2006) while the water temperatures were similar. The emission rate was almost linearly increased

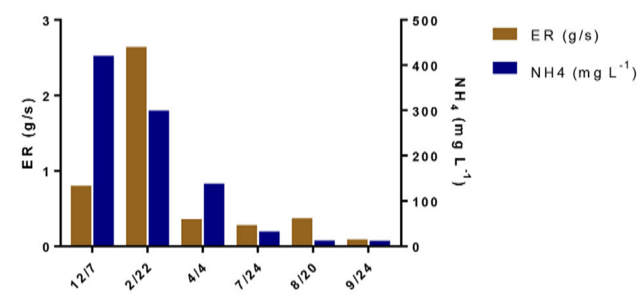

FIGURE 5 | Ammonia emission rates and corresponding total ammoniacal concentrations of the lagoons.

TABLE 3 | Summary of temperatures, wind speed, and NH_3emission rates from the swine farm.

Sampling period†	**Mean air temperature**‡ **(°C)**	**Wind speed (m s^{-1})**	**Average NH_3 concentration**§ **(μL L^{-1})**	**NH_3 emission rate (g s^{-1})**#
12/7/2006 12:15–14:55	15.3 ± 0.6	2.67 ± 0.92	0.29 ± 0.05	0.80 ± 0.16
2/22/2007 15:45–16:30	22.4 ± 0.05	8.33 ± 2.20	0.27 ± 0.03	2.64 ± 0.32
4/4/2008 10:15–12:30	16.9 ± 2.7	3.46 ± 1.03	<MDL¶	< 0.15
7/24/2008 11:50–18:20	29.3 ± 1.1	1.59 ± 0.75	<MDL	< 0.12
8/20/2008 11:10–15:30	30.9 ± 1.0	2.38 ± 1.06	<MDL	< 0.13
9/16/2008 14:10–16:15	22.5 ± 0.6	3.52 ± 1.23	<MDL	< 0.05
9/17/2008 9:30–10:45	19.2 ± 0.6	3.01 ± 1.12	<MDL	< 0.04

† *New wastewater treatment system started operation December 9, 2006.*
Mean air temperatures measured at 2 m above ground on site during measurement period.
§NH_3 concentration.
¶ *<MDL, below minimum detection limit (<8.1 μL L^{-1}-m or about 0.08 μL L^{-1} for a 100-m path length).*
#*Calculated assuming 8.1 μL L^{-1}-m for all PICs.*

with wind speed. Note that during this initial startup period, the effects of the new treatment system on lagoon water quality were minimal.

After 1.5 years of operation the total emissions (combining animal houses, treatment plant, and lagoons) of the treatment plant were <0.15 g s^{-1} [0.22 kg N wk^{-1} (1,000 kg LW)$^{-1}$] for April 2008 (**Table 3**). Astonishingly, even for hot days in July and August 2008, the downwind NH_3 concentrations were below its detection limit. Comparing to swine lagoon NH_3 emission rates with similar water temperature of 30.2°C [2.4 kg N wk^{-1} (1,000 kg LW)$^{-1}$, (Szogi et al., 2006)], this reduction in total NH_3 emission was indeed remarkable.

Compared to the average of the first two emissions for the periods of December 2006 and February 2007 (1.72 g s^{-1}), total NH_3 emissions from the April 2008 to September 2008 periods were reduced by 94%. In comparison, the new treatment plant removed an average of 96.5% of the soluble NH_4-N contained in the raw flushed manure with most of the removal (84%) occurring at the biological N treatment step (Vanotti et al., 2018). Concentrations of NH_4-N in the lagoons were also reduced significantly during the emission evaluation period from 371 to 485 mg L^{-1} in 2006 to lower than 65 mg L^{-1} in 2008 (Vanotti et al., 2018).

Low NH_3 emission rates similar to those recorded in 2008 were also found in earlier studies (Aneja et al., 2008b) of a first generation version of the same wastewater treatment system (Vanotti et al., 2007) that was retrofitted in a 4,360-finishers production unit and removed 98.7% of the soluble NH_4 from liquid waste. According to Aneja et al. (2008b), the NH_3 emissions for this first-generation wastewater treatment plant were 0.02 kg N wk^{-1} (1,000 kg LW)$^{-1}$ in the warm season and 0.0004 kg N wk^{-1} (1,000 kg LW)$^{-1}$ in the cold season. These low NH_3 emissions represented reductions of 94.4% for the warm season and 99.0% for the cool season with respected to a conventional lagoon system (Aneja et al., 2008a). On a separate study of the same first-generation wastewater treatment system, Szogi et al. (2006) found that total annual NH_3 emission from the converted lagoon was 1,210 kg N y^{-1} and equivalent to 0.10 kg N wk^{-1} (1,000 kg LW)$^{-1}$. In contrast the NH_3 emissions from the conventional lagoon totaled 12,540 kg N y^{-1} [1.22 kg N wk^{-1} (1,000 kg LW)$^{-1}$]. Compared with the conventional lagoon, annual NH_3 emissions from the converted lagoon were reduced 90% (Szogi et al., 2006).

The NH_3 concentration in the air inside the houses was also reduced as a result of recycling cleaner water for pit recharge under the animal houses (Szogi and Vanotti, 2008). Compared with the previous conventional lagoon management, the new treatment system lowered NH_3 concentrations in the house exhaust air by an average of 75.1%. Even though the wastewater treatment had instant effect on reducing NH_3 in the manure, and the NH_3 in the recycled water used for flushing the barns, the beneficial effects on emissions at the farm level were not seen immediately; the dirty liquid in the former lagoons needed time to be cleaned up. Actual lagoon 1 volume was 9,565 ± 750 m^3 and received 11,552 m^3 per year of the clean plant effluent (Vanotti et al., 2018). In this study, one year of continuous operation of the wastewater treatment plant was necessary to observe marked differences in NH_3 emissions at the farm level.

SUMMARY

The NH_3 emissions from a 5,145-head finishing swine farm before and after the new manure treatment system implementation were measured using both micrometeorological VRPM and static chamber techniques. The NH_3 emission flux from the lagoons decreased from 32.8 to 7.1 kg-N ha^{-1} d^{-1}. The reduction in the total farm NH_3 emission was even more dramatic: All downwind NH_3 concentrations 1.5 year after starting up of the new treatment system were all below MDL. The dramatic decrease in total ammoniacal nitrogen in the lagoon validated this exceedingly low NH_3 emission from the farm.

AUTHOR CONTRIBUTIONS

KR designed and conducted the emission experiments, collected and post-processed data, and wrote the draft. MV, AS, JL, and PM conducted water quality work, data summarization and processing. All authors participated in writing of the manuscript.

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Swine manure biogas production improvement using pre-treatment strategies: Lab-scale studies and full-scale application

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ABSTRACT

The paper deals a case study of solid-liquid separation (SLS) approaches for swine manure biogas recovery in a system configured to treat solid fraction on Continuous Stirred Tank Reactor (CSTR) and liquid fraction on Covered lagoon biodigester (CLB) in a large scale. At the same time, scale down reactors on laboratory scale were operated under same conditions. Biogas productivity of full-scale CSTR showed an average of 0.65 ± 0.23 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$, while CLB was around 0.18 ± 0.05 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$. The results of lab-scale can predict digestion capability and methane recovery for full-scale system. Economy in biogas plant construction and operation, biogas generation constantly and digestate treatment system implementation are the mainly benefits to use SLS followed different reactor configurations. Biological desulfurization process was dependent of biogas retention time. By the way, operational improvements made on large scale have allowed an advance in the productivity and biogas quality.

1. Introduction

Swine farming has changed from small family farms to large concentrated animal feeding operations (CAFO's) to increase swine production at lower cost and sanitary benefits (Moses and Tomaselli, 2017; Wang et al., 2019). It is an important activity for Brazil's economy, being result of work in development, modernization, and intensification this sector. Brazil appears as 4th largest in global majors' producer of swine meat, with 4.1 million tons annually (USDA, 2021).

Considering the large production and consequently huge amounts of swine manure, anaerobic digestion (AD) is considered an environmentally friendly technology that combines biogas production and sustainable waste management, being a key process in any swine manure treatment system (Tápparo et al., 2020). Besides, due to large volume of swine manure produced in a restricted area, traditional methods of disposal, like land application, posed pressure on the environment. This situation represents a risk to the expansion of swine farming and consequently as an economic activity (Kunz et al., 2009b).

AD has intensified in Brazil in view of the low cost, easy operation, high efficiency in reducing odors by using covered lagoons biodigesters (CLB) and energy use of biogas. However, these biodigesters have limitations, such as work better for liquid manure with less than 3% total solids, low organic loading rate (approximately 0.5 $kg_{VS}{\cdot}m^{-3}d^{-1}$) (Wu, 2013), biogas production varies seasonally, because reactors are not heated and depend of ambient temperatures (Khanal et al., 2019). Furthermore, microorganisms and liquid have limited contact since there is no agitation, biogas yield (0.36 $m^3{\cdot}kg_{VS}^{-1}$) is lower than others technologies and periodic cleaning is necessary due to fixed solid accumulation (Cantrell et al., 2008; Yu and Schanbacher, 2010).

In contrast, others different reactor models were developed, and its use is emerging. One example is the continuous stirred tank reactor (CSTR) that is widely used to substrates with high solids concentration, especially for the treatment of high-strength liquid animal manure (Mao et al., 2015). Microorganisms are suspended due to complete mixing in CSTR and this condition offers good substrate-sludge contact, increasing mass transfer resistance and consequently AD occurs swiftly (Tauseef et al., 2013; Wei et al., 2019). Combined with temperature control (mesophilic and thermophilic), this design provides higher biogas

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productivity (>1.5 m^3_{biogas} $m^3_{reactor}$ d^{-1}), that in comparison to CLB the increase is over 10-fold (Cantrell et al., 2008). The feed should contain 3 to 10% total solids and short retention time (< 20 days) for swine manure. Nevertheless, this reactor design own relatively high capital, energy consumes, and need mechanical parts maintenance (Kress et al., 2018) and these drawbacks decrease the use in development countries.

On the other hand, because the low concentration of volatile solids of swine manure (between 1 and 5%) and high volume the maintenance of temperature a CSTR is difficult and consequently the energy recovery would be affected (Yang et al., 2015). Increasing solid concentration of substrate, through co-digestion (Tápparo et al., 2018) or using preliminary SLS are options to swine manure large-scale plants economic feasibility using of CSTR (Vu et al., 2016).

Previous studies evidence that using SLS processes, such mechanical separations, screens, gravity settling are ways for improve biogas generation. Additionally, the fractions of solid-liquid separation have different methane yields and afford more effective options for reactor design and operational conditions for swine wastewater treatment (Amaral et al., 2016; Chen et al., 2015; Yang et al., 2016).

Considering this background, the goal of this study is evaluating the process performance and biogas production of one combined system where after a solid-liquid separation unit, the solid fractions of swine manure are forwarded to CSTR and the liquid fraction directed to CLB comparing in lab and full-scale.

2. Material and methods

The research was conducted at a full-scale swine manure treatment system, localized in Videira, Santa Catarina – Brazil (27°02′S 51°05′W) and in laboratory of Embrapa Suínos e Aves, Concórdia, Santa Catarina – Brazil (27°18′S 51°59′W). The swine farm is a farrow to wean unit, with 5320 breeder sows. Swine manure treatment system consist in one solid-liquid separation unit (SLS), include a sieve with rotating brushes (2 mm sieves) and a settling tank (40 m^3), followed by one biodigester module where the liquid fraction is forwarded to a CLB and the solid fraction to a CSTR reactor. The configuration of the manure treatment system follows the Brazilian Patent of SISTRATES (PI1100464-9 A2) (Kunz et al., 2015). Data collection were in the years of 2018 and 2020. A flowchart of treatment system is shown on Fig. 1.

2.1. Biochemical biogas potential (BBP)

BBP was measured on samples of solid retained in sieve (SRS), settling tank sludge (StS), CLB sludge (CLBS), and supernatant (SN). Sampling was carried out on different seasons, respectively, summer (1), autumn (2) winter (3) and spring (4). Experiments were conducted in triplicate using 250 mL batch reactors bound to 500 mL eudiometer tubes. The system was maintained at a mesophilic temperature (37 °C). The biogas yield was evaluated until it became stable, which was indicated when the daily biogas production was less than or equal to 1% of the total biogas that had been produced, according to (VDI 4630, 2016). Inoculum containing mesophilic anaerobic microorganisms was prepared and used according to (Steinmetz et al., 2016). All samples were analyzed for total nitrogen (TKN), total carbon (TC), K, TP, total solids (TS), and volatile solids (VS).

2.2. Lab-scale reactors

The CSTR used was made of acrylic, had a working volume of 21 L and was jacketed, allowing water circulation for temperature control. Temperature was controlled by means of a thermostatic bath (JULABO, Model M8) kept in mesophilic range (37 °C ± 1). The daily feed was composed by a mixture of approximately 7% of CLBS, 10% of SRS and 83% of StS. The composition of CSTR feed was response of SLS module and necessity of CLB sludge discard.

CLB was made of acrylic, had a working volume of 17 L, without temperature control. Feed composition of this reactor was the supernatant after the SLS module as described in Fig. 1. Both reactors were manually fed intermittently once a day and biogas production was measured using Milligascounter (model MGC-1 V3.3 PMMA, Ritter, Germany). Methane concentration was evaluated using BIOGAS 5000 (Geotech, UK). The strategy for CLB start-up used 20% of working

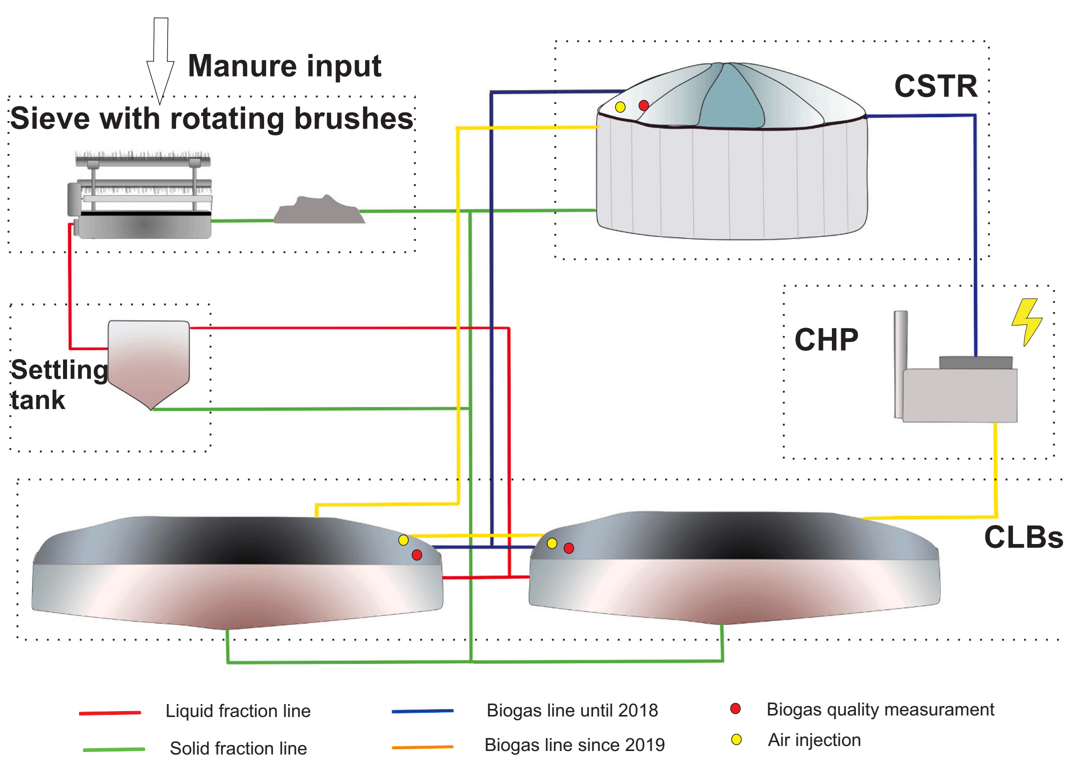

Fig. 1. Schematic of solid-liquid separation unit and biodigesters fed.

volume (3.4 L) of sludge and 80% (13.6 L) of effluent from a CLB reactor fed with swine manure from a commercial farm. The CSTR was started with 12 L of effluent (digestate) from a full-scale CSTR fed with solid fraction from a solid-liquid process unit installed in a swine manure treatment process from a commercial farm.

2.3. Full-scale reactors

The full-scale CSTR has a working volume of 700 m^3 and is operated under mesophilic conditions and built in concrete. The heating system was assembled to use the heat losses of a 300 kVA CHP unit used for electrical energy generation. CSTR mixing was performed by a side shaft, mixing took place for 5 min on every 30 min. The two CLB were built using a PEAD geomembrane and has a volume of 2500 m^3 each one, its flow is divided equally for these digesters. The CLBs are operated without heating system.

Thermal mass flow meters, model FT2 (Contech®, Brazil) were used for biogas quantification of both biodigesters. Biogas composition (CH_4, CO_2, O_2 and H_2S) was measured every 2 h using AwiFLEX (Awite®, Germany). The biogas analyzer includes the biological desulfurization module "AwiDESULF" (Awite, Germany), which automatically doses ambient air based on the measurements of H_2S and O_2 in the biogas. The system uses a combination of artificial intelligence (Fuzzy Logic) and conventional PI-control to regulate oxygen concentration in the biogas between 0.2 and 1%. As the digester headspace is used to achieve biological desulfurization by micro-aeration, no additional constructions are needed to use the system. For both reactors, feed conditions were the same as described for lab-scale reactors.

2.4. Analytical methods

TC was determined by CNHS elemental analyzer (model Flash 2000, Thermo Fisher Scientific, (Massachusetts, USA) following the manufacturer's instructions. Total phosphorus was determined using spectrophotometric molybdovanadate method on a spectrophotometer Cary 50 (Agilent, Santa Clara, USA). TKN was measured by Kjeldahl method on a digester and distillation unit model Kjeltec 8100 (Foss, Hilleroed, Denmark). The correlation between volatile fatty acids (VFA) and alkalinity (TA) was determined according Liebetrau and Pfeiffer (2016) by titration with sulfuric acid 0.05 mol L^{-1} in an automatic titrator, (model 848, Titrino plus, Metrohm, Switzerland), previously, samples were centrifuged (5488*g* for 7 min). TS, VS, pH and K were determined according Standard Methods (APHA, 2012).

3. Results and discussion

3.1. Samples characteristics and BBP assays

The characteristics of SRS, StS, CLBS and SN feedstock showed variation in all parameters, and in particular in BBP and CH_4 concentration in all collections due to temporal and seasonal changes (Table 1). Amaral et al. (2016) demonstrated that the methane potential of the StS fraction was approximately two times higher than the methane potential of the SN fraction. In the present work it was observed that 1 ton of StS fraction could produce about 16.2 ± 5.2 Nm^3 of biogas, while for SN fraction this production was reduced to 4.3 ± 1.4 Nm^3 (Table 1) and while for CLB the production was 3.0 ± 1.5 Nm^3. The SRS present the highest biogas production per fresh matter, around 96 ± 17 Nm^3. Despite the SN fraction showing a low methane production (fresh matter) and VS concentration, comparing with StS, it presents higher bioavailable carbon (Amaral et al., 2016). The swine manure characteristics and consequently biogas generation is linked with the inherent fluctuations of production process. The water consumption by swine's, animal number in the installations, swine growing phase, nutritional aspects and manure storage time directly influence concentration and biodegradability of manure (Amaral et al., 2016; Kunz et al., 2009a).

The differences in biodegradability and biogas potential of swine manure fractions enables the use of different reactors to recover biogas of each fraction. Due SRS and StS are produced in smaller volumes than SN it is easier to increase temperature and assure operation in mesophilic range. The settling process is an interesting strategy to improve biogas production, because using concentrated manure as a feedstock a smaller size digester is required to produce the same volume of biogas than it would need when using the raw swine slurry. Thus, the costs with reactor construction and heating system are lower (Deng et al., 2014). Moreover, sedimentation is an attractive option because of lower cost and simpler technology compared to others SLS, like centrifugation and pressurized filtration (Hjorth et al., 2011; Hollas et al., 2019).

The substrates that feed the CSTR reactor in this work concentrate the most part of nutrients (e.g. N, P, K) (Table 1), which consequently are preserved in digestate, producing high-quality organic fertilizer compared with digestate of swine manure. Therefore, CSTR digestate has a high agronomic value compare to the swine manure digestate.

3.2. Lab-scale reactors performance

The performance parameters of CSTR and CLB are presented in

Table 1
Swine manure samples characteristics. Volatile solids, biogas yield, N, P K and C each fraction.

Samples		BBP (NL_{biogas} kg_{VSadd}^{-1})			CH_4 (%)	TS (g kg^{-1})	VS (g kg^{-1})	TKN (g kg^{-1})	K (g kg^{-1})	TP (g kg^{-1})	TC (g kg^{-1})
Solid retained in sieve (SRS)	1	461	±	41	65	220.83	189.43	2.75	1.19	9.99	102.0
	2	449	±	45	59	250.55	176.42	2.60	0.90	6.40	107.5
	3	485	±	19	53	248.47	205.83	2.67	1.17	10.34	107.3
	4	645	±	9	40	210.90	183.20	1.81	1.07	4.84	111.6
	Average	510	±	91	54 ± 10	233 ± 20	189 ± 13	2.5 ± 0.4	1.1 ± 0.2	7.9 ± 2.7	107 ± 0.4
Settling tank sludge (StS)	1	314	±	36	66	44.30	31.44	4.45	1.10	1.60	16.8
	2	274	±	19	71	67.42	53.54	2.80	0.91	1.43	29.3
	3	296	±	12	58	75.23	62.82	3.48	1.16	2.08	32.0
	4	424	±	43	54	67.90	51.40	3.99	1.07	0.78	11.1
	Average	327	±	66	62 ± 02	64 ± 13	50 ± 13	3.7 ± 0.7	1.1 ± 0.1	1.5 ± 0.5	22 ± 1.0
Covered lagoon biodigester sludge (CLBS)	1	121	±	14	68	13.08	7.52	1.88	0.51	0.51	4.2
	2	129	±	11	73	30.92	23.20	3.91	0.69	1.08	11.3
	3	218	±	57	50	30.97	18.73	4.09	0.68	1.32	10.1
	4	178	±	48	44	38.46	23.62	4.64	0.80	1.58	13.0
	Average	162	±	45	59 ± 14	28 ± 11	18 ± 08	3.6 ± 1.2	0.7 ± 0.1	1.2 ± 0.5	9.7 ± 0.4
Supernatant (SN)	1	509	±	30	58	7.09	4.90	1.3	0.8	0.15	1.99
	2	666	±	33	67	8.61	5.80	0.9	0.4	0.42	2.86
	3	715	±	13	44	10.63	7.25	1.4	0.7	0.21	3.01
	4	492	±	3	60	15.52	11.33	1.5	0.7	0.31	5.08
	Average	595	±	111	57 ± 10	10 ± 4	7 ± 3	1.3 ± 0.3	0.7 ± 0.1	0.3 ± 0.1	3.3 ± 1.3

Fig. 2A and B, respectively. Table 2 summarizes digestate analyses for lab-scale and full-scale digestion processes.

CSTR biogas productivity showed 0.61 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$, and biogas yield was around 0.46 $NL_{biogas}\ g^{-1}_{vsadd}$, while the BBP (based on Table 1) is approximately 0.32 $NL_{biogas}\ g^{-1}_{vsadd}$. The methane concentration ranged from 59 to 68%. In contrast CLB presented 0.2 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$ and 0.9 $NL_{biogas}\ g^{-1}_{vsadd}$ (BBP was 0.7 $NL_{biogas}\ g^{-1}_{vsadd}$) in stationary period with 70% of methane concentration. Due to SLS process, the manure forwarded to CLB contains only between 0.5 and 1% of VS (Table 1). This low concentration of solids and high bioavailability of carbon present in this fraction explains the higher biogas yield of CLB compared to CSTR, that is fed with food residues, nondigested fed and pig hair residues, that present low biodegradability (Amaral et al., 2016; Zhang et al., 2014).

The BBP predictions (Table 1) with lab results for CSTR and CLB biogas yield showed that experimental values were 43% and 28% higher than predicted values, respectively. Notwithstanding, for CSTR, the mixture of this substrates probably had shown a synergic effect, when compared with digestion of individual samples. As shown in Table 2, the pH for both reactors remained between 7.26 and 7.86 and VFA/TA ratio did not rise above the critical value of 0.4, indicating good process stability. Biogas yield and productivity showed an initial transitory state, attributed to the star-up to the process establishment with a balance between hydrolytic, fermentative bacteria and methanogenic archaea (Bouallagui et al., 2010).

3.3. Full-scale reactors performance

CSTR feeding consist in a mixture of SRS, StS and CLBS, as aforementioned. The flow rate was 20 ± 5 $m^3\ d^{-1}$ with a VS concentration of 60.6 ± 10.8 kg m^{-3}. Until operational day 152, the OLR was 1.69 ± 0.34 $kg_{\ VSadd}\ m^{-3}\ d^{-1}$, biogas productivity showed 0.65 ± 0.23 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$, and biogas yield was 0.38 ± 0.14 $NL_{biogas}\ g^{-1}_{vsadd}$. Between 121 and 155 operation days a maintenance in agitation system was performed and the reactor was opened. For this maintenance it was necessary to reduce about 33% of reactor volume and this occurred during winter time, causing the decrease of temperature in the biodigester up to 10 °C. During this time biogas production was not registered, after the maintenance conclusion the process was restarted. Biogas productivity decreased to 0.48 ± 0.18 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$ and biogas yield to 0.29 ± 0.10 in an OLR of 1.68 ± 0.24 (Fig. 2 C). This difference was attributed to the decrease in biodigester temperature. Already, CLB was feeding with the SN, in a flow rate of 76 ± 29 $m^3\ d^{-1}$ (each biodigester), with VS concentration of 11 ± 2.5 kg m^{-3}. Biogas productivity was 0.18 ± 0.05 $NL_{biogas}\ L^{-1}{}_{reactor}\ d^{-1}$ and 0.56 ± 0.27 $NL_{biogas}\ g^{-1}_{vsadd}$ with OLR 0.35 ± 0.16 $kg_{VSadd}\ m^{-3}\ d^{-1}$ (Fig. 2 D). A decrease in biogas yield between the days 165–184, was due to the temperature decline of 5 °C. A decrease in the biodigester internal temperature had impact on degradation efficiency and consequently in biogas production, a decline of 7 °C could reduce up to 30% the biogas production (Schmidt et al., 2019). One of the main CLB characteristics is the absence of heat systems, consequently a variation on biogas production is expected. This reactor configuration is indicated to geographical regions with moderate or elevated year-round temperatures. However, due to the economy in construction and operational simplicity, its use justifies the lowest AD efficiency compared with others configuration (Yu and Schanbacher, 2010).

3.4. Lab-scale versus full-scale results

The comparison of results for continuous reactors (CSTR and CLB) showed that the lab-scale predict full-scale biogas yield and biogas productivity. The variability of biogas yield and biogas productivity are attributed to the lack in heating and HRT fluctuations in the AD system, that can be attributed to manure composition and volume. This is caused by limitations of large-scale process control, leading to fluctuations in operating parameters (e.g. variations in HRT, biodigester external temperature influence and variations in substrate composition). In the lab-scale reactors a higher process control was provided comparing to full-scale, thus allowing to predict the operating parameters of the full-scale reactor.

Moreover, must be highlighted that the HRT and solids retention

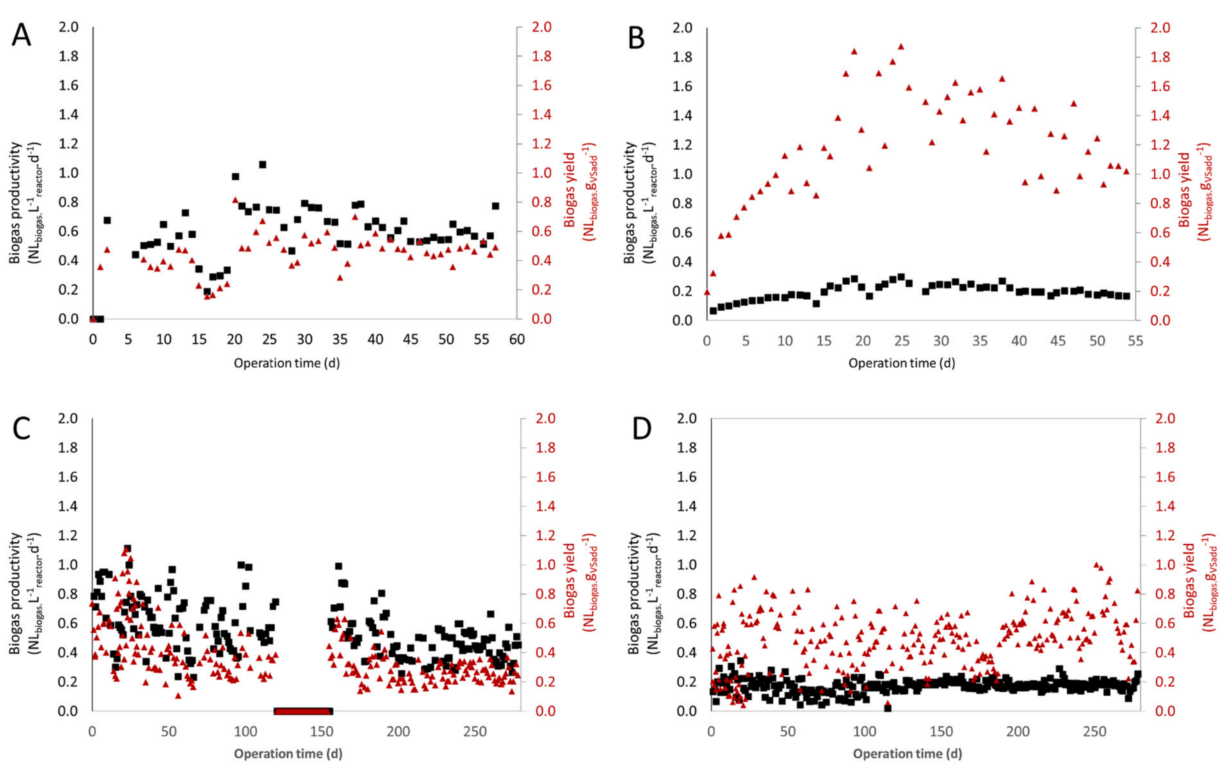

Fig. 2. Biogas yield and productivity of reactors studied. A. CSTR lab-scale. B CLB lab-scale. C. CSTR full-scale. D. CLB full-scale.

Table 2
CSTR and CLB reactor performance and efficiency on volatile solid reduction from full and lab scale.

		pH			HRT (d)			OLR ($kg_{vsadd}\cdot m^{-3}{}_{reactor}\cdot d^{-1}$)			T (°C)			VS removal (%)			VFA/TA ($mg_{HAC}\cdot mg_{CaCO3}^{-1}$)		
Lab Scale	BLC	7.46	±	0.22	30	±	0	0.165	±	0	24	±	1.2	71	±	7	0.289	±	0.064
	CSTR	7.76	±	0.20	36	±	1.5	1.340	±	0.09	37	±	1.0	58	±	7	0.117	±	0.009
Full Scale	BLC	7.37	±	0.17	28	±	11	0.356	±	0.163	23	±	3.5	63	±	8	0.330	±	0.116
	CSTR	7.72	±	0.26	38	±	8	1.618	±	0.418	34	±	4	55	±	4	0.122	±	0.045

time is different in CLB. Therefore, the inoculum used in start-up of lab-scale reactor can be contributed with biogas production, and maximizing the biogas yield, since the inoculum is from a covered lagoon biodigester operating for 13 years. Nevertheless, around 55 to 60% and 60 to 70% of the VS for CSTR and CLB, respectively, were degraded to biogas in both digester scales. Lower VFA/TA ratios and high pH stability were also observed which means that anaerobic production efficiency of biogas in a lab-scale was reproduced in full-scale digester.

Furthermore, based on CSTR full-scale stability, the OLR could be increased, without process failure, increasing global biogas production and energy conversion. The Fig. 3 shows the global biogas production of the full-scale system where it could be observed that in approximately one year of monitoring 370,000 m^3 of biogas were produced, being used in the CHP unit. SRS and StS represented 10% of the volume of initial wastewater and contained around 27% of organic materials, methane and nutrients and contributed with around 30% of global biogas generation.

3.5. Learning with large-scale biological desulfurization

Desulfurization is required to avoid damages to combustion equipment. H_2S in combination with humidity can form corrosive acids, which is besides its toxicity the main reason to remove it, before the gas is used. The injection of small amounts of oxygen or ambient air into the biodigester headspace is a well-known and comparatively cheap way to reduce the H_2S concentration significantly. To avoid formation of sulfuric acid (H_2SO_4) and other compounds, like SO_2, as acid rain precursor, it is important limit the oxygen content in biogas. In addition, it is known that excess of oxygen and the generation of sulfide inhibits methanogenesis (Hilton and Oleszkiewicz, 1988). Over the period of full-scale monitoring, it was possible to identify daily variations in the quality of biogas influenced by operational variations of the system. The high data resolution (biogas sampling every 2 h) allowed a better understanding of H_2S generation and to make decisions regarding changes in the plant to benefit the microaerobic desulfurization system.

One of the perceptions was the direct influence of the biogas residence time (BRT) on the biodigester headspace in the H_2S concentration. The Fig. 4a shows H_2S concentration at biogas headspace in the CLB that would occur daily during the operation of the electric generator group (CHP unit). At some periods, the generation of electric energy occurred in a discontinuous way (around 8 to 12 h d^{-1}) according to the biogas availability. In these periods, the flow variation of biogas consumption oscillated between zero (when CHP turned off) to 55 $m^3\ h^{-1}$ (when CHP turned on) and this produced an H_2S concentration increasing effect approximately from 50 to 500 ppm.

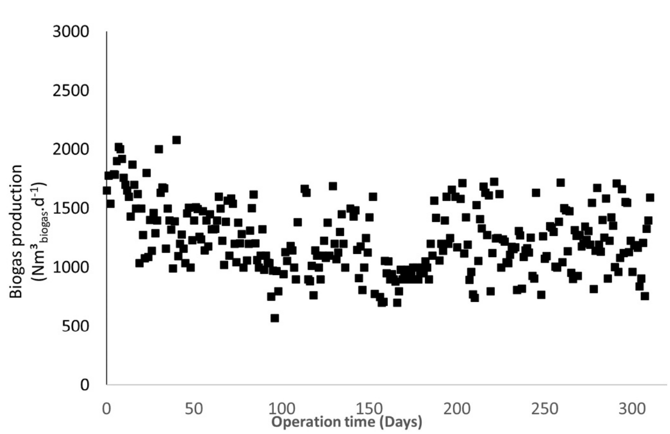

Fig. 3. Global biogas production of swine manure anaerobic digestion.

Another possible explanation for the growth of H_2S during the day could related to the manure loading in the digesters, since the operational management of the manure and cleaning of the swine facilities occurs manually during the morning period. It is known that the generation of H_2S is dependent on the organic loading in the digester, since the generation of the component gases of biogas is dependent on the of substrate composition (Hilton and Oleszkiewicz, 1988; Stams et al., 2003).

However, the Fig. 4a shows a strong correlation between H_2S and BRT. The concentration changes at the same moments when the consumption of biogas increases (CHP turn on) indicating that the BRT is the most important condition for the process control. The efficiency of biological desulfurization is dependent on the contact time of the sulfide oxidizing microorganisms with the biogas with oxygen levels below 1%. There is also a dependency on headspace homogeneity and microorganism abundance, but usually for a H_2S removal efficiency higher than 90% the BRT must be longer than 5–10 h (Khoshnevisan et al., 2017).

This BRT influence in the gas composition can also be observed by comparing the periods in which there was an inversion of the biogas flows. In the period prior to December 1, 2018, the biogas pipeline has been connected following the flow CLB ➔ CSTR ➔ CHP unit. The CSTR headspace volume is 530 m^3 resulting in an estimated BRT of 7–10 h. The CLB biogas capacity is 5600 m^3 that corresponding to the BRT of 66–100 h. After December 1, the flow changed to the sequence CSTR ➔ CLB ➔ CHP unit. The Fig. 4b present the temporal concentration of H_2S (gray line) and CH_4 (black line) at the CHP intake in the period between January 31, 2018 and March 31, 2020. The inversion of the flow allowed an increase in stability, resulting in a decrease in the average concentration of H_2S from 763 ± 823 to 457 ± 379 mg L^{-1} and an increase in the CH_4 concentration from 61.5 ± 2.4 to 65.2 ± 1.9%. This increase in the concentration of methane was expected, as it is already known that the better stability on sulfur oxidizing bacteria activity promotes a lower concentration of H_2S in the digester and thereby reduces competition with methanogenic microorganisms (Hilton and Oleszkiewicz, 1988; Mulbry et al., 2017; Ramos et al., 2014a, 2014b; Stams et al., 2003).

The robustness and efficiency of the desulfurization process was also observed. Fig. 4c shows the H_2S profile over 120 h of operation in which there happens an electric power failure to the desulfurization equipment and interrupted oxygen supplementation in the headspace of the CLB. Immediately the concentration of H_2S increased from 14 to 4388 ppm in a 56-h interval. After correcting the operational problem that generated the power interruption to the desulfurizer system, the H_2S concentration returned to the previous level after 32 h. Although a maximum stable value was not verified, it is possible to infer that the biological desulfurization presented an efficiency of H_2S removal above 98%. This is similar of the efficiency reported by Mulbry et al. (2017) that shown 99% removal efficiency for similar micro-aeration system in plug-flow reactor for dairy manure.

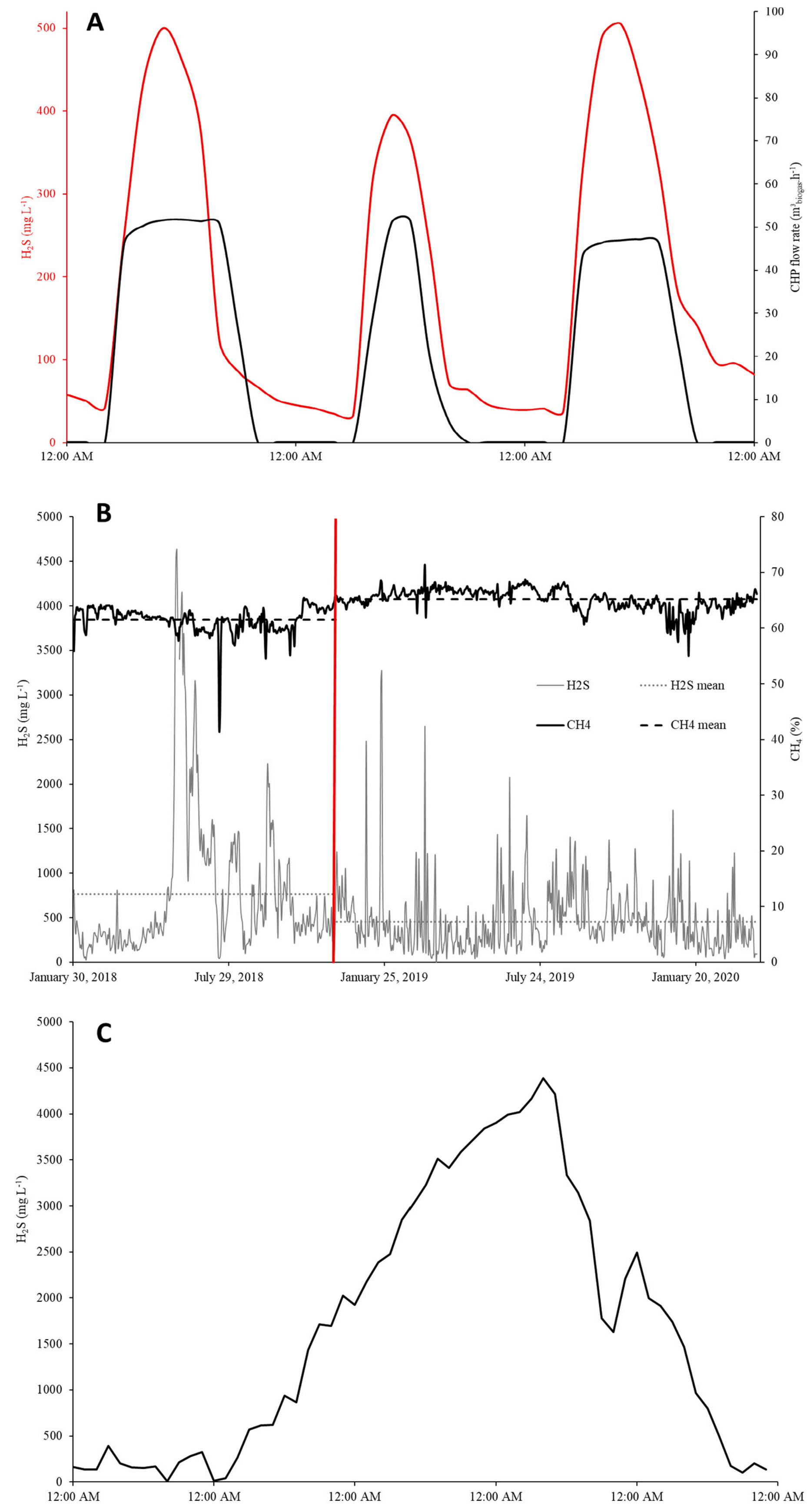

Fig. 4. Desulfurization process monitoring. A. Variations in the H_2S content at biogas headspace in the CLB in relation of electric generator group (CHP unit) operation. B. temporal concentration of H_2S (gray line) and CH_4 (black line) at the CHP intake from January 2018 to March 2020. C. hydrogen sulfide profile over 120 h during interrupted oxygen supplementation in the headspace of the CLB.

4. Conclusion

The SLS as a pre-treatment is an interesting alternative to enable the treatment of swine manure fractions using different reactor configurations. The main benefits are: 1) CSTR size reduction 2) Stability in biogas generation in CSTR reactor, treating the solid fraction 3) Solid over loading reducing effect for CLB improving biogas generation 4) Possibility to implement a digestate treatment system for CLB that presents more diluted effluent due to previous SLS. Moreover, the biogas monitoring allowed a better understanding to gas production and concentration variation. The gas concentration and H_2S removal was BRT dependent.

CRediT authorship contribution statement

Deisi Cristina Tápparo: Conceptualization; Formal analysis; Investigation, Writing - original draft. **Daniela Cândido**: Conceptualization; Formal analysis; Investigation; Writing - review & editing. **Ricardo Luis Radis Steinmetz**: Conceptualization; Investigation; Writing - review & editing. **Christian Etzkorn**: Writing - review & editing. **André Cestonaro do Amaral**: Conceptualization; Investigation; Writing - review & editing: **Fabiane Goldschmidt Antes**: Conceptualization; Investigation; Writing - review & editing. **Airton Kunz**: Conceptualization; Funding acquisition; Investigation; Project administration, Writing - review & editing.

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ATTACHMENT 62

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Effluent treatment from biogas plants[1,2]

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Summary – The generation of energies from the biomethane fuel generated by the anaerobic bio-digestion process of animal effluents has received special attention in Brazil as a promising alternative and sustainable source of energy. However, technical and economic limitations persist in the process, mainly regarding the correct recycling of effluents. This prospective study analyzes the technical and economic feasibility of five alternatives available in the national market for the correct treatment and disposal of effluents generated in a biogas plant to be installed in the Southern region. The results obtained from the estimation of technical performance and use of production inputs and factors and consequent net present value (LPV) for the technologies were accounted for in order to estimate the respective economic impacts on the central biogas plant. The information generated can support the energy sector and the production chain of pig farming, as well as the formulation of public policies for renewable energies from biogas.
Keywords: agribusiness, biomethane, costs, prospecting, economic viability.

Effluent wastewater treatment from biogas plants

Abstract – Biomethane production from the anaerobic digestion of animal wastes has regained popularity in Brazil and therefore is receiving especial attention lately. Biomethane is an alternative source of renewable energy that can contribute to local agribusiness economy whereas decreasing our dependency on petroleum. Therefore, large scale biogas plants have been considered toincrease methane productivity to volumes that can make it commercially competitive. Despite the fact that biogas plants technologies are well established, little effort has been placed on how to treat the final wastewater adequately in order to maintain the sustainability of the process without

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jeopardizing its economic feasibility. The aim of this case-study was to determine the economic impacts of five most conventional wastewater treatment approaches available at local market on a specific pilot-scale biogas plant designed to operate in Southern Brazil. The benefits and technical limitations of each wastewater treatment approach were taken into consideration during analyses. The economic analyses were estimated based on net present value (NPV). Overall, the information gathered in this case-study can serve as guidance to decision makers during the development of newer biogas plants. Moreover, the results can assist public policies towards the development of sustainable agribusiness that is capable of producing biomethane as promising and profitable source of renewable energy.
Keywords: agribusiness, biomethane, costs, prospecting, economic feasibility.

Introduction

The generation of energy from the production of biogas with animal waste has grown in the world and in Brazil (DEUBLEIN; STEINHAUSER, 2011; PLANO..., 2012), with great potential to generate additional income and reduce agricultural and agro-industrial costs. There is also growing interest from governments, technology institutions, energy companies (electricity and gas), suppliers of machinery and equipment and specialized technical consulting services. Biogas also emerges as a potential solution to reduce greenhouse gas (GHG) emissions in agribusiness. Therefore, the projects in this area align with the Brazilian government's agenda for mitigating the effects of agricultural activities on climate change and is placed as an alternative to initiatives aimed at sustainable rural development, also attracting the interest of institutions with local action, such as the municipal governments and the technical assistance and rural extension agencies.

Although the use of biogas as an alternative source of energy is not recent, technical and economic limitations still persist in the process, especially with regard to the correct destination of the final effluent. Although they are conventionally considered a reliable source of fertilizers, effluents can promise the quality of water resources in regions with high herd concentration and little availability of agricultural area – in the case of many producing regions in the South of the country. In recent years, many viable initiatives have emerged for the production and use of animal waste biogas. One of these initiatives foresees the implementation of a central biogas plant in the South and with the participation of a public company in the energy sector, a foreign agency of international technical cooperation and the local city council.

This prospective study analyzed the technical and economic feasibility of five treatment routes and final disposal of effluents produced by a biogas plant for the Southern region, available in the market. The realization occurred under the technical cooperation contract between Embrapa Swine and Poultry and the German International Cooperation Agency (GIZ). The original project foresees the installation of a central power generation plant with a power of 500 kW to 1 MW with the biogas of pig, cattle and broiler manure waste from 78 agricultural establishments (BLOCK, 2011; OAK, 2011). Figure 1 shows the schematic design of the proposal, which provides as a strategy for the management of effluents (A and B) its transport for application as a biofertilizer and the composting and drying of silt, and it is up to the municipal government to bear the transport costs (from waste to the plant and effluents to agricultural areas).

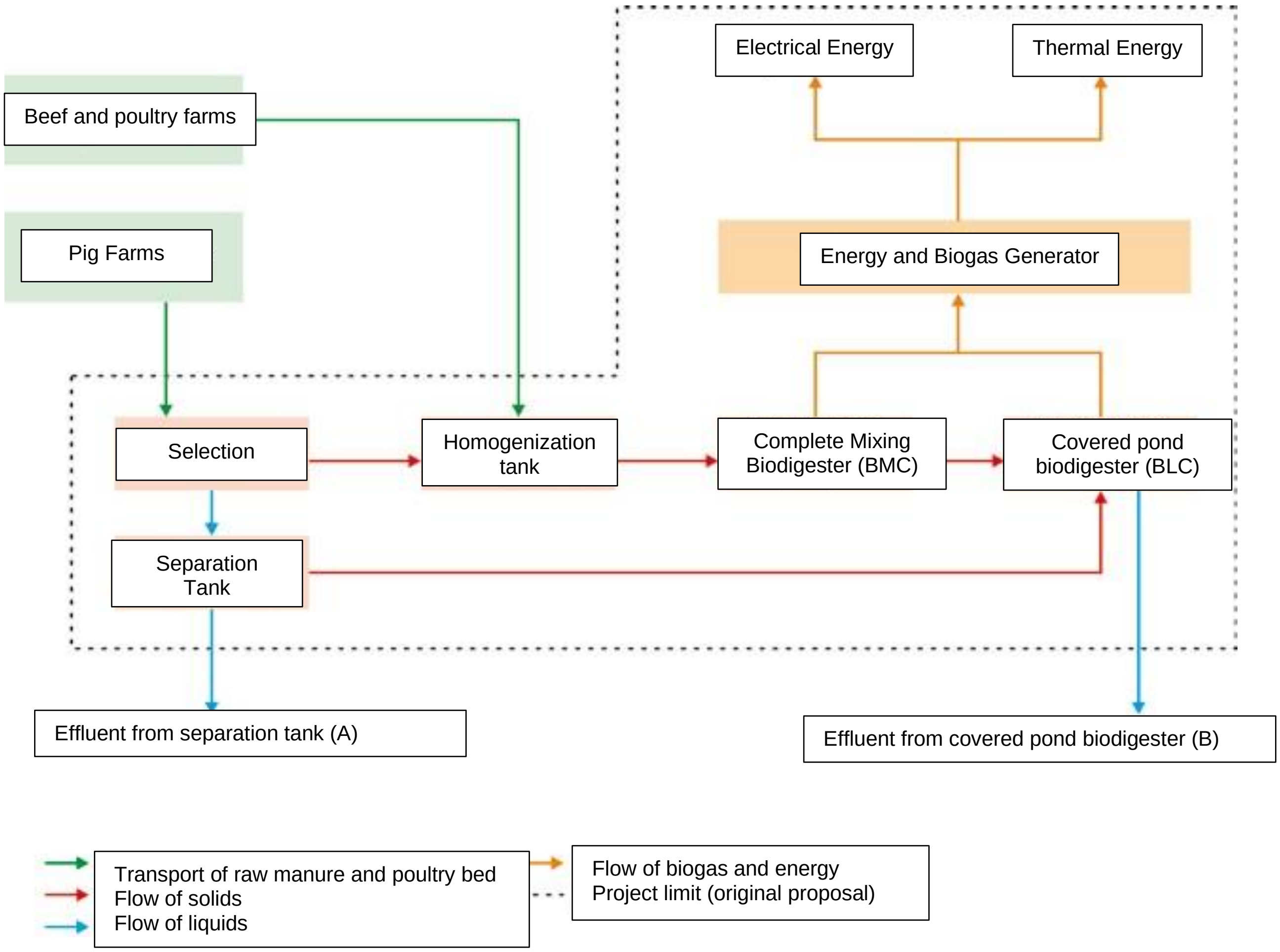


Figure 1. Schematic design of the biogas plant implementation project.
Source: Block (2011) and Carvalho (2011).

Scope of the study and technologies considered

The study did not propose to analyze the overall viability of the project, but only the impact of the five effluent treatment routes on the economic value generated by the plant. The treatment technologies are as follows:

1) Transport of effluents for application in the soil as liquid biofertilizer.
2) Separation of silt with decanter, transportation of liquid effluent for application in soil as biofertilizer and drying of silt with the use of raphyus bags for sale of the by-product (solid organic fertilizer).
3) Swine Effluent Treatment System (Sistrates) with disposal of effluents treated in water bodies (or reuse) and drying of silt of the phosphorus module and use of raphyus bags for sale of the by-product in the form of calcium phosphate.
4) Separation of silt with sand filter, followed by treatment of the liquid by reverse osmosis and subsequent disposal in water bodies (or reuse), transport of untreated liquid effluents for application in soil as biofertilizer

and drying of sand filter silt using raphyus bags for sale of the by-product – solid organic fertilizer, for example.

5) Use of effluents in place of water supplied by the public network or artesian well in the process of manufacturing fluid fertilizers and subsequent sale of the product in the fertilizer market. This is not exactly an alternative, but a new business.

A flowchart of technological alternatives is presented in Figure 2. The use of liquid effluents (A and B in Figure 1) for composting was discarded because of the absence of carbon as substrate for the biological process (HIGARASHI, 2012). The drying of the effluent with the heat from the plant's generators was also discarded because this process does not present technical feasibility (BLOCK, 2012a).

Methodology

The methodology used is prospective and allowed estimating the technical performance and the use of production materials and factors (capital and labor) to calculate the net present value (LPV) of the five technological alternatives analyzed. Following the concept of incremental cash flow, the NPV of the technological alternatives should be counted in the value generated by biogas plants (GALESNE et al., 1999)[10]. A qualitative evaluation was also developed for each alternative, addressing the benefits, actions, disadvantages and potential risks. The case study concludes with considerations focused on public policy and the challenges for innovation in projects of this nature.

To estimate the annual input volume of waste, dry matter and nutrients, the following information was used: a) initial analysis of the herd volume of the 78 producers planned to participate in the project (BLOCK, 2011); b) average dry matter value of liquid waste

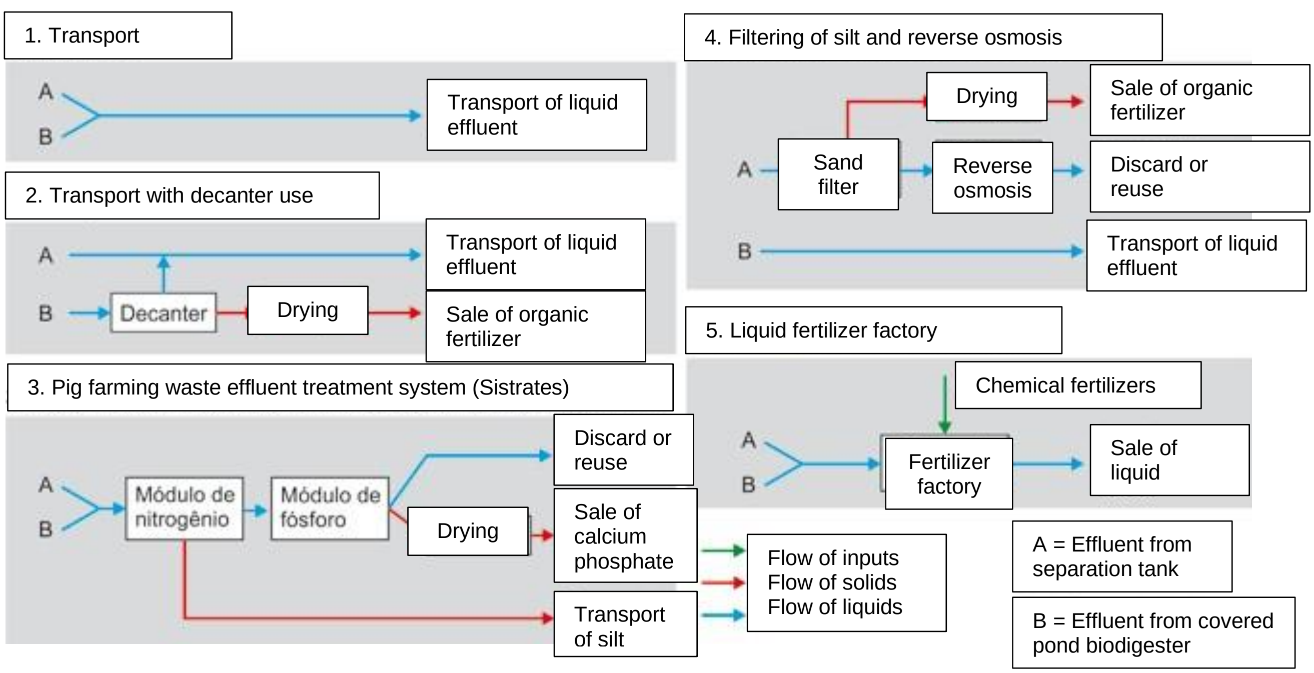


Figure 2. Technological alternatives for the treatment and final disposal of effluents from a central biogas plant in the South region.

[10] Discount rate of 10% per year or 0.797% per month, project life of 120 months and residual value of investments of 10%.

of pigs equal to 3.46%, obtained from the survey of 23 samples (RAMME, 2011); and c) consultation with previous surveys carried out by Embrapa Swine and Poultry (OLIVEIRA et al.,1993) and the Soil Chemistry and Fertility Commission of Rio Grande do Sul and Santa Catarina states (MANUAL..., 2004) to establish the concentration of nitrogen nutrients (N), phosphates (P_2O_5) and potassium (K_2O).

The volume and characteristics of the effluents were estimated based on the technical performance coefficients projected for the plant (BLOCK, 2011). The characteristics of effluents, silt and by-products of alternative technologies were estimated by means of technical information obtained in the development of swine effluent treatment systems, such as Sistrates[11] (KUNZ et al., 2009; MIELE et al., 2011) and reverse osmosis (BLOCK, 2012b). For the liquid fertilizer plant, field technical data (COZZO, 2012) were used. Table 1 presents the characteristics of effluents, used to determine the use of inputs and production factors by volume of effluent to be treated and estimate the initial investment necessary for the implementation of each alternative.

The fertilizer value was calculated from the concentration of nutrients in effluents, silt and by-products, and from the market price of fertilizers. Urea was considered to contain

Table 1. Characteristics of effluents, slides and by-products of the central biogas plant and the technological alternatives analyzed.

Effluent, silt and by-products	**Daily volume**	**Dry material (%)**	**N (kg/m³)**	**P_2O_5 (kg/m³)**	**K_2O (kg/m³)**
Biogas plant					
Effluent from the separation tank (A)	239 m³	0,81	2,74	1,53	1,92
Effluent from the covered pond biodigester (B)	147 m³	7,11	6,17	5,70	3,30
Treatment alternative no. 1 - Transportation					
Liquid effluent	386 m³	3,21	4,05	3,12	2,45
Treatment alternative no. 2 - Transport and decanter					
Liquid effluent	341 m³	1,46	4,10	3,50	1,70
Organic compost (dry decanter sod)	17 t	60,00	7,63	30,02	5,96
Treatment alternative no. 3 - Sistrates					
Disposal or reuse of water (standard allocation)	294 m³	0,81	0,01	0,01	0,84
N removal module silt	19 m³	3,21	0,14	0,58	0,12
Calcium phosphate (after drying)	12 t	60,00	0,00	34,88	0,00
Treatment alternative nº 4 – Filtering and reverse osmosis					
Liquid effluent	204 m³	1,70	7,98	3,18	7,61
Disposal or reuse of clean water	172 m³	0,00	0,00	0,00	0,00
Organic fertilizer (dry silt from sand filter)	2 t	60,00	97,57	113,34	14,28
Treatment alternative nº 5 – Fertilizer plant					
Liquid fertilizer (0-10-10)	597 t	Nd	0,00	100,00	100,00

Source: prepared with data from Block (2011, 2012a, 2012b), Cozzo (2012), Kunz et al. (2009), Manual... (2004), Miele et al. (2011), Oliveira et al. (1993) e Ramme (2011).

[11] The Patent of Sistrates was filed in February 2011 with the INPI (PTO), as PI (Application for Invention), protocol no. 012110000133.

44% N; triple superphosphate contains 42% P_2O_5; and potassium chloride (KCl) contains 60% K_2O. It is emphasized that this value is a “shadow price” given to effluents, silt and by-products, because they have no market price (except fluid fertilizer). Thus, they are illustrative values that allow comparing effluents and by-products on the same basis (value in NPK) and do not necessarily constitute revenue, sales price or production cost. The economic benefit provided by effluents is equal to the fertilizer value when they are applied in their own areas. When applied in third-party areas, they have an equal value to the sales price, which can suffer deductions of up to 100%.

The study considered as a baseline for the comparison of technological alternatives the cost of transporting effluents from the central plant to the crops of the region for application in the soil as a biofertilizer (Figure 2, alternative 1), deducing the economic benefit generated by these effluents. This cost is determined by the equipment used, by the price of the inputs (diesel, tires and maintenance, labor and truck value) and mainly by the average distance traveled. To estimate the average distance, a model was developed based on the supply and demand of nutrients in the plant's implantation region and in the 127 municipalities with distance of up to 150 km, in addition to the willingness of farmers in the region to accept the effluents from the plant. The data sources and information used were: data from the Municipal Livestock Survey (PPM) (IBGE, 2012b) to size the pig herd; b) estimation of the supply of manure and nutrients (NPK) by pig herds from the average concentration of 3.46% of dry matter, average daily excretion of 8.6 L/head/day and average nutrient concentration of 3.13 kg/m³ for N, 2.68 kg/m³ for P_2O_5 and 1.63 kg/m³ for K_2O (MANUAL..., 2004; OLIVEIRA et al., 1993; RAMME, 2011)[12]; c) data from the Municipal Agricultural Survey (PAM) (IBGE, 2012a) to size the planted area (ha) of the main crops; d) estimation of nutrient demand (NPK) based on fertilization recommendations for maintenance and correction of nutrient contents in the soil as a function of NPK extraction capacity by agricultural crops cultivated in the region, considering the critical phosphorus element (NICOLOSO; CORREA, 2012); e) the willingness of farmers in the region to accept the effluents of the plant and the consequent percentage of the available area. As they are unknown variables, a simulation was made with several levels of acceptance (from 100% to 10% of the planted area). The estimated unit cost of effluent transport (R$/km and R$/m³) was based on Miele et al. (2011) and Sandi et al. (2011, 2012), being considered only 165 working days without rain per year (NATIONAL INSTITUTE OF METEOROLOGY, 2012) – the application of effluents cannot be performed on rainy days. Table 2 shows the market prices used in this study.

Baseline to compare alternatives: effluent transport

Alternative no. 1, transportation of effluents from the biogas plant for application in the soil as a biofertilizer of crops, is the one of lowest technological complexity and greater flexibility (MAYERLE, 2011)[13] and represents the first strategy for management of swine manure in Brazil and is also the management strategy of the effluents proposed by the project (CARVALHO, 2011). Therefore, this option was considered the main baseline for comparison with the other alternatives.

The benefits of this alternative are the use of nutrients contained in waste and effluents (NPK) and its important role as a soil conditioning agent (carbon and other nutrients), reducing farmers’ expenses

[12] Since ppm data do not discriminate the rearing system and the type of pig herd, it was decided to use daily excretion and the average concentration of herd nutrients.

[13] Logistic setup with centralized production of biogas and electricity and decentralization in waste collection and effluent distribution (MAYERLE, 2011).

Table 2. Market prices used in this study.

Item	Unit	Price[1]
Organic fertilizer from decanter[2]	R$/t	16.27
Organic fertilizer from rapid sand filter[2]	R$/t	170.91
Purchased electricity	R$/kWh	0.264
Electricity sold	R$/kWh	0.140
Liquid fertilizer (00-10-10)	R$/t	519.82
Solid fertilizer (00-25-25)	R$/t	1,214.54
Calcium phosphate from Sistrates[2]	R$/t	28.84
Freight of the inputs to the liquid fertilizer plant	R$/t	35.47
Liquid fertilizer freight (100 km one way)	R$/t	58.000
Freight of by-products sold (35 km one way)[3]	R$/t	15.85
Calcium hydroxide	R$/kg	0.50
KCl	R$/t	1,262.22
MAP	R$/t	1,491.00
Diesel oil	R$/L	2.12
Polymers	R$/kg	25.00
Raphyus bags for drying silt	R$/bag	10.00
Operator salaries (includes charges)	R$/month	1,353.00
Triple superphosphate	R$/t	1,328.74
Tannins	R$/L	1.85
Urea	R$/t	1,197.13

[1] Average values for the period of Jan./2011to May/2012, updated by IGP-DI for Jun./ 2012.

[2] FOB price calculated, considering 60% of dry matter and discount of 74% in relation to fertilizer value, according to discount verified in the aviary market.

[3] For the sale of organic fertilizer and calcium phosphate, we considered the distance of 35 km from the plant's installation site to the neighboring municipality where it is cooperative that produces and markets organic fertilizers by composting agro-industrial waste.

Source: prepared with data from the National Agency of Petroleum, Natural Gas and Biofuels (2012), Cozzo (2012) and Santa Catarina (2012)

with mineral fertilizers and also positively impacting crop productivity[14]. In situations in which the effluent is applied in its own areas, it can be affirmed that the effluent holder will benefit from its fertilizer value equivalent to the reduction of expenses with chemical fertilizers. However, when the effluent is applied in third-party areas, the benefit will be equivalent to the selling price, not the fertilizer value. The selling price is very variable, but can suffer deductions of up to 100% in relation to fertilizer value. In the marketing of poultry for consumption, the deductions vary from 66% to 92% in relation to the fertilizer value. The main limiting factor of this alternative is the high concentration of pig herds and other confined farms in relation to the low availability of nearby agricultural areas for the recycling of waste in an environmentally correct manner. This situation does not occur in all producing regions, but is particularly characteristic of the traditional swine production areas of Southern Brazil, in the case of the region chosen for the plant installation.

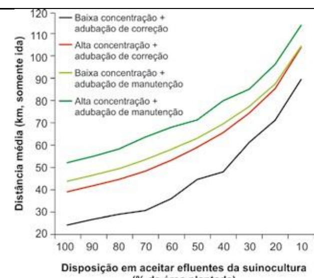
[14] This economic analysis did not consider the positive effects on soil quality and crop productivity, only their fertilizer value (reduction of chemical fertilizer expenses) or their sales price when applicable.

The soils of the region and its surroundings are characterized by the low aptitude for mechanized agriculture (high slope and rocky), which limits the agricultural area available for biofertilizer application, and by high natural fertility. Given this and the need for the enterprise (plant) to be sustainable in the long term, the nutrient demand had to be calculated based on the recommendations of fertilization to maintain the nutrient contents in the soil as a function of the capacity of NPK extraction by agricultural crops cultivated in the region. As there is no up-to-date and available information on the level of soil fertility in the region, the fertilizer demand in the first years of the project was considered to be potentially higher than that used in this study, given the need to correct soil fertility. However, over the lifetime of the project, fertilizer demand should fall to the estimated levels necessary for soil maintenance, keeping the agricultural scenario of the region unchanged and correctly following agronomic recommendations.

In addition to the supply of NPK via biofertilizers of the plant, it is also necessary to compute the supply of biofertilizers in the municipalities potentially receiving the effluents of the plant, so that the problem of excess nutrients is not only transferred to the other municipalities. In view of the recommendations for fertilization (correction vs. maintenance), the variability of nutrient concentration of effluents and the levels of willingness of farmers to accept the effluents of the plant, great variability and uncertainty should be predicted in relation to the distance traveled for the transport of effluents. Figure 3 shows the simulation of the model described above, which predicts that the distance for the transportation of effluents from the central biogas plant can vary from 24 km to 115 km (one way)[15].

Average distance (km, one way)

Low concentration + corrective fertilizer
High concentration + corrective fertilizer
Low concentration + maintenance fertilizer
High concentration + maintenance fertilizer



Willingness to accept effluents from pig farming (% of planted area)

Figure 3. Weighted average distance to transpose the effluents of the central biogas plant as a function of the willingness to accept the effluents of swine culture, the concentration of nutrients in the effluents and the agronomic recommendation for fertilization.

The large range of distances covered (Figure 3) is reflected in the cost of transport of effluents, which may present great variability. These projections are aligned with the transportation costs of liquid pig waste with the use of tanker trucks that participated in the surveys by Sandi et al. (2011, 2012) in municipalities in the South region. It is important to compare the distance required to meet the agronomic recommendation as a function of the willingness in accepting the effluents (Figure 3) with the maximum distance that guarantees the viability of the transport as a function of the condition of the effluent appraisal. The transport and subsequent application of effluents in third-party areas are only feasible at distances lower than that necessary to meet the agronomic recommendation, even when the availability to accept effluents is 100% of the planted area (Table 3).

[15] The study did not consider the supply of manure or NPK by the bovine herd because most of it is not reared in confinement – waste is absorbed by grazing areas. It was also not considered the herd of broilers or other birds because the poultry bed can be marketed at greater distances than pig manure, being, in many cases, acquired by organic fertilizer factories. If the supply of NPK by these herds were considered, a significant increase in the average distance covered should be predicted.

Table 3. Distance required to meet the agronomic recommendation due to the willingness to accept effluents, maximum distance for economic viability of transportation due to the condition of effluent stocking and respective transport costs.

Criteria	Average distance (Km, one way)	Transportation Cost	
		R$/km	R$/m^3
Distance to meet the agronomic maintenance recommendation using 100% of the planted area	44	3.72	16.32
Distance to meet the agronomic recommendation of maintenance and high acceptance of effluent (70% of the planted area)	54	3.64	19.55
Distance to meet the agronomic recommendation of maintenance and low acceptance of effluent (30% of the planted area)	78	3.53	27.50
Economically viable distance for application in own area (cost = fertilizer value)	73	3.55	26.01
Economically viable distance for application in third-party area (cost = market price with average discount of 74% on fertilizer value)	15	4.52	6.96

Based on these considerations, it is believed that these are the most likely conditions of removal of effluents from the central biogas plant: a) predominance of good quality soils requiring only maintenance fertilization; b) low concentration of nutrients in waste; c) low willingness to accept effluents from the biogas plant; d) great variability and uncertainty regarding the distance traveled for the transport of effluents (Figure 3) and, consequently, in transport costs (Table 3); e) application in a third-party area, with high discount on its fertilizer value, also subject to great variability and uncertainty[16].

The main consequence for the baseline of this study is that the net cost[17] of effluent transportation will most likely be positive (cost of transport > economic benefit), subject to great variability, which can be one of the main sources of uncertainty of a project with such characteristics. Since it is not possible to determine the standard value, we chose to use two scenarios for the analysis:
a) scenario 1, of low net transport cost, of 1.50 R$/m^3 (defined from consultation with pig producers in the Southern region with large scales of accommodation and which does not have an agricultural area); b) scenario 2, of high net transport cost, of 2.71 R$/m^3, which would represent an increase of 80% over scenario 1.

Both scenarios were considered conservative (underestimating the net cost of transportation) given the characteristics described above, the trends observed in the relationship between herd and agricultural area (IBGE, 2012a, 2012b) and also the future behavior of prices (especially diesel oil). It is also important to highlight the possibility of biogas generated being used as a fuel source to move the truck fleet, which could be an attractive alternative for reducing costs and atmospheric emissions (GHGs). At the moment, this technology is still incipient and commercially unavailable in Brazil. In addition, a reduction in the plant's biogas supply is expected, which would consequently reduce project revenues.

[16] The high volatility of exchange rates and the price of fertilizers on the international market also directly impacts this variable.

[17] The net cost of transporting the effluent is equivalent to the transport cost deducted from the fertilizer value of the effluent when it is applied in its own areas or minus the sales price of the effluent when it is applied in third-party areas.

Results

Alternative no. 1, of effluent transport, requires the lowest initial investment, of R$ 357,000, a value that refers to the construction of a wastewater storage pond with a volume of 30,900 m³, for a period of 80 days[18]. Investments in its own fleet were not accounted for, assuming that the enterprise would outsource freight services[19]. Thus, the main component of the cost are freight expenses (Tables 4 and 5). In Scenario 1, it is the lowest cost option of approximately R$2.08/m³ of effluent (Tables 4 and 6), as well as the lowest impact on the value generated by the central biogas plant, with a reduction in the LPV of the plant of R$ 1.8 million in ten years (Table 7). Scenario 2 presents a cost of R$3.29/m³ of effluent (Table 6), with a negative impact on the value generated by the plant, estimated at R$ 2.9 million over ten years (Table 7).

The main benefit of this alternative is the possibility of using biofertilizer, given its low cost and greater flexibility of use for small distances. However, it can present disadvantages when contributing to GHG emissions and by detaining large agricultural areas of their own to benefit from fertilizer value. It would also present high cost when considering greater transport distances.

Alternative no 2, transport with separation of silt with decanter, was considered a variant of the alternative no 1, not being properly a distinct technological option. This option requires the second lowest investment

Item	**Transport**	**Transport + Decanter**	**Sistrates**	**Filtering + Reverse Osmosis**	**Liquid Fertilizer Factory**
Investment required	357,115	585,288	917,571	1,547,157	4,256,422
Expenses (cash outflows)	19,437	38,173	24,287	23,659	8,893,285
Transport of silt and effluent	17,380	15,334	869	9,201	0
Electricity	0	1,514	6,542	3,153	5,185
Other inputs and services	1,104	16,489	9,665	1,732	8,848,471
Maintenance	893	1,463	3,823	6,446	10,833
Labor	0	2,706	2,706	2,706	24,630
Other items	60	665	682	421	4,167
Depreciation and amortization	4,634	7,596	11,908	20,078	55,238
Total cost	24,071	45,768	36,195	43,737	8,948,523
Income from by-products (FOB)(1)	0	8,506	10,580	8,257	9,309,112
Freight of by-products sold(2)	0	8,286	5,816	766	1,038,676
Result	-24,071	-45,548	-31,431	-36,247	-678,087

(1) The volume and characteristics of the by-products are shown in Figure 3 and Table 3.

(2) The distance of 35 km was considered, only gone, for the sale of organic fertilizer and calcium phosphate, and 100 km for the fluid fertilizer.

18 18 Equivalent to the 120 days required by the legislation [technical standard of the State Foundation for Environmental Protection Henrique Luiz Roessler - RS (2014)] minus 40 days of hydraulic retention time (HRT) in the central biogas plant. This storage structure is also necessary for effluents generated on rainy days, during which it is not possible to apply them in crops.

19 If the option is by its own fleet, the investment should be increased by approximately R$ 774,000, referring to the acquisition of three tanker trucks.

Table 5. Initial investment and cost of technological alternatives for scenario 2 (R$/month).

Item	Transport	Transport + Decanter	Sistrates	Filtering + Reverse Osmosis	Liquid Fertilizer Factory
Investment required	357,115	585,288	917,571	1,547,157	4,256,422
Expenses (cash outflows)	33,434	50,523	24,987	31,069	8,893,285
Transport of silt and effluent	31,378	27,685	1,569	16,611	0
Electricity	0	1,514	6,542	3,153	5,185
Other inputs and services	1,104	16,489	9,665	1,732	8,848,471
Maintenance	893	1,463	3,823	6,446	10,833
Labor	0	2,706	2,706	2,706	24,630
Other items	60	665	682	421	4,167
Depreciation and amortization	4,634	7,596	11,908	20,078	55,238
Total cost	38,069	58,118	36,895	51,147	8,948,523
Income from by-products (FOB)[1]	0	8,506	10,580	8,257	9,309,112
Freight of by-products sold[2]	0	8,286	5,816	766	1,038,676
Result	-38,069	-57,899	-32,131	-43,657	-678,087

[1] The volume and characteristics of the by-products are shown in Figure 3 and Table 3.

[2] The distance of 35 km was considered, only gone, for the sale of organic fertilizer and calcium phosphate, and 100 km for the fluid fertilizer.

Table 6. Operational expenses and total cost of technological alternatives considered for scenarios 1 and 2 (R$/m³ of effluent).

Scenario	Income from By-products	Transport	Transport + Decanter	Sistrates	Filtering + Reverse Osmosis	Liquid Fertilizer Factory
Scenario 1	Operational expenses	1.68	3.29	2.10	2.04	768.55
	Total Cost	2.08	3.95	3.12	3.77	772.32
Scenario 2	Operational expenses	2.89	4.36	2.16	2.68	768.55
	Total Cost	3.29	5.02	3.18	4.41	772.32

Table 7. Net present value (LPV) of technological alternatives for scenarios 1 and 2 (R$ thousand)[1].

Scenario	Income from By-products	Transport	Transport + Decanter	Sistrates	Filtering + Reverse Osmosis	Liquid Fertilizer Factory
Investment Required		357	585	918	1,547	4,256
Scenario 1	No	-1,841	-3,504	-2,754	-3,311	
	Yes		-3,487	-2,387	-2,733	-52,087
Scenario 2	No	-2,920	-4,456	-2,808	-3,882	
	Yes		-4,439	-2,440	-3,304	-52,087

[1] Discount rate of 0.797% per month, project life of 120 months and residual investment value of 10%.

R$ 585,000, a value that refers to the construction of a wastewater storage pond with a volume of 27,300 m³, for a period of 80 days, and the installation of a decanter equipment (flow of 15 m³/h) for separation of silt from the covered pond biodigester. Investments with their own fleet were also not considered. The use of a decanter implies higher expenses with chemical and maintenance, but has shown little effectiveness in reducing freight expenses. This option was the one with the highest cost and the greatest negative impact on the value generated by the project. The revenue that could be obtained from the sale of by-products (organic fertilizer) has little importance in relation to costs and does not change these results (Tables 4, 5, 6 and 7).
Effluent treatment alternatives require greater initial investment compared to transport alternatives – Sistrates requires R$ 918,000 in investments, while reverse osmosis requires R$ 1,547,000. In addition to being more costly in terms of equipment, this latter option also requires the construction of a storage pond of the portion of untreated effluents for a period of 80 days, with a volume of 16,400 m³. The main components of the cost of treatment in Sistrates are chemical and electric energy, which account for almost half of the total costs and reduce the supply of energy to the market. Reverse osmosis equipment, although more energy efficient, has higher costs with depreciation and amortization of capital and maintenance expenses (especially in filter replacement), which together account for more than half of total costs (Tables 4 and 5). It is important to highlight that the reverse osmosis equipment does not treat the entire effluent, but only the effluent of the separation tank (A in Figure 2 and Table 3), with up to 2% organic dry matter, limit for this technology (BLOCK, 2012b). A significant portion of the cost (at least 20%) occurs because of the transport of untreated effluent (effluent from the covered pond biodigester, B in Table 1), which presents more than 2% of organic dry matter.
Sistrates presented treatment costs lower than the reverse osmosis option in the two scenarios analyzed (Tables 4, 5 and 6), as well as lower negative impact on the value generated by the central biogas plant (Table 7)[20]. In relation to the baseline, Sistrates has the lowest cost of scenario 2, approximately R$3.18/m³ of effluent (Table 6) and the lowest negative impact on the value being generated by the plant, approximately R$ 2.8 million in 10 years without obtaining revenue stemmed from the sale of by-products, and R$ 2.4 million with revenues (Table 7). The recipe that can be obtained from the sale of by-products (calcium phosphate and organic fertilizer) changes little the performance of these alternatives, especially in the case of Sistrates. It is important to note that in all options, except in Sistrates, there is a significant impact by the increase in the value of effluent removal freight in the total costs (scenario 1 x scenario 2).
Among the main benefits of Sistrates are the possibility of reducing GHG emissions, reuse of water or release in receiving bodies and the use of the generated by-product (calcium phosphate) as an input for fertilizers and feed. The advantages of this system are the efficiency of nutrient removal and the reduction of the area necessary for the application of animal production residues and consequent reduction of transport costs. As disadvantages, it presents high initial investment value, high operating cost and consumption of electricity and chemical inputs. The removal of N and P makes this technology less attractive because it reduces the supply of nutrients for agriculture. In addition, it is a technology still in the validation phase and is subject to fluctuations in the price of electricity and phosphate.
Reverse osmosis has characteristics similar to those of Sistrates. Its main

[20] This condition is maintained even when electricity tariffs practiced in the market (R$ 264.00/MWh) are considered instead of the cost of selling the energy used in this study (estimated at R$ 140.00/MWh).

benefits are the possibility of reducing GHG emissions, the production of clean water (about 40% of the total effluent) and the generation of organic fertilizer, being able to access value-added markets. The advantages of this system are the reduction of the volume of effluent transported, the reduction of the area necessary for the application of animal production waste and also the availability of clean water. However, it has the same disadvantages as Sistrates, besides not treating the entire effluent of the biogas plant (62%).

The alternative that requires the greatest investment is the implementation of a fluid fertilizer plant, with a projected total value of R$ 4.3 million and the capacity to produce 240,000 t/year. This option includes facilities and equipment in the amount of R$ 3.7 million and the construction of a wastewater storage pond for a period of 120 days, with a volume of 46,300 m^3 and a value of R$ 536 thousand, necessary because of the seasonality of demand due to the calendars of agricultural harvests.

It should be considered that this is not exactly a treatment alternative, but a new business in the fertilizer segment, which can be located in central biogas plants, using bio digestion effluents in industrial processes in place of clean water[21]. Therefore, the costs of the fluid fertilizer plant were not compared to those of the other treatment alternatives. Chemical inputs for the manufacture of fluid fertilizers (MAP, Urea and KCl) represent 99% of production costs (Tables 5 and 6) and 89% of costs when considering freight to the rural producer.

This option is extremely sensitive to the freight costs of the products sold. Because there is no return freight (specialized transport), one should charge the round-trip routes, unlike the case of solid fertilizer. In addition, this is a product intended for farmers with an area of more than 50 ha (COZZO, 2012).

These limitations also make the option of the liquid fertilizer plant dependent on the area of crops around the plant. Considering the formulation that uses the largest amount of effluent[22, it] would be necessary to produce 215,000 tons per year, which would require an area of 537,000 hectares of crops in establishments with more than 50 hectares. For the economic viability of the project, this area should therefore be at a distance of up to 50 km, or 100 km round trip (COZZO, 2012), which is not the case in the project region (IBGE, 2006, 2012a). The unfavorable logistics of the region chosen for the project and the low availability of agricultural areas directly impact the expected return of a fluid fertilizer plant (COZZO, 2012). This conclusion is maintained even in the situation in which resources from the central biogas plant to the fertilizer plant equivalent to the expenses with transportation of effluents occur (subsidy from one project to another). The advantage of this alternative that provides the sale of fluid fertilizers is to add value to the effluent, enabling transportation to greater distances (up to about 50 km). However, it requires high investment and operational cost with chemical and electricity. In addition, its priority audience is producers with areas greater than 50 ha and specific machinery is needed for the application of fertilizer in the soil.

Environmental legislation, cost-sharing and co-responsibility

In view of the differences in scale, it should be considered that the posture of environmental agencies in relation to central biogas plants will be guided by greater caution and a degree of requirement (release, supervision and monitoring and technological standard, for example) significantly more restrictive than with small and medium geographically dispersed pig farmers. As pointed out, the predominant option of

[21] It takes 565 L to 647 L of water (or effluent) to produce one ton of fluid fertilizer, depending on the formulation.
[22] Formulation 00-10-10 requires 647 L of effluent for each ton of fertilizer produced

management of pig waste in Brazil is its transport for later application as biofertilizer in crops. The cost is borne by the swine producers, and there is no co-responsibility of the other actors in the production chain (especially agro-industries), and in many producing regions there is a significant contribution of public resources to subsidize the transport of waste (SANDI et al., 2011, 2012)[23]. It is understood that the entry of new sectors in the biogas segment, such as energy, should be guided by the internalization of the costs involved in the transport or treatment of effluents. The original proposal to divide tasks in this plant project in the South region does not solve the problem. On the contrary, it increases the impacts on the municipal budget. In a survey conducted in the municipality, the subsidy reached 74.6% of the cost of distribution of waste in farms that needed to transport at distances greater than 8 km (SANDI et al., 2012), which represents annual expenditure for the municipal budget of R$ 111,000.

The public incentive to implement central biogas plants should consider that it is not possible to enable new ways of generating value on pig production (from meat to energy and fertilizer) without reserving part of this added value to the correct management of effluents, preferably considering the reduction of GHG emissions and pressure on water resources. The regulatory framework of auctions for the purchase of energy from renewable sources should consider such issues. In this sense, the scope of the project should be expanded to include the transport or treatment of effluents.

From the point of view of the agricultural establishment, it is important to reinforce that the producer will incur costs to participate in a project of this nature and magnitude, especially labor and investments to reduce the volume of water in waste and the length of their stay in the facilities (RAMME, 2011). Initiatives of this nature, taken in the agricultural establishment and affecting the resources of the producer, not only increase the efficiency of the plant in the generation of biogas but reduce the costs of transport or treatment of effluents. For this to happen, incentives should be established or explained, through financial remuneration or the possibility of increasing the herd through technical analysis on a case-by-case basis. It is understood that it would be convenient and promising to develop a contractual arrangement, similar to the integration of production with agro-industries, for the determination of standards and delivery volumes and, mainly, to determine economic advantages for the pig farmer from quality parameters of the supplied waste (especially in terms of volatile solids and dry matter).

Final considerations

The option of replacing small biodigesters of covered pond, geographically dispersed by rural properties, with central biogas plants allows significant gains in technical efficiency (CANTRELL et al., 2008). However, this option represents an increase in transportation costs and, above all, the risks of pollution of water resources, in view of the limitations of inspection and the economic incentive for effluents to be overapplied in areas near the plant. Regarding the alternatives analyzed for the correct management and final disposal of effluents from a central biogas plant, there are also dilemmas to be considered. The transport alternative and its variant, the use of decanter for silt separation, on the one hand provide the agricultural sector with the nutrients contained in pig farming waste[24], reducing costs with chemical inputs and ultimately reducing

[23] In addition, they represent a permanent focus of tensions, such as difficulties in meeting demand, political pressures to prioritize beneficiaries and impact on the budget of small municipalities.

[24] In many cases, the full use of nutrients is impaired because of the way effluents are used in the soil (ammonia loss, leaching and percolation) and the non-observance of agronomic fertilization recommendations.

the country's dependence on fertilizer imports. On the other hand, this option contributes to GHG emissions (ÁLVARES JUNIOR; LINKE, 2002; BRAZIL, 2009), either by emissions in transport (CO2), or by emissions from the use of waste applied to the soil (N2O). Moreover, as pointed out, this alternative increases the risk of pollution of water resources.

Treatment options reduce the polluting potential of pig farming and biogas plants of water resources and GHG emissions (BORTOLI et al., 2012). On the other hand, they reduce the surplus of electricity generated and do not return to the agricultural sector the nutrients contained in the waste (except organic fertilizer and calcium phosphate).

The combined use of central biogas plants with fluid fertilizer plants can be an alternative for the utilization of nutrients contained in waste without impacting water resources. As the effluent starts to make up a product to be sold (liquid fertilizer), that is, there is a price to be paid by the farmer, there is no economic incentive for its excess application, contrary to agronomic recommendations, in the case of the liquid effluent transported. However, a fertilizer plant consumes electricity and generates GHG emissions through transportation (ÁLVARES JUNIOR; LINKE, 2002; BRAZIL, 2011), and another limitation of this alternative is its maximum radius of activity, up to 50 km. Thus, even in regions with low herd/area ratio there is a limit of fertilizer production scale, which limits the scale of the central biogas plant to half or one third of the size proposed in the project.

Another issue that should be analyzed in projects of central biogas plants, which permeates all the alternatives described here, refers to the sanitary implications, which should be evaluated and discussed with the multiple actors of the process, as well as agricultural defense and control agencies and agro-industries.

In this study, quantitative data and qualitative questions were presented and discussed to prospect the technical and economic feasibility of various effluent treatment routes for the project of implementation of a biogas plant in the South region. What is evident is that the viability of the plant goes through the logistics strategy adopted and the stocking of effluents, but, above all, it passes through the dimensioning of the scale of the central plant, which should consider the availability of agricultural areas or the adoption of treatment technologies. Unfavorable logistics and low availability of agricultural areas limit their scale or raise the costs with the correct destination of effluents. Under the energy business's focus, this negatively affects profitability and LPV and prompts the change of project location to more favorable regions. Already under the focus of local public policy focused on rural development, it is possible that environmental demands and tensions in agriculture require interventions to deal with effluents from animal production, even if they have to operate with lower profitability than that accepted by the market.

Finally, it should be considered that the biogas segment has been characterized by accelerated and increasing incorporation of innovations, which tend to increase efficiency or reduce the costs of the technological alternatives analyzed. These innovations should also expand the range of options available, such as the use of biogas as a substitute for diesel oil in the transport of effluents or the development of modern and efficient physical-chemical and biological systems for the treatment of effluent from biodigesters. In both cases, significant implications for the results of this study can be expected.

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ATTACHMENT 63

DIGESTATE MANAGEMENT

OUR SOLUTIONS

Benefits & Impacts

Reduce digestate handling costs, improve gas production and digester throughput, concentrate nutrients, and reclaim clean water for reuse or discharge.

ULTRAFILTRATION (UF)

- Recover 65-85% of digestate as a transparent filtrate free of pathogens and suspended solids with very low levels of phosphorus and organic nitrogen
- Filtrate is ideal for high-volume application to fields and will not clog irrigation systems
- Concentrate is rich in undigested volatile solids and bacteria that can be recycled to the digester to increase solids retention time, thereby boosting gas production and allowing higher volumetric throughput of feedstock
- Concentrate is also a phosphorus-rich slurry that can be cost effectively transported to fields further away or used to produce value-added composts and fertilizers

TWO-STEP REVERSE OSMOSIS (TSRO)™

- UF filtrate is converted into clean water for reuse or discharge and a concentrated liquid fertilizer
- Water reclamation reduces digestate hauling, spreading, and lagoon storage requirements while decreasing freshwater consumption

FACILITY BENEFITS

- Use UF to create low-phosphorus filtrate for cost-effective land application or sewer discharge
- Increase daily loading rates to boost tip fee revenues while maintaining efficient gas production
- Reduce digestate disposal costs
- Reduce regulatory and operating risks of land application

HOW IT WORKS

Technology & Performance

ULTRAFILTRATION

- Proven, polymer-free membrane technology with demonstrated results at commercial digester
- Stainless steel tubular design handles high temperatures, high solids, high viscosities, and extremes in pH
- Designed to last 10-15 years in challenging applications with little downtime, maintenance, or repair
- Cleans quickly and automatically using standard cleaning solutions
- Can operate in batch or continuous modes and be mounted horizontally or vertically

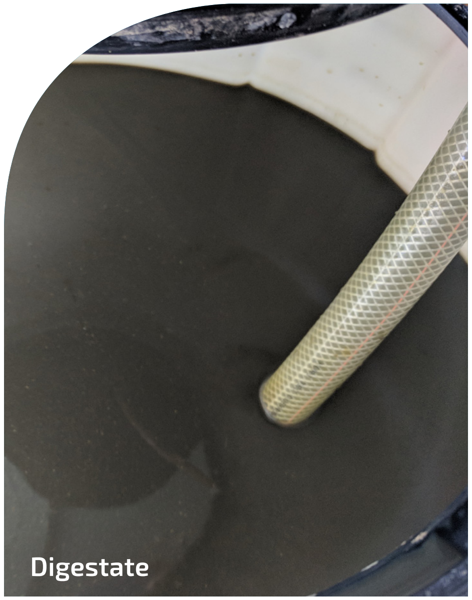
TWO-STEP REVERSE OSMOSIS

- Unique combination of Forward Osmosis (FO) and Reverse Osmosis (RO) membrane technologies can reduce UF filtrate volume by 65-90%
- Highly reliable, non-biological, and polymer-free solution to purify UF filtrate
- Produces clean water ready for reuse or discharge and a value-added product high in ammonia nitrogen and potassium
- Highly automated design requires minimal operator attention with demonstrated results at commercial digester

UF Equipment

TSRO Equipment

PHONE: 844.934.4378 | **WEBSITE:** digestedorganics.com | **EMAIL:** info@digestedorganics.com



FOOD WASTE DIGESTATE ULTRAFILTRATION

PARAMETER	UNITS	DIGES-TATE	UF FILTRATE	UF CONC.	PERCENT CONCENTRATED BY UF (%)
Total solids	%	2.79	1.32	5.66	53%
Total suspended solids	mg/L	18,400	140	55,500	99%
Total volatile solids	mg/L	16,000	4,140	38,600	74%
Total Kjeldahl nitrogen	mg/L	2,000	1,090	3,780	46%
Phosphorus	mg/L	445	15	814	97%
Potassium	mg/L	1,060	888	682	16%

Data from filtering digestate food waste with 70% filtrate recovery.

Contact us today for a quote
844.934.4378

We will help you...

- Reduce digestate disposal costs
- Boost revenues by processing more feedstock and making more biogas
- Increase digester efficiency and throughput

JUST ASK OUR CUSTOMERS...

"Ultrafiltration is a useful tool for biogas developers and digester operators. We believe it can be used to concentrate feedstocks like manure prior to digestion, thereby reducing hauling costs, and concentrate volatile solids and bacteria in digestate that can be returned to the digester to enhance gas production while minimizing hydraulic residence time. This means we can digest manure from more cows through the same size digester, reducing the capital costs and footprint for digestion facilities."

- Dan Nemke, CTO at Dynamic Group

EXPLORING Filtration

Call to learn how a Digestate Filtration System can help your facility. We will assess your current situation and recommend the best solution for you.

TESTING Filtration

We will pilot test your feedstock at our facility in Michigan and provide a thorough report and proposal for a full-scale system.

IMPLEMENTING Filtration

We deliver, install, and commission your new filtration system and teach employees how it works, ensuring it meets your expectations.

MAINTAINING Filtration

The filtration system will become an integral part of your facility, allowing you to streamline operations, significantly reduce costs, and increase your revenues.

PHONE: 844.934.4378 | **WEBSITE:** digestedorganics.com | **EMAIL:** info@digestedorganics.com

ATTACHMENT 64

US008673046B1

(12) **United States Patent**
Szogi et al.

(10) **Patent No.: US 8,673,046 B1**
(45) **Date of Patent: Mar. 18, 2014**

(54) **PROCESS FOR REMOVING AND RECOVERING PHOSPHORUS FROM ANIMAL WASTE**

(75) Inventors: **Ariel A. Szogi**, Florence, SC (US); **Matias B. Vanotti**, Florence, SC (US); **Patrick G. Hunt**, Florence, SC (US)

(73) Assignee: **The United States of America, as represented by the Secretary of Agriculture**, Washington, DC (US)

(*) Notice: Subject to any disclaimer, the term of this patent is extended or adjusted under 35 U.S.C. 154(b) by 404 days.

(21) Appl. No.: **12/905,226**

(22) Filed: **Oct. 15, 2010**

Related U.S. Application Data

(63) Continuation-in-part of application No. 12/026,346, filed on Feb. 5, 2008, now abandoned.

(51) **Int. Cl.**
C05F 3/00 (2006.01)

(52) **U.S. Cl.**
USPC **71/21**; 71/11; 71/12; 71/15; 210/669; 210/666

(58) **Field of Classification Search**
USPC 71/21
See application file for complete search history.

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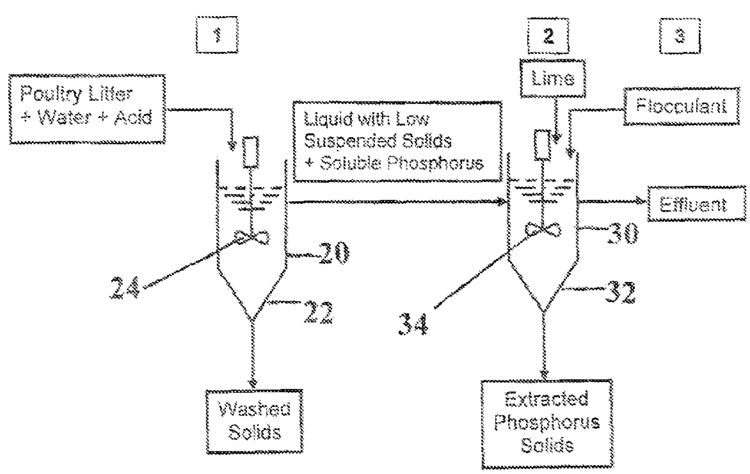
Primary Examiner — Patricia L Hailey
Assistant Examiner — Colette Nguyen
(74) *Attorney, Agent, or Firm* — John D. Fado; G. Byron Stover; Gail E. Poulos

(57) **ABSTRACT**

A process for removing phosphorus from solid poultry or animal wastes involving (a) mixing solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue (having a N:P ratio of at least more than 4 expressed on an elemental basis), (b) separating the liquid extract from the washed solid residue to form separated liquid extract and separated washed solid residue, (c) mixing the separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0, (d) mixing the liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid, and (e) separating the precipitated phosphorus solids from the liquid to form separated phosphorus solids and separated liquid. The process is conducted at a temperature greater than about 5° C. and less than about 50° C. The solid poultry or animal wastes are not pretreated prior to mixing the solid poultry or animal wastes with water and acid; such pretreatments include those that may cause the complete or partial loss of oxidizable organic carbon and nitrogen.

21 Claims, 9 Drawing Sheets



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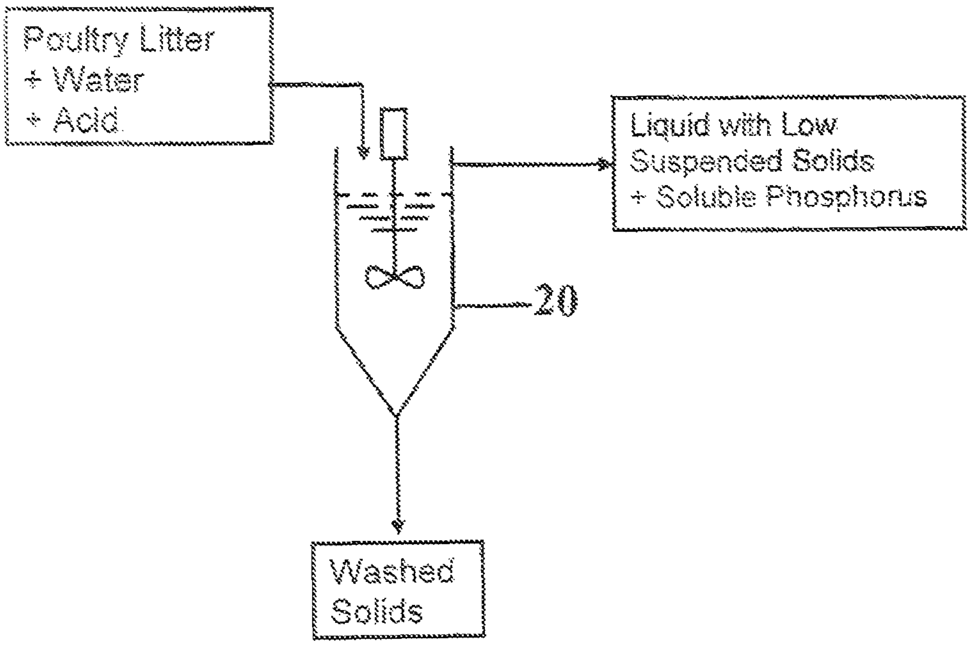

Poultry Litter
+ Water
+ Acid
Liquid with Low
Suspended Solids
+ Soluble Phosphorus
20
Washed
Solids

FIG. 1

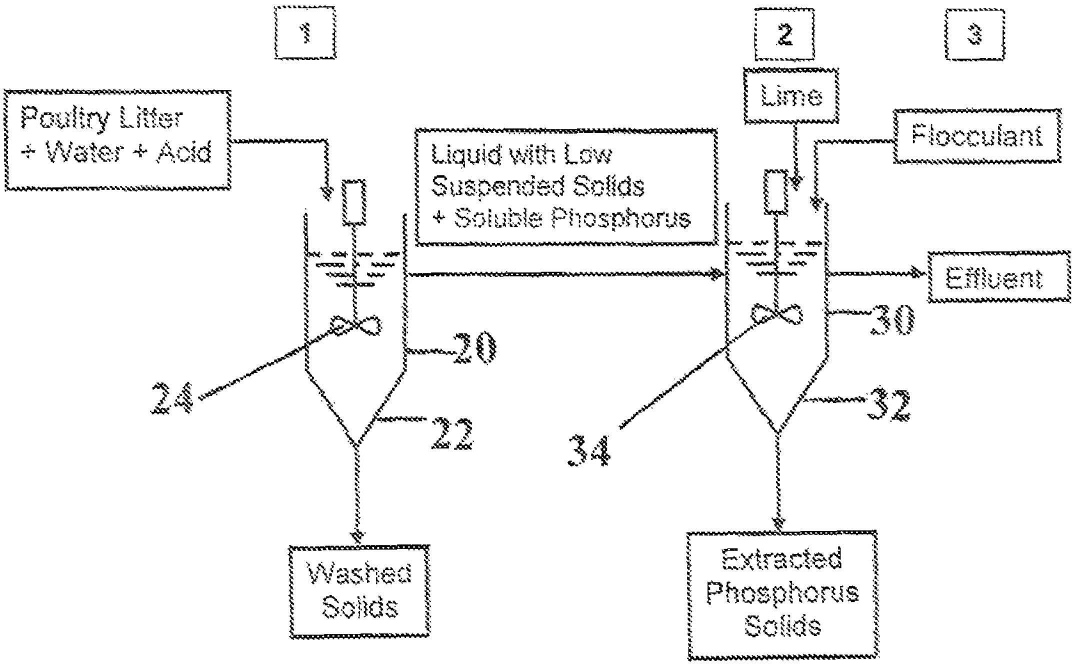

1
2
3
Poultry Litter
+ Water + Acid
Liquid with Low
Suspended Solids
+ Soluble Phosphorus
Lime
Flocculant
Effluent
20
22
24
30
32
34
Washed
Solids
Extracted
Phosphorus
Solids

FIG. 2

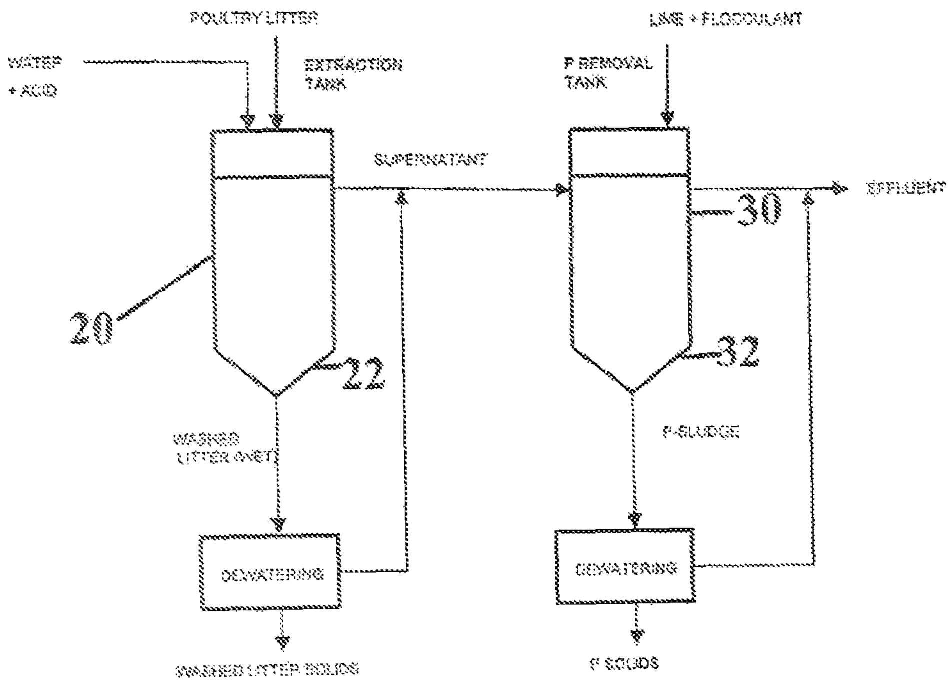

POULTRY LITTER
WATER
+ ACID
EXTRACTION
TANK
SUPERNATANT
LIME + FLOCCULANT
P REMOVAL
TANK
EFFLUENT
20
22
30
32
WASHED
LITTER (WET)
P-SLUDGE
DEWATERING
DEWATERING
WASHED LITTER SOLIDS
P SOLIDS

FIG. 3

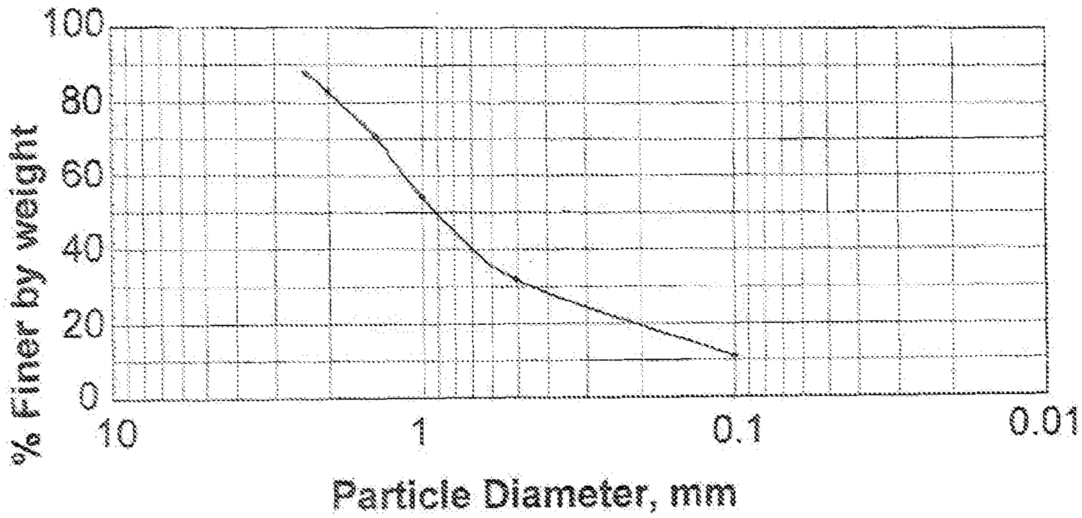

100
80
60
40
20
0
% Finer by weight
10
1
0.1
0.01
Particle Diameter, mm

FIG. 4

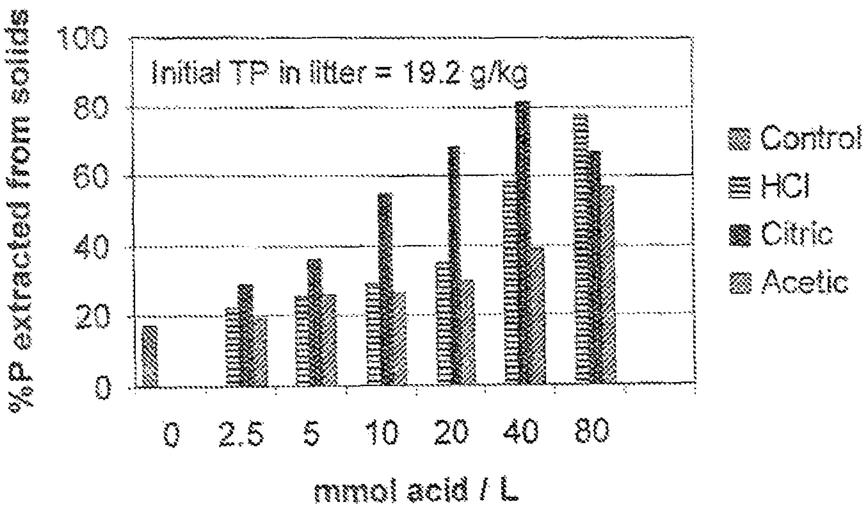

%P extracted from solids
100
80
60
40
20
0
Initial TP in litter = 19.2 g/kg
Control
HCl
Citric
Acetic
0
2.5
5
10
20
40
80
mmol acid / L

FIG. 5

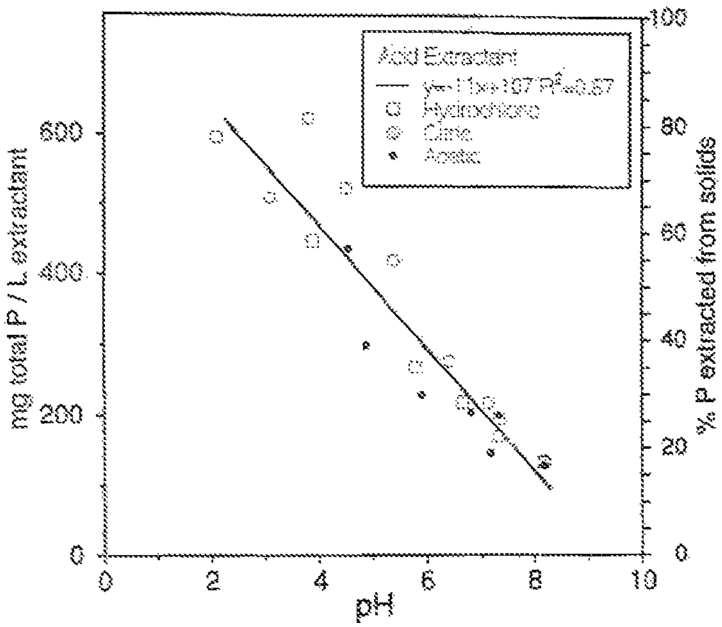

Acid Extractant
Hydrochloric
Citric
Acetic
mg total P / L extractant
% P extracted from solids
pH
0
200
400
600
0
20
40
60
80
100
0
2
4
6
8
10

FIG. 6

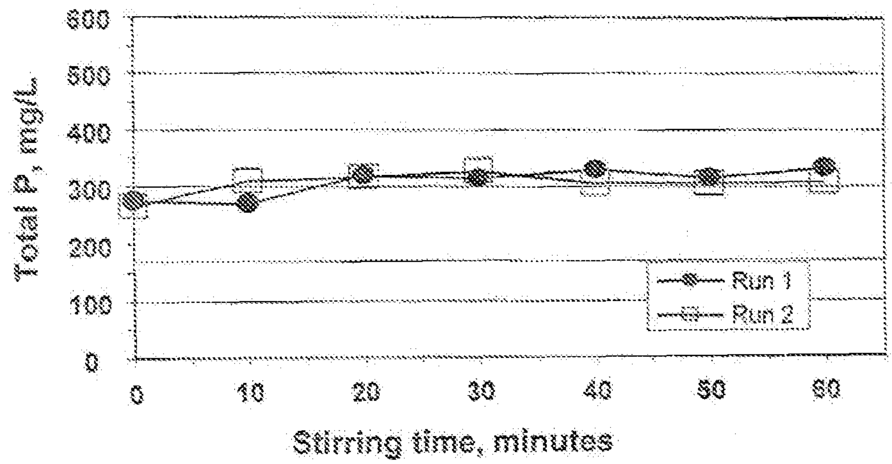

600
500
400
300
200
100
0
Total P, mg/L
0
10
20
30
40
50
60
Stirring time, minutes
Run 1
Run 2

FIG. 7

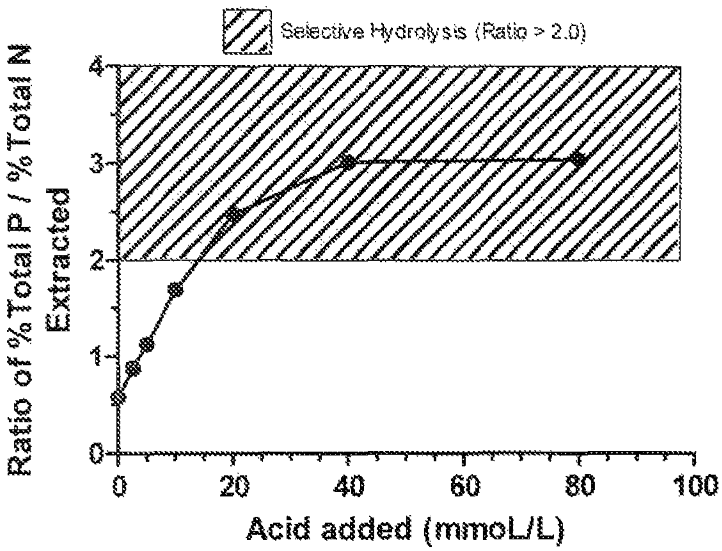
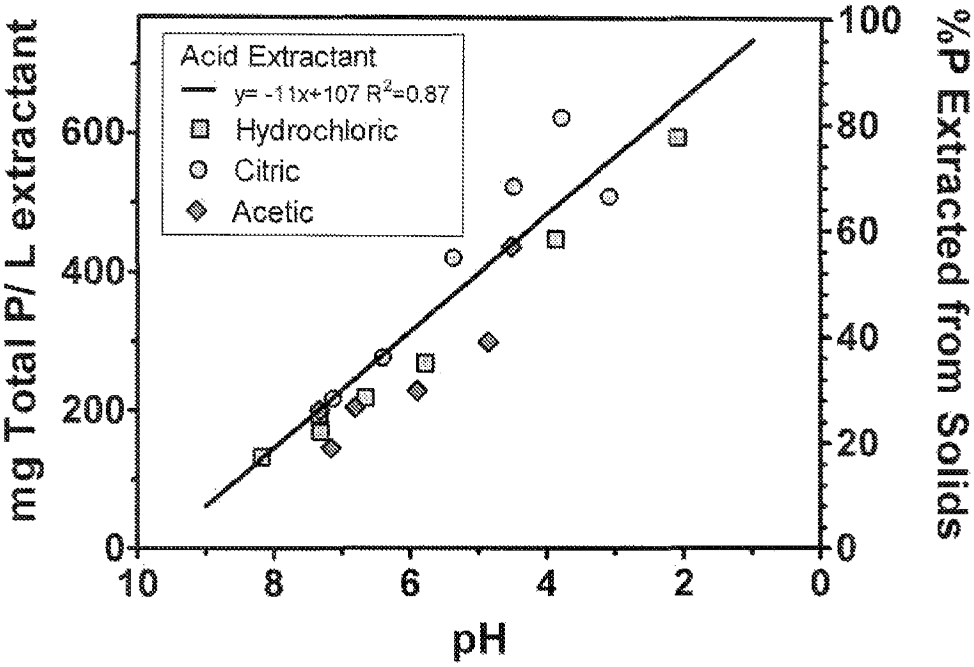

Selective Hydrolysis (Ratio > 2.0)
Ratio of % Total P / % Total N Extracted
4
3
2
1
0
0
20
40
60
80
100
Acid added (mmoL/L)

FIG. 8



Acid Extractant
y= -11x+107 R²=0.87
Hydrochloric
Citric
Acetic
mg Total P/ L extractant
%P Extracted from Solids
pH
0
200
400
600
10
8
6
4
2
0
20
40
60
80
100

FIG. 9

PROCESS FOR REMOVING AND RECOVERING PHOSPHORUS FROM ANIMAL WASTE

REFERENCE TO RELATED APPLICATION

This application claims the benefit of U.S. patent application Ser. No. 12/026,346, filed 5 Feb. 2008, which is incorporated herein by reference in its entirety.

FIELD OF THE INVENTION

The invention relates to a process for extracting and recovering phosphorus from animal wastes.

BACKGROUND OF THE INVENTION

Animal production, a major component of the U.S. agricultural economy, is at risk because of both real and perceived environmental problems. Dramatic advancements are required to protect the environment, save this vital industry, and maintain food security. Municipal and agricultural waste disposal is a major problem. For agricultural animals, the animals are confined in high densities and lack functional and sustainable treatment systems. Confined livestock produce approximately 1329 million pounds of recoverable manure phosphorus annually with about 70% (approximately 925 million pounds) in excess of on-farm needs. This livestock production system was developed in the early and mid 20^{th} century prior to the current trend in high concentrated livestock operations. One of the main problems in sustainability is the imbalance of nitrogen (N) and phosphorus (P) applied to land (Edwards, D. R., and T. C. Daniel, Bioresource Technology, 41: 9-33 (1992)). Nutrients in manure are not present in the same proportion needed by crops, and when manure is applied based on a crop's nitrogen requirement, excessive phosphorus is applied resulting in phosphorus accumulation in soil, phosphorus runoff, and eutrophication of surface waters (Sharpley et al., J. Soil Water Conserv, 62: 375-389 (2007); Heathwaite, L., et al., J. Environ. Qual., 29: 158-166 (2000); Sharpley, A., et al., J. Environ. Qual., 29: 1-9 (2000); Edwards and Daniel, Bioresource Technology, 41: 9-33 (1992)).

Phosphorus build up in soils to excessively high levels due to animal manures often results in eutrophication and pollution of surface waters due to intense application of animal manures to land (Edwards and Daniel, 1992; USEPA, 1992; Heathwaite et al., 2000; Sharpley et al., 2000). This is a national problem affecting dairy, poultry, and swine production systems. Consequently, a substantial amount of manure phosphorus needs to be moved at least off the farms and some needs to be transported longer distances beyond county limits to solve accumulation and distribution problems of this nutrient (USDA-ERS, Agricultural Outlook, September 2000, p. 12-18). Manure nutrients in excess of the assimilative capacity of land available on farms are an environmental concern often associated with confined livestock production. The ability to extract-phosphorus from manure will be critical to poultry and livestock producers to accomplish manure utilization through land application without elevating soil phosphorus levels when land is limited. In addition, the aspect of phosphorus reuse is becoming important for the fertilizer industry because the world phosphorus reserves are limited (Smil, V., Annu. Rev. Energy Environ., 25(1):53-88 (2000)). According to the Potash and Phosphate Institute, the United States annual consumption of inorganic phosphorus for crop production is about 3700 million pounds (Potash and Phosphate Institute, 2002, Plant nutrient use in North American agriculture, Technical Bulletin 2002-1). On the other hand, for the U.S. as a whole, confined livestock produces about 1,329 million pounds of recoverable manure phosphorus annually with about 70% (about 925 million pounds) in excess of on-farm needs (Kellogg, R. L., et al., Manure nutrients relative to the capacity of cropland and pastureland to assimilate nutrients: Spatial and temporal trends for the United States, NRCS and ERS GSA Publ. No. nps00-0579. Washington, D.C., 2000). Therefore, reuse of phosphorus recovered from animal waste could substitute about 25% of the phosphorus now obtained from mining.

Farmers obtain nutrients for their crops from inorganic commercial fertilizers and from organic sources such as animal manure and biosolids from wastewater treatment plants. Inorganic nitrogen and phosphorus compounds are water soluble and readily available to plants. Most organic nutrient sources contain both inorganic forms of nutrients and forms that must first be mineralized or decomposed to become available to plants. The movement of nitrogen and phosphorus through soil are different. If nitrogen is converted to the highly water soluble nitrate-nitrogen form, and it is not used during plant growth, it can move through the soil-water system and be vulnerable to leaching into groundwater. Soil amended with large quantities of organic or inorganic phosphorus may generate significant amounts of soluble phosphorus that can be readily transported by surface and subsurface runoff and groundwater leachate.

A further problem with the management of human and animal waste is the loss of nutrients. Phosphates and nitrates are fundamental nutrients which determine the possibility for plant and animal life to occur. They are taken up by plants and the plants are eaten by animals. Subsequently they should return to the soil as manure in a normal agricultural cycle, but in the present situation in most cases they end up washed into the sea, whether they are simply dumped in a river or go through a municipal wastewater treatment

The lack of closure of the nutrient cycle is a major environmental problem, especially in the case of phosphates which, at present, are considered a mineral resource to be extracted. Excess of phosphates in the seas causes eutrophication. The depletion of the mineral phosphate resources is a problem which will become important in the near future (Scrivani et al., Solar trough concentration for fresh water production and waste water treatment, Desalination, 206: 485-493 (2007))

In livestock operations, the crop acreage is typically calculated to allow for uptake by the crops of the applied nitrogen from the soil, thus minimizing movement of nitrogen in ground and surface water beyond the farm's boundaries.

Unlike carbon and nitrogen, phosphorus cannot volatilize from the system. Crops typically take up less phosphorus from the soil than that applied in the manure because the acreage has been calculated for nitrogen removal, which requires less acreage. The soil absorbs phosphorus but over time reaches saturation. Additional application of phosphorus can cause release of phosphorus to surface waters beyond the farm's boundaries, risking oxygen depletion of water organisms. Measures for reducing phosphorus content of manure must be considered.

Phosphorus inputs accelerate eutrophication when it runs off into fresh water and has been identified as a major cause of impaired water quality (Sharpley et al., 2000). Eutrophication restricts water use for fisheries, recreation, industry, and drinking due to the increased growth of undesirable algae and aquatic weeds and resulting oxygen shortages caused by their death and decomposition. Also many drinking water supplies throughout the world experience periodic massive surface blooms of cyanobacteria. These blooms contribute to a wide range of water-related problems including summer fish kills, unpalatability of drinking water, and formation of trihalomethane during water chlorination. Consumption of cyanobacteria blooms, or water-soluble neuro- and hepatoxins released when these blooms die, can kill livestock and may pose a serious health hazard to humans. Recent outbreaks of the dinoflagellate *Pfiesteria piscicida* in near-shore waters of the eastern United States also may be influenced by nutrient enrichment. Although the direct cause of these outbreaks is unclear, the scientific consensus is that excessive nutrient loading helps create an environment rich in microbial prey and organic matter that *Pfiesteria* and menhaden (target fish) use as a food supply. In the long-term, decreases in nutrient loading will reduce eutrophication and will likely lower the risk of toxic outbreaks of *Pfiesteria*-like dinoflagellates and other harmful algal blooms. These outbreaks and awareness of eutrophication have increased the need for solutions to phosphorus run-off.

Past research efforts on phosphorus removal from wastewater using chemical precipitation have been frustrating due to the large chemical demand and limited value of by-products such as alum sludge, or because of the large chemical demand and huge losses of ammonia at the high pH that is required to precipitate phosphorus with calcium (Ca) and magnesium (Mg) salts (Westerman and Bicudo, Tangential flow separation and chemical enhancement to recover swine manure solids and phosphorus, ASAE Paper No. 98-4114, St. Joseph, Mich., ASAE, 1998; Loehr et al., Development and demonstration of nutrient removal from animal wastes, Environmental Protection Technology Series, Report EPA-R2-73-095, Washington, D.C., EPA, 1973). Other methods used for phosphorus removal include flocculation and sedimentation of solids using polymer addition, ozonation, mixing, aeration, and filtration (see U.S. Pat. No. 6,193,889 to Teran et al). U.S. Pat. No. 6,153,094 to Craig et al. teaches the addition of calcium carbonate in the form of crushed limestone to form calcium phosphate mineral. The patent also teaches adsorbing phosphorus onto iron oxyhydroxides under acidic conditions.

Continuing efforts are being made to improve agricultural, animal, and municipal waste treatment methods and apparatus. U.S. Pat. No. 5,472,472 and U.S. Pat. No. 5,078,882 (Northrup) disclose a process for the transformation of animal waste wherein solids are precipitated in a solids reactor, the treated slurry is aerobically and anaerobically treated to form an active biomass. The aqueous slurry containing bioconverted phosphorus is passed into a polishing ecoreactor zone wherein at least a portion of the slurry is converted to a beneficial humus material. In operation the system requires numerous chemical feeds and a series of wetland cells comprising microorganisms; animals, and plants. See also U.S. Pat. Nos. 4,348,285 and 4,432,869 (Groeneweg et al); U.S. Pat. No. 5,627,069 to Powlen; U.S. Pat. No. 5,135,659 to Wartanessian; and U.S. Pat. No. 5,200,082 to Olsen et al (relating to pesticide residues); U.S. Pat. No. 5,470,476 to Taboga; and U.S. Pat. No. 5,545,560 to Chang.

One of the main problems in sustainability of poultry production is the imbalance between nitrogen and phosphorus in the waste (Edwards and Daniel, USEPA, 2001). Nutrients in manure are not present in the same proportion needed by crops. The mean N:P ratio in manure is generally lower than the mean N:P ratio taken up by major grain and hay crops (USDA, 2001). To solve the problem of a phosphorus buildup in soil and increased potential for phosphorus losses through runoff and subsequent eutrophication of surface waters, efforts are being made to immobilize phosphorus or find alternative uses for poultry litter such as burning and gasification and transport to agricultural lands with low levels of phosphorus. Current methods for handling phosphorus in waste include immobilization, see for example U.S. Pat. No. 6,923,917; gasification (Sheth, A. C., and A. D. Turner, Trans. ASAE, 45(4):1111-1121 (2002)), precipitation, see U.S. Pat. No. 7,005,072; litter transport to agricultural lands with low levels of phosphorus (Jones, K., and G. D'Souza, Agric. Resour. Econ. Rev., 30(1):56-65 (2001); Kelleher, B. P., et al., Bioresour. Technol., 83(1) 27-36 (2002); Keplinger, K. O., and L. M. Hauck, Impacts of livestock concentration and application rate restrictions on manure utilization, ASAE/CSAE Meeting Presentation, Paper No. 042204. ASAE, St. Joseph, Mich., 2004); anaerobic digestion by combustion (USDOE-NREL, 2000, Biomass co-firing: A renewable alternative for utilities, NREL/FS-570-28009, DOE/GO-102000-1055, U.S. Department of Energy, National Renewable Energy Laboratory), etc.

While various systems have been developed for treating solid animal waste for the removal of phosphorus, there still remains a need in the art for a more effective treatment system for the phosphorus. The present invention, different from prior art systems, provides a system for extracting phosphorus from solid animal manure using a selective extraction and subsequent recovery.

SUMMARY OF THE INVENTION

In accordance with the present invention, there is provided a process for removing phosphorus from solid poultry or animal wastes involving: mixing the solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein the washed solid residue has a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in the solid poultry or animal wastes; separating the liquid extract from the washed solid residue to form separated liquid extract and separated washed solid residue; mixing the separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0; mixing the liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid; and separating the precipitated phosphorus solids from the liquid to form separated phosphorus solids and separated liquid.

Also in accordance with the present invention, there is provided a material produced by a process involving: mixing solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein the washed solid residue contains a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in the solid poultry or animal wastes; separating the liquid extract from the washed solid residue to form-separated liquid extract and separated washed solid residue; mixing the separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0; mixing the liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid; and separating the precipitated phosphorus solids from the liquid to form separated phosphorus solids and separated liquid. The material is the separated phosphorus solids and contains greater than

about 10% P_2O_5, greater than about 10% Ca, less than about 5% N, and less than about 5% K as K_2O.

Still in accordance with the present invention is a material produced by a process involving: mixing solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue; and separating the liquid extract from the washed solid residue to form separated liquid extract and separated washed solid residue. The material is the separated, washed solid residue and contains a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in the solid poultry or animal wastes.

BRIEF DESCRIPTION OF THE DRAWINGS

FIG. **1** is a schematic drawing of the step **1** of the quick wash process of poultry litter showing phosphorus extraction showing an extraction tank **20** as described below.

FIG. **2** is a schematic drawing of the quick wash process showing: (1) mixing solid poultry or animal wastes with water and acid to perform a selective hydrolysis for phosphorus extraction in an extraction tank **20** to form a separated liquid extract and separated washed solid residue; (2) treatment of separated liquid extract with lime and flocculant in a phosphorus removal tank **30**, and to form precicpitated phosphorus solids (3) in removal tank **30** as described below.

FIG. **3** is a schematic drawing of the field prototype system for solid manure quick wash showing a field prototype system for a solid manure quick wash process including an extraction tank **20**, and a phosphorus removal tank **30** as described below.

FIG. **4** is a graph showing particle size distribution of homogenized broiler litter used in field prototype experiments as described below. Each data point is the mean of three replicates.

FIG. **5** is a graph showing extraction of phosphorus from poultry liter using acids at seven concentration levels as described below.

FIG. **6** is a graph showing the pH on total phosphorus extracted from broiler litter as described below. Total phosphorus concentration increased with decreasing pH of mineral and organic acids extracting solutions; more than about 50% of total phosphorus was extracted with respect to initial total phosphorus content in broiler litter at pH lower than 5 of the extracting acid solutions. The % phosphorus extracted from solids corresponds with values in FIG. **5**. Variables in regression line $y=-11x+107$ are x=pH and y=% phosphorus extracted from solids.

FIG. **7** is a graph showing effect of stirring time on total phosphorus concentration in the extract as described below. Broiler litter was extracted with citric acid solution at pH about 4.5 (step **1**). Data show that total phosphorus concentration stays stable in supernatant liquid with stirring time between about 20 and 60 minutes. Data points are average concentrations of two separate runs using the field prototype.

FIG. **8** shows effect of citric acid treatment on the ratio of percent total phosphorus extracted to percent total nitrogen extracted from poultry litter as described below. For purposes of the present invention, selective hydrolysis is defined as any hydrolysis reaction that, relative to the initial phosphorus and nitrogen contents in the solid poultry or animal wastes, allows extraction of at least twice as much of the phosphorus (in percentage terms) than the nitrogen (y-axis≥2.0, preferably >2.0); in other words, allows extraction of at least twice as much of phosphorus (in percentage terms) than nitrogen (e.g., 55% phosphorus and 25% nitrogen extracted).

FIG. **9** (another version of FIG. **6**) is a graph showing the pH on total phosphorus extracted from broiler litter as described below. Total phosphorus concentration increased with decreasing pH of mineral and organic acids extracting solutions; more than about 50% of total phosphorus was extracted with respect to initial total phosphorus content in broiler litter at pH lower than 5 of the extracting acid solutions. The % phosphorus extracted from solids corresponds with values in FIG. **5**. Variables in regression line $y=-11x+107$ are x=pH and y=% phosphorus extracted from solids.

DETAILED DESCRIPTION OF THE INVENTION

Land application of large amounts, of solid animal wastes is an environmental concern often associated with excess phosphorus in soils and potential pollution of water resources. Recovery of phosphorus from solid waste was developed for extraction and recovery of phosphorus from solid poultry or animal wastes (e.g., aminal solid manures and poultry manure). The term solid poultry or animal wastes includes any materials containing a mixture of poultry or animal urine, feces, undigested feed, and optionally bedding material. The invention can use different types of poultry manure such as litter (manure mixed with bedding material) or cake (manure with minimal bedding material). The invention can also use different types of animal wastes such as manure mixed with bedding materials (such as in deep bedding systems for pig or cow rearing) or animal wastes with minimal bedding material (such as scraped or centrifuged manure or manure collected with belt systems).

The process generally includes three steps: (1) phosphorus extraction, (2) phosphorus recovery, and (3) phosphorus recovery enhancement. In the first step (FIG. **1**), solid poultry or animal wastes (e.g., animal solid manures or poultry manure) is washed by mixing it with water and acid in a reactor vessel at a pH lower than about 5.0 (e.g., lower than 5.0) and higher than about 3.0 (e.g., higher than 3.0); preferably at a pH of about 3.1 (e.g., 3.1) to about 5.0 (e.g., 5.0), preferably at a pH of about 3.1 (e.g., 3.1) to about 4.5 (e.g., 4.5) to form a liquid extract and a washed solid residue. The washed solid residue is settled and is dewatered to prevent unnecessary carbon and nitrogen oxidation and digestion; the washed solid residue contains the oxidizible organic carbon and nitrogen fraction that would be digested and oxidized if the animal solid manure or poultry manure had been instead washed by mixing it with water and acid at a low pH (e.g., below 3.0) or that would be lost by ignition (or that would be lost if the animal solid manure or poultry manure had been incinerated before being mixed with water and acid at a low pH (i.e., below 3). The first step extracts at least 60% (e.g., at least 60%) of the phosphorus contained in the original solid poultry or animal wastes (in other words, no more than 40% of the original phosphorus remains in the washed solid residue while the balance is in the liquid extract), preferably at least about 65% (e.g., at least 65%), preferably at least 67%, preferably at least 68%, preferably at least 81%; and the first step extracts no more than about 30% (e.g., 30%) of the nitrogen contained in the original solid poultry or animal wastes (in other words, no more than 70% of the original phosphorus remains in the washed solid residue while the balance is in the liquid extract), preferably no more than 22.0%, preferably no more than 26.8%, preferably no more than 27.4%. The washed solid residue has a N:P ratio (expressed on an elemental basis) of at least more than about 4 (e.g., more than 4; preferably at least 4.1, more preferably at least 4.3, preferably at least 4.4, preferably at least 5.5, preferably at least 9.8, preferably at least 11.1). The washed solid residue contains no more than about 40% (e.g., no more than 40%) of the total phosphorus (P and/or P_2O_5) in the original untreated waste (preferably no more than 37.3%, preferably no more than 35.2%, preferably no more than 29.9%, preferably no more than 26.9%, preferably no more than 26.7%, preferably no more than 26.3%, preferably no more than 25.9%, preferably no more than 25.7%, preferably no more than 24.7%, preferably no more than 22.5%, preferably no more than 18.4%, preferably no more than 17.6%, preferably no more than 16.7%, preferably no more than 16.6%, preferably no more than 16.1%, preferably no more than 15.9%, preferably no more than 15.7%, preferably no more than 15.5%, preferably no more than 14.1%, preferably no more than 13.4%); based on Table 10: the washed solid residue contains less than about 5% (e.g., less than 5%) P_2O_5, preferably less than 4%, preferably less than 3%, preferably less than 2%, preferably no more than 1.15%, preferably no more than 1.06%, preferably no more than 0.96%; the washed solid residue contains less than about 5% (e.g., less than 5%) P, preferably less than 4%, preferably less than 3%, preferably less than 2%, preferably less than 1%, preferably no more than 0.52%, preferably no more than 0.50%, preferably no more than 0.46%, preferably no more than 0.42%, preferably no more than 0.40%. The washed solid residue contains about 100% (e.g., 100%) of the carbon in the original untreated waste; based on Table 10: the washed solid residue contains about 34% to about 42% carbon. The washed solid residue contains at least about 60% (e.g., at least 60%) of the nitrogen in the original untreated waste, preferably at least 66.1%, preferably at least 68.7%, preferably at least 74.9%, preferably at least 77.4%, preferably at least 78.3%, preferably at least 80.4%, preferably at least 81.1%, preferably at least 82.5%, preferably at least 83.4%, preferably at least 83.7%, preferably at least 85.4%, preferably at least 85.8%, preferably at least 86.7%, preferably at least 87.0%, preferably at least 88.2%, preferably at least 88.6%, preferably at least 89.9%, preferably at least 97.8%; based on Table 10: the washed solid residue contains less than about 5% (e.g., less than 5%) N, preferably less than 4%, preferably less than 3%, preferably no more than 2.16%, preferably no more than 2.10%, preferably no more than 1.96%, preferably no more than 1.82%, preferably no more than 1.76%. The washed solid residue contains no more than about 20% (e.g., no more than 20%) of the potassium in the original untreated waste, preferably no more than about 15% (e.g., no more than 15%), preferably no more than 14.6%, preferably no more than 14.4%, preferably no more than 13.8%, preferably no more than 13.3%, preferably no more than 13.0%; based on Table 10: the washed solid residue contains, less than about 5% (e.g., less than 5%) K, preferably less than 4%, preferably less than 3%, preferably less than 2%, preferably less than 1%, preferably no more than 0.50%, preferably no more than 0.48%, preferably no more than 0.44%, preferably no more than 0.39%, preferably no more than 0.38%. The washed solid residue contains no more than about 20% (e.g., no more than 20%) of the K_2O in the original untreated waste, preferably no more than about 15% (e.g., no more than 15%), preferably 14.7%, preferably no more than 14.5%, preferably no more than 13.8%, preferably no more than 13.2%, preferably no more than 12.9%; based on Table 10: the washed solid residue contains less than about 5% (e.g., less than 5%) K_2O, preferably less than 4%, preferably less than 3%, preferably less than 2%, preferably less than 1%, preferably no more than 0.59%, preferably no more than 0.57%, preferably no more than 0.52%, preferably no more than 0.47, preferably no more than 0.45%. This first step produces a liquid extract containing low suspended solids of less than about 3.5 g/L (e.g., less than 3.5 g/L; preferably less than about 3 g/L (e.g., less than 3 g/L)) and extracted soluble phosphorus (e.g., generally more than about 600 mg/L (more than 600 mg/L); preferably at least about 613 mg/L (at least 613 mg/L)); the liquid extract contains the phosphorus in the original material less the phosphorus in the washed solid residue. The washed solid manure residue is subsequently separated from the liquid extract and dewatered; unnecessary carbon and nitrogen oxidation and digestion are prevented by dewatering the residue.

The liquid extract is transferred to a second vessel where phosphorus is recovered in steps **2** and **3** (FIG. **2**). In step **1**, organically bound phosphorus is first converted to soluble-P by selective, hydrolysis reactions using mineral or organic acids. This process hydrolyzes organic phosphorus-containing compounds rapidly in order to extract the phosphorus; more phosphorus goes into solution than nitrogen under the acidic conditions utilized (e.g., 3>pH<5). This step also releases phosphorus from insoluble inorganic phosphate complexes. Therefore, for purposes of the present invention, selective hydrolysis is defined as any hydrolysis reaction which allows extraction of at least twice as much of phosphorus (in percentage terms) than nitrogen (Table 2 and FIG. **8**; see also Table 11). The selective hydrolysis and solubilization of phosphorus compounds is obtained by using organic acids such as citric, oxalic, malic, etc., mineral acids such as hydrochloric or sulfuric, for example, or a mixture of both mineral and organic acids or their precursors. The acids used in the process can be produced using different acid precursors that consist of organic substrate including animal waste transformed into acid compounds by bacterial, yeast, or fungal microorganisms for example, such as *Thiobacillus* sp., *Arthrobacter paraffineus, Candida* sp., and *Aspergillus niger*. Furthermore, any mineral acid or organic acid can be used in the selective hydrolysis step. Although the preferred acids for quick wash are those acids which do not add phosphorus or nitrogen, the use of acids such as nitric, ethyldiamintetracetic, sulfuric or phosphoric may be used during the process of the present invention to fortify the final extracted product with nitrogen, sulfur or phosphorus.

In step **2**, phosphorus is precipitated by addition of an alkaline earth base, such as for example lime (calcium hydroxide), magnesium hydroxide, calcium oxide, magnesium oxide, and mixtures thereof, to the liquid extract to a pH range of not less than about 8.0 (e.g., not less than 8.0) to not more than about 11.0 (e.g., not more than 11.0), preferably not less than about 9.0 (e.g., not less than 9.0) to not more than about 11.0 (e.g., not more than 11.0), to form an alkaline earth metal-containing phosphorus compound.

In step **3**, an organic flocculant is added into the second vessel to enhance precipitation and phosphorus grade of the precipitated product (steps **2** and **3** may occur simultaneously or sequentially; preferably sequentially). After a settling period, of less than about 30 minutes (e.g. less than 30 minutes) the precipitated phosphorus-rich solid is removed from the bottom of the second vessel while the supernatant liquid is recycled back into the quick wash system or land applied. The flocculant is a poly-electrolyte and is added at less than about 10 ppm (e.g., less than 10 ppm) to increase the yield of filtering. One example of a filtering device is a 0.84 m×0.84 m×0.13 sieve box with a 0.6 wire mesh bottom and a commercial polypropylene non-woven fabric (Dupont E.I. de Nemours, N.J.). One of ordinary skill in the art could readily determine any other filter that would be useable in the process of the present specification. The present invention produces a phosphorus fertilizer material (i.e., the precipitated phosphorus-rich solid after steps **2** and **3**) that contains: (1) greater than about 10% P_2O_5 (e.g., more than 10%; preferably 10.91% or more; preferably 10.95% or more; preferably 11.06% or more; preferably 1.1.16 or more %; preferably at least 11.21%); (2) greater than about 4% P (e.g., more than 4%; preferably 4.57% or more; preferably 4.61% or more; preferably 4.70% or more; preferably 4.79% or more; preferably at least 4.83%); (3) less than about 5% N (e.g., less than 5% N; preferably less than 4.5%; preferably less than about 4% (e.g., less than 4%); preferably 3.64% or less; preferably 3.61% or less; preferably 3.54% or less; preferably 3.47% or less; preferably no more than 3.44%); (4) less than about 5% K as K_2O (e.g., less than 5%; preferably less than about 4.5% (e.g., less than 4.5%); preferably less than about 4.0% (e.g., less than 4.0%); preferably less than about 3.5% (e.g., less than 3.5%); preferably less than about 3.0% (e.g., less than 3.0%); preferably less than about 2.5% (e.g., less than 2.5%); preferably less than about 2.0% (e.g., less than 2.0%); preferably less than about 1.5% (e.g., less than 1.5%); preferably less than 1.188%; preferably less than 1.164%; preferably less than 1.128%; preferably less than 1.08%; preferably no more than 1.068%); (5) less than about 4% K (e.g., less than 4%; preferably less than about 3.5% (e.g., less than 3.5%); preferably less than about 3.0% (e.g., less than 3.0%); preferably less than about 2.5% (e.g., less than 2.5%); preferably less than about 2.0% (e.g., less than 2.0%); preferably less than about 1.5% (e.g., less than 1.5%); preferably less than 0.99%; preferably less than 0.97%; preferably less than 0.94%; preferably less than 0.90%; preferably no more than 0.89%); (6) less than about 40% C (e.g., less than 40%; preferably less than 36.26%; preferably less than 36.11%; preferably less than 35.90%; preferably less than 35.60%; preferably no more than 35.54%); (7) more than about 10% Ca (e.g., more than 10%; preferably more than 10.27%, preferably more than 10.54%; preferably more than 11.22%; most preferably more than 11.89, preferably at least 12.117%); (8) less than about 2% Na (e.g., less than 2%; preferably less than about 1% (e.g., less than 1%); preferably less than 0.34%, preferably less than 0.33%, preferably less than 0.31%, preferably less than 0.29%, preferably no more than 0.28%); (9) less than about 2% Mg (e.g., less than 2%; preferably less than about 1.5% (e.g., less than 1.5%); preferably less than 0.70%, preferably less than 0.69%, preferably no more than 0.68%). Furthermore, this phosphorus product is only about 15% (e.g., 15%) of the initial volume of the poultry litter.

In addition, the remaining washed solid residue has a more balanced nitrogen to phosphorus ratio that is environmentally safe for land application and use by crops. As an alternative, washed litter residue can be digested for methane production or utilized as bedding especially in areas where bedding material is in short supply.

The process is generally conducted at an ambient temperature greater than about 5° C. and less than about 50° C. (e.g., greater than 5° C. and less than 50° C.; preferably greater than about 10° C. and less than about 45° C. (e.g., greater than 10° C. and less than 45° C.); greater than about 10° C. and less than about 40° C. (e.g., greater than 10° C. and less than 40° C.); greater than about 10° C. and less than about 35° C. (e.g., greater than 10° C. and less than 35° C.); greater than about 10° C. and less than about 30° C. (e.g., greater than 10° C. and less than 30° C.); in other words the process does not require heat and the process is not exothermic.

Poultry litter used in the following experiments consisted of wood chip bedding plus manure accumulated during bird production. Broiler litter for Examples 1 and 2 below was collected from a 27,400-bird broiler house in Sumter County, South Carolina. At the time of sampling, the litter was being used by the fifth consecutive flock (approximately 6.5 flocks per year). Two composite litter samples were taken in approximately two 12-meter transects covering the width of the house. Composite samples were placed in 20-liter plastic sealed containers and stored in the freezer until preparation for laboratory experiments.

Broiler litter used for field prototype experiments was collected from a 25,000-bird broiler house. At the time of sampling, the house was empty and between the second and third flock (5 flocks per year). Two large composite litter samples were taken in two transects along the house, in its center section between water lines, and placed in 160-L containers. The containers were sealed, transported and placed in cold storage of about <2 degrees centigrade. Two 15.2 kg samples were prepared for field prototype experiments. In average, the two samples contained approximately 28.6 (±0.6) % moisture, approximately 26.2 (±0.04) mg/kg TKN, and approximately 15.5 (±3.8) mg/kg total phosphorus (Table 1 below). Prior to field prototype tests, broiler litter was ground and homogenized using a chipper (Yard Machines 5HP model, MTD LLC, Cleveland, Ohio). Average particle size distribution of chipped poultry litter is shown in FIG. **4**.

Analysis of supernatant liquid was, performed according to Standard Methods for the Examination of Water and Wastewater (APHA, 0.1998, Standard Methods for the Examination of Water and Wastewater, 20th edition. Washington, D.C., American Public Health Association, American Water Works Association, and Water Environment Federation). Total phosphorus and Total K nitrogen were determined in liquid and solid samples using the automated ascorbic acid method (Standard Method 4500-P F) and the phenate method (Standard Method 4500-NH_3 G) adapted to digested extracts (Technicon Instruments Corp., 1977, Individual/simultaneous determination of nitrogen and/or phosphorus in BD acid digests (dialyzer), Industrial method 337-74W/B, Tarrytown, N.Y.), respectively. Total nitrogen is the sum of total K nitrogen plus nitrate-nitrogen. Nitrate nitrogen was also determined using Standard Method 4500-NO_3^-F; it represented less than about 3% of total nitrogen. The pH of the supernatant liquid was measured electrometrically using a combination pH electrode. Total suspended solids (TSS) were determined by retaining solids on a glass-fiber filter (Whatman grade 934AH, Whatman Inc., Clifton, N.J.) dried to approximately 105° C. (Standard Method 2540 D). Moisture in solids was determined using a microwave moisture analyzer (Omnimark Instrument Corp., Tempe, Ariz.). Elemental analysis of recovered phosphorus-rich solids for total carbon and nitrogen was done by dry combustion (Leco Corp., St. Joseph, Mich.) and for phosphorus, calcium, magnesium, potassium, and sodium by inductively coupled plasma (ICP) from nitric acid plus H_2O_2 digested extract (Peters, J., et al., Recommended methods of manure analysis, University of Wisconsin-Extension Publication A3769, 2003).

The process can be carried out in batch mode using a single vessel to do the mixing and settling in step **1** or steps **2** and **3** (FIG. **2**) or adapted for continuous operation using two separate vessels, to do the mixing first and then the settling (FIG. **3**).

The following examples are intended only to further illustrate the invention and are not intended to limit the scope of the invention as defined by the claims. Unless defined otherwise, all technical and scientific terms used herein have the same meaning as commonly understood by one of ordinary skill in the art to which the invention belongs. The term "about" is defined as plus or minus ten percent; for example, about 100° F. means 90° F. to 110° F. Although any methods and materials similar or equivalent to those described herein can be used in the practice or testing of the present invention, the preferred methods and materials are now described. Poultry litter is used as a model for solid animal or poultry manure to demonstrate the invention.

Example 1

Organic and inorganic acids were tested for their potential to extract phosphorus from poultry litter. Poultry litter samples were prepared by grinding and passing through a sieve of about 5.8 mm. Aqueous solutions of acetic, citric, and hydrochloric acids were added to about 2.00 grams of ground and sieved poultry litter samples in a ratio of about 1:2.5 w/v at concentration levels of about 0, 2.5, 5, 10, 20, 40, and 80 mmoles/liter. The solutions and litter were mixed in a reciprocating shaker at about 135 oscillations/minute at ambient temperature of about 23° C. for approximately 1 hour. Subsequently solids and liquid were separated by centrifuge at about 2000×g for about 5 minutes. The liquid supernatant was decanted and analyzed for pH, total phosphorus (TP), and total Kjeldahl nitrogen (TKN). Solids were air dried at about 40° C. and analyzed for total Kjeldahl and total phosphorus. The experiment was repeated and the treatment control consisted of extraction with distilled water. Treatment efficiency of the various acid treatments was established by comparison of phosphorus extraction relative to initial phosphorus content in untreated poultry litter (Technicon Instruments Corp., 1977), respectively. The ground and sieved poultry litter contained approximately 17.1±0.2% moisture, approximately 35.10±0.02 mg/kg of total kjeldahl nitrogen, and approximately 19.2±0.2 mg/kg total phosphorus (Table 1).

Both mineral and organic acids extracted phosphorus from poultry litter (FIG. **5**). During extraction, a significant portion of total phosphorus in poultry litter was released from the manure solids. Total phosphorus extraction rates increased with increasing acid concentrations. At approximately 40 mmol/L concentration of acid, about 81% of the initial total phosphorus content in broiler litter was extracted. In contrast, the distilled water (control) extracted only about 20%. In addition to the concentration of acid, the type of acid made a difference. Citric acid was surprisingly more efficient at extracting phosphorus than HCl or acetic acid at similar molar applications (approximately 2.5 to 40 mmol/L). High extraction efficiencies (>70%) were also possible with HCl, but required molar rates that were double (approximately 80 mmol/L).

Even though phosphorus extraction surprisingly increased from approximately 17% to approximately 81% with increased citric acid treatment in the range of approximately 0 to approximately 40 mmol/L, nitrogen extraction was surprisingly not greatly affected (Table 2). Nitrogen contained in litter was extracted much less efficiently than phosphorus. For instance, about 81% of initial total phosphorus in litter was extracted in treatment 5 at about pH 3.8 (approximately 40 mM citric acid) but only about 27% of nitrogen was extracted (Table 2). Thus, the litter wash residue surprisingly resulted in a nitrogen:phosphorus ratio of approximately 9.8. This is surprisingly about 5-fold higher than the nitrogen:phosphorus ratio of the untreated litter (nitrogen:phosphorus ratio of about 2.1). Furthermore, this is in the range of nitrogen:phosphorus ratio required for balanced fertilization of crops for both nitrogen and phosphorus.

The percentage of phosphorus extracted from solids increased linearly with decreasing pH ($y=-11x+107$, $R^2=0.87$, $n=19$, $P<0.0001$, FIGS. **6** and **9**). Using the equation one can estimate a range of percent phosphorus extracted; for example where pH=3.1, $y=(-11)(3.1)+107=73\%$, or where pH=5.0, $y=(-11)(5.0)+107=52\%$. The pH required to extract % P from solids decreased linearly with increasing phosphorus extraction ($x=(y-107)/-0.11$, $R^2=0.87$, $n=19$, $P<0.0001$, FIGS. **6** and **9**). Using the equation one can estimate the pH required to extract a percentage of phosphorus extracted from solids; for example, where % P extracted from solids is 55%, $x=(55\%-107)/-11=4.73$, or where % P extracted from solids is 60%, $x=(60\%-107)/-11=4.27$. Although the quick wash process consistently extracted more than about 50% of total phosphorus when the pH of the acid solution-broiler litter mixture was lower than 5 units, similar percentages of phosphorus from broiler litter were extracted at different acid concentrations (FIG. **5**). Thus, the amount of acid added in the process to extract a specific amount of phosphorus can be controlled by setting a specific end point pH using a pH controller. The process includes any pH range along the curve shown in FIG. **6** or **9** (or defined by the above equation); for example pH range of 3.1 to 4.0, or 3.2 to 4.1, or 3.3 to 4.2, etc. The process also includes any % P extracted from solids range along the curve shown in FIG. **6** or **9** (or defined by the above equation); for example 50-60%, or 51-61%, or 52-62%, etc.

Although other mineral and organic acids can be used for the present invention, such as for example, sulfuric, malic, oxalic, phosphoric, nitric ethyldiamintetracetic, etc., the preferred acids are those that do not add phosphorus or nitrogen during the process of extracting phosphorus. Therefore, the use of acids such as phosphoric nitric, or ethyldiamintetracetic is not recommended.

The treated litter (washed solids; washed solid residue) left at the end of the process can now be used for land application at rates based on the nitrogen crop requirements without accumulation of excess phosphorus in the soil. Using data from Edwards and Daniel (1992), a nitrogen:phosphorus ratio of 5.2:1 would be needed to match Kentucky bluegrass specific nutrient uptake needs, which can be delivered with a phosphorus extraction at pH 4.5 (nitrogen:phosphorus=5.5). Higher nitrogen:phosphorus ratios needed for cotton (6.2:1), corn (7.5:1) or wheat (10.7:1) can be obtained at pH<4.5 (Table 2).

TABLE 1

Broiler litter characteristics.

Experiment	Moisture %	Total Phosphorus g/kg	Total Nitrogen g/kg	Nitrogen: Phosphorus Ratio
Examples 1 and 2 Sample 1	17.6	19.4	34.6	1.8
Examples 1 and 2 Sample 2	16.6	19.1	35.5	1.9
Mean[1]	17.1(0.2)	19.2(0.2)	35.1(0.02)	1.9
Field Prototype Sample 1 (Run 1)	29.3	12.8	25.9	2.0
Field Prototype Sample 2 (Run 2)	27.9	18.2	26.5	1.5
Mean	28.6(0.6)	15.2(3.8)	26.2(0.04)	1.7

[1]Mean value (standard deviation)

TABLE 2

Effect of citric acid treatment on pH of the extraction solution-solids mixture, total P and N extracted, and N:P ratio in solid residue left after washing poultry litter.

Treatment	pH mixture	Acid mmol/L	Total P extracted[1] g/kg litter	Total P extracted[1] %	Total N extracted[2] g/kg litter	Total N extracted[2] %	% Total P Extracted/ % Total N Extracted	N:P Ratio in Washed Litter
0	8.2	0.0	3.3[3]	17	10.2	29.1	0.58	1.2
1	7.1	2.5	5.5	29	11.6	33.1	0.88	1.3
2	6.4	5	6.9	36	11.1	31.7	1.13	1.4
3	5.4	10	11	55	11.4	32.5	1.69	2.5
4	4.5	20	13	68	9.6	27.4	2.48	5.5
5	3.8	40	16	81	9.4	26.8	3.02	9.8
6	3.1	80	13	67	7.7	22.0	3.05	11.1

[1]Total P extracted = P extraction relative to initial P content in litter (19.2 g/kg).
[2]Total N extracted = TKN extraction relative to initial TKN content in litter (35.1 g/kg).
[3]Data are the means of two replicates.

Example 2

To demonstrate the removal and recovery of phosphorus from the liquid extract, which includes steps **2** and **3** of the process, generated by litter washing (step **1**) (FIG. **2**), approximately 64 grams of poultry litter, as prepared in Example 1, was mixed with approximately 1.6 liters of 20 mM citric acid solution in a ratio of 1.25 w/v and stirred for about one hour with a magnetic stirrer. After the mixture settled for about 20 minutes, the liquid extract was separated from washed litter by decantation and transferred to separate laboratory vessels. To one half of the vessels, hydrated lime ($Ca(OH)_2$) was added, to the other half, lime and flocculant was added. Hydrated lime in water was added in various amounts until the pH of the mixed liquid reached set points of approximately 6, 7, 8, 9, 10, or 11 units (Treatments 1-6, respectively); a control treatment with no lime addition was included (Treatment 0). The recovery of phosphorus was enhanced by adding an organic flocculant to clump the fine particles of the phosphorus precipitate (Step **3**). The organic flocculant was an anionic polymer (polyacrylaminde) Magnafloc 120 L with an approximately 34% mole charge and approximately 50% active ingredient (CIBA Specialty Chemicals Water Treatment, Inc., Suffolk, Va.). This flocculant was added at a rate of approximately 7.0 mg/L (active ingredient). For both lime only and lime plus flocculant addition, the liquid supernatant was decanted and analyzed for pH, total phosphorus, and total Khejdahl nitrogen. Solids were air dried at about 40° C. and analyzed for total Khejdahl nitrogen. Treatment efficiency of the various lime and flocculant treatments was expressed as percentage of phosphorus extraction relative to initial phosphorus content. All tests were conducted in duplicate.

A 20 mmol/L citric acid extract solution was selected for step **1** to further recovery of phosphorus with hydrated lime. This liquid extract contained a high total phosphorus concentration of about 600 mg/L at about pH 4.7 (Table 3, Treatment 0) and low total suspended solids (approximately 2.1 g/L) after liquid-solid separation by decantation. In step **2**, total phosphorus was removed from solution by precipitating soluble phosphorus compounds under alkaline conditions. Addition of hydrated lime decreased total phosphorus until a pH of approximately 8.0 units was obtained (Table 3).

Subsequent addition of a flocculant improved the percentage of total phosphorus removed at pH higher than 8.0 (Table 4). A small amount of an organic flocculant was added at a rate of about 7 mg/L (active ingredient) to all treatments to enhance thickening and phosphorus grade of the precipitated product (Step **3**). Results in Table 4 surprisingly show an increase of the amount of phosphorus extracted and higher phosphorus grade of the precipitate by addition of hydrated lime followed by flocculant enhancement. The highest phosphorus recovery rate and grade in the precipitate (about 18.8% P_2O_5) was surprisingly obtained when the pH reached a value of about 10.0 units.

The enhancing effect of organic flocculant addition on total phosphorus content of the precipitate is summarized in Table 5 at three hydrated lime levels (pH approximately 8, 9, and 10) with and without application of polymer after citric acid (approximately 20-mM) extraction. From these results, surprisingly more than 65% of total phosphorus in poultry litter can be recovered by the addition of hydrated lime and small amounts of organic flocculant (Steps **2** and **3**).

TABLE 3

Quick wash process (Step 2), hydrated lime application for recovery of extracted soluble phosphorus from broiler litter. Data show total phosphorus concentration in liquid extract and corresponding percentage of total phosphorus removed by increasing pH with hydrated lime after phosphorus extraction (Step 1) with citric acid solution (1:25).

Treatment[1]	pH	$Ca(OH)_2$ applied g/L liquid	Total phosphorus mg/L	Total phosphorus removed from liquid extract[2] %
0	4.7	0.0[3]	613	0
1	6.0	1.4	381	39
2	7.0	2.0	299	51
3	8.0	2.6	215	65
4	9.0	3.1	251	59
5	10.0	3.7	303	51
6	11.0	4.1	236	62

[1]Treatment of the liquid was done by addition of hydrated lime (2% $Ca(OH)_2$ in water) to obtain a specific pH.
[2]Total P Removed = P recovered from liquid fraction relative to initial P concentration in liquid extract (613 mg/L).
[3]Data are the average of two replicates.

TABLE 4

Quick wash process (Steps 2 and 3), hydrated lime and flocculant application for recovery of extracted soluble phosphorus from broiler litter. Data show total phosphorus recovered per unit weight of broiler litter and phosphorus grade of the recovered phosphorus. Step 1 (P extraction), was carried out using citric acid solution (1:25 w/v).

Treatment[1]	pH	$Ca(OH)_2$ applied g/L liquid	$Ca(OH)_2$ applied g/kg litter	Total phosphorus recovered[2] g/kg litter	Total phosphorus recovered[2] %	Phosphorus grade in precipitate % P_2O_5
0	4.7	0.0[3]	0.0	0.5	2.8	1.4
1	6.0	1.4	36	6.5	33.6	14.9
2	7.0	2.0	50	8.1	42.3	11.9
3	8.0	2.6	65	11.7	61.0	17.6
4	9.0	3.1	78	13.0	67.5	17.2
5	10.0	3.7	93	13.9	72.5	18.8
6	11.0	4.1	104	13.5	70.4	14.4

[1]Treatment of the liquid was done by addition of hydrated lime (2% $Ca(OH)_2$ in water) to obtain a specific pH. An anionic polymer (polyacrylamide) was added at a rate of 7 mg/L (active ingredient) to all treatments to enhance precipitation.
[2]Total P recovered = P recovered from liquid fraction relative to initial P content in litter (19.2 g/kg).
[3]Data are the average of two replicates.

TABLE 5

Increased total phosphorus recovery in the-quick wash process using anionic polyacrylamide polymer application. Results are compared to total phosphorus recovered without polymer addition. For lime treatment, refer to table 4.

Lime Treatment	pH[1]	Total P Recovered[2] Without polymer g/kg litter	With polymer[3]	Recovery Increase with Polymer[4] %
3	8	10.0[4]	11.7	14.0
4	9	9.1	13.0	30.0
5	10	7.7	13.9	45.0

[1]Specific pH values obtained using hydrated lime (2% $Ca(OH)_2$ in water).
[2]Total P recovered = P removal from liquid fraction relative to initial P content in litter (19.2 g/kg).
[3]Anionic polyacrylamide, 37% charge, applied at a constant rate (7 mg/L active ingredient).
[4]Data are the average of two replicates.

Example 3

A field prototype system was developed to evaluate the process of the present invention to extract and recover phosphorus from poultry litter. The system included two connected reactor vessels (FIG. **3**). The extraction vessel **20** in the sequence was the phosphorus extraction reactor that consisted of an approximately 378-liter tank with a conical bottom **22**, a mixer **24**, and a pH controller (not shown). Once liquid reacted with solids, stirring was stopped to let solids settle. After settling of solids, the supernatant from tank **20** was pumped to a second vessel, a phosphorus removal tank **30**. The tank **30** in the sequence was the phosphorus recovery reactor that consisted of a second about 378 liter tank with a conical bottom **32**, mixer (not shown) and pH controller (not shown). The unit was completed with a smaller 115 gallon tank (not shown) with a mixer and pump used to stir and inject the hydrated lime solution into the tank **30**. Solid and liquid sampling was done in duplicate. Phosphorus extraction was performed by adding citric acid, approximately 10% w/w to a stirred mixture of approximately 15.2 kg of broiler litter, prepared as in Example 1, and water in a ratio of approximately 1:25 w/v inside the extraction reactor **20**. Addition of citric acid stopped when the pH of the mixture reached a set point of approximately 4.5. The extraction mixture was sampled about every 10 minutes during about a sixty minute, stirring period to determine the minimum stirring time required to reach a stable total phosphorus concentration in the extraction liquid; total phosphorus was determined in supernatant after about a 24 hour settling of unfiltered samples. The treated litter solids were removed from the bottom of the phosphorus extraction tank **20** after, about a twenty minute-settling period and further dewatered through a filter. The filter was a 0.84 m×0.84 m×0.13 sieve box with a 0.6 wire mesh bottom and a commercial polypropylene non-woven fabric (Dupont E.I. de Nemours, N.J.).

The supernatant from the phosphorus reactor was pumped into the phosphorus recovery reactor tank **30** and hydrated lime; about 10% $Ca(OH)_2$, was injected and mixed pH controller (not shown) stopped the lime injection when the pH of the mixed liquid reached a set point of about 9.0 in the first experiment or about 10 in the second experiment. Once the desired pH was reached, about 15 mg/L of anionic polyacrylamide, a flocculant, was injected and mixed to enhance phosphorus recovery. The precipitated solids were removed from the bottom, of the tank-after an approximately 30 minute settling period and dewatered through a filter as described above. The dried P-solids were analyzed for phosphorus, carbon, nitrogen, calcium, magnesium, potassium, and sodium content.

The prototype experiment was based on the acid and alkaline endpoint pH values that were determined in Examples 1 and 2 to extract, and recover more than about 65%, of total phosphorus from poultry litter. This procedure avoided an excessive chemical application. Consequently, in the prototype experiment, phosphorus was extracted from broiler litter using citric acid solution at approximately pH 4.5. The first tested component was the effect of stirring time on amount of phosphorus extracted from the slurry formed by mixing litter and extracting liquid (Step **1**). Extracted total phosphorus concentration remained stable (approximately 300-330 mg/L) at pH of approximately 4.5 with stirring time of about 20 minutes or more (FIG. **7**). From these results, it was confirmed that stirring time of about 20 to about 60 minutes (e.g., 20 to 60 minutes) is sufficient to obtain a stable total phosphorus extracted concentration during extraction process at a pH of approximately less than 5.0.

Phosphorus extraction performance of the prototype system under field conditions (Table 6) was surprisingly consistent with performance obtained in the laboratory (FIGS. **5** and **6**). Phosphorus-extraction efficiencies of approximately 65 to approximately 75% with respect to initial total phosphorus in broiler litter were surprisingly obtained with pH treatment of approximately 4.5 for both runs. As a result of phosphorus extraction, the average nitrogen:phosphorus ratio is better for crop utilization. As an alternative, the dried washed litter could be reused in the broiler house as bedding in geographic areas where bedding materials are in short supply or digested for methane production.

After settling in the phosphorus extraction tank, the supernatant liquid had a low total suspended solids (TSS) concentration of approximately <3.5 g/L, with respect to the total suspended solids concentration of the extraction slurry of approximately 28.7 g/L. This clarified liquid was pumped to the phosphorus recovery tank reactor and treated with hydrated lime and flocculant. This treatment surprisingly recovered approximately 92 to 89% of phosphorus extracted in step **1**. The complete process surprisingly recovered >60% of the initial total phosphorus in broiler litter; higher phosphorus recovery rates were obtained at a pH of approximately 10.0 (Table 6).

Before dewatering, mean initial moisture of the phosphorus sludge was about 96.3% (Table 7). After filtration, the sludge-had a mean moisture content of about 88.8%. The drying process was further accelerated by placing the phosphorus sludge in a greenhouse. The mean moisture content declined to about <10% in the subsequent thirteen days after filtration.

The prototype performance confirmed laboratory results that about >60% of the total phosphorus content of poultry litter can be surprisingly recovered using the quick wash process of the present invention (Table 6). The phosphorus grade of the product obtained in the prototype was lower (about 11.1% P_2O_5=4.85 mg P/100 grams×2.29) than the precipitate obtained in the laboratory (Tables 3 and 8). For example, on a dry matter basis, litter treated in the prototype had a lower mass and lower phosphorus concentration per volume of extracting solution.

In average, the precipitate contained relatively large amounts of phosphorus, carbon (C), nitrogen (N), and calcium (Ca), and small amounts of magnesium (Mg), potassium (K), and sodium (Na) (Table 8). Thus, the resulting molar ratio was about 1:7.0:1.6:1.4 for P:C:N:Ca.

An additional characteristic of the recovered phosphorus product was its surprisingly reduced bulk volume. The recovered phosphorus product (average dry bulk density of about 780 g/dm^3) surprisingly had about 17% of the initial volume of poultry litter. Therefore, the recovered phosphorus product can be transported more economically off the farm for use as a fertilizer material.

TABLE 6

Performance of field prototype to remove phosphorus from poultry litter using the quick wash process.

	Litter Before Wash			Extraction			Recovery		
Run	Total P g/kg litter	N:P Ratio[1]	N:P Ratio Washed Litter	pH	Total P[2] g/kg litter	%	pH	Total P[3] g/kg litter	%
1	12.8	2.0	4.4	4.5	8.3	65	9.0	7.7	60
2	18.2	1.5	4.1	4.5	13.7	75	10.0	12.2	67
Average	15.5	1.75	4.3	4.5	11	70	9.5	10.0	64

[1]Initial N content in litter: 2.59 and 2.65 g/kg for run 1 and 2, respectively.

[2]Total P extracted = P extracted relative to initial P content in litter before wash.

[3]Total P recovered = P recovered in precipitated solids relative to initial P content in litter after flocculant application.

TABLE 7

Percent moisture of phosphorus sludge before and after dewatering.

Sludge Phosphorus Dewatering	Percent Moisture g per 100 g Run 1	Run 2	Mean
Initial Moisture[1]	96.0	96.5	96.3
After Filtering[2]	89.0	88.6	88.8
Air Dried[3]	10.1	9.1	9.6

[1]Sludge obtained after decantation of liquid after flocculant addition (step 3)

[2]Dewatering for 24 hours after filtration through polypropylene non-woven filter fabric.

[3]Air dried for 13 days after dewatering in greenhouse, average temperature = 37 degrees C. and relative humidity = 54%

TABLE 8

Percent elemental composition of the solid precipitate produced from poultry litter using the quick wash process.[1]

	Percent Composition		
Constituent	Run 1	Run 2 g per 100 g	Mean
Phosphorus	4.61	4.79	4.70 (0.13)
P_2O_5[2]	11.16	10.95	11.06 (0.15)
Carbon	35.60	36.11	35.90 (0.36)
Nitrogen	3.61	3.47	3.54 (0.10)
Calcium	11.89	10.54	11.22 (0.95)
Magnesium	0.70	0.68	0.69 (0.01)
Potassium	0.90	0.97	0.94 (0.05)
K_2O[3]	1.08	1.16	1.12 (0.06)
Sodium	0.29	0.33	0.31 (0.03)

[1]Data for run 1 and run 2 obtained at pH 9 and 10, respectively (table 6) expressed as oven dry values. Values in parenthesis are standard deviations.

[2]Phosphorus grade expressed as P_2O_5 = % P × 2.29.

[3]Potassium grade expressed as K_2O = % K × 1.20

Example 4

This Example demonstrated that the manure wash treatment was also surprisingly effective to remove P from other animal manure besides poultry litter. A 64-g hog manure sample was mixed with about 1.6 L of 10-mM citric acid solution and stirred for approximately one hour. Similar to Examples 1 and 2 above, after the manure-liquid extract mixture settled for about 20 minutes, the liquid extract was separated from the washed litter by decantation and transferred to separate laboratory vessels. Hydrated lime was added to the vessels in various amounts until the pH of the mixed liquid reached set points of about 6, 7, 8, 9, 10 and 11 units (Treatments 1-6, respectively); the test included a control (treatment 0) with no lime addition. The recovery of P was enhanced (step **3**) by adding the same flocculant as in experiment 2 (7.0 mg L^{-1} active ingredient) to all six lime treatments and control. Liquid supernatant was decanted and analyzed for pH, TP, and TKN; solids were air dried at 40° C. and analyzed for TKN and TP. The tests were conducted in duplicate.

Table 9 shows experimental data supporting that the quick wash process can be surprisingly used for swine manure treatment and other fresh animal manures. In step **1**, phosphorus from fresh manure was extracted at pH 4.5 when mixed with 10-mM citric acid (Table 9, treatment 0). Results in Table 9 show a surprising increase of the amount of phosphorus recovered by addition of hydrated lime (step **2**) and organic flocculant (step **3**). The highest phosphorus recovery rate (6.4 g/kg manure) was surprisingly obtained when the pH reached a value between 9.0 and 10.0 units. Thus, about 90% of total phosphorus in swine manure can be surprisingly recovered by the addition of hydrated lime and small amount of organic flocculant. From this example, we concluded that the quick wash treatment can be used for P extraction and recovery from animal manures other than poultry litter.

TABLE 9

Quick wash process (steps 2 and 3), hydrated lime and flocculant application for recovery of phosphorus from swine manure after phosphorus extraction (step 1) with citric acid solution. Data show phosphorus recovered fresh swine manure by increasing pH with hydrated lime and organic flocculant addition.

		Total P Recovered	
Lime Treatment	pH[1]	g/kg manure	%
0	4.5	0.0	0
1	6.0	2.2	31
2	7.0	4.7	66
3	8.0	6.2	87
4	9.0	6.4	90
5	10.0	6.4	90
6	11.0	6.3	89

[1]Specific pH treatment was obtained by addition of hydrated lime (2% $Ca(OH)_2$ in water). An anionic polymer (polyacrylamide) was added at a rate of 7 mg/L (active ingredient) to all treatments to enhance precipitation.

[2]% Total P recovered = P recovered relative to initial P content in fresh swine manure (7.1 g/kg). Solids content of fresh manure = 30%.

[3]Data are the average of two replicates.

TABLE 10

(A) Percent composition of raw poultry litter; (B) percent composition of washed solid residue; and (C) percent of each constituent remaining in the washed solid residue with respect to initial content in raw poultry litter [C = (B/A) * 100].

	Percent Composition[1]								
	(A) Raw Poultry Litter			(B) Washed Solid Residue[2]			(C) Remaining in Washed Solid Residue		
Constituent	Run 1	Run 2	Mean[3] g per 100 g	Run 1	Run 2	Mean	Run 1	Run 2 %	Mean
Phosphorus	1.28	1.82	1.55 (0.38)	0.50	0.42	0.46 (0.06)	39.1	23.1	31.1 (11.3)
P_2O_5[4]	2.93	4.17	3.55 (0.88)	1.15	0.96	1.06 (0.13)	39.1	23.1	31.1 (11.3)
Carbon	39.30	35.36	37.33 (2.79)	41.00	35.49	38.25 (3.90)	104.3	100.4	102.3 (2.8)
Nitrogen	2.59	2.65	2.62 (0.04)	2.10	1.82	1.96 (0.20)	81.1	68.7	74.9 (8.8)
Calcium	1.93	1.87	1.90 (0.04)	1.50	0.32	0.91 0.83)	77.7	17.1	47.4 (42.9)
Magnesium	0.49	0.71	0.60 (0.16)	0.13	0.13	0.13 (0.00)	26.5	18.3	22.4 (5.8)
Potassium	2.71	3.61	3.16 (0.64)	0.39	0.48	0.44 (0.06)	14.4	13.3	13.8 (0.8)
K_2O[5]	3.25	4.33	3.79 (0.76)	0.47	0.57	0.52 (0.07)	14.5	13.2	13.8 (0.9)
Sodium	0.87	1.14	1.01 (0.19)	0.17	0.20	0.19 (0.02)	19.5	17.5	18.5 (1.4)
Sulfur	0.65	0.8	0.73 (0.11)	0.27	0.25	0.26 (0.01)	41.5	31.3	36.4 (7.3)

[1]Data expressed as oven dry values.
[2]Solid residue obtained after washing raw poultry litter using the quick wash process at pH = 4.5.
[3]Values in parenthesis are standard deviations.
[4]Phosphorus grade expressed as P_2O_5 = % Phosphorus × 2.29.
[5]Potassium grade expressed as K_2O = % Potassium × 1.20.

TABLE 11

(A) Percent composition of raw poultry litter; (B) percent composition of washed solid residue; and (C) percent of phosphorus and nitrogen gv
Percent Composition[1]

			(A) Raw Poultry Litter			(B) Washed Solid Residue[2]			(C) Remaining in Washed Solid Residue		
Acid mmoL/L	pH Mixture	Constituent	Run 1	Run 2	Mean[3] g/100 g	Run 1	Run 2	Mean	Run 1	Run 2 %	Mean
0	8.2 (0.1)	Phosphorus	1.93	1.91	1.92 (0.01)	1.99	1.61	1.80 (0.27)	103.1	84.3	93.7 (13.3)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.55	1.95	2.25 (0.42)	73.7	54.9	64.3 (13.3)
2.5	7.1 (0:1)	Phosphorus	1.93	1.91	1.92 (0.01)	1.61	1.60	1.61 (0.01)	83.4	83.8	83.6 (0.2)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.11	2.16	2.14 (0.04)	61.0	60.8	60.9 (0.1)
5	6.4 (0.1)	Phosphorus	1.93	1.91	1.92 (0.01)	1.58	1.54	1.56 (0.03)	81.9	80.6	81.2 (0.9)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.17	2.31	2.24 (0.10)	62.7	65.1	63.9 (1.7)
10	5.4 (0.1)	Phosphorus	1.93	1.91	1.92 (0.01)	1.08	0.84	0.96 (0.17)	56.0	44.0	50.0 (8.5)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.32	2.55	2.44 (0.16)	67.1	71.8	69.4 (3.4)
20	4.5 (0.1)	Phosphorus	1.93	1.91	1.92 (0.01)	0:50	0.51	0.51 (0.01)	25.9	26.7	26.3 (0.6)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.71	2:93	2.82 (0.16)	78.3	82.5	80.4 (3.0)
40	3.8 (0.0)	Phosphorus	1.93	1.91	1.92 (0.01)	0.34	0.27	0.31 (0.05)	17.6	14.1	15.9 (2.5)
		Nitrogen	3.46	3.55	3.51 (0.06)	2.97	3.13	3.05 (0.11)	85.8	88.2	87.0 (1.6)
80	3.1 (0.1)	Phosphorus	1.93	1.91	1.92 (0.01)	0.32	0.30	0.31 (0:01)	16.6	15.7	16.1 (0.6)
		Nitrogen	3.46	3.55	3.51 (0.06)	3.11	3.75	3.43 (0.45)	89.9	105.6	97.8 (11.1)

[1]Data expressed as oven dry values.
[2]Solid residue obtained after washing raw poultry litter using the quick wash process at increasing concentrations of acid.
[3]Values in parenthesis are standard deviations.

Van Slyke (U.S. Pat. No. 6,916,426) discloses to extract ammonium, phosphorus and potassium from an animal waste slurry to form ureates of potassium and ammonium in crystalline form. Van Slyke further discloses that a substantial amount of potassium is extracted as ureates of potassium using flocculation before they degrade. Therefore, the solid material disclosed by Van Slyke contains substantial amounts of the potassium, nitrogen and phosphorus that was contained in the original animal waste sludge. Our fertilizer phosphorus product material is low in potassium (e.g., potassium content of less than 1% in Table 8) and low in nitrogen (e.g., nitrogen content of less than 4 in Table 8) because the acid treatment that we apply with our process would solubilize and destroy the potassium ureates, and the potassium remains in solution in the liquid extract. Our subsequent alkaline addition to the liquid extract reaching a pH between 8 and 11 does not recover significant amounts of the solubilized potassium that resulted from the destruction of the potassium ureates at acid pH. Therefore, our phosphorus fertilizer product contains low concentrations of potassium (e.g., Table 8, where total potassium is at most 0.97%). In contrast, our process does not involve ureates; there is also no flocculation of our initial animal wastes prior to or during our acid addition and/or lime addition.

Our process does not involve anaerobic digestion, composting, or direct combustion processes as disclosed in Kelleher et al., Bioresource Technology, 83: 27-36 (2002).

The solid poultry or animal wastes utilized by the process of the present invention are not incinerated before or during our process; in other words the present invention does not concern incinerated materials (e.g., incineration ash in JP 2000189927) which are devoid of oxidizable organic carbon and nitrogen and therefore are not solid organic wastes. The present process does not involve the addition of ammonium sulfate nor the production of aluminum phosphates.

The pre-treatment of solids using wet oxidation or fenton oxidation according to Kida (JP 20033200199) destroys the organic matter and solubilizes phosphorus. Nitrogen and organic substances in the sludge are removed by nitrification-denitrification. A wet oxidative pre-treatment to solid poultry or animal waste is contrary to our teaching of removing, phosphorus while conserving most of the carbon and nitrogen in the washed litter residue. In addition, our process does not use nitrification-denitrification to destroy carbon and nitrogen compounds form the solid poultry and animal waste. Furthermore, Kida is different from our process because Kida applies hydrochloric acid to a pH of less than 2 to remove phosphorus only from the ash of the deposit of undigested sludge already separated from supernatant liquid; in contrast, in our process the acid (e.g., at a pH lower than 5.0 and higher than 3.0) is first applied to the entire mass of poultry litter or animal waste to remove phosphorus prior to separation of the formed liquid extract and soluble phosphorus from the washed solid residue.

The process of Higashida (U.S. Pat. No. 5,378,257) is different from our process since it does not form (1) a liquid extract and soluble phosphorus and (2) a washed solid residue, and mixing said liquid extract with an alkaline earth base (e.g., calcium hydroxide). Higashida's process is unrelated to our process of removing phosphorus from solid organic wastes because Higashida does not separate solids from liquid. In Higashida, the Waste material is simply processed by adding nitric acid, crushing the material, adding quicklime, and drying it. In addition, Higashida's process destroys organic matter, which is contrary to the conservation of organic carbon and nitrogen in our process. Higashida teaches that waste matter (e.g., sewage) is oxidized with nitric acid incorporated in it (column 3, lines 30-31; column 4, lines 23-25). Therefore, Higashida's discloses a process that oxidizes, corrodes and destroys the organic matter of organic waste materials. In contrast, our process does not oxidize waste material since our process does not utilize a pH lower than 3.0 where unnecessary carbon and nitrogen digestion would occur which would destroy oxidizable organic carbon and nitrogen.

The method disclosed by Angell (U.S. Pat. No. 5,422,015) adds to solid waste a combination of acid plus a base that produces a strong exothermic reaction that elevates the temperature to at least 70° C. Angell's method has the purpose of disinfecting the waste but not the separation of phosphorus from the waste; phosphorus may even be added for binding heavy metals (see Claim **18**) to produce a material rich in nitrogen, phosphorus, and potassium (Claim **19**). Angell's method is different and unrelated to our process since Angell's method uses the combination of a strong acid and a strong base to create an exothermic reaction when in contact with the waste. In contrast, our process uses acid at such low concentrations that its reaction with the waste does not produce heat. Also, Angell's method requires temperatures of at least 70° C., while our process removes phosphorus from poultry litter and animal waste at ambient temperature of less than about 50° C. Furthermore, Angell's process is an exothermic process developed to kill pathogens but it does not separate phosphorus. Unlike Angell, our process does not involve adding acid and base at the same time and to the same material.

Cabello-Fuentes (U.S. Patent Application Publication No. 2004/0025553) discloses a process for treating sludge involving a first step of disinfecting the sludge by heating the sludge at a temperature of between about 50° C. to 100° C., preferably at 80° C., by means of a heat exchanger, and adding mineral acids in order to decrease the pH of the mass to 3.0 or less, thus guaranteeing that all pathogen microorganisms are destroyed. In contrast, our process does not involve heating solid wastes to a temperature of between about 50° C. to 100° C. and adding acids to decrease the pH of the mass to 3.0 or less.

Our process has, in part, the following advantages: It extracts and recovers phosphorus from organic solid wastes (poultry and animal manures) without the need for destroying the organic carbon as it is usually done during incineration or acid digestion of organic wastes. In addition to conserving the carbonaceous matter, our process conserves most of the nitrogen through a selective hydrolysis reaction. Thus it produces a material with elemental nitrogen to phosphorus ratio (N:P) of more than 4 that is optimal for use in crop production and helps to prevent the eutrophication of surface waters. This washed material contains most of the original carbon, most of the original nitrogen and a reduced amount of the original phosphorus which is more desirable for poultry and livestock producers—such as poultry farmers in the Chesapeake Bay area, Arkansas, and other areas with intensive poultry production—that have problems disposing poultry litter without contaminating soils and water resources with phosphorus. Therefore, poultry producers can use our process to wash the poultry waste to remove only the deleterious constituent for environmental compliance—the phosphorus—and maintaining in the washed residue the desirable constituents that benefit their crops within their operation consisting of the nitrogen with important savings in nitrogen fertilizer cost and the organic carbon which helps to build the organic matter in the soil and improves soil health, water retention, and resistance to drought. Another advantage of our process is that the extracted phosphorus is transferred into a concentrated calcium phosphate fertilizer product that can be easily transported away from areas with excess phosphorus due to intensive poultry and livestock production and be used effectively to substitute for mined phosphate fertilizer used in crop and horticulture production. Another advantage of our process is that heating is not needed and it can be optimally performed at ambient temperatures between 5° and 50° C.

All of the references cited herein, including U.S. Patents, are incorporated by reference in their entirety. Also incorporated by reference in their entirety are the following references: Bolan, N., et al., The management of phosphorus in poultry litter, Proc. New Zealand Poultry Industry Annual Conf., Oct. 7-9, 2008, Palmerson North, NZ, p. 1.56-168; Cantrell, K. B., et al., Plant nutrients and bioenergy via a new quick wash procedure for livestock manures, pp. 1238-1244, Proc. Beltwide Cotton Conf., San Antonio, Tex., Jan. 5-8, 2009; Donatello et al., "Production of Technical Grade Phosphoric Acid from Incinerator Sewage Sludge Ash", Waste Management, 30: 1634-1642 (2010); Jackson et al., "Trace Element Speciation in Poultry Litter", Journal of Environmental Quality, 32: 535-540 (2003); Moore, P. A., 2002, Best management practices for poultry manure utilization that enhance agricultural productivity and reduce pollution, p. 89-123, In J. L. Hatfield and B. A. Stewart (eds.), Animal waste utilization: Effective use of manure as a soil resource, Lewis Publishers/CRC Press, Boca Raton, Fla.; Szogi, A. A., et al., Fertilizer effectiveness of phosphorus recovered from broiler litter, Agron. J., 102(2): 723-727 (2010); Szogi, A. A., et al., Agronomic effectiveness of phosphorus materials recovered from manure, 13th RAMIRAN Int'l. Conf., Jun. 11-14, 2008, Albana, Bulgaria, pp. 52-56; Szogi, A. A., et al., Phosphorus recovery from poultry litter, Trans. ASABE, 51(5): 1727-1734 (2008); Szogi, A. A., and M. B. Vanotti, Prospects for phosphorus recovery from poultry litter, Bioresource Tech. 100:5461-5465 (2009); Szogi, A. A., et al., Effectiveness of recovered manure phosphorus as plant fertilizer, pp. 133-136, Proc. 1st Int'l. Symp. on Management of Animal Residuals, Mar. 11-13, 2009, Florianopolis, Brazil (SIGERA); Szogi, A., and M. Vanotti, Closing the loop for nutrients in livestock wastes: Phosphorus recovery from animal manure, 2008 ASA Annual Mtgs., Oct. 5-9, 2008, Houston, Tex.; Szogi, A. A., et al., Distribution of phosphorus in an Ultisol fertilized with recovered manure phosphates, p. 95-98, In: Proceedings 19th World Soil Congress, Aug. 1-6, 2010, Brisbane, Australia, Published in DVD; Bolan N., et al., The management of phosphorus in poultry litter, In: Proceedings 19th World Soil Congress, p. 317-320, Aug. 1-6, 2010, Brisbane, Australia, Published in DVD.

Thus, in view of the above, the present invention concerns (in part) the following:

A process for removing phosphorus from solid poultry or animal wastes comprising (or consisting essentially of or consisting of):

(a) mixing said solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein said washed solid residue has a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in said solid poultry or animal wastes,

(b) separating said liquid extract from said washed solid residue to form separated liquid extract and separated washed solid residue,

(c) mixing said separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0,

(d) mixing said liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid, and

(e) separating said precipitated phosphorus solids from said liquid to form separated phosphorus solids and separated liquid;

wherein said process is conducted at a temperature greater than about 5° C. and less than about 50° C., wherein said solid poultry or animal wastes are not pretreated prior to step (a), and wherein steps (c) and (d) are either sequential or simultaneous.

The above process, wherein said process comprises mixing said solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than 3.0. The above process, wherein said process comprises mixing said solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher or equal to 3.1.

The above process, wherein said process is conducted at ambient temperature.

The above process, wherein said process is conducted at a temperature greater than about 5° C. and less than about 45° C. The above process, wherein said process is conducted at a temperature greater than about 5° C. and less than 45° C. The above process, wherein said process is conducted at a temperature greater than about 5° C. and less than about 40° C.

The above process, wherein said solid poultry or animal wastes are not incinerated before or during said process.

The above process, wherein said-acid is selected from the group consisting of a mineral acid, an organic acid, and mixtures thereof and their precursors.

The above process, wherein said acid is selected from the group consisting of citric acid, oxalic acid, malic acid, hydrochloric acid, sulfuric acid, and mixtures thereof. The above process, wherein said acid is citric acid.

The above process, wherein said alkaline earth base is selected from the group consisting of calcium hydroxide, magnesium hydroxide, calcium oxide, magnesium oxide, and mixtures thereof.

The above process, wherein said steps (c) and (d) are sequential.

The above process, wherein said steps (c) and (d) are simultaneous.

The above process, wherein said washed solid residue contains the oxidizible organic carbon and nitrogen fraction that would be digested and oxidized if said solid poultry or animal wastes had instead been washed by mixing said-solid poultry or animal wastes with water and acid at a pH below about 3.0 or that would be lost if said solid poultry or animal wastes had instead been incinerated before being mixed with water and acid.

The above process, wherein said solid poultry or animal wastes are not pretreated (e.g., incinerated, or treated with flocculant) prior to said mixing said solid poultry or animal wastes with water and acid.

A material produced by a process comprising (or consisting essentially of or consisting of):

(a) mixing solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein said washed solid residue contains a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in said solid poultry or animal wastes,

(b) separating said liquid extract from said washed solid residue to form separated liquid extract and separated washed solid residue,

(c) mixing said separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0,

(d) mixing said liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid, and

(e) separating said precipitated phosphorus solids from said liquid to form separated phosphorus solids and separated liquid,

wherein said process is conducted at a temperature greater than about 5° C. and less than about 50° C., and wherein said solid poultry or animal wastes are not pretreated prior to step (a), and wherein steps (c) and (d) are either sequential or simultaneous;

wherein said material is said separated phosphorus solids and contains greater than about 10% P_2O_5, greater than about 10% Ca, less than about 5% N, and less than about 5% K as K_2O.

The above material, wherein said material contains greater than 10% P_2O_5.

The above material, wherein said material contains greater than 10% Ca.

The above material, wherein said material contains less than 4.5% N. The material, wherein said material contains less than 4% N.

The above material, wherein said material contains less than 4.5% K as K_2O. The material, wherein said material contains less than 4% K as K_2O. The material, wherein said material contains less than 3.5% K as K_2O. The material, wherein said material contains less than 3% K as K_2O. The material, wherein said material contains less than 2.5% K as K_2O. The material, wherein said material contains less than 2% K as K_2O.

The above material, wherein said washed solid residue contains the oxidizible organic carbon and nitrogen fraction that would be digested and oxidized if said solid poultry or animal wastes had instead been washed by mixing said solid poultry or animal wastes with water and acid at a pH below about 3.0 or that would be lost if said solid poultry or animal wastes had instead been incinerated before being mixed with water and acid.

A material produced by a process comprising (or consisting essentially of or consisting of):

(a) mixing solid poultry or animal wastes with water and acid at a pH lower that about 5.0 and higher than about 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, and

(b) separating said liquid extract from said washed solid residue to form separated liquid extract and separated washed solid residue;

wherein said material is said separated washed solid-residue and contains a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in said solid poultry or animal wastes.

The above material, wherein said material contains less than about 5% (e.g., less than 5%) P_2O_5. The material, wherein said material contains less than 5% P_2O_5. The material, wherein said material contains less than 4% P_2O_5. The material, wherein said material contains less than 3% P_2O_5. The material, wherein said material contains less than 2% P_2O_5.

The above material, wherein said washed solid residue contains the oxidizible organic carbon and nitrogen fraction that would be digested and oxidized if said solid poultry or animal wastes had instead been washed by mixing said solid poultry or animal wastes with water and acid at a pH below about 3.0 or that would be lost if said solid poultry or animal wastes had been instead incinerated before being mixed with water and acid.

Other embodiments of the invention will be apparent to those skilled in the art from a consideration of this specification or practice of the invention disclosed herein. It is intended that the specification and examples be considered as exemplary only, with the true scope and spirit of the invention being indicated by the following claims.

We claim:

1. A process for selectively removing phosphorus from solid poultry or animal wastes containing phosphorus, nitrogen and carbon, said process comprising:

(a) mixing said solid poultry or animal wastes with water and acid at a pH lower than about 5.0 and higher than 3.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein said washed solid residue has a N:P ratio of at least more than 4 expressed on an elemental basis and contains no more than about 40% of the total phosphorus in said solid poultry or animal wastes,

(b) separating said liquid extract from said washed solid residue to form separated liquid extract and separated washed solid residue,

(c) mixing said separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0,

(d) mixing said liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid, and

(e) separating said precipitated phosphorus solids from said liquid to form separated phosphorus solids and separated liquid;

wherein said process is conducted at a temperature greater than about 5° C. and less than about 50° C., wherein said solid poultry or animal wastes are not pretreated prior to step (a), wherein said solid poultry or animal wastes are not incinerated before or during said process, and wherein steps (c) and (d) are either sequential or simultaneous.

2. The process of claim **1**, wherein said process comprises mixing said solid poultry or animal wastes with water and acid at a pH of 3.8 to 4.5.

3. The process of claim **1**, wherein said process comprises mixing said solid poultry or animal wastes with water and acid at a pH lower than about 5.0 and higher or equal to 3.1.

4. The process of claim **1**, wherein said process is conducted at ambient temperature.

5. The process of claim **1**, wherein said process is conducted at a temperature greater than about 5° C. and less than about 45° C.

6. The process of claim **1**, wherein said process is conducted at a temperature greater than about 5° C. and less than 45° C.

7. The process of claim **1**, wherein said process is conducted at a temperature greater than about 5° C. and less than about 40° C.

8. The process of claim **1** wherein said acid is selected from the group consisting of a mineral acid, an organic acid, and mixtures thereof and their precursors.

9. The process of claim **1**, wherein said acid is selected from the group consisting of citric acid, oxalic acid, malic acid, hydrochloric acid, sulfuric acid, and mixtures thereof.

10. The process of claim **1**, wherein said acid is citric acid.

11. The process of claim **1**, wherein said alkaline earth base is selected from the group consisting of calcium hydroxide, magnesium hydroxide, calcium oxide, magnesium oxide, and mixtures thereof.

12. The process of claim **1**, wherein said steps (c) and (d) are sequential.

13. The process of claim **1**, wherein said steps (c) and (d) are simultaneous.

14. The process of claim **1**, wherein said washed solid residue contains the oxidizible organic carbon and nitrogen fraction that would be digested and oxidized if said solid poultry or animal wastes had been washed by mixing said solid poultry or animal wastes with water and acid at a pH below about 3.0 or that would be lost if said solid poultry or animal wastes had been incinerated before being mixed with water and acid.

15. The process of claim **1**, wherein said process comprises mixing said separated liquid extract with an alkaline earth base to a pH of 9.0 to 11.0.

16. The process of claim **1** wherein said acid does not add phosphorus or nitrogen during said process.

17. The process of claim **1**, wherein said process is conducted at a temperature of greater than about 10° C. and less than about 45° C.

18. The process of claim **1**, wherein said process is conducted at a temperature of greater than 10° C. and less than 45° C.

19. The process of claim **1**, wherein the percent phosphorus removed from said solid poultry or animal wastes is determined by the following formula: $y=-11x+107$ where x is pH and y is percent phosphorus extracted from said solid poultry or animal wastes.

20. A process for removing phosphorus from solid poultry or animal wastes comprising:

(a) mixing said solid poultry or animal wastes with water and acid at a pH between 3.1 and lower than about 5.0 to form (i) a liquid extract that contains suspended solids of about 3.5 g/L and soluble phosphorus and (ii) a washed solid residue, wherein said washed solid residue has a N:P ratio of 4.1 or more expressed on an elemental basis and contains no more than 40% of the total phosphorus in said solid poultry or animal wastes,

(b) separating said liquid extract from said washed solid residue to form separated liquid extract and separated washed solid residue,

(c) mixing said separated liquid extract with an alkaline earth base to a pH of about 8.0 to about 11.0,

(d) mixing said liquid extract with a flocculant to form (i) precipitated phosphorus solids with P_2O_5 content greater than about 10% and (ii) a liquid, and

(e) separating said precipitated phosphorus solids from said liquid to form separated phosphorus solids and separated liquid.

21. The process according to claim **20**, wherein said acid does not add phosphorus or nitrogen during said process.

* * * * *

ATTACHMENT 65

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From a PI's Perspective: How We Made a T2 Success

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Abstract – A team of scientists (Drs. Ariel A. Szogi, Matias B. Vanotti, and Patrick G. Hunt) from the Agricultural Research Service (ARS) - Coastal Plains, Soil, Water & Plant Research Center in Florence, S.C., invented a new treatment process, called "quick wash," to extract and recover phosphorus from poultry litter and animal manure solids.[1] This invention led us down the path to an award-winning technology transfer process.[2] As part of this process, a new and unexpected use for the technology emerged that was different from what we had imagined—to the extent that it helped build a new business model for our commercial partner. The purpose of this paper is to describe and illuminate, from the perspective of a principal investigator, what made the process successful and the lessons we learned along the way.

(Keywords: federal laboratory; Department of Agriculture; technology transfer; commercialization; licensing; marketing; technology; principal investigator; business models)

The Inspiration Behind the Effort

Environmental problems, the potential scarcity of phosphorus resources, and the value of recovered phosphorus products were the drivers to developing "quick wash," a method to recover the

[1] Ariel A. Szogi, Matias B. Vanotti, and Patrick G. Hunt. "United States Patent: 8,673,046 - Process for removing and recovering phosphorus from animal waste," March 18, 2014. See http://www.ars.usda.gov/SP2UserFiles/Place/60820500/Manuscripts/2014/pat8673046.pdf.

[2] FLC Excellence in Technology Transfer Award, 2015. See https://www.federallabs.org/index.php?tray=award_detail&cid=FLCawrd902&tid=1FLtop207.

phosphorus in livestock manure that consists of the rapid removal and recovery of phosphorus in solid form.

Nutrient pollution, caused by too much nitrogen and phosphorus in the environment, is one of America's most widespread, costly and challenging environmental problems, impacting many sectors of the U.S. economy that depend on clean water. The repeated application of untreated manures on soil can cause excess phosphorus accumulation in soils, and its subsequent loss through soil runoff or leaching can result in the pollution of surface waters. For this reason, widespread phosphorus pollution of waterways can occur in regions with concentrated livestock production. As result of phosphorus pollution, algal blooms in drinking water sources can drastically increase treatment costs and generate shortages in water supplies.

Phosphorus, an essential element for life on Earth, is a finite resource since mined phosphates are the main source in the production of phosphorus fertilizers. The demand for mined phosphorus is escalating worldwide due to both increasing food demand and human population. Inevitably, the future demand for mined phosphorus will exceed its supply capacity. Globally, the remaining phosphorus is found in various waste streams. These waste streams include large quantities of effluents rich in phosphorus from municipal, industrial, and livestock production sources. Therefore, phosphorus in these waste streams, if economically recovered, can contribute to a sustainable management of phosphorus resources.

The quick wash process mitigates both of these problems because phosphorus is selectively extracted from solid manure or municipal biosolids prior to land application. In layman's terms, the process takes manure—be it from a chicken or a human—and "washes" it, separating out the phosphorus, but leaving most of the nitrogen behind. This means that both the nitrogen-rich manure and the extracted phosphorus can be more judiciously applied, resulting in more effective fertilizer with less harmful phosphorus runoff as well as surplus phosphorus that can be reused in markets that need it, thus mitigating the shortage. The quick wash process selectively recovers more than 80 percent of the phosphorus from solid waste while leaving most of the nitrogen in the washed solid residue. Consequently, the washed solid residue has a more balanced nutrient composition that is safe for land application and is better balanced to match the specific nutrient needs of crops. Also, fertilizer tests of the recovered phosphate obtained with the quick wash method demonstrated that it is a good plant fertilizer. The concentrated phosphorus material contains more than 90% of its phosphorus in plant available form, which provides a recycled phosphorus source for use as a crop fertilizer on phosphorus-deficient croplands.

Our lab's role in finding a solution to these problems was crucial for more reasons than one. For our agency, it helps meet the objectives of addressing manure management problems that harm the environment and maximizing nutrient recovery.[3,4] For the private sector, we addressed a challenge deemed impossible—to develop a totally new technology capable of meeting multiple and strict environmental standards at a low cost.

Although many similar processes are not profitable, we quickly realized that this one would be profitable. Eventually, economic incentives such as government subsidies, environmental credits, and tipping fees have been considered as possible additional incentives for wide adoption and integration of this new method to reduce phosphorus pollution from animal production activities. Thus, phosphorus recovery in a concentrated, usable form would allow a more economical long-distance transfer of manure nutrients, while reducing agronomic nitrogen and phosphorus imbalances and the adverse effects of soil nutrient losses on the environment.

Lightning in a Bottle: Going Viral

To promote this technology, we made numerous presentations at professional scientific meetings, and two journal papers were published in peer-reviewed engineering journals.[5,6] However, the online ARS news releases were the most effective promotional technique, attracting worldwide attention.

An ARS News & Events story describing the quick wash process, entitled "Mining Manure for Phosphorus,"[7] was released online by the ARS Information Staff. The news release announced ARS's interest in finding business partners to move the product to market. We did not anticipate that when

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[3] USDA. National Program 214 AGRICULTURAL & INDUSTRIAL BYPRODUCTS ACCOMPLISHMENT REPORT 2009-Szogi, A.A., Vanotti, M.B., Hunt, P.G. 2015. Phosphorus recovery from pig manure solids prior to land application. Journal of Environmental Management. 157:1-72013. October 2013. http://www.ars.usda.gov/SP2UserFiles/Program/214/NP214AccomplishmentRpt2009-2013FINAL.pdf.

[4] Christina Woods. NP 214 Agricultural and Industrial Byproducts. Report, April 2012. http://www.ars.usda.gov/SP2UserFiles/Subsite/sciQualRev/NP214%20Panel%20Report.pdf.

[5] A.A. Szogi, M.B. Vanotti, and P.G. Hunt. "Phosphorus Recovery From Poultry Litter." American Society of Agricultural and Biological Engineers, Transactions of the ASABE Vol. 51(5): 1727-1734. See http://www.ars.usda.gov/SP2UserFiles/Place/60820000/Manuscripts/2008/Man785.pdf.

[6] Szogi, A.A., Vanotti, M.B., Hunt, P.G. 2015. Phosphorus recovery from pig manure solids prior to land application. Journal of Environmental Management. 157:1-7. doi:10.1016/j.jenvman.2015.04.010.

[7] Ann Perry, "Mining Manure for Phosphorus." ARS news story. February 29, 2008. http://www.ars.usda.gov/is/pr/2008/080229.htm

published in 2008, this technology would become viral. Yet, as a result of this news release we received nine requests for information from entrepreneurs in North America, Europe, and Asia.

The ARS patent application and related information for the quick wash process were provided after signing confidentiality agreements with each interested party. The preparation of these confidentiality agreements was facilitated and overseen by our Technology Transfer Coordinator. (ARS has a Technology Transfer Coordinator at each Area Office; in this case, the Coordinator covered all ARS research units within the southeastern United States).

Within two years after the ARS news release, we had four meetings with individual U.S. entrepreneurs interested in our invention; however, the only one that followed up with additional consultations and meetings with the goal of licensing our invention was Renewable Nutrients, LLC (RN), a small business located in North Carolina.

This interaction led to material transfer agreements, licensing, verification of the technology by independent consultants, and current commercialization.

The Technology Transfer

Initially, a technical consultant working with RN contacted our Research Center at Florence. This consultant read about quick wash from the ARS online news article and called the Florence lab to request more technical information, which was provided upon the signing of a confidentiality agreement.

After this first contact, company representatives attended a meeting at the Florence laboratory, where the technology was showcased. Since RN representatives did not know about ARS technology transfer programs, Florence scientists consulted with the Technology Transfer Coordinator and referred the RN CEO to the Office of Technology Transfer (OTT) at ARS Headquarters in Beltsville, Maryland. Along with technical consultations with the ARS team and the OTT, a Material Transfer Agreement (MTA) was used to determine if wash litter and the recovered phosphorus could be granulated for commercialization. ARS licensing specialists provided instructions regarding how to apply for an exclusive USDA license for the pending patent of the quick wash process.

Upon obtaining the exclusive license, the first step was to advertise in the *Federal Register* the notice of intent to grant exclusive license of the USDA invention to RN. The exclusive license was later granted after 30 days from the date of this published notice since ARS-OTT did not receive any written evidence and argument from other potential investors interested in licensing the invention. The exclusive license was granted five months later once negotiations with the OTT office in Beltsville were completed. In accordance with USDA-ARS policies and procedures, the exclusive license agreement between USDA-ARS and Renewable Nutrients, which was granted August 10, 2010, is confidential because of conflict-of-interest rules.

Finding a New Purpose for Quick Wash

The ARS team worked closely with RN to develop approaches for commercializing this new technology. Our role was to provide innovation, scientific knowledge, and improvement of the technology; RN's role was to design and develop commercial units of the technology. ARS's expectations were to effectively transfer the new technology after verification at pilot scale. RN's expectations were to reach the market with a reliable and proven technology to recover phosphorus that has competitive advantages for commercialization.

Originally we conceived of quick wash as a treatment to be used in the agricultural market with poultry litter, but our research has shown that the approach is equally effective with municipal biosolids (i.e., sewage). In addition to the agricultural market, RN realized the value of this technology for municipal disposal systems and changed its business plan to commercialize quick wash in the municipal wastewater treatment market. For this market, RN's business model now consists of sublicensing the technology to each municipal treatment plant. The technology is being marketed as Quick Wash™.

Bringing Quick Wash to Market

Jeff Dawson, CEO of Renewable Nutrients, worked to secure investors for commercialization. He interacted with local leaders to get the quick wash technology placed on a pilot scale in two municipal treatment plants.

Traditional technologies for phosphorus removal (specifically in the wastewater treatment sector) involve the addition of some form of binding agent to a facility's influent stream. Ferric chloride or aluminum sulfate, for example, are introduced and bind to phosphorus molecules, which then settle with solids material and ultimately depart the facility through its biosolids disposition program. This "chemical" removal of phosphorus can cost a facility anywhere from several thousand dollars to hundreds of thousands of dollars annually, depending upon the facility size and the amount of phosphorus that must be removed. Furthermore, this approach to phosphorus removal only serves to embed this vital nutrient into the facility's biosolids output, and in many areas of the country the biosolids must be transported to disposal sites and landfilled due to land application restrictions for phosphorus.

In recent years, a few technologies have emerged to not just "remove" phosphorus from waste streams, but recover this nonrenewable resource. Quick Wash™, however, has surfaced as the only multi-stream and truly scalable phosphorus recovery methodology in the marketplace. While nearly all of the competitive options for phosphorus recovery concentrate on the liquid side streams of wastewater treatment facilities, Quick Wash™ can remove and recover phosphorus from a facility's side stream or solid stream with recovery rates exceeding 95%. In addition, the system can be effectively deployed in small treatment facilities as well as very large operations, whereas most other nutrient recovery platforms are limited (due to their requisite level of capital investment) to large facility applications.

The following chart provides a net present value (NPV) cost comparison over a 20-year period for various phosphorus removal or recovery options, including Quick Wash™. The NPV cost includes upfront capital expense, annual maintenance fees, labor, and chemical expenses for a typical 11-MGD wastewater treatment facility.

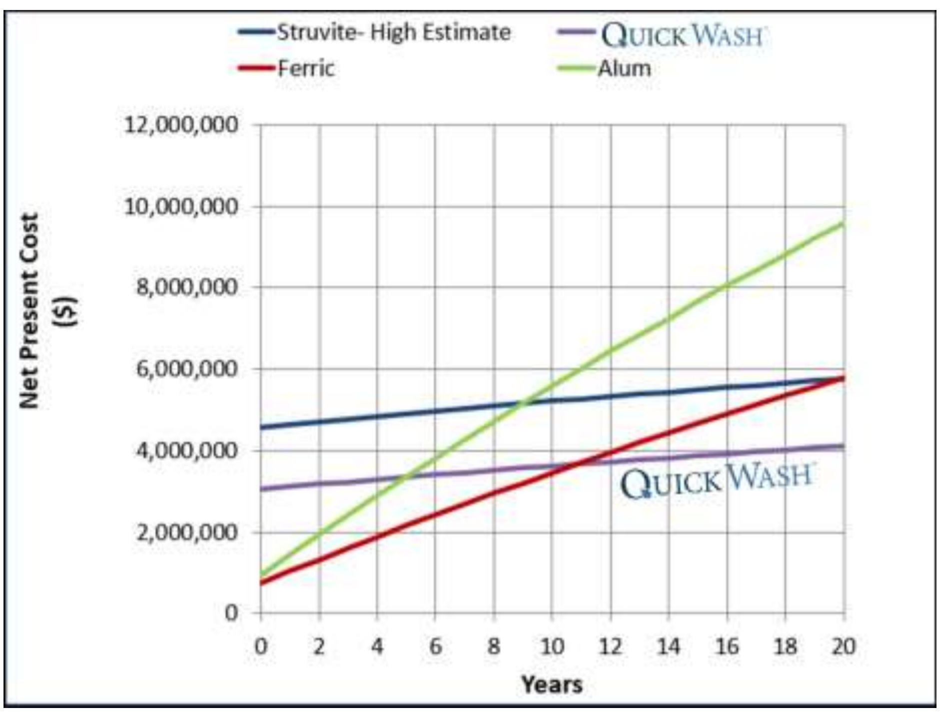


Figure 1. 20-Year NPV Cost Comparison of Phosphorous Removal/Recovery Options

The value proposition of Quick Wash is not simply limited to the removal and recovery of phosphorus. As already mentioned, Quick Wash can replace expensive chemical removal technologies. The system can also significantly reduce the amount of phosphorus in a plant's biosolids output, allowing for land application of the biosolids material and the elimination of transportation and landfill fees. Finally, facilities can sell the recovered phosphorus, a program that can serve as an incremental revenue stream, and participate in nutrient credit trading opportunities.

In the latter half of 2014, Renewable Nutrients designed and constructed a mobile pilot, at a cost of USD $500,000, for its Quick Wash phosphorus extraction and recovery system. The purpose of the mobile pilot was to demonstrate the Quick Wash technology and prove its performance in various scenarios and sizes of wastewater treatment facilities. The pilot was subsequently deployed in early 2015 at the Ephrata Wastewater Treatment Plant in Ephrata, Pennsylvania (a small 2-MGD facility).

Following Ephrata, the Renewable Nutrients team conducted eight additional pilot operations in the Mid-Atlantic, Southeast and Midwest areas, including:

- Westminster, Md.
- Raleigh, N.C.
- Chapel Hill, N.C.
- Greenville, N.C.
- Neoga, Ill. (a large swine production operation).

Currently, RN is marketing Quick Wash™ for the recovery of phosphorus from both animal and human waste through different techniques such as the company website and blog, and exhibiting at national technical conferences for water treatment professionals, such as the Mid-Atlantic Biosolids Association, the Water Environment Federation, and the American Water Works Association.

Our team at the Research Center has hit on a creative marketing technique: We worked with our IT staff to develop a bar code that participants can scan and immediately access a promotional video that RN produced about Quick Wash™.[8] The bar code was included in posters presented at scientific meetings by ARS Florence scientists. We thought it was a good idea to include a link to this video clip because our most recent technical presentations contain results of tests carried out at the request of the licensee. The ARS technology transfer policy permits its inventors, where practicable, to participate

[8] Video, "Renewable Nutrients Quick Wash™ Overview." March 19, 2015. https://youtu.be/VOq2mGh24js.

in the development of their inventions by providing technical assistance to licensees. We have been providing this assistance through monthly conference calls between scientists and RN personnel.

Figure 2. Bar Code Linking to RN Promotional Video

Looking Back: What We Learned

Trust is Key

Renewable Nutrients was the right partner because they trusted the science behind the quick wash project. A mutual trust developed between ARS and RN as a result of reviewing and confirming through technical advisors that the information included in the patent application was a sound scientific approach for phosphorus recovery from animal waste. When RN decided to change its business plan to commercialize Quick Wash™ in the municipal wastewater treatment market, it contacted the ARS Florence scientists to determine if the invention could be used for municipal waste treatment. Once again, we referred the CEO of RN to consult with the OTT office. This time, the ARS patent advisor who prepared the patent application confirmed to RN that the invention also covered use of the quick wash process for recovering phosphorus from municipal waste streams. At RN's request, our team provided technical assistance by performing laboratory tests to demonstrate the feasibility of recovering phosphorus from municipal sludge and biosolids. These lab tests showed that more than 80% of the phosphorus contained in municipal wastes can be recovered using the quick wash process. In addition, the process was tested by a third party—an environmental engineering consulting firm hired by RN—that satisfied RN's expectations of using the process to recover phosphorus from municipal sludge and biosolids. In addition, the consulting engineers developed a marketing program and a mobile pilot plant to test Quick Wash™ onsite at the municipal plant.

Patience Is a Virtue

The major challenge was to maintain the licensee's and scientists' interest in developing the technology during the long time before the technology reached commercialization. It took seven years from the initial ARS news release to start commercialization of the technology. It took the first two of the seven years to license it. Since the technology was licensed while the patent was pending, plus RN's difficulty selling it in the animal waste treatment market, it took RN another four years to redirect its marketing strategy. It is important to mention that RN started to heavily invest in developing and marketing the technology for use in municipal waste treatment once the patent was officially issued in 2014. The commitment of the ARS scientists to provide technical assistance to RN was extremely important in helping RN shift its commercialization plan for the municipal market.

We Could Still Be More Agile

If we could do this tech transfer process over again, we would try to be better prepared to extend the focus of our invention beyond the research laboratory. This could have helped to transfer and commercialize the technology faster. A program for customer discovery and identification of real problems in the different sectors of the waste management industry could have helped ARS have a better idea of the business side of science while helping the licensee to discover an alternative market for the quick wash technology.

In 2015, the ARS started a program called ARS Innovation Corps (I-Corps @ ARS) to help scientists for faster transfer and commercialization of technology. The I-Corps @ ARS pilot consists of a set of activities and programs designed to help ARS scientists broaden the impact of their research by extending their focus beyond the laboratory to the end product of their work. I-Corps @ ARS is based on the NSF I-Corps™ and basically combines experience from established entrepreneurs with a curriculum on market opportunities and innovation.

If we had had this training years ago, we could have learned to be more "agile" with our tech transfer.

About the Authors

Ariel A. Szogi and Mattias B. Vanotti are soil scientists, and Patrick G. Hunt holds the title of collaborator at the Agricultural Research Service's Coastal Plain Soil, Water and Plant Conservation Research Center.

Notice

ATTACHMENT 66

QUICKWASH®

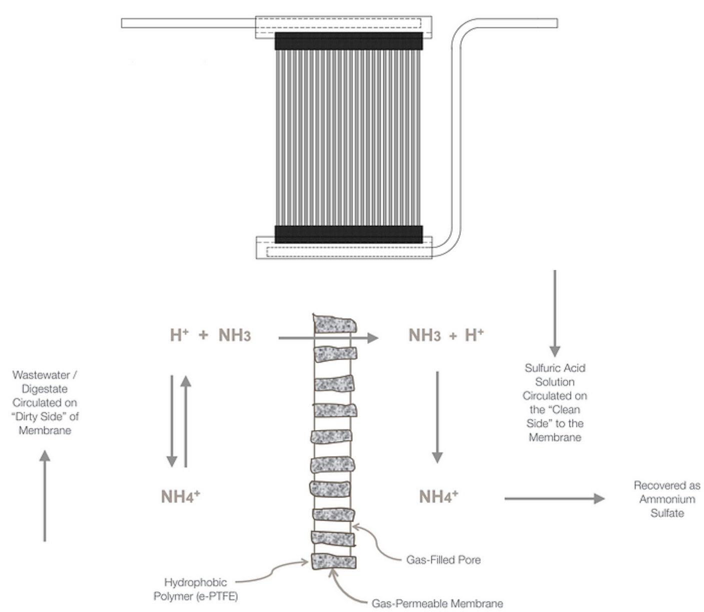
Nitrogen Removal & Ammonia Recovery

Renewable Nutrients is rethinking Nitrogen Removal & Ammonia Recovery with a proprietary system that draws on exclusive patented technology to extract and recover ammonia in either a liquid or gaseous state found in biosolids and other waste streams.

Renewable Nutrients patented Nitrogen Removal & Ammonia Recovery process is disruptive to current technologies as it recovers the ammonia for beneficial use and by its very nature can be operated with no impact to the biological process.

With the Renewable Nutrients patented Ammonia Recovery process in place, an existing facility will:

- Reduce inputs of energy, carbon, alkalinity
- Recover rather than destroy a valuable resource
- Produce a high quality, saleable product
- Improve the quality of the effluent by enhancing existing nutrient removal processes
- Allow the reduction of ammonia alkalinity
- Potentially free existing capacity and allow the rerating of treatment plants in lieu of additional capital investment
- Provide a solution for additional capacity in plants with limited footprint for expansion



Need more details? Contact us

We are here to assist. Contact us by phone, email or via our Social Media channels.

Contact Us

ATTACHMENT 67

Separation of Ammonia and Phosphate Minerals from Wastewater using Gas-Permeable Membranes

WRRI Water Sustainability Through Nanotechnology Symposium
March 15, 2017 – Raleigh, NC

Matias Vanotti

Collaborators: Patrick Dube, Ariel Szogi

United States Department of Agriculture, ARS

Florence, SC

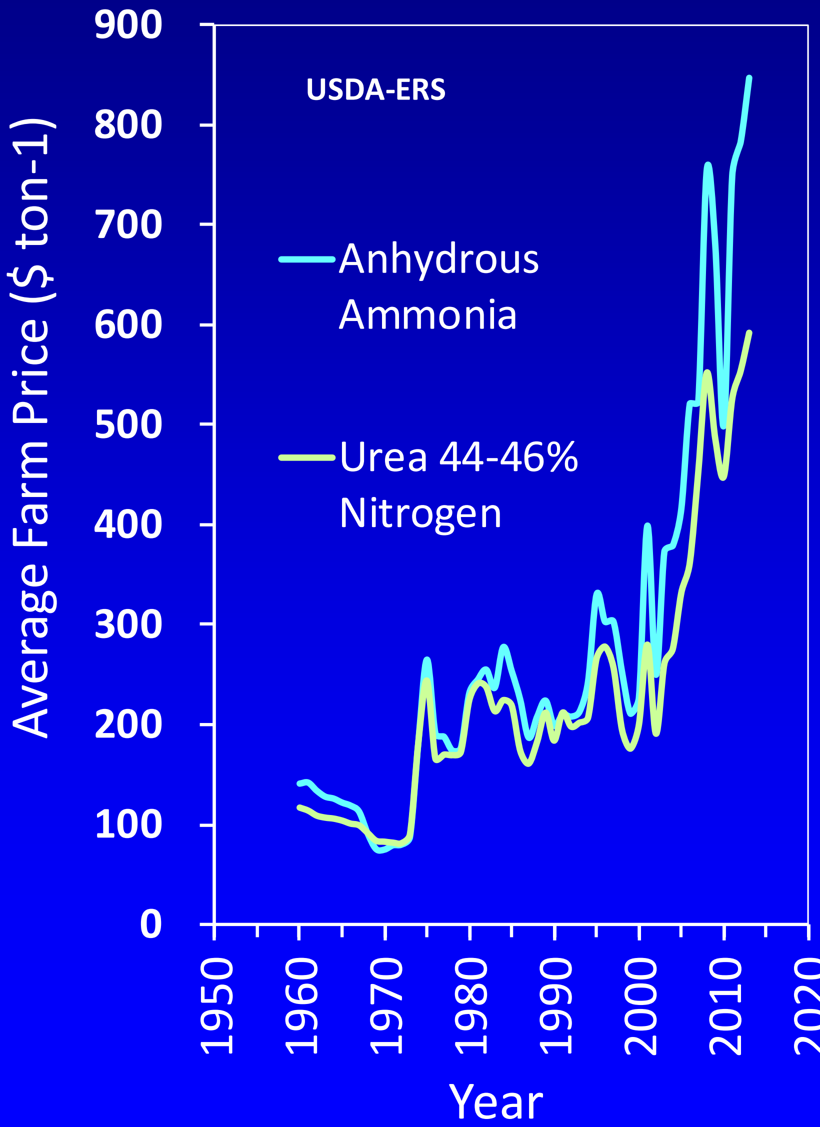
Presentation outline

Recent development at USDA of systems and methods to recover N, P and value-added materials from wastes

- 1. Improved ammonia recovery from liquid with gas-membranes
- 2. Simultaneous N and P recovery with membranes
- 3. Recovery of ammonia without chemicals

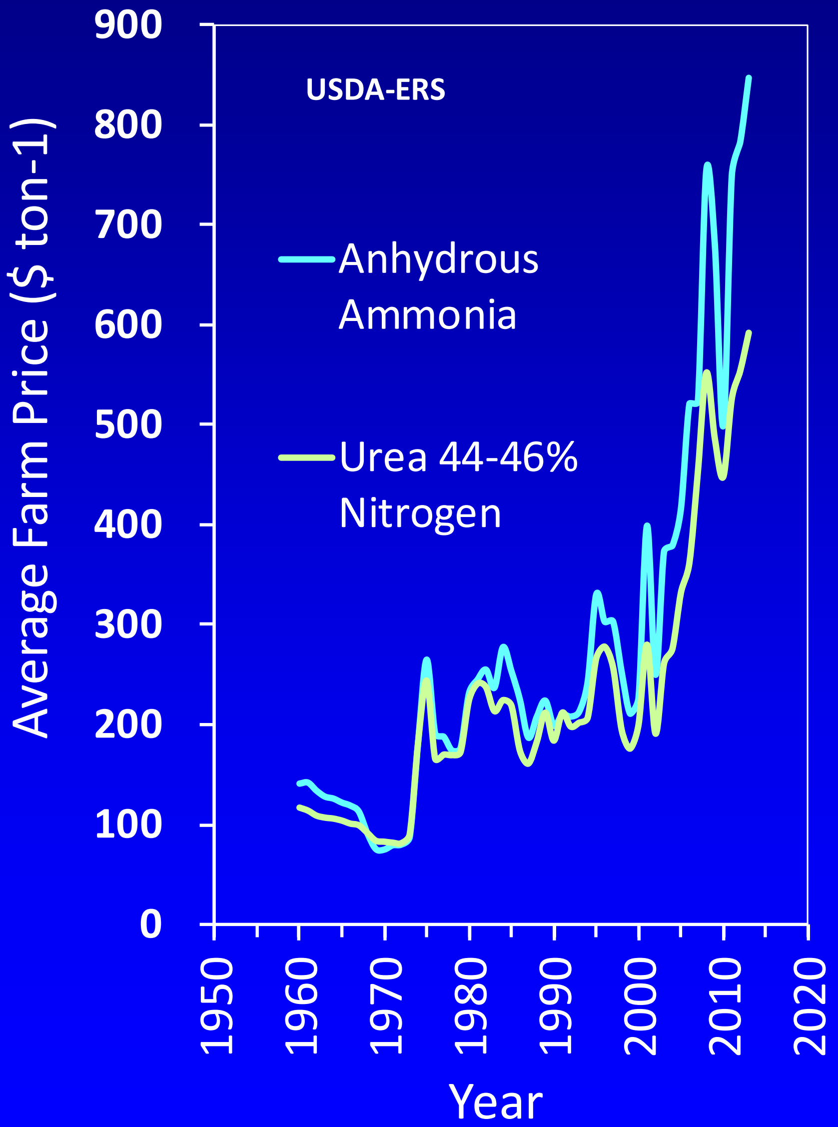
Why recover N?

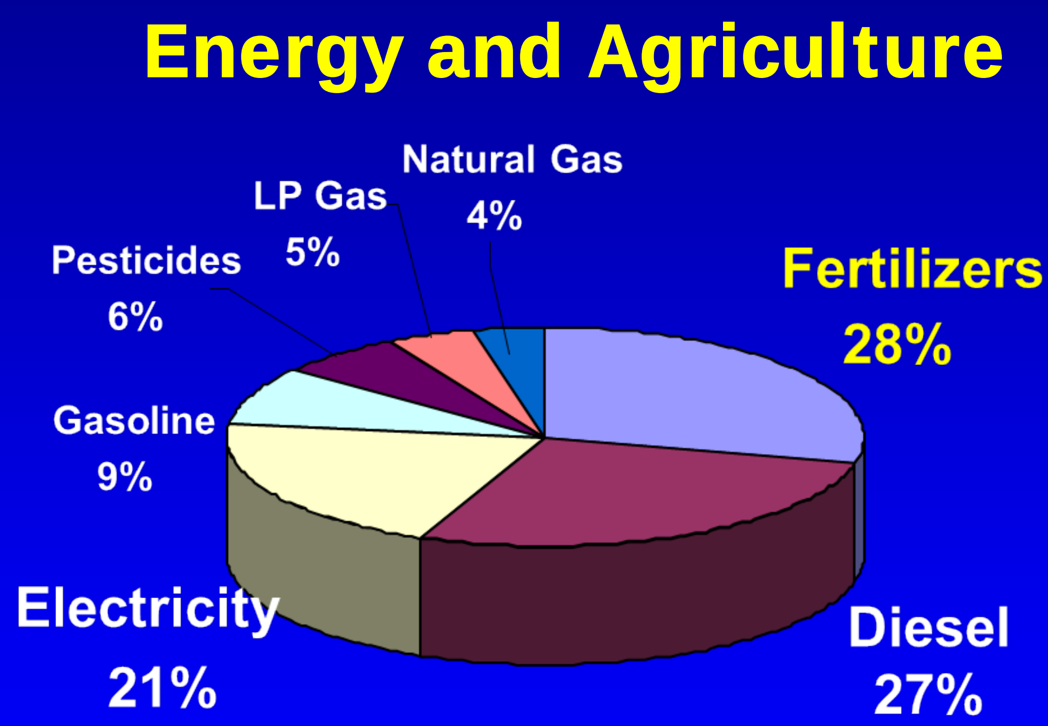
Escalating U.S. Fertilizer Costs



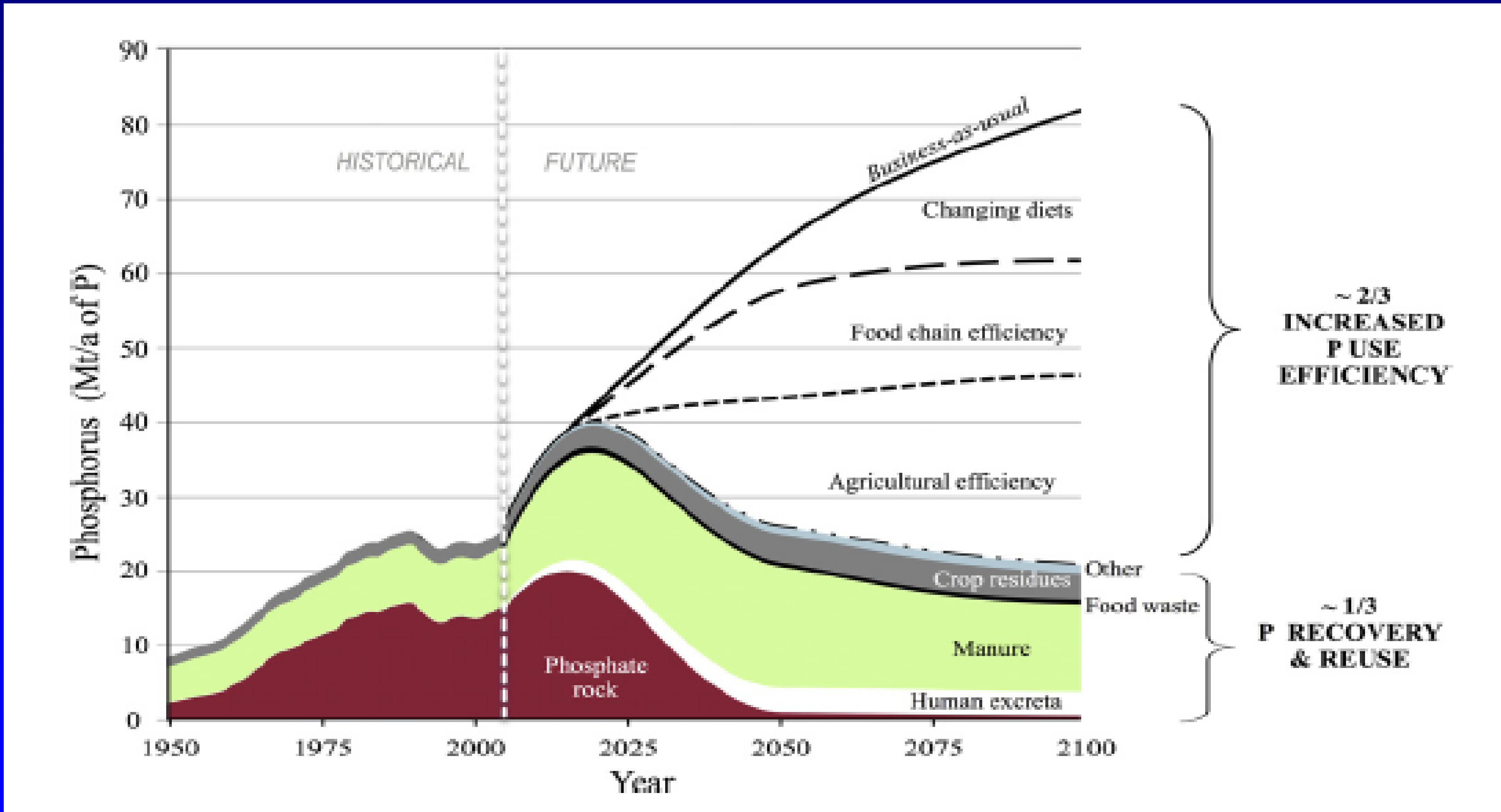
Why recover N?

Escalating U.S. Fertilizer Costs



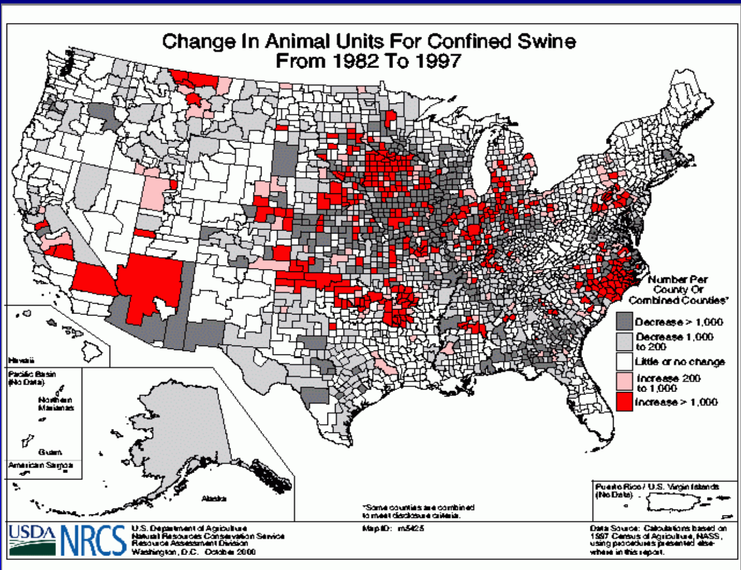


Why recover phosphorus?



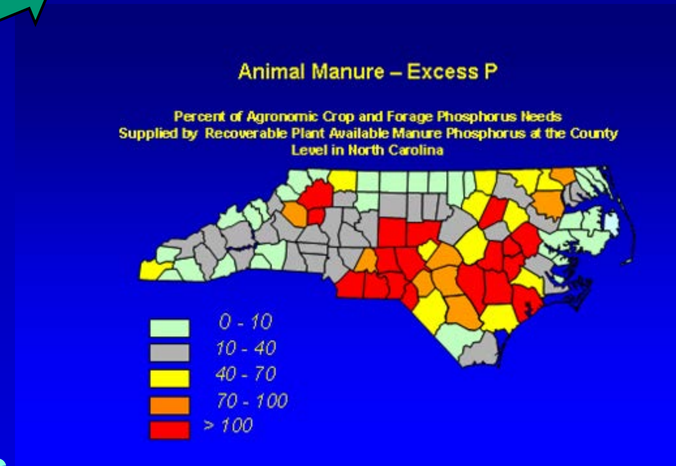
Cordell, D., et al., Chemosphere 84:747-758

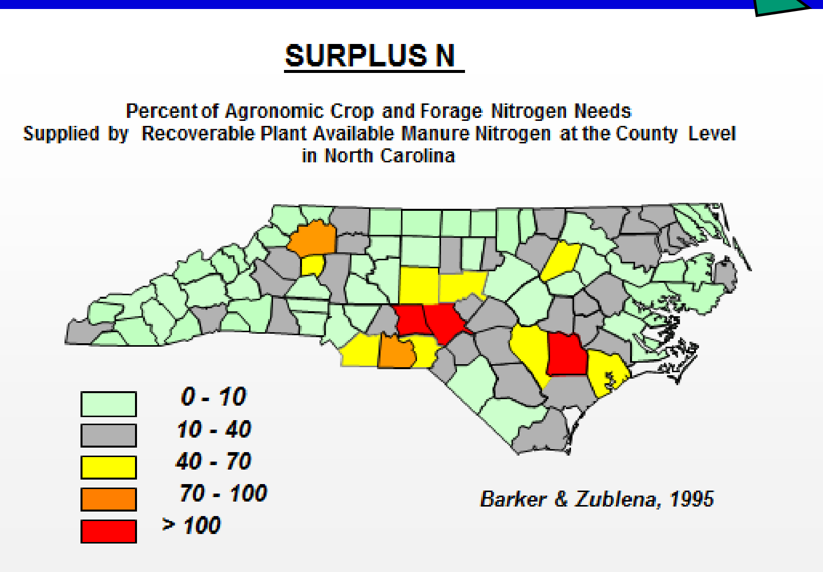
Animal Manure – Surplus N & P , Ammonia emissions in areas of concentrated animal production



North Carolina produces approximately 750 million chickens, 40 million turkeys, 3.5 billion table eggs, and 19 million hogs per year.

Surplus Phosphorus





Ammonia Emissions

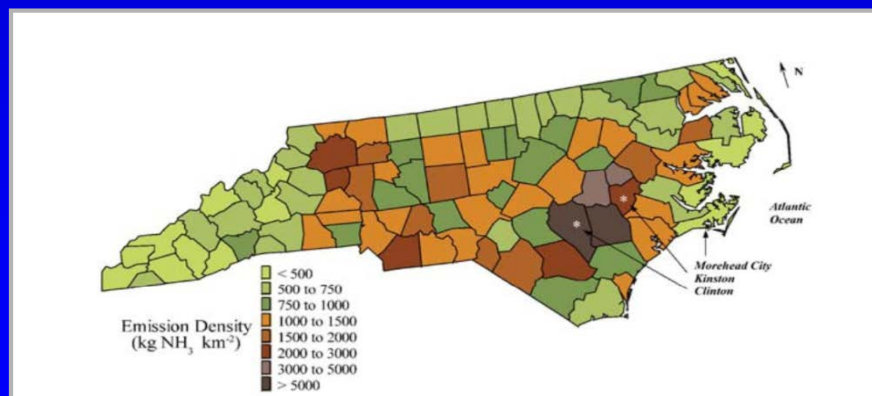
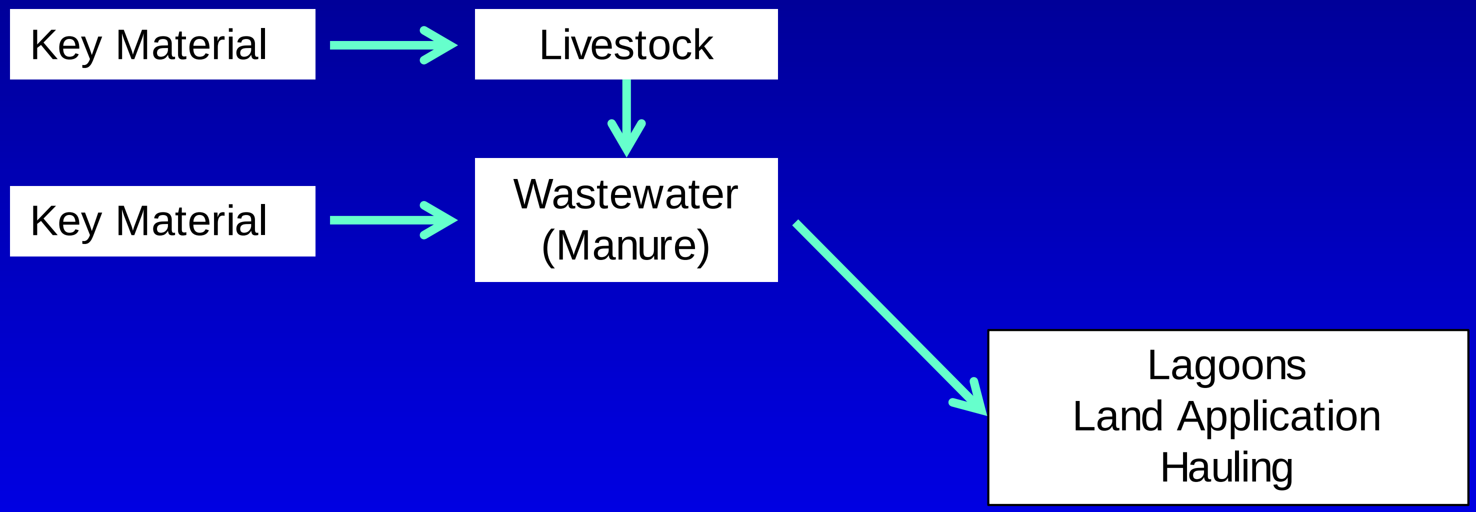


Fig. 1. County scale NH_3 emission density for North Carolina along with measurement sites. Livestock activity data represent 2000 levels. All other activity data represent 1996 levels.

Walker et al., Atmos. Environ. 38:1235-1246

Value Chain without Solution



The Technology

What do you do?

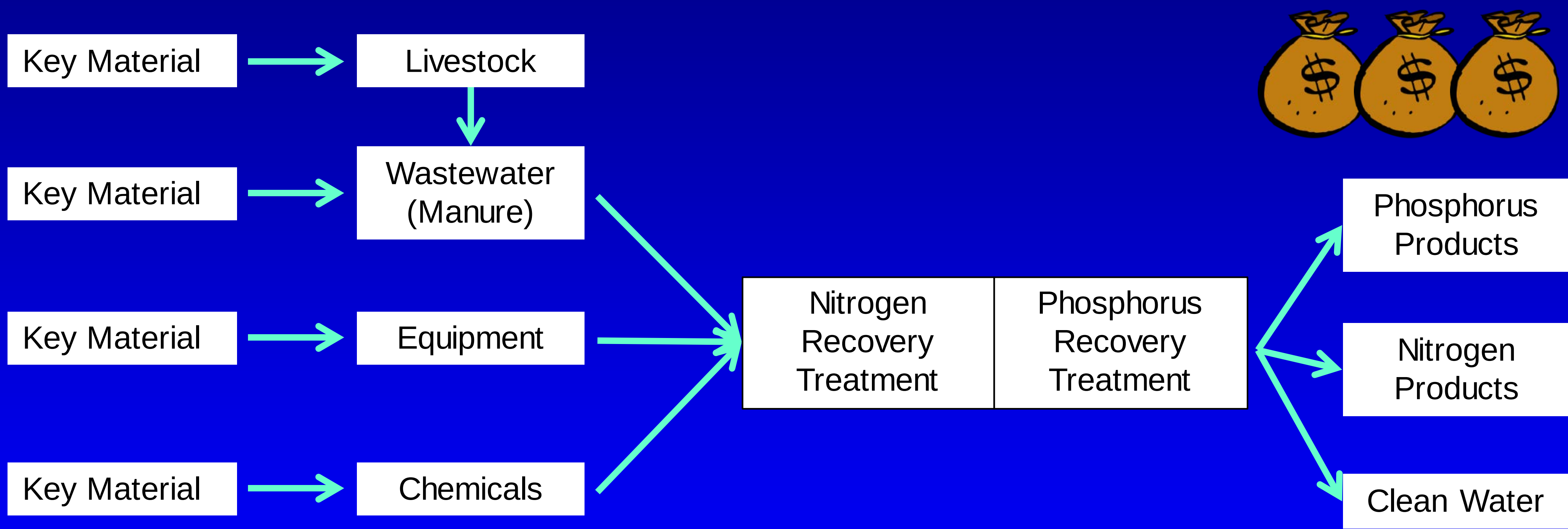
- Our technology simultaneously removes and recovers both nitrogen and phosphorus from manures and wastewaters.

Why do you do it?

- This creates value added products from wastes and helps society with a cleaner environment.

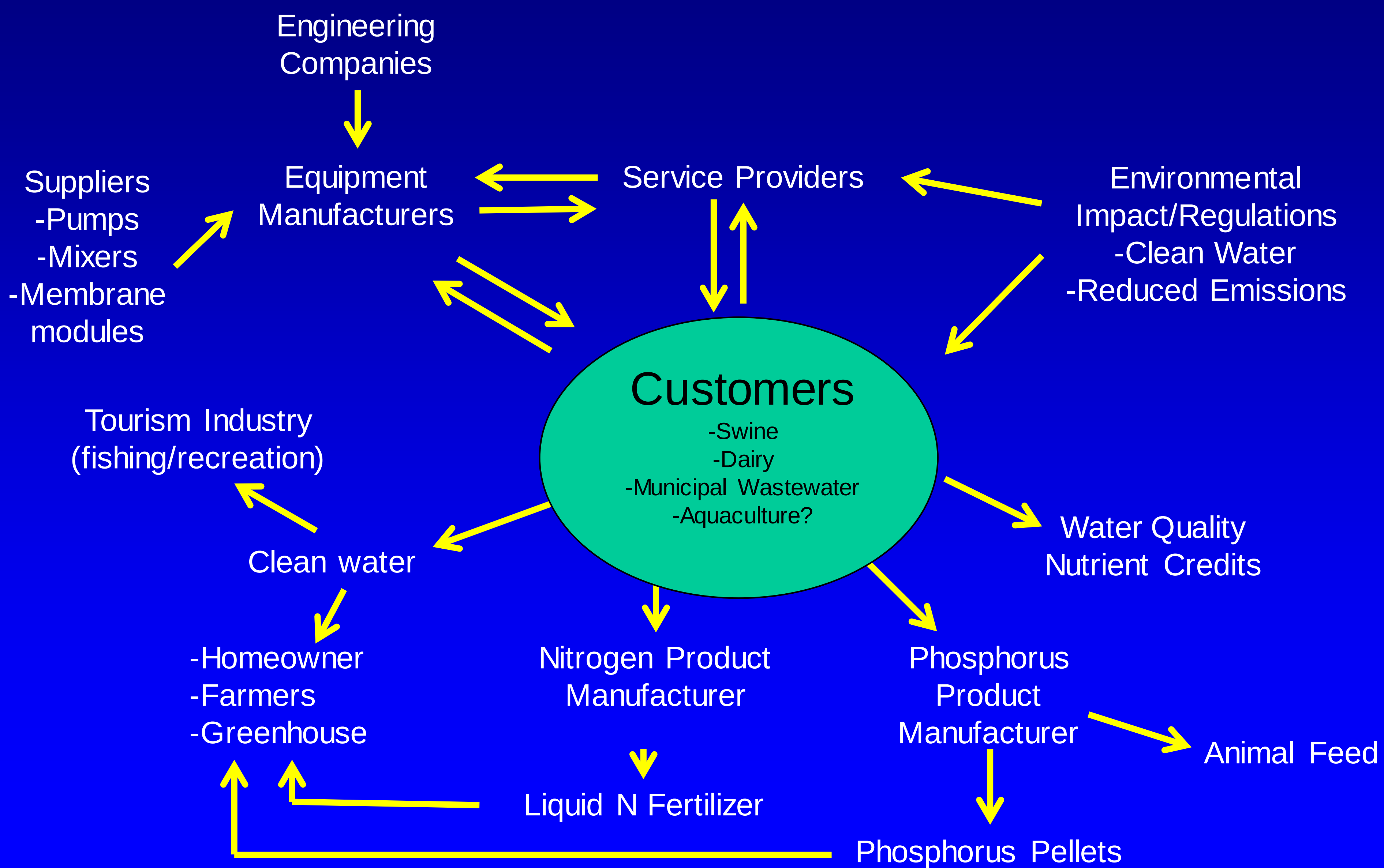
Wastewater (Manure) → Nitrogen Recovery Treatment | Phosphorus Recovery Treatment → Phosphorus Products / Nitrogen Products / Clean Water

Value Chain with Solution

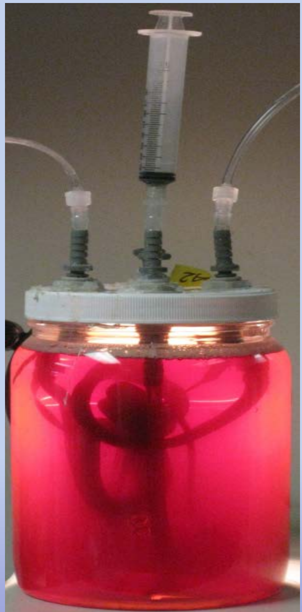


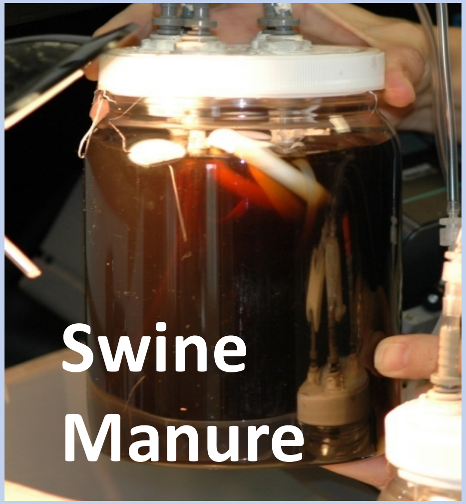
Ecosystem Map With Solution

How your product interacts with the world once it is in the hands of the customer



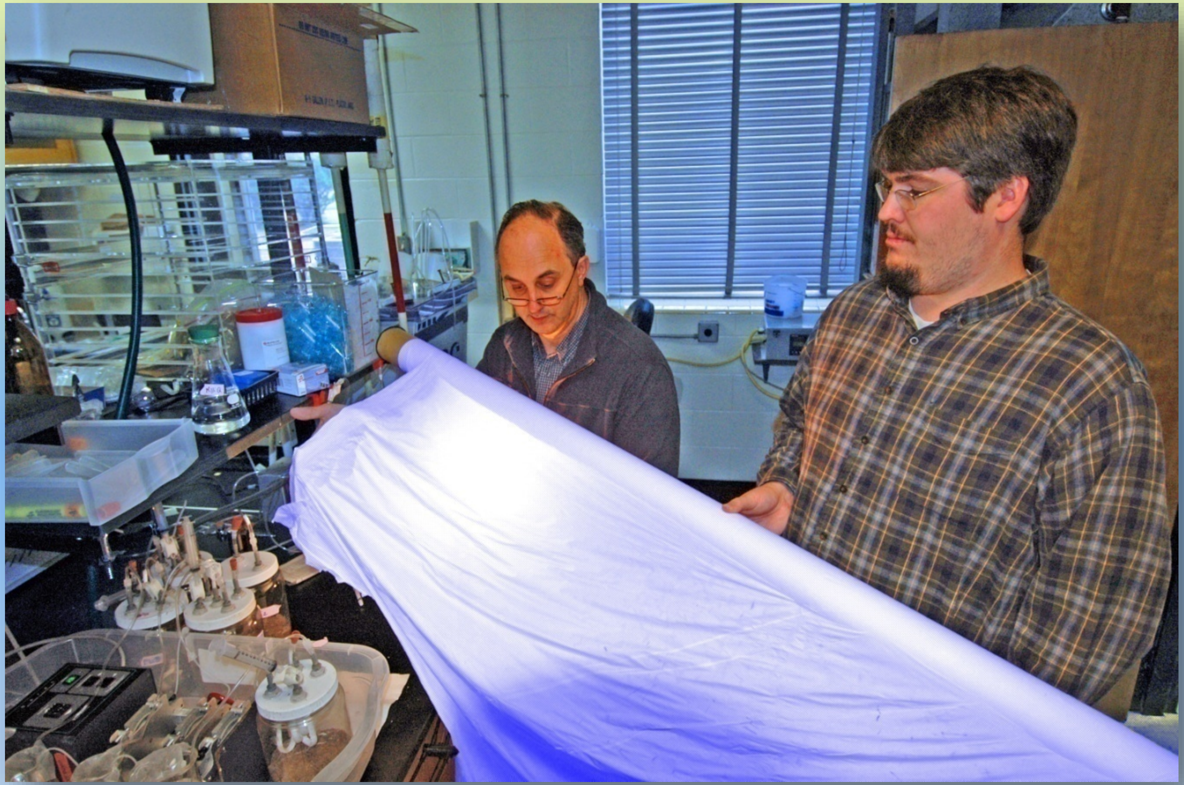
New technology: Recovery of Ammonia from Manure

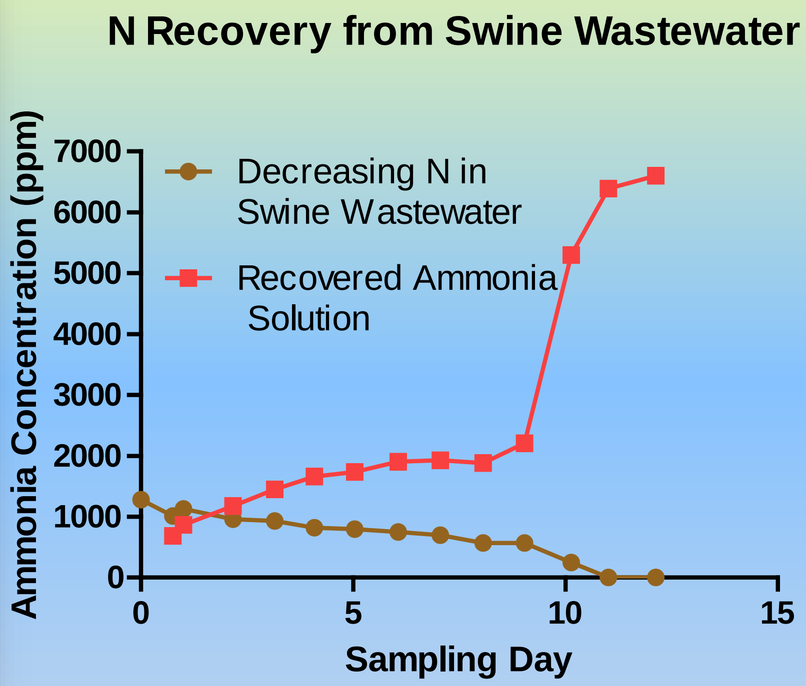




- ❑ Ammonia is separated using gas-permeable membranes
- ❑ Applications include liquid manures and air in livestock houses
- ❑ Product is liquid fertilizer with 50,000 to 100,000 ppm N

Recovery and Concentration of Ammonia





- **Ammonia permeation through microporous, hydrophobic membranes**
- **Reduced ammonia emissions from livestock operations**
- **Product is ammonia solution with > 50,000 ppm N**

Gas-permeable membranes

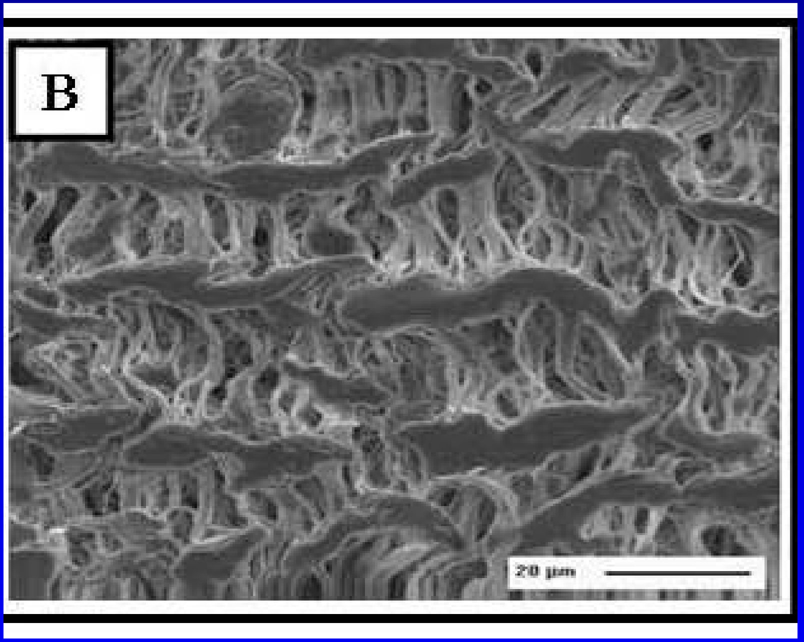
- **Medical uses:** Used in membrane oxygenators to imitate the function of the lungs in cardiopulmonary bypass, to add oxygen to, and to remove carbon dioxide from the blood (Gaylor, 1988).

- **Clothing & shoe industries:** Used to provide waterproof and <u>breathable</u> fabrics in sportswear and footware (i.e. *GORE-TEX® Products, 1968)*

For this research we used gas-permeable membranes made of expanded polytetrafluoroethylene (ePTFE)

PTFE is stretched to form a strong, porous material

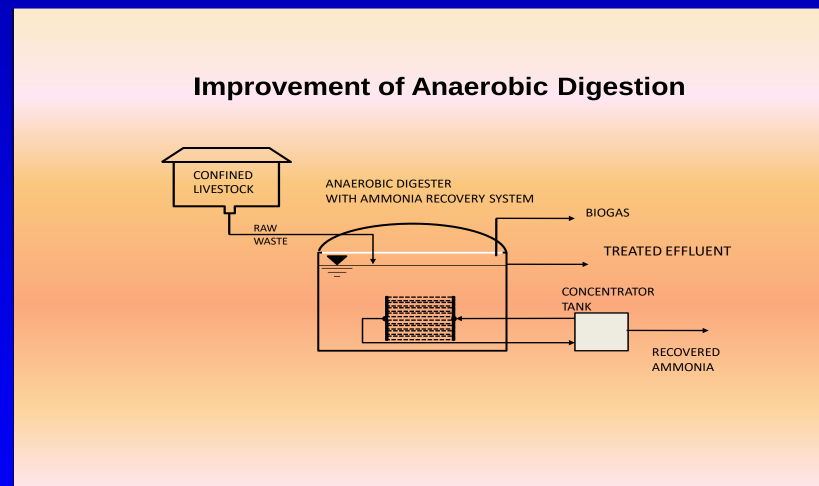
Manufacture of Gas Permeable Membrane

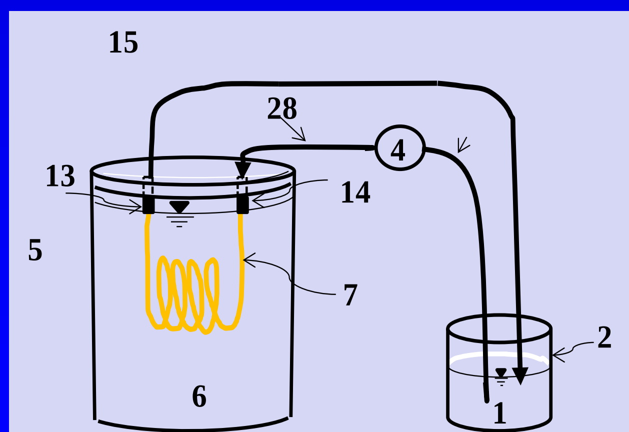


Gas Permeable Membrane
Microscopic structure (SEM)

Recovery of Ammonia from Liquid Manure with Gas-permeable Membranes

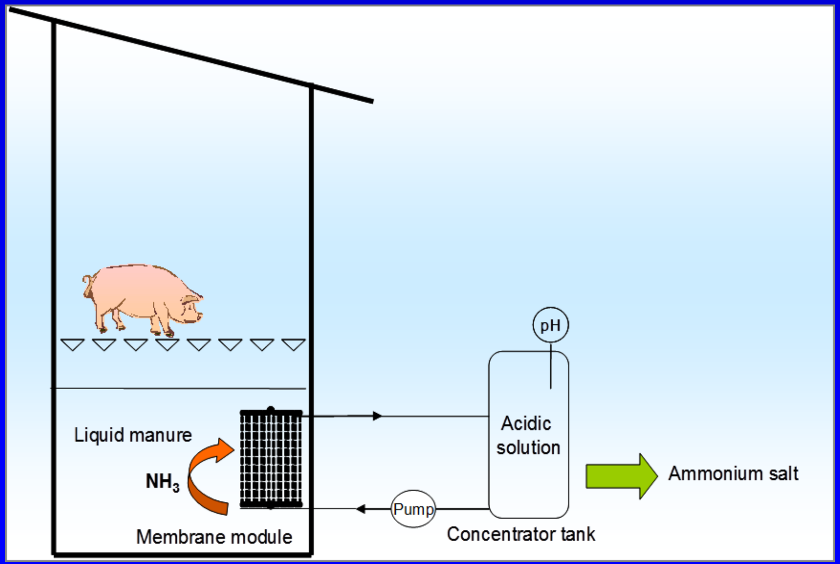
- Technology captures ammonia emissions
- Produces liquid fertilizer with > 50,000 ppm nitrogen



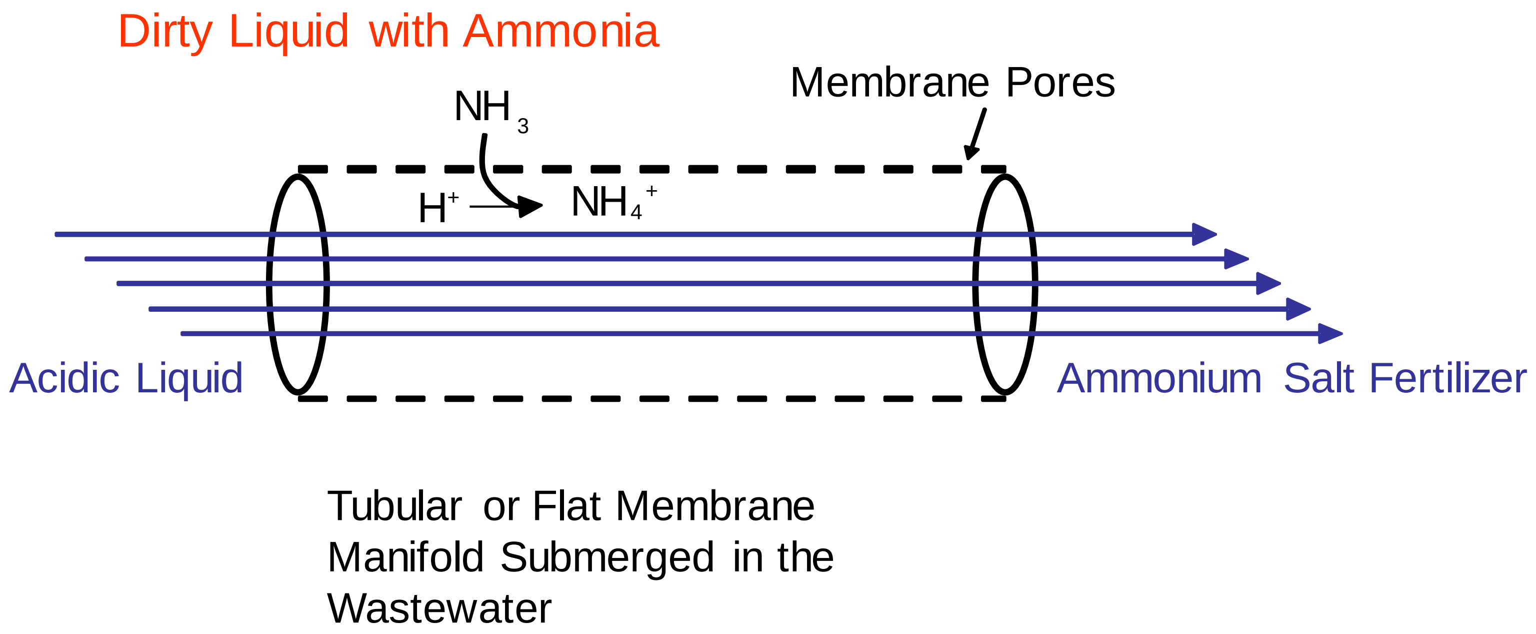


WHAT IS INTENDED TO DO?

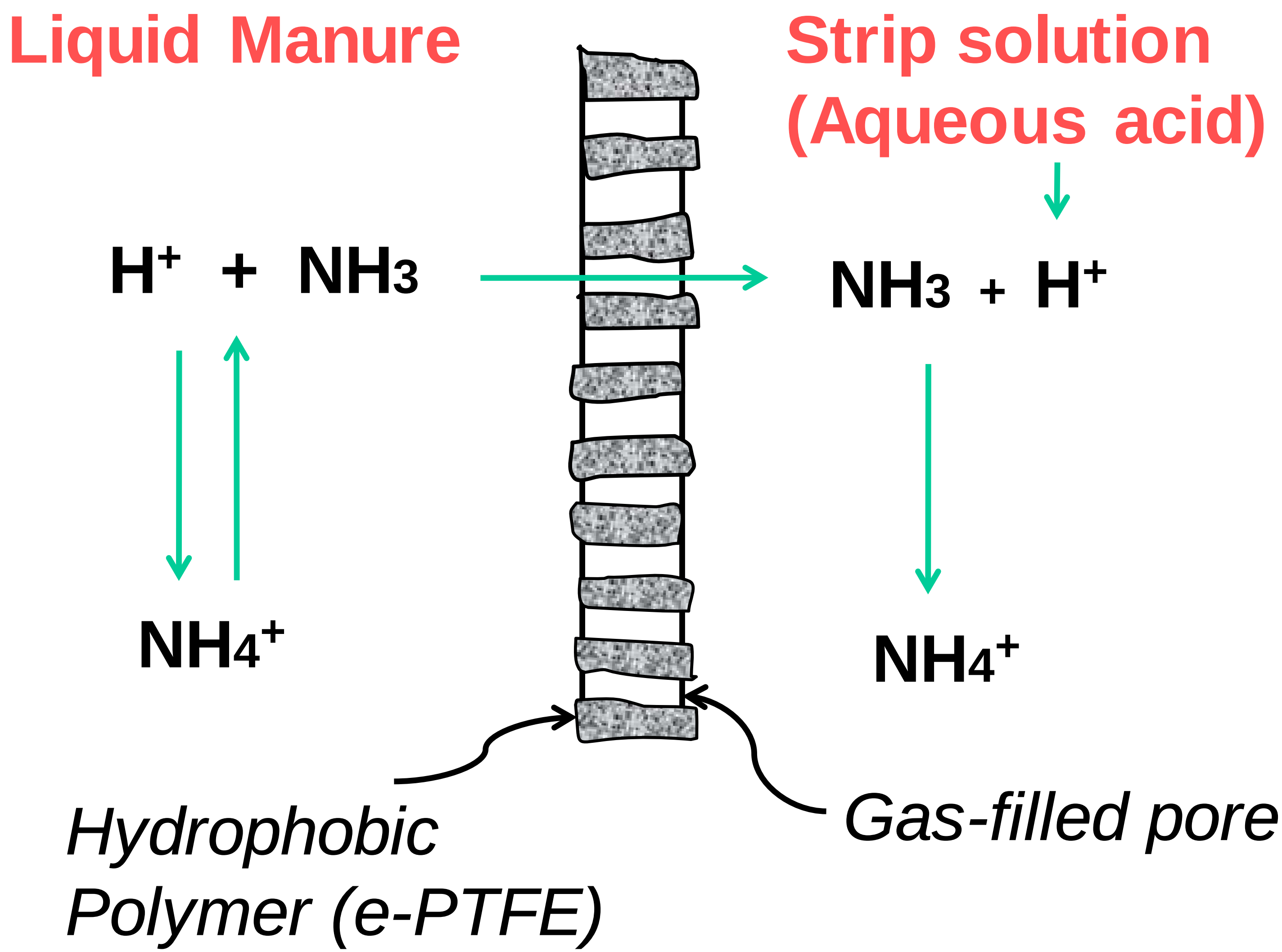
- Removal of ammonia gas from the liquid manures before it escapes into the air.
- Nitrogen is recovered from liquid manures in a concentrated, purified form



Concept of Ammonia Capture from Wastewater using Gas Permeable Membrane

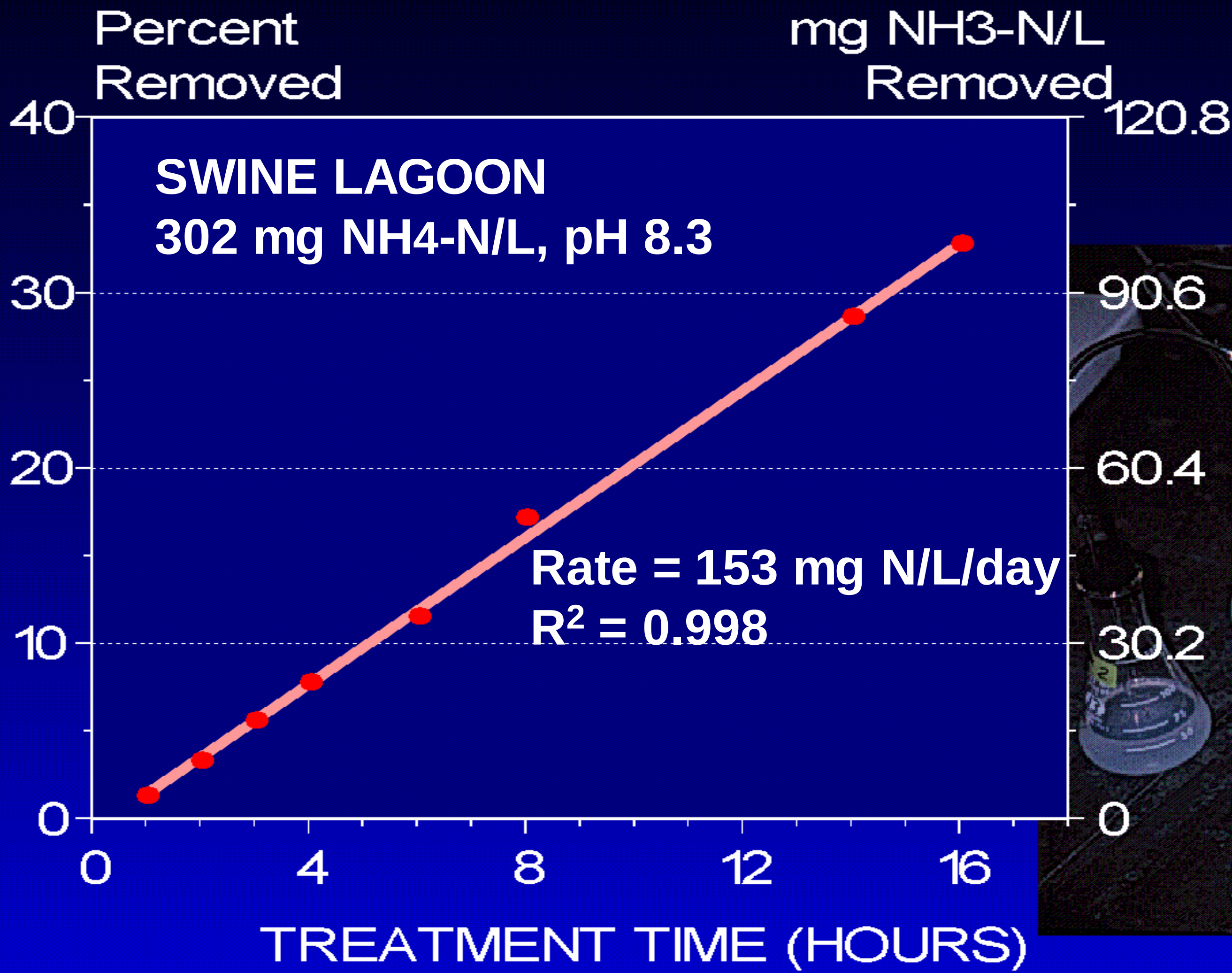


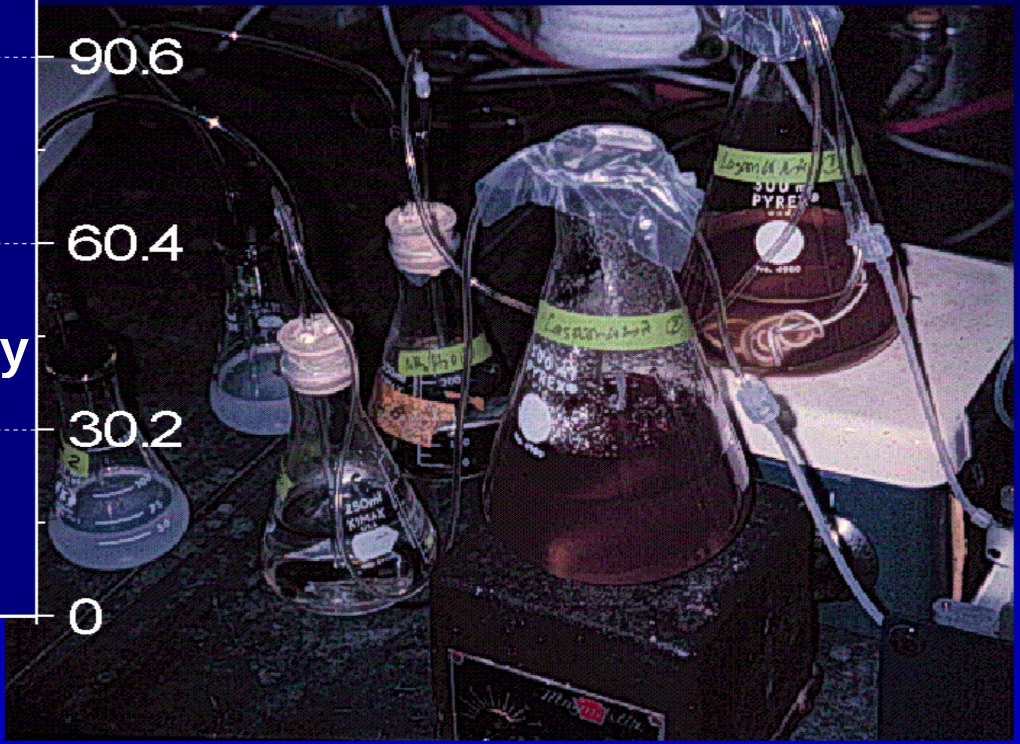
Gas-permeable membrane system:
The ammonia gas (NH3) passes through



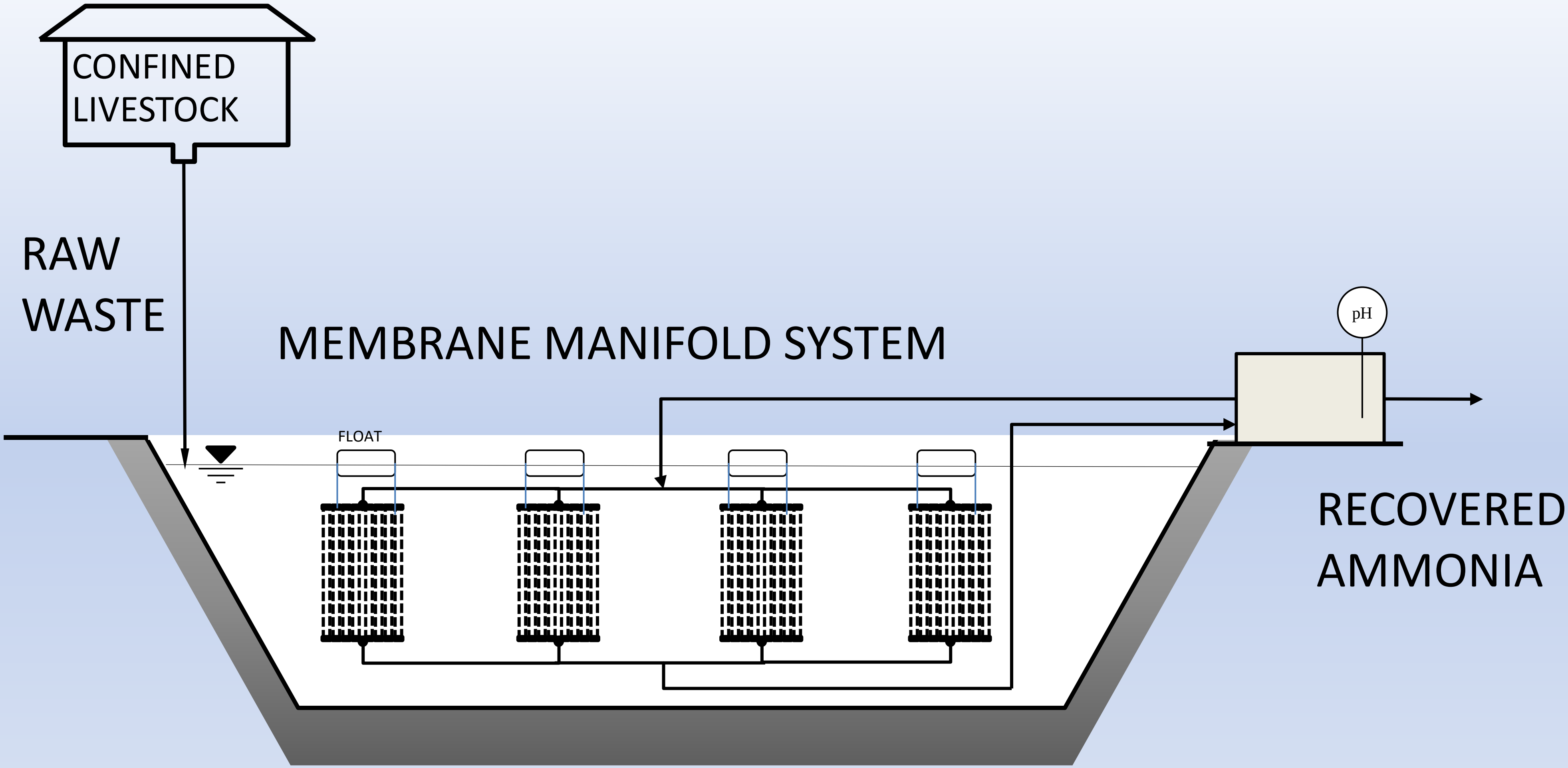
Does it work?

Ammonia removal from animal waste using gas permeable membranes



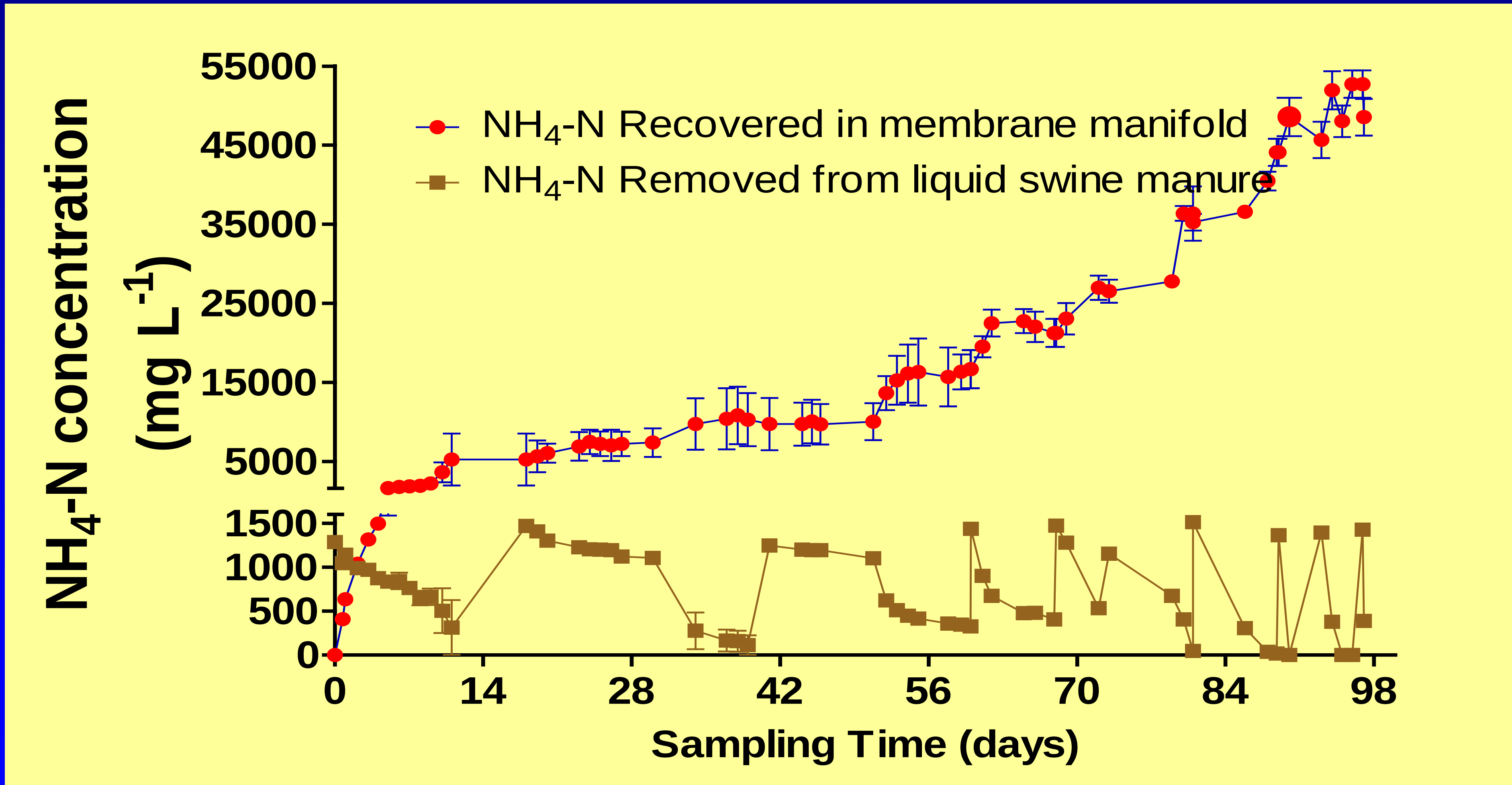


Retrofit of manure storage units to harvest the ammonia



Anaerobic Livestock Wastewater Lagoon with
Ammonia Recovery System

Recovery and Concentration of Ammonia from Liquid Swine Manure using Gas Membranes (10 batches using same stripping solution)



Recovered NH_4-N was concentrated to 53,000 ppm

Microporous gas-permeable membrane :
In tests, the soluble carbon did not pass through

Synthetic Wastewater

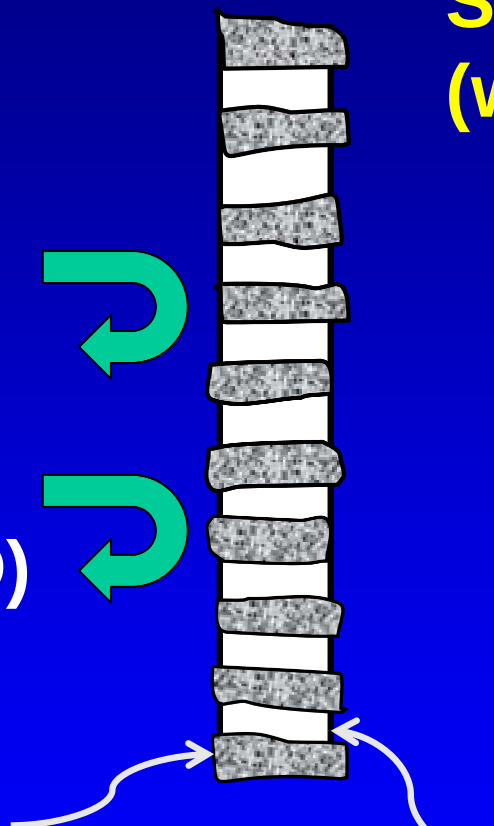
Glucose
(500 ppm COD)

KHP
(1000 ppm COD)

Strip solution (water)

H_2O

0 ppm COD



Hydrophobic Polymer (e-PTFE)

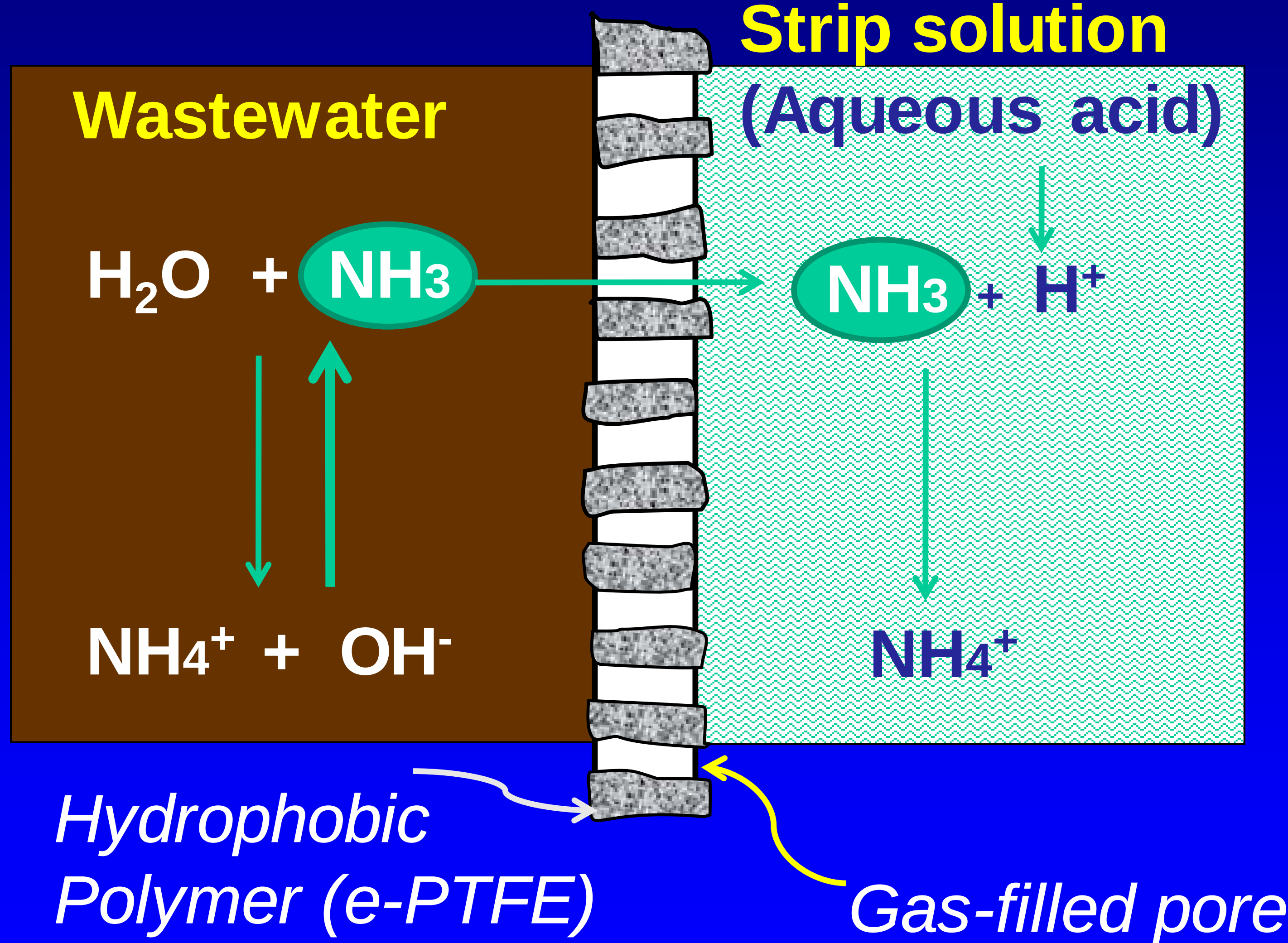
Gas-filled pore

Design Parameter: Effect of wastewater pH:

Time (hours)	Initial Source pH = 8.3			Initial Source pH = 10.0		
	Mass NH_4-N in Trap (mg)	NH_4-N Recovery from Source (%)	pH of Trap	Mass NH_4-N in Trap (mg)	NH_4-N Recovery from Source (%)	pH of Trap
0	0	**0**	1.08	0	**0**	1.08
1	0.86	**1.0**	1.11	7.82	**8.7**	0.99
2	2.44	**2.7**	0.98	26.51	**29.4**	1.16
3	3.72	**4.1**	0.99	38.60	**42.9**	1.28
4	4.77	**5.3**	1.1	48.86	**54.3**	1.6
5	5.39	**6.0**	1.0	56.40	**62.7**	1.8

N Recovery was ~ 1.2 % per hour at pH 8.3 and 13% per hour at pH 10 (increased 10 times)

Gas-permeable membrane used for separation of free ammonia (NH_3)

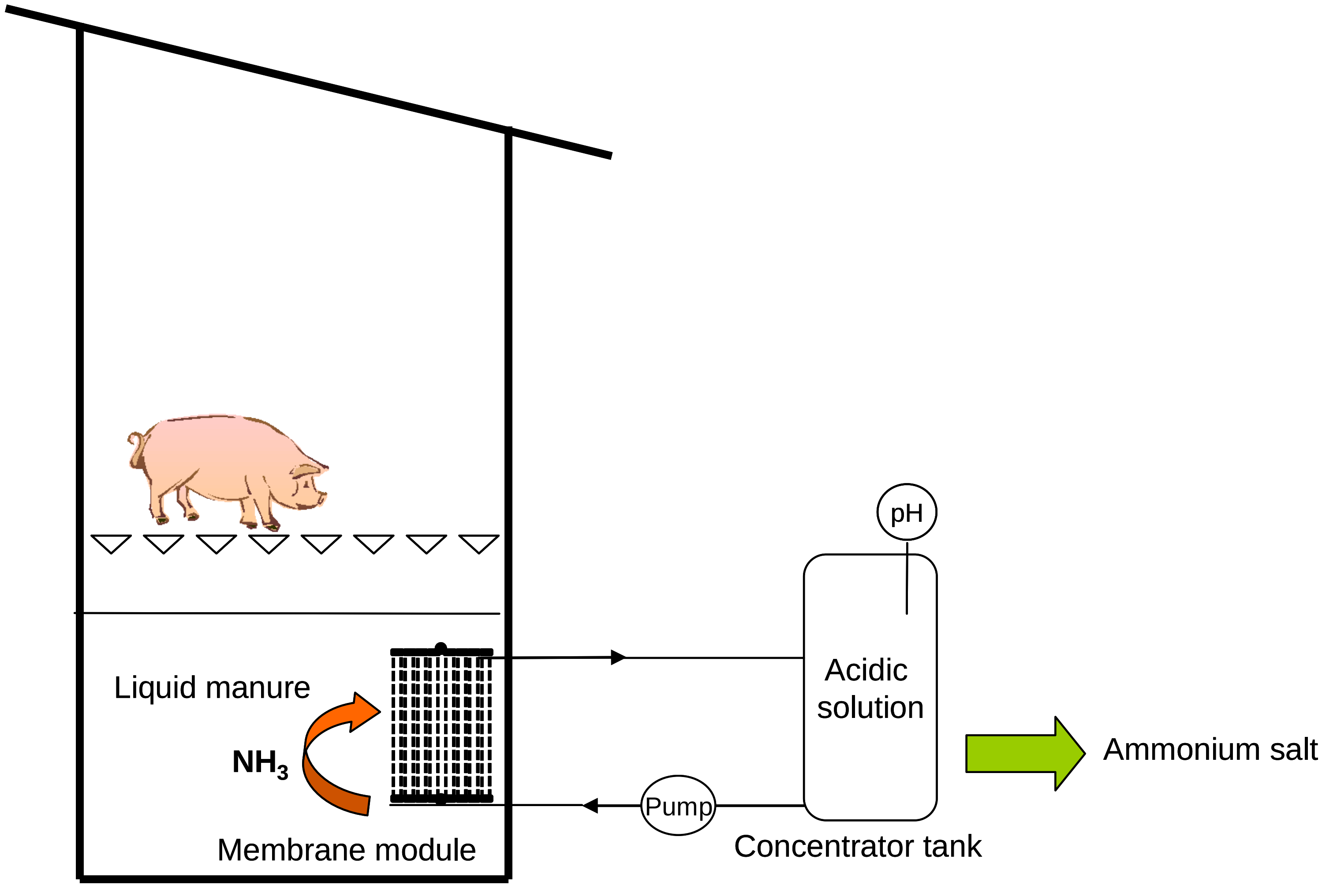


Design Parameter: Effect of waste strength

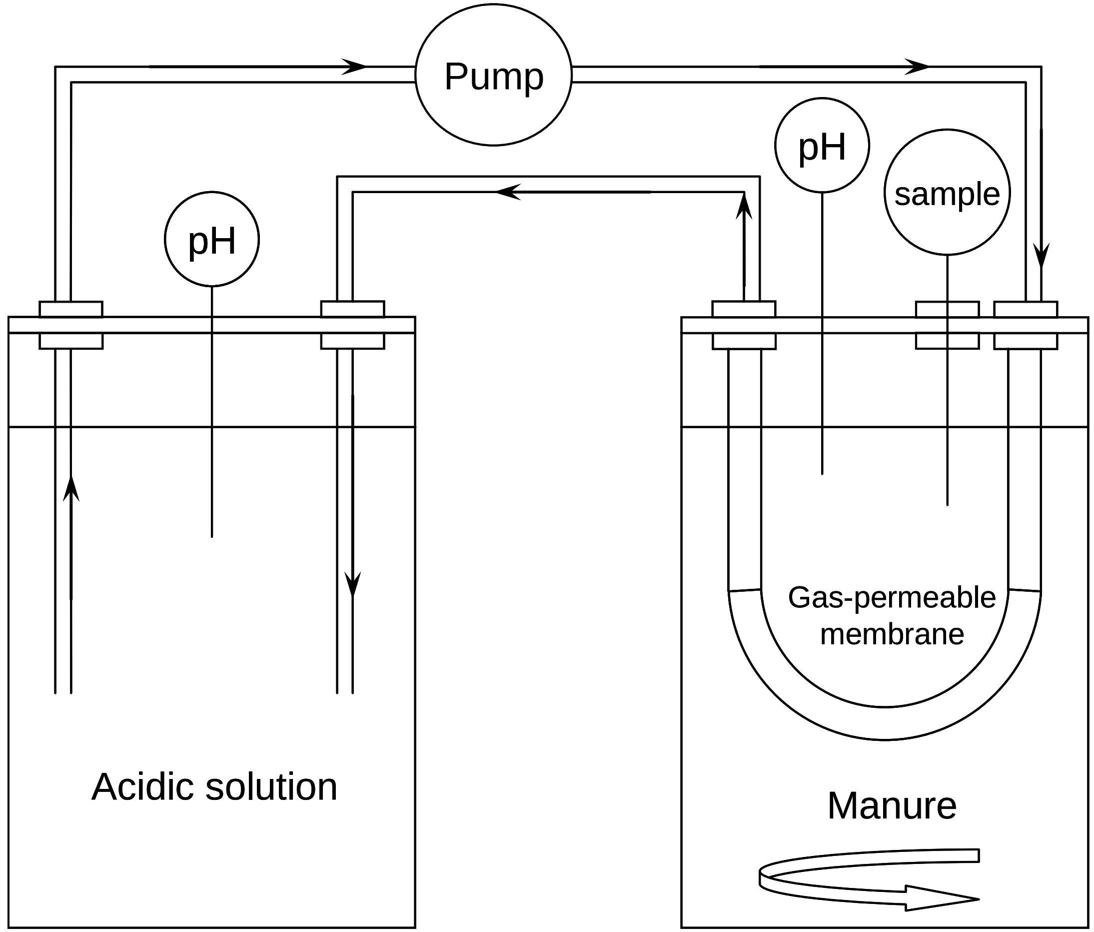
Swine manure characteristics

Manure strength	Swine Farm Type	pH	NH_4-N mg/l	TKN mg/L	EC (mS)	COD mg/L	TS g/L	VS g/L
Low	Piglet	8.64	1065	1345	8.470	4519	4.89	2.58
Medium	Farrow-finish w/ separation	7.57	1680	2743	14.080	24405	17.41	10.33
High	Finishing	7.52	2285	3699	16.980	34081	29.87	20.13

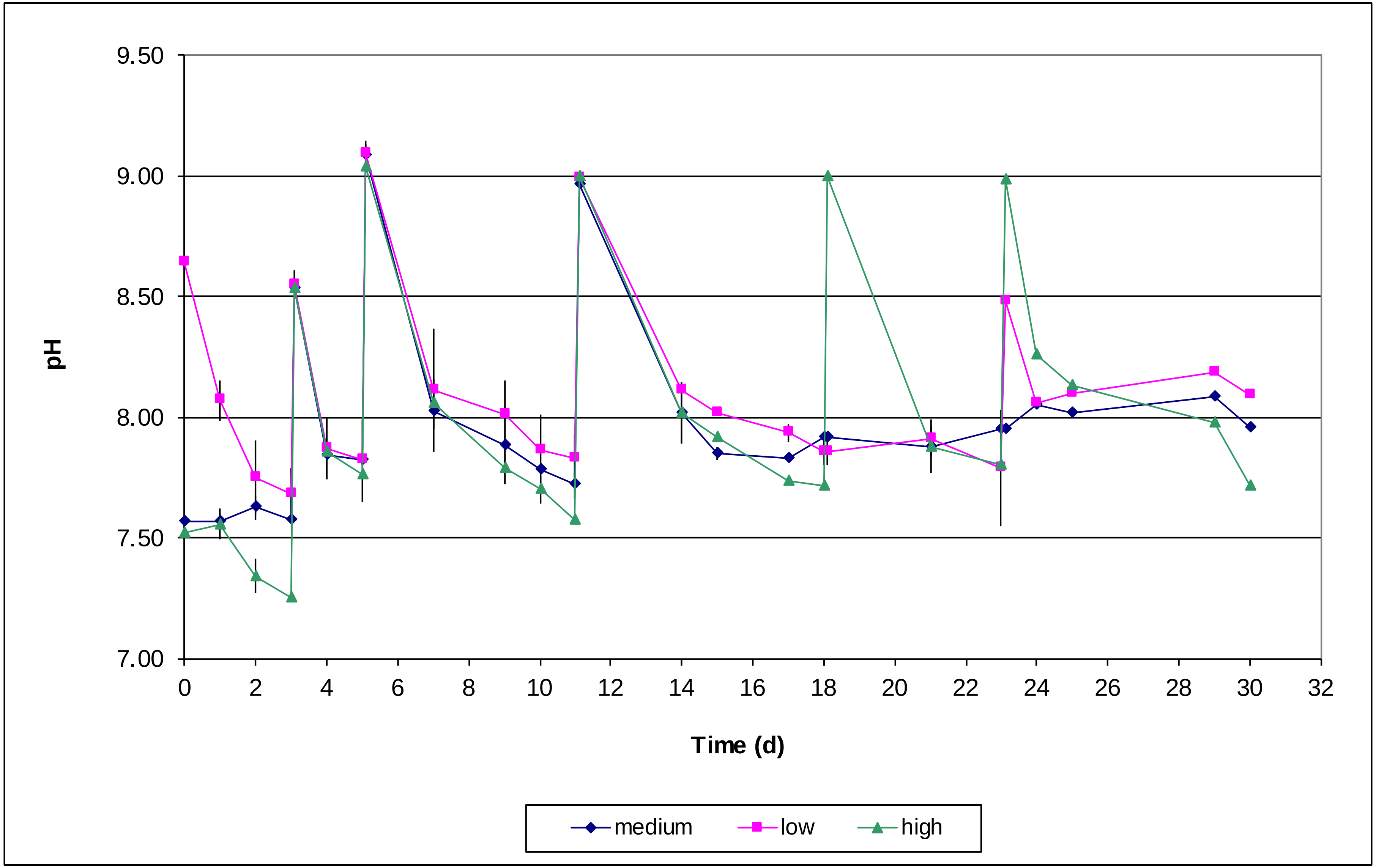
Ammonia recovery from livestock manure using gas-permeable membrane module and concentrator tank (Closed loop system).



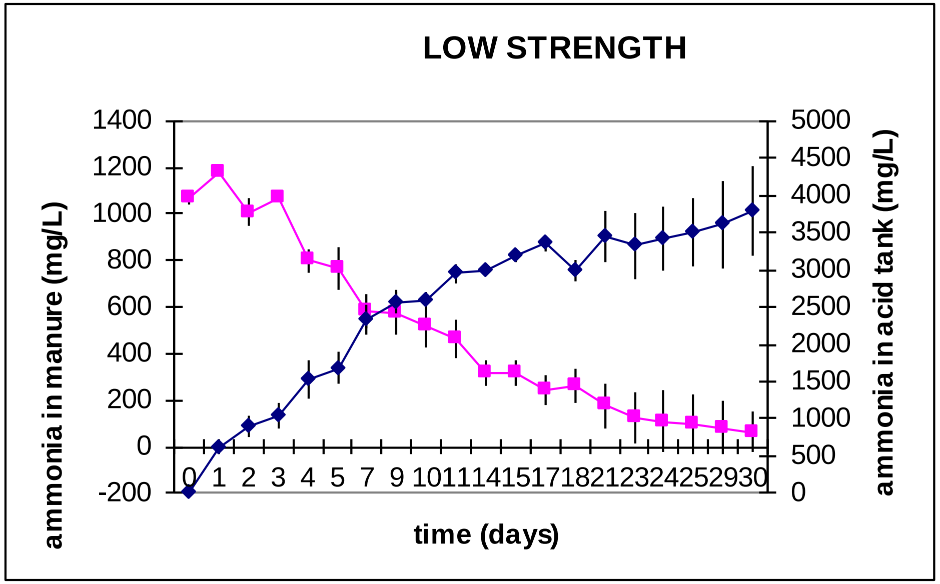
Experimental device for ammonia capture from manure using gas-permeable membranes (closed loop).

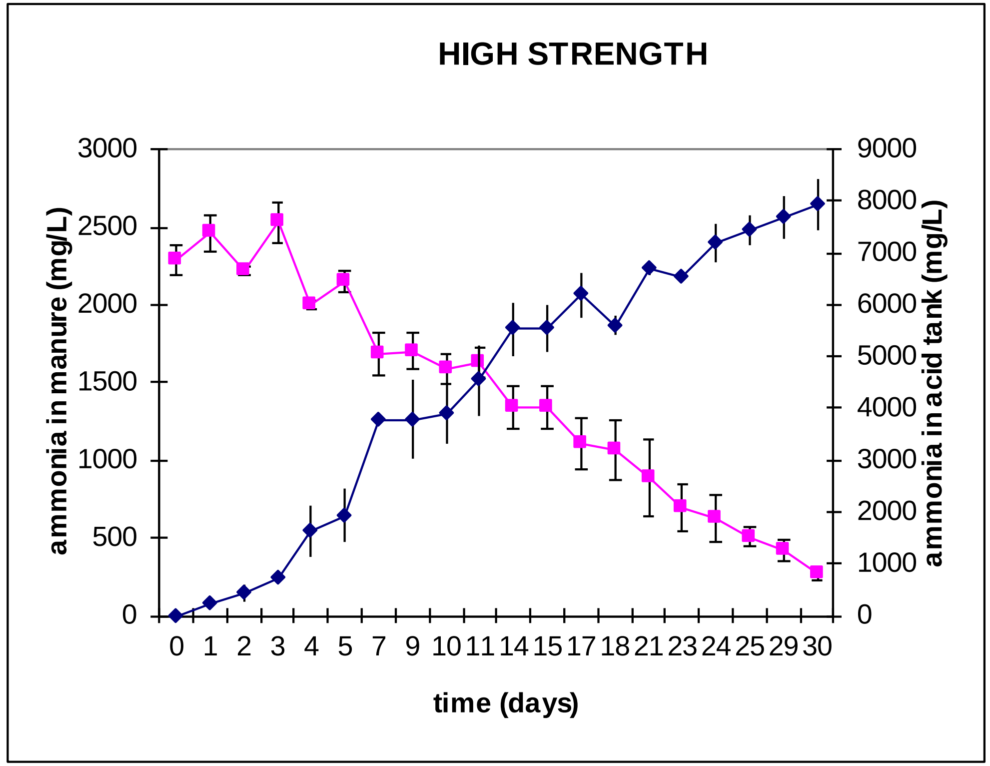


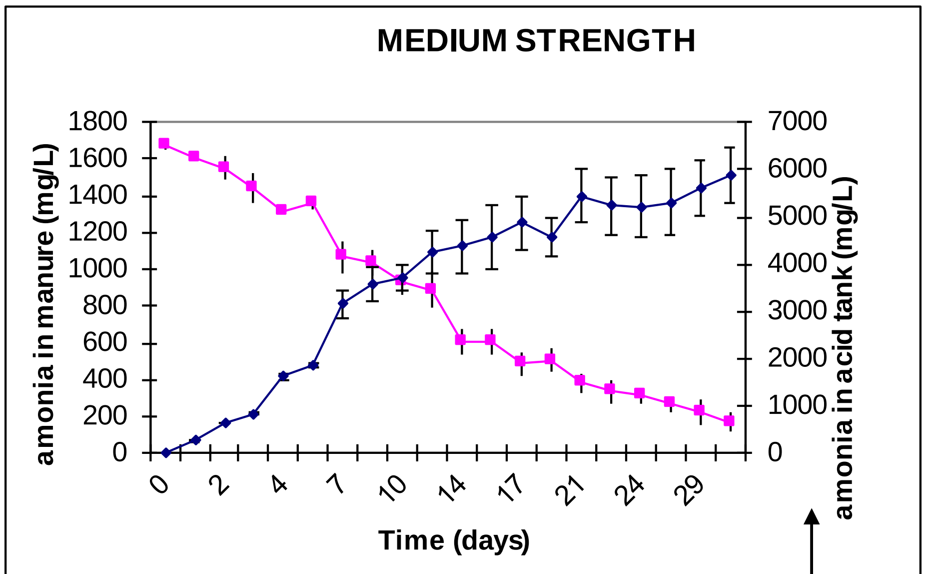
Process pH adjusted with alkali (7.7 to 9)



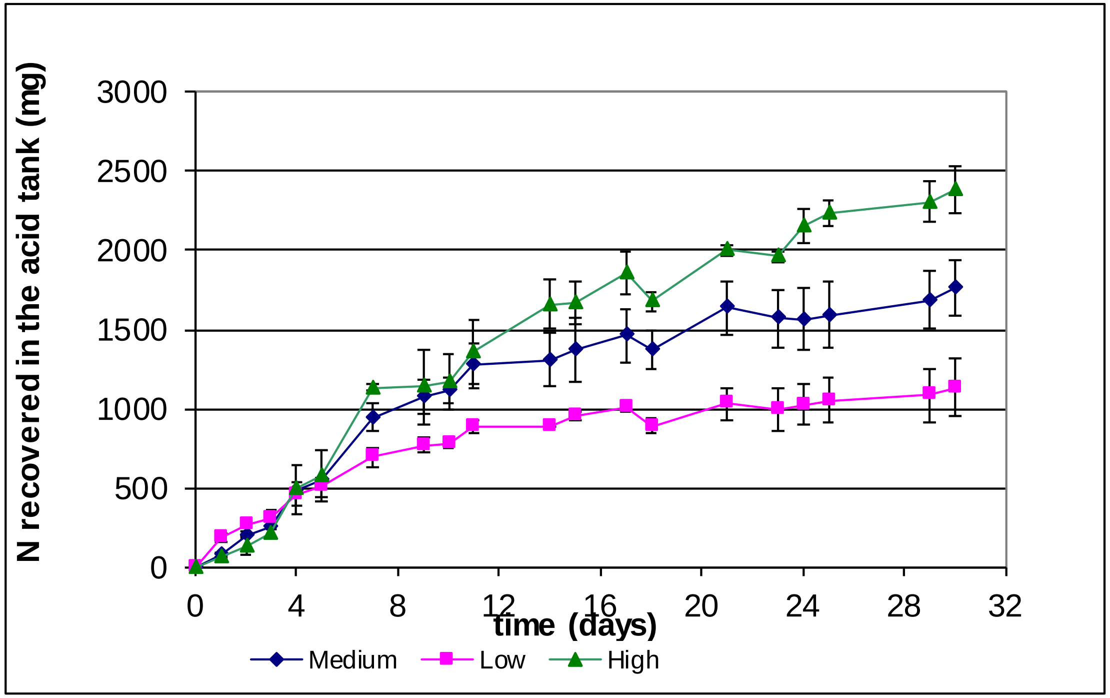
Removal of ammonia in the manures and recovery in the acid tank







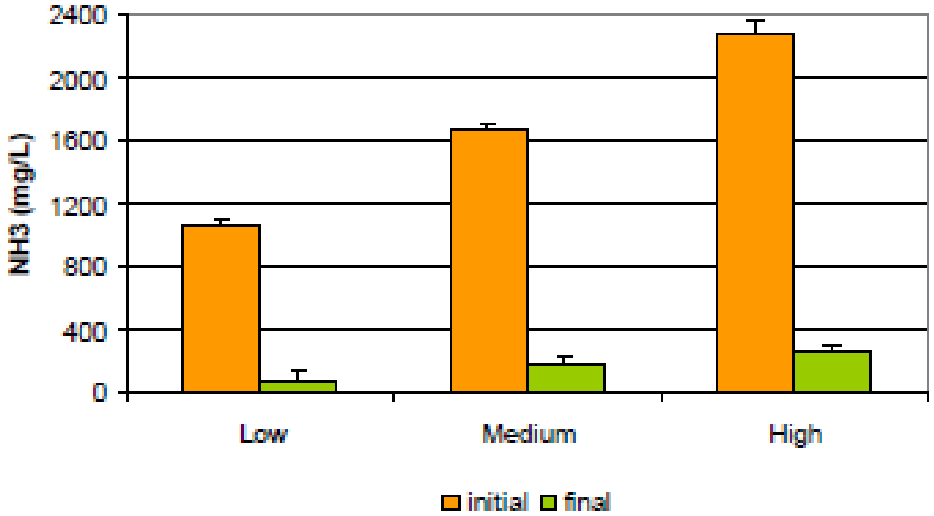
Ammonia recovery rate increases with manure strength



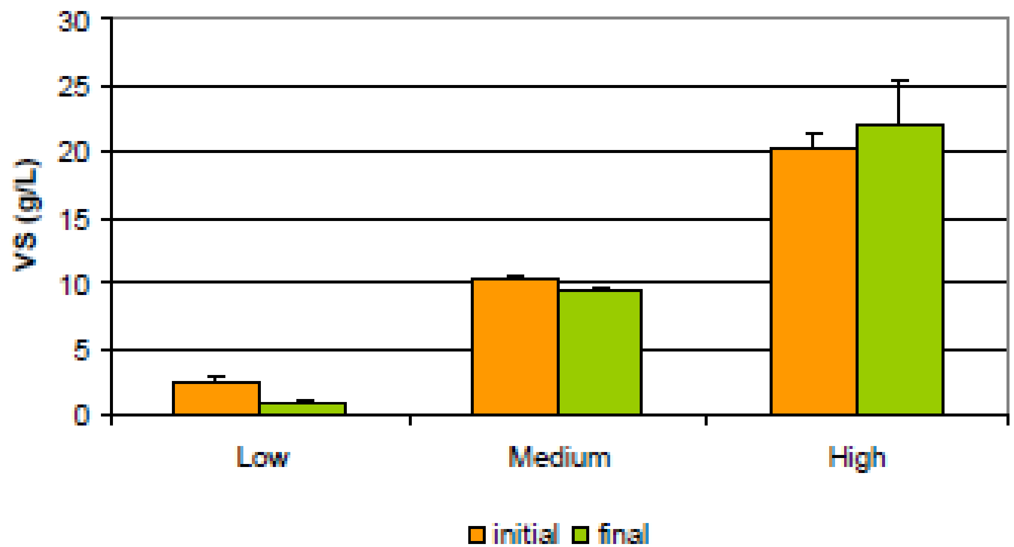
Manure strength	Initial NH4 mg N/L	NH4 removed %	NH4 recovery %	NH4 recovery rate (mg/L/d)
low	1385	94	87	74
medium	2184	90	90	92
high	2971	88	90	194

M.C. Garcia and M.B. Vanotti, Waste Management 2014 (in press)

Ammonia was removed

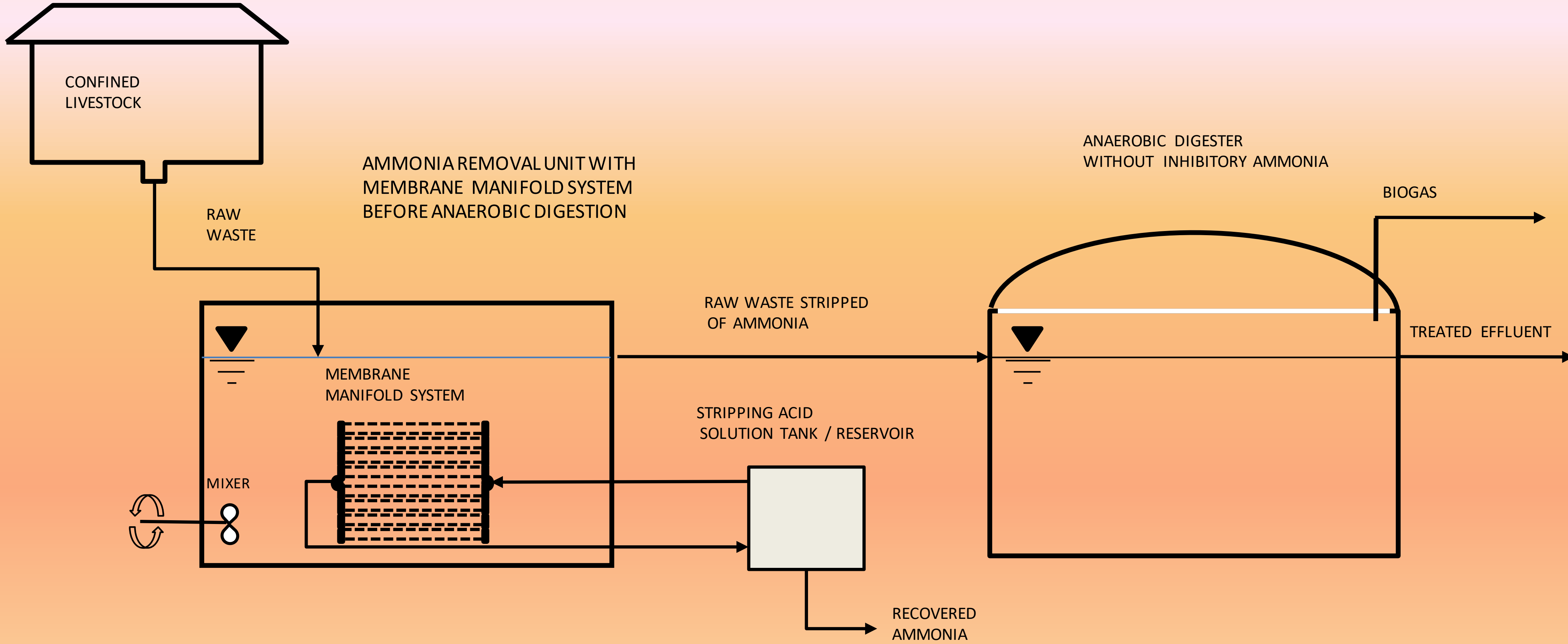


but carbon (volatile solids) was not removed



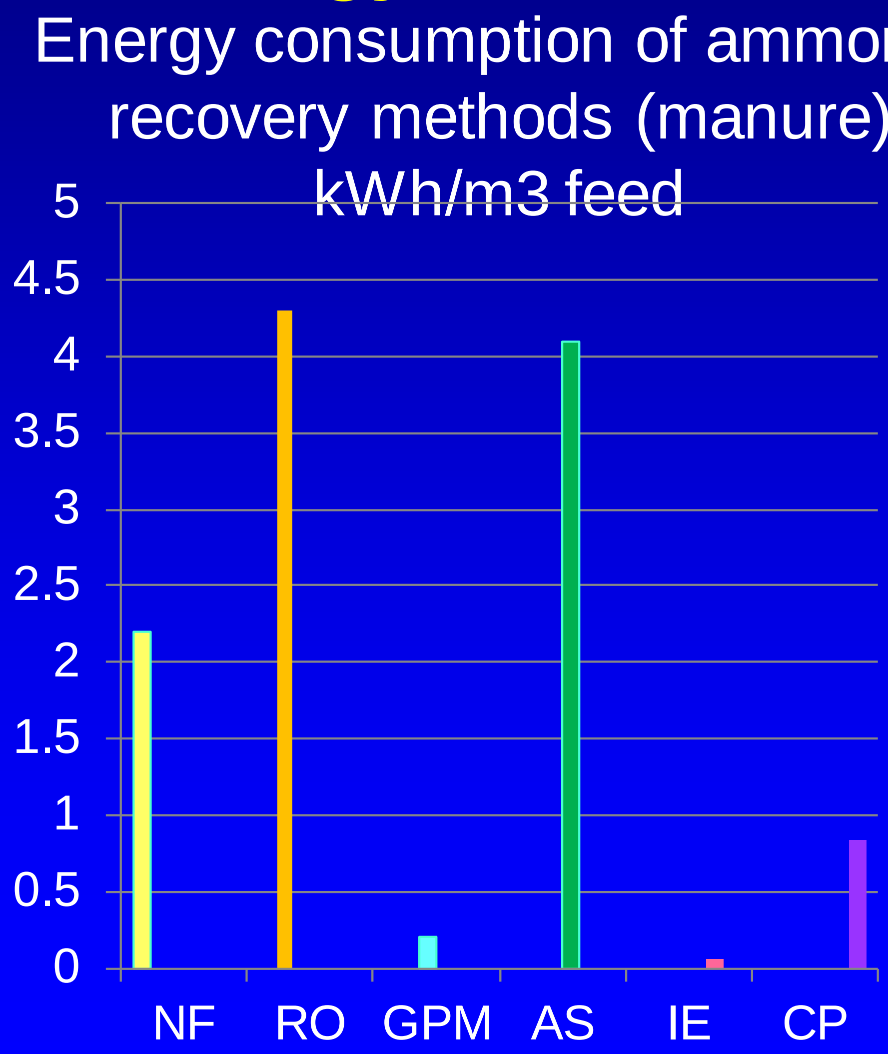
Technology can be combined with <u>anaerobic digestion</u> to recover both the ammonia and the energy from manure.

Ammonia Recovery System with Anaerobic Digestion



Zarebska et al. (2015) Ammonium fertilizers production from manure: A critical review.

The gas-permeable membrane method had very low energy demand



NF= nanofiltration

RO = reverse osmosis

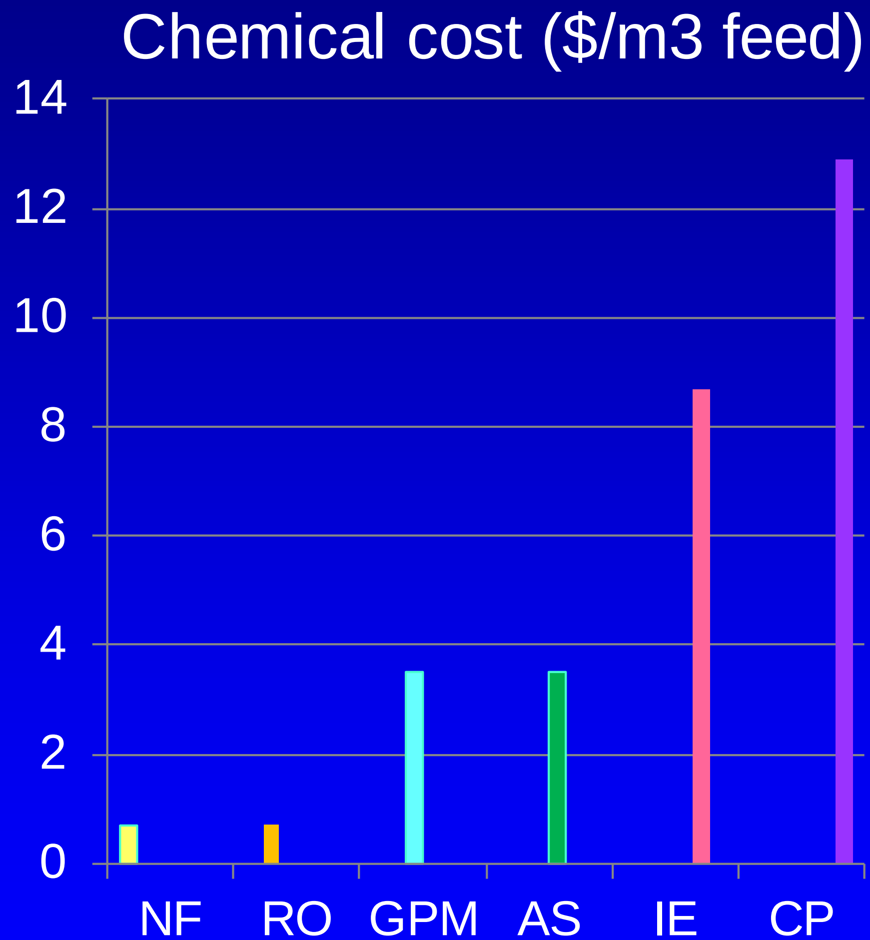
GPM = gas permeable memb.

AS = air stripping

IE = ion exchange/ zeolites

CP = Chemical precipitation

The gas-permeable membrane method (MD) had high chemical demand (NaOH to increase pH)



NF= nanofiltration

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IE = ion exchange/ zeolites

CP = Chemical precipitation

Zarebska et al. (2015) Ammonium fertilizers production from manure: A critical review.

Design Parameter: Effect of aeration

Two ways can be used to increase manure pH and N recovery efficiency by the gas-permeable membrane system:

1. **Add alkali chemicals (OH^-)**

2. **Low-rate aeration**

$$HCO_3^- + \text{air} \rightarrow OH^- + CO_2$$

$$NH_4^+ + OH^- \rightarrow NH_3 + H_2O$$

Design Parameter: Effect of aeration

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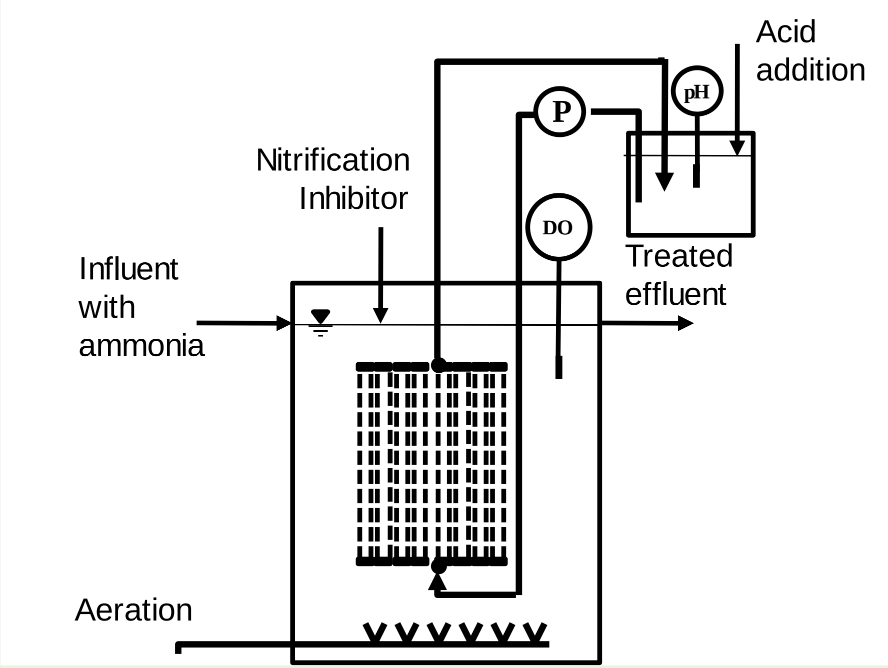
$$HCO_3^- + air \rightarrow OH^- + CO_2$$
$$NH_4^+ + OH^- \rightarrow NH_3 + H_2O$$

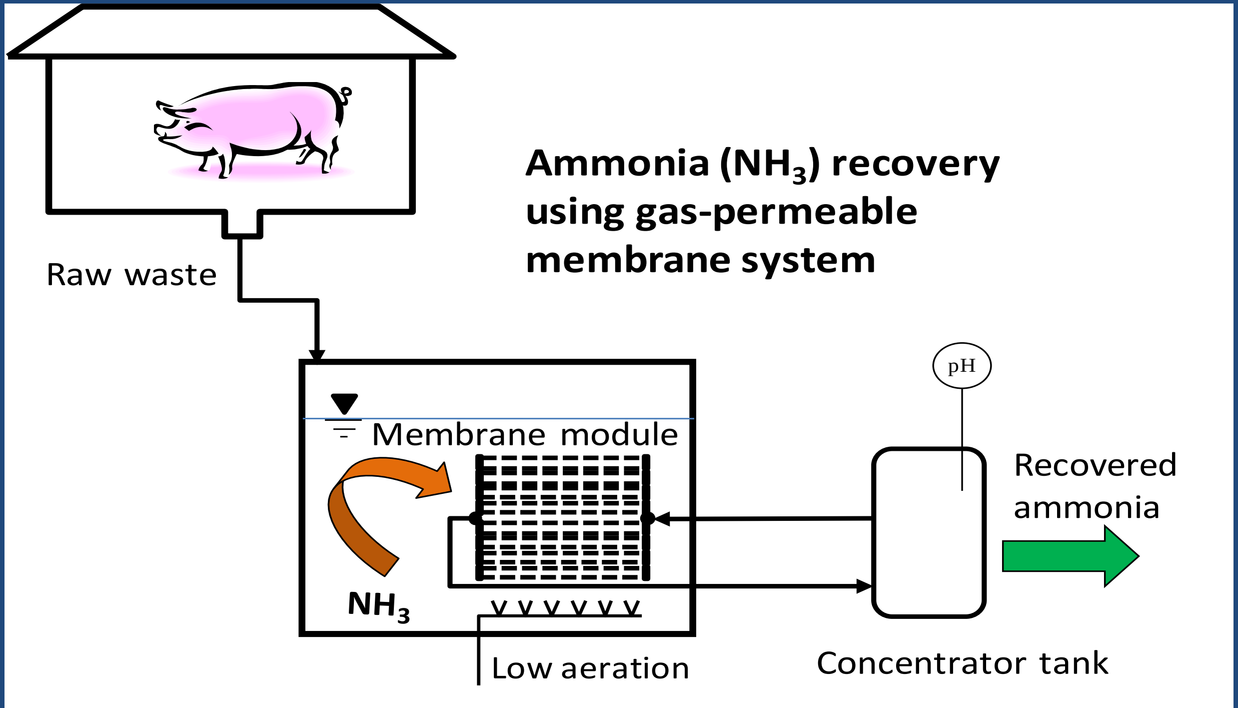


- **Aeration increases manure pH about 1 unit**
- **The aeration rate must be low to inhibit nitrification**
- **Nitrification inhibitor can be used (< 10 ppm)**

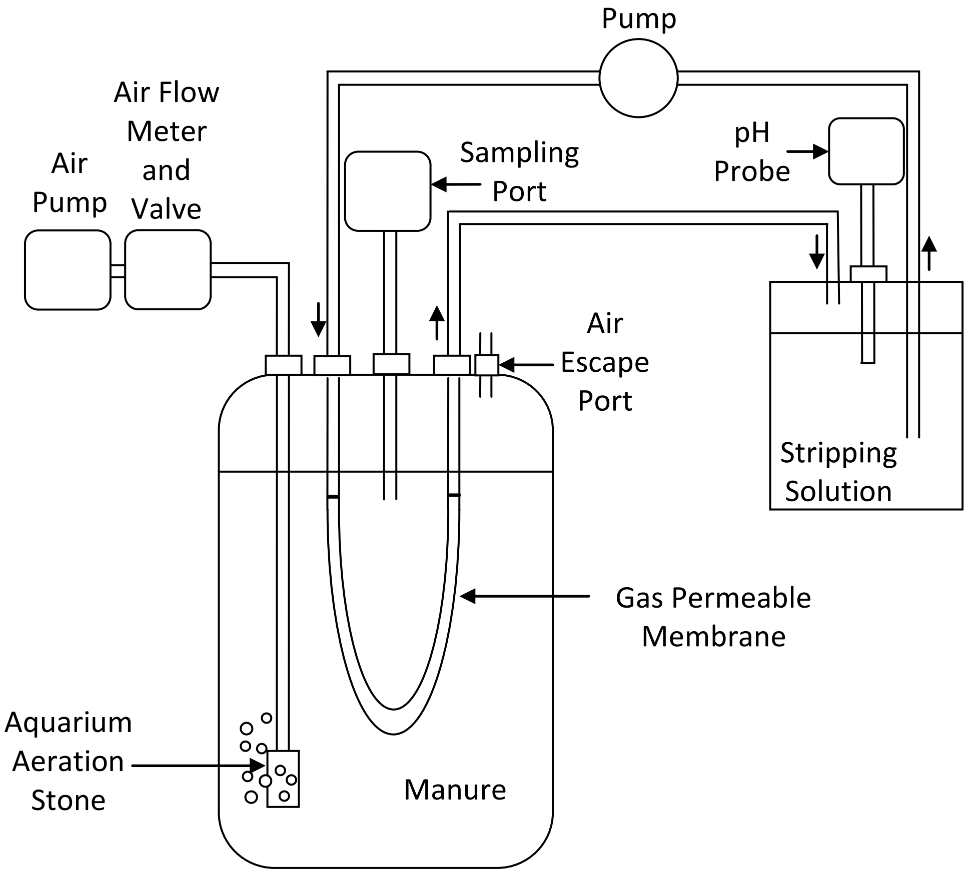
Vanotti and Szogi. US 9,005,333

Recovery of Ammonia from Liquid Manure with Gas-permeable Membranes

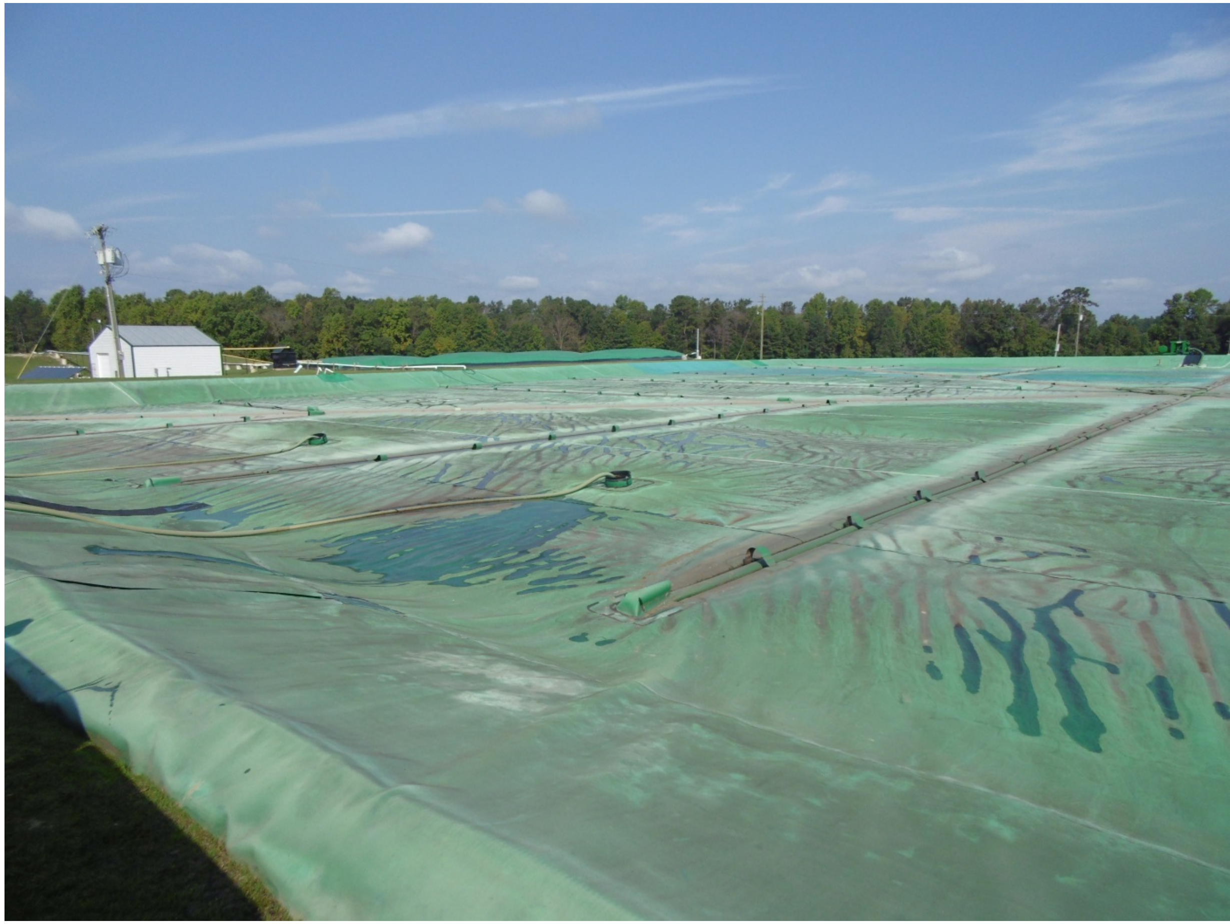
- **T**echnology recovers ammonia from liquid manure
- Produces liquid fertilizer with > 50,000 ppm nitrogen
- US Patent in 2015: "Systems and Methods for Reducing Ammonia Emissions from Liquid Effluents and for Recovering the Ammonia" (US 9,005,333, Vanotti,M.B., and Szogi,A.A)



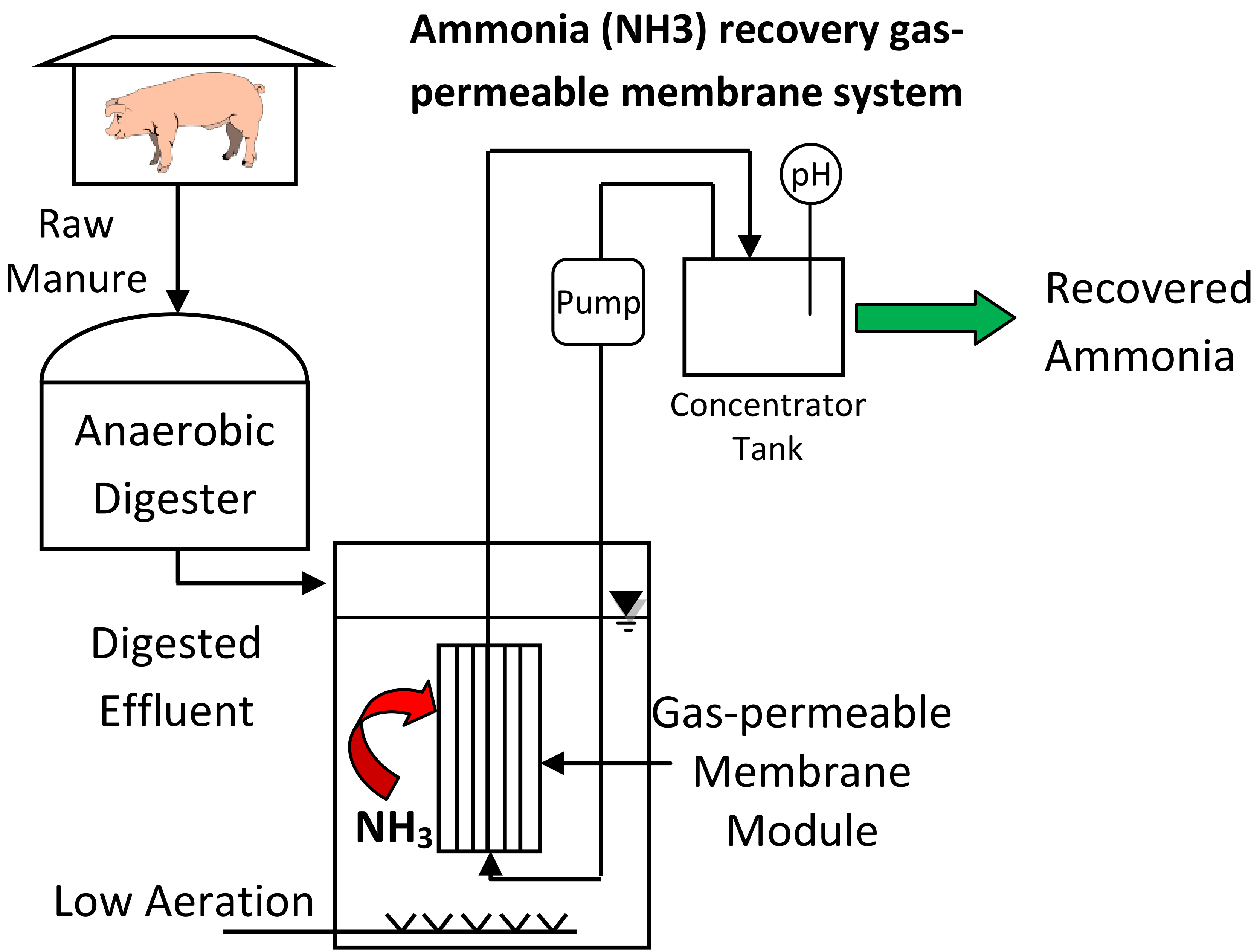
Experimental device for ammonia capture from manure using gas-permeable membranes (closed loop).



N recovery: Effect of low-rate aeration Covered lagoon effluent, North Carolina

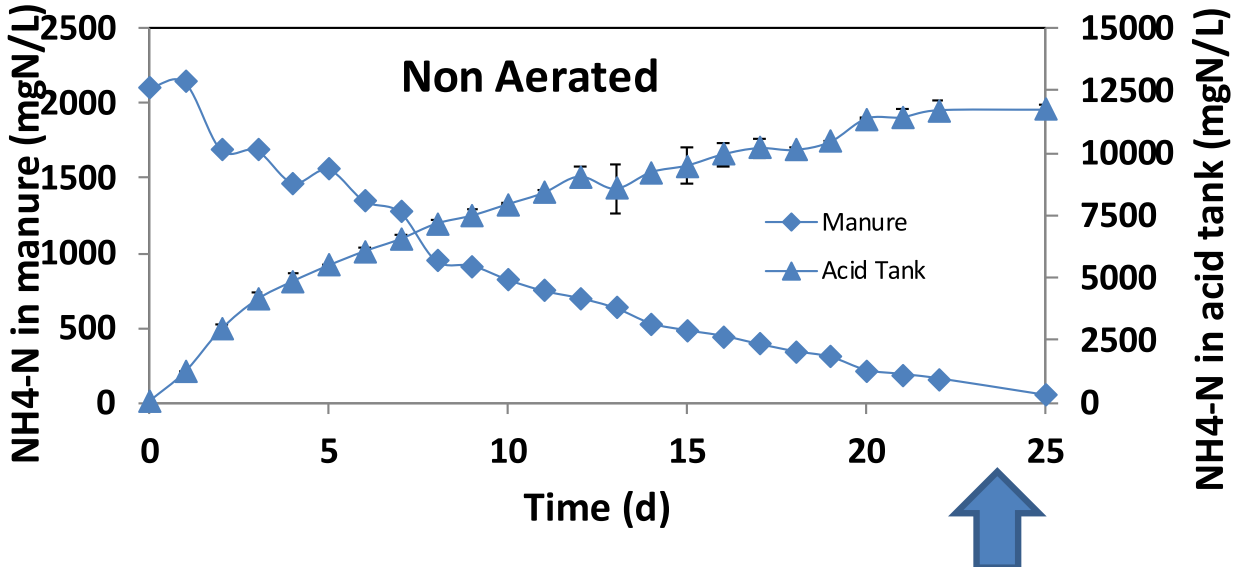


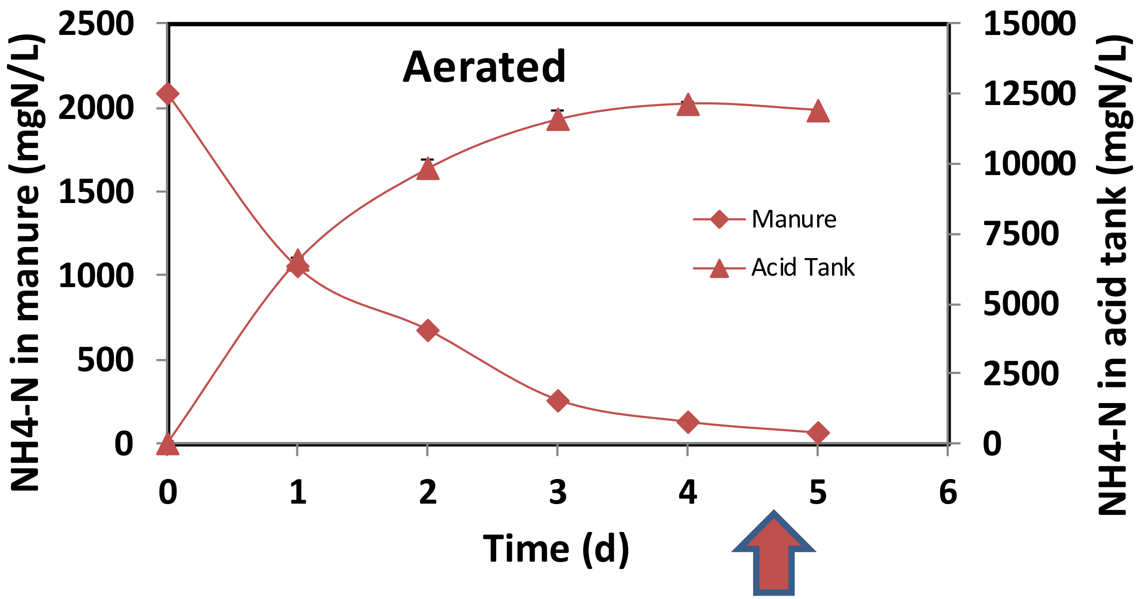
Liquid with 2,100 mg/L NH4-N


Ammonia (NH3) recovery gas-permeable membrane system
Raw Manure
Anaerobic Digester
Digested Effluent
Pump
pH
Concentrator Tank
Recovered Ammonia
NH_3
Gas-permeable Membrane Module
Low Aeration

Changes in ammonia concentration in manure and the N recovery tank

Covered anaerobic lagoon effluent, NC





Mass Balances of the Recovery of Ammonia - anaerobic digester effluent

	Treatment Time	Initial NH_4^+ in Manure	Remaining NH_4^+ in Manure	NH_4^+ removed from Manure	NH_4^+ recovered in acidic solution	NH_4^+ removal efficiency	NH_4^+ recovery efficiency	NH_4^+ Volatilized in air
	(days)	**---------------------------mg N----------------------**				**----------------%-----------------**		
Aerated	5	3133 (151)	96 (29)	3037	2979 (2)	97	98	2
Non Aerated	25	3157 (132)	71 (19)	3086	2936 (40)	98	95	5

Contents lists available at ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

ELSEVIER

Enhancing recovery of ammonia from swine manure anaerobic digester effluent using gas-permeable membrane technology

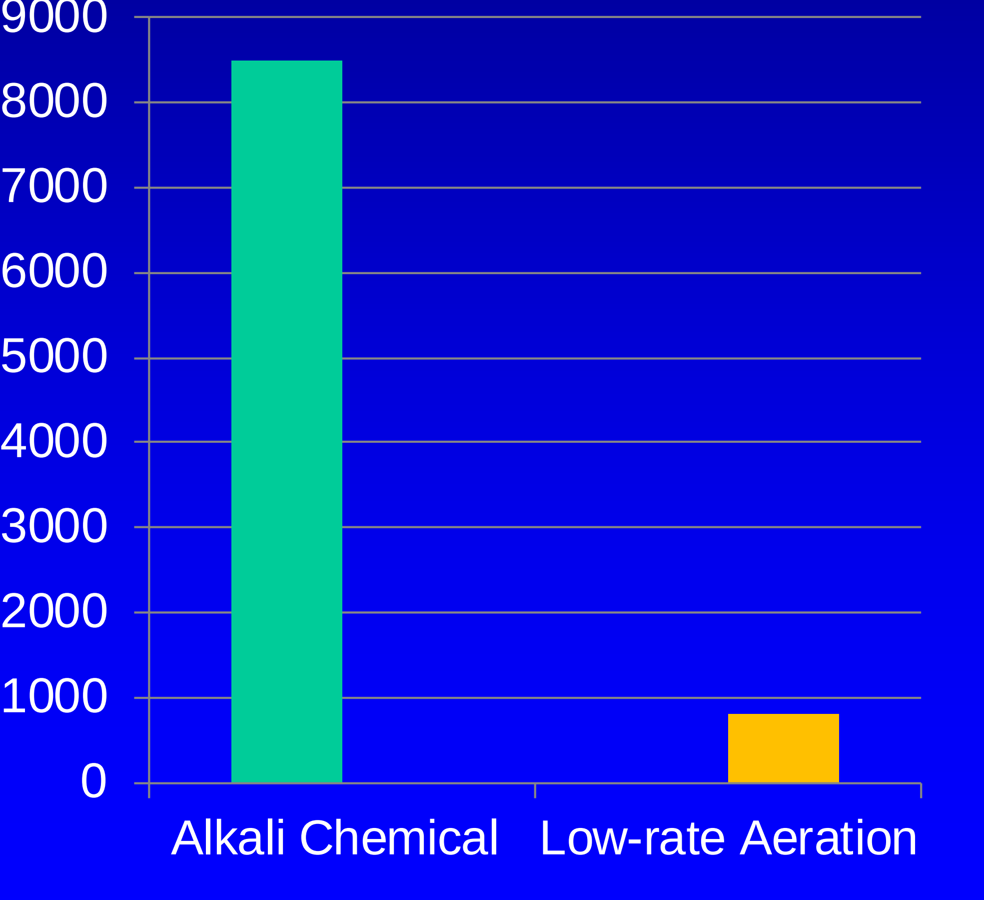
P.J. Dube [a,*], M.B. Vanotti [a], A.A. Szogi [a], M.C. García-González [b]

[a] United States Department of Agriculture, Agricultural Research Service, Coastal Plains Soil, Water and Plant Research Center, 2611 W. Lucas St, Florence, SC 29501, USA
[b] Agriculture Technological Institute of Castilla and Leon (ITACyL), Valladolid, Spain

ARTICLE INFO ABSTRACT

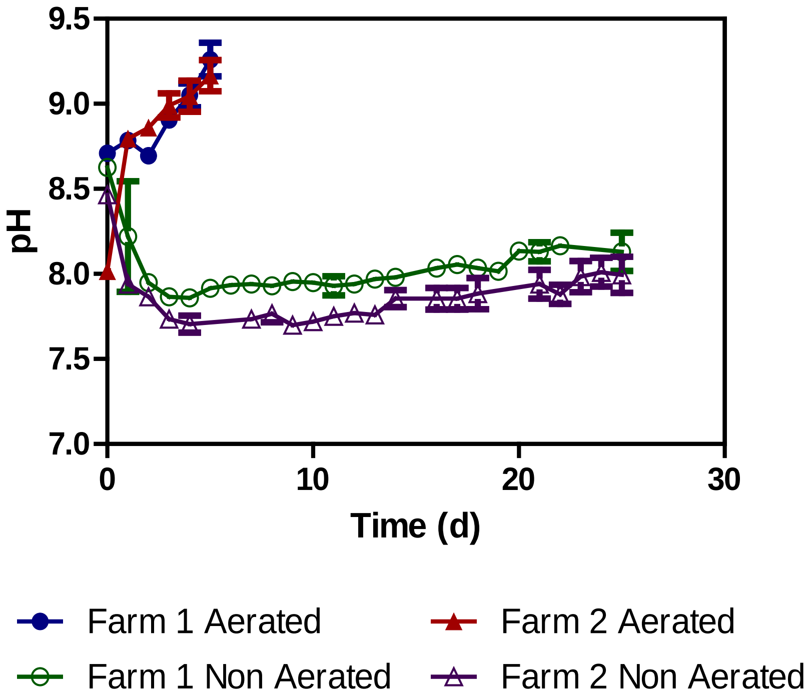
Significant cost reductions can be achieved with new concepts and research

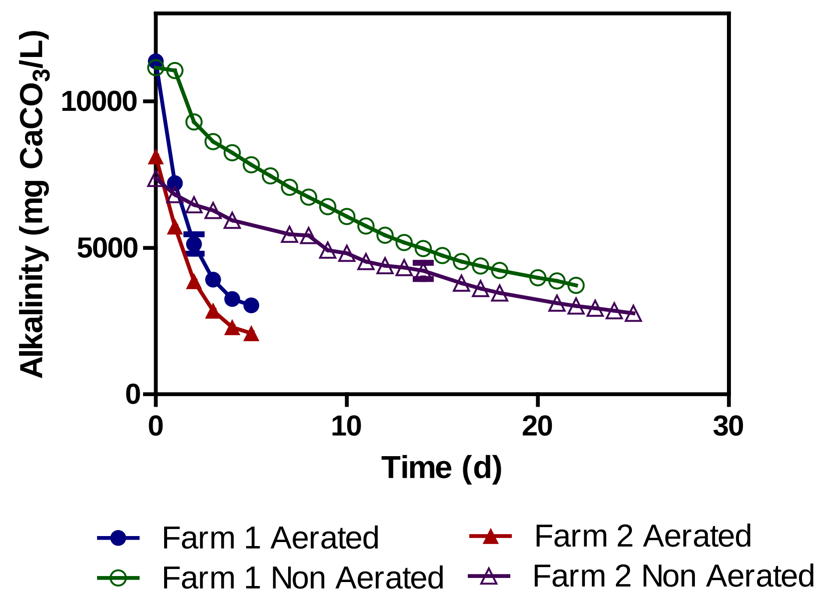
Operational cost of NH_3 recovery using gas-permeable membranes ($/4000 pigs/year)



Changes in pH and alkalinity of manure during N recovery process

Covered anaerobic lagoon effluent, NC





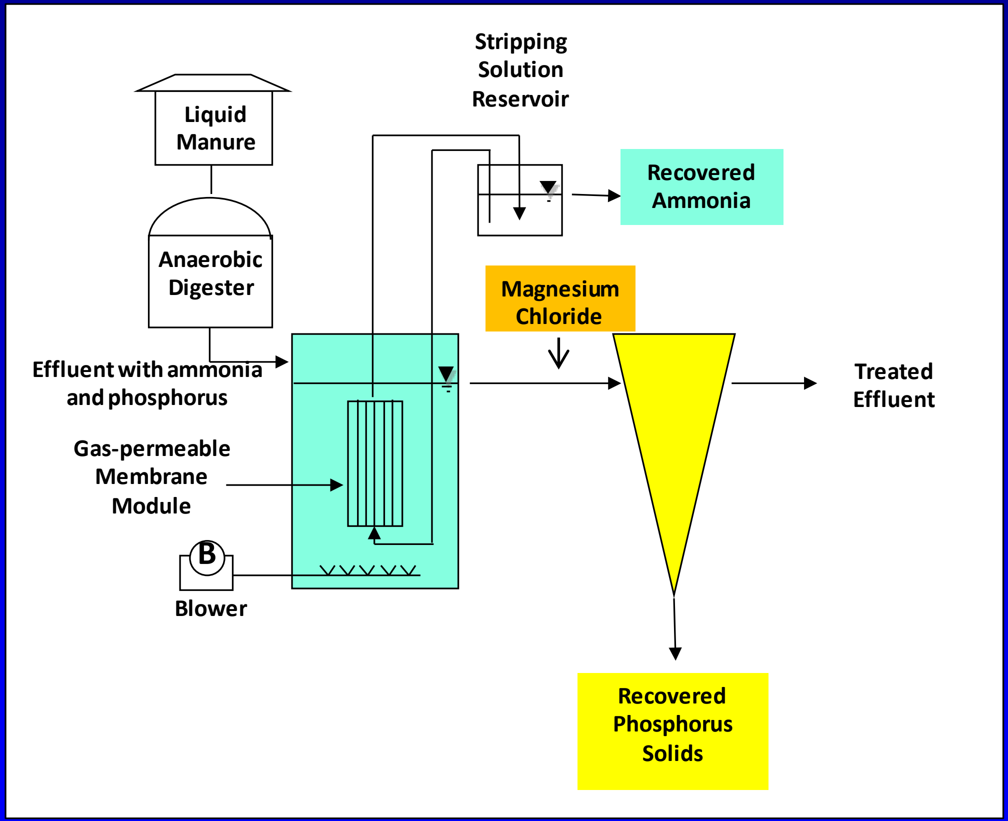
Key finding

- The process removes ammonia and alkalinity and increases pH.

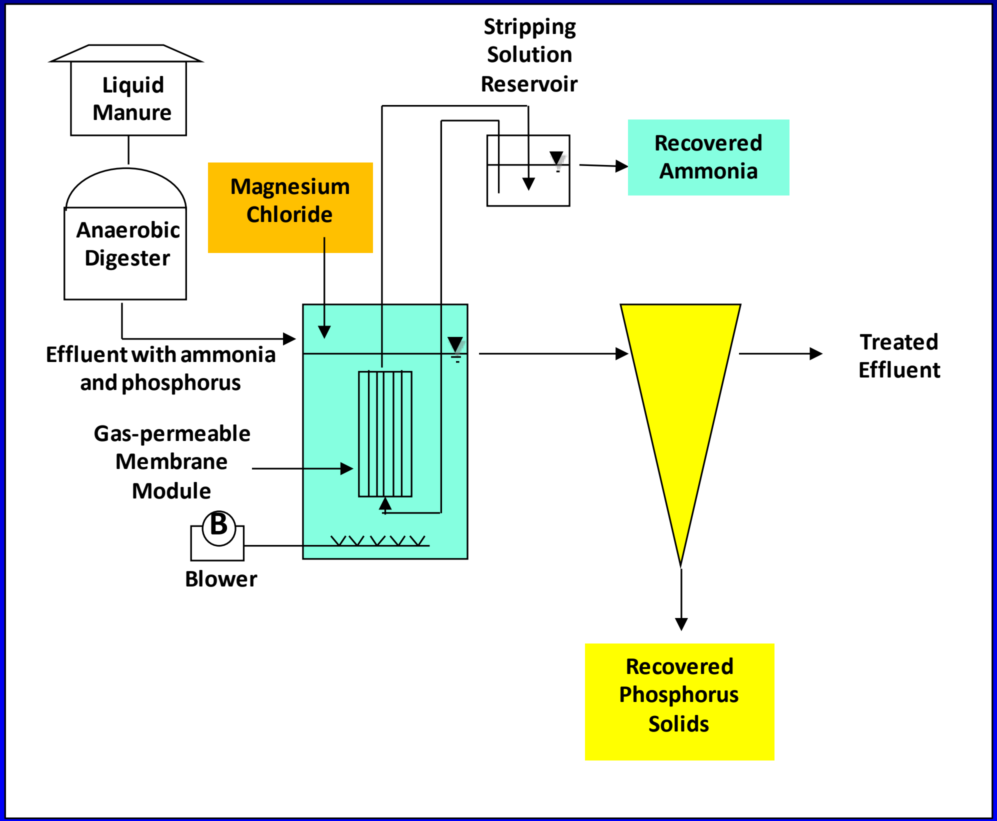
- These are ideal conditions for phosphorus precipitation and recovery

Recovery of ammonia and phosphorus from animal manure

Configuration 1



Configuration 2



Influent P concentration: 150-200 mg/L

Influent N concentration: 1500-2000 mg/L

US Pat. Appl. 62/169,387 (USDA 6/1/2015)

For Mg phosphates, two potential forms that can precipitate in liquid systems that contain Mg^{2+}–NH_4^+–PO_4^{3-} and a high Mg/Ca ratio are struvite and newberyite

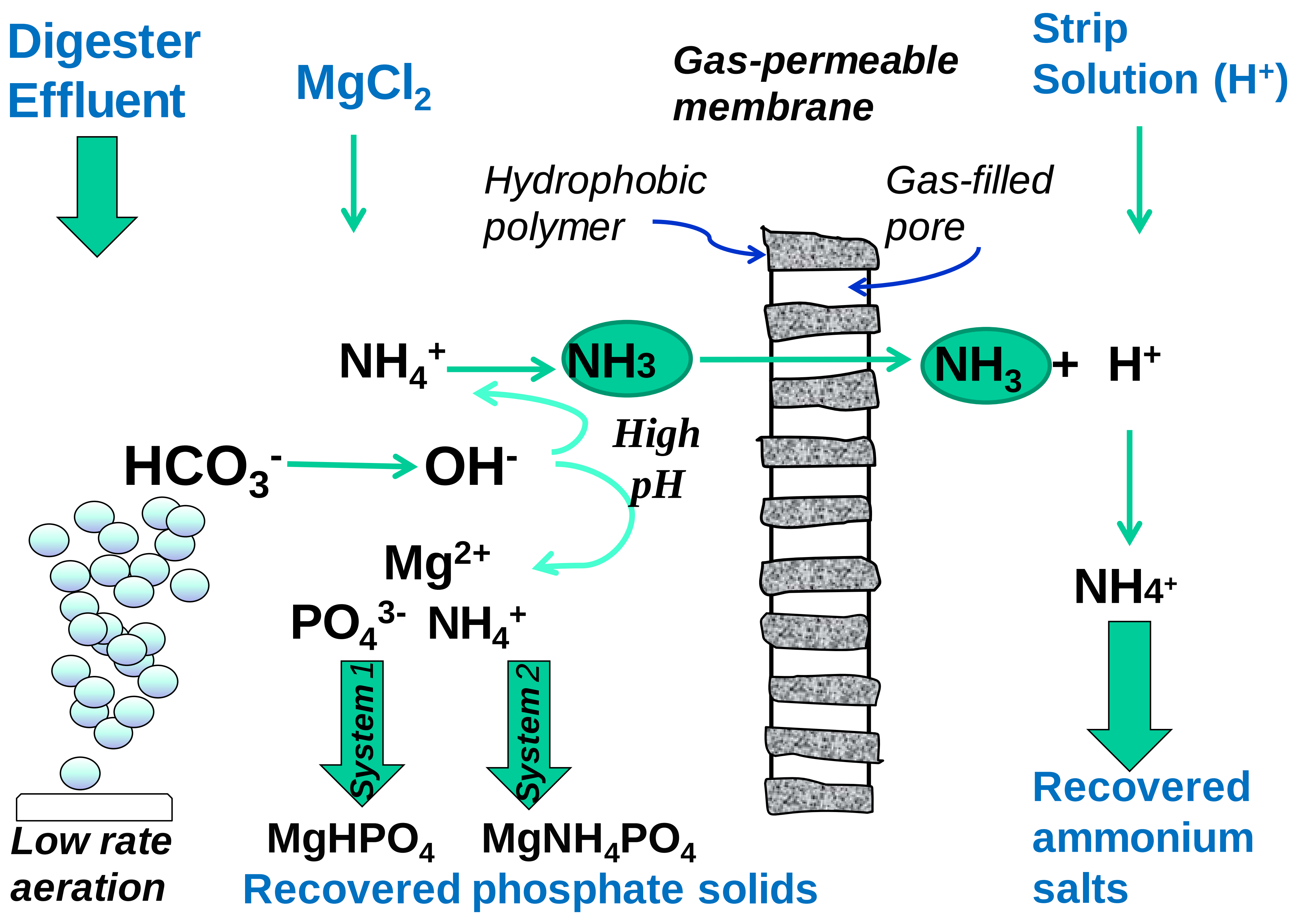
(Boistelle et al., 1983; Abbona et al., 1988; Muster et al., 2013).

Struvite

$Mg^{2+} + H_2PO_4^- + NH_3 \rightarrow \mathbf{MgNH_4PO_4} + H^+$

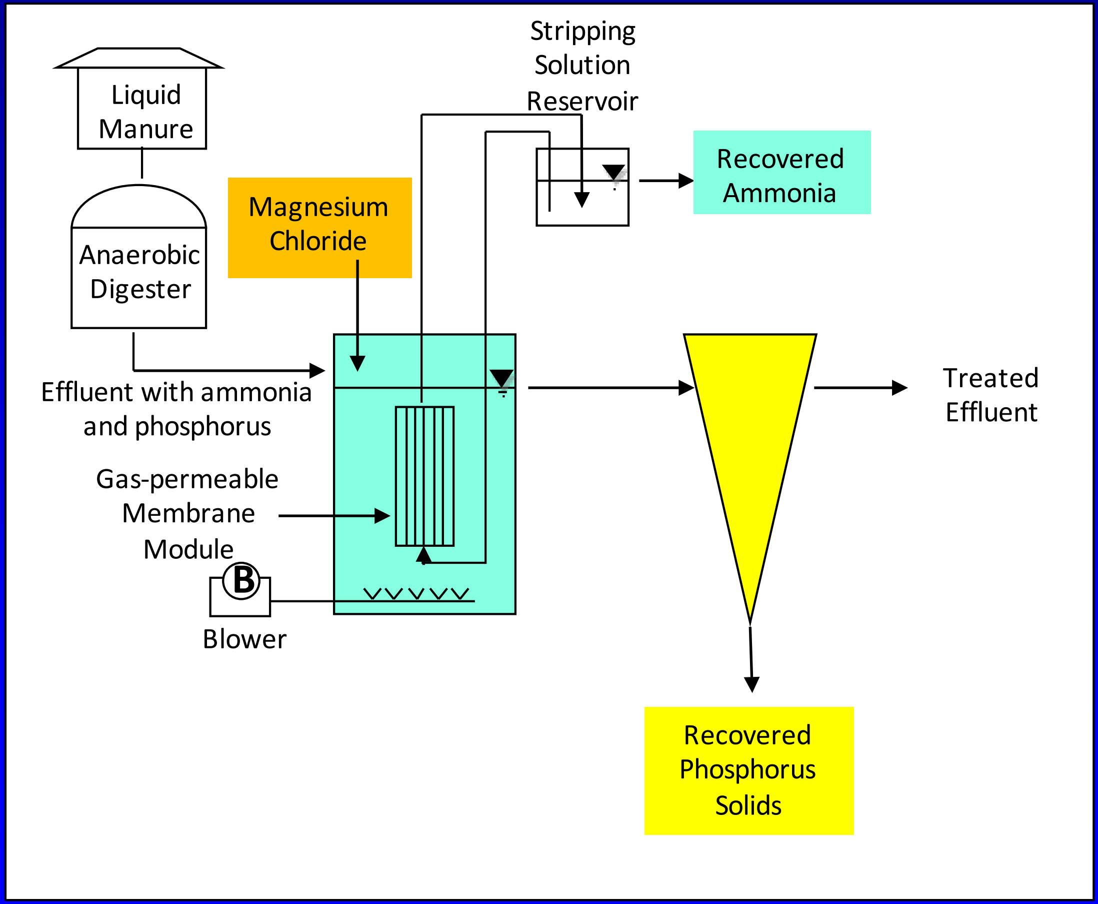
Newberyite

$Mg^{2+} + H_2PO_4^- \rightarrow \mathbf{MgHPO_4} + H^+$



Digester Effluent
MgCl2
Gas-permeable membrane
Strip Solution (H+)
Hydrophobic polymer
Gas-filled pore
NH4+
NH3
NH3 + H+
HCO3-
OH-
High pH
Mg2+
PO4 3- NH4+
System 1
System 2
NH4+
Low rate aeration
MgHPO4
MgNH4PO4
Recovered phosphate solids
Recovered ammonium salts

Nitrogen and Phosphorus Recovery Configuration 2: MgCl2 added to N reactor (no alkali added)



Influent P = 446 mg/L
Influent pH =8.4

pH after aeration = 9.5
N recovery = 91%
P recovery = 100%

Configuration 2 with MgCl2 added (without NaOH)

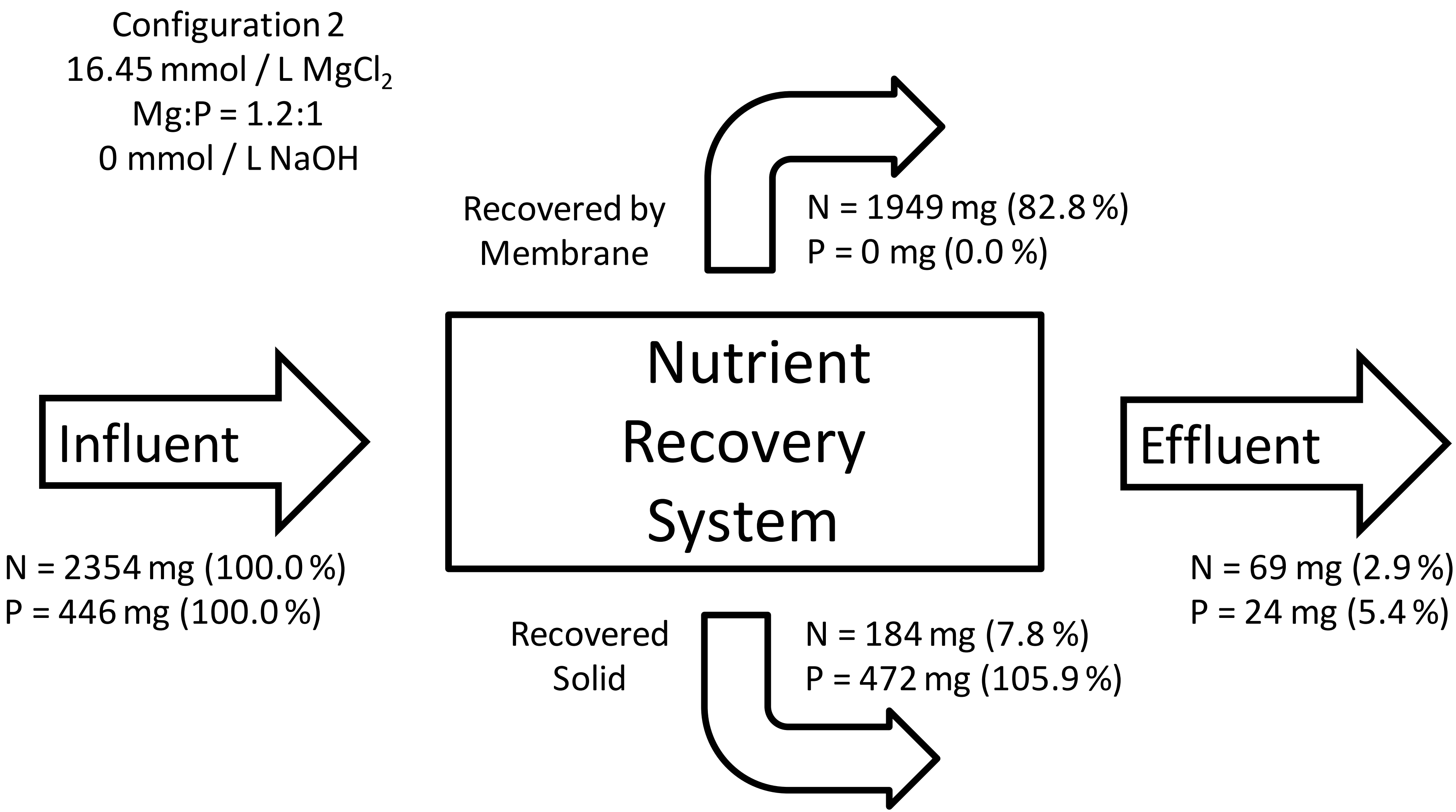
	Concentrations		MASS BALANCE				
Nutrient	**Influent Concentration**	**Effluent Concentration**	**Initial Manure**	**Recovered Solid**	**Recovered by Membrane**	**Effluent**	**Total Recovery**
	mg/L		**Percentages**				
N	2354	69.2	100%	7.7%	83.1%	2.9%	90.5%
P	446	23.5	100%	104.3%	0%	5.3%	104.3%

Configuration 2 with MgCl2 added (without NaOH)

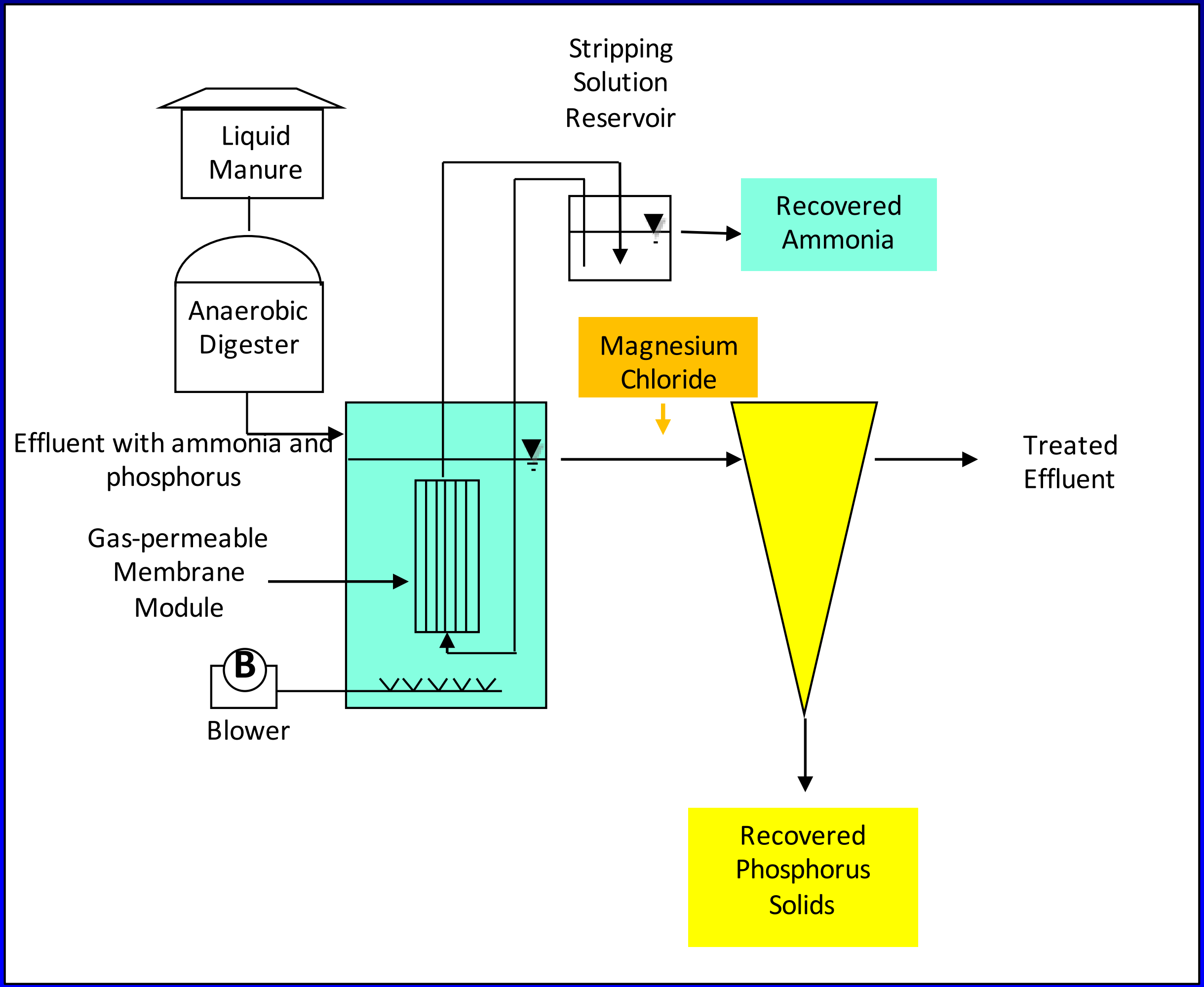
	Concentrations		MASS BALANCE				
Nutrient	Influent Concentration	Effluent Concentration	Initial Manure	Recovered Solid	Recovered by Membrane	Effluent	Total Recovery
	mg/L		Percentages				
N	2354	69.2	100%	7.7%	83.1%	2.9%	**90.5%**
P	446	23.5	100%	104.3%	0%	5.3%	**104.3%**

Composition of Recovered Solid					
N	P_2O_5	Mg	Ca	K	Plant Available P (Citrate soluble)
Percentages, %					
4.5	**26.4**	**10.0**	2.0	1.7	99.00

Struvite = 5.7 N : 29 P_2O_5 : 10 Mg


Configuration 2
16.45 mmol / L $MgCl_2$
Mg:P = 1.2:1
0 mmol / L NaOH
Recovered by Membrane
N = 1949 mg (82.8 %)
P = 0 mg (0.0 %)
Influent
Nutrient Recovery System
Effluent
N = 2354 mg (100.0 %)
P = 446 mg (100.0 %)
N = 69 mg (2.9 %)
P = 24 mg (5.4 %)
Recovered Solid
N = 184 mg (7.8 %)
P = 472 mg (105.9 %)

Nitrogen and Phosphorus Recovery Configuration 1: MgCl2 added after N removal (no alkali added)



Influent P = 446 mg/L
Influent pH =8.4

pH effluent after N recovery = 9.3
P recovery = 93.2%

US Pat. Appl. 62/169,387 (USDA 6/1/2015)

Recovered Phosphates (Configuration 1)

- P recovered as High-Grade Magnesium Phosphate
- 99.7% plant available (standard citrate test)

Chemical Composition

Constituent	Percentage
P_2O_5	46.4%
Magnesium	17.1%
Calcium	0.4 %
Potassium	1.7 %
Nitrogen	1.8 %

Newberyite ($MgHPO_4.3H_2O$)
41% P_2O_5 and 14% Mg

Triple superphosphate = 46% P_2O_5; Rock phosphate = 27-36% P2O5

Configuration 1 with Municipal Side Stream Wastewater (after AD of sludges)

	Concentrations		MASS BALANCE				
Nutrient	Influent Concentration	Effluent Concentration	Initial Manure	Recovered Solid	Recovered by Membrane	Effluent	Total Recovery
	mg/L		Percentages				
N	731	123	100%	2.4%	90.5%	16.7%	**92.3%**
P	147	6	100%	79.2%	0%	4.1%	**79.2%**

- Results obtained were **consistent** using **swine and municipal side-stream** digester effluents
  - Composition similar to rare bio-mineral NEWBERYITE that is found in guano deposits

Composition of Recovered Phosphate Minerals (Swine Effluent)					
N	P_2O_5	Mg	Ca	K	Plant Available P
Percentages, %					
1.8	46.4	17.1	0.4	1.8	99.7

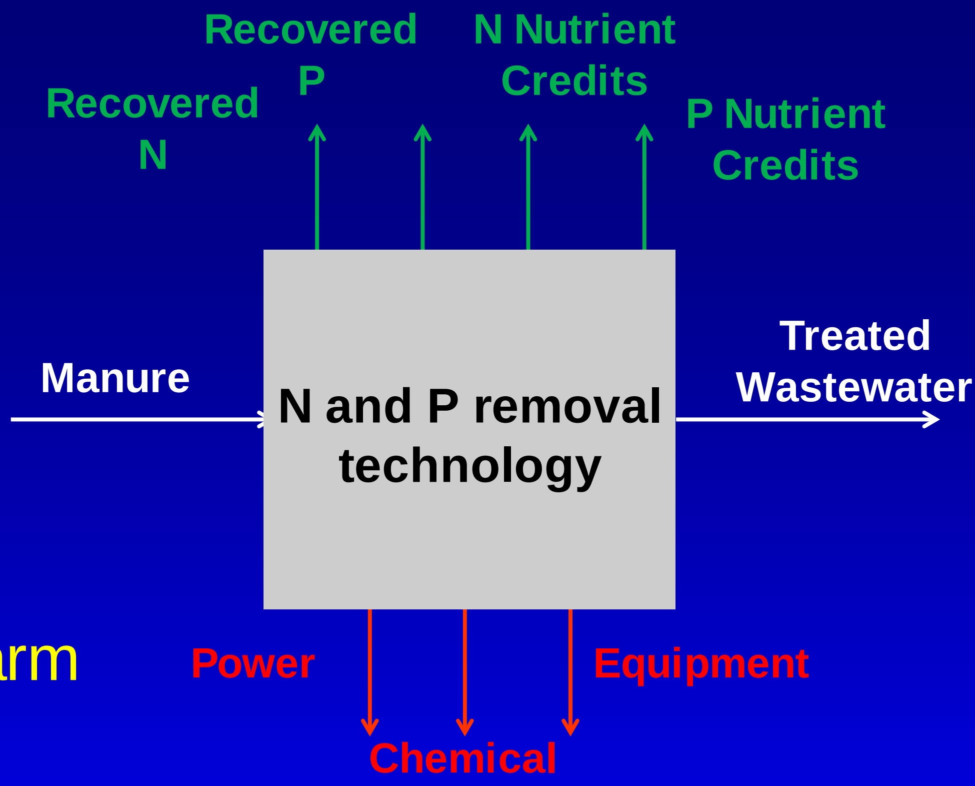
Composition of Recovered Phosphate Minerals (Municipal Centrate) James River WWTP, Virginia					
N	P_2O_5	Mg	Ca	K	Plant Available P)
Percentages, %					
2.8	44.1	13.6	0.9	0.7	98.5

Triple superphosphate = 46% P_2O_5; Rock phosphate = 27-36% P2O5

Struvite = 5.7 N : 29 P_2O_5 : 10 Mg **Newberyite 41 P_2O_5 : 14 Mg**


Municipal Centrate
P precipitate
Mg Cl2 + Na OH
without NH3
with NH3

Ecosystem Cost Map



5200-head swine farm (finishing)

Capital and Operational Costs	
Equipment, Chemical, Power	$57,168.47 / year
Potential Revenue	
Sale of fertilizer products (N & P)	$58,538.63 / year
Additional Revenue: Sale of Non-point Nutrient Credits (2:1 trading ratio)	$ 61,449.93 / year

Conclusions

- Phosphorus recovery was combined with ammonia recovery using gas-permeable membranes
- Aeration destroyed carbonates, increased pH, and enhanced N capture
- The process provided approximately 100% phosphorus recovery efficiencies
- With substantial ammonia capture, the recovered P contained very-high phosphate grade (biomineral newberyite)

USDA Agricultural Research Service

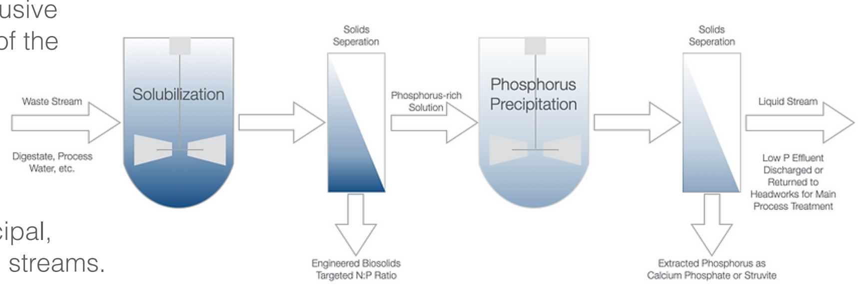
https://www.ars.usda.gov/southeast-area/florence-sc/coastal-plain-soil-water-and-plant-conservation-research/

ATTACHMENT 68

QUICK WASH®

Phosphorus Extraction & Recovery

Renewable Nutrients is rethinking phosphorus with Quick Wash®, a proprietary system that draws on exclusive technology to extract and recover 95% or more of the phosphorus found in biosolids.



Quick Wash is equally effective at treating municipal, industrial, and agricultural waste solids and side streams.

- Extracts and recovers more than 95% of phosphorus

Quick Wash® is effective at extracting and recovering 95% or more of the phosphorus from a treatment facility's solid stream or side stream.

- Alternative treatment locations

Quick Wash® is flexible in solving phosphorus issues. It is the only system that can treat your solids pre digestion, post digestion, or post dewatering.

- Reduces or eliminates phosphorus recycle load

When deployed on the side stream of thickening or dewatering operations, Quick Wash® can reduce or eliminate the recycle flow of phosphorus.

- Reduces polymer and metal salts

Quick Wash® can reduce or even eliminate a facility's reliance on costly metal salts for phosphorus removal. The system also helps to increase dewatering efficiencies and reduce polymer consumption.

- Reduces disposal costs

Quick Wash® lowers the phosphorus content of liquid and dewatered biosolids — making the dewatering process more efficient, which lowers the volume of solids for disposal. Quick Wash® can reduce or eliminate biosolids landfilling fees by making the solids more viable

- Eliminates struvite scaling

Eliminates the costly maintenance fees associated with struvite scaling in digesters and treatment pipes.

- Increases revenue

Facilities can sell the recovered phosphorus, participate in nutrient credit trading opportunities, and potentially create locally blended custom fertilizers.

- Meet EPA nutrient TMDL requirements

Quick Wash® can help facilities meet increasingly stringent nutrient loading regulations.

- Represents a new, sustainable source of phosphorus

Quick Wash® represents a new and sustainable source of phosphorus to meet accelerating demand, while providing a solution that helps curtail phosphate pollution in groundwater and waterways.

Need more details? Contact us

We are here to assist. Contact us by phone, email or via our Social Media channels.

Contact Us

ATTACHMENT 69

National Hog Farmer Article

National Hog Farmer

Renewable Nutrients announces pilot of Quick Wash process in the animal ag sector

Aug 12, 2015

Renewable Nutrients LLC, a firm focused on nutrient extraction, recovery, and reuse, today announced that it will be piloting its Quick Wash phosphorus extraction and recovery process later this month at Walk Stock Farm, a large swine production farm near Neoga, Ill.

“This will mark our first in-field operation in the animal agriculture sector, outside of small, bench scale tests in the laboratory,” says Jeff Dawson, President and chief executive officer of Renewable Nutrients. “We couldn’t be more pleased in the interest that Walk Stock Farm has expressed in finding a scalable, economical solution to recovering phosphorus from their swine waste. The Renewable Nutrients team is eager to test and prove Quick Wash’s effectiveness and practicality for on-farm operations,” Dawson adds.

Quick Wash has been successfully piloted at waste water treatment facilities throughout the mid-Atlantic, including Ephrata, Pa., Westminster, Md., and Greenville, Chapel Hill and Raleigh, N.C. The upcoming pilot at Walk Stock Farm represents the first time Quick Wash will be tested and utilized to extract and recover phosphorus from animal manure in a real-world, on-farm scenario. “We have been following Renewable Nutrients’ rollout of their Quick Wash technology within the waste water treatment industry, and we feel that it's a very promising solution for reducing and recovering the phosphorus loads in our hog manure," says Roger Walk, president of Walk Stock Farms. "If Quick Wash can extract an acceptable level of phosphorus from our manure, we feel we will have a very sustainable — perhaps future-proof — solution for managing our manure and transporting and spreading it on our crop fields. We are excited to host this first animal agriculture-specific pilot of the Quick Wash system, and we look forward to seeing it in operation and reviewing its performance," Walk adds.

Animal and plant growing operations throughout the country (specifically in areas close to or leading to fragile watersheds) are under increasing pressure to employ best farming practices that eliminate or lessen nutrient runoff from crop fields where manure has been applied. Given a tool — like Quick Wash — to extract and recover phosphorus from animal manure, farmers can spread the resultant low or no-phosphorus manure without fear of phosphorus runoff or soil saturation. Quick Wash also provides farmers with a means of selling the extracted phosphorus on the open market or engaging in the trading of nutrient credits.

"Quick Wash has surfaced as a proven and practical methodology for reducing phosphorus loads in waste water treatment plant sludge and effluent," says Mike Schmid, chief marketing officer at Renewable Nutrients. "It only makes sense to extend this new technology to the animal agriculture sector, where it can basically eliminate the occurrence of non-point source runoff of phosphorus, and provide farmers with an efficient, scalable means of extracting and recovering such a valuable nutrient from their animal waste."

LINK TO ARTICLE

ATTACHMENT 70

RENEWABLE
NUTRIENTS

Innovative Nutrient Recovery Technologies

- Renewable Nutrients is a privately owned US company currently commercializing exclusive, proprietary & patented nutrient recovery technologies in the municipal, industrial, and agricultural markets.

- Renewable Nutrients is a full service technology company with the capacity to license, sell, design, and build nutrient recovery technologies for full scale implementation

- Renewable Nutrients owns exclusive licenses for multiple USDA patents for **Phosphorus & Nitrogen** recovery technologies commercially known as Quick Wash®

RENEWABLE NUTRIENTS

- Renewable Nutrients has redefined nutrient recovery with Quick Wash, a proprietary patented system that draws on exclusive patented technology to extract and recover **Phosphorus & Nitrogen** from municipal, agricultural, and industrial markets

- Renewable Nutrients has completed numerous Quick Wash pilots and trials in municipal, agricultural, and industrial applications and has effectively proven the performance of Quick Wash

- Quick Wash technologies are cost effective and completely customizable based on the costumers needs for nutrient extraction and recovery with the ability to focus on **Phosphorus & Nitrogen individually or as a combined recovery technology.**

- Quick Wash is poised to be the solution for treatment, recovery & recycling of high strength **Phosphorus & Nitrogen** waste streams in any food waste digestate, municipal sludge or centrate, agricultural applications, and other high strength waste streams

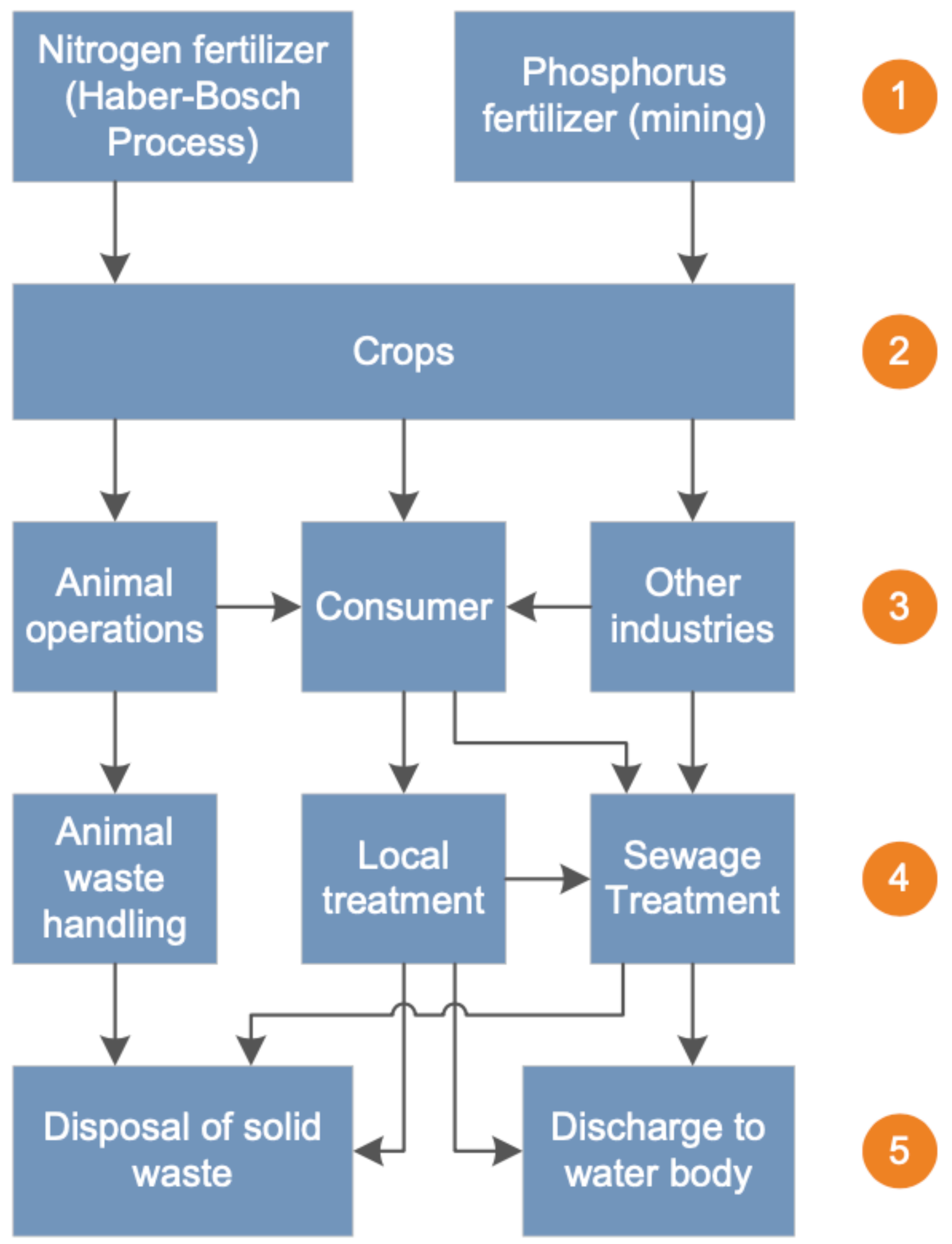
QUICKWASH®

IP Portfolio Summary

Patent	Description	Transformative Invention	Market Applicability
Phosphorus	Phosphorus extraction & recovery	**Yes** - moves facilities from simply sequestering P to biosolids to P extraction & recovery. Yields a recovered commodity (calcium phosphate)	Municipal Industrial Agriculture
Ammonia Liquid	Passive ammonia recovery from liquid waste streams	**Yes** – drives efficiency and reduces cost of nitrification/ de-nitrification operations. Yields a recovered commodity (ammonium sulfate)	Municipal Industrial Agriculture
Ammonia Gaseous	Passive ambient ammonia recovery from a gaseous ammonia-rich environment	**Yes** – potential to competley transform current costly ventilation practices in confined animal agriculature	Agriculture
Combined **Phosphorus & Ammonia**	Protects the IP when combining both phosphorus and ammonia recovery	**Yes** – aids in the operational efficiency of phosphorus recovery and yields two commodities (calcium phosphate & ammonium sulfate)	Municipal Industrial Agriculture

QUICK WASH®

- More than 16,000 municipal waste treatment plants in US
- Animal production is a major component of the US economy
- Existing solutions to water treatment and disposal of animal waste are costly and ineffective
- Concern for public health risks and treatment costs of contaminated water supplies
- Increasing demand for fertilizer and nutrients and limited world phosphorus reserves
- Major environmental risks around important bodies of water like the Chesapeake Bay, Susquehanna River, Lake Michigan and the Gulf of Mexico etc.
- Current "old" technology does not produce usable and marketable co-products



General scheme of nutrient pathways

QUICK WASH®

Competitive Technology Comparison

Nutrient Removal/Recovery Technologies

	Chemical	EBPR	Algea	Struvite	Renewable Nutrients
Characteristics	Alum, Ferric Chloride, Cerium Chloride	A2/), 4-5 Stage Bardenpho, Biological/ Chemical, SBR	Clearas, ABNR. Algae Wheel, Lagoon Settling Systems	Ostara, Multi-Form Harvest, CNP-Airprex, Phospaq, Crystalactor, NuReSys	QUICK WASH®
Side Stream vs Main Stream	Both	Side Stream	Mainstream	Side Stream	Both
Solids Stream Application	No	No	No	No	Yes
Phosphorus Removal	Yes	Yes	Yes	Yes	Yes
Phosphorus Recovery	No	No	Yes	Yes	Yes
Nitrogen Removal	No	No	No	No	Yes
Ammonia Recovery	No	No	No	No	Yes
Sludge Bulkin	Yes	Yes	No	No	No
Byproduct Recovered	N/A	N/A	Algae Biomass	Struvite Fertilizer	Calcium Phosphate - Hydroxylapatite / Ammonia Sulphate
% Efficiency of Recovery	N/A	N/A	>95%	45-90%	>95%
Scalability	Small to Large Facilities	Typically Larger Facilities >10MGD	Scales from small to large facilities	Large facilities - only economical when sidestream contributes large % of total P	Scales from small to large

"The EPA will actively promote those Municipal Biosolids management practices that provide for the beneficial use of Biosolids while maintaining or improving environmental quality and protecting public health" -- Federal Register Vol. 49, p 24358

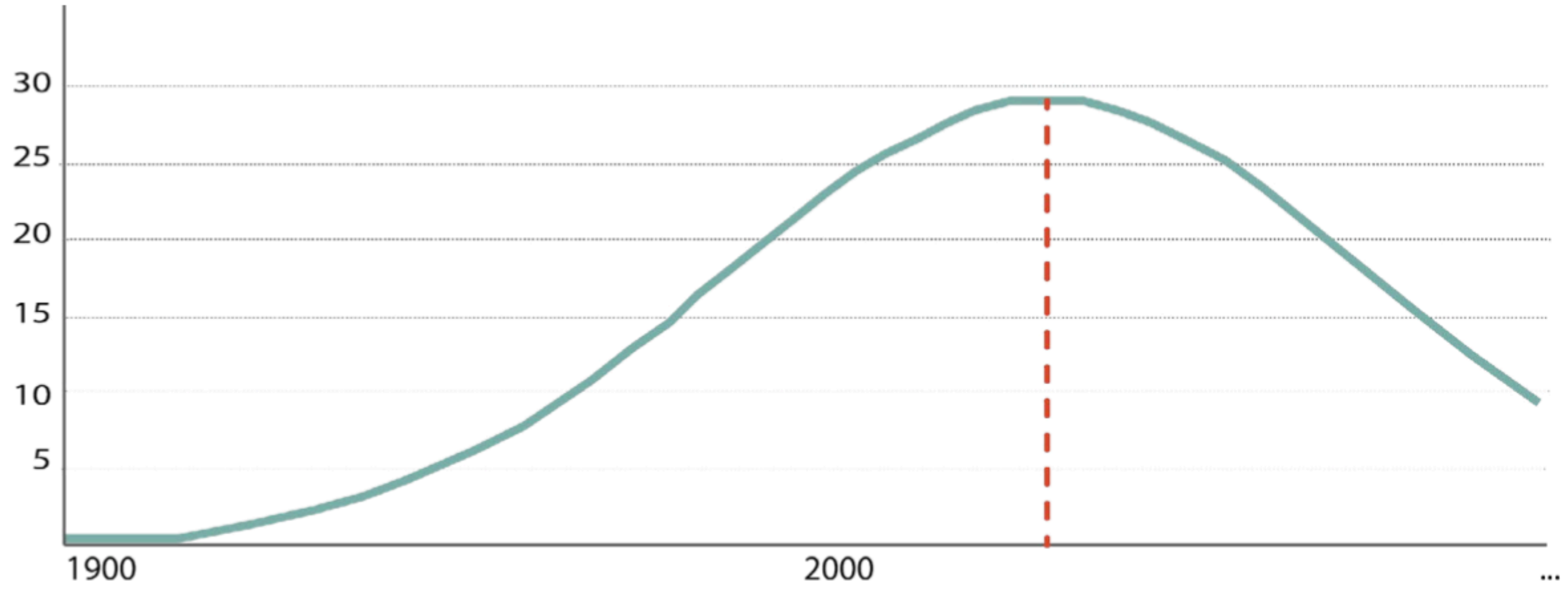
Environmental Issue	Scope
Hazards of Disposing of Nutrient rich Waste Biosolids from Wastewater Treatment Plants	• (WWTPs) in U.S. generate approximately 7 million dry tons of sludge each year • Causes more pollution per acre than other "fertilizer" and 4x more phosphorous
Contamination from animal waste	• Over 10.2 million tons of poultry litter alone are generated in the US
Public Health Risks and Treatment Costs from Contaminated Drinking Water Supplies	• Current disposal methods result in contamination of surface waters • Problems compounded by population growth through increased storm water runoff, municipal wastewater discharges, and air carried pollutants
Limited World Phosphorus Reserves	• World will reach peak Phosphorus mining production in 50 to 100 years • Necessity to sustain life • No synthetically reproducible alternative for phosphorus; technology to extract and reuse phosphorus is appealing

The Environmental Crisis

Demand for phosphorus has greatly expanded since 1960 and appears to be continuing on this trend

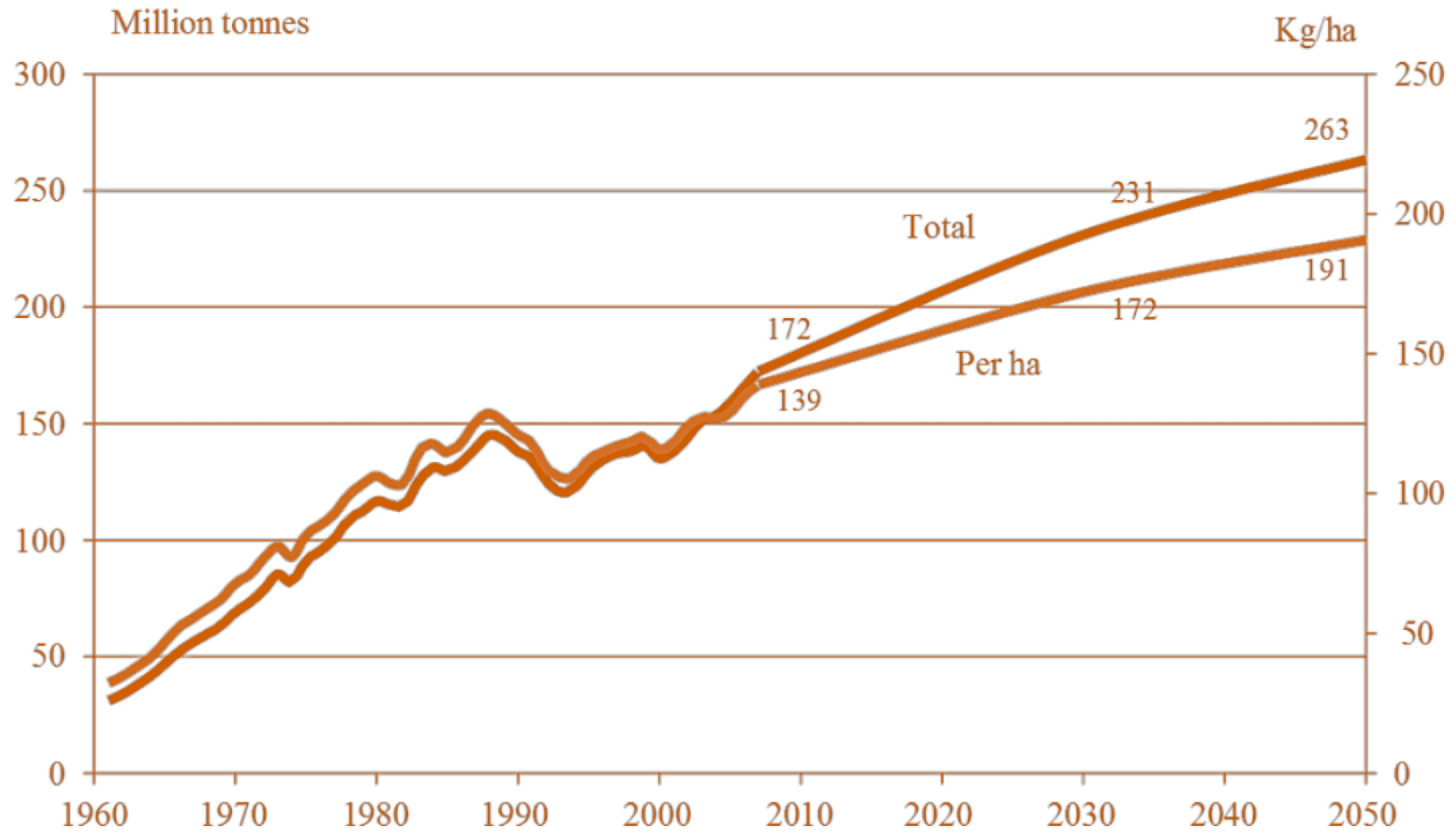
WORLD PHOSPHATE ROCK PRODUCTION (TONS)

*Source: USGS



Peak Phosphorus refers to a time when production reaches a maxcimum, and high quality, highly accesible reserves have been depleted

WORLD PHOSPHATE ROCK PRODUCTION (TONS)



World fertilizer use: past and projected (nitrogen, phosphorus and potassium aggregated). Source: FAO, World

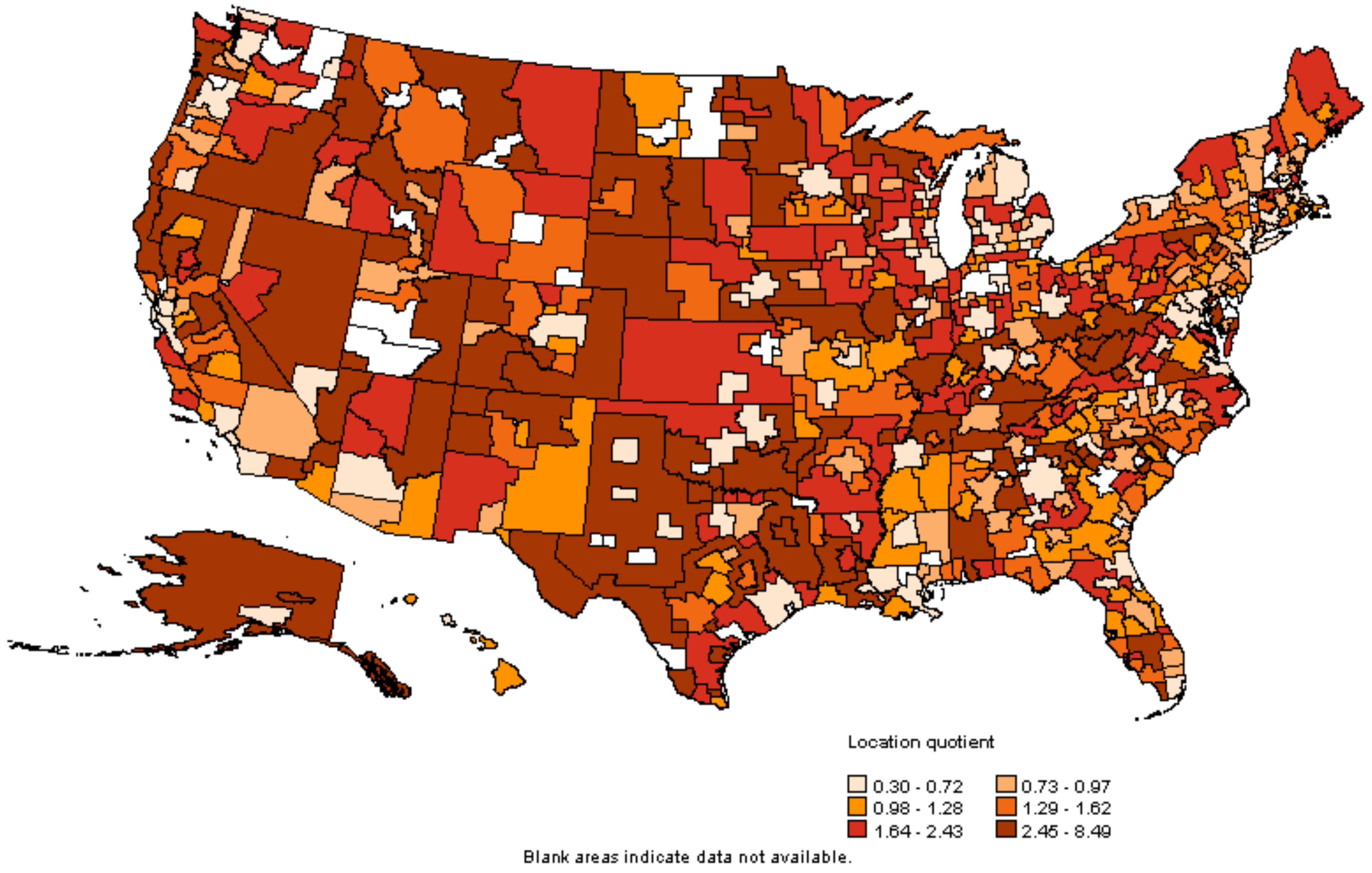
QUICK WASH®

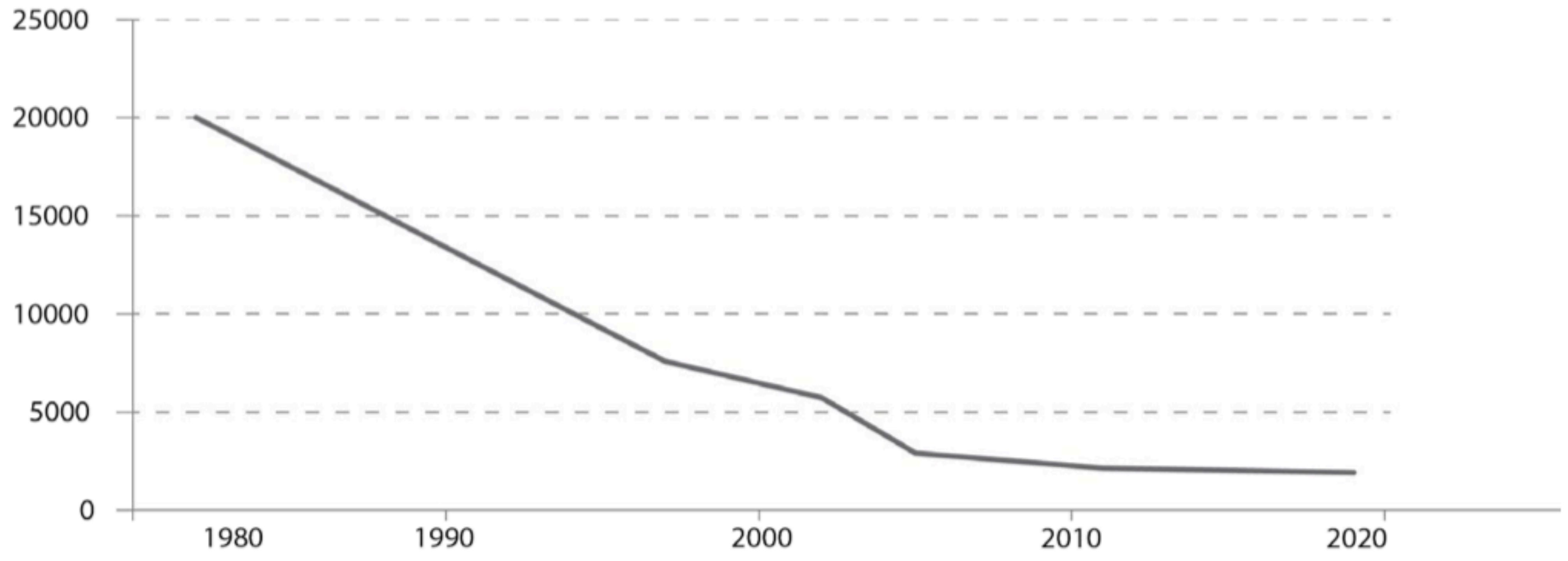
Key Drivers for Adoption of Nutrient Recovery

Environmental	Regulatory	Operational
•Eutrophication of waterways •Drinking water concerns •Growth of toxic algae •Depletion of phosphorus reserves •Impact to fisheries, aquatic species, and food supply	•Facility operational permitting based on nutrient effluent management •Reduction of phosphorus effluent limits to < 1ppm and some will be required to reach "ultra-low" levels for discharge under 0.04 mg/l •Facilities operating near impaired waterways to see phosphorus effluent limits < 0.5ppm •Reduction in permitted "free ammonia" ($NH3$) effluent levels •Farmers forced to store or dispose of animal manure vs land application	•Reducing the level of phosphorus in biosolids allows facilities and farmers to land apply vs dispose biosolids •Drives greater efficiency of biological nutrient removal by reducing overall phosphorus and nitrogen load in the facility •Eliminates costly struvite scaling •Reduced biosolids production •Incremental revenue gain through sale of recovered coproducts

Key Drivers for Adoption of Nutrient Recovery

Municipal System Operators (16,000 +) are located across the entire United States, creating a market and customer base for Renewable Nutrients with regards to waste treatment.

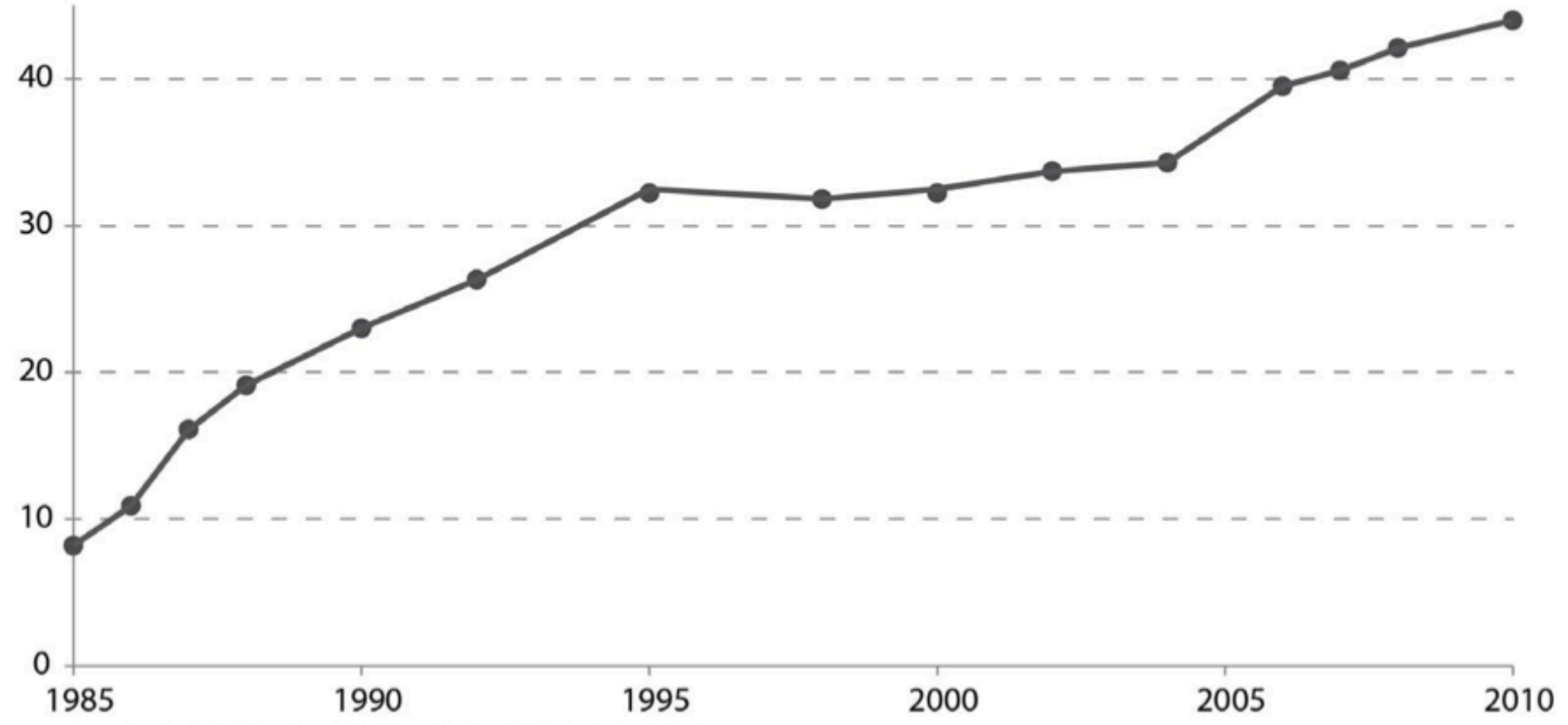




US LANDFILL DECLINE

*Source: Waste & Recycling News

The number of US landfill has been declining since the late 1970s,



US AVERAGE TIPPING FEE

*Source: Waste & Recycling News

The decreasing number of landfills has led to increases in tipping fees at an average of $1.24 per ton across the US each year. In the future, landfill may not be the least expensive biosolid management alternative

RENEWABLE NUTRIENTS

Technology Summary

QuickWash®

Nutrient Extraction & Recovery

LIFT©

isle

USDA

- A combined application of both **Phosphorus & Nitrogen** extraction and recovery technologies to solve the total nutrient issue for high strength streams without additional treatment steps
- The use of existing tankage, piping, and pumping is highly possible and recommended
- We anticipate that regulatory agencies will enforce more stringent discharge requirements for both **Phosphorus & Nitrogen**
- Economy of scale can be achieved by implementing both exclusive **Phosphorus & Nitrogen** technologies
- Renewable Nutrients is the only company with a combined IP protected **Phosphorus & Nitrogen** recovery offering in addition to individual **Phosphorus & Nitrogen** recovery solutions
- Utilizing Quick Wash technologies will enable the extraction, recovery, and reuse of valuable nutrients in an efficient and cost effective offering.

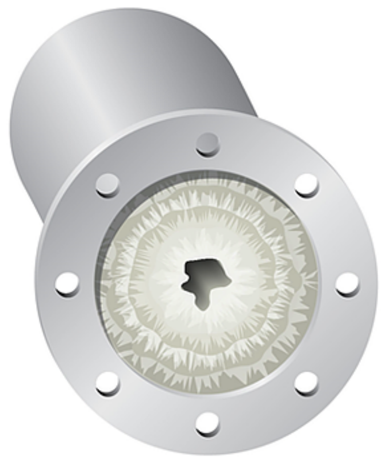
Configurations/Applications

- Side Stream from dewatering (aerobic or anaerobic)
- Solids Prior to dewatering
- Prior to thickening & anaerobic digestion
- Pre-treat a high **Phosphorus & Nitrogen** industrial discharge
- Animal agriculture applications
- Other custom configurations
- Renewable Nutrients has designed multiple process flow diagrams for different applications of the Quick Wash process

QuickWash®

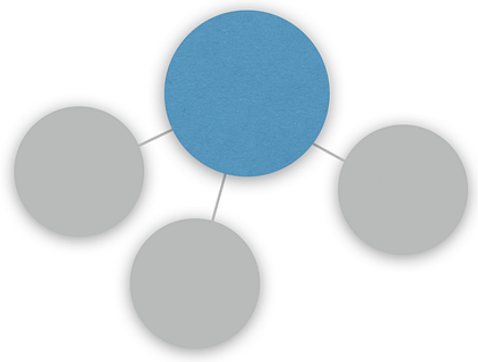
Phosphorus Extraction Benefits

- Removes and recovers more than 95% of **Phosphorus**
- Reduce phosphorus concentration in high strength recycle streams such as digester supernatant liquid waste
- Reduce polymer & metal salts
- Reduce disposal costs
- Eliminate struvite scaling
- Increase revenue from sales of phosphorus by-product
- Meet EPA nutrient TMDL requirements



QUICK WASH®

Nitrogen / Ammonia Extraction Benefits



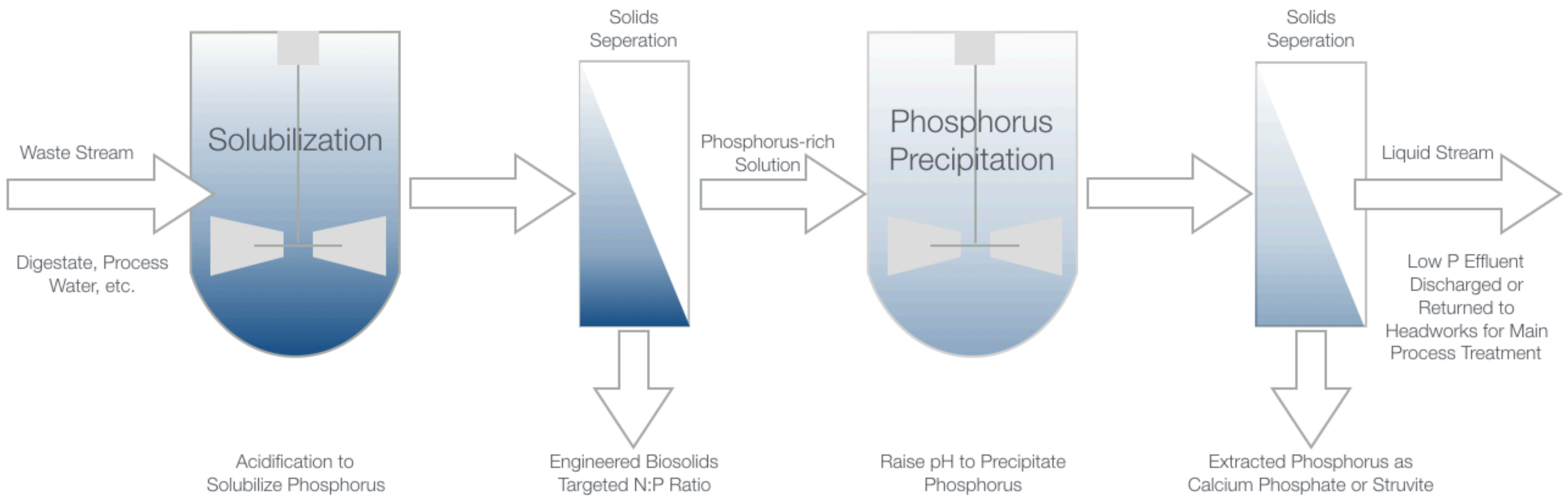
- Removes and recovers more than 95% of **Nitrogen**
- Reduce inputs of energy, carbon, alkalinity
- Recover rather than destroy a valuable resource
- Produce a high quality, marketable ammonium compound product ie ammonium sulfate or ammonium citrate
- Improve the quality of the effluent by enhancing exiting nutrient removal processes
- Allow the reduction of ammonia alkalinity
- Potentially free existing capacity and allow the re-rating of treatment plants in lieu of additional capital investment
- Provide a solution for additional capacity in plants with limited footprint for expansion

QuickWash®

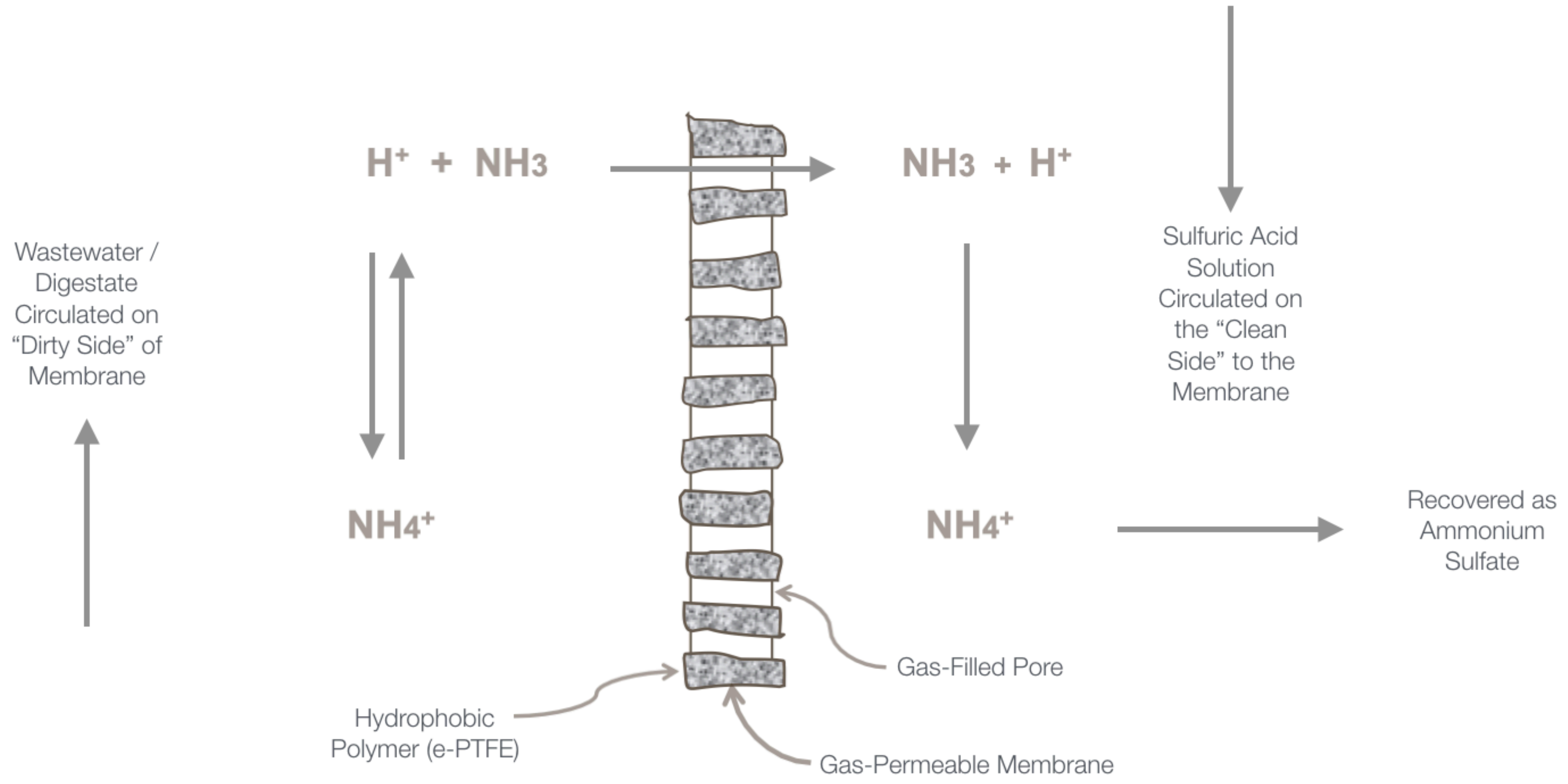
A Multitude of Intangible Benefits

Description	Discussion	Value
Ability to treat high-strength side stream digestate, centrate, filtrate without complex or expensive equipment	Depends on application. Compared to other technologies, the RN technology can serve as an enabling technology, solving the problems that are generated from anaerobic digestion	Possibly worth millions depending on specific project application and compared to alternative technologies
Production of beneficial use byproducts	Our standard design produces calcium phosphate and ammonium sulfate. Both are valuable and popular in the agronomics/agriculture sector for fertilizer and other uses.	We recognize the byproducts do not generate significant revenue. But they generate revenue rather than a byproduct that requires further cost for disposal. Value in tens of thousands to hundreds of thousands annually depending on system size
Elimination of struvite formation	Many agencies are struggling with the effects of scaling and struvite formation in piping, pumps, and equipment. By recovering the P, and not allowing it back into the plant to form struvite scaling.	Many agencies are spending hundreds of thousands, even millions, annually to mitigate struvite issues. The value of elimination of struvite is significant.
Small footprint	Unlike some technologies, the RN technologies do not require a great deal of space	Many other technologies are disqualified due to space requirements. Having a technology with a small footprint requirement may save significant project dollars.
Low cost of operations	The input chemicals are relatively inexpensive and the power costs are low. Depending on the hydraulic grade considerations for a specific site, much of the process can flow via gravity	Sulfuric acid is a low cost chemical in comparison to many, as is lime or caustic. Because this is a chemical solution, there are chemical costs, but they are simple, affordable chemicals.
Fast process	Compared to other technologies, the RN technologies are fast acting, therefore not requiring large tankage for long HRT's. This reduces CapEX and OpEx.	Compared to many, the RN technologies are fast. Speed is related to cost. The value in treatment costs is significant.
Major opportunity to reuse existing facilities	Existing tankage, pumps, and other facilities can be reused with ease to reduce costs	The equipment required for the technology is simple and is not proprietary. Therefore, existing tanks, mixers, pumps, piping and other facilities can be repurposed reducing project costs and saving money.
Flexibility in targets for performance	Adjustments in stoichiometric can allow for flexibility in performance and results and costs	Facilities can be sized and operated to achieve specific targets. The flexibility is valuable and allows for a system to follow changing conditions. The value of this flexibility is significant.
Technology is more robust than biological process - chemistry vs biology	With biology, a rogue contaminant, or rapid changes in conditions, can upset the active organism causing treatment failure. Chemistry is more predictable and reliable.	Other technologies that rely on bacteria, algae, microorganisms, or other biological presence have struggled to treat nutrient rich water (particularly) high strength due to the sensitive nature of the organisms. Having a reliable process that is more robust is priceless.
Processes will be more familiar to operations staff	Most operations staff are familiar with the use of pH modifications to manipulate treatment processes. Not much new to learn with RN tech.	Reduction in training and an acceleration in process acceptance has significant value.
Site specific Benefits	Depending on conditions and finanical opportunities ie. Insurance & nutrients trading	Potential to be game changing for a privately owned facility

Quick Wash Phosphorus Extraction and Recovery extracts phosphorus from municipal, agricultural, or industrial waste streams in solids or liquids (effluent) and recovers the phosphorus in the form of Calcium Phosphate or Struvite.



- Ammonia **LIQUID** technology relates to a system and methods for the removal, recovery and use of ammonia from ammonia-containing liquid effluents such as animal and municipal wastewater.
- Ammonia **GASEOUS** technology relates to a system and method for the removal of gaseous nitrogen to reduce emissions from systems that produce gaseous nitrogen.



RENEWABLE NUTRIENTS

QUICKWASH®

Pilot/Trial & Data Summary

QuickWash®

Pilot/Trial Large Scale Locations

Facility	Size	Type	Treatment
Ephrata, PA	2 MGD	Municipal	Solid Stream - Feed to BFP
Westminster, MD	5 MGD	Municipal	Solid Stream - Feed to Dewatering
Raleigh, NC	60 MGD	Municipal	Side Stream - Filtrate from BFP
Greenville, NC	14 MGD	Municipal	Side Stream Filtrate from BFP & Solid Stream - Feed to BFP
Chapel Hill, NC	8 MGD	Municipal	Side Stream - Filtrate from Rotary Press & Solid Stream - Feed to Rotary Press
Walk Stock Farm, IL	Pit	Agrigultural	Raw Swine Manure - Pit
Fertilizer Manufacturer, PA	Industrial	Industrial	Byproduct of Manufacturing Process
Smithfield Foods, NC	Lagoon	Agricultural	Raw Swine Manure - Lagoons
Waste to Energy Project, OH	40 KGD	Industrial	Animal Waste Digestate
Perrysburg, OH	8 MGD	Municipal	Side Stream - Filtrate from BFP
Mercer County, OH	Pit	Agricultural	Raw Swine Manure - Pit

QuickWash®

Pilot/Trial Sample Bench Locations

Facility	Type	Treatment
Big Ox Energy	Industrial	Animal Waste Digestate
CleanBay	Industrial	Animal Waste Digestate
Abtech	Industrial	Digestate
Feed Energy Corp	Industrial	Animal Waste Digestate
Chicago (MWRD), IL	Municipal	Side Stream - Filtrate from BFP
Denver, CO	Municipal	Side Stream - Filtrate from BFP
Canton, OH	Municipal	Side Stream - Filtrate from BFP
Algonquin, IL	Municipal	Side Stream - Filtrate from BFP
Canton, OH	Municipal	Side Stream - Filtrate from BFP
West Goshen, PA	Municipal	Side Stream - Filtrate from BFP
Charlotte, NC	Municipal	Side Stream - Filtrate from BFP

QUICK WASH®

Sample Municipal Performance QWP

Facility	P Extraction %	P Recovery %	Stream Characteristics
Raleigh, NC	98%	>99%	Side Stream - Filtrate from BFP
Greenville, NC	98%	>99%	Side Stream Filtrate from BFP & Solid Stream - Feed to BFP
MWRD	95%	>99%	Post Centrate Stream
Chapel Hill, NC	99%	>99%	Solid Stream BFP

SUBURBAN TESTING LABS

Select 3rd Party Laboratory Analytical Results
Select On-Board Pilot Analytical Results
Quick Wash Pilot Operations

QUICK WASH®

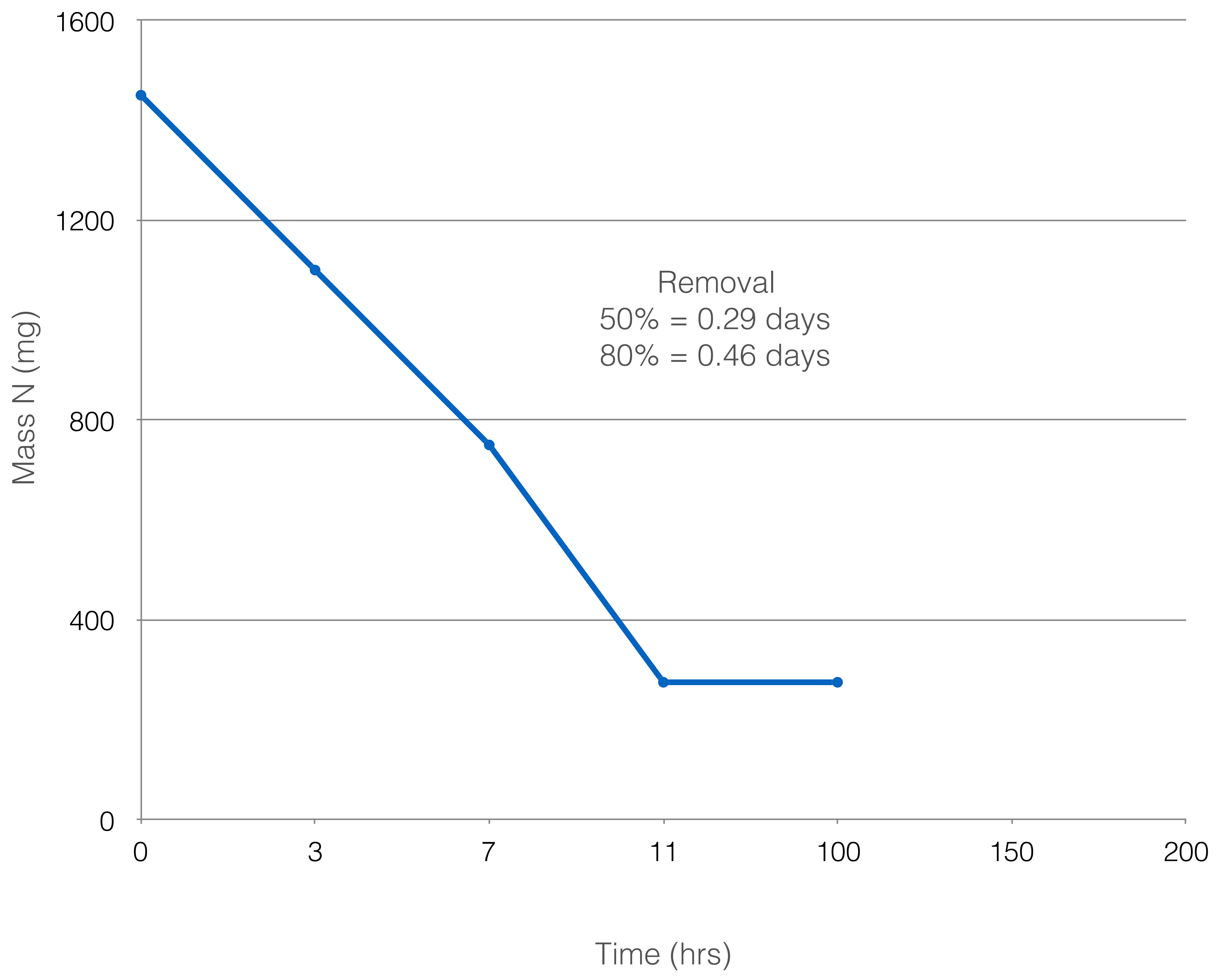
Sample Agricultural / Industrial Performance QWP

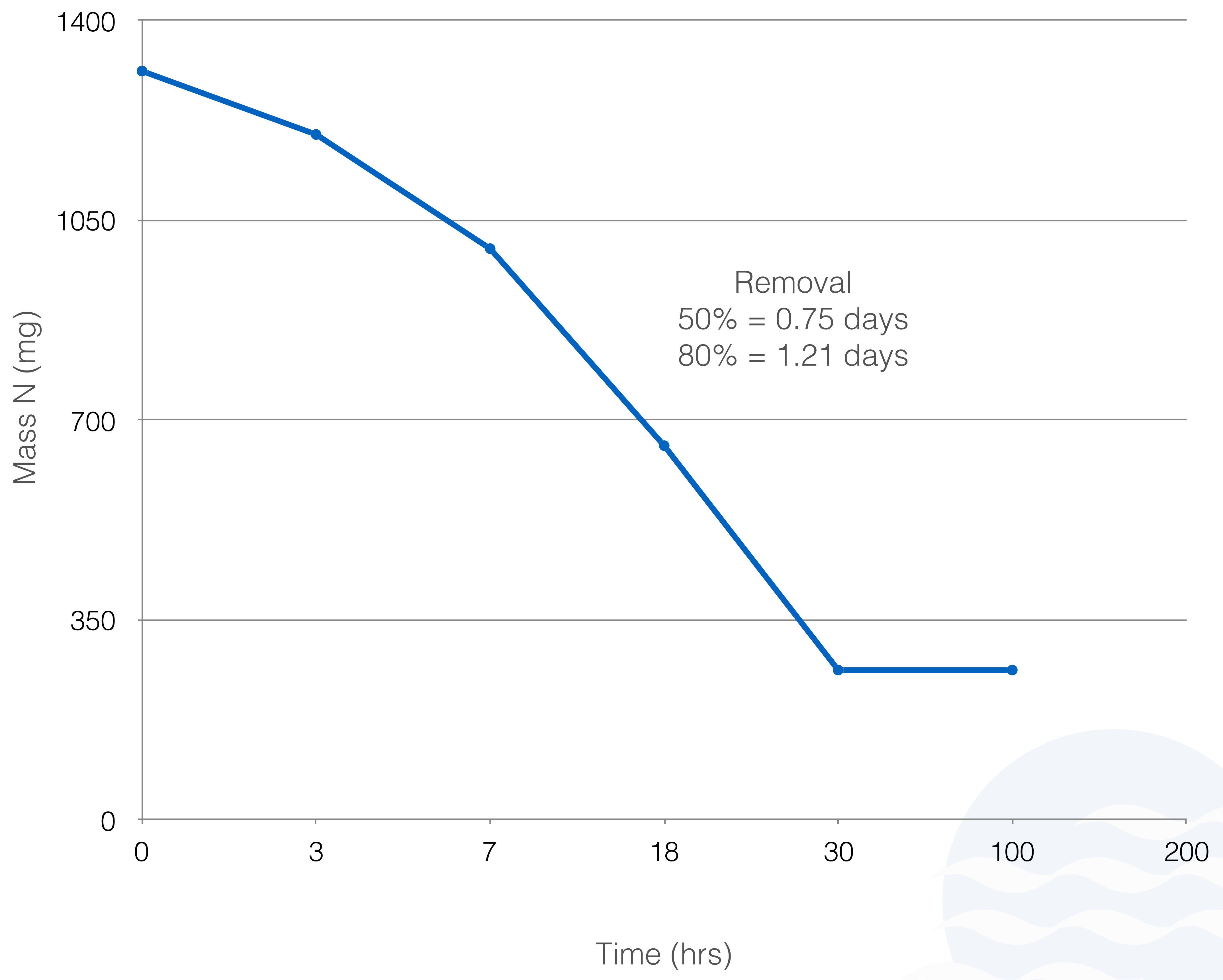
Facility	P Extraction %	P Recovery %	Stream Characteristics	Situation
Walk Stock Farm, IL	89%	>99%	Raw swine manure in sow and finisher pits	Desire to expand and operation, but limited by land availability to apply manure due to P restrictions
Fertilizer Manufacturer, PA	56%	>99%	Dewatered poultry litter	Recover P to mix into custom liquid fertilizer blends for specific markets
Smithfield Foods, NC	81%	>99%	Raw swine manure in lagoon	Beneficial reuse of P for applications beyond fertilizer
Waste to Energy Project, OH	88%	>99%	Swine Manure digestate	Remove P at Waste to Energy Facility prior to effluent discharge

SUBURBAN TESTING LABS

Select 3rd Party Laboratory Analytical Results
Select On-Board Pilot Analytical Results
Quick Wash Pilot Operations

Sample Municipal Performance - Side Stream QWN





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"Leading the way in nutrient recovery"

ATTACHMENT 71

environments

MDPI

Article

Differences in Microbial Communities and Pathogen Survival Between a Covered and Uncovered Anaerobic Lagoon

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Abstract: Anaerobic lagoons are a critical component of confined swine feeding operations. These structures can be modified, using a synthetic cover, to enhance their ability to capture the emission of ammonia and other malodorous compounds. Very little has been done to assess the potential of these covers to alter lagoon biological properties. Alterations in the physicochemical makeup can impact the biological properties, most notably, the pathogenic populations. To this aim, we performed a seasonal study of two commercial swine operations, one with a conventional open lagoon, the other which employed a permeable, synthetic cover. Results indicated that lagoon fecal coliforms, and *Escherichia coli* were significantly influenced by sampling location (lagoon vs house) and lagoon type (open vs. covered), while *Enterococcus* sp. were influenced by sampling location only. Comparisons against environmental variables revealed that fecal coliforms (r^2 = 0.40), *E. coli* (r^2 = 0.58), and *Enterococcus* sp. (r^2 = 0.25) significantly responded to changes in pH. Deep 16S sequencing of lagoon and house bacterial and archaeal communities demonstrated grouping by both sampling location and lagoon type, with several environmental variables correlating to microbial community differences. Overall, these results demonstrate that permeable synthetic covers play a role in changing the lagoon microclimate, impacting lagoon physicochemical and biological properties.

Keywords: anaerobic lagoons; permeable cover; microbial communities; pathogens; *Enterococcus*; *Escherichia coli*

1. Introduction

Anaerobic lagoons remain the preferred option of manure treatment for confined swine production systems in the Southeastern United States. These earthen structures, utilized for both passive treatment and storage, are aimed at reducing the organic load of fresh manure and consequently, concentrating the nutrients contained within these waste materials. These nutrients, combined with anaerobic conditions, provide a suitable growth environment for a variety of microorganisms, including a number of pathogenic bacteria [1]. In the Southeastern U.S., liquid manure is collected under the barns using slotted floors and a shallow pit filled daily or weekly with the supernatant lagoon effluent. Any excess lagoon liquid not used for filling the shallow pit is land applied on spray fields during the crop season. Therefore, pathogens can be reintroduced into the barns with recycled lagoon liquid or deposited into the surrounding environment during land application of the lagoon wastewater, where they may eventually infect livestock or truck crops, thereby potentially entering the food chain [2].

While the construction of anaerobic lagoons tend to follow general engineering design criteria [2], swine operators have the discretion to add additional safeguards and management measures as long as such modifications continue to meet federal, state, and local regulations [3]. For instance, swine operations adjacent to communities may opt to employ synthetic lagoon covers for the control of ammonia and other malodorous compounds [4,5]. These covers can be permeable (e.g., geotextile, foam, straw) or impermeable (e.g., plastic, wood, concrete), and despite differences in cover composition, they all serve a similar purpose—to reduce emissions. One benefit to permeable covers is their ability to allow oxygen penetration, resulting in microclimate formation at the cover/lagoon interface [6,7], and such microclimates have been documented to result in the formation of biofilms [7], enhance protozoa fauna populations [6], and alter nutrient cycling patterns [4].

Given the reliance on anaerobic lagoons by the swine industry as a waste treatment measure, significant research has been conducted into understanding pathogen fate [8], nutrient cycling [9], and emissions [10] in these systems. Many of these studies have focused on open (i.e., uncovered) lagoons, primarily because they dominate the treatment landscape. Despite research demonstrating that lagoon covers utilized in swine production reduce ammonia and malodor emissions, there remains a paucity of information regarding the microbial community composition of covered lagoons, and the potential for synthetic covers to impact pathogenic populations.

Given the lack of information on the microbial communities that populate covered lagoons, and whether these lagoons can control pathogenic populations, this study was conducted with two major objectives: (i) determine potential differences in pathogen kill rates and (ii) assess microbial community differences between a covered and uncovered lagoon. A third objective, if differences are identified in the first two objectives, is to determine the relationship between environmental factors and the noted differences between the two types of lagoon systems.

2. Materials and Methods

2.1. Site Description and Sample Collection

Two commercial swine finishing operations were chosen for this study. The first operation, supporting between 2100 and 2200 animals per cycle, had an uncovered 0.55 ha lagoon, while the second operation, supporting between 1200 and 1500 animals per cycle, functioned with a synthetic, permeable membrane covering the 0.4 ha lagoon, details of which have been previously described [6]. The covered lagoon operation employed a flush tank recirculation system, while the open lagoon operation employed a shallow pit with a pull-plug flushing system for moving waste out of the house. Samples were collected seasonally, starting with a spring sampling in April of 2017, and ending with a winter sampling in February of 2018. Samples from both lagoons and houses were performed in triplicate. For the uncovered house, samples were collected from the recirculation pump, while samples collected from the covered house were collected inside, during the flushing event. Lagoon samples were collected from the top of the water column at three separate locations.

2.2. Sample Analysis

Dissolved oxygen and temperature were recorded on site using a YSI ProODO optical dissolved oxygen meter (YSI Incorporated, Yellow Springs, OH, USA) prior to transport and storage of lagoon liquid samples on ice. Additional wastewater analyses, which included total suspended solids (TSS), volatile suspended solids (VSS), pH, ammonium (NH_4-N), and total Kjeldahl-N (TKN) were performed according to Standard Methods for the Examination of Water and Wastewater [11]. Anions (Cl, SO_4-S, NO_3-N, NO_2-N) were measured by chemically suppressed ion chromatography using a Dionex 2000 Ion Chromatograph according to ASTM Standard Method D4327-11 [12], while cations (Ca, K, Mg, and Na) were measured according to ASTM Standard Method D6919-09 [13].

2.3. Pathogen Detection

Escherichia coli, fecal coliforms, and *Enterococcus* sp. were enumerated on CHROMAgar E. coli (CHROMagar, Paris, France), mFC (Sigma, St. Louis, MO, USA), and mE (Becton Dickinson, Franklin Lakes, NJ) agar, respectively. To determine colony-forming units (CFU), wastewater samples were serially diluted in sterile phosphate-buffered saline (PBS) and spiral plated in triplicate on the corresponding plates. All incubations were done aerobically, at temperatures and times as follows: *E. coli* at 37 °C for 24 h; fecal coliforms at 44.5 °C for 24 h; and *Enterococcus* sp. at 37 °C for 48 h. Due to the variability in suspended solids from sample to sample, all CFU were adjusted per gram of volatile suspended solids (CFU/gVSS) prior to log10 normalization for statistical analysis purposes.

2.4. DNA Extraction

A total of 2 mL of each wastewater sample was set aside for DNA extraction using a Qiagen Allprep PowerViral DNA/RNA Kit (Qiagen Sciences Inc, Germantown, MD). A total of 200 µL of each sample was used per extraction using protocol modifications designed to extract DNA only (no RNA) from wastewater and manure samples (i.e., no β-mercaptoethanol added to buffer solutions, and DNase steps skipped). The remaining wastewater samples were archived at −80 °C. DNA purity was determined by absorbance at 260 and 280 nm using a spectrophotometer, and quantity was determined fluorometrically using a Qubit dsDNA assay kit (ThermoFisher Scientific, Waltham, MA, USA).

2.5. Deep 16S sequencing and Analysis

Deep 16S sequencing of the V3–V4 region was performed on an Ion Torrent PGM sequencer, using a 316v2 chip and Hi-Q View sequencing reagents. Barcoded bacterial 341F (5′-CCTAYGGGRBGCASCAG-3′) and 806R (5′-GGACTACNVGGGTWTCTAAT-3′), and archaeal ARC787F (5′-ATTAGATACCCSBGTAGTCC-3′) and ARC1059R (5′-GCCATGCACCWCCTCT-3′) primers were designed according to the Ion Amplicon Library Preparation Fusion Methodology (Life Technologies, Carlsbad, CA, USA), and included 12 base pair error-correcting Golay barcodes [14]. Primers were synthesized by Integrated DNA Technologies (IDT, Coralville, IA, USA). Individual amplicon libraries for bacterial and archaeal community analysis were generated by PCR using the following protocol: activation of enzyme at 94 °C for 3 min, followed by 40 cycles of denaturation at 94 °C for 30 s, annealing at 58 °C for 30 s, and elongation at 68 °C for 45 s. Amplicons were quantified using a Qubit Fluorometer (Invitrogen, Carlsbad, CA, USA), quality controlled on an Agilent 2100 BioAnalyzer (Agilent, Santa Clara, CA, USA), and amplicons from each sample were mixed in equimolar amounts prior to sequencing.

Full-length forward- and reverse-direction sequencing libraries for each sample were verified for read quality, assembled, and analyzed using the Ion Reporter v5.10 platform and metagenomics workflow (ThermoFisher Scientific, Waltham, MA, USA). Operational taxonomic units (OTUs) were assigned at a cutoff of 97% for genus identification using the curated MicroSEQ 16S v2013.1 and Greengenes v13.5 [15] reference libraries. For determination of substrate utilization for methanogenesis, archaeal families were sorted into three groups: acetoclastic, hydrogenotrophic, and methylotrophic. *Methanosaetaceae*, *Methanosarcinaceae*, and *Methermicoccaceae* were classed as acetoclastic. The *Methanomassiliicoccaceae* was classed as methylotrophic. The remaining were classed as hydrogenotrophic.

2.6. Statistical Analysis

All statistical analyses were performed using Minitab 17 (Minitab Incorporated, State College, PA). Analysis variance (ANOVA) was conducted using the general linear model, with pairwise comparisons using Fisher's Least Square Difference Method (LSD); difference between any two means was considered significant with $p < 0.05$. Regressions of bacterial CFUs (log10 CFU/gVSS) with environmental variables were performed using a linear model. Non-metric multidimensional scaling (NMS) of microbial community population data was performed in PC-Ord v.6 (MJM Software Design, Gleneden Beach, OR, USA).

3. Results and Discussion

3.1. Wastewater Characteristics

Wastewater physicochemical characteristics are summarized in Table 1. Results are consistent with the wastewater properties of other swine-studied anaerobic lagoons [8,9]. Seasonal effects were documented for temperature, with summer samples showing significantly higher ($p < 0.05$) temperatures than other samples, and dissolved oxygen, with spring samples (0.74 ± 0.07 SE mg L^{-1}) demonstrating significantly higher dissolved oxygen (DO) levels ($p < 0.05$) than fall samples (0.37 ± 0.01 SE mg L^{-1}). Sampling location (i.e., lagoon vs house) was significant ($p < 0.05$) for pH, temperature, TSS/VSS, and TKN. Lastly, when examining the lagoons, the type of system (i.e., open versus covered) demonstrated significant differences ($p < 0.05$) for pH and TKN. Covered lagoons (1009 ± 24 SE mg L^{-1}) have more than double the TKN of open lagoons (473 ± 44 SE mg L^{-1}), this may be explained by the TKN levels originating in the animal houses that feed into those lagoons. TKN in the house feeding into the covered lagoon had mean TKN levels of 3632 ± 339 SE mg L^{-1}, while the house feeding into the open lagoon had mean TKN levels of 1306 ± 211 SE mg L^{-1}. These results differ from those of VanderZaag et al. [4], which showed no significant difference between TKN levels of a covered lagoon system when compared to an open control lagoon system filled from the same wastewater source.

Table 1. Fisher pairwise comparisons of lagoon and house physicochemical characteristics.

Season	System	Site	pH	DO	Temp	Total Suspended Soils	Volatile Suspended Solids	Total Kjeldahl Nitrogen
				mg L^{-1}	°C		mg L^{-1}	
Spring	Open	Lagoon	8.11 [a1]	0.60 [bcde]	20.5 [i]	189 [e]	166 [d]	630 [gh]
		House	7.24 [d]	0.96 [a]	21.5 [h]	3200 [e]	3125 [d]	1741 [e]
	Cover	Lagoon	6.92 [h]	0.66 [bcd]	20.6 [i]	293 [e]	289 [d]	906 [fg]
		House	7.33 [def]	0.86 [ab]	26.1 [c]	6275 [de]	5975 [cd]	2445 [d]
Summer	Open	Lagoon	7.86 [b]	0.92 [a]	27.3 [b]	149 [e]	124 [d]	276 [i]
		House	7.24 [efg]	0.25 [gh]	28.3 [a]	15,600 [d]	13,900 [c]	1850 [e]
	Cover	Lagoon	7.16 [fg]	0.53 [def]	27.0 [b]	410 [e]	285 [d]	985 [f]
		House	6.67 [i]	0.23 [h]	27.5 [b]	100,100 [a]	89,600 [a]	4823 [a]
Fall	Open	Lagoon	7.72 [bc]	0.40 [efgh]	20.9 [i]	276 [e]	237 [d]	399 [hi]
		House	7.35 [def]	0.33 [fgh]	22.2 [k]	325 [e]	265 [d]	546 [hi]
	Cover	Lagoon	7.28 [efg]	0.35 [fgh]	21.9 [gh]	483 [e]	410 [d]	1110 [f]
		House	7.04 [gh]	0.40 [defgh]	24.1 [e]	51,400 [b]	44,850 [b]	3848 [b]
Winter	Open	Lagoon	7.76 [bc]	0.55 [cdef]	23.3 [f]	467 [e]	413 [d]	588 [ghi]
		House	7.36 [def]	0.81 [abc]	25.6 [d]	1927 [e]	1755 [d]	1088 [f]
	Cover	Lagoon	7.41 [de]	0.53 [def]	19.1 [j]	427 [e]	373 [d]	1033 [f]
		House	7.13 [fgh]	0.53 [cdefg]	25.5 [d]	41,375 [c]	37,250 [b]	3410 [c]

[1] Means followed by the same letter are not significantly different at $p = 0.05$.

Analysis of cation and anion concentrations of swine wastewater are found in Table 2. While NO_2-N and NO_3-N were assayed, they were below detectable limits throughout the course of the study. No significant seasonal effects were noticed amongst samples, although sampling location was significant for all cations and anions detected, with significantly increased concentrations ($p = 0.05$) in swine houses. When examining the lagoon system used, SO_4-S was significantly higher ($p < 0.05$) in the open lagoons (30.8 ± 4.9 SE mg L^{-1}) as compared to covered lagoons (4.5 ± 0.6 SE mg L^{-1}); conversely, NH_4-N was significantly increased ($p < 0.05$) in covered lagoons (858 ± 11 SE mg L^{-1}) as compared to open lagoons (379 ± 38 SE mg L^{-1}). As already noted for TKN, the higher NH_4-N concentrations in the covered lagoon most likely were a consequence of higher N loading in the more concentrated wastewater derived from the house feeding into it.

Table 2. Fisher pairwise comparisons of lagoon and house anions and cations (mg L^{-1})[1].

Season	System	Site	Cl	NH_4-N	PO_4-P	SO_4-S	Ca	K	Mg	Na
Spring	Open	Lagoon	491.8 ^{f}[2]	490.0 ^{h}	18.7 ^{efg}	41.3 ^{c}	74.5 ^{g†}	720.1 ^{fgh}	29.6 ^{f}	254.4 ^{e}
		House	750.8 ^{c}	1182.2 ^{c}	33.3 ^{cdef}	8.4 ^{d}	123.6 ^{bcd}	983.8 ^{c}	34.9 ^{cdef}	389.8 ^{b}
	Cover	Lagoon	444.6 ^{g}	842.4 ^{de}	94.0 ^{b}	5.4 ^{d}	113.7 ^{cd}	658.5 ^{ghi}	57.2 ^{cd}	214.9 ^{f}
		House	918.3 ^{a}	1420.0 ^{b}	47.5 ^{cd}	63.8 ^{b}	82.3 ^{efg}	1363.9 ^{b}	43.7 ^{cdef}	431.2 ^{a}
Summer	Open	Lagoon	407.5 ^{h}	204.1 ^{j}	7.9 ^{g}	40.0 ^{c}	48.4 ^{h}	598.4 ^{i}	35.9 ^{def}	195.4 ^{f}
		House	543.0 ^{e}	639.8 ^{g}	10.5 ^{fg}	9.3 ^{d}	74.4 ^{g}	831.1 ^{de}	51.6 ^{cdef}	259.9 ^{e}
	Cover	Lagoon	440.7 ^{gh}	807.7 ^{ef}	8.4 ^{g}	6.5 ^{d}	26.1 ^{i}	644.1 ^{hi}	33.8 ^{ef}	206.4 ^{f}
		House	665.8 ^{d}	754.2 ^{f}	25.5 ^{cdefg}	2.9 ^{d}	191.3 ^{a}	1038.9 ^{c}	60.2 ^{cd}	345.1 ^{c}
Fall	Open	Lagoon	423.6 ^{gh}	312.5 ^{i}	16.5 ^{fg}	39.0 ^{c}	65.2 ^{gh}	666.3 ^{ghi}	46.9 ^{cdef}	203.6 ^{f}
		House	489.7 ^{f}	426.3 ^{h}	21.5 ^{defg}	32.7 ^{c}	85.9 ^{efg}	758.6 ^{ef}	52.9 ^{cdef}	239.0 ^{e}
	Cover	Lagoon	430.0 ^{gh}	885.1 ^{d}	39.5 ^{cde}	4.0 ^{d}	103.0 ^{de}	708.8 ^{fgh}	55.9 ^{cde}	198.0 ^{f}
		House	828.3 ^{b}	1580.4 ^{a}	100.8 ^{b}	118.6 ^{a}	198.6 ^{a}	1562.5 ^{a}	104.2 ^{b}	424.9 ^{a}
Winter	Open	Lagoon	424.8 ^{gh}	509.2 ^{h}	17.8 ^{efg}	2.8 ^{d}	77.8 ^{fg}	690.4 ^{fgh}	44.6 ^{cdef}	200.3 ^{f}
		House	579.7 ^{e}	886.2 ^{de}	46.3 ^{cd}	BDL[3]	101.4 ^{def}	883.7 ^{d}	62.2 ^{c}	293.4 ^{d}
	Cover	Lagoon	433.3 ^{gh}	898.0 ^{de}	46.7 ^{cd}	2.2 ^{d}	134.8 ^{bc}	736.9 ^{fg}	54.8 ^{cde}	203.9 ^{f}
		House	777.2 ^{c}	1512.3 ^{ab}	217.9 ^{a}	46.5 ^{c}	140.9 ^{b}	1627.7 ^{a}	290.5 ^{a}	422.7 ^{a}

[1] F, NO_2-N, and NO_3-N were below detectable limits (<2 mg/L); [2] Means followed by the same letter are not significantly different at $p = 0.05$; [3] BDL, below detectable limits (<2 mg/L).

3.2. Pathogen Reduction

Fecal coliforms, *E. coli*, and *Enterococcus* sp. were identified and enumerated in all samples (Supplementary Table S1). The highest rates of enumeration were found in animal houses, and at no point were CFU rates higher in a lagoon when compared to its respective house.

3.2.1. House vs. Lagoon

Comparisons between animal houses and their respective lagoon can be found in Figure 1. Differences in bacterial levels (Figure 1; Supplementary Table S1) between the houses and wastewater lagoons demonstrate that transfer of wastewater from the houses to the lagoons results in significant reductions to all three bacterial indicators measured. Given that all CFUs were adjusted based on volatile suspended solid levels, these reductions are independent of the solids concentration of the wastewater. Significant relationships ($p < 0.05$) were observed between fecal coliforms, *E. coli*, and *Enterococcus* sp. with pH, N (TKN; NH_4), chloride, K, and Na. These chemical properties demonstrated significantly higher concentrations in the houses as compared to the lagoons (Tables 1 and 2). These results are supported by Viancelli et al. [16] that similarly documented reductions in total coliforms and *E. coli* after movement of swine manure to anaerobic lagoons, a result that may be due to reductions in organic material leading to increased competition for resources.

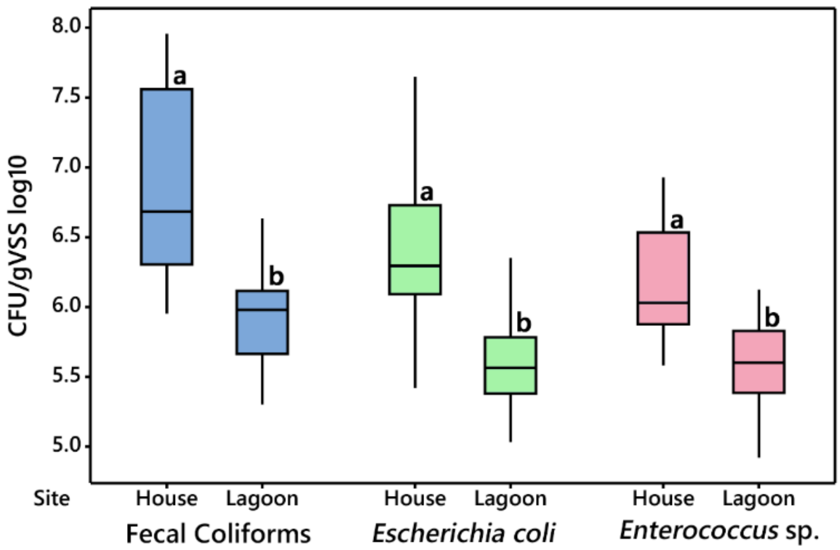

Figure 1. Comparison of colony-forming units (CFU)/gVSS $\log_{10}$ values between animal houses and lagoons, for fecal coliforms (blue), Escherichia coli (green), and Enterococcus sp. (red). Means followed by the same letter are not significantly different at $p = 0.05$.

3.2.2. Open vs. Covered Lagoon

Comparisons between open and covered lagoons can be found in Figure 2. Fecal coliform densities in the open lagoon ranged from 5.41 to 6.35 CFU/gVSS $\log_{10}$ to 5.73 to 7.04 CFU/gVSS $\log_{10}$ in the covered lagoon. *E. coli* ranged from 5.13 to 5.83 CFU/gVSS $\log_{10}$ in the open lagoon, to 5.34 to 6.47 CFU/gVSS $\log_{10}$ in the covered lagoon. *Enterococcus* sp. counts ranged from 4.88 to 5.94 CFU/gVSS $\log_{10}$ in the open lagoon, to 5.44 to 5.86 CFU/gVSS $\log_{10}$ in the covered lagoon. The CFU counts are listed in Supplementary Table S1.

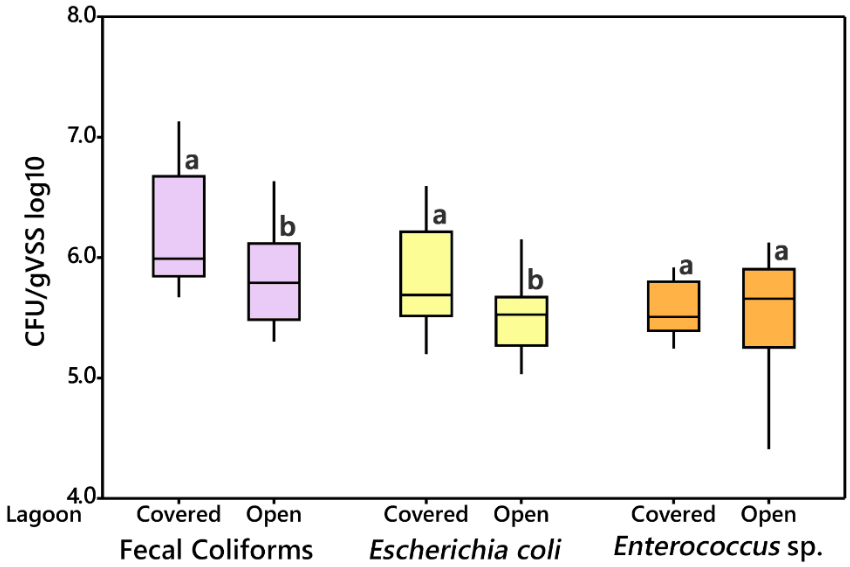

Figure 2. Comparison of CFU/gVSS $\log_{10}$ values between the covered and open lagoon, for fecal coliforms (blue), Escherichia coli (green), and Enterococcus sp. (red). Means followed by the same letter are not significantly different at $p = 0.05$.

Analysis of variance examining seasonal, site specific, and sampling location effects demonstrated that all three variables play significant roles in pathogen reduction. While seasonal patterns emerged in CFU counts for all three measured bacterial populations, with highest densities tending to be in the summer samplings, and the lowest densities observed during the winter, only fecal coliform counts demonstrated a significant relationship with temperature (r^2 = 0.485, p = 0.003). Examination between pathogen counts and physicochemical characteristics revealed significant relationships between pH and fecal coliforms (r^2 = 0.404, p = 0.008), *E. coli* (r^2 = 0.577, p = 0.001), and *Enterococcus* sp. (r^2 = 0.248, p = 0.05). For *E. coli*, significant relationships between TKN (r^2 = 0.261, p = 0.04), chloride (r^2 = 0.471, p = 0.003), potassium (r^2 = 0.272, p = 0.04), and sodium (r^2 = 0.394, p = 0.009) were also identified. No further influences on bacterial counts by physicochemical parameters were noted.

Additionally, for the lagoons, it appears that the addition of a cover had a significant impact on fecal coliform and *E. coli* levels, resulting in increased CFUs. It is possible that these higher bacterial densities in the covered lagoon may be due to solar radiation. Reductions in solar radiation have been demonstrated to result in increased bacterial counts [8], and may be a contributing factor in the increased bacterial counts in the studied covered lagoon. For the open lagoon, pH was significantly higher as compared to its covered counterpart (Table 1), and Curtis et al. identified that pH levels over 7.5, combined with sunlight, reduced fecal coliform levels [17]. *E. coli* thrive in a relatively neutral pH range, up to around pH 7.75, after which they begin to become stressed [18]. It should be noted that the open lagoon had pH ranges at or above this 7.75 pH value and could be contributing to the lower CFU counts observed. While increased pH may contribute to reductions in bacterial pathogens, it also results in increased ammonia volatilization. To counter this phenomenon, acidification is employed to reduce ammonia emissions from swine wastewater [19], and if modest reductions in pH (by two to three units) can also achieve significant pathogen reduction levels, it may provide producers with an additional means to reduce environmental impacts. This was demonstrated by Odey et al. who utilized lactic acid fermentation to inactivate fecal coliforms in human fecal sludge by reducing the pH to 3.9 [20]. Additionally, *E. coli* is considered a major reservoir of antibiotic resistance genes [21], so any employable means to reduce *E. coli* CFUs could prove to be a treatment capable of disrupting the cycle of antimicrobial resistance of animal origin.

3.3. Microbial Community Composition

Microbial community analysis using non-metric multidimensional scaling (Figures 3 and 4) revealed significant differences in the bacterial and archaeal population structures of the open and covered systems. While the samples taken from the lagoon and house of the open system showed a high degree of similarity, as evidenced by their overlapping groupings (Figures 3 and 4), the lagoon and house from the covered system neither overlapped with the open system, or each other. This pattern was similar in both the bacterial and archaeal NMS plots, and indicate larger differences in the population structure of the covered system. A number of environmental relationships correlate with these differences for bacterial populations (Figure 3), and are as follows: along the first axis, pH (r^2 = 0.354), TSS/VSS (r^2 = 0.467), TKN (r^2 = 0.513), K (r^2 = 0.532), and Na (r^2 = 0.338); and along the second axis, chloride (r^2 = 0.316), pH (r^2 = 0.266), and Na (r^2 = 0.258). Both TKN and suspended solids have been previously demonstrated to correlate with bacterial community structure [22]. Archaeal populations (Figure 4) correlated with several environmental variables along the first axis, Ca (r^2 = 0.468), K (r^2 = 0.468), and TKN (r^2 = 0.422). Calcium has been demonstrated to impact anaerobic digestion at concentrations as low as 100 mg L^{-1} [23], while potassium has been reported as toxic to acetate-utilizing methanogens [24].

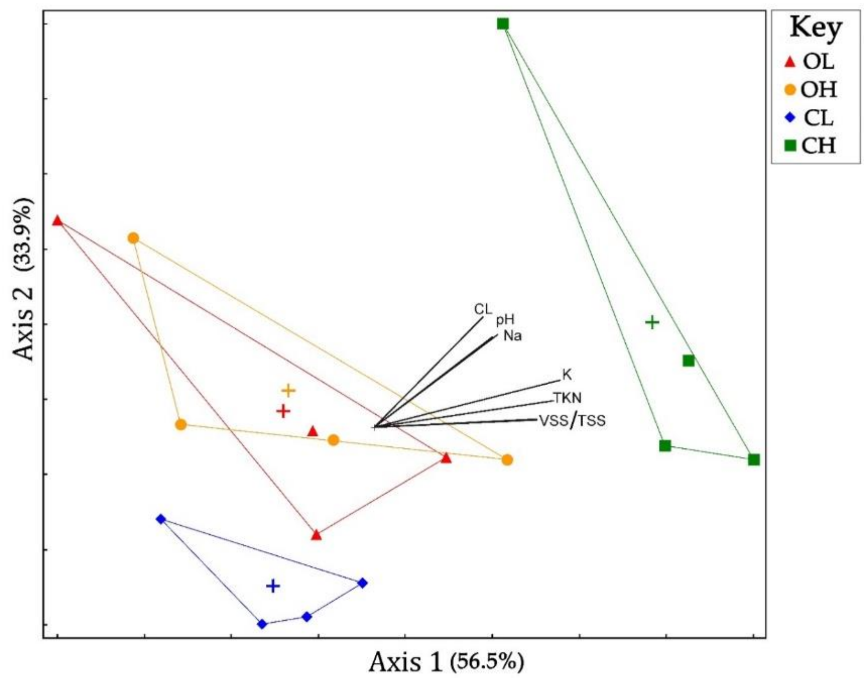

Figure 3. Non-metric multidimensional scaling (NMS) plot of microbial communities (based on relative abundances of bacterial families identified). Only explanatory environmental variables with a combined $r^2 > 0.45$ for both axes are included as vectors. Centroid for each group is marked by (+). O = open; C = cover; L = lagoon; H = house.

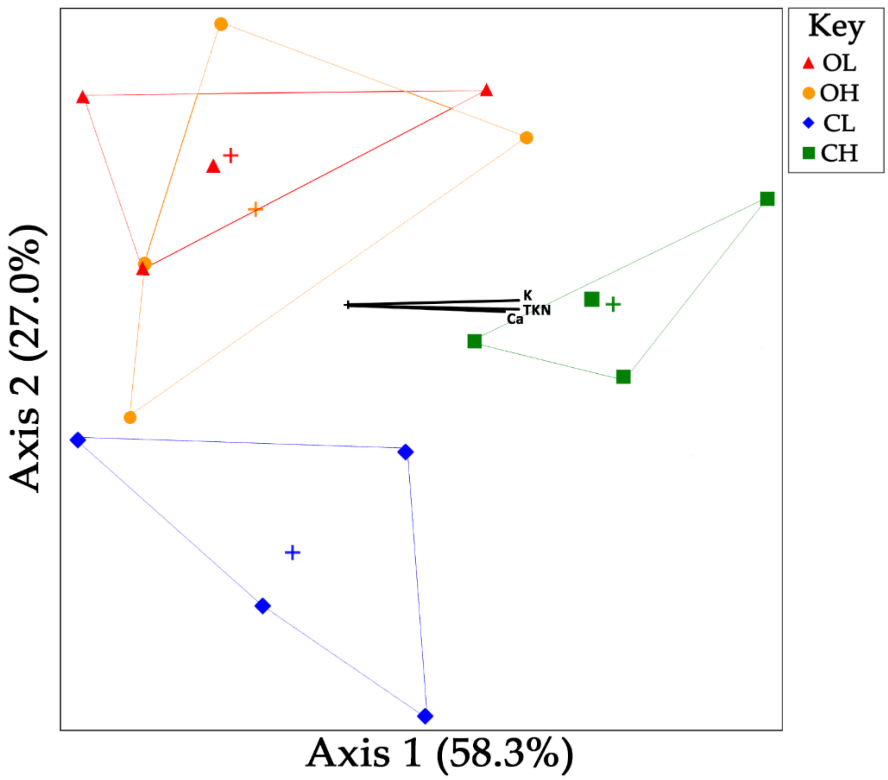

Figure 4. Non-metric multidimensional scaling (NMS) plot of archaeal communities (based on relative abundances of archaeal families identified). Only explanatory environmental variables with a combined $r^2 > 0.45$ for both axes are included as vectors. Centroid for each group is marked by (+). O = open; C = cover; L = lagoon; H = house.

The bacterial populations of both the covered and open lagoons demonstrate similarity to lagoons previously reported [22,25]. Of the 231 families identified in the 16 waste samples collected over the course of the study, using the universal bacterial primer set, only 22 bacterial families (9.5%) were represented in all 16 samplings. However, these bacterial families account for an average of 62.2% (± 4.2% SE; range 33.1% to 86.3%) of the OTU sequences classified in each sample (Figure 5; Supplementary Material Table S2). A total of 34 (14.7%) bacterial families were represented in all 8 lagoon samplings (see Supplementary Material Table S2). Of these 34 families, several were previously reported as being ubiquitous in analyzed anaerobic swine lagoons [22], such as *Ruminococcaceae*, *Chlostridiaceae*, *Lachnospiraceae*, *Peptostreptococcaceae*, and *Synergistaceae*. One noticeable difference is that while previous studies demonstrated high levels of *Chromatiaceae*, in this particularly study, this family went unidentified in the covered lagoon samples. The *Chromatiaceae*, also referred to as purple sulfur bacteria, rely primarily on phototrophic growth [26], and their growth in open lagoons is often quite evident, particularly when the lagoons adopt a purplish to red hue [27]. This family accounted for approximately half the OTU sequences for the open lagoon in the spring (56.3%) and summer (55.0%) samplings (see Figure 5). The greenish tint of the covered lagoon samples compared to the purplish tint of the open lagoon samples during sampling lent support to these findings. These findings potentially correlate with the significantly higher levels of SO_4-S in the open lagoons as compared to the covered lagoons, due to sulfate oxidation by purple sulfur bacteria [28]. These results are similarly reflected in the identification of *Desulfomicrobiaceae*, a family of sulfate reducers [29], only in samples taken from the open lagoon system. Additional sulfate reducers belonging to the families *Desulfobacteraceae* (8 of 8), *Desulfobulbaceae* (8 of 8), and *Desulfovirbrionaceae* (7 of 8) were found in a majority of all lagoon samples [30]. Additionally, while primers 341F and 806R were designed as bacterial-specific, they have been known to pick up archaeal sequences [31]. This led to the identification of the *Methanobacteriaceae*, an archaeal family of hydrogenotrophic (H_2/CO_2) methane (CH_4) producers, and the *Methanosaetaceae*, a family of archaeal acetoclastic methanogens, both of which were found in all 16 samples. The identification of these two archaeal families is of particular importance given the interest of the pork industry to use impermeable lagoon covers to trap methane for energy production in their "manure-to-energy" program initiative [32].

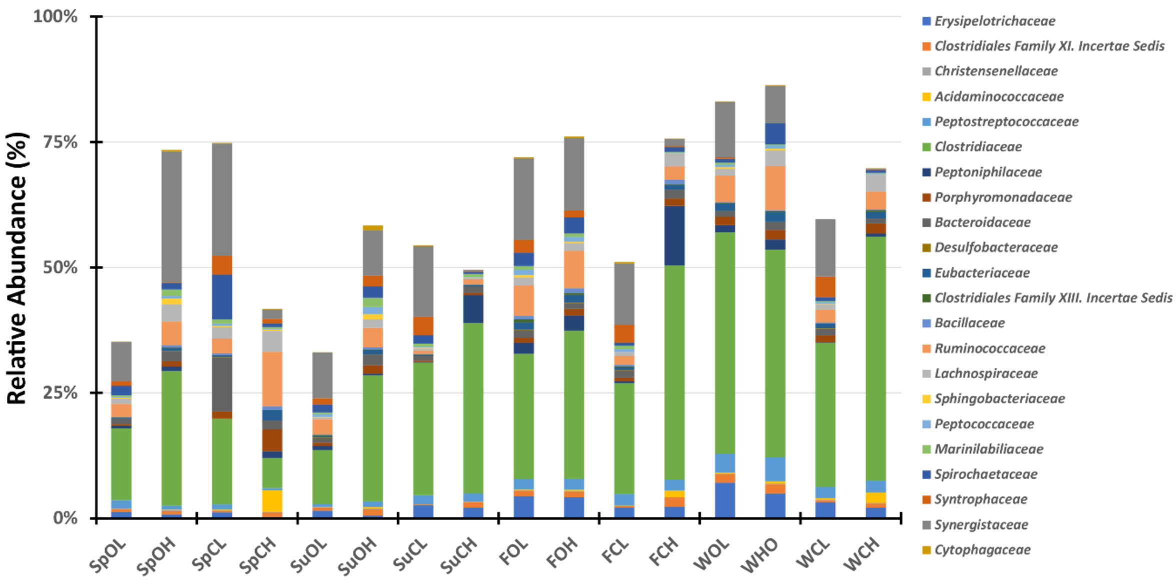

Figure 5. Bacterial community structure, shown as relative abundance. The legend listing selected bacterial families is displayed to the right of the chart. Samples are distinguished by columns. Sp = spring; Su = summer; F = fall; W = winter; O = open; C = cover; L = lagoon; H = house.

A closer look at the archaeal community composition (Figure 6; Supplementary Table S3), using archaeal-specific primers confirmed the presence of OTUs classified to *Methanobacteriaceae* and *Methanosaetaceae,* as well OTUs classified to five other methanogenic archaeal families, identified in all 16 samples: *Methanospirillaceae*, *Methanomicrobiaceae*, *Methanosarcinaceae*, *Methanocorpusculacea*, and the methylotropic *Methanomassiliicoccaceae.* The family *Thermofilaceae* was also identified in all 16 samples, bringing the number of families found in all 16 samples up to eight. When looking at just the eight lagoon samples, a total of 11 families were identified, and include the above-mentioned eight, as well as *Methanpyraceae*, *Methanocalculaceae*, and *Thermococcaceae*. The remaining classified OTUs were assigned to families not found in all samples, and typically found in low percentages (often less than 1%). Examination of archaeal families in relation to sampling source reveals a number of associations (Supplementary Figure S4). For example, both the *Methanosaetaceae* and *Methanoregulaceae* associate closely with the closed lagoon samples, while the *Methanospirillaceae*, *Methanocorpusculaceae*, and *Methanopyraceae* closely associate with the open lagoons and houses. The *Methanobacteriaceae,* on average the most identified archaeal family across all samples (Mean: 34.5%; SE: ± 6.0%), associate most closely with the closed house samples.

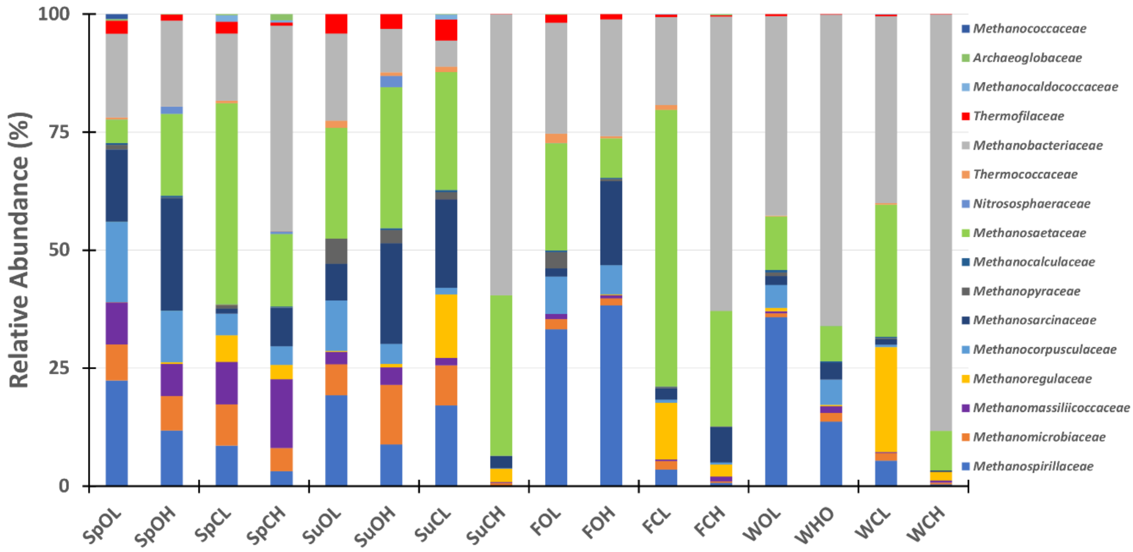

Figure 6. Archaeal community structure, shown as relative abundance. The legend listing specific families is displayed to the right of the chart. Samples are distinguished by columns. Sp = spring; Su = summer; F = fall; W = winter; O = open; C = cover; L = lagoon; H = house.

Of all the archaeal families identified, a majority of the OTUs corresponded to three, *Methanospirillaceae*, *Methanobacteriaceae*, and *Methanosaetaceae*, with the first two classified as being hydrogenotrophic methanogens, and the third classified as acetoclastic methanogens. Overall, our results demonstrate that while hydrogenotrophic methanogens make up the largest segment of methanogens in the two systems studied, acetoclastic methanogens also make up a sizeable portion of the overall methanogenic community. Seasonally, methylotrophic methanogenic OTUs were highest in the spring, acetoclastic methanogenic OTUs were highest in the summer and fall, and hydrogenotrophic methanogenic OTUs peaked in the winter (Figure 7). These OTUs point to both the open and covered lagoons as having significant potential for methane production—a process likely supported by the anaerobic conditions of the lagoons and houses, as indicated by low DO measurements (Table 1).

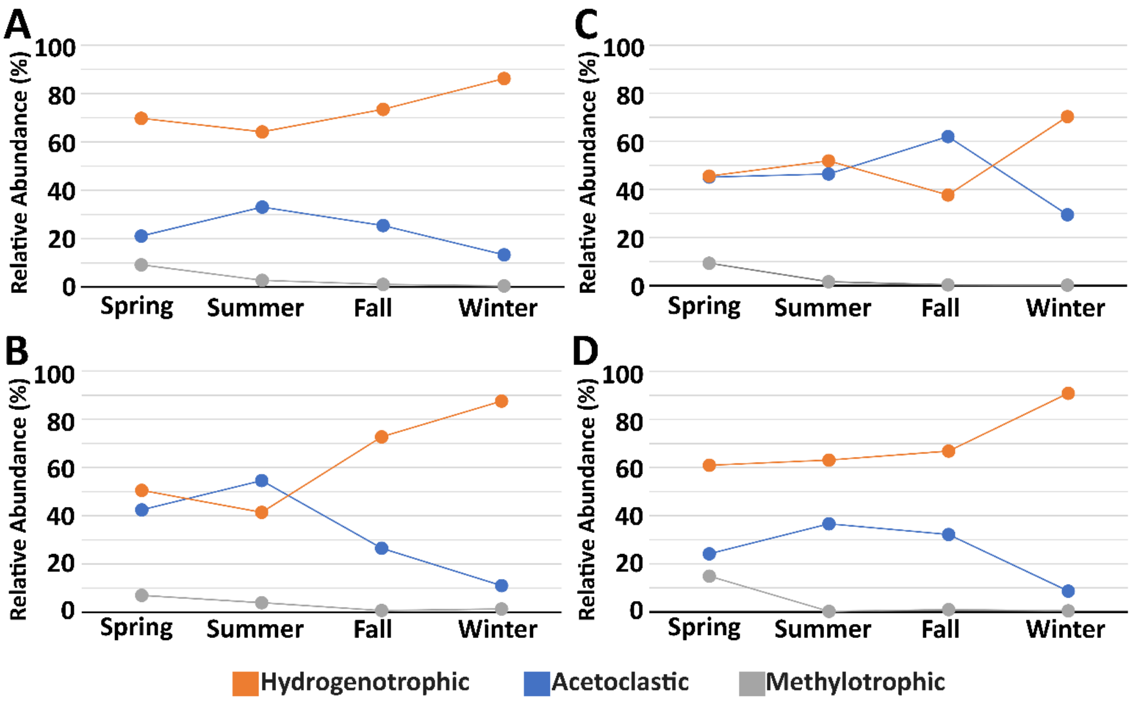

Figure 7. Relative abundance of classified operational taxonomic units (OTUs) potentially involved in methanogenic pathways. (**A**) Open Lagoon; (**B**) Open House; (**C**) Covered Lagoon; (**D**) Covered House.

4. Conclusions

While synthetic covers provide an option for swine producers to reduce odor emissions from anaerobic lagoons, there have been few studies focused on analyzing the biological responses to the microclimates generated at the cover/lagoon interface. Several wastewater physicochemical characteristics demonstrated seasonal variation, while additional differences were seen in comparisons by sampling site (lagoon vs. house) and by the type of lagoon system employed (open vs. covered). Fecal coliforms, *E. coli*, and *Enterococcus* sp., all demonstrated significant relationships with pH. When looking at fecal coliforms and *E. coli*, significant differences in CFU were identified seasonally, by sampling site, and type of lagoon system. Enterococcus sp. were unaffected by the lagoon system employed.

Microbial community analysis identified over 200 bacterial families, with 10.4% represented in all 16 samples, and an additional 19 archaeal families were identified, with eight represented by OTUs in all 16 samples. Evidence for the potential for sulfate-reduction, acetoclastic, hydrogenotrophic, and methylotrophic methanogenesis in the lagoons was demonstrated by the identification of microbial populations responsible for those processes across all lagoon samples. The in-depth sequence analysis of methanogenic communities indicates the potential for—or presence of—methane production from these anaerobic lagoons, although inhibitory concentrations of several nutrients such as Ca and K, need to be accounted for if lagoons are converted for biogas capture with impermeable covers.

Supplementary Materials: The following are available online at http://www.mdpi.com/2076-3298/6/8/91/s1, Figure S4: NMS ordination plot, as seen in Figure 4, demonstrating lagoon and house community structure in relation to individual archaeal family relative abundances, Table S1: Fisher pairwise comparisons of lagoon and house pathogen levels (CFU/gVSS log10), Table S2: Relative abundances of OTUs identified using universal bacterial primer set, presented as relative abundances (%) Only bacterial families are counted in Figure 5 and discussion involving bacterial family identification, Table S3: Relative abundances of OTUs identified using archael primer set, presented as relative abundances (%).

Author Contributions: Individual contributions were as follows: conceptualization and methodology, T.F.D.; investigation, T.F.D. and D.M.C.R.; formal analysis, T.F.D. and A.A.S.; writing—original draft preparation, TFD.; writing—review and editing, T.F.D., D.M.C.R., A.A.S.

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Conflicts of Interest: The authors declare no conflict of interest.

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ATTACHMENT 72

NC Department of Environmental Quality
Division of Water Resources
Animal Feeding Operations

Facilities with Permitted Animal Waste Digesters

Permit No.	Facility Name	County
AWI090025	Storms Farm	Bladen
AWS310014	DM Farms Sec 3 Sites 1-3, Wendy 3-8	Duplin
AWI310015	Magnolia III, DM Section 4 Sites 1-4, Section 3 Sites 4-5	Duplin
AWI310017	DM Farms Sec 2 Sites 1-4	Duplin
AWI310035	Waters Farm 1-5, M&M Rivenbark	Duplin
AWI310039	Benson Farm	Duplin
AWI310048	Stockinghead Creek Farm, LLC Farm	Duplin
AWI310077	Circle K Farm I & II	Duplin
AWI310082	Vestal I&II Farm	Duplin
AWI430029	W. Thomas Butler Farms, LLC	Harnett
AWI510016	Barham Farms Home Farms 48A	Johnston
AWS820005	Kilpatrick Farm 1, 2, 4 & 5 & Merritt Farm	Sampson
AWS820077	Magnolia 4, Melville I & II, DELL DM Section 1 Site 4	Sampson
AWI820466	Farm 2037 and 2038	Sampson
AWI960067	White Oaks Farm Inc	Wayne
AWI990031	Loyd Ray Farms, Inc.	Yadkin

10/12/2021

ATTACHMENT 73

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

EXTERNAL CIVIL RIGHT COMPLIANCE OFFICE
OFFICE OF GENERAL COUNSEL

January 18, 2017

Dear Colleague:

All applicants for and recipients of EPA financial assistance[i] have an affirmative obligation to comply with federal civil rights obligations.[ii] EPA's External Civil Rights Compliance Office (formerly Office of Civil Rights (OCR), within the Office of General Counsel, (ECRCO),[iii] also has a duty to ensure that applicants for and recipients of federal financial assistance ("EPA recipients") comply with federal civil rights laws in their programs or activities that apply for and receive federal financial assistance (including subrecipients of EPA financial assistance). All persons, regardless of race, color, national origin, age, disability or sex, are entitled to receive the benefits of and participation in the programs and activities of EPA recipients without discrimination.[iv] EPA ensures compliance with federal civil rights laws in several ways – through complaint investigations, compliance reviews, technical assistance, community engagement, and policy formulation.[v]

Strong civil rights compliance and enforcement are essential. Furthermore, enforcement of civil rights laws and environmental laws are complementary and can be achieved in a manner consistent with sustainable economic development and that ensures the protection of human health and the environment.

The purpose of this letter is to introduce the **U.S. EPA's External Civil Rights Compliance Office Compliance Toolkit ("Toolkit")**, which is a clarification of existing law and policy intended to provide guidance to promote and support EPA recipients' compliance with federal civil rights laws. With this letter, we are issuing **Chapter 1 of the Toolkit**, which highlights the application of the federal civil rights laws and the legal standards used in investigating and resolving civil rights complaints at EPA. In addition, we are including a companion **Frequently Asked Questions (FAQs)** document to assist in responding to potential questions addressed in Chapter 1.

What is the purpose of the Toolkit?

The overall purpose of the Toolkit is to support and advance our external civil rights compliance and enforcement efforts. We have now finalized the External Civil Rights Compliance Office Strategic Plan for FY 2015-2020 ("Strategic Plan") to promote mission-critical program accountability through measurable goals.[vi] The Strategic Plan is one part of a multi-prong approach to prompt, effective and efficient docket management that includes a Case Resolution Manual together with a Strategic Case Assessment Management Plan,[vii] and deployment of EXCATS,[viii] which is ECRCO's internal docket management system. The Toolkit is previewed in the Strategic Plan to support EPA's goals of enhancing its strategic docket management and developing a proactive compliance program.[ix]

We designed the Toolkit to help you comply with your federal civil rights obligations. The information, guidance, and examples or hypotheticals are intended to assist you in conducting your programs and activities in a nondiscriminatory manner. We created the Toolkit with an understanding that you build a civil rights program around a legal analytical framework that depends upon the legal standards pursued and the nature of facts gathered, such as, direct or indirect/circumstantial evidence. In other words, we recognize that a "one-size-fits-all" approach to civil rights compliance may not adequately address all of your needs. You may have different civil rights concerns in communities within your jurisdiction, different amounts of resources, and different organizational structures.

The Toolkit does not address every scenario that may arise under federal civil rights laws; nor does the Toolkit come with a guarantee that you will not receive a civil rights complaint if you abide by and implement the guidance contained within it. The Toolkit may not apply in a particular situation based upon the circumstances, and EPA retains discretion to adopt approaches on a case-by-case basis that differ from those discussed in the Toolkit where appropriate. Importantly, the Toolkit does not change in any way, your obligation to comply with applicable environmental laws or create any new legal rights or responsibilities.

The Toolkit is a "living document." EPA may revise it from time to time to make improvements, reflect emerging case law or reflect policy changes in EPA's approach to implementing federal civil rights laws.

In introducing the Toolkit, EPA affirms its commitment to work with EPA recipients to achieve their compliance with federal civil rights laws; that is, for recipients to operate and administer their programs and activities in a manner free from discrimination. We look forward to issuing additional Toolkit Chapters that address other civil rights compliance areas. Please do not hesitate to contact me if you have questions relating to the content of this letter and the Toolkit, or if we can otherwise assist you in your federal civil rights compliance efforts.

Sincerely,

Lilian S. Dorka
Director
EPA External Civil Rights Compliance Office
Office of General Counsel

[i] "Applicant means any entity that files an application or unsolicited proposal or otherwise requests EPA assistance." 40 C.F.R. § 7.25. Generally, a recipient means an entity that receives financial assistance from EPA. EPA regulations define recipient as follows:

> Recipient means, for the purposes of this regulation, any State or its political subdivision, any instrumentality of a State or its political subdivision, any public or private agency, institution, organization, or other entity, or any person to which ***Federal financial assistance is extended directly or through another recipient***, including any successor, assignee, or transferee of a recipient, but excluding the ultimate beneficiary of the assistance. 40 C.F.R. § 7.25 (emphasis added).

[ii] *See, e.g.*, http://apply07.grants.gov/apply/forms/sample/SF424B-V1.1.pdf.

[iii] This document generally references EPA throughout. Within EPA, ECRCO is the primary office that enforces federal civil rights laws.

[iv] EPA enforces and ensures compliance with federal civil rights laws that together prohibit discrimination on the bases of race, color, national origin (including limited-English proficiency), disability, sex and age. The five federal civil rights laws that we enforce are as follows: Title VI of the Civil Rights Act of 1964 (42 U.S.C. §§ 2000d *et seq.*); Title IX of the Education Amendments of 1972, as amended (20 U.S.C. §§ 1681 *et seq.*); Section 504 of the Rehabilitation Act of 1973, as amended (29 U.S.C. § 794); Age Discrimination Act of 1975 (42 U.S.C. §§ 6101 *et seq.*); and the Federal Water Pollution Control Act Amendments of 1972, Pub. L. 92-500 § 13, 86 Stat. 903 (codified as amended at 33 U.S.C. §§ 1251 *et seq.* (1972)). *See also* 40 C.F.R. Parts 5 and 7 (EPA's nondiscrimination regulations).

[v] EPA is required to seek the cooperation of applicants and recipients in securing compliance EPA's nondiscrimination regulations and is available to provide help in that regard. 40 C.F.R. § 7.105. Members of the public who believe that he or she or a specific group of persons have been discriminated against may file a complaint alleging discrimination in violation of federal civil rights laws. 40 C.F.R. § 7.120. In such cases, EPA is authorized to investigate and resolve these complaints, as a part of its responsibility to develop and administer a means of ensuring compliance with federal civil rights laws. *See Alexander v. Sandoval*, 532 U.S. 275, 293 (2001) (holding that there is no private right of action to enforce disparate impact regulations promulgated under Title VI). EPA is also authorized to initiate compliance reviews to determine compliance with the civil rights laws enforced by EPA. *See* 40 C.F.R. §§ 7.110, 7.115. This regulatory provision is incorporated by reference in the regulations implementing other statutes enforced by ECRCO. *See* 40 C.F.R. § 5.605. *See also* External Civil Rights Compliance Office Strategic Plan (2015-2020), at 12 (https://www.epa.gov/sites/production/files/2017-01/documents/final_strategic_plan_ecrco_january_10_2017.pdf).

[vi] *See* Strategic Plan at 5.

[vii] See ECRCO Case Resolution Manual (https://www.epa.gov/sites/production/files/2017-01/documents/final_epa_ogc_ecrco_crm_january_11_2017.pdf).

[viii] *See* Strategic Plan at 11 (discussing EXCATS).

[ix] *Id.* at 13.

January 18, 2017

U.S. EPA's EXTERNAL CIVIL RIGHTS COMPLIANCE OFFICE COMPLIANCE TOOLKIT

CHAPTER 1: Application of the federal civil rights laws and the civil rights legal standards used in investigating and resolving civil rights complaints at EPA

I. Who is Covered by Federal Civil Rights Laws?

Federal civil rights laws apply to the programs and activities of applicants for and recipients of federal financial assistance.[1] EPA's nondiscrimination regulation[2] defines a "recipient" to include both public[3] and private entities, such as a State, public or private agency, institution, organization, or other entity or person to which Federal financial assistance is extended.[4]

Applicants for EPA financial assistance must submit an assurance with their applications stating that they will comply with federal civil rights laws.[5] In turn, the acceptance of EPA financial assistance is an acceptance of federal civil rights obligations.[6] Some programs and activities involve more than one recipient of EPA financial assistance. The "primary recipient" is the entity that directly receives the federal financial assistance. The primary recipient then may distribute the funds to a separate entity, known as a "subrecipient,"[7] to carry out a program or activity. Whether you are a primary recipient or subrecipient, you are covered by and must conform your

State Department of Environmental Quality (SDEQ) is the recipient of an EPA Brownfields revolving loan fund grant. SDEQ makes a subgrant to one of its counties, Green County, to carry out cleanup activities at a brownfield site within the county. Therefore, Green County is a subrecipient of EPA financial assistance.

conduct to federal civil rights laws.[8] Generally, a recipient can also include one that is a successor (*e.g.* one who legally acquires the rights and obligations of another through merger, buy-out, or other means), transferee (*i.e.*, one to whom a transfer of property has been made), or assignee (*i.e.*, one to whom an assignment – a transfer of rights – is made) of EPA financial assistance.[9]

As a recipient, you also may not release yourself of your federal civil rights obligations by hiring a contractor or agent to perform or deliver assistance to beneficiaries. EPA's regulations clearly state that prohibitions against discriminatory conduct, whether intentional or through facially neutral means that have a disparate impact, apply to a recipient, whether committed directly or through contractual or other arrangements.[10]

II. What is Covered by Federal Civil Rights Laws?

Civil rights laws prohibit discrimination in "any program or activity" of recipients of federal financial assistance. With regard to certain recipients, such as public institutions, the "program or activity" that Title VI covers encompasses the entire institution and not just the part of the institution that receives the federal financial assistance.[11] For example, many state environmental agencies receive federal funding for their regulatory and environmental protection functions. Those agencies should be aware that all actions, not just permitting decisions, taken by state agencies funded by EPA are subject to federal civil rights laws.

Note: If in a given circumstance you are complying with applicable environmental laws, that fact alone does not necessarily mean that you are complying with federal civil rights laws.

It is also important to note that civil rights laws and environmental laws function separately. Thus if, in a given circumstance, you are complying with applicable environmental laws that fact alone does not necessarily mean that you are complying with federal civil rights laws.

III. Analyzing Discrimination Complaints at EPA

Federal civil rights laws prohibit recipients from intentionally discriminating[12] based on race, color, national origin, disability, sex and age. In addition, federal law authorizes federal agencies to enact "rules, regulations, or orders of general applicability" to achieve the law's objectives.[13] The Supreme Court has held that such regulations may validly prohibit practices that have a *disparate impact* on protected groups. This includes policies, criteria or methods of administering programs that are neutral on their face but have the effect of discriminating.[14] Therefore, both intentionally discriminatory actions (as discussed in section A below) and actions that have the effect of subjecting individuals to discrimination (as discussed in section B below) are prohibited.[15]

In 1973, EPA issued such nondiscrimination regulations and revised them in 1984.[16] Under these regulations, recipients of EPA financial assistance are prohibited from taking actions in their programs or activities that are intentionally discriminatory and/or have a discriminatory effect. EPA regulations also prohibit retaliation and intimidation.[17] No applicant, recipient nor other person may intimidate, threaten, coerce, or engage in other discriminatory conduct against anyone because he or she has either taken action or participated in an action to secure rights protected by the non-discrimination statutes that the EPA External Civil Rights Compliance Office (ECRCO) enforces.[18]

A complainant does not have the burden to cite to specific evidence supporting the claim of discrimination, but may wish to provide supporting information for its complaint. A complainant reports what he or she believes is an act violating federal civil rights laws by an EPA recipient of financial assistance. EPA is not in an adjudicatory role, evaluating evidence produced by opposing sides. Rather, if the jurisdictional criteria in 40 C.F.R. § 7.120 have been established (*see also* ECRCO's Case Resolution Manual, at § 2.4),[19] EPA will investigate the allegations about its recipient to determine if a federal civil rights violation has occurred, even absent specific supporting evidence from a complainant.

A. What constitutes intentional discrimination (disparate treatment)?

Federal civil rights laws prohibit recipients from intentionally discriminating in their programs and activities based on race, color, or national origin, disability, age, or sex. This is also referred to as *disparate treatment*. A claim of intentional discrimination alleges that a recipient intentionally treated individuals differently or otherwise knowingly cause them harm because of their race, color, or national origin, disability, age or sex. Intentional discrimination requires a showing that a "challenged action was motivated by an intent to discriminate."[20] Evidence of "bad faith, ill will or any evil motive of the part of the [recipient] is not necessary.[21] Evidence in a disparate treatment case will generally show that the recipient was not only aware of the complainant's protected status, but that the recipient acted, at least in part, because of the complainant's protected status.[22]

Various methods of proof are available to organize evidence to show whether intentional discrimination has occurred. These methods are described briefly below and one or more of these methods may apply to the facts in an investigation. EPA will evaluate the "totality of the relevant facts" including direct, circumstantial, and statistical evidence to determine whether intentional discrimination has occurred.[23]

The clearest case of intentional discrimination involves direct evidence, such as with a policy or decision that is discriminatory on its face. For example, a policy or decision that includes explicit language requiring individuals or groups of one race to be treated differently from individuals or groups of another race – such as explicitly conditioning the receipt of benefits or services on the race, color, or national origin of the beneficiary – evidences an express classification and thus, direct evidence of intentional discrimination. Comments or conduct by decision-makers that express a discriminatory motive, such as racist or similar discriminatory statements or actions, are also direct evidence that can establish intentional discrimination.

Intentional discrimination also occurs when a policy or decision that is facially neutral (for instance, if the language used does not explicitly differentiate between groups on

SDEQ has a policy on its website stating that it provides fair and equal access to its programs and activities and does not discriminate based on race, color, national origin, disability, sex, age or any other protected category under law. SDEQ is aware that individuals in the community with physical mobility disabilities wish to participate in a public meeting regarding a proposed environmental action; however, SDEQ decides to hold the meeting at a facility that is inaccessible to those individuals because the facility is more centrally located for SDEQ staff. This action, though based on an apparently neutral rationale, may constitute a viable intentional disability discrimination.

the basis of race) is administered by the recipient in a discriminatory manner that is motived, at least in part, by the race, color, national origin, disability, age or sex of the alleged victims of discriminatory treatment.

SDEQ determines to hold a public hearing on the permitting of a controversial landfill in Green County Township. SDEQ decides it will hold public hearings in different sections of the Township to cover the two main areas of town. SDEQ holds two hearings in the East Section, a predominantly white part of the Township and one hearing several miles away in the West Section, a predominately African-American part of the Township.

The East Section hearings are held during the daytime, as well as in the evening after work hours, and both hearings provide three-hour time slots for community comments. The hearing that is held in the West Section is held during the day hours only and limits comments from the community to one hour. Armed security officers also attend the West Section hearing.

SDEQ's decision to hold three public hearings appears to reflect an effort to provide access to all areas of the Township. However, the fact that the hearing in the West Section provides significantly less time for community comment and is scheduled and staffed differently than the two hearings in the East Section raises different treatment concerns. Given these facts, SDEQ's actions may result in a viable claim of disparate treatment based on race.

Direct proof of discriminatory motive is often unavailable. However, EPA will consider both direct and circumstantial evidence of discriminatory intent. For example, evidence to be considered may include:

- statements by decision makers,
- the historical background of the events in issue,
- the sequence of events leading to the decision in issue,
- a departure from standard procedure (*e.g.*, failure to consider factors normally considered),
- legislative or administrative history (*e.g.*, minutes of meetings),
- the foreseeability of the consequences of the action,
- a history of discriminatory or segregated conduct.[24]

Finally, disparate treatment can be shown based on evidence of a substantial disparate impact on a protected group, together with other evidence of motive, such as that listed in the bulleted list above, showing that the recipient acted with discriminatory intent.[25]

SDEQ granted a permit to operate a cement grinding facility. Plaintiffs timely filed an intentional discrimination complaint against SDEQ under Title VI. Plaintiffs alleged: 1) the operation of a cement grinding facility would have a disparate impact upon the predominantly minority community of Waterfront South; 2) SDEQ was well-aware of the potential disproportionate and discriminatory burden placed upon that community and failed to take measures to avoid, minimize, or mitigate that burden; and 3) SDEQ had historically engaged in a statewide pattern and practice of granting permits to polluting facilities to operate in communities where most of the residents are African-American and/or Hispanic to a greater extent than in predominately white communities. These facts may establish circumstantial evidence of intentional discrimination.

An offshore oil spill has caused contamination affecting a Vietnamese community in Green County Township. The spill has contaminated the local beachfront and killed fish and waterfowl. SDEQ does not provide initial response to the incident until four days after receiving notification of the spill, exposing the community to health effects, including stinging eyes, rashes, nausea, dizziness, headaches, coughs and other *respiratory symptoms. The response time has resulted in additional adverse impacts, such as economic impact to the local fishing industry and food supply from the fish kill. During the past few years, SDEQ has provided initial response to six other environmental events, including two oil spills within 12 to 24 hours of being notified. Each of those events occurred in areas outside of the Vietnamese community, in areas with a majority white population. These facts may establish a viable discrimination complaint from the Vietnamese community based on disparate treatment.*

Additionally, in situations where direct proof of discriminatory motive is unavailable, EPA may analyze claims of intentional discrimination using the Title VII[26] burden shifting analytic framework established by the Supreme Court in *McDonnell Douglas Corp. v. Green*.[27] This framework is usually most applicable where a complaint is about one or a few individuals, and involves easily identifiable, similarly-situated individuals not in the protected class. To establish a prima facie case of disparate treatment under the *McDonnel Douglas* framework, EPA must determine that:

(1) the complainant is a member of a protected class;

(2) the complainant was eligible for the recipient's program, activity or service;

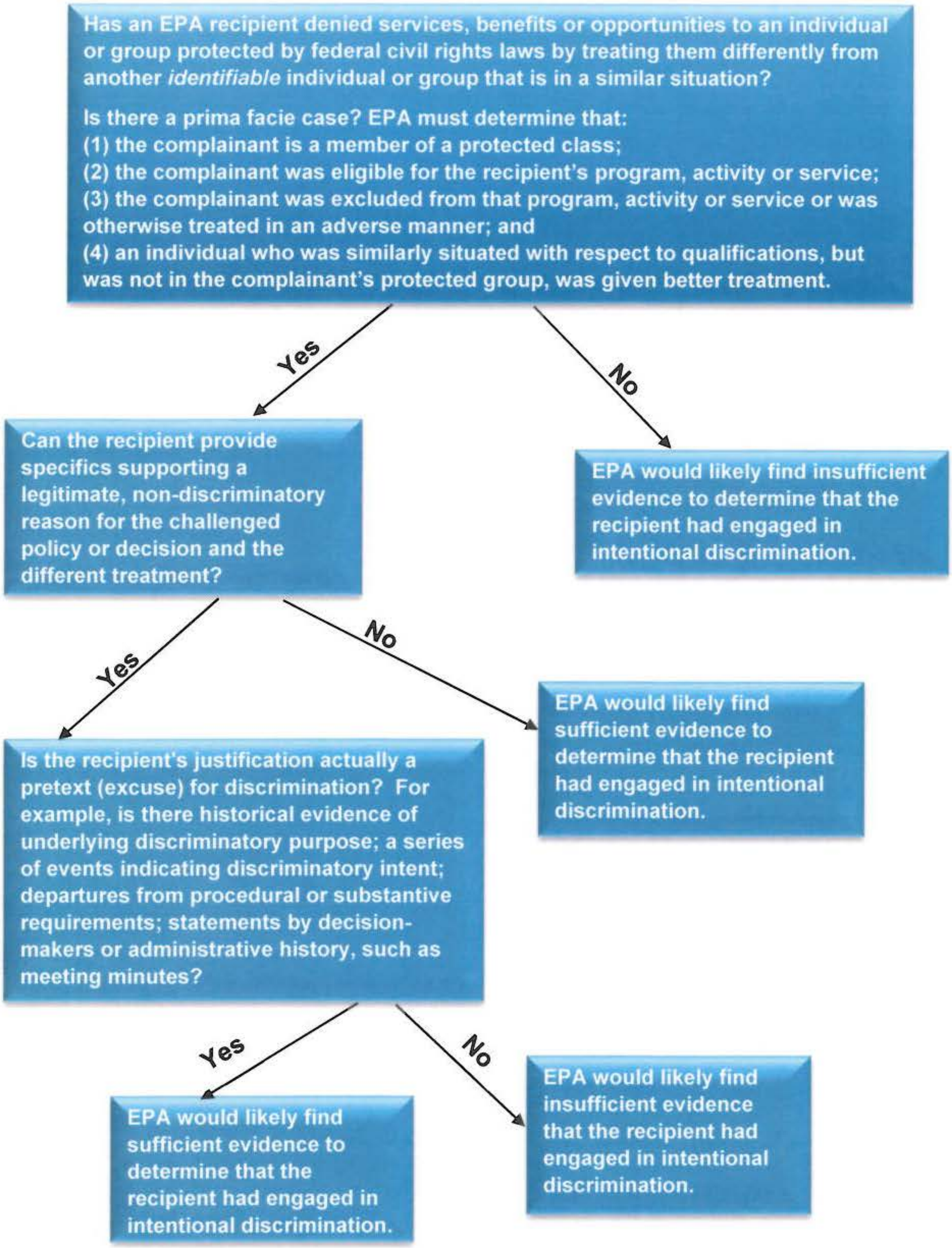
(3) the complainant was excluded from that program, activity or service or was otherwise treated in an adverse manner; **and**

(4) an individual who was similarly situated with respect to qualifications, but was not in the complainant's protected group, was given better treatment.

If a prima facie case of disparate treatment is established, the recipient then has the burden of producing a legitimate, non-discriminatory reason for the challenged policy or decision and the different treatment.[28] If the recipient articulates such a reason, EPA must then determine if there is evidence that the proffered reason is false, *i.e.*, that the nondiscriminatory reason or reasons or the defendant gives for its actions are not the true reasons and are actually a pretext for discriminatory intent.[29]

The chart below illustrates this burden-shifting framework as applied in an administrative complaint.

Intentional Discrimination – McDonnell-Douglas Framework



Similar principles may be used to analyze claims that a recipient has engaged in a *pattern or practice* – or systemic violations – of unlawful discrimination.[30] A showing of more than the mere occurrence of isolated, accidental or sporadic discriminatory acts may prove such claims.[31] In such cases, EPA would look to determine if the recipient regularly engaged in less favorable treatment of a protected group in some aspect of its program as part of its standard policy or operating procedure.[32] A standard policy or operating procedure may be established by a strong statistical disparity that affects a large number of individuals.[33] Statistical evidence can sometimes serve by itself to establish a prima facie case of a pattern or practice of unlawful discrimination[34] but in many cases, statistics are coupled with anecdotal evidence of an intent to treat the protected class unequally.[35] Once the existence of such a discriminatory pattern has been shown, it may be presumed that every disadvantaged member of the protected class was a victim of the discriminatory policy, unless the recipient can rebut the inference that its standard operating policy or operating procedure is discriminatory.[36]

Finally, it is important to understand that establishing that a recipient acted because of race, color, or national origin does not mean that the recipient's actions automatically violate Title VI. Race may be used when a governmental entity has a compelling interest supporting its use, and that use is narrowly tailored to support the stated compelling interest.[37] EPA regulations recognize circumstances under which recipients' consideration of race may be permissible, including providing remedies to those injured by past discrimination.[38]

B. What constitutes disparate impact discrimination?

The second primary method for proving a federal civil rights violation is based on federal nondiscrimination regulations and is known as the *disparate impact* or *discriminatory effects* standard.[39] As noted previously, EPA and other federal agencies are authorized to enact regulations to achieve the law's objectives in prohibiting discrimination. For example, EPA regulations state:

> A recipient shall not use criteria or methods of administering its program or activity which have the effect of subjecting individuals to discrimination. ...[40]

In a disparate impact case, EPA must determine whether the recipient uses a facially neutral policy or practice that has a sufficiently adverse (harmful) and disproportionate effect based on race, color, or national origin. This is referred to as the prima facie case. To establish an adverse disparate impact, EPA must:

(1) identify the specific policy or practice at issue;
(2) establish adversity/harm;[41]
(3) establish disparity;[42] **and**
(4) establish causation.[43]

The focus here is on the consequences of the recipient's policies or decisions, rather than the recipient's intent.[44] The neutral policy or decision at issue need not be limited

SDEQ issued a Clean Air Act permit for the construction and operation of a power station to be located in Green County Township. Although the site for the power station is zoned for industry, the majority of residents in the immediate vicinity of the power station are African-American. If those residents have reason to believe that SDEQ's permitting of the power station will cause them to suffer adverse health impacts at comparatively higher rates than other communities without a significant African-American population, then this may potentially raise a viable disparate impact claim and provide a reason to file a federal civil rights complaint.

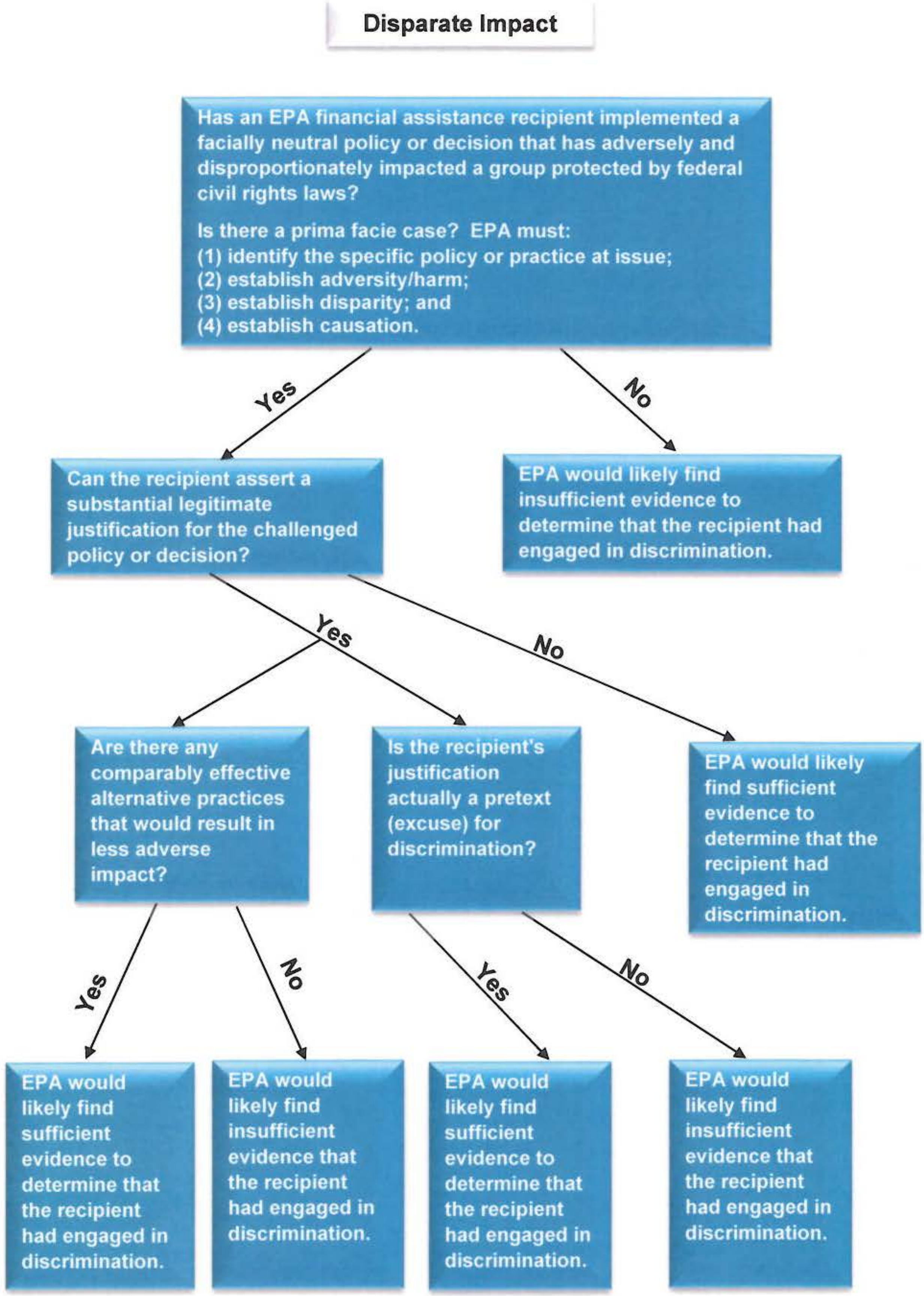
to one that a recipient formalizes in writing, but also could be one that is understood as "standard operating procedure" by recipient's employees.[45] Similarly, the neutral practice need not be affirmatively undertaken, but in some instances could be the failure to take action, or to adopt an important policy.[46]

If the evidence establishes a prima facie case of adverse disparate impact, as discussed above, EPA must then determine whether the recipient has articulated a "substantial legitimate justification" for the challenged policy or practice.[47] "Substantial legitimate justification" in a disparate impact case, is similar to the Title VII employment concept of "business necessity," which in that context requires a showing that the policy or practice in question is demonstrably related to a significant, legitimate employment goal.[48] The analysis requires balancing recipients' interests in implementing their policies with the substantial public interest in preventing discrimination.

Although determining a substantial legitimate justification is a fact-specific inquiry, EPA will generally consider whether the recipient can show that the challenged policy was "*necessary* to meeting a goal that was *legitimate, important, and integral to the [recipient's] institutional mission*" in order to establish a "substantial legitimate justification."[49] EPA will evaluate whether the policy was "necessary" by requiring that the justification bear a "manifest demonstrable relationship" to the challenged policy.[50] As part of its assessment, EPA will generally consider not only the recipient's perspective, but the views of the affected community in its assessment of whether a permitted facility, for example, will provide direct, economic benefits to that community.

If a recipient shows a "substantial legitimate justification" for its policy or decision, EPA must also determine whether there are any comparably effective alternative practices that would result in less adverse impact. In other words, are there "less discriminatory alternatives?"[51] Thus, even if a recipient demonstrates a "substantial legitimate justification," the challenged policy or decision will nevertheless violate federal civil rights laws if the evidence shows that "less discriminatory alternatives" exist.

The chart below illustrates the analysis that EPA utilizes in the investigation of a case involving disparate impact.


Disparate Impact
Has an EPA financial assistance recipient implemented a facially neutral policy or decision that has adversely and disproportionately impacted a group protected by federal civil rights laws?
Is there a prima facie case? EPA must:
(1) identify the specific policy or practice at issue;
(2) establish adversity/harm;
(3) establish disparity; and
(4) establish causation.
Yes
No
Can the recipient assert a substantial legitimate justification for the challenged policy or decision?
EPA would likely find insufficient evidence to determine that the recipient had engaged in discrimination.
Yes
No
Are there any comparably effective alternative practices that would result in less adverse impact?
Is the recipient's justification actually a pretext (excuse) for discrimination?
EPA would likely find sufficient evidence to determine that the recipient had engaged in discrimination.
Yes
No
Yes
No
EPA would likely find sufficient evidence to determine that the recipient had engaged in discrimination.
EPA would likely find insufficient evidence that the recipient had engaged in discrimination.
EPA would likely find sufficient evidence to determine that the recipient had engaged in discrimination.
EPA would likely find insufficient evidence that the recipient had engaged in discrimination.

1. *Disparate Impact: Adversity prong of prima facie case-NAAQS Example*

Referring back to the power station example cited above, this section will focus on the adversity portion of the *prima facie* case analysis, as this issue has been the topic of previous EPA draft guidance papers.[52]

Under these facts, assume that EPA has jurisdiction over a complaint. The complaint alleges that SDEQ's issuance of a construction and operating permit for the power station under its Clean Air Act permitting program has resulted in discrimination. The complaint asserts that SDEQ's action has caused a disparate impact based on alleged adverse health effects that are occurring or will occur from the power station's emission of pollutants for which EPA has established National Ambient Air Quality Standards (NAAQS). In addition, for the purpose of this example, assume that the area in which the power station is located is designated as being in attainment for all of the NAAQS.

In analyzing the complaint, EPA will follow the disparate impact analysis framework in the discussion and chart above. EPA will consider the information provided in the complaint, including any information pertinent to whether the air quality in the area in question does not meet the NAAQS. EPA will examine whether site-specific information demonstrates the presence of adverse health effects from the NAAQS pollutants, even though the area is designated attainment for all such pollutants and the facility recently obtained a construction and operating permit that ostensibly meets applicable requirements.[53] For instance, EPA's assessment would seek to establish whether a localized adverse health impact, as indicated by the NAAQS, exists in the area at issue and has been (or will be) caused by the emissions from the power station even though the impact of the facility had previously been modeled to demonstrate that the source met the criteria for obtaining a construction permit. (Note that some NAAQS, especially those that are source-oriented in nature, are more likely to be associated with localized air quality impacts than those that are more regional.) The localized adverse health impact may result from the increased emissions from the power station, but was not identified at the time of the permit review.

EPA's assessment of such evidence will likely, but not always, be based on gathering pre-existing technical data, including data generated by air quality monitors, general air quality assessments, records from source-specific permitting actions, and information provided by a complainant, rather than EPA generating new data. Such an assessment would not seek to reexamine the characteristics of the NAAQS itself. Rather, EPA's purpose in seeking such evidence is to assess whether a policy or practice of a grant recipient is preventing the area in question from benefiting from the protection of the NAAQS.

Two critical points about the preceding discussion warrant clarification.

1. The fact that the area is designated as in attainment with the NAAQS and that the recent permitting record shows that emissions from the facility would not

cause a violation of the NAAQS would be insufficient by themselves to find that no adverse impacts are occurring for purposes of Title VI and other federal civil rights laws. EPA's investigation would seek to ascertain the existence of such adverse impacts (*e.g.*, violations of the NAAQS) in an area regardless of the area's designation and the prior permitting record. As stated previously, compliance with environmental laws does not necessarily constitute compliance with federal civil rights laws.[54]

2. Complainants do not bear the burden of proving adversity. EPA recognizes that it is responsible for conducting an investigation of the allegations to determine if there is adverse impact.[55]

That said, to the extent that a complainant is able to provide precise allegations and quantified information about the location and nature of the adverse impact from higher-than expected concentrations of the NAAQS pollutant, EPA may be in a better position to conduct a timely and responsive investigation of that complaint. Accordingly, EPA encourages complainants to provide as much information to EPA as they are able to and as early in the process as possible.[56]

For example, a complainant could – but is not required to – provided ECRCO with information that shows a localized adverse health impact based on air monitoring data or air quality modeling that has been prepared using publicly available simple screening tools. (See Air Quality Dispersion Modeling - Screening Models | Support Center for Regulatory Atmospheric Modeling (SCRAM) | US EPA). Complainants may also be able to provide ECRCO with information about relevant university research, or a public interest or industry investigation that has been reported.

EPA will determine if a health-based NAAQS is likely not being met at the location in question, and whether the likely localized violation of a NAAQS is due, at least in part, to the impact of the particular source of air pollution that has recently obtained permits to construct and operate. While the complainant does not bear the burden of proof, any relevant information that the complainant provides could assist the Agency in its analysis.[57]

2. *Disparate Impact – Municipal Solid Waste Landfill Permit example*

SDEQ, which has an approved State program to issue permits to municipal solid waste landfills, renewed a permit to operate a municipal solid waste landfill in State Center, a city in Green County. The facility site is located near neighborhoods that are predominately Latino. Representatives of the neighborhoods filed a civil rights complaint with EPA alleging race and national origin discrimination by SDEQ in reissuing the permit.

The complainants allege the following based on local, recent census data: Green County is 8% Latino and 92% white, African-American and other groups; within State Center, 20% are Latino; and close to the site of the facility, the population is 67% Latino.

Complainants state that during the public participation process leading up to the permitting of the facility, the community raised concerns about anticipated adverse health effects from the facility. Complainants assert that the facility was not appropriately managing waste, which resulted in water run-off polluting the drinking water. Complainants also assert that SDEQ ignored those community concerns. They allege that SDEQ's actions disparately impact Latinos because the Latino population near the facility site is disproportionately affected when compared with other groups in the greater State Center and Green County by adverse health effects stemming from the site. The alleged adverse health effects include headaches; dizziness; burning eyes, nose and throat; nausea; seizures and other chronic illnesses.

In addition, complainants allege that they suffer at a disproportionate level other adverse effects, including economic (e.g., depressed property values); nuisance odors; increased truck traffic and noise; vermin and other vectors.

Given these facts, again assume that EPA has jurisdiction over the complaint and it involves alleged adverse effects that are occurring at the State Center facility, which is regulated under Subtitle D of the Resource Conservation and Recovery Act (RCRA).

In analyzing the complaint, EPA will follow the disparate impact analysis framework in the discussion and chart above. Thus, to find a prima facie violation, EPA's investigation must establish by a preponderance of the evidence that SDEQ's permitting action resulted in adverse and disparate effects on the Latino community identified in the complaint.

EPA will analyze available data, including site-specific data, to determine whether it shows sufficient adverse health effects from site-related pollutants. As mentioned in the NAAQS example, EPA's assessment of health effects will likely, but not always, be based on gathering pre-existing technical data, including information provided by a complainant, rather than generating new data.[58] With respect to the non-health harms alleged (*e.g.*, economic, traffic, noise), Title VI allows agencies to consider whether these effects are occurring and, if so, whether they are sufficiently harmful to support a violation finding.[59]

EPA will consider whether SDEQ's methods of administering the programs at issue subjected the Latino community to disproportionate harm. In evaluating disproportionality, EPA must evaluate population or demographic information of the impacted community as compared to an appropriate comparison population that is similarly situated. The exact areas EPA will evaluate, including distance from the site and specific population centers, will necessarily vary based on the facts and circumstances of each case.

If EPA finds that SDEQ's actions in this case caused adverse and disproportionate impacts on Latinos, SDEQ has the opportunity to justify its permitting actions. To justify the action, the SDEQ must offer evidence that its policy or decision in question is demonstrably related to a significant, legitimate goal related to its mission. For example, have SDEQ's actions resulted in a benefit delivered directly to the affected community, such as public health or environmental benefits? Are there broader interests, such as economic development, as a result of the permitting action that would serve as an acceptable justification? Are the benefits delivered directly to the affected population and is the broader interest legitimate, important, and integral to SDEQ's mission? Will the Latino community, in fact, realize any of these benefits? In evaluating the justification, EPA would likely consider not only SDEQ's perspective, but also the views of the affected community, as appropriate.

Assuming SDEQ establishes such justification, EPA must further look to determine whether there are less discriminatory alternatives; that is, approaches that cause less disparate impact but are practicable and comparably effective in meeting the needs identified by recipient. For example, EPA may find evidence that SDEQ had the capacity to prevent any adverse and disproportionate effects by requiring that the facility be operated in a manner that would eliminate or mitigate its disproportionate impact; by modifying permit conditions or employing practicable mitigation measures to lessen or eliminate the demonstrated adverse disparate impacts; or by not renewing the permit. EPA will also examine whether the asserted justification is merely a pretext or excuse for discrimination.

[1] 40 C.F.R. § 7.15.

[2] 40 C.F.R. Part 7 (https://www.gpo.gov/fdsys/pkg/CFR-2011-title40-vol1/pdf/CFR-2011-title40-vol1-part7.pdf).

[3] A federal agency is not a recipient under federal civil rights laws.

[4] *See* 40 C.F.R. § 7.25.

[5] 40 C.F.R. § 7.80(a)(1).

[6] *Id.*

[7] The term "subrecipient" generally refers to an entity that receives federal financial assistance from EPA through a primary recipient. *See* http://www.grants.gov/web/grants/learn-grants/grant-terminology.html#S (definition of subrecipient).

[8] A recipient is not the same as a beneficiary (*i.e.*, one who is entitled to receive a benefit). An ultimate beneficiary of any program or activity is not considered to be a recipient. *See* 40 C.F.R. § 7.25. *See also U.S. Dep't of Transp. v. Paralyzed Veterans*, 477 U.S. 597, 606-07 (1986). Federal civil rights obligations apply those who receive the aid, but do not apply to those who benefit from the federal financial assistance. *See id.* at 607. Beneficiaries do not enter into any formal contract or agreement with the federal government where compliance with federal civil rights laws is a condition of receiving the federal financial assistance. *See id.* at 605.

[9] *See* 40 C.F.R. § 7.25.

[10] 40 C.F.R. § 7.35(a), (b).

[11] *See* 40 C.F.R. § 7.25.

[12] *See Alexander v. Choate*, 469 U.S. 287, 293 (1985); *Guardians Ass'n. v. Civil Serv. Comm'n*, 463 U.S. 582 (1983). *Alexander* and *Guardians* are Title VI cases. However, Title VI is the model for several subsequent laws that prohibit discrimination on other grounds in federally assisted programs or activities, including Title IX (discrimination in education programs prohibited on the basis of sex) and Section 504 (discrimination prohibited on the basis of disability). *See Paralyzed Veterans*, 477 U.S. at 600 n.4; *Grove City Coll. v. Bell*, 465 U.S. 555, 566 (1984) (Title IX was patterned after Title VI); *Consol. Rail Corp. v. Darrone*, 465 U.S. 624 (1984) (Section 504 patterned after Titles VI and IX). Accordingly, courts have "relied on case law interpreting Title VI as generally applicable to later statutes," *Paralyzed Veterans*, 477 U.S. at 600 n.4.

[13] 42 U.S.C. § 2000d-1.

[14] *Guardians*, 463 U.S. 582; *Alexander*, 469 U.S. at 292-94; *see Elston v. Talladega Cty. Bd. of Educ.*, 997 F.2d 1394, 1406 (11th Cir. 1993). Under the disparate impact analysis, a recipient, in violation of agency regulations, uses a neutral procedure or practice that has a disparate impact on individuals of a particular race, color, or national origin, and such practice lacks a "substantial legitimate justification." *Larry P. v. Riles*, 793 F.2d 969, 983 (9th Cir. 1984); *New York Urban League v. New York*, 71 F.3d 1031,

1038 (2d Cir. 1995); *Elston*, 997 F.2d at 1407. Title VI disparate impact claims are analyzed using principles similar to those used to analyze Title VII disparate impact claims. *Young by and through Young v. Montgomery Cty. (Ala.) Bd. of Educ.*, 922 F. Supp. 544, 549 (M.D. Ala. 1996).

[15] The discussion of legal standards in this document focuses on Title VI because the majority of complaints received by ECRCO allege discrimination based on race, color, or national origin. Importantly, the analyses under other civil rights laws are not always the same. For example, section 504 requires "reasonable accommodation," an obligation not discussed in this chapter. 40 C.F.R. § 7.60.

[16] 38 Fed. Reg. 17968 (1973), as amended by 49 Fed. Reg. 1656 (1984) (codified at 40 C.F.R. Part 7).

[17] Specifically, the regulation states:

> No applicant, recipient, nor other person shall intimidate, threaten, coerce, or discriminate against any individual or group, either:
> (a) For the purpose of interfering with any right or privilege guaranteed by the Acts or this part, or
> (b) Because the individual has filed a complaint or has testified, assisted or participated in any way in an investigation, proceeding or hearing under this part, or has opposed any practice made unlawful by this regulation. 40 C.F.R. § 7.100

EPA plans to issue further information on the prohibition of retaliation and intimidation in the future.

[18] *See* 40 C.F.R. § 7.100. Any individual alleging such harassment or intimidation may file a complaint with EPA. EPA would investigate such a complaint if the situation warranted.

[19] ECRCO Case Resolution Manual, at § 2.4 (https://www.epa.gov/sites/production/files/2017-01/documents/final_epa_ogc_ecrco_crm_january_11_2017.pdf).

[20] *Elston*, 997 F.2d at 1406.

[21] *Williams v. City of Dothan*, 745 F.2d 1406, 1414 (11th Cir. 1984).

[22] Congress has prohibited acts of intentional discrimination based on the protected bases identified in Section I. These protections are statutory, not constitutional, and the analysis under the civil rights statutes at issue here may differ from the different levels of protections the Equal Protection Clause provides to classifications based on sex; disability; and race, color, and national origin.

[23] *See Washington v. Davis*, 426 U.S. 229, 242 (1976).

[24] *See Arlington Heights v. Metro. Hous. Redevelopment Corp.*, 429 U.S. 252 at 266-68 (1977) (evaluation of intentional discrimination claim under the Fourteenth Amendment).

[25] *Elston*, 997 F.2d at 1406. *Arlington Heights,* 429 U.S. at 266-68 (proof of disproportionate impact on an identifiable group can satisfy the intent requirement if it tends to show that some invidious or discriminatory purpose underlies the policy). The first text box example is based on *S. Camden Citizens in Action v. N.J. Dep't of Envtl. Prot.*, 254 F. Supp. 2d 486, 497-498 (D.N.J. 2003) (reversed on other grounds, case history omitted).

[26] Title VII of the Civil Rights Act of 1964 § 7, 42 U.S.C. § 2000e *et seq.* (1964).

[27] *McDonnell Douglas Corp. v. Green*, 411 U.S. 792 (1973). *See Baldwin v. Univ. of Texas Med. Branch at Galveston*, 945 F. Supp. 1022, 1031 (S.D. Tex. 1996); *Brantley v. Indep. Sch. Dist. No. 625, St. Paul Pub. Sch.*, 936 F. Supp. 649, 658 n.17 (D. Minn. 1996).

[28] The recipient's explanation of its legitimate reason(s) must be clear and reasonably specific. Not every proffered reason will be legally sufficient to rebut a prima facie case. *See Texas Dep't of Cmty. Affairs v. Burdine*, 450 U.S. 248, 254-55, 258 (1981).

[29] *See Burdine,* 450 U.S. at 255-56; *Brooks v. Cty. Comm'n of Jefferson Cty.*, 446 F.3d 1160, 1162-63 (11th Cir. 2006).

[30] *See Int'l Bhd. of Teamsters v. United States*, 431 U.S. 324, 336 (1977).

[31] *Id.; EEOC v. Joe's Stone Crab, Inc.*, 220 F.3d 1263, 1286-87 (11th Cir. 2000).

[32] *Teamsters*, 431 U.S. at 336.

[33] *Teamsters*, 431 U.S. at 336, 339 n.20; *Craik v. Minn. State Univ. Bd.*, 731 F.2d 465, 470 (8th Cir. 1984).

[34] *Teamsters*, 431 U.S. at 336; *Hazelwood Sch. Dist. v. United States*, 433 U.S. 299, 307–08 (1977) ("Where gross statistical disparities can be shown, they alone may in a proper case constitute prima facie proof of a pattern or practice of discrimination.").

[35] *Mozee v. Am. Commercial Marine Serv. Co.*, 940 F.2d 1036, 1051 (7th Cir. 1991); *EEOC v. O & G Spring & Wire Forms Specialty Co.*, 38 F.3d 872, 876 (7th Cir. 1994) (citing *Teamsters*, 431 U.S. at 340).

[36] *Teamsters*, 431 U.S. at 361-2.

[37] *Parents Involved in Cmty. Schs. v. Seattle Sch. Dist. No. 1*, 551 U.S. 701, 720 (2007).

[38] 40 C.F.R. § 7.35(a)(7).

[39] *Guardians,* 463 U.S. at 582; *Choate*, 469 U.S. at 293. Many subsequent cases have also recognized the validity of Title VI disparate impact claims. *See Villanueva v. Carere*, 85 F.3d 481 (10th Cir. 1996); *New York Urban League v. New York*, 71 F.3d 1031, 1036 (2d Cir. 1995); *Chicago v. Lindley*, 66 F.3d 819 (7th Cir. 1995); *David K. v. Lane*, 839 F.2d 1265 (7th Cir. 1988); *Gomez v. Illinois State Bd. Of Educ.*, 811 F.2d 1030 (7th Cir. 1987); *Georgia State Conference of Branches of NAACP v. Georgia*, 775 F.2d 1403 (11th Cir. 1985); *Larry P. v. Riles*, 793 F.2d 969 (9th Cir. 1984). United States v. Maricopa Cty, 915 F. Supp. 2d 1073, 1081 (D. Ariz. 2012) (plaintiff properly stated a disparate impact claim where limited-English proficient Latino inmates had diminished access to jail services such as sanitary needs, food, clothing, legal information, and religious services). In addition, by memorandum dated July 14, 1994, the Attorney General directed the Heads of Departments and Agencies to "ensure that the disparate impact provisions in your regulations are fully utilized so that all persons may enjoy equally the benefits of [f]ederally financed programs." Attorney General Memorandum on the use of the Disparate Impact Standard in Administrative Regulations under Title VI of the Civil Rights Act of 1964 (July 14, 1994) (Attorney General July 14, 1994 Memorandum on the use of the Disparate Impact Standard in Administrative Regulations Under Title VI | AG | Department of Justice).

[40] 40 C.F.R. § 7.35(b).

[41] Adversity exists if a fact specific inquiry determines that the nature, size, or likelihood of the impact is sufficient to make it an actionable harm.

[42] In analyzing disparity, EPA analyzes whether a disproportionate share of the adversity/harm is borne by individuals based on their race, color, national origin, age, disability or sex. A general measure of disparity compares the proportion of persons in the protected class who are adversely affected by the challenged policy or decision and the proportion of persons not in the protected class who are adversely

affected. *See Tsombanidis v. W. Haven Fire Dep't*, 352 F.3d 565, 576-77 (2d Cir. 2003). When demonstrating disparity using statistics, the disparity must be statistically significant.

[43] *See N.Y.C. Envtl. Justice All. v. Giuliani*, 214 F.3d 65, 69 (2d Cir. 2000) (plaintiffs must "allege a causal connection between a facially neutral policy and a disproportionate and adverse impact on minorities").

[44] *Lau v. Nichols*, 414 U.S. 563, at 568 (1974).

[45] If as part of a recipient's permitting of a facility, a recipient makes a decision with respect to the siting of a facility; such decision may not intentionally discriminate or have a discriminatory effect on a protected population. The regulation states:

> A recipient shall not choose a site or location of a facility that has the purpose or effect of excluding individuals from, denying them the benefits of, or subjecting them to discrimination under any program or activity to which this part applies on the grounds of race, color, or national origin or sex; or with the purpose or effect of defeating or substantially impairing the accomplishment of the objectives of this subpart. 40 C.F.R. § 7.35(c).

[46] *See, e.g., Maricopa Cty.*, 915 F. Supp. 2d at 1079 (disparate impact violation based on national origin properly alleged where recipient "failed to develop and implement policies and practices to ensure [limited English proficient] Latino inmates have equal access to jail services" and discriminatory conduct of detention officers was facilitated by "broad, unfettered discretion and lack of training and oversight" resulting in denial of access to important services).

[47] *Georgia State Conf.*, 775 F.2d at 1417.

[48] *Wards Cove Packing Inc. v. Antonio*, 490 U.S. 642, 659 (1989); *Griggs v. Duke Power Co.*, 401 U.S. 424, 433-36 (1971). Notably, the concept of "business necessity" does not transfer exactly to the Title VI context because "business necessity" does not cover the full scope of recipient practices that Title VI covers, which applies far more broadly to many types of public and non-profit entities. *See Texas Dept. of Hous. and Cmty. Affairs v. Inclusive Communities Project*, 135 S. Ct. 2507, 2522-24 (2015) (recognizing the limitations on extension of the business necessity concept to Fair Housing Act complaints).

[49] *Elston*, 997 F.2d at 1413 (emphasis added); *See* EPA Draft Revised Guidance for Investigating Title VI Administrative Complaints Challenging Permits, 65 Fed. Reg. 39,667, 39683 (2000) (Draft Revised Investigation Guidance) ("Determining what constitutes an acceptable justification will necessarily be based on the facts of the case. Generally, the recipient would attempt to show that the challenged activity is reasonably necessary to meet a goal that is legitimate, important, and integral to the recipient's institutional mission.") (https://www.epa.gov/sites/production/files/2013-09/documents/frn_t6_pub06272000.pdf).

[50] Georgia State Conf., 775 F.2d. at 1418.

[51] *Elston*, 997 F.2d at 1407.

[52] In its 2000 *Draft Revised Investigation Guidance*, EPA stated that a demonstration in the permitting context that construction of a stationary source will not cause a violation of health-based NAAQS creates a rebuttable presumption that no adverse impacts are caused by the environmental permit at issue with respect to the relevant NAAQS pollutant for purposes of Title VI. That presumption could be overcome with other relevant information about the area. *See* Draft Revised Investigation Guidance, 65 Fed. Reg. at 39,680-81. Stakeholders raised concerns that EPA should more clearly distinguish between environmental compliance and civil rights compliance. Consequently, in 2013, EPA proposed to clarify that the Agency would no longer apply a rebuttable presumption in such a context and instead would consider whether an area was attaining NAAQS concurrently with other information, such as the

presence of "hot spots." *See* Adversity and Compliance with Environmental Health-Based Thresholds, 78 Fed. Reg. 24,739 (2013) (https://www.gpo.gov/fdsys/pkg/FR-2013-04-26/pdf/2013-09922.pdf). Following its review of comments on the 2013 draft, as well as subsequent external engagement with interested stakeholders, EPA will apply the approach described here. This approach supersedes the corresponding discussions in the two prior Federal Register notices and eliminates application of the rebuttable presumption.

Both prior positions and the approach described here are predicated on the application of health-based environmental standards such as the NAAQS. Under the Clean Air Act, a primary NAAQS must, in the judgment of the Administrator, protect public health with an adequate margin of safety. This judgment is based on a thorough review of the available scientific literature, including assessments of sensitive sub-populations. The NAAQS and its underlying science are then reviewed periodically to ensure that they remain sufficiently protective. Implementation of a NAAQS requires proper characterization of air quality, generally involving the use of ambient monitors over time, in order to determine whether the NAAQS are being met.

[53] Separately, complainants who believe the permits were issued in error may seek to appeal those permit decisions under administrative or judicial procedures applicable under a state permitting program. In addition, parties may petition EPA to object to a Title V operating permit. These procedures and remedies are distinct from a complaint under civil rights laws, and they are not addressed in the Toolkit.

[54] *See, e.g.*, 78 Fed. Reg. at 24,742; 65 Fed. Reg. at 39,680 (2000).

[55] EPA will exercise its reasonable enforcement discretion to determine whether a violation has occurred.

[56] In evaluating and receiving a complaint and supporting information from complainants, ECRCO will assist the complainant in understanding ECRCO's jurisdiction and the complainant's nondiscrimination rights under the statutes and regulations enforced by ECRCO according to its Role of Complainants and Recipients in the Title VI Complaints and Resolution Process policy paper (May 4, 2015) (https://www.epa.gov/ocr/epas-title-vi-policies-guidance-settlements-laws-and-regulations) and Case Resolution Manual (https://www.epa.gov/sites/production/files/2017-01/documents/final_epa_ogc_ecrco_crm_january_11_2017.pdf).

[57] This example addresses how compliance with environmental health-based thresholds relates to "adversity" in the context of disparate impact claims about environmental permitting. The approach described here does not address allegations about intentional discrimination, allegations about pollutants that are not addressed by NAAQS, most non-permitting fact patterns, or technology- and cost-based standards. However, the principle described here or another similar approach may apply in other contexts where appropriate. Furthermore, this approach in no way diminishes EPA's emphasis on informal resolutions of federal civil rights complaints, which may be undertaken before completion of the analysis described here. In addition, as outlined above, adversity is only one part of the disparate impact analysis.

[58] ECRCO may give due weight to relevant adverse impact analyses and disparity analyses submitted by recipients or complainants that, at a minimum, generally conform to accepted scientific approaches. The weight that ECRCO gives to any evidence and the extent to which ECRCO may rely on it in its decision will likely vary depending upon:

- relevance of the evidence to the alleged impacts;
- the validity of the recipient's methodologies;
- the completeness of the documentation that is submitted by the recipient;
- the degree of consistency between the methodology used and the findings and conclusions; and
- the uncertainties of the input data and results.

Consequently, EPA experts would undertake a scientific review of submitted materials. If the analyses submitted meet the factors above, ECRCO will not seek to duplicate or conduct such analyses, but instead will evaluate the appropriateness and validity of the relevant methodology and assess the overall reasonableness of the outcome or conclusions at issue.

If ECRCO's review reveals that the evidence contains significant deficiencies with respect to the factors above, then the analysis will likely not be relied upon in ECRCO's decision.

[59] EPA has substantial discretion to determine the types of harms, on a case by case basis, that warrant investigatory resources and are sufficiently harmful to violate Title VI: "Title VI had delegated to the agencies in the first instance the complex determination of what sorts of disparate impact upon minorities constituted sufficiently significant social problems, and were readily enough remediable, to warrant altering the practices of the federal grantees that had produced those impacts." *Choate*, 469 U.S. at 293–94; *see also Alexander v. Sandoval*, 532 U.S. 275, 306 (2001) (Stevens, J., dissenting). And lower courts have consistently recognized and deferred to agency interpretations of the disparate impact standard. *See, e.g.*, *United States v. Maricopa Cty*, 915 F. Supp. 2d 1073, 1080 (D. Ariz. 2012) (citing *Auer v. Robbins*, 519 U.S. 452, 461 (1997)) (agency interpretation of its own regulations "controlling unless plainly erroneous or inconsistent with the regulations"). Historically, EPA has focused primarily on those impacts that could fall under a recipient's authority.

January 18, 2017

FREQUENTLY ASKED QUESTIONS (FAQs) FOR CHAPTER 1 OF THE U.S. EPA'S EXTERNAL CIVIL RIGHTS COMPLIANCE OFFICE COMPLIANCE TOOLKIT

1) Why is EPA issuing a Civil Rights Compliance Toolkit ("Toolkit")?

The overall purpose of the Toolkit is to support and advance the External Civil Rights Compliance Office's (ECRCO) proactive compliance and enforcement efforts. ECRCO ensures that applicants for and recipients of EPA federal financial assistance comply with federal civil rights laws[1] in their programs or activities in several ways –through complaint investigations, compliance reviews, technical assistance, community engagement, and policy formulation. Accordingly, EPA is issuing the Toolkit to clarify existing law and policy and to provide guidance to promote and support applicant and recipient compliance with federal civil rights laws.

In issuing the Toolkit, EPA affirms its commitment to work with its financial assistance applicants and recipients to help achieve their compliance with federal civil rights laws, that is, that applicants for and recipients of financial assistance operate and administer their programs and activities in a manner free from discrimination. The Toolkit also provides members of the public with information about the civil rights laws and implementing regulations that ECRCO enforces and how those laws are enforced.

2) What does the Toolkit contain?

The Toolkit contains information and policy guidance to inform recipients about how EPA evaluates whether they are complying with their legal obligations pursuant to federal civil rights laws,[2] including through discussion and clear examples of the application of foundational civil rights legal standards (*i.e.*, intentional discrimination and disparate impact) used in investigating and resolving civil rights complaints at EPA.

The Toolkit is a "living document." EPA may revise it from time to time to make improvements, reflect emerging case law or reflect policy changes in EPA's approach to implementing federal civil rights laws.

3) Who is covered by federal civil rights laws?

Federal civil rights laws apply to the programs and activities of applicants for and recipients of federal financial assistance[3] as well as any subrecipients[4] who receive funds from a recipient to carry out its programs and activities. EPA's nondiscrimination regulation defines a recipient to include both public and private entities, including any State, public or private agency, institution, organization, or other entity or person to which federal financial assistance is extended.[5]

4) What is covered by federal civil rights laws?

Civil rights laws prohibit discrimination in "any program or activity" of recipients of federal financial assistance. With regard to certain recipients, such as public institutions, the "program or activity" that Title VI covers encompasses the entire institution and not just the part of the institution that receives the federal financial assistance.[6] For example, many state environmental agencies receive federal funding for their regulatory and environmental protection functions. Those agencies should be aware that all actions, not just permitting decisions, taken by state agencies funded by EPA are subject to federal civil rights laws.

5) What conduct is prohibited by federal civil rights laws and EPA's nondiscrimination regulation?

Recipients of EPA financial assistance are prohibited from taking actions in their programs or activities that are intentionally discriminatory and/or have a discriminatory effect. Violations of federal civil rights laws can result not only from intentional discrimination, but from discrimination based on disparate impact, *i.e.*, policies and practices that are neutral on their face, but have the effect of discriminating.[7] In addition, recipients may not intimidate, threaten, coerce, or engage in other discriminatory conduct against anyone because he or she has either taken action or participated in an action to secure rights protected by the non-discrimination statutes ECRCO enforces.[8]

6) What is intentional discrimination?

Intentional discrimination (or different treatment) occurs when a recipient intentionally treated individuals differently or otherwise knowingly cause them harm because of their race, color, national origin, disability, age or sex. Intentional discrimination requires a showing that a challenged action was motivated by an intent to discriminate but does not require showing bad faith, ill will, or evil motive.

7) What is disparate impact?

Disparate impact (or discriminatory effect) results when a recipient uses a facially neutral procedure or practice that has a significantly adverse (harmful) and disproportionate effect based on race, color, or national origin. In a disparate impact case, the focus is on the consequences of the recipient's policies or decisions, including the failure to take action, rather than the recipient's intent.

If there is evidence of adverse disparate impact, EPA must then determine whether the recipient has asserted a "substantial legitimate justification" for the challenged policy or practice. "Substantial legitimate justification" in a disparate impact case, is similar to the Title VII concept of "business necessity," which requires a showing that the policy or practice in question is demonstrably related to a significant, legitimate employment goal.[9] The analysis requires balancing recipients' interests in implementing their policies with the substantial public interest in preventing discrimination. If there is no such showing, EPA would likely find that the recipient has engaged in discrimination. If the recipient makes such an assertion, EPA must also determine if there are any "equally effective alternative practices" that would result in less adverse impact and/or whether the asserted justification is not just an excuse for discrimination. If EPA makes such a determination about available alternatives or finds pretext, it would likely find that discrimination occurred.

8) What legal standard does EPA apply in its civil rights investigations?

EPA utilizes the "preponderance of the evidence" (more likely than not) standard in its investigations to determine whether or not a recipient has violated federal civil rights laws.

9) Does compliance with environmental laws in a given situation equate to compliance with federal civil rights laws?

No. If in a given circumstance a recipient is in compliance with applicable environmental laws that fact alone does not necessarily mean that the recipient is in compliance with federal civil rights laws.

10) Does the EPA apply a "rebuttable presumption" to the adversity prong of its disparate impact analysis?

EPA addresses this issue directly in the Toolkit through an example involving issuance of permits authorizing construction and operation of a power station. To put this question in context, EPA, in its 2000 *Draft Revised Investigation Guidance*, stated that a demonstration in the permitting context that construction of stationary source will not cause a violation of the health-based National Ambient Air Quality Standards (NAAQS) creates a rebuttable presumption that no adverse impacts are caused by the environmental permit at issue with respect to the relevant NAAQS pollutant for purposes of Title VI.[10] In 2013, EPA proposed to clarify that the Agency would no longer apply a rebuttable presumption in such a context and instead would consider whether an area was attaining NAAQS concurrently with other information, such as the presence of "hot spots."[11]

Following its review of comments on the 2013 draft, as well as subsequent external engagement with interested stakeholders, EPA will apply the approach to adversity that is discussed in the Toolkit. Specifically, EPA will examine whether site-specific information demonstrates the presence of adverse health effects from NAAQS pollutants, even though the area is designated attainment for all such pollutants and the facility recently obtained a construction and operating permit that ostensibly meets applicable requirements. EPA's assessment would seek to establish whether a localized adverse health impact, as indicated by the NAAQS, exists in the area at issue and has been (or will be) caused by the emissions from the power station even though the impact of the facility had previously been modeled to demonstrate that the source met the criteria for obtaining a construction permit. As stated previously, compliance with environmental laws does not necessarily constitute compliance with federal civil rights laws.

While the adversity example in the Toolkit involves permits authorizing construction and operation of a power station, the approach described here or another similar approach may apply in other contexts where appropriate. Ultimately, this approach supersedes the corresponding discussions in the two prior Federal Register notices and eliminates application of the rebuttable presumption.

11) What types of harm does EPA consider when determining whether there has been an adverse and disproportionate impact on individuals?

EPA's nondiscrimination regulation does not define discriminatory effects but simply states that a recipient may not administer its program or activity in a manner which has the effect of subjecting individuals to discrimination because of their race, color, national origin, age, disability status, or sex.[12] This language encompasses a broad range of effects caused by a recipient's administration of its program. Therefore, in analyzing a claim of disparate impact, EPA will consider environmental harms and adverse health effects (*e.g.*, asthma and other respiratory illnesses, cancer, cardiac disease, stroke, allergies, *etc.*) that have allegedly been caused disproportionately based on race, color, or national origin, by a recipient's policy or practice. EPA will also consider non-health harms, including, among other things, economic (*e.g.,* depressed property values), nuisance odors, traffic congestion, noise and vermin. With respect to the non-health harms alleged (*e.g.*, economic, traffic, noise), Title VI allows agencies to consider whether these effects are occurring and, if so, whether they are sufficiently harmful to support a violation finding.[13]

[1] EPA's ECRCO is responsible for enforcing several civil rights laws which, together, prohibit discrimination on the basis of race, color, national origin (including on the basis of limited-English proficiency), sex, disability, and age, by applicants for and recipients of federal financial assistance from EPA.

[2] Note: The Toolkit is a guidance document and does not add requirements to applicable federal civil rights laws. The Toolkit is not a rule; it is not legally enforceable; and it does not create or confer legal rights or legal obligations upon any member of the public, recipient, the EPA, state and local governments, tribes, or any other agency. For instance, it includes references to statutes, regulations and case law, but it does not change or substitute for any legal requirements contained in those sources. While EPA has made every effort to ensure the accuracy of the information discussed in the Toolkit, the relevant statutes, regulations, and other legally binding requirements determine your obligations as a recipient. In the event of a conflict between the discussion in the Toolkit and any statute or regulation, the Toolkit would not control.

The Toolkit does not address every scenario that may arise under federal civil rights laws; nor does the Toolkit come with a guarantee that you will not receive a civil rights complaint if you abide by and implement the guidance contained within it. The Toolkit may not apply in a particular situation based upon the circumstances, and EPA retains discretion to adopt approaches on a case-by-case basis that differ from those discussed in the Toolkit where appropriate. Importantly, the Toolkit does not change in any way, your obligation to comply with applicable environmental laws.

[3] 40 C.F.R. § 7.15.

[4] The term "subrecipient" generally refers to an entity that receives federal financial assistance from EPA through a primary recipient. *See* http://www.grants.gov/web/grants/learn-grants/grant-terminology.html#S (definition of subrecipient).

[5] *See* 40 C.F.R. § 7.25.

[6] *Id.*

[7] The discussion of legal standards in this document and the Toolkit, generally, focuses on Title VI because the majority of complaints received by ECRCO allege discrimination based on race, color, or national origin. Importantly, the analyses under other civil rights laws are not always the same. For example, section 504 requires "reasonable accommodation," an obligation not discussed in this chapter. 40 C.F.R. § 7.60.

[8] *See* 40 C.F.R. § 7.100.

[9] *Wards Cove Packing Inc. v. Antonio*, 490 U.S. 642, 659 (1989); *Griggs v. Duke Power Co.*, 401 U.S. 424, 433-36 (1971). Notably, the concept of "business necessity" does not transfer exactly to the Title VI context because "business necessity" does not cover the full scope of recipient practices that Title VI covers, which applies far more broadly to many types of public and non-profit entities. *See Texas Dep't of Hous. and Cmty. Affairs v. Inclusive Communities Project*, 135 S. Ct. 2507, 2522-24 (2015) (recognizing the limitations on extension of the business necessity concept to Fair Housing Act complaints).

[10] *See* Draft Revised Guidance for Investigating Title VI Administrative Complaints Challenging Permits, 65 Fed. Reg. 39,667, 39,680-81 (June 27, 2000).

[11] *See* Adversity and Compliance with Environmental Health-Based Thresholds, 78 Fed. Reg. 24,739 (April 26, 2013).

[12] 40 C.F.R. § 7.35(b).

[13] EPA has substantial discretion to determine the types of harms, on a case by case basis, that warrant investigatory resources and are sufficiently harmful to violate Title VI: "Title VI had delegated to the agencies in the first instance the complex determination of what sorts of disparate impact upon minorities constituted sufficiently significant social problems, and were readily enough remediable, to warrant altering the practices of the federal grantees that had produced those impacts." *Alexander v. Choate*, 469 U.S. 287, 293-94 (1985); *see also Alexander v. Sandoval*, 532 U.S. 275, 306 (2001) (Stevens, J., dissenting). And lower courts have consistently recognized and deferred to agency interpretations of the disparate impact standard. *See, e.g.*, *United States v. Maricopa Cty*, 915 F. Supp. 2d 1073, 1080 (D. Ariz. 2012) (citing *Auer v. Robbins*, 519 U.S. 452, 461 (1997)) (agency interpretation of its own regulations "controlling unless plainly erroneous or inconsistent with the regulations"). Historically, EPA has focused primarily on those impacts that could fall under a recipient's authority.

ATTACHMENT 74

Julius L. Chambers Center for Civil Rights
What great things are you working on?

THE ENVIRONMENTAL LAW CLINIC
MORNINGSIDE HEIGHTS LEGAL SERVICES, INC.
COLUMBIA UNIVERSITY SCHOOL OF LAW
435 WEST 116TH STREET • NEW YORK, NY 10027

EARTHJUSTICE

<u>CONFIDENTIAL ATTORNEY WORK PRODUCT– DRAFT</u>
March 4, 2019
Via e-mail to Christine.Lawson@ncdenr.gov and Swinepermit.comments@ncdenr.gov
DWR Animal Operations
Attn: Swine General Permit
1636 Mail Service Center
Raleigh, NC. 27699-1636

Re: Draft Swine Waste Management System General Permit (AWG100000)

Dear DWR Animal Operations:

We submit these comments on the draft Swine Waste Management System General Permit (AWG100000) ("Draft Permit") on behalf of the North Carolina Environmental Justice Network ("NCEJN"), the Rural Empowerment Association for Community Help ("REACH"), Waterkeeper Alliance, Inc. ("Waterkeeper"), the Crystal Coast Waterkeeper and the North Carolina State Conference of the NAACP. To avoid redundancy, we incorporate by reference the attached comments submitted on our clients' behalf on December 21, 2018, as part of the stakeholder process.[1]

As an initial matter, and in connection with our representation of NCEJN, REACH and Waterkeeper in the 2014 administrative complaint alleging racial discrimination filed against DEQ with the U.S. Environmental Protection Agency (EPA) under Title VI of the Civil Rights Act of 1964 ("Title VI complaint"), we refer you to the "Background" section of those comments, which summarizes information provided to DEQ in 2013 concerning the racially discriminatory impacts of the current general permit.[2] Those impacts include air and water pollution emanating from the open pits of waste and sprayfields in which DEQ-permitted swine operations store and disperse billions of gallons of feces, urine and other waste. The consequences of this system are not just environmental, but also racially discriminatory, because they disproportionately burden non-white North Carolinians.

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[1] *See NCEJN, REACH, and Waterkeeper Alliance Stakeholder Comments on General Permit (AWG 100000) (Dec. 21,*2018). These comments are herein incorporated by reference and attached hereto as Exhibit 1. *See also Sound Rivers, Cape Fear River Watch, Winyah Rivers Foundation, Crystal Coast Waterkeeper, Crystal Coast Riverwatch, White Oak-New Riverkeeper Alliance, Haw River Assembly, Yadkin Riverkeeper, Catawba Riverkeeper Foundation, Broad River Alliance, and Mountain True, Stakeholder Comments on General Permit (AWG 100000) (Dec. 21, 2018); National Resources Defense Council and Center for Biological Diversity, Stakeholder Comments on General Permit (AWG 100000)* (Dec. 21, 2018), which are herein incorporated by reference and attached hereto as Exhibits 2 and 3.
[2] *See* Complaint by NCEJN, REACH, and Waterkeeper Alliance Against North Carolina DEQ (EPA File No. 11R-14-R4), September 3, 2014 (hereinafter, "Complaint," attached hereto as Exhibit 4).

DEQ continues violating Title VI because the agency has failed to exercise its authority to provide adequate protections for the health and welfare of surrounding communities and, knowing the risks and impacts of the lagoon and sprayfield system in eastern North Carolina, failed to exercise its duty to include terms to identify and protect those communities in the draft Swine General Permit.We urge DEQ to begin planning now for the transition of North Carolina's swine industry from the antiquated lagoon and sprayfield system to a more sustainable foundation for farming in the state; at a minimum, though, the Draft Permit must comply with the mandates of Title VI, and should be changed to include the following:

1) At page 1, an additional provision should be added to the new section listing the "[r]easons for requiring application for an individual permit," that expressly recognizes the agency's mandate to comply with Title VI, as follows:
   - Subsection (h) "*a determination that the operation contributes to cumulative and/or discriminatory impacts on communities of concern.* [3]

2) Also at page 1, the permit duration period should be shortened from 5 to 2 years ("*This General Permit shall be effective from October 1, 2019 until September 30, 2021*"). This change is necessary given DEQ's public representation that the Environmental Justice geographic mapping tool ("EJ Tool") it is developing will not be ready to conduct a disparate impact and cumulative impacts analysis of facilities operated by applicants for the 2019 General Permit. Limiting the permit's duration to two years would demonstrate DEQ's bona fide commitment to implement the EJ Tool and conduct the necessary analyses within the next two years, make appropriate changes to the 2021 General Permit, and require swine operators to adopt less discriminatory alternative means of waste disposal within a reasonable timeframe. The 2-year time period would also afford the opportunity for DEQ to consider additional evidence of impacts by considering PLAT and groundwater monitoring data envisioned in this draft permit, as well as water monitoring data now collected pursuant to the Settlement Agreement DEQ reached with NCEJN, REACH, and Waterkeeper.[4]

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[3] Our definition of "communities of concern" is based upon EPA guidelines, which refer to "populations of concern" and "vulnerability" defined by "characteristics of individuals or populations that place them at increased risk of an adverse health effect.*" See Risk Assessment Forum, U.S. Envtl. Prot. Agency, Framework for Cumulative Risk Assessment*, EPA/630/P-02/001F (2003). EPA considers a number of factors to determine vulnerability: susceptibility/sensitivity, differential exposure, differential preparedness, and differential ability to recover. *Id.,* at 39. In the North Carolina context, these factors would include the density of hog operations and proximity to other polluting sources, such as poultry operations.
[4] *See* Settlement Agreement, Case 11R-14-R4, attached hereto as Exhibit 5; DEQ, Stocking Head Creek Watershed Study, https://deq.nc.gov/about/divisions/water-resources/water-resources-data/water-sciences-home-page/shc-study.

In addition to and in support of the above requests, and in light of DEQ's response to the changes we requested during the stakeholder process to protect communities of concern, the pork industry's responses to those requests, and the subsequent changes in the Draft Permit, we direct DEQ's consideration to the following:

1. **EPA's "Letter of Concern"**

In January 2017, the EPA issued a "letter of concern" to DEQ as part of its investigation of our clients' Title VI complaint.[5] The 23-page letter describes results not only from EPA's site visits to Duplin, Sampson, Northampton and Pender Counties, where it interviewed "over 60 residents living near industrial swine operations permitted under the Swine Waste General Permit,"[6] but also its assessment of over two decades of scientific research and "reports published by or with federal agencies."[7] The EPA notes that the

> adverse impacts on nearby residents from the lagoon spray field method of treatment and disposal of waste from industrial swine operations are documented in numerous peer reviewed scientific studies, including more than thirty conducted in North Carolina.... [T]he reports provide consistent support for the occurrence of potential health hazards (e.g., eye, nose, and throat irritation; headaches; respiratory effects including asthma exacerbation; waterborne disease) at industrial swine operations and in their waste. Even while there is significant uncertainty regarding the levels of exposure in nearby communities to the identified contaminants and the risk of health effects attributable to those exposures, the risk for specific health effects in communities near industrial swine operations is a concern.
>
> North Carolina's 1994 Swine Odors Task Force stated "It is not surprising to learn that living near a swine operation can affect mental health" when discussing a Duke University study of "the moods of people exposed to odors from commercial swine operations in North Carolina. Forty-four neighbors of hog operations ... had less vigor and were significantly more tense, depressed, angry, fatigued, and confused.[8]

The letter then describes the EPA's consideration of the disparate impact analysis conducted by Drs. Steve Wing and Jill Johnston.[9] EPA warned DEQ of its "deep concern about the possibility that African Americans, Latinos, and Native Americans have been subjected to discrimination as the result of NC DEQ's operation of the Swine Waste

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[5] *Letter from EPA to DEQ in Case 11R-14-R4*, January 12, 2017 (attached hereto as Exhibit 6).
[6] *Id.* at 3.
[7] *Id.* at 6.
[8] *Id.* at 6.
[9] *See* Steve Wing and Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians* (rev. Oct. 19, 2015), submitted to EPA in Case-11R-14-R4 (attached hereto as Exhibit 7).

General Permit program" in violation of Title VI and EPA's ensuing regulations.[10] EPA also expressed "grave concern" that our clients' members and staff had suffered intimidation from pork industry representatives in their efforts to get DEQ to address the harmful impacts from swine operations.[11]

EPA's letter concludes with seven sets of "preliminary" recommended actions. While all seven are critical, these three most directly relate to our requests:

- Conduct an assessment of current Swine Waste General Permit to determine what changes to the Permit should be made in order to substantially mitigate adverse impacts to nearby residents. Determine which changes are currently within NC DEQ's authority to make and develop a timetable for adopting them. For Permit changes necessary to substantially mitigate the adverse impacts that NC DEQ cannot adopt, determine the source of the impediment to their adoption.

- Conduct an assessment of current regulations applicable to facilities operating under the Swine Waste General Permit to determine what if any changes to the regulations would be required to substantially mitigate adverse impacts to nearby residents. Determine which changes are currently within NC DEQ's authority to make and develop a timetable to adopt them. For regulatory changes necessary to substantially mitigate the adverse impacts that NC DEQ cannot adopt, determine the source of the impediment to their adoption.

- Conduct an assessment of current mitigation technologies that would satisfy NC DEQ' s performance criteria for new or expanding industrial swine operations and what if any impediments exist to adopting those technologies.

It has been more than two years since EPA made the above recommendations. It has been more than a decade since such "mitigation technologies" were identified. It is unacceptable for DEQ to refuse to comply with its obligations under Title VI for another five years.

2. **The Title VI Settlement**

As part of the May 2018 settlement reached with DEQ, the agency agreed to a number of provisions to comply with its obligations under Title VI, including establishment of an effective Title VI program, improvements in environmental and public health protections in the Draft Permit, inclusion of impacted community members in the stakeholder input process, and development of an Environmental Justice geographical information tool ("EJ tool") to "allow DEQ programs to conduct environmental justice analyses" as a part of

[10] *See* Exh. 6 at 1.
[11] *Id.* at 8.

permitting.[12] At DEQ's February 4, 2019 public meeting regarding the EJ tool, the Department's Title VI Administrator, Sarah Rice, indicated that the tool will not be ready for implementation as part of the 2019 General Permit renewal.

It is necessary to address some apparent misunderstanding about the General Permit's discriminatory nature. During the Draft Permit public hearing on February 19, 2019, N.C. House Representative Jimmy Dixon incorrectly stated that claims of racially discriminatory impacts from "our animal facilities" are "misrepresented facts" because "if you take our animal facilities and you measure one half mile from them, it is 62% white." We set aside for the moment questions regarding what Rep. Dixon meant by "our animal facilities." We also question the relevance of Rep. Dixon's selection of a half mile as the appropriate metric for consideration, given that:

1) Significantly fewer people (less than one tenth as many) live within a half mile of a permitted swine operation than live within three miles. People living between a half mile and three miles away from any one CAFO are impacted by a higher concentration of CAFOs (sometimes dozens at once) than those living within a half mile of their closest CAFO;[13]

2) Decades of scientific research demonstrates air (including odor) and water pollution and concomitant adverse health effects experienced by residents within a radius greater than one-half mile of swine CAFOs in eastern North Carolina;[14]

[12] *See* Exh. 5. The agreement also contains other steps that DEQ must take, including a proposed implementation rule for the Violation Points System (see below at 26)(Exh. 5 at 5) and terms regarding acceptance of third-party data (*See*, *Letter from Sheila Holman to Complainants*, May 3, 2018, attached hereto as Exhibit 8).
[13] *See* Fliss et. al, *Comments on Swine General Permit (AWG 100000),* March 4, 2019.
[14] *See e.g.*, Melva Okun, Envtl. Res. Program, UNC School of Public Health, *Human Health Issues Associated with the Hog Industry* (1999); Todd Cole and Steve Wing, *Concentrated Swine Feeding Operations and Public Health: A Review of Occupational and Community Health Effects*, 108 Envtl. Health Persp. 685-699 (2000); Steve Wing, Rachel Avery Horton, Stephen W. Marshall, Kendall Thu, Mansoureh Tajik, Leah Schinasi and Susan S. Schiffman, *Air Pollution and Odor in Communities Near Industrial Swine Operations*, 116 Envtl. Health Persp. 1362 (2008); Wendee Nicole, *CAFOs and Environmental Justice: The Case of North Carolina*, 121 Envtl. Health Persp. A182, A186 (2013); Steve Wing, Rachel Avery Horton and Kathryn M. Rose, *Air Pollution from Industrial Swine Operations and Blood Pressure of Neighboring Residents*, 121 Envtl. Health Persp. 92 (2013); Steve Wing and Jill Johnston, *Industrial Hog Operations in North Carolina Disproportionately Impact African-Americans, Hispanics and American Indians* (rev. Oct. 19, 2015) and published studies referenced therein (Exh. 7); Virginia T. Guidry, Alan C. Kinlaw, Jill Johnston, Devon Hall and Steve Wing, *Hydrogen Sulfide Concentrations at Three Middle Schools Near Industrial Livestock Facilities*, 27 J. Expo. Sci. Environ. Epidemiol. 174 (2017); Julia Kravchenko et al, *Mortality and Health Outcomes in North Carolina Communities Located in Close Proximity to Hog Concentrated Animal Feeding Operations*, 79 N.C. Med. J. 278 (2018).

3) Complaints of horrible odor, flies, buzzards and other burdens from neighboring CAFOs have been made by the REACH members who live more than a half mile from multiple CAFOs (both swine and poultry); and

4) All four nuisance cases tried so far against Smithfield (which resulted in multi-million- dollar jury verdicts) involved residents living more than a half mile from swine operations covered under DEQ's General Permit.

It is, however, critical to address a more fundamental misrepresentation underlying Rep. Dixon's statement. Latino, Native American and African Americans constitute a minority of North Carolina's population as compared to whites and as explained in the methodology section of Drs. Wing and Johnston's analysis (see Exhibit 7), "a larger *proportion* of non-Hispanic Whites in North Carolina live in remote rural areas than do Blacks."[15] However, the "percentages of [people of color, defined as everyone other than non-Hispanic whites], Blacks, Hispanics, and American Indians living within 3 miles of [an industrial hog operation] are 1.38, 1.40, 1.26 and 2.39 times higher than the percentage of non-Hispanic Whites, respectively."[16] These disparities are "highly statistically significant,"[17] which means they are not by chance or random.

So while it is correct that most people living near swine CAFOs in North Carolina are white—because whites are still the racial majority in the state and a larger proportion of that white population lives in rural areas—it is incorrect to conclude that there is no racially disparate impact from these facilities. Such a statement ignores the analysis and methodology required to determine disparate impact,[18] as well as the fact that although non-whites make up a minority of the total population, the communities living near permitted hog operations have disproportionately high percentages of black, Latino and Native American residents. Furthermore, the fact that there are also white people living in those communities does not contradict or rebut the statistical evidence presented by our clients to DEQ and acknowledged by the EPA that these operations disproportionately burden communities of color. Indeed, white residents living in proximity to hog operations would also benefit from cleaner air and water if DEQ required Permittees located near communities of concern to employ stronger protections and superior waste disposal technologies, but the agency's legal obligation under Title VI is to mitigate the demonstrated disparate impact on North Carolinians of color.

Given these facts and DEQ's obligation to comply with Title VI, the EJ tool must be designed, developed and deployed not only to identify whether a permit applicant's barns, lagoons and sprayfields are in close proximity to vulnerable populations (as measured by those neighbors' demographic and health indicators) and to a cluster of

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[15] Wing and Johnston, *supra* n.9, Exh. 7 at 1 (emphasis added).
[16] *Id*. at 6.
[17] *Id.*
[18] *Id.*

other facilities and other sources of pollution, but also to assess potential cumulative impacts from other polluting operations within the three-mile radius.[19]

The EPA has defined "cumulative impacts" as the "[t]otal exposure to multiple environmental *stressors* . . . , including exposures originating from multiple *sources*, and traveling via multiple pathways over a period of time."[20] Incorporated here by reference and attached as Exhibit 10 is our September 2, 2016 letter brief to the EPA on cumulative impacts from hog CAFOs covered by DEQ's General Permit, which addresses with supporting research three stressors that contribute to the General Permit's cumulative adverse impact:

> First, EPA must account for the cumulative impacts of more than 2,000 IHOs [industrial hog operations] in a relatively small geographic area. Eastern North Carolina is more densely populated with hogs than anywhere else in the United States. Second, EPA must analyze the environmental contamination and associated risks to human health stemming from the cumulative impacts of IHOs and the many industrial poultry operations clustered in this same region. Finally, any analysis of the adverse impacts of IHOs in eastern North Carolina would be grossly inadequate without consideration of the affected community's pre-existing vulnerability, which results from racial discrimination, poverty, and other factors.[21]

Consideration of cumulative impacts is particularly critical given the co-location of the swine and poultry industries in the state. As of 2018, there were 516 million birds confined in industrial operations in North Carolina, up from 147 million in 1997, according to a recent report by the Environmental Working Group and Waterkeeper Alliance.[22] The report found that industrial hog and poultry farms in the state are highly concentrated together, with 93 percent of poultry operations located within just three miles of 20 or more other poultry and swine operations.[23] Pollution from both swine and poultry

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[19] *See also* G.S. 143-215.1(b)(2) ("The Commission shall also act on all permits so as to prevent violation of water quality standards due to the cumulative effects of permit decisions. Cumulative effects are impacts attributable to the collective effects of a number of projects and include the effects of additional projects similar to the requested permit in areas available for development in the vicinity. All permit decisions shall require that the practicable waste treatment and disposal alternative with the least adverse impact on the environment be utilized.").

[20] *Draft Title VI Guidance for EPA Assistance Recipients Administering Environmental Permitting Programs* and *Draft Revised Guidance for Investigating Title VI Administrative Complaints Challenging Permits*, 65 Fed. Reg. 39650, 39684 (June 27, 2000).

[21] *Letter to Lilian Dorka, Interim Director, EPA Office of Civil Rights in Case 11R-14-R4*, September 2, 2016, at 9 (attached hereto as Exhibit 9 with names of declarants redacted).

[22] Soren Rundquist & Don Carr, *Under the Radar: New Data Reveals NC Regulators Ignored Decade-Long Explosion of Poultry CAFOs* (2019), https://waterkeeper.org/wp-content/uploads/ 2019/02/EWG_ Waterkeeper-NC-CAFO_Report_C05.pdf.

[23] *Id.*

facilities is now contaminating the same waterbodies,[24] which are already suffering from high nutrient loads. Poultry manure, which is often applied as fertilizer, contains many of the same nutrients as hog manure, including nitrogen and phosphorus. At this point, statewide, industrialized poultry operations cause even more nutrient pollution than hogs.[25] The presence of both poultry and hog facilities in the same areas raises the risks of harms to water quality. Therefore, the co-location of industrial hog and poultry operations demands that the permit require the assessment of cumulative impacts from both kinds of operations – to ensure that decisions take account of real-world conditions.

In sum, in order to comply with Title VI, DEQ must take significant steps in the 2019 General Permit to address the permit's discriminatory and cumulative impacts. The above requested changes on page 1 of the Draft Permit are two necessary steps.

As described in our December 21, 2018 stakeholder comments, important recent events make this the most critical time for major changes in the Swine General Permit, including Smithfield's announcement of its plans for methane recapture at many of its operations; the increased frequency and severity of storms and rainfall; and the growing body of scientific literature that demonstrates that industrial hog operations using the lagoon and sprayfield system threaten human health and the environment. We applaud the positive changes that DEQ has made in the Draft Permit, and offer the following concerning other specific conditions and terms:

Condition I.1: Definition of Discharge and Storm Standards

We support DEQ's effort to update the definition of the 25-year, 24-hour rainfall event from reliance on an outdated 1960's era bulletin to more recent NOAA standards. However, the Draft Permit undercuts DEQ's recognition that the science and the facts have evolved since the 1990s by allowing facilities to be "designed, constructed, operated, and maintained to contain all waste plus the runoff from a 25-year, 24-hour rainfall event ***at the time of construction for the location of the facility***" (emphasis added). As storms hit North Carolina more frequently and more severely, the state needs to ensure that it updates its interpretation of the standard to protect the environment and communities from harmful manure waste runoff. In addition, DEQ must follow the science and require all operations to adhere to the same updated and uniform storm standard rather than outdated definitions established "at the time of construction."[26]

[24] *Id.*
[25] *Id.*
[26] Given the effects of climate change and the increasing frequency of even 100-year storms, if lagoons were to be built today, they should meet the even stricter standards of 100-year/24-hour rainfall events. As indicated above, however, we are not requesting the significant investment that this standard might require. We call on DEQ to plan for the transition from lagoons and sprayfields and in the meantime, recognize that a standard developed for a 25-year/24-hour storm event is grossly inadequate and not sufficiently protective of the waters or people of the state.

Hurricanes, storms, and extreme precipitation events in North Carolina have increased in both severity and frequency due to climate change.[27] Researchers project that average winter precipitation in North Carolina will increase up to 25% in the next 50 years at the current rate of warming.[28] The number of "heaviest 1%" rainfall events along the Mid-Atlantic coast rose nearly 30% from 1958 to 2016.[29] Since 1999, North Carolina endured at least four hurricanes or tropical storms that qualify as 100-year storms: Floyd in 1999,[30] Irene in 2011,[31] Matthew in 2016,[32] and Florence in 2018.[33] During Hurricane Floyd, Wilmington experienced a 24-hour record of 15.06 inches of rain.[34] Even according to updated NOAA rainfall standards, this exceeds the 100-year, 24-hour rainfall frequency of 12.7 inches.[35] Hurricane Irene produced enough rain over the northeast portion of the state to exceed a 25-year rainfall event.[36] Hurricane Matthew, which hit North Carolina just 15 years after Floyd, resulted in record levels of precipitation. Rainfall in Fayetteville, in the region of the state most concentrated with swine CAFOs, exceeded the town's 1000-year, 24-hour rainfall event of 8.6 inches by 6 inches.[37] Most recently, Hurricane

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[27] U.S. Global Change Res. Program (USGCRP), 2 *Fourth National Climate Assessment: Impacts, Risks, and Adaptation* (2018); Gabriele Villarini & Gabriel Vecchi, *Projected Increases in North Atlantic Tropical Cyclone Intensity from CMIP5 Models*, 26 J. of Climate 3232, 3232-3240 (2013); Enrico Scoccimarro et al., *Intense Precipitation Events Associated with Landfalling Tropical Cyclones in Response to a Warmer Climate and Increased CO2*, 27 J. of Climate 4642, 4647-4654 (2014); Donald Wuebbles et al., *CMIP5 Climate Model Analyses: Climate Extremes in the United States*, 95 Am. Meteorological Soc'y J. (2014); Brian A. Colle et al., *Historical Evaluation and Future Prediction of Eastern North American and Western Atlantic Extratropical Cyclones in the CMIP5 Models During the Cool Season*, 26 J. of Climate 6882, 6882-6903 (2013).
[28] USGCRP, *supra* n.27.
[29] *Id.*; Russ S. Schumacher & Richard H. Johnson, *Characteristics of U.S. Extreme Rain Events During 1999–2003,* 21 Weather Forecast 69, 69–85 (2006).
[30] NOAA Nat'l Weather Serv., *Event Overview, Hurricane Floyd Storm Summary,* https://www.weather.gov/mhx/Sep161999EventReview (last accessed Feb. 22, 2019).
[31] NOAA Nat'l Weather Serv., *Event Overview, Hurricane Irene August 26-27, 2011*, https://www.weather.gov/mhx/Aug272011EventReview (last accessed Feb. 22, 2019).
[32] NOAA Nat'l Weather Serv., *Hurricane Matthew, October 8-9, 2016 Summary*, https://www.weather.gov/mhx/MatthewSummary (last accessed Feb. 22, 2019).
[33] Gavin Off, *Rain Gauge Map: Charlotte Totals at 72 Locations. Some Areas see more than 10 Inches since Friday,* The Charlotte Observer (Sept. 15, 2018), https://www.charlotteobserver.com /news/local/article218458300.html.
[34] NOAA Nat'l Weather Serv., *Event Overview, Hurricane Floyd Storm Summary,* https://www.weather.gov/mhx/Sep161999EventReview (last accessed Feb. 22, 2019).
[35] NOAA Atlas 14, Volume 2, version 3.0, 2004 revised 2006, https://hdsc.nws.noaa.gov /hdsc/pfds/pfds_map_cont.html; *See also* https://www.nws.noaa.gov/oh/hdsc /PF_documents/Atlas14_Volume2.pdf.
[36] N.Y. Times, *Flooding, Power Failures, Rainfall and Damage From Hurricane Irene* (Aug. 31, 2011), https://archive.nytimes.com/www.nytimes.com/interactive/2011/08/27/us/preparations-for-hurricane-irene-and-reports-of-damage.html?ref=us.
[37] Elena Gooray*, Hurricane Matthew Brought 1,000-Year Record Rainstorms to North Carolina*, Pacific Standard (Oct. 12, 2016), https://psmag.com/news/hurricane-matthew-brought-1000-year-record-rainstorms-to-north-carolina#.gtjrmj8h7; *See also* NOAA National Weather Service, *AEP Storm Analysis Hurricane Matthew October 2016 (2016),* https://www.nws.noaa.gov/ohd/hdsc/aep_storm_analysis/AEP_HurricaneMatthew_October2016.pdf .

Florence dumped 8.84 inches of rain within 24-hours in Fayetteville, also exceeding the current storm definition.[38]

The heavy rainfall associated with extreme storms cause severe flooding. For example, multiple rivers and streams in North Carolina reached 100-year flood volumes during both Hurricane Floyd[39] and Hurricane Matthew.[40] During Hurricane Floyd, there were 14 distinct 500-year or greater floods in the eastern part of the state.[41] Sudden inundation of rainfall into uncovered swine manure lagoons caused the volume of waste to exceed the facilities' holding capacity, leading to structural failures and overflowing of waste matter onto flooded fields and into waterways. Hurricane Floyd flooded 45 swine lagoons, causing algal blooms and mass fish die-offs.[42] In 2016 with Hurricane Matthew, at least 14 hog lagoons flooded and two breached. After Hurricane Florence, North Carolina DEQ reported that 6 lagoons experienced structural damage and 33 lagoons overtopped, with an additional 10 lagoons operating at a level where overtopping was likely because the structure had no room to store additional liquid.[43] Overall, the agency reported hundreds of incidents representing environmental threats at animal operations due to flooding and inundation in the wake of the storm.[44]

With heavy rainfall, nutrients and disease-causing agents from sprayfields enter nearby streams through surface and subsurface runoff.[45] These pollutants, such as phosphorus and nitrogen, already have caused significant damage in streams and other water bodies

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[38] Gavin Off, *Florence Rain Gauge Map: Totals for More than 400 Locations Across North Carolina*, The Charlotte Observer (Sept. 16, 2018), https://www.charlotteobserver.com/news/state/north-carolina/article218495840.html

[39] Jerad D. Bales et al., *Two Months of Flooding in Eastern North Carolina, September-October 1999: Hydrologic, Water quality, and Geologic Effects of Hurricanes Dennis, Floyd, and Irene,* Water-Res. Investigation Rep. No. 00-4093, U.S. Dept. of the Interior, U.S. Geological Surv. (2000), https://pubs.usgs.gov/wri/wri004093/flooding.html.

[40] FEMA, *Hydrologic Analysis of Hurricane Matthew's Impact on Dam Safety in North Carolina and South Carolina (*Aug. 2018), https://www.fema.gov/media-library-data/1535042937481111942dab7f7f79e5f561f3e0bc0a2d9c/NCSCDamsHydrologicSummary_FINAL_8-14-18_dz.pdf.

[41] *Id.*

[42] Steve Wing et al., *The potential impact of flooding on confined animal feeding operations in eastern North Carolina*, 110 Envtl. Health Persp. 387, 387-391 (2002).

[43] North Carolina DEQ, *DEQ Dashboard (*Oct. 9, 2018), https://deq.nc.gov/news/deq-dashboard#animal-operations---swine-lagoon-facilities (these numbers understate the problem by failing to take into account the practice of lowering lagoons in advance of storms, which also raises the risk of significant runoff during storm event).

[44] North Carolina DEQ, *Hurricane Incident Tracking Application,* https://ncdenr.maps.arcgis.com/apps/webappviewer/index.html?id=c73b17df1fa8400998c69da505f36eb8.

[45] Robert Evans et al., *Subsurface Drainage Water Quality from Land Application of Swine Lagoon Effluent*, 27 Transactions Am. Soc'y Agric. & Biological Eng'rs 473 (1984); Phil Westerman et al., *Swine Manure and Lagoon Effluent Applied to a Temperate Forage Mixture: II. Rainfall Runoff and Soil Chemical Properties*, 16 J. Envtl. Quality 106 (1987).

in eastern North Carolina.[46] In addition, pathogens and bacteria from the runoff fecal matter contaminate drinking water sources and threaten human health. In the weeks following Hurricane Florence, nearly 30% of tested drinking water wells contained fecal coliform bacteria at levels above the EPA health-based standard.[47] Drinking water with elevated levels of bacterial contaminants increases the risk of infection and illness.

Extreme rainfall in the future will cause even more runoff from swine waste management systems, polluting the environment and harming public health. DEQ's interpretation of the requirement that lagoons must meet a 25-year/24-hour storm event must keep up with the science and ensure that animal operations are prepared for the increasing severity of rainfall events.

It is inappropriate to ask only that facilities comply with historic standards from ten or twenty years ago when a lagoon was built, because these facilities were not designed to sustain the magnitude of extreme storms today. Prior to the adoption of the 25-year, 24-hour rainfall event developed by the National Oceanic and Atmospheric Administration in 2004, the General Permit used a definition from a 1960's technical bulletin.[48] This standard would have applied to any operation constructed prior to the adoption of the NOAA 2004 definition. A comparison of the precipitation quantities at various storm frequencies demonstrates the increasing probability of experiencing extreme rainfall events. The 1960's rainfall frequency atlas defines a 50-year, 24-hour rainfall event in Fayetteville, NC as just under 7 inches of rain.[49] In the updated atlas, a rainfall event producing between 6.16 to 7.19 inches is predicted to occur every 25 years rather than every 50 years.[50] A 25-year storm standard that would have applied to past permitted facilities now reflects the maximum amount of rain from a 10-year storm.

Operating and maintaining facilities based on historic standards and climate conditions is no longer adequate to limit waste runoff. In Fiscal Year 2017-2018, the large majority of permit violations discovered during Division of Water Resources (DWR) and Division of Soil and Water Conservation (DSWC) inspections related to manure management

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[46] Kenneth Stone et al., *Water Quality Status of a USDA Water Quality Demonstration Project in the Eastern Coastal Plain*, 50 J. Soil & Water Conservation 567 (1995); James W. Gilliam et al., *Contamination of Surficial Aquifers with Nitrogen Applied to Agricultural Land,* Water Resources Res. Inst., Univ. of N.C., Rep. No. 306 (1996).
[47] Lisa Sorg, *Monday numbers- A Closer Look at Private Drinking Water Wells post-Florence,* NC Policy Watch (Jan. 14, 2019), http://www.ncpolicywatch.com/2019/01/14/monday-numbers-a-closer-look-at-private-drinking-water-wells-post-florence/.
[48] David M. Hershfield, *Technical Paper No. 40, Rainfall Frequency Atlas of the United States for Durations from 30 Minutes to 24 Hours and Return Periods from 1 to 100 Years*, U.S. Dept. of Commerce & Weather Bureau (1961), https://www.nws.noaa.gov/oh/hdsc/PF_documents/TechnicalPaper_No40.pdf.
[49] *Id.*
[50] Precipitation Frequency Estimate in Fayetteville, NC, NOAA Atlas, https://hdsc.nws.noaa.gov/hdsc/pfds/pfds_map_cont.html (search by location at latitude: 35.057 and longitude: -78.876).

issues.[51] 20% of all permit violations at swine operations (24 out of 121) were due to discharges from animal waste management systems; 83% of these discharges reached state surface waters.[52] Another 29% of violations documented "inadequate freeboard," in which lagoon structures held waste volumes above the maximum operating level.[53]

Condition I.4: Amendments

We support the new requirement that amendments be included as part of the CAWMP and be submitted to the Division's Central Office within 30 days. Under the 2014 permit, Permittees were not required to submit an amendment to the Division Office "unless specifically requested." This change will lead to greater transparency and accountability by allowing public access to records of amendments, which will no longer be kept only at the facility. This additional transparency is crucial, allowing community members who are affected by pollution to access information on changes to waste management activities, including land application, and to learn of any changes in a reasonable time. This will reduce the likelihood that amendments will lead to adverse effects on the environment.[54]

Condition I.12: Setbacks

We encourage DEQ to adopt stronger setbacks for land application of waste. The current requirements contained in the Draft Permit continue to be inadequate for protecting human health and the environment. DEQ should act on its authority to implement more protective standards for land application than those specified by statute.[55]

Abundant scientific research has demonstrated that land application of waste impairs water quality, supporting the need for strong setbacks.[56] Overapplication and application to saturated soils can cause contaminants to enter waters through runoff. [57] Significantly, even when waste is applied at recommended application rates, contamination can still

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[51] *Dept. of Envtl. Quality, Annual Repost on Animal Waste Management Permitting, Inspection and Compliance Activities* July 1, 2017 through June 30, 2018 (2019), https://www.ncleg.gov/documentsites/committees/BCCI-6658/Reports/FY%202018-19/DEQ_DWR_Animal_Waste_Management_Annual_Report-2019-01-28.pdf (last accessed Feb. 24, 2019).
[52] *Id.* at 5, *see* Table 6.
[53] *Id.* (freeboard is the required depth between the upper edge of the lagoon, usually an elevated diversion terrace constructed around the perimeter of the lagoon, and maximum allowed level of liquid manure storage).
[54] Comments submitted in the stakeholder process, Exh. 1 at 7 (removal of amendments) were also intended to promote transparency. Consistent with that intent, we support the language in the Draft Permit, which clarifies that amendments must be submitted to the Division's Central Office within 30 days.
[55] Statutory requirements set floors for setback distances. See 15A NCAC 02T .0108(b) and 15A NCAC 02T.1304(b)(5); *See also* Exh. 3, at 27-28.
[56] *See* studies cited *infra* n. 57-61.
[57] JoAnn M. Burkholder et al., *Impacts of Waste from Concentrated Animal Feeding Operations on Water Quality*, 115 Envtl. Health Persp. 308 (2007), http://dx.doi.org/10.1289/ehp.8839.

occur.[58] Research has also linked land application to the presence of antimicrobial residues in stream water,[59] and surface waters near sprayfields have been found to contain high concentrations of fecal indicator bacteria.[60] A study in North Carolina revealed that 22% of the nitrogen in waste applied to sprayfields was lost to offsite transport, suggesting that nitrogen could be contaminating nearby water resources.[61] As mentioned in previous comments, North Carolina needs particularly stringent requirements for setbacks due to its high concentration of industrial animal operations and its unique hydrological and geological features.[62] Porous soil and high water tables increase the risk that nutrients from waste application will contaminate water sources, including drinking water.

Despite its need for especially strong protections for water resources, North Carolina's requirements for setbacks are lower than in other states. In this Draft Permit, setbacks for wells remain only 100 feet, which is significantly lower than other states' requirements. For example, as mentioned in past comments, South Dakota requires setbacks of at least 250 feet from private wells,[63] South Carolina requires a 200-foot setback from any drinking well,[64] and Wisconsin requires a 1,000-foot setback from community wells.[65] In addition, Indiana recently required that CAFO waste management systems be located a minimum of 1,000 feet from public water supply wells.[66] Due to the vulnerability of drinking water sources in many areas of North Carolina, land application should only be allowed to occur a minimum of 500 feet from wells, rather than only 100 feet. North Carolina, with its particularly vulnerable water resources, should have even stricter standards than these other states.

While we welcome the inclusion of specific setbacks for land application from water bodies in this Draft Permit, the distances fall far short of what is necessary. The setback requirements prior to 1997 have been recognized to be inadequate; therefore, older facilities should not continue to be held to these weaker standards. The setbacks should

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[58] *Id.*

[59] UNC Chapel Hill, Dep't of City & Reg'l Planning, Econ. Dev. Workshop, *Identifying Opportunities and Impacts for New Uses of Hog Waste in Eastern North Carolina*, 12-13 (2013), www.ncgrowth.unc.edu/wp-content/uploads/2014/06/OpportunitiesAndImpactsOfHogWasteInEasternNC.pdf

[60] Christopher D. Heaney, et al. *Source Tracking Swine Fecal Waste in Surface Water Proximal to Swine Concentrated Animal Feeding Operations*, 511 Sci. Total Env't. 1 (2015) https://www.ncbi.nlm.nih.gov/pmc/articles/PMC4514616/

[61] Jeffrey DeBerardinis, *Nitrogen Mass Balance for Spray Fields Fertilized with Liquid Swine Waste*, 67 (2006), https://cdr.lib.unc.edu/indexablecontent/uuid:612a684e-41c1-464b-bc98-3b6b1ad16247

[62] *See* Exh. 1, at pp. 8-9; *see also* Exh. 2, at 39-43.

[63] South Dakota Department of Environment and Natural Resources, General Water Pollution Control Permit for Concentrated Animal Feeding Operations, SDG-100000 (April 15, 2017) at 1.1.26.

[64] S.C. Reg. 61-43 Part 100 https://www.clemson.edu/extension/camm/regulations/r61-43.pdf; https://www.scdhec.gov/environment/water-quality/water-quality-agriculture-permits-and-compliance/agricultural-permits-3.

[65] Wisconsin Administrative Code Chapter N243.15 https://docs.legis.wisconsin.gov/code/admin_code/nr/200/243/II/11

[66] 327 Indiana Administrative Code 19-12-3.

equal the distance necessary for preventing contamination of waterbodies, which is DEQ's obligation. Older facilities should not be given more leeway at the expense of environmental quality and impacts on nearby community residents. In addition, having one standard for land application for all facilities will make compliance and enforcement more straightforward.

Even a 75-foot setback from surface waters for all facilities would be inadequate. The EPA NPDES manual requires that setbacks for manure application for large CAFOs must be "a minimum" of 100 feet "from surface waters and conduits to surface waters."[67] Some states have recognized the need for even greater protections. For example, Indiana requires setback distances of 300 feet from surface waters, as of 2018.[68]

Condition II.4: Nutrient Management Plan

We strongly support the new language clarifying that agronomic rates for land application must account for "all nutrient sources," rather than only nutrients from the swine waste being applied. This change will help ensure that land application does not occur at greater than agronomic rates, since analyses prior to land application should account for all potential nutrient sources, including commercial fertilizer, effluent, sludge, and waste from other animals.

Preventing excessive application of nutrients is critical since the watersheds impacted by land application already suffer from high nutrient loads. For example, nutrient enrichment has been highlighted as the primary threat to water quality in both the Tar Pamlico River Basin and the Neuse River Basin.[69] Excessive nutrient enrichment harms the environment by contributing to fish kills, algal blooms, and contamination of drinking water.[70] High nutrient loads in surface and groundwater indicate that land application has failed to occur at proper agronomic rates, leading to excess nutrients being leached into waterways.[71] Moreover, ensuring proper land application practices, and

[67] NPDES *Permit Writers' Manual for CAFOs*, (2012), https://www3.epa.gov/npdes/pubs/cafo_permitmanual_entire.pdf.
[68] 327 IAC 19-12-3.
[69] Div. of Water Res., *Neuse River Basinwide Water Quality Plan* § 17.1.4 (2009), https://deq.nc.gov/about/divisions/water-resources/planning/basin-planning/water-resource-plans/neuse-2009. ("Excessive nutrient loading is ultimately the primary stressor in the Neuse River basin."); Div. of Water Res., *Tar-Pamlico River Basinwide Water Quality Management Plan* § 7.3. (2010), https://files.nc.gov/ncdeq/Water%20Quality/Planning/BPU/BPU/Tar_Pamlico/Tar%20Pam%20Plans/2010%20Plan/TAR_SummaryFinal.pdf. ("Tar Pamlico River Basin-Nutrient enrichment of the water bodies within this basin continues to be the main water quality issue."); *See also* 15A NCAC 2B.0255, .0256 (agricultural nutrient control goal and strategy for the Tar-Pamlico basin); 15A NCAC 2B.0236, .0237 (agricultural nutrient control goal and strategy for the Neuse basin).
[70] Michael Mallin, *Impacts of Industrial Animal Production on Rivers and Estuaries*, 88 Am. Scientist 26 (Jan.–Feb. 2000).
[71] The nutrient load in waterways is also affected by the deposition of volatilized ammonia. Calculations of agronomic rates will be grossly inadequate until they take into account this significant nutrient source. It is vital that DEQ require facilities to do so in the next permit.

thus preventing water contamination from leached nutrients before it occurs, is especially critical due to the lack of comprehensive water monitoring requirements for CAFOs.

This proposed change is especially important given the rapid rise in the number of poultry operations in the same areas where hog operations are located, discussed above.[72] Under these conditions, application rates must take into account potential impacts of the co-location of poultry and swine facilities. The revision to Condition II.4 is a welcome move in that direction, since it will require that all nutrient sources be accounted for when determining agronomic rates for land application.

Condition II.7: Operator in Charge ("OIC")

We support the clarification regarding the OIC's responsibilities, which are reasonable measures to ensure that all land application is properly inspected, monitored, documented, and supervised in a timely manner. The additional requirement that if neither the OIC nor designated back-up is present during land-application of waste then the OIC or designee must inspect the application site within 24 hours of the application, should be amended to require not just inspection but completion of the requisite inspection records within that time period. Inclusion of OIC oversight in the permit, and the removal of the overbroad affirmative defense "loophole" is necessary to reduce the risk of discharge and other permit violations and will improve accountability and transparency.

Condition II.10: Disposal of Mortalities

We applaud DEQ's decision to clarify that "burial is not recommended for disposal of dead animals." This provision of the Draft Permit is consistent with guidance published by the North Carolina Department of Agriculture and Consumer Services ("DACS"), which deems burial a "limited on site disposal option due to flooded conditions and often minimal depth to seasonal high water table."[73] However, we urge DEQ to impose meaningful safeguards to protect public health and the environment in situations in which burial occurs.

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[72] *See supra* n. 20-22.

[73] N.C. Depart. of Agric., *Mass Animal Mortality Management Plan for Catastrophic Natural Disasters,* 2 (2016), http://www.ncagr.gov/disaster/documents/massmortalityguicanceplan.pdf. In emergency conditions that require burial, DACS recommends that operators maintain at least 12 inches of soil between the bottom of the burial hole and the seasonal high water table; maintain at least 3 feet of soil covering any buried animal; maintain at least 300 feet from any existing stream, public body of water, or public water supply well, and at least 100 feet from any other type of existing well; and locating a burial site "so as to minimize the effect of stormwater runoff." N.C. Depart. of Agric., *Animal Burial Guidelines During a Declared Emergency*, 2 (2011), http://www.ncagr.gov/oep/Storms/ANIMAL_BURIAL_GUIDELINES_April_2011.pdf.

Burial of dead animals can significantly impair water quality.[74] Indeed, burial pits often release more pollution than municipal and industrial sewage plants.[75] As animal carcasses decay, they release nutrients (nitrogen and phosphorus), chloride, disease-causing agents present in animal waste, ammonia, and nitrates into the environment—potentially rendering groundwater unsafe to drink.[76] This pollution continues until carcasses delay completely, a process that can take up to two years.[77]

Improper burial also reduces the quality of life and threatens the health of those living nearby. Areas surrounding industrial animal facilities already host high numbers of flies, birds, and rodents—all of which may feed on improperly buried carcasses.[78] Because animal carcasses can carry antimicrobial-resistant pathogens,[79] improper burial facilitates the movement of these pathogens into nearby communities and may lead to the development of antibiotic-resistant bacteria.[80]

We urge DEQ to impose groundwater-monitoring requirements near burial sites, as discussed below. Hurricane Florence recently caused the deaths of more than 5,500 hogs, nearly double the number killed during Hurricane Matthew.[81] Given the increasing frequency and intensity of extreme weather events due to climate change,[82] severe storms and mass mortalities are likely to continue in the future.

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[74] Qi Yuan et al., *Potential Water Quality Impacts Originating from Land Burial of Cattle Carcasses*, 456-457 Sci. Total Environ. 246 (2013).

[75] Anja Coors et al., *Removal of Estrogenic Activity from Municipal Waste Landfill Leachate Assessed with a Bioassay Based on Reporter Gene Expression*, 37 Envtl. Science. Tech. 3430, 3430-3434 (2003); C.E.J.R. Desbrow et al., *Identification of Estrogenic Chemicals in STW effluent. 1. Chemical Fractionation and* in vitro *Biological Screening*, 32 Envtl. Science. Tech. 1549, 1549-1558 (1998); Peter Splenger et al., *Substances with Estrogenic Activity in Effluents of Sewage Treatment Plants in Southwestern Germany. 2. Biological Analysis*, 20 Soc'y of Envtl. Toxicology Chemistry 2133, 2133-2141 (2001).

[76] Hilda Hatzell, *Effects of Waste-disposal Practices on Ground-water Quality at Five Poultry (Broiler) Farms in North-central Florida, 1992-93*. Water-Res. Investigation Rep. No. 95-4064, U.S. Dept. of the Interior, U.S. Geological Surv. (1995); Lee M. Myers et al. *Impact of Poultry Mortality Pits on Farm Groundwater Quality,* Ga. Inst. of Tech. (1999); William Ritter & A. E. M. Chirnside, *Impact of Dead Bird Disposal Pits on Ground-water Quality on the Delmarva Peninsula*, 53 Bioresour Tech 105, 105-111 (1995).

[77] Qi Yuan et al., *supra* n. 74.

[78] Gordon Nichols, *Fly Transmission of Campylobacter*, 11 Emerg. Infect. Dis. 361, 361-364 (2005); Dana Cole et al., *Free-living Canada Geese and Antimicrobial Resistance*, 11 Emerg. Infect. Dis. J.935, 935-938 (2005); D.J. Hanzler & H.M. Opitz, *The Role of Mice in the Epizootiology of* Salmonella Enteritidis *Infection on Chicken Layer Farms*, 36 Avian Diseases 625, 625-631 (1992).

[79] Ellen Silbergeld et al., *Industrial Food Animal Production, Antimicrobial Resistance, and Human Health*, 29 Annu. Rev. Public Health 151, 151-169 (2008).

[80] Julia R. Barrett, *Airborne Bacteria in CAFOs: Transfer of Resistance from Animals to Humans*, 113 Envtl. Health Persp. A116, A116-A117 (2005); Mary J. Gilchrist et al., *The Potential Role of Concentrated Animal Feeding Operations in Infectious Disease Epidemics and Antibiotic Resistance*, 115 Envtl. Health Persp. (2007).

[81] Tom Polansek, *Hog Deaths, Manure Flooding from Florence Seen Surpassing 2016 Hurricane*, Reuters (Sept. 18, 2018), https://www.reuters.com/article/us-storm-florence-hogs/hog-deaths-manure-flooding-from-florence-seen-surpassing-2016-hurricane-idUSKCN1LY36W

[82] Cynthia Rosenzweig et al., *Climate Change and Extreme Weather Events; Implications for Food Production, Plant Diseases, and Pests,* 2 Global Change & Human Health 89, 89-104 (2001).

Operators likely require more than 24 hours to develop an appropriate plan for managing mass mortalities. During Florence, high levels of precipitation and flooding continued long after initial hog deaths occurred. As a result, operators who planned to bury dead animals were likely unprepared to take prompt and responsible action, since burying dead animals in flooded soil directly contributes to release of pollutants into groundwater.

To protect public health and the environment, DEQ should work with the State Vet pursuant to its authority under N.C. Gen. Stat. 106-403 to prohibit mass burials during rainfall events and within floodplains. In addition, DEQ should increase the minimum distance of the burial site to the nearest groundwater well to a quarter mile (1,320 feet).[83] Information about mortalities should be added to the annual certification form. Additional challenges associated with mass burial should be addressed on a case-by-case basis.[84]

Condition II.19: Application of Waste in Wind Conditions

We applaud DEQ's decision to prohibit facilities from applying waste in wind conditions that cause or might reasonably be expected to cause waste to reach surface waters or wetlands or cross boundary lines or field boundaries. The Draft Permit provides an objective standard for land application and clarifies that operators are responsible for ensuring that waste does not in fact reach surface waters or wetlands or cross property lines or field boundaries. This provision is particularly important because certain wind conditions can carry mist containing disease-causing agents into vulnerable ecosystems, including surface waters and wetlands, and across property lines threatening human health.

Livestock waste contains harmful bacteria, viruses, and other pathogens that can infect humans and cause serious illness. In CAFO waste slurries, these disease-causing microbes can survive for weeks at a time.[85] Surface applications of bacteria-rich wastes to nearby fields are imprecise, leading to deposition of fecal matter and associated microbes and contaminants downstream. In the days following applications, harmful microbes have

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[83] The Washington State General Permit prohibits natural decomposition within 1,320 feet or less from any groundwater well, spring, sinkhole, or body of surface water, and in an area that has a seasonally high water table, seasonal floods, or within a hundred-year floodplain. We urge DEQ to implement similar restrictions on mass burial during a catastrophic mortality event to prevent environmental and health harms. *See* Wash. State Dept. of Ecology, *Response to Comments for the Concentrated Animal Feeding Operation National Pollutant Discharge Elimination System and State Waste Discharge General Permit and Concentrated Animal Feeding Operation State Waste Discharge General Permit*, 47 (Jan. 18, 2017), available at: https://ecology.wa.gov/DOE/files/5f/5f39ec9a-7687-442e-b8f1-1376ba2a4687.pdf.
[84] *Id.*
[85] Jane L. Mawdsley et al., *Pathogens in Livestock Waste, Their Potential for Movement Through Soil and Environmental Pollution*, 2 Applied Soil Ecology 1, 1–15 (1995).

been detected in groundwater and streams far from the site of application.[86] The area of potential exposure is much larger than the immediate area surrounding the site of application.

People living and working near application sites suffer exposure to these microbes and pollutants for extended periods, putting them at higher risk of health complications. Researchers have found swine fecal particles inside residential homes located near swine facilities in eastern North Carolina.[87] Contact with and consumption of food in contact with fecal particles containing bacteria and viruses can lead to infections. Wind-dispersed ammonia and the associated odor also pose significant threats to human health and wellbeing, causing symptoms that include headaches, nausea, hives, anxiety, irritated and dry eyes and throat, and depression.[88] Requiring facilities to limit and prevent spreading of airborne fecal matter and pollutants from spray mist is crucial to protect community health.

Condition II.23: Ceasing land application after storm warnings/watches
The Draft Permit expands land application up to 12 hours after storm watches and warnings, rather than four hours as in the stakeholder draft permit and the current permit. We oppose this change and urge the Department to revert to the original 4-hour window after which to cease land application following issuance of a National Weather Service (NWS) Hurricane Warning, Tropical Storm Warning, or a Flood Watch/Flash Flood Watch. It appears that the window was expanded from 4 to 12 hours at the swine industry's request, based only on industry's unsubstantiated claim that the NWS has begun issuing warnings and watches 36 hours in advance of a storm's arrival. This condition must ensure that waste has enough time to incorporate into the soil before precipitation in order to prevent waste from entering storm runoff from fields. Given the increasing intensity and severity of storms and rainfall, and the overwhelming impacts from flooding from Hurricanes Matthew and Florence in 2018, any reduction in the amount of time that sprayed waste can be absorbed by crops before a significant rain event is unconscionable and ignores scientific climate evidence described in our comments on Condition I.1. above. While storm watches and warnings might now be issued 36 rather than 24 hours in advance, this is not necessarily always the case, as the National Weather Service says warnings are "generally within 36 hours" and "are expected within 36 hours."[89] Requiring cessation of land application within 4 hours of the NWS

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[86] Stuart R. Crane et al., *Bacterial Pollution From Agricultural Sources: A Review*, 26 Transactions of the American Society of Agricultural Engineers 858, 858–866 (1983); Jane L. Mawdsley et al., *Pathogens in Livestock Waste, Their Potential for Movement Through Soil and Environmental Pollution*, 2 Applied Soil Ecology1, 1–15 (1995); *see* Mallin, *supra* n.69.
[87] Expert Report of Shane Rogers, Ex. 6 to Resp. in Opp'n to Mot. Partial Summ. J., *In re: NC Swine Farm Nuisance Litigation*, No. 5:15-cv-00013-BR 62-72 (E.D.N.C. May 5, 2017), ECF No. 330-6.
[88] U.S. EPA, *Managing Manure Nutrients at Concentrated Animal Feeding Operations* 64 (2004), https://www3.epa.gov/npdes/pubs/cafo_manure_guidance.pdf; Susan S. Schiffman, *Livestock Odors: Implications for Human Health and Well-Being*, 76 J. Animal Sci. 1,343, 1,343 (1998).
[89] Hurricane and Tropical Storm Watches, Warnings, Advisories and Outlooks, https://www.weather.gov/safety/hurricane-ww (last visited Feb 19, 2019).

watch or warning is a critical safeguard to prevent runoff and discharge. This change should be reversed.

Conditions II.18, II.24, III.2, and III.3: Requirements for Installation of Automated Technology

In North Carolina, antiquated, lower-tech systems for managing swine waste continually lead to permit violations that jeopardize public health and environmental quality. Automated technology that is affordable and widely available could help address these pervasive and severe violations.
We again encourage DEQ to plan now for the transition to more sustainable alternatives to the lagoon and sprayfield system. While the agency continues authorizing the use of the lagoon and sprayfield system, however, we support new language in Conditions II.18, III.2.c., and III.3 clarifying when the Division will use its authority to require automatic flow monitoring equipment, automated waste-level monitors and recorders, and automated rain gauges and recorders are necessary, though these provisions don't go far enough. The permit should require that all Permittees install these devices, instead of requiring them only on a case-by-case basis. Mandatory installation of these three kinds of devices would improve Permittees' ability to comply with several permit provisions. Automatic flow meters improve the quality of recordkeeping and calculations needed for proper maintenance of lagoons and sprayfields. Automated waste-level monitors and recorders enable Permittees to monitor and record lagoon waste levels with greater accuracy and consistency. The increasing frequency of severe weather events makes the use of these devices all the more important, since they enable data collection during storms, when it is most crucial. Similarly, automatic rain gauges are a simple, inexpensive way to improve the accuracy of records of precipitation events. These devices leave less room for human error and intentional non-compliance from self-monitoring and self-reporting.

At minimum, if installation of automatic flow monitoring equipment, automated waste-level monitors and recorders, and automated rain gauges and recorders continues to be on a case-by-case basis, the permit should state that circumstances beyond those explicitly listed in these conditions can lead DEQ to require these technologies. Language should be added to Conditions II.18, III.2, and III.3 to make clear that devices can be required as deemed necessary by the Director.

Furthermore, Condition II.24 should be modified to require that all Permittees install devices to stop irrigation during precipitation events. We are opposed to Permittees being given the choice between installing automated equipment and committing to the presence of an Operator in Charge (OIC) during land application of waste. These two options do not provide equivalent protections, since the written commitment to have an OIC present does not address the possibility for willful noncompliance. It is already a permit violation to land apply waste during precipitation events. Under the current permit, Permittees already have an obligation to avoid this practice and the OICs, their backups, or someone under their supervision is already required to inspect "the land application site as often as necessary to insure that the animal waste is land applied in accordance with the CAWMP."[90] This human element has proven repeatedly inadequate to prevent violation of the permit. Requiring installation of devices to automate compliance for all Permittees would help combat violations of the prohibition on land application during precipitation, which have been widely observed by Waterkeeper Alliance and community members. In addition, for those Permittees who elect to or are required to install these devices, they should be required to complete installation within six months, instead of 12 months. This shorter timeframe would ensure that devices to stop irrigation during precipitation events would be in place prior to the hurricane season of 2020. It is bad enough that the industry will enter the 2019 hurricane season without such technology in place.

In addition, all Permittees should be required to notify DWR in writing when devices covered by Conditions II.18, II.24, III.2, and III.3 have been installed. Finally, Condition II.18 on flow monitoring equipment should contain language similar that in Conditions II.24, III.2, and III.3 requiring that devices be properly maintained in line with the manufacturer's instructions and warranties.

Condition II.28: Crop Removal

Although we support the effort to clarify crop removal requirement, 24 months is far too long to adequately protect against decomposition and the return of nutrients into the soil. As discussed below, we support a more reasonable time-frame for the removal of hay bales (either in the form of actual removal or, absent a reasonably short time frame for actual removal, then for proper storage) on/near CAWMP fields.[91]

Timeline for Harvesting

Allowing cut crops to lie on the land results in decomposition. As crop residues decompose, nitrogen and phosphorus are released back into the soil through N mineralization and then nitrification, becoming available for leaching in the process.[92]

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[90] Permit AWG 100000, II.17.

[91] *See e.g.,* N.C.G.S. § 143-215.10B(c)(7) (2014) (requiring waste management plans regarding waste utilization that "assure a balance between nitrogen application rates and nitrogen crop requirements").

[92] Deanna Osmond & Jihoon Kang, *Nutrient Removal by Crops in North Carolina* (2008), https://content.ces.ncsu.edu/static/publication/js/pdf_js/web/viewer.html?slug=nutrient-removal-by-crops-in-north-carolina (last visited Feb 20, 2019) ("Nutrients in plants that are left in the field will partially resupply nutrient reserves in the soil as they decompose…Estimates of nutrient depletion, therefore, should take into account only the nutrients removed with the harvest portion of the plant."); Deanna Osmond, *Nitrogen Management and Water Quality* (2017), https://content.ces.ncsu.edu/nitrogen-management-and-water-quality#section_heading_10566 (last visited Feb 20, 2019) (depending on the crop, N may temporarily be immobilized by microbial activity for a period, but eventually N mineralization frees up N from microbial biomass. Mineralized N becomes available to plants for uptake or to leaching through nitrification or, particularly in areas with high-water-table soils, enters the atmosphere as gaseous nitrous

Crops and soil conditions found in eastern North Carolina where the highest concentration of these operation are located lend themselves to increased N mineralization of decomposing crop residues.[93] In the 24 months that the current provision language allows crop residues to remain after cutting, significant amounts of nitrogen and phosphorus once stored in crops will leave the decomposing plant residues and return to the soil in a leachable form.[94] If the objective of applying waste to crops is to take up nutrients from the soil to prevent leaching and to remove these nutrients from the system, this provision should be amended to clarify that the harvest of cut residues must be completed within 1 to 2 weeks.[95]

A more appropriate time frame in which to harvest cut crops would be 1-2 weeks, and not more than 6 months.[96] The shorter time period would also improve hay and silage

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oxide (a greenhouse gas) through denitrification. P in crop residues is mostly in forms already bioavailable to plants and will likely join the soil solution or become adsorbed to soil particles upon decomposition from crop residues; P in the soil solution or adsorbed to soil particles is available for leaching or loss to water through erosion).

[93] Baoqing Chen et al., *Soil Nitrogen Dynamics and Crop Residues. A review*, 34 Agron. Sustain. Dev. 429–442 (2014) (N mineralization, and therefore the risk of leaching, is impacted by crop composition and soil type. Even during decomposition of crops like hay with high C:N ratios, microbes might at first immobilize N, but decomposition process eventually lead to a net N mobilization for these crop residues); T. M. Egelkraut, D. E. Kissel & M. L. Cabrera, *Effect of Soil Texture on Nitrogen Mineralized from Cotton Residues and Compost*, 29 J. of Envtl Quality; Madison 1518 (2000) (sandier soils like those in eastern North Carolina mineralize more N than soils comprised of higher percentages of silt and clay, possibly because sandy soils have less aggregate protection of decomposing organic matter, freeing up more of the crop residues to decomposition and, consequently, N mineralization and leaching).

[94] I. K. Thomsen et al., *Net Mineralization of Soil N and 15N-ryegrass Residues in Differently Textured Soils of Similar Mineralogical Composition*, 33 Soil Biology and Biochemistry 277–285 (2001) (N mineralization of decomposing crop residues can begin on day one of cutting and contact with the soil and continue, though at a decreasing rate, until the residues are completely decomposed. Ryegrass decomposition studies show continued N mineralization during the entire study period of seven months, at which time, as much as 36% of N from the ryegrass residues was recovered in the soil as mineral N); H. Shindo & T. Nishio, *Immobilization and Remineralization of N Following Addition of Wheat Straw into Soil: Determination of Gross N Transformation Rates by 15N-ammonium Isotope Dilution Technique*, 3 Soil Biology and Biochemistry 425–432 (2005) (wheat straw decomposition can transition from immobilization to mineralization in a matter of weeks, if not sooner. The N remineralisation rates of wheat straw have been documented at 0.71, 0.55 and 0.29 mg N kg–1 day–1 after 7, 28 and 54 days, respectively—indicating a faster rate of return occurs in the first week).

[95] A. P. Schaffers, M. C. Vesseur & K. V. Sykora, *Effects of Delayed Hay Removal on the Nutrient Balance of Roadside Plant Communities*, 35 J. of Applied Ecology 349–364 (1998) (studies examining the loss of nutrients from cut hay left at the soil surface demonstrate that a substantial proportion of N and P is lost from decomposing plant matter in just a few weeks; after 6 weeks, more P is lost than N); Mahbubeh Zarabi & Mohsen Jalai, *Rate of Nitrate and Ammonium Release From Organic Residues*, 20 Compost Science & Utilization; Abingdon 222–229 (2012) (studies of other types of crop residues similarly found a rapid initial nutrient loss in the first one to five weeks of decomposition, followed by a continued but slower rate of nutrient loss).

[96] Chen et al., *supra* n. 93 (of crop residues researched, all crop residues leading to immediate N mineralization or net N mineralization were studied in less than six months); G. Maltais-landry & E. Frossard, *Similar Phosphorus Transfer from Cover Crop Residues and Water-soluble Mineral Fertilizer to Soils and a Subsequent Crop*, 393 Plant and Soil 193–205 (2015) (the vast literature investigating the fate of N and

quality. Most growers harvest and remove hay and other feed crops after cutting within a week, if not much sooner.[97] Furthermore, research from USDA Extension programs advocates for "Haylage in a Day" practices that consolidate cutting, drying, and removal steps within a single day to allow for enough drying out time to avoid rot while maximizing the amount of total digestible nutrients in the hay or silage.[98]

Timeline for Removal after Harvesting
Once crops on land application sites have been harvested, they should be removed from contact with the land and, if kept onsite, stored in such a way that minimizes the chance of decomposition and return of nutrients to the soil in order to guarantee compliance with the CAWMP. At a minimum, this Condition should clarify that if harvested crops are not physically removed from the land application site then operators must remove harvested crops from contact with the ground when stored on the land application site. The same timeframes for decomposition of crop residues described above applies to harvested crops. Therefore, crops should be removed according to the definition described above within 1-2 weeks of harvest, and at the very least no later than 6 months after harvest. Again, this timeline is reasonable, as most growers now cut, harvest, and remove their hay all in a week, if not in a day.

Conditions for Proper Removal via Onsite Storage
The closer in contact soil particles and plant residues are, the faster decomposition occurs.[99] The optimal conditions for decomposition are high water content, small residue size with high surface area for soil contact, and high temperatures. Therefore, to minimize the chance of returning nutrients to the soil—and to improve the quality of stored hay—the permit provision should specify the conditions for storing hay as a form of "removal" after it has been cut and harvested from the land application site.

Hay exposed to moisture and/or sunlight will decompose faster. Contact between hay and soil is the most critical aspect of hay decomposition and spoilage and should be eliminated.[100] Bales stored outside, exposed to the elements, on the ground and without a cover can see a sharp increase in moisture content, as much as 120% for the outer 3 inches, which can speed up weathering and decomposition of the hay.[101] In areas of high precipitation or humidity, improper storage can lead to greater than 50% loss of dry

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P during decomposition processes predominantly span just 2-6 months, during which time decomposition processes occur most rapidly with significant initial nutrient losses).

[97] Schaffers, Vesseur, and Sykora, *supra* n. 95.

[98] Paul Craig, *Haylage in a Day Penn State Extension* (2016), https://extension.psu.edu/haylage-in-a-day (last visited Feb 20, 2019); Matthew Digman et al., *Best Practices to Hasten Field Drying of Grasses and Alfalfa* 8 (2011).

[99] S. J. Giacomini et al., *Simulating the Effects of N Availability, straw Particle Size and Location in Soil on C and N Mineralization*, 301 Plant and Soil; Dordrecht 289–301 (2007).

[100] Don Ball et al., *Minimizing Losses in Hay Storage and Feeding*, https://extension.msstate.edu/sites/default/files/topic-files/beef-publications/beef-publications-landing-page/minhaylosses.pdf (last visited Feb 24, 2019).

[101] *Id.*

matter, which is detrimental to forage quality and efforts to prevent leaching.[102] Over the course of just several months, storing bales outside unprotected in the eastern United States can lead to decomposition and extensive and costly loss of feeding value for at least the outer 5-6 inches of hay, with greater losses at the bottom of bales due to contact with the soil.[103]

To avoid speeding up decomposition, the permit should require that hay be stored either 1) offsite or 2) if onsite, on pallets, pads 4-8 inches deep of rocks 1-3 inches in diameter, concrete, or some other base to raise the hay off of bare ground; securely covered as a group or as individual bales with a woolen blanket, flannelette sheet, plastic sheeting at least 6 mil. thick, net wrap, tarp, or other cover that protects bales from moisture while ideally letting the hay breath; and in high density bales of at least 10 pounds per cubic foot if a round bale. Even bales individually wrapped in covers should not be stored directly on the ground.[104] Bales stored outside in rows should run up and down well-drained, gentle slopes to avoid trapping water, should be stored with ends of bales butted tightly together, and should have at least three feet between the rows.[105] These conditions are in line with best storage practices for preserving hay and silage quality and are widely recommended by extension agencies and industry entities to improve crop quality during storage.[106]

In addition to these criteria for any storage on site, the permit should set a limit for how long bales can be stored onsite. Even with the best practices described above, nutrient loss from bales stored for 12-18 months can be double that of bales stored for less than 9 months.[107] Therefore, this permit should limit onsite storage of hay bales to a maximum of one year.

Condition III.2, 3b: Criteria for Monitoring and Recording Waste Levels

We urge DEQ to make automated monitoring/recording of waste levels mandatory for all facilities, instead of requiring it on a "case by case" basis. Given the increase in extreme

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[102] *Id.*
[103] *Id.*
[104] *Id.*
[105] *Id.*
[106] William Edwards, *Hay Storage Options: How do They Stack Up?*, Farm Progress (2017), https://www.farmprogress.com/forage/hay-storage-options-how-do-they-stack (last visited Feb 24, 2019); Ball et al., *supra* n.100; Brian Holmes, *Dry Round Hay Bale Storage Costs* 3 (2004), https://fyi.extension.wisc.edu/forage/dry-round-hay-bale-storage-costs/; Raymond Huhnke, *Round Bale Hay Storage*, http://pods.dasnr.okstate.edu/docushare /dsweb/Get/Rendition-6342/BAE-1716web.pdf (last visited Feb 24, 2019); Bob Schultheis, *Hay Storage & Feeding Management* (2013), http://extension.missouri.edu/webster/documents/presentations/2013-03-28_RegionalHaySchool/2013-03-29_Hay_Storage_and_Feeding_Management-BobSchultheis-screen.pdf; N.C. Cooperative Extension, *Hay Storage and Feeding Losses* (2018), https://duplin.ces.ncsu.edu/2018/04/hay-storage-and-feeding-losses/ (last visited Feb 27, 2019).
[107] Huhnke, *supra* n.106.

weather events due to climate change, the accuracy and precision of waste levels are increasingly important and time-sensitive. Automatic monitors make information available when it matters most. They are also more accurate and can store data, which can simplify the Permittee's job, while providing an accurate record at relatively low cost.

We appreciate the Division's removal of the freeboard violation pre-condition from III.2(c) as a step in the right direction. We note what appears to be an inadvertent failure to delete this sentence from III.2(c): "The Director may determine that installation of automated waste level monitors is not required if the Permittee can demonstrate that preventative measures were taken to avoid the violations and that the violations resulted from conditions beyond the Permittee's control." Because of the elimination of the freeboard violation requirement, this sentence makes no sense here.

Condition III.8: Mortality Records

Increasing the regularity of recordkeeping from monthly to weekly is an important step towards more effective oversight and, when necessary, investigation relating to the mortality and disposal of animals. The particular odors, vermin, pestilence, and other adverse impacts from the collection and disposal of dead hogs continue to be a central element in complaints from residents and were raised in the Title VI complaint.[108] Additionally, this increased oversight of animal mortality at permitted facilities is consistent with DEQ's new provisions in Condition II.10 regarding disposal of dead animals, which incorporate specific record-keeping and reporting requirements, and help improve oversight and transparency.

Condition III.9(f): Waste Samples Following Discharge

This Draft inexplicably deletes the provision requiring facilities to obtain waste samples within 48 hours after first knowledge of a discharge from a lagoon to surface waters.[109] This provision needs to be restored, as it is fundamental to ensuring accountability and transparency to the public, as well as to securing accurate and timely data regarding the potential impact of any discharge, which becomes more difficult to assess if sampling is delayed. During the stakeholder process, industry representatives objected to requiring sampling within 48 instead of 72 hours on the grounds that labs are not open on weekends. We requested sampling happen within 24 hours of knowledge of discharge.

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[108] *See* Exh. 2, at 16, 23, 30.
[109] *See* Draft Swine Waste Management System General Permit for Stakeholder Process, III.9(f)(48 hour standard); Swine Waste Management Permit (2014), III.9(f) (72 hour standard).

Condition III.10-III.14: Groundwater Monitoring

These provisions reaffirm the Division's existing regulatory authority to require a Permittee to undertake any necessary monitoring and reporting to protect surface water, groundwater, and wetlands. By its express terms, this provision requires individualized assessment and provides an important "first-line" of defense from any actual or potential adverse off-site impacts of permitted operations. The Division must retain the full range and scope of potential monitoring and reporting requirements, since this provision is designed to identify possible impacts, and any necessary monitoring, testing, or mitigation, as early as possible.

Support for Groundwater Monitoring in the 100-year Floodplain

We strongly support the additional provisions in Condition III.10-III.11 that require on-site groundwater monitoring at facilities with structures in the 100-year floodplain. The high-water table, low organic matter content, and sandy nature of these soils create conditions in which lagoon pits and spray field manure applications can leach nitrogen, phosphorus, and bacteria into groundwater and surface waterways. Studies have repeatedly found elevated nutrient levels below spray fields and downstream from lagoons in North Carolina, not just in the 100-year floodplain.[110] North Carolina watersheds with CAFOs have significantly higher concentrations of ammonium, nitrate, and total N than those without CAFOs.[111]

Facilities operating in the 100-year floodplain are more vulnerable to the risks of groundwater contamination due to flooding and high water tables. The Legislature recognized this risk in 1995, when it passed N.C. Gen. Stat. § 106-802(a2), prohibiting construction of new animal waste management systems in the 100-year floodplain, and again in 2007 when it amended N.C. Gen. Stat. § 143-215.10I to establish superior technology performance criteria and the Lagoon Conversion Program which came with significant financial investment.[112] In the last two decades, the risks associated with operating a facility in the 100-year floodplain have only increased. In light of climate change and more intense and frequent severe weather, CAFO operations in the 100-year floodplain are even more vulnerable to flooding and the surrounding area will be more at risk of groundwater and surface water contamination.[113]

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[110] Michael A. Mallin et al., *Industrial Swine and Poultry Production Causes Chronic Nutrient and Fecal Microbial Stream Pollution*, 226 Water, Air and Soil Pollution; Dordrecht 1–13 (2015).
[111] Stephen Harden, *Surface-Water Quality in Agricultural Watersheds of the North Carolina Coastal Plain Associated with Concentrated Animal Feeding Operations* (2015).
[112] To date, nearly $19 million in funds allocated by the North Carolina General Assembly has been spent in the Lagoon Conversion Program. *See* David Williams, Deputy Director, Division of Soil & Water Conservation, *Overview: 100-Year Floodplain Swine Buyout*, December 1, 2016.
[113] *See also* Exh.2, at 40.

Conditions to Trigger Groundwater Monitoring Outside of the 100-year Floodplain

The groundwater monitoring proposals in this Condition are a necessary step toward improved monitoring and enforcement to support the health and well-being of North Carolinians. However, the permit should specify additional key conditions that will trigger groundwater monitoring of facilities outside the 100-year floodplain, and these triggers should target the CAFOs most at risk of polluting and the areas most at risk of being polluted. Such triggers should include the following:

1. *Operations that employ burial as a mortality management strategy* – We applaud that this permit disfavors burial of mortalities, discussed above regarding Condition II.10. If facilities are still using burial, they should monitor groundwater for relevant contaminants. Livestock burial sites produce leachate that can contaminate groundwater, especially in areas with elevated water tables and sandy soils.[114] Complete decomposition of animal carcasses can take two years, and even as long as ten years depending on environmental conditions.[115] During the period of decomposition, animal carcasses can release greenhouse gases and leachates containing high levels of chemical contaminants including high electrical conductivity, turbidity, biochemical oxygen demand, ammonium-nitrogen, total dissolved solids, and Cl^- and SO_4^{2-} concentrations.[116] Elevated chloride levels in particular are good indicators of burial-related groundwater contamination. Burial sites can also lead to groundwater contamination from pathogens and bacteria including fecal coliform, *C. perfringens*, and, especially, *E. coli*.[117] Additionally, bacteria like sulfate-reducing bacteria can also serve as indicators of groundwater contamination from livestock burial sites. [118] Lastly, antibiotics are also likely present in the leachate from decomposing carcasses.[119]

2. *Scale and environmental context of a facility* – Studies indicate that the concentration of facilities in a given area, the amount of area covered by wetlands, and the amount of land under a sprayfield system together correlate with measurable effects of CAFO waste manures on surface water quality. In the shallow aquifer system of the North Carolina Coastal Plain, groundwater discharge

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[114] Ron Fleming & Rachel Freedman, *Water Quality Impacts of Burying Livestock Mortalities* (2003), https://www.ridgetownc.com/research/documents/fleming_carcassburial.pdf.
[115] *Id.*
[116] *Id.*; Man Jae Kwon et al., *Impacts of Leachates from Livestock Carcass Burial and Manure Heap Sites on Groundwater Geochemistry and Microbial Community Structure*, 12 PLOS One e0182579 (2017); T. D. Glanville et al., *Soil Contamination Caused by Emergency Bio-Reduction of Catastrophic Livestock Mortalities*, 198 Water, Air and Soil Pollution; Dordrecht 285–295 (2009).
[117] Ha Kyung Joung et al., *Nationwide Surveillance for Pathogenic Microorganisms in Groundwater near Carcass Burials Constructed in South Korea in 2010*, 10 Int. J. Environ Res. Public Health; Basel 7126–43 (2013).
[118] Kwon et al., *supra* n.116.
[119] Geon-ha Kim & Sudipta Pramanik, *Biosecurity procedures for the environmental management of carcasses burial sites in Korea*, 38 Environmental Geochemistry and Health; Kew 1229–1240 (2016).

to receiving streams contributes about 50-60 percent of the annual stream flow in the region.[120] Therefore, not only do the findings of groundwater studies bear weight in considering the impacts of CAFOs on groundwater quality, surface water studies in the region hold relevance, as well, as proxy indicators of groundwater quality. Watersheds with lower percentages of wetlands and higher swine barn densities and/or higher total acres of land application area have the most significant measurable impacts on surface water quality.[121] Therefore, groundwater-monitoring requirements should be triggered in watersheds with 1) a high concentration of CAFOs or a high amount of land area sprayed with animal waste *and* 2) a low percentage of wetland area.[122] This requirement should be triggered by a facility's proximity to a community of concern, discussed above at 7-8.

3. *Lagoons with a bottom elevation less than a minimum of 2 feet above the seasonal high water table* –These facilities are not compliant with the Natural Resources Conservation Service Waste Storage Facility Pond Criteria (313-Practice Standard), which requires that all ponds have a bottom elevation that is a minimum of 2 feet above the seasonal high water table.[123] Such lagoons are more at risk of leaching of nutrients and bacteria with little barrier between the waste they contain and the shallow aquifer systems in the Coastal Plains.[124] Therefore, triggering monitoring for these facilities would ensure that their operations are not endangering the health of communities who depend on water downstream for drinking water.

4. *Facilities upstream from drinking water sources* – Given the abundance of scientific evidence that waterways in watersheds dense with CAFOs have measurable negative nutrient and bacterial characteristics resulting from CAFO operations, facilities operating in close proximity upstream from drinking water sources should have to monitor their impacts on water quality.[125]

5. *Non-compliance history* – As directed by NC Gen Stat § 143-215.6E (2014), the North Carolina Environmental Management Commission under DEQ has the authority and obligation to develop a Violation Point System applicable to permits for animal waste management systems for swine farms that monitors permit violations and considers the relative threat of each violation to groundwater and

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[120] Harden, *supra* n. 111.
[121] *Id.*
[122] *Id.* DEQ could readily operationalize these standards by adopting a groundwater monitoring trigger for facilities in watersheds with 1) less than 14% wetland cover and 2) greater than 3 barns/mile or 52 total acres to which swine waste is applied.
[123] *See* NRCS, NHCP, Conservation Practice Standard No. 313, *Waste Storage Facility* 3 (May 2016), https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs143_026465.pdf.
[124] R. Hermanson et al., *Nitrogen Use by Crops and the Fate of Nitrogen in the Soil and Vadose Zone – A Literature Search*, Pub. No. 00-10-015 at 131 (2000), https://fortress.wa.gov/ecy/publications/SummaryPages/0010015.html.
[125] *See supra*, n. 110-111.

surface water quality, public health, and environment.[126] Violations that harm one of these three factors "shall receive the most points and shall be considered significant violations."

In the May 2018 settlement, DEQ agreed to draft a proposed implementation rule for the Violation Points System by May 3, 2019.[127] Consistent with a Violation Points System, this permit should require facilities with more than one significant permit violation or a certain number of violation points in the permit timeframe to monitor their facilities for the duration of the permit according to the standards in this Draft Permit.[128] Facilities that have non-compliance issues and acquire a high number of violation points have an elevated risk of contaminating groundwater as the conditions of this Draft Permit and, eventually, the point system are in place to protect environmental quality. Violating these regulations, therefore, is accompanied by an increased risk of negative impacts on environmental and water quality. Monitoring groundwater quality provides an avenue for protecting human health near non-compliant facilities and ensuring a transition to robust compliance.

6. <u>*Documented impacts on groundwater quality*</u> – DEQ should require groundwater monitoring at facilities where there is documentation that operations have impaired groundwater. At minimum, this permit should retain language proposed in the stakeholder process requiring groundwater monitoring "when any of the following conditions exist, including but not limited to: a. evidence that groundwater impacts to public or private water wells are occurring off-site; b. evidence of migration of contaminated groundwater to off-site property or properties; c. evidence of surface water impacts via groundwater."[129]

   For the Violation Points System to yield its intended results, DEQ must detect and enforce non-compliance issues negatively impacting water, health, and environmental quality. Of the 177 violations found during FY 2017-18, only 20 resulted in the initiation of an enforcement action.[130] Unpermitted discharges from the systems and evidence of over application were two of the three most common violations, both of which can degrade groundwater quality. The Wilmington Regional Office staff only had the capacity to spend an average of 56 minutes on each inspection, which is insufficient given the necessary inspection components.

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[126] *See* 15A NCAC 02T.0120(a) ("The Division shall consider an applicant's compliance history in accordance with G.S. 143-215.1(b)(4)b.2 and with the requirements contained in this Rule for environmental permits and certifications issued pursuant to Article 21.").
[127] *See* Exh. 5 at 5.
[128] Groundwater monitoring should be tied to significant violations, *see* GS 143.215.6E(a)(1) ("violations that involve the greatest harm to … groundwater"), as well as the accumulation of lesser violations that heighten the risk to groundwater resources.
[129] Draft Swine Waste Management System General Permit for Stakeholder Process, III.10.
[130] Dept. of Envtl. Quality, *supra* n.51.

In light of DEQ's stretched capacity for enforcement and recent budget cuts,[131] in the event that a third party can demonstrate acceptable evidence of offsite impact on groundwater quality, groundwater monitoring should be triggered. The Division should always assess the credibility of evidence of groundwater contamination, but when credible evidence is presented, it should be considered regardless of who presents it. Additionally, absent comprehensive groundwater monitoring by the regulating agency, the evidence necessary to trigger groundwater monitoring will be impossible to obtain unless rigorous, credible third-party evidence collected via approved sampling protocols is accepted.

7. *Covered Lagoons/Digesters* –N.C. Gen. Stat. § 143-215.10I(b) prohibits issuance or modification of a permit "to authorize the construction, operation, or expansion of an animal waste management system that serves a swine farm that employs an anaerobic lagoon as the primary method of treatment and land application of waste by means of a sprayfield as the primary method of waste disposal." As should be made clear by attached Exhibits 11 and 12[132], this would apply to lagoon covers and digesters, which would have to comply with the following:

> The Commission may issue a permit for the construction, operation, or expansion of an animal waste management system that serves a swine farm under this Article only if the Commission determines that the animal waste management system will meet or exceed all of the following performance standards: (1) Eliminate the discharge of animal waste to surface water and groundwater through direct discharge, seepage, or runoff. (2) Substantially eliminate atmospheric emission of ammonia. (3) Substantially eliminate the emission of odor that is detectable beyond the boundaries of the parcel or tract of land on which the swine farm is located. (4) Substantially eliminate the release of disease-transmitting vectors and airborne pathogens. (5) Substantially eliminate nutrient and heavy metal contamination of soil and groundwater.

N.C. Gen. Stat. § 143-215.10I(b) (2007-523, s. 1(a).) We note that this statute has already been violated where DEQ has allowed Permitees to install lagoon covers and digesters that fail to comply with these statutory performance criteria. DEQ must require those operations to undertake

[131] Will Doran, *As NC Pollution Concerns Grow, so do Environmental Budget Cuts*, Raleigh News & Observer, https://www.newsobserver.com/news/politics-government/state-politics/article174769781.html (last visited Feb 27, 2019).

[132] *See Smithfield Swine Farm Biogas Strategy*, admitted on Feb. 22, 2019 as Defendant's Exhibit 707 in *McGowan et al v. Murphy-Brown LLC d/b/a/ Smithfield Hog Production,* (E.D.N.C. Southern Division, 7:14-cv-00182), attached hereto as Exhibit 10; and attached Exhibit 11, the December 2, 2018 testimony of Kraig Westerbeek, Vice President Environment and Support Operations, Murphy-Brown LLC from *Gillis et al. v. Murphy-Brown*, LLC, E.D.N.C. No. 7:14-cv-00185 (Exh. 9 referenced at 281).

groundwater monitoring and forbid any further covers or digesters under the General Permit.

Anaerobic digestion makes nutrients more bioavailable to plants, increasing the content of leachable nitrogen in effluent and leading to a greater risk of groundwater contamination.[133] After anaerobic digestion, less effluent needs to be applied to the land to achieve the same fertilization effects since more of the nutrients are usable for plant uptake. Bioavailable nutrients are not locked up in microbes or absorbed to soil, making them likely to leach if not taken up by plants. Plant uptake of nutrients is relatively inefficient.[134]

Anaerobic digestion increases the nitrogen load in effluent by inhibiting ammonia off-gassing from the lagoon. Therefore, if a lagoon and sprayfield system with a digester continues to spray the same amount of waste on a field as before installing a digester (which often already occurs in amounts exceeding the soil's and plants' capacities to hold onto the nutrients), more bioavailable nutrients will be introduced into the system, increasing the amount of nutrients that can be leached out of the system. Furthermore, any leaching directly from the lagoon into the soil or groundwater will contain more bioavailable nutrients that stimulate eutrophication and result in negative health consequences.

Likewise, since the digester system increases the concentration of ammonium and ammonia in the lagoon, these systems produce an increase in ammonia volatilization from CAFO waste that can increase the risk of groundwater contamination nearby.[135] Digestate slurry volatilizes—and therefore redeposits—more ammonia than regular swine slurry.[136] Scientists estimate that 80-90% of total ammonia emitted from livestock operations is redeposited within roughly 6 miles of the source, and 20% is redeposited within roughly a half a mile.[137] Ammonia can be dry deposited on soil and plant surfaces or dissolved in water particles and precipitated back to land and waterways. In both cases, this increased volatilization not only presents atmospheric emissions concerns, but also represents an additional source of nitrogen entering waterways adjacent to CAFO operations with digesters. For the reasons described above, facilities with digesters pose an elevated risk of surface and groundwater nutrient contamination.

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[133] Joe Harrison et al., *Transformation and Agronomic Use of Nutrients From Digester Effluent*, eXtension.org (2013), https://articles.extension.org/pages/67900/transformation-and-agronomic-use-of-nutrients-from-digester-effluent (last visited Feb 20, 2019).
[134] For example, in North Carolina, average nitrogen losses are roughly 50% of the total amount applied to the land due to leaching, volatilization, and denitrification. Osmond and Kang, *supra* n.130.
[135] Mallin et al., *supra* n.110; Robert Nkoa, *Agricultural benefits and environmental risks of soil fertilization with anaerobic digestates: a review*, 34 Agronomy for Sustainable Development 473–492 (2014).
[136] Nkoa, *supra* n.134.
[137] *Id.*

Therefore, installing a covered lagoon/anaerobic digester system should trigger groundwater-monitoring requirements.

Sampling Parameters for Groundwater Monitoring

We strongly support quarterly sampling of monitored facilities described in Attachment B referred to in Condition III.13, as seasonality in both environmental conditions and CAFO activities influences detected impacts, with samples in winter or early spring significantly underestimating impacts from swine CAFO lagoon and sprayfield systems.[138] Additionally, we support the proposed monitoring parameters specified in Attachment B. While the specified parameters will provide DEQ with the necessary information to monitor CAWMP outcomes and ensure compliance with the General Permit, these parameters alone limit DEQ's ability to link swine operations to water quality indicators.

We propose expanding the parameters to include sodium, potassium, and nitrite-N concentrations, which can be used in combination with nitrate-N concentrations to isolate the impacts of swine CAFOs as opposed to poultry CAFOs or other agricultural activities.[139] Likewise, the use of genetic markers and isotope tracers would improve groundwater monitoring with respect to isolating the impacts of a particular operation. Furthermore, groundwater-monitoring parameters should include Total Kjeldahl Nitrogen so that sample results enable calculation of total nitrogen levels using nitrate-N, nitrite-N, and Total Kjeldahl N. We also recommend adding more up-to-date pathogen indicators—including the *E. coli* or *enterococci* standards, which EPA recommended replace fecal coliform indicators three decades ago[140]—as well as parameters to detect antibiotic resistant bacteria.

Lastly, we encourage DEQ to clarify the sampling procedures and standards necessary to obtain accurate and high-quality groundwater monitoring data. For instance, facilities directed to monitor water quality on a case-by-case basis should also install monitoring wells both upstream and downstream from each lagoon/storage pond. The National *Field Manual for the Collection of Water Quality Data* published by the U.S. Geological Survey is a good blueprint for standardizing appropriate groundwater monitoring protocols.[141] Groundwater quality data and water level data should be entered into DEQ's GIS database and made available to the public.

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[138] Michael A. Mallin & Matthew R. McIver, *Season Matters When Sampling Streams for Swine CAFO Waste Pollution Impacts*, 16 J. of Water and Health; London 78–86 (2018).

[139] Harden, *supra* n. 111 (swine CAFOs in North Carolina tend to have higher sodium + potassium concentrations (commonly between 11 and 33 mg/L) and δ15N values of nitrate + nitrite (commonly between 11 and 26 ‰) relative to streams that have general, non-CAFO agricultural inputs).

[140] U.S. Environmental Protection Agency, *Ambient Water Quality Criteria for Bacteria*, 5 (1986).

[141] Franceska Wilde, *Collection of Water Samples (ver. 2.0): U.S. Geological Survey Techniques of Water-Resources Investigations* §4.2 (2006), http://pubs.water.usgs.gov/twri9A4/ (last visited Feb 24, 2019).

Although there is room to go farther, DEQ has made significant improvements to monitoring groundwater quality that will provide needed information and help protect human and environmental health.

Condition III.15: Facility Record Keeping

Lengthening the record retention period at facilities from 3 to 5 years is consistent with terms agreed upon in the Title VI settlement and with practices in other states. This expansion will improve accountability and transparency and make possible a more complete and longitudinal history to evaluate permit compliance. For these reasons, it is also vital that these provisions apply to all Permittees. DEQ should also facilitate more online record-keeping and allow for Permittees to convert these records from paper to digital and for the records to be filed with the Agency rather than exclusively stored on site. These changes would ease the compliance burden operators bear, while also greatly improving transparency and enhancing accessibility to information that impacts the environment and public health.

Condition III.18: Annual Certification

We strongly support the requirement that the annual certification be filed by all Permittees. This was also a core element of the Title VI settlement and is designed to increase accountability, oversight, and transparency, and minimizes any reporting burden. However, the proposed Certification form should be revised to require additional basic information about each permitted operation, including:

(a) each manure hauler by name and address;
(b) which and how much additional nutrient loads were added (including but not limited to, sludges, unused feedstuff, leachate, milk waste, septage, and commercial fertilizer); and
(c) the number of animal mortalities at each operation.

The requirement that these Annual Certifications be kept on file at the Department and made available to the public will ensure that the information most important to the public—relating to potential environmental and public health impacts of permitted operations—is up-to-date, complete, and readily accessible. We also support any effort to streamline or expedite the production, collection, dissemination, and availability of these records through digital or online methods.

Conditions III.17, 19-21: Notices to DWR and to the Public after Discharges

We appreciate the clarification that the larger discharges trigger all the requirements of lesser discharges. However, Condition III.17 should be changed to require facilities to contact DWR within a shorter time—12 as opposed to 24 hours. Other states have even stricter requirements for reporting dangerous events. In Ohio, permittees must report potentially dangerous spills and discharges within 30 minutes of discovery, and in Illinois, permittees must report discharges into the waters of the state to a hotline "immediately upon discovery." Given that the heaviest concentration of swine operations are in eastern North Carolina—where drainage conditions and other access to surface waters increase the contamination risks substantially in the event of a lagoon breach, overflow, or spill—and given the high number of lagoons and sprayfields in the floodplain, these changes to the public notice provisions are critical for DEQ to ensure transparency and accountability.

We also urge DEQ to require issuance of the press release within 24 as opposed to 48 hours after a discharge of 1,000 or more gallons of waste reaches surface waters. Two days is too long for the public to have to wait to obtain information of this magnitude, particularly given the speed and ease of using of digital media to distribute this information. We also urge DEQ to include the additional provision included in the stakeholder draft that the press release reference actions taken to prevent further discharge and a facility contact person and phone number. That information would ensure greater transparency and direct accountability to the public.

Condition IV.1: Unannounced Inspections

This new language is an important clarification of DEQ's existing authority and aligns with best practices used in other states to incentivize compliance.[142] This change is also consistent with the broader goals expressed by communities of concern and other environmental advocates throughout the stakeholder process for greater oversight and monitoring of permitted operations by DEQ.

Thank you for your consideration.

[142] *See Ind. Dep't of Envtl. Mgmt., Guidance Manual for Indiana's Confined Feeding Program* at 93 ("An investigation of a complaint or a spill requires no prior notification.) (Dec. 2014), available at https://www.in.gov/idem/cfo/files/guidance_manual_cfo_program.pdf; Okla. Stat. tit. 2, § 20-14(A) (2018) (providing that the State Board of Agriculture "shall make at least one unannounced inspection per year of every swine feeding operations licensed pursuant to the Oklahoma Swine Feeding Operations Feeding Act").

Respectfully submitted,

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ATTACHMENT 75

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Asthma Symptoms Among Adolescents Who Attend Public Schools That Are Located Near Confined Swine Feeding Operations

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Abstract

OBJECTIVES—Little is known about the health effects of living in close proximity to industrial swine operations. We assessed the relationship between estimated exposure to airborne effluent from confined swine feeding operations and asthma symptoms among adolescents who were aged 12 to 14 years.

METHODS—During the 1999–2000 school year, 58 169 adolescents in North Carolina answered questions about their respiratory symptoms, allergies, medications, socioeconomic status, and household environments. To estimate the extent to which these students may have been exposed during the school day to air pollution from confined swine feeding operations, we used publicly available data about schools (n = 265) and swine operations (n = 2343) to generate estimates of exposure for each public school. Prevalence ratios and 95% confidence intervals for wheezing within the past year were estimated using random-intercepts binary regression models, adjusting for potential confounders, including age, race, socioeconomic status, smoking, school exposures, and household exposures.

RESULTS—The prevalence of wheezing during the past year was slightly higher at schools that were estimated to be exposed to airborne effluent from confined swine feeding operations. For students who reported allergies, the prevalence of wheezing within the past year was 5% higher at schools that were located within 3 miles of an operation relative to those beyond 3 miles and 24%

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higher at schools in which livestock odor was noticeable indoors twice per month or more relative to those with no odor.

CONCLUSIONS—Estimated exposure to airborne pollution from confined swine feeding operations is associated with adolescents' wheezing symptoms.

Keywords

asthma; environmental health; epidemiology; school age children; school health

During the past 2 decades, the process of raising swine and other livestock has grown into a major industry in the United States. Production has shifted from smaller, family-owned farms to larger, industrialized confined animal feeding operations (CAFOs). Animals in North Carolina's industrialized operations are raised in confinement buildings, housing hundreds to thousands of hogs per operation. Residues of food additives, bedding, dried waste, and animal dander are vented from confinement buildings, and animal waste from the confinement houses is flushed into on-site cesspools, where it begins to decompose and aerosolize anaerobically before being sprayed onto nearby land. There are concerns about the health impacts of exposure to particulate matter, antibiotic residues, volatile organic compounds, and bioaerosols that are present in air that is downwind from confinement buildings, waste lagoons, and spray fields.[1–4]

In occupational settings, adverse respiratory symptoms and changes in bronchial responsiveness and lung function have been observed among confinement building workers.[5–12] Studies that have compared swine CAFO neighbors with other rural residents showed that neighbors reported more frequent respiratory symptoms and mucosal membrane irritation.[13] This literature about health impacts of residential exposures that arise from CAFOs focuses on adults[2,13–15] and may describe inadequately the potential respiratory health effects among children, who may experience notably different physical, educational, and social impacts from such exposures. We designed this research to assess the relationship between self-reported wheezing symptoms among adolescents who were aged 12 to 14 years and estimated exposure to airborne effluent from swine CAFOs.

METHODS

This study combined data about adolescents' respiratory health symptoms, data from a survey of school environments, and location data about swine CAFOs and public schools in North Carolina. Random-intercepts binary regression models were used to estimate prevalence ratios (PRs) that assessed the association between airborne swine pollutants and the prevalence of wheezing symptoms.

North Carolina School Asthma Survey Data

During the 1999–2000 school year, the North Carolina Department of Health and Human Services conducted a statewide respiratory health surveillance project to assess the prevalence of respiratory symptoms among middle school–aged children.[16] Approximately 67% (128 568 of 192 248) of all eligible students participated in the survey, which included core wheezing questions from the International Study of Asthma and Allergies in Childhood questionnaire, a standardized and validated instrument that combines a traditional written questionnaire with a series of video scenes that show children with asthma symptoms.[17–20] To complete the video-based survey questions, students viewed a sequence of video vignettes that showed adolescents experiencing asthma-related symptoms; each scene was followed by time to complete a written survey question, allowing each student to indicate whether he or she had experienced symptoms like those illustrated in the scene.[19,20] We analyzed the prevalence of any wheezing symptoms within the past year ("current wheezing"), as determined by responses to questions about wheezing at rest, waking at night as a result of wheezing, exercise-induced wheezing, and severe wheezing attacks. The definition of current wheezing used here is consistent with that applied in previous analyses of the North Carolina School Asthma Survey (NCSAS) data.[16,21–23]

To evaluate whether the estimated exposure had an impact other asthma-related outcomes, we assessed "severe wheezing" using responses to survey questions about waking at night as a result of wheezing and having a severe wheezing attack during the past year; considered the severe wheezing symptoms to be frequent when they occurred at least once per month ("frequent severe wheezing"); and evaluated physician-diagnosed asthma, medical care, and behavioral consequences of asthma-related symptoms.

Each adolescent also answered questions about age, race, Hispanic ethnicity, allergies, socioeconomic status, cigarette smoking history, and home environment. We included age as a continuous variable (centered at 13) and categorized all other variables: race (black/white); Hispanic ethnicity (yes/no); allergies to cat, dog, dust, grass, or pollen (yes/no); ever smoked cigarettes (yes/no); number of other smokers in household (0, 1, 2, or ≥3); and use of a gas stove at home (<1 time per month vs ≥1 times per month). Socioeconomic status was assessed using responses to a question about payment for lunch at school, with lower economic status designated by receiving free or reduced-price lunch at school compared with paying full price for lunch or bringing lunch to school.

School Environment Data

During the 2003–2004 school year, we mailed 4 copies of a survey to principals of 337 public schools and asked each to distribute the surveys to current school employees. More than 800 anonymous survey respondents, employed in 265 (79%) of the targeted schools, answered questions about their observations of the environmental conditions in and around the school buildings. The survey responses indicated whether there was visible evidence of the presence of cockroaches, rodents, or mold and noticeable odors from indoor (eg, mold) and outdoor (eg, nearby industries) sources of airborne pollutants. Responses were used to create school-level indicator variables for the presence of indoor respiratory irritants and sources of outdoor air pollution from agriculture and industries that are located near the school. Because of concerns about response bias resulting from social and political conflict surrounding industrial swine production in North Carolina, we asked survey respondents to answer a question about livestock odor generically rather than about odor specifically arising from swine operations. When we received >1 survey from a single school, schools were categorized as positive for a given survey question when any respondent reported the given condition.

Swine CAFO Exposure Estimates

Estimates of exposure to airborne pollution from 2343 swine CAFOs were generated using data from permits that were issued by the North Carolina Division of Water Quality to all CAFOs that house at least 250 animals and use a liquid waste management system. Records contained mandatory information about each CAFO facility, including geographic coordinates and the number, type, and weight of animals (called steady-state live weight [SSLW]) at each operation.[3,24] CAFO operators who filed applications for liquid waste management permits with the state agency provided latitude and longitude coordinates of their operations; the coordinates were verified and corrected, when necessary, when state inspectors visited the operations, although the extent to which the information was corrected by agency inspectors was not recorded in the data (S. Lewis, personal communication, 2002).

Separate exposure estimates were developed on the basis of distances between schools and swine CAFOs and of survey responses about noticeable odors from livestock farms. Distances and geographic directions between schools and CAFOs were calculated using the formulas given by Goldberg et al[25] and Sinnott,[26] respectively. We used calculations of proximity to create 3 metrics of potential exposure for each school: (1) distance to the nearest operation; (2) SSLW within 3 miles; and (3) a weighted SSLW based on the distance between the school and nearby swine CAFOs, the SSLW of each operation, and the proportion of wind measurements in the direction from the operation to the school. We obtained measurements of wind speed and direction recorded at 16 automated weather stations located throughout the state from the State Climate Office of North Carolina (Raleigh, NC). Hourly averages from January 1999 through December 1999 and from the weather station located nearest each school–CAFO pair were used to compute the proportion of time when the wind was blowing from the operation to the school. Weighted SSLW values for each CAFO within 3 miles of a school were the product of the squared inverse of the distance between the school–CAFO pair, the operation's SSLW value, and the proportion of time that regional wind measurements indicated that wind was blowing from the operation toward the school. For each school, weighted SSLW values were summed and the schools were assigned categories of low, medium, and high exposure on the basis of tertiles of the distribution of values among schools with 1 or more swine CAFOs located within 3 miles. A 3-mile radius was selected on the basis of previous research about the impacts of swine CAFOs on health and quality of life among neighbors who live within a 2-mile radius[2,13]; for this research, we expanded the potential zone of exposure to 3 miles because odors from swine CAFOs sometimes are reported at distances of >2 miles.

Study Population

Students in 499 public schools participated in NCSAS, and each student provided data about his or her respiratory health. Schools in 14 counties that did not contain a swine CAFO or border a county with at least 1 swine CAFO (*n* = 45), schools within the city limits of the 6 cities with populations >100 000 (*n* = 61), schools within 5 miles of the state border (*n* = 18), schools with <25 students surveyed (*n* = 34), schools that had closed or relocated since 2000 (*n* = 11), and schools that did not respond to the survey about in-school environmental conditions (*n* = 72) were excluded from our study. The remaining 265 public schools were included in our study. From these 265 schools, a total of 73 305 boys and girls who were aged 12 to 14 years responded to NCSAS. Of those, 58 169 (79%) who reported black or white race and provided complete data for all asthma survey variables of interest constituted our final study population.

Statistical Analyses

Multivariate analyses were conducted separately for individuals with and without self-reported allergies to cat, dog, dust, grass, and/or pollen. To assess the relationship between the prevalence of wheezing symptoms and the estimates of in-school exposure, we used random-intercepts binary regression. This method accounted for the hierarchical clustering of student-level data within schools. Specifically, we used a variation of the generalized linear mixed model $E(Y x) = \exp(\alpha + \Sigma\beta x)$ similar to those described by Singer[27] and McLeod,[28] in which the student's outcome is modeled by a combination of student-level (level 1) and school-level (level 2) models. The student-level model was defined as

$$\log_e(P_{ij}) = \beta_{0j} + \beta_1 x_{1j} + \beta_2 x_{2j} + \ldots + \beta_n x_{nj}, \quad \text{(level 1)}$$

where P_{ij} is the probability of outcome y = 1 for individual i in school j, p_{ij} ~ binomial; β_{0j} is school-specific intercept (intercept for school j); and β is the effect of individual-level predictor x_{ij}. Level 1 models included student-level variables for age, gender, race, Hispanic ethnicity, economic status, allergy status, cigarette smoking experience, number of other smokers in the household, and use of a gas kitchen stove at home. The school-level (level 2) model was defined as

$$\beta_{0j} = \beta_0 + \mu_1 z_1 + \mu_2 z_2 + \ldots + \mu_n z_m + \mu_{0j}, \quad \text{(level 2)}$$

where β_0 is the mean of school-level means for outcome y (ie, fixed intercept); μ is the effect of school-level predictor z_j; z_j is the school-level predictor for school j; $\mu_{0j} \sim N(0, \tau_{00})$; and τ_{00} is between-school variance. The level 2 models included main exposure variable(s) and indicator variables for rural school locale, survey-reported presence of indoor respiratory irritants (cockroaches, rodents, mold visible, mold odor, or flooding of school buildings within the past 5 years), and survey-reported industry other than a swine CAFO located near the school. The level 2 model, substituted into the level 1 model, results in a final 2-level random-intercepts model,

$$\log_e(P_{ij}) = \beta_{0j} + \beta_1 x_{1j} + \beta_2 x_{2j} + \ldots + \beta_n x_{nj} + \mu_1 z_1 + \mu_2 z_2 + \ldots + \mu_n z_m + \mu_{0j},$$

where μ_{0j} is the random intercept term. Associations were estimated as PRs (exp[μ]) using SAS statistical software version 8.2 (SAS Institute Inc, Cary, NC).

RESULTS

More than 26% (15 250 of 58 169) of students who participated in NCSAS during the 1999–2000 school year reported wheezing during the past year (ie, current wheezing). Table 1 shows adjusted PRs for individual-and school-level characteristics. Of the individual-level characteristics, the highest PR was observed for self-reported allergy status (PR: 2.20; 95% confidence interval [CI]: 2.14–2.27). Variations in the prevalence of current wheezing by school-level characteristics and indicators of school-specific environmental health conditions were less pronounced.

Of the 265 schools, 66 (25%), including 10 518 (18%) surveyed students, were located within 3 miles of at least 1 (range: 1–27) swine CAFO. More than 50% of the schools were within 7 miles of the nearest operation (median: 6.7 miles; range: 0.22–42.0 miles). The average SSLW capacity of operations that were located within 3 miles of a school was slightly lower than that of operations that were located beyond 3 miles (556 283 lb vs 605 139 lb), and, overall, the SSLW capacity of swine CAFOs increased with increasing distance from the nearest surveyed school (β [SE] per mile = 15 948 [4791]). On the basis of the environmental health surveys and according to survey respondents, livestock odor was noticeable outside buildings in 86 (33%) schools and inside the buildings in 39 (15%) schools.

Table 2 presents adjusted PRs for wheezing using each exposure measure separately for students with and without allergies. PRs were 1.05 (95% CI: 1.00–1.10) and 1.02 (95% CI: 0.94–1.11) for adolescents who did and did not have allergies, respectively, and attended schools that were located within 3 miles of the nearest swine CAFO. PRs were approximately unity for schools that were closer than 2 miles, compared with schools with no nearby swine CAFOs, and were 1.12 (95% CI: 1.04–1.19) and 1.08 (95% CI: 0.95–1.21), respectively, for students who did and did not have self-reported allergies and attended schools that were located between 2 and 3 miles from the nearest operation. Associations with SSLW and the weighted SSLW exposure categories also tended to be highest for the low exposure groups and closer to unity for higher exposure groups compared with schools with no nearby swine CAFOs. Basing potential exposure estimates on survey-reported livestock odor resulted in 20 fewer schools' and 3315 fewer adolescents' being considered unexposed. The prevalence of current wheezing was 24% and 21% higher among allergic and nonallergic students, respectively, at schools in which livestock odor was noted inside the school building 2 or more times per month relative to the prevalence at schools without any survey reports of livestock odor.

Table 3 presents adjusted associations between school proximity within 3 miles of a swine CAFO and alternative asthma outcomes as well as functional consequences of asthma-related symptoms. Results indicate that larger proportions of adolescents who attended school near at least 1 swine CAFO experienced respiratory symptoms, physician diagnosis, asthma-related medical treatment, activity limitations, and missing school because of their symptoms. In the population of all students, the largest PRs were observed for physician-diagnosed asthma (PR: 1.07; 95% CI: 1.01–1.14), medication use (PR: 1.07; 95% CI: 1.00–1.15), and visit to a physician or an emergency department or hospitalization (PR: 1.06; 95% CI: 1.00–1.12). Most associations were slightly higher in adolescents with self-reported allergies; however, the PR for physician-diagnosed asthma was higher among students without (PR: 1.14; 95% CI: 1.01–1.26) compared with those with (PR: 1.06; 95% CI: 0.99–1.12) self-reported allergies. Adjusted associations between these outcomes and the presence of livestock odor in and around the schools indicate only slightly elevated proportions of wheezing symptoms, physician diagnosis, use of asthma-related medical care, activity limitations, and missed school among students in schools where employees reported noticeable livestock odor (Table 4). When school-level exposures were assigned on the basis of reported livestock odor (Table 4), the PRs for severe wheezing (PR: 1.05; 95% CI: 1.00–1.10) and frequent severe wheezing (PR: 1.06; 95% CI: 0.98–1.14) were higher than when exposure was assigned on the basis of distance to the nearest swine CAFO (severe wheeze, ≤3 miles: 1.02 [95% CI: 0.97–1.07]; frequent severe wheeze, ≤3 miles: 1.01 [95% CI: 0.92–1.09]; Table 3).

DISCUSSION

We observed elevated prevalences of current wheezing among 12- to 14-year-old students who attended public schools near swine CAFOs, especially among students with self-reported allergies. Such associations are plausible, given that swine CAFOs are sources of bioaerosols, endotoxins, and other airborne asthma triggers. The availability of standardized symptom data and the independence of symptom and exposure data strengthen confidence in the validity of our findings. Overall, estimates of excess current wheezing symptoms among students who attended schools nearby swine CAFOs are as high as 24% among students who attended schools where livestock odor was reported outside as well as inside 2 or more times per month. Excess prevalence of current wheezing tended to be greater among students who reported allergies. Although the majority of the estimates are small in relative terms, the increases are important in absolute terms because of the high prevalence of asthma-related symptoms in this age group; the impact that symptoms have on adolescents' ability to attend school and participate in social, recreational, and physical activities; and the costs and burdens of symptom-related medical care. In these data, the effect estimates for swine CAFO exposures are of similar magnitude to the effects that have been estimated for established risk factors for wheeze, such as age, race, gender, economic status, Hispanic ethnicity, exposure to secondhand cigarette smoke, and use of a gas stove at home.

We estimated potential exposure on the basis of distance and a mailed survey. Although distance is a crude measure of exposure, our findings suggest a consistent trend toward higher symptom prevalence, especially among adolescents with allergies, at schools that were between 2 and 3 miles of a swine CAFO. The finding that schools that were located within 2 miles had a lower prevalence of current wheezing may reflect the lack of a direct relationship between exposure to etiologically active agents and distance. Use of distance and SSLW as exposure measures does not take account of waste management and sanitation practices of swine CAFOs, ages and conditions of the facilities' equipment, localized weather patterns, topography surrounding the school, school building structure, and ventilation practices, all of which may affect the quantity and the duration of the exposures. In addition, swine CAFO practices such as waste and sanitation procedures may be influenced by population density, land availability, and other features of the communities in which the operations are located, although we do not know the extent to which this occurs. Indeed, results of analyses that used exposure metrics of increasing complexity failed to show a monotonic dose-response relationship between the exposure and current wheezing, further suggesting that if the exposure is associated with an increase in respiratory symptoms, then relevant exposure may not correlate directly with the factors that we used for our distance-based exposure categories.

The higher prevalence of current wheezing among students who attended schools that were located 2 to 3 miles from the nearest swine CAFO compared with the prevalence among students who attended schools within 2 miles also may be attributable to exposures that were experienced at home, in the communities where students lived, and in other locations that could not be assessed in our study. In many of the rural areas in North Carolina, students may live many miles from the public schools that they attend. As the distance between the school and the CAFO becomes small, few homes can be equally close or closer to a CAFO; as the distance increases, more of the students' homes can be located closer to a CAFO than the distance between the CAFO and the school, and school-based exposure estimates will underestimate students' total swine CAFO exposures. In addition, reports of odor from swine CAFOs tend to be more common in early morning and evening hours rather than in the daytime, when students are in school. Although this phenomenon may not affect exposures in geographic areas where both schools and homes are far from CAFOs, identifying exposure as the distance between a school and a CAFO may be more problematic in regions where schools are located very near or within several miles of CAFOs if exposure varies throughout the day. Previous research that was conducted in a rural population of school-aged children who may have experienced swine farm exposures at home indicated a higher prevalence of asthma-related symptoms among children who lived on farms where swine were raised than among children who lived on farms where swine were not raised and among children who did not live on farms,[29] although the extent to which exposures that resulted from residence on a swine farm were attributable to performing chores or occupation-like tasks, rather than simply living close to swine, are unknown. Although information about adolescents' household farming exposures are unavailable in our study population, the majority of swine in North Carolina are raised in nonresidential, factory farm settings; therefore, the proportion of children who perform chores or live on swine farms is expected to be low.

Results of analyses of the distance-based measures of each exposure suggest lower prevalence of wheezing among students who attended schools that were located nearest to CAFOs and located in areas with the highest density of swine compared with those in the highest exposure categories. To assess potential misclassification of exposure, we excluded from all analyses schools with reported livestock odor from the unexposed distance-based categories, schools that were located beyond 3 miles of swine CAFO from the exposed survey-based categories, and schools for which survey respondents specifically identified livestock odor as arising from poultry and found no notable differences in the direction, magnitude, or precision of the PRs generated. An alternative explanation for the lower prevalence of wheezing among students in schools that were located nearby swine CAFOs may be the hygiene hypothesis, which postulates that early-life exposures and childhood infections may confer protection against hay fever, atopy, and asthma.[30,31] Specifically, rural living and early-life exposures to allergens, irritants, and other bioaerosols on farms may be associated with lower rates of atopy and asthma.[29,32–38] In our study, the prevalence of wheezing was slightly lower (−1.2%) in rural compared with non-rural schools. Although we could not assess early-life exposures, higher exposures to animal dander and bacterial endotoxin during early developmental stages among individuals who attend schools closest to swine CAFOs and therefore often live in rural areas could provide some resistance to exposures later in childhood and lead to lower prevalence of wheezing during adolescence compared with students who attend schools farther away.

Twenty-one percent (*n* = 72) of schools were excluded from our final analysis because of nonparticipation in our mailed survey about in-school environmental conditions. When we compared the populations of schools that participated and those that did not, we found differences in mean distance to the nearest swine CAFO (participating schools: 8.7 miles; nonparticipating schools: 8.0 miles), percentage of nonwhite enrollment (participating schools: 36%; nonparticipating schools: 42%), and percentage of enrolled students who received subsidized school lunches (participating schools: 48%; nonparticipating schools: 51%). Systematic differences between participating and nonparticipating schools in levels of exposure and prevalences of asthma-related symptoms could have influenced our findings.

We received up to 7 completed surveys per school, and for each survey question, we assigned an exposure to a school when any respondent indicated the presence of the exposure. This method of classifying schools' environmental conditions and, in particular, the presence of livestock odor at the school was sensitive to the number of surveys completed and returned from each school and did not take into account the variation in survey responses from a single school. Our intention was to survey employees in several occupations who would be familiar with different aspects of the school building and students' behaviors: teacher, administrator, maintenance or custodial staff, and school nurse or health care personnel. Previous literature about the economic, political, and social impacts of a strong swine industry presence in communities in Iowa and North Carolina suggested that residents who live near swine CAFOs may be reluctant to voice their concerns for fear of social ostracism or conflict in their communities.[39–42] Although our school survey was anonymous and designed to minimize risks for deductive disclosure of respondents' identities, we recognize the possibility that respondents may have underreported livestock odor out of concern for expressing their opinions, and we cannot know fully the extent to which our survey reports were influenced by the social and political context in the communities in which the schools were located.

Lack of data on medical risk factors, environmental asthma triggers, and classification of allergic status on the basis of survey reports rather than of a clinical assessment of atopy are limitations of this study. Because students self-identified asthma-related symptoms, our current wheezing variable may include other respiratory symptoms that the respondents experience and mistake for the symptoms that were illustrated in the video scenes. Cross-sectional asthma-related symptom data and survey-based exposure data prohibit specific assessment of temporal relationships between the symptoms and exposures evaluated here. Our findings are vulnerable to systematic error if students with asthma-related symptoms changed their environments or behaviors because of symptoms that were caused by exposure to airborne pollution that arose from swine CAFOs; such a systematic error would lead to underestimation of associations between swine CAFOs and asthma symptoms.

CONCLUSIONS

This research was designed to estimate exposures to a source of air pollution that is of great concern to swine CAFO neighbors and to investigate relationships between school exposures and respiratory health of middle school–aged children. Our findings identify a plausible association between exposure to airborne pollution from swine CAFOs and wheezing symptoms among adolescents. Environmental pollution measurement and standardized clinical information about asthma symptoms and atopic status could help to determine better the magnitude and the temporality of the relationships between swine CAFO emissions and respiratory symptoms. Our findings should be used by public health personnel who are interested in understanding possible adverse respiratory health consequences of an important rural environmental exposure.

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Abbreviations

CAFO	confined animal feeding operation
PR	prevalence ratio
NCSAS	North Carolina School Asthma Survey
SSLW	steady-state live weight
CI	confidence interval

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TABLE 1

Characteristics of North Carolina School Asthma Survey Participants and Public Schools in North Carolina

	N	Students Who Reported Current Wheezing, *n* (%)	PR (95% CI)[a]
Total	58 169	15 250 (26.2)	—
Age, y[b]			
12	17 905	4873 (27.2)	1.06 (1.04–1.08)
13	28 130	7268 (25.8)	1.00[c]
14	12 134	3109 (25.6)	0.95 (0.93–0.96)
Race			
White	43 590	10 919 (25.1)	1.00
Black	14 579	4331 (29.7)	1.04 (1.01–1.08)
Gender			
Male	28 342	6798 (24.0)	1.00
Female	29 827	8452 (28.3)	1.07 (1.04–1.10)
SES indicator			
Lunch not subsidized	41 719	10 088 (24.2)	1.00
Lunch subsidized	16 450	5162 (31.4)	1.16 (1.12–1.20)
Hispanic ethnicity			
No	54 827	14 236 (26.0)	1.00
Yes	3342	1014 (30.3)	1.11 (1.06–1.16)
Allergies			
No	31 480	5149 (16.4)	1.00
Yes	26 689	10 101 (37.9)	2.20 (2.14–2.27)
Ever smoked			
No	40 632	9154 (22.5)	1.00
Yes	17 537	6096 (34.8)	1.35 (1.31–1.39)
No. of other smokers in household[b]			
0	27 662	6138 (22.2)	1.00
1	16 079	4447 (27.7)	1.09 (1.07–1.10)
2	10 209	3178 (31.1)	1.18 (1.15–1.21)
≥3	4219	1487 (35.3)	1.29 (1.24–1.34)
Frequency of gas kitchen stove use			
Less than once per more	45 546	11 384 (25.0)	1.00
Once per month or more	12 623	3866 (30.6)	1.14 (1.11–1.17)
Rural school locale			
No	30 154	8074 (26.8)	1.00
Yes	28 015	7076 (25.6)	0.96 (0.92–1.00)
In-school asthma triggers[d]			
No	4619	1147 (24.8)	1.00
Yes	53 550	14 103 (26.3)	1.03 (0.95–1.11)
Location near non-livestock industry[e]			

	N	Students Who Reported Current Wheezing, *n* (%)	PR (95% CI)[a]
No	52 184	13 603 (26.1)	1.00
Yes	5985	1647 (27.5)	1.06 (0.99–1.13)

PR indicates prevalence ratio; SES, socioeconomic status.

[a]Adjusted for all individual-level and school-level covariates in the table.

[b]Included in the model as a continuous variable.

[c]Referent category.

[d]Environmental Health Survey responses about cockroaches, rodents, mold, and/or flooding in school buildings (no: 24 schools; yes: 241 schools).

[e]Environmental Health Survey responses about non-livestock industries located near the school (No: 236 schools; Yes: 29 schools).

TABLE 2

Associations Between the Prevalence of Wheezing and Exposure to Confined Swine Feeding Operations by Adolescents' Self-Reported Allergic Status, North Carolina

	Total No. of Schools	Self-Reported Allergies (n = 26 689)			No Self-Reported Allergies (n = 31 480)			All (N = 58 169)		
		Total No. of Students	Wheeze, n (%)[a]	PR (95% CI)[b]	Total No. of Students	Wheeze, n (%)	PR (95% CI)[b]	Total No. of Students	Wheeze, (%)	PR (95% CI)[c]
Current wheeze			10 101 (37.9)			5149 (16.4)			15 250 (26.2)	
Miles to nearest swine CAFO										
>3	199	21 898	8145 (37.2)	1.00	25 753	4138 (16.1)	1.00	47 651	12 283 (25.8)	1.00
≤3	66	4791	1956 (40.8)	1.05 (1.00–1.10)	5727	1011 (17.7)	1.02 (0.94–1.11)	10 518	2967 (28.2)	1.04 (0.99–1.09)
2 to ≤3	22	1865	822 (44.1)	1.12 (1.04–1.19)	2107	396 (18.8)	1.08 (0.95–1.21)	3972	1218 (30.7)	1.10 (1.02–1.18)
≤2	44	2926	1134 (38.8)	1.01 (0.95–1.07)	3620	615 (17.0)	0.99 (0.89–1.09)	6546	1749 (26.7)	1.01 (0.95–1.07)
Hog pounds (in millions) within 3 miles of school										
None	199	21 898	8145 (37.2)	1.00	25 753	4138 (16.1)	1.00	47 651	12 283 (25.8)	1.00
0.1 to <2.0	42	3342	1388 (41.5)	1.07 (1.01–1.12)	4017	713 (17.8)	1.03 (0.93–1.12)	7359	2101 (28.6)	1.05 (1.00–1.11)
2.0 to <5.0	12	733	294 (40.1)	1.04 (0.93–1.14)	858	150 (17.5)	0.99 (0.81–1.16)	1591	444 (27.9)	1.01 (0.91–1.12)
≥5.0	12	716	274 (38.3)	1.00 (0.89–1.11)	852	148 (17.4)	1.04 (0.85–1.23)	1568	422 (26.9)	1.02 (0.91–1.13)
Exposure category										
None	199	21 898	8145 (37.2)	1.00	25 753	4138 (16.1)	1.00	47 651	12 283 (25.8)	1.00
Low	21	1655	711 (43.0)	1.10 (1.03–1.18)	1922	359 (18.7)	1.09 (0.95–1.23)	3577	1070 (29.9)	1.09 (1.01–1.18)
Medium	22	1741	771 (40.8)	1.04 (0.97–1.12)	2139	378 (17.7)	1.01 (0.89–1.13)	3880	1089 (28.1)	1.03 (0.96–1.11)
High	23	1395	534 (38.3)	1.01 (0.93–1.08)	1666	274 (16.5)	0.97 (0.84–1.10)	3061	808 (26.4)	1.00 (0.92–1.08)
Livestock odor										
None	179	19 055	7188 (37.7)	1.00	22 438	3694 (16.5)	1.00	41 493	10 882 (26.2)	1.00
Outside school only	47	4625	1766 (38.2)	1.04 (0.98–1.09)	5593	843 (15.1)	0.94 (0.85–1.02)	10 218	2609 (25.5)	1.00 (0.95–1.06)
Outside + inside <2 times/mo	36	2745	1022 (37.2)	0.99 (0.93–1.06)	3137	550 (17.5)	1.04 (0.93–1.15)	5882	1572 (26.7)	1.01 (0.94–1.07)
Outside + inside ≥2 times/mo	3	264	125 (47.4)	1.24 (1.03–1.44)	312	62 (19.9)	1.21 (0.85–1.57)	576	187 (32.5)	1.23 (1.01–1.44)

[a] Any wheeze in the past 12 months (current wheeze).

[b]Adjusted for individual-level characteristics (gender, age, race, Hispanic ethnicity, economic status, smoking status, exposure to second-hand smoke at home, and use of a gas stove more than once per month) and school-level characteristics (rural locale, indoor air quality, and reports of other non-livestock industries nearby).

[c]Adjusted for variables listed above plus self-reported allergy to cats, dogs, dust, grass, and/or pollen.

TABLE 3

Associations Between the Prevalence of Asthma-Related Symptoms and School Location Within 3 Miles of a Confined Swine Feeding Operation by Adolescents' Self-Reported Allergic Status, North Carolina

	PR (95% CI) for ≤3 vs >3 Miles From Nearest Swine CAFO		
	Self-Reported Allergies (*n* = 26 689)	**No Self-Reported Allergies (*n* = 31 480)**	**All (*N* = 58 169)**
Wheezing symptoms			
Current wheeze	1.05 (1.00–1.10)	1.02 (0.94–1.11)	1.04 (0.99–1.09)
Current wheeze without physician diagnosis	1.08 (1.01–1.15)	0.99 (0.90–1.08)	1.04 (0.98–1.11)
Severe wheeze[b]	1.01 (0.96–1.07)	1.05 (0.96–1.14)	1.02 (0.97–1.07)
Frequent severe wheeze[a]	1.02 (0.92–1.11)	0.97 (0.80–1.14)	1.01 (0.92–1.09)
Physician-diagnosed asthma	1.06 (0.99–1.12)	1.14 (1.01–1.26)	1.07 (1.01–1.14)
Medical care			
Asthma-related physician visit, emergency visit, and/or hospitalization in past year	1.06 (1.00–1.13)	1.03 (0.92–1.13)	1.06 (1.00–1.12)
Asthma medication use in past year	1.09 (1.00–1.18)	1.03 (0.88–1.18)	1.07 (1.00–1.15)
Functional consequences of symptoms			
Activity limitations in past year as a result of asthma symptoms	1.09 (1.01–1.16)	—[b]	—
Missed school in past year as a result of asthma symptoms	1.06 (0.98–1.14)	—	—

[a]Among individuals with current wheeze.

[b]Nonconvergent model.

TABLE 4

Associations Between the Prevalence of Asthma-Related Symptoms and the Presence of Livestock Odor at the School by Adolescents' Self-Reported Allergic Status, North Carolina

	PR (95% CI) for Livestock Odor Reported Outside or Inside School Building Versus No Reported Odor		
	Self-Reported Allergies (*n* = 26 689)	**No Self-Reported Allergies (*n* = 31 480)**	**All (*N* = 58 169)**
Wheezing symptoms			
Current wheeze	1.03 (0.98–1.07)	0.99 (0.91–1.06)	1.01 (0.97–1.06)
Current wheeze without physician diagnosis	1.04 (0.97–1.10)	0.99 (0.90–1.07)	1.01 (0.96–1.07)
Severe wheeze[a]	1.06 (1.01–1.12)	1.00 (0.91–1.08)	1.05 (1.00–1.10)
Frequent severe wheeze[a]	1.04 (0.95–1.14)	1.10 (0.92–1.28)	1.06 (0.98–1.14)
Physician-diagnosed asthma	1.00 (0.94–1.06)	1.04 (0.93–1.15)	1.01 (0.95–1.06)
Medical care			
Asthma-related physician visit, emergency visit, and/or hospitalization in past year	0.99 (0.94–1.05)	1.01 (0.91–1.10)	1.00 (0.95–1.05)
Asthma medication use in past year	1.03 (0.96–1.11)	1.02 (0.89–1.15)	1.03 (0.96–1.10)
Functional consequences of symptoms			
Activity limitations in past year as a result of asthma symptoms	1.02 (0.96–1.08)	—[b]	—
Missed school in past year as a result of asthma symptoms	1.02 (0.94–1.09)	—	—

[a]Among individuals with current wheeze.

[b]Nonconvergent model.

ATTACHMENT 76

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Residential proximity to concentrated animal feeding operations and allergic and respiratory disease

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ABSTRACT

Background: Air emissions from concentrated animal feeding operations (CAFO) have been associated with respiratory and allergic symptoms among farm workers, primarily on swine farms. Despite the increasing prevalence of CAFOs, few studies have assessed respiratory health implications among residents living near CAFOs and few have looked at the health impacts of dairy CAFOs.
Objectives: The goal of this study was to examine objective and subjective measures of respiratory and allergic health among rural residents living near dairy CAFOs in a general population living in the Upper Midwest of the United States.
Methods: Data were from the 2008–2016 Survey of the Health of Wisconsin (SHOW) cohort (n = 5338), a representative, population based sample of rural adults (age 18 +). The association between distance to the nearest CAFO and the prevalence of self-reported physician-diagnosed allergies, asthma, episodes of asthma in the last 12 months, and asthma medication use was examined using logistic regression, adjusting for covariates and sampling design. Similarly, the association between distance to the nearest CAFO and lung function, measured using spirometry, was examined using multivariate linear regression. Restricted cubic splines accounted for nonlinear relationships between distance to the nearest CAFO and the aforementioned outcomes.
Results: Living 1.5 miles from a CAFO was associated with increased odds of self-reported nasal allergies (OR = 2.08; 95% CI: 1.38, 3.14), lung allergies (OR = 2.72; 95% CI: 1.59, 4.66), asthma (OR = 2.67; 95% CI: 1.39, 5.13), asthma medication (OR = 3.31; 95% CI: 1.65 6.62), and uncontrolled asthma, reported as an asthma episode in last 12 months (OR = 2.34; 95% CI: 1.11, 4.92) when compared to living 5 miles from a CAFO. Predicted FEV1 was 7.72% (95% CI: −14.63, −0.81) lower at a residential distance 1.5 miles from a CAFO when compared with a residence distance of 3 miles from a CAFO.
Conclusions: Results suggest CAFOs may be an important source of adverse air quality associated with reduced respiratory and allergic health among rural residents living in close proximity to a CAFO.

1. Introduction

Over the last several decades, large livestock farms, including concentrated animal feeding operations (CAFOs), have increasingly replaced small farms across the globe. The change in normative agricultural practices from smaller farms to large-scale farming productions, while more efficient for meat production, may also increase risk of adverse respiratory health or other outcomes among communities living in rural communities. CAFOs increase both the quantity and concentration of airborne particulates, gases, and vapors associated with farming (Schiffman et al., 2001). More than 400 compounds have been found in and around CAFO facilities, including volatile organic compounds (VOCs), endotoxins, ammonia, and hydrogen sulfide (Schiffman et al., 2001). While respiratory health effects among CAFO farm workers are well documented (Douglas et al., 2018; Kirkhorn and Garry, 2000; Radon, 2006), less is known about the extent to which CAFO air emissions affect the health of nearby residents.

Beyond increasing air emissions, potential for increased exposure to emerging antibiotic resistance microorganisms and outbreaks of zoonotic viral and bacterial pathogens have drawn attention to potential health risks among residents living near CAFOs (Gilchrist et al., 2007; Li et al., 2015; Rogers and Haines, 2005). Several agents, such as ammonia, hydrogen sulfide, endotoxins, and viral and bacterial pathogens from animal manure can be absorbed by dust particles and stay

Abbreviations: OR, Odds ratio; CI, Confidence interval; FEV1, Forced expiratory volume in one second

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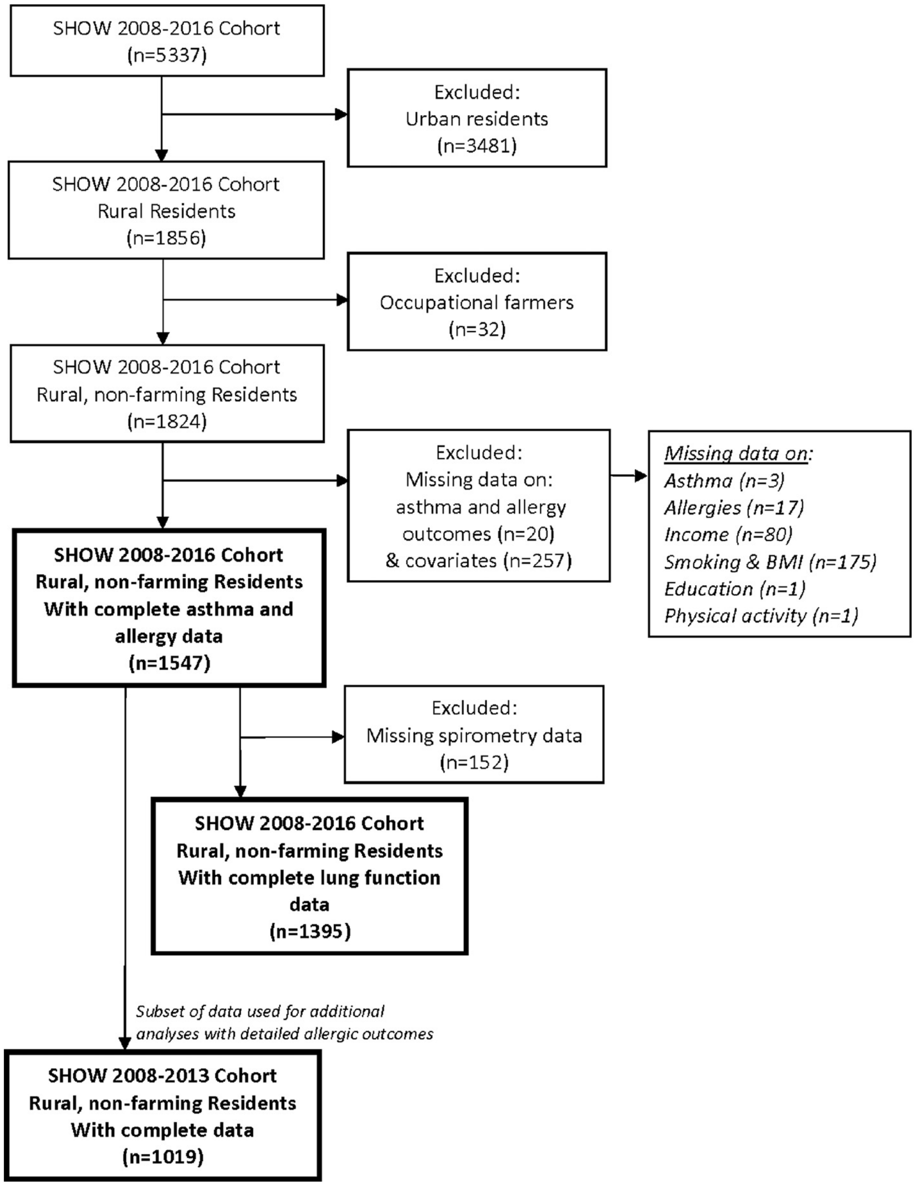

Fig. 1. Flow chart of the study sample, depicting exclusion criteria and sample size.

airborne for long periods and travel several miles, potentially exposing nearby residents to elevated levels of livestock-related agents (Cole et al., 2000; Omland, 2002; Dungan, 2010).

Three studies in the United States (U.S.) found the prevalence of asthma to be higher among children and adolescents attending schools (Mirabelli et al., 2006; Sigurdarson and Kline, 2006), and living (Pavilonis et al., 2013), near swine CAFOs. Studies among adults have found more mixed results. Two ecological studies among adults in the Netherlands (Hooiveld et al., 2016) and Greece (Michalopoulos et al., 2016) found null results when assessing residential proximity to livestock farms with allergy and asthma outcomes. Yet, an ecological study in North Carolina, U.S. found the prevalence of wheezing to be higher among adults living near swine CAFOs (Wing and Wolf, 2000). Two studies in rural Germany found the number of animal houses near a residence and measured ammonia levels to be associated with decreased lung function in adults (Radon et al., 2007a; Schulze et al., 2011). However, only measured ammonia levels were associated with sensitization of allergies (Schulze et al., 2011).

Three Netherlands studies found mixed results using general practice electronic medical records (EMR) to identify cases and controls of asthma and allergies. Inverse associations were found between distance to the nearest farm and asthma, allergies, and COPD (Borlée et al., 2015; Smit et al., 2014) and negative associations between the numbers of livestock farms within 1000 m of residence and lung function (Borlée et al., 2017). Yet living within 1000 m of > 11 farms had increased odds of wheezing and COPD (Borlée et al., 2015), and measured ammonia was associated with decreased lung function (Borlée et al., 2017). The only adult study in the U.S. to use EMR found living near a CAFO was associated with increased odds of asthma medication use and asthma-related hospitalizations (Rasmussen et al., 2017).

Several of the aforementioned studies (Sigurdarson and Kline, 2006; Wing and Wolf, 2000; Michalopoulos et al., 2016) consisted of people living near 2–3 identified livestock operations, small regions consisting of a few rural towns in Germany (Radon et al., 2007a; Schulze et al., 2011) or a rural county in the U.S. (Pavilonis et al., 2013). While studies in the Netherlands (Borlée et al., 2015, 2017; Smit et al., 2014) have used population-based study samples using electronic medical records from general practices, only one study in the United States has attempted to done so by using asthma hospitalization, emergency, and medication data from Geisinger Clinic in Pennsylvania (Rasmussen et al., 2017). Generating generalizable results from clinic data in the United States can be challenging as those who do not seek medical care due to inconvenience, cost, or lack of insurance go unreported.

The number of studies on the effect of CAFO air emissions exposure on respiratory health among nearby residents is limited and results are inconsistent. Furthermore, many prior studies have grouped exposure to CAFOs, removing individually variability. This study advances understanding of public health implications of CAFOS by using cubic spline regression to examine the association between residential proximity to CAFOs and respiratory health effects in order to account for non-linearity and retain individual levels of exposure. This study uses a well-characterized, rural sample of Wisconsin residents. Wisconsin ranks second after California as the state with the largest number of dairy cows (USDA, 2017); over 90% of its CAFOs being dairy CAFOs (WDNR, 2016). To our knowledge, no studies to date have looked at respiratory effects among residents living near dairy CAFOs.

2. Materials and methods

2.1. Study sample

Data came from the 2008–2016 Survey of the Health of Wisconsin (SHOW) state-wide sample of adults ages 18 and older ($n = 5338$). SHOW participants are randomly selected using a probability sampling proportion to size with replacement (PPSWR) approach (Nieto and Peppard, 2010). Between 2008 and 2013, a two-stage probability-based cluster sampling was used to randomly select census block groups (stage 1) and household addresses (stage 2) annually within strata of region and poverty level (Nieto and Peppard, 2010). SHOW 2014–2016 cohort was designed as a three-year sample instead of an annual sample as in prior years. A three stage cluster-sampling approach was employed. One county per strata was randomly selected within strata of county mortality rates, followed by random selection of census block groups by poverty status strata. Then 30–35 residential households were randomly selected via US postal service listings.

SHOW recruits 400–1000 participants every year. Across all years of the study, on average 67% of individuals who screen eligible complete each study component (interview and exam). However, participation rates vary from 47% in some urban communities to > 80% in some rural communities.

Fig. 1 describes the analytic sample selected for this study which includes a subset of 1856 (35%) rural participants among the 5338 SHOW subjects. Participants were considered rural if their residence was located in rural census block group defined by the U.S. Census Bureau as having fewer than 2500 people (U.S. Census Bureau, 2015). Additionally, 32 subjects who reported farming as their current occupation were excluded due to increased likelihood of occupational contact with livestock. While livestock contact could be assessed as a surrogate of a higher level of exposure to CAFOs, the number of individuals with occupational exposure was too small to examine this sub-population separately. Since those with livestock contact may or may not live near a CAFO, they were excluded to reduce confounding. Subjects with missing data on any of the respiratory outcomes or confounders of interest were also excluded from analyses, resulting in a final sample size of 1547 for asthma and allergy outcomes, and 1395 for objectively measured lung function outcomes. Detailed allergy data was only collected for 2008–2013 SHOW cohort, resulting in $n = 1019$ for detailed allergy analyses. All residential household addresses were geocoded using CENTRUS software (Pitney Bowes Inc., Stamford, CT) and linked to the nearest CAFO using ArcGIS v10.3 software (ESRI, Redlands, CA).

2.2. Concentrated animal feeding operations (CAFOs)

Data on CAFO location, type (dairy cow, hog, chicken, or turkey), years of operation and total animal units are maintained by the Wisconsin Department of Natural Resources' (WDNR) and Department of Agriculture, Trade and Consumer Protection (DATCP) under the Wisconsin Pollutant Discharge Elimination System (WPDES) program. WPDES falls under the Clean Water Act (CWA) National Pollutant Discharge Elimination System (NPDES) which requires states to regulate point source pollution to waters of the entire United States. CAFOs are defined by the CWA [Section 502(14)] as point sources, thus requiring a discharge permit and monitoring by WPDES.

CAFOs are defined as an animal feeding operation (AFO) where the following conditions are met: 1) animals are confined for a total of 45 days or more in any 12-month period and 2) animals do not have access to crops, vegetation or forage growth in the normal growing season. AFOs that have 1000 or more animal units (1 animal unit = 1000 pounds of live animal weight) are considered a large CAFO (1000+ cattle, 700+ dairy cows, 2500+ swine, 55,000+ turkeys). Medium CAFOs (300–999 cattle, 200–699 dairy cows, 750–2499 swine, 16,500–54,999) are additionally regulated under WPDES if the facility has a manmade ditch or pipe that carries manure or wastewater to surface water or if the animals come into contact with surface water that passes through the area where they are confined (40 CFR § 122.23 (b), 2012).

According to publicly available data downloaded from WDNR WPDES program there were a total of 284 CAFOs operating in Wisconsin in 2016. Ninety percent (244 large, 2 medium) were dairy CAFOs, followed by swine (5 large, 9 medium), beef (10 large, 3 medium), poultry (1 medium, 10 small). Publicly available data were limited, therefore additional data including the location, start date, and end date of all permitted CAFOs established between 2007 and 2015 was obtained via an open records request to the Wisconsin DATCP. The DATCP data was used to ensure CAFOs were in existence during SHOW participants' year of participation in the study (when residential address and health data were collected). Supplementary Fig. 1 from the WDNR shows the proportion of CAFOs by animal type has remained stable over the last decade, with over 90% of the CAFOs in Wisconsin being dairy.

Residential proximity to the nearest CAFO was used as a proxy to estimate potential exposure to air emissions from CAFOs. Distance from a participant's residence to the nearest CAFO was calculated using the "Near" tool in ArcGIS (ESRI, Redlands, CA). Participants were linked by cohort year to the nearest CAFO, only including CAFOs that were in existence during both the year they participated AND the year prior.

2.3. Allergy, asthma, and lung function

Self-report history of respiratory allergies and asthma was collected during in-home interviews. Current allergies were defined as having reported "yes" to the survey question "Do you still have allergies or hay fever?" as a follow-up to the question "Has a doctor or other health professional ever told you that you had allergies or hay fever?" Allergy type was defined based on response to the question "Where do allergy symptoms occur?" For this analysis individual with nasal, sinus, lung, eye, and skin as sites of allergies most likely to be triggered by CAFO air emissions were included. Those reporting digestion, food, or insect allergies were unlikely to be related to proximity to CAFOs and were defined as not having respiratory allergies.

Participants were defined as having current asthma if they responded yes to the survey question "Do you still have asthma?" which is a follow-up to the question "Has a doctor or other health professional ever told you that you had asthma?" Those who report having current asthma are also asked "During the last 12 months, have you had an episode of asthma or an asthma attack?" and if they have taken prescription medication to prevent or stop asthma attacks within the last 30 days.

Forced expiratory volume in 1 s (FEV1) and forced vital capacity (FVC) were measured via spirometry using an electronic peak flow meter (Jaeger AM, Yorba Linda, CA), and validated protocol (Richter et al., 1998). Trained technicians gave study participants explicit directions on how to breathe into the spirometry device. Measurements were considered valid if two FEV1 and FVC readings were within 10% of the highest value measured. FEV1 to FVC ratio (Tiffeneau index) and percent predicted FEV1 (FEV1 divided by predicted FEV1) were also assessed to account for inter-individual variability in lung function measurement. Predicted FEV1 was calculated using sex, race, age, and height as defined by the NHANES general U.S. population (Hankinson et al., 1999).

2.4. Covariates and confounding

Self-reported demographic data including age (years), gender (male vs. female), education (high school or less, some college, and bachelor's degree or higher) and household income were gathered via personal interviews. Poverty to income ratios were calculated using U.S. Department of Health and Human Services poverty guidelines and the midpoint of the household income range identified by the participant. Body mass index (BMI) was calculated from measured weight and height as kg/m^2. Physical activity was defined as Metabolic Equivalent of Task (MET)-minutes/week of moderate or vigorous activity using self-report data from a modified International Physical Activity Questionnaire - IPAQ (Craig et al., 2003). Income, BMI and MET-minutes/week were used as continuous variables in all statistical models, but log transformed to adjust for skewness. Additional self-reported questionnaire items assessed as potential confounders include: home smoking policy, household pets, smell of mildew or mold inside, and the use of any pesticides inside the home in the last 12 months. Sensitivity analyses were also run to test for potential confounding by previously identified environmental sources of allergies and respiratory health in the population (Schultz et al., 2017) residential proximity to the nearest primary or secondary roadway and industry were also examined.

2.5. Statistical analysis

Restricted cubic splines functions were applied to the residential distance in order to account for nonlinear relationships between distance to the nearest CAFO and respiratory health. Knots were placed at the minimum, maximum, and 25th, 50th, 75th percentiles of the distance variable (0.24, 6.17, 9.07, 17.9, 69.9 miles). Univariate as well adjusted multiple linear (lung function outcomes) and logistic (allergic and asthma outcomes) regression models were used to examine associations between residential proximity to a CAFO and respiratory health. Potential confounders selected a priori from the literature. Covariates that did not change the main effect estimate by > 10% were excluded from the multivariate models. An adjusted odds ratio (OR) or an adjusted beta-coefficient value with two-sided p-value < 0.05 was regarded as statistically significant. To acquire estimates from the spline regression, comparisons were made between different residential distances, while holding confounders constant. Residential distances of interest were chosen a priori from literature estimating air pollution and distance from CAFOs (Schinasi et al., 2011; Williams et al., 2011; Wilson and Serre, 2007; Wing et al., 2013; Michalopoulos et al., 2016), and from univariate spline regression trends between distance to nearest CAFO and each outcome. SAS version 9.4 (SAS Institute Inc. Cary, NC) was used for all statistical analyses. All adjusted analyses included sampling weights to account for sampling design, response rates and spatial clustering.

3. Results

Descriptive characteristics of the study population by residential proximity to the nearest CAFO are presented in Table 1. The majority of the study population (72%) lived > 5 miles from a CAFO, 4% (n = 65) lived < 1.5 miles of a CAFO and 23% (n = 361) lived 1.5–5 miles from a CAFO. Those living near a CAFO (< 1.5 miles) were more likely to be males, never-smokers, younger, less educated and diagnosed with asthma when compared with those living middle-distance (1.5–5 miles) and far (> 5 miles) from a CAFO. Those living near a CAFO were also less likely to live near a major roadway and have allergies when compared to the populations living middle-distance and far from a CAFO (Table 1). Unadjusted cubic spline plots revealed the log odds of asthma and allergy outcomes decreased, and lung function increased, as distance from a CAFO increased, leveling off at around 5 miles (Supplementary Fig. 2). Therefore, results include comparisons between distances of 1–3 miles compared with 5 miles from a CAFO.

Close residential proximity to a CAFO (living within 1–3 miles) remained positively associated with reporting any allergy symptoms even after controlling for gender, age, BMI, smoking status, education, income, pet ownership (Fig. 2). Mold in the home, smoking policy in the home, indoor chemical use, and residential proximity to an industrial site and roadway did not change the main effects and were not included in final models. Odds of allergies was > 2-fold when comparing living 1 and 1.5 miles from a CAFO to 5 miles from a CAFO (OR = 2.55; 95% CI: 1.49, 4.36 and OR = 2.02; 95% CI: 1.33, 3.08) and decreased as distance from a CAFO increased. Similar associations were seen among those with nasal- and lung-specific allergies, with the strongest associations seen with lung allergies. The adjusted odds of lung allergies was consistently > 2-fold higher among those living 1–3 miles from a CAFO when compared to those living 5 miles from a CAFO. Supplementary Tables 1 and 2 show results of all distance comparisons made for the previously mentioned allergy outcomes, along with current allergies assessed with the entire 2008–2016 cohort. While results indicate residential proximity is associated with eye and dermal allergies, none of the results were statistically significant (Supplemental Table 2).

Residential proximity to a CAFO was similarly associated with asthma and asthma control measures, including one or more asthma attacks in the last 12 months or taking asthma medication. Reporting current asthma was consistently about 1.8–1.9 times greater among those living 1–3 miles versus 5 miles from a CAFO (Fig. 3). The odds of ever being diagnosed with asthma was 3.11 (95% CI: 1.49, 4.36) and 2.67 (95% CI: 1.33, 3.08) when comparing 1 and 1.5 miles from a CAFO to 5 miles from a CAFO. Similar to the associations seen with current and nasal-specific allergies, the odds of doctor diagnosed asthma and asthma medication use decreased as distance from a CAFO increased. Those living 1, 1.5, 2, 2.5 miles from a CAFO, asthma medication was 4, 3, 2.5, and 2 times greater, respectively, when compared to those living 5 miles from a CAFO; all associations statistically significant. Odds of an asthma attack were consistently 2-fold higher at 1–3 miles versus 5 miles from a CAFO, with the odds being 2.34 (95% CI: 1.11, 4.92) times higher at 1.5 miles versus 5 miles from a CAFO.

Among the SHOW 2008–2013 cohort, the odds of reporting both allergies of nose or lungs and current asthma was 2.67 (95% CI: 0.97, 6.38) times greater and 2.14 times greater among those living 1 and 1.5 miles from a CAFO when compared to those living 5 miles from a CAFO (Fig. 2). Associations were lower at 2 and 2.5 miles but increased again to 2.74 (95% CI: 1.43, 5.23) when comparing 3 miles to 5 miles from a CAFO. This finding suggests that those in this study population with the presence of asthma or allergies may have allergic asthma. Results of all distance comparisons made with the aforementioned asthma outcomes can be seen in Supplementary Table 3. Similar directional associations are seen when distances of 1–3 miles are compared with 3, 4, and 6 miles as a reference value instead of 5 miles.

FEV1 percent predicted and FEV1/FVC were significantly lower among individuals living 1–3 miles from a CAFO when compared to those living5 miles from CAFO (Fig. 3).While not statistically significant, Fig. 4 shows FEV1 percent predicted was 11.31 L/s (95% CI: 0.51, 23.14) lower at 1 mile, and 7.00 L/s (95% CI: 2.26, 16.26) lower at 1.5 miles, when compared with 5 miles from a CAFO. The difference in FEV1 percent predicted decreased at 2 and 2.5 miles versus 5 miles until it reached 0 when comparing 3 miles versus 5 miles from a CAFO. FEV1/FVC was 0.039 (95% CI: 0.008, 0.07) lower at 1 mile, and 0.027

Table 1
Characteristics of the study population.

	Total study sample (n = 1547)	Residential distance from nearest CAFO: ≤1.5 miles 2.4 km (n = 65)	1.5–5 miles 2.4–8 km (n = 361)	≥5 miles 8 km (n = 1121)	
	N	%	%	%	p-trend
Gender					0.82
Male	682	47.7	44.3	43.8	
Female	865	52.3	55.7	56.2	
Age (in years)					0.48
18–39	320	23.1	18.8	21.1	
40–59	711	44.6	50.1	44.7	
60–94	516	32.3	31.0	34.2	
Race					0.12
White (non-Hispanic)	485	98.5	93.9	92.3	
Non-white	42	1.5	6.1	7.7	
Education					0.67
H.S./GED or less	475	38.5	31.0	30.2	
Some college	606	36.9	38.2	39.6	
Bachelors or higher	466	24.6	30.7	30.2	
Income					0.0001
< $25,000	246	6.2	11.6	17.8	
$25,000–$49,999	401	43.1	23.8	25.6	
$50,000–$99,999	590	35.4	45.7	35.9	
> $99,999	310	15.4	18.8	20.7	
Smoking status					0.84
Current	247	13.8	15.0	16.4	
Former	488	27.7	32.1	31.6	
Never	812	58.5	52.9	52.0	0.39
BMI					
< 25	381	20.0	28.0	23.8	
25–30	501	38.5	29.9	32.8	
> 30	665	41.5	42.1	43.4	
Physical activity					
< 600 met min/wk	392	24.6	27.7	24.6	0.50
≥600 met min/wk	1155	75.4	72.3	75.4	
Proximity to major roadway					0.02
< 300 m	493	20.0	28.5	33.6	
≥300 m	1054	80.0	71.5	66.4	

CAFO: concentrated animal feeding operation; km: kilometer; N: number; H.S.: high school; GED: General Education Development test; BMI: body mass index; wk.: week
p-trend: statistical significance by Chi-square test.

(95% CI: 0.003, 0.051) lower at 1.5 miles, when compared with 5 miles from a CAFO. Results of all distance comparisons, including FEV1 and FVC outcomes, can be found in Supplementary Table 4.

4. Discussion

These findings add to the emerging body of literature regarding public health impacts of concentrated animal feeding operations among rural populations. Much of the existing research has been conducted in Europe. This one of the first studies to examine how rural respiratory health is potentially influenced by farming practices in a general population based sample of adults in the United States. Among this well-characterized population-based sample, household proximity to a CAFO was associated with numerous respiratory outcomes including increased odds of self-reported allergies and asthma, and decreased lung function.

The use of cubic splines to explore nonlinear relationships between proximity to a CAFO and respiratory health outcomes was a strength of this study. Associations between residential proximity within 3 miles of a CAFO and increased prevalence of allergies, asthma, and decreased lung function were observed. Each of the respiratory outcomes followed a similar nonlinear relationship with distance from CAFOs and a 5 mile reference cut point was determined based on visual plots of the cubic spline functions of distance to the nearest CAFO regressed by each respiratory outcome separately. The non-linearity relationship seen in respiratory outcomes is not surprising as levels of constituents from air emissions from point sources (i.e. airports, roadways, industries, livestock facilities) tend follow a similar exponential decay as distances from the sources increase (Batterman et al., 2014; Dungan, 2010; Hadlocon et al., 2015; Maantay et al., 2009; Moreira et al., 2005; O'Shaughnessy and Altmaier, 2011; Polidori et al., 2010; Zhou and Levy, 2007).

Study findings are consistent with, and add strength to other U.S.-based studies of asthma and allergy symptoms among people living near AFOs or CAFOs. Pavilonis et al. (2013) found cumulative exposure to AFOs < 3 miles from residence was associated with an increased odds of asthma (1.51 $p = 0.014$) and asthma medication or wheeze (1.38 $p = 0.023$) among school age children. Similarly, Rasmussen et al. (2017) found adult asthmatics recruited from a clinic based sample and living within 3 miles of a CAFO compared > 3 miles had increased odds of ordering asthma medications (OR = 1.11 (95% CI: 1.04, 1.19)) and asthma hospitalizations (OR = 1.29; 95% CI: 1.15, 1.46). The smaller farm sizes may have contributed to the smaller effect sizes seen in Pavilonis et al. (2013) study. Not to mention, diagnosis of pediatric asthma is based on symptoms, which vary throughout a child's life, and also day to day (Asher et al., 2012; Jacob et al., 2017; Yang et al., 2017). The focus on hospitalizations and emergency department visits (Rasmussen et al., 2017) may have underestimated asthma events by excluding those who live near CAFOs but do not seek medical care due to being uninsured, financially insecure, or far from services.

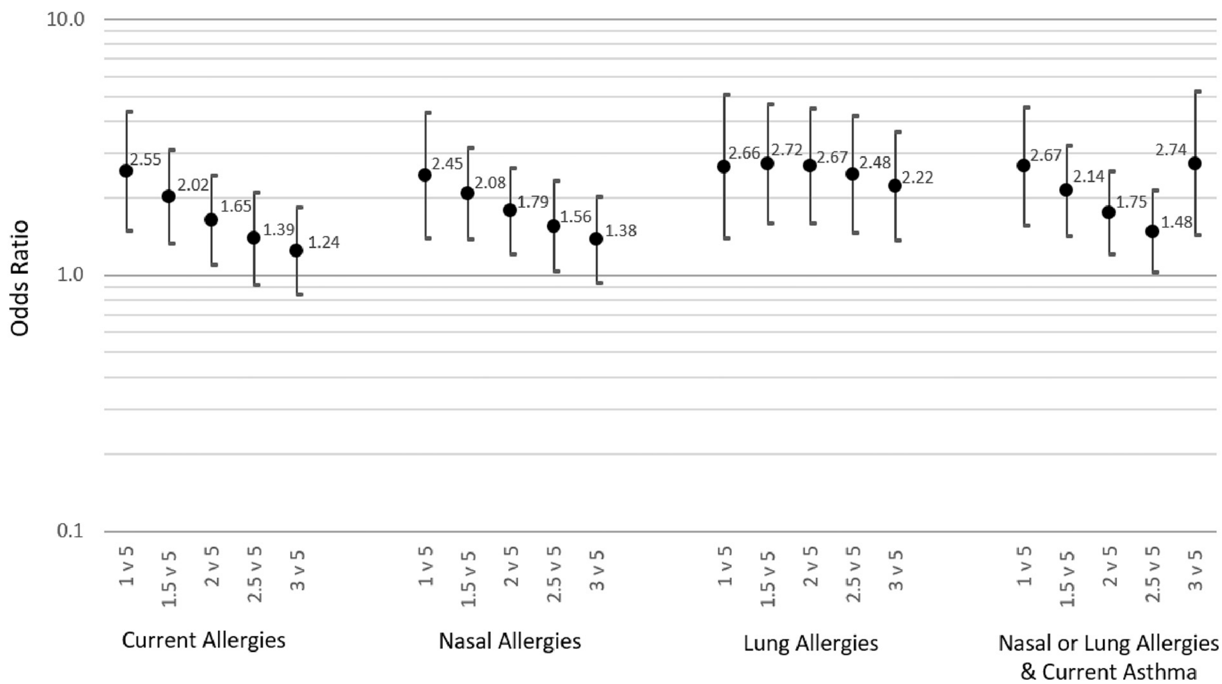

Fig. 2. Results of logistic regression assessing allergic outcomes by restricted cubic spline of residential distance to the nearest CAFO. Residential distances of 1, 1.5, 2, 2.5 and 3 miles (1.6, 2.4, 3.4, 4.0, 4.8 km) from a CAFO were compared with a residential distances of 5 miles (8.0 km) from a CAFO. Models are adjusted for gender, age, poverty to income ratio, education, BMI, smoking status, pet ownership and proximity to major roadways.

Results showed stronger associations with doctor diagnosed asthma than with current asthma. Discrepancies could be due to several factors, including a lack of clarity regarding the survey question assessing current asthma. Cross-tab frequencies on current asthma and asthma medication in the last 12 months revealed several participants reported not having current asthma because it is under control from taking asthma medication. Discrepancies between current asthma and doctor diagnosed asthma are not uncommon and can be due to several other factors including misdiagnosis, remission and relapse of asthma (Aaron et al., 2017).

Current allergies of any type and nasal allergies were 2.5 times higher at 1 mile from a CAFO, and decreased to 1.3 times higher at 3 miles from a CAFO when compared to 5 miles from a CAFO. Lung allergies remained 2.2–2.6 times higher at distances 1–3 miles from a CAFO when compared to 5 miles. The ability to assess allergy by type is a unique contribution, and something few studies have been able to do. Our study confirms findings from a few U.S. studies that have looked at proximity to CAFOs and allergies or allergy-like symptoms. Wing and Wolf (2000) found those living within 2 miles of a CAFO had increased prevalence of running nose, coughing, headache, itchy eyes, running nose, and sore throat. Mirabelli et al. (2006) found stronger associations with adolescents attending schools within 3 miles of a CAFO and asthma when stratified by those with allergies.

Findings in the U.S, are largely in contrast to those found in Europe,

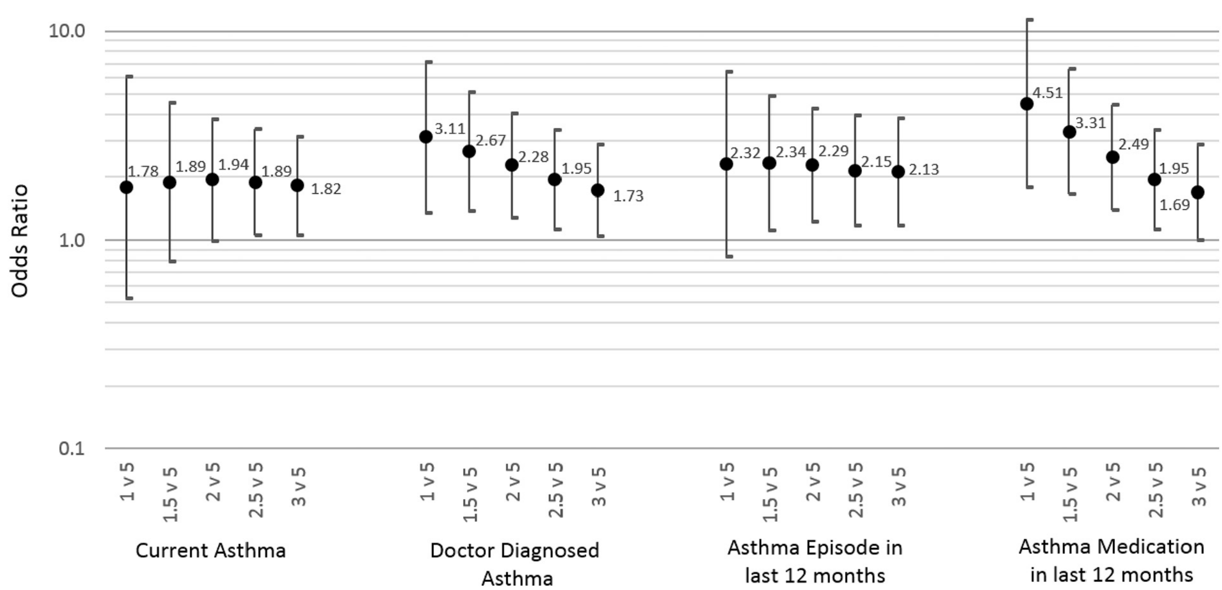

Fig. 3. Results of logistic regression assessing asthmatic outcomes by restricted cubic spline of residential distance to the nearest CAFO. Residential distances of 1, 1.5, 2, 2.5 and 3 miles (1.6, 2.4, 3.4, 4.0, 4.8 km) from a CAFO were compared with a residential distances of 5 miles (8.0 km) from a CAFO. Models are adjusted for gender, age, poverty to income ratio, education, BMI, smoking status, pet ownership and proximity to major roadways.

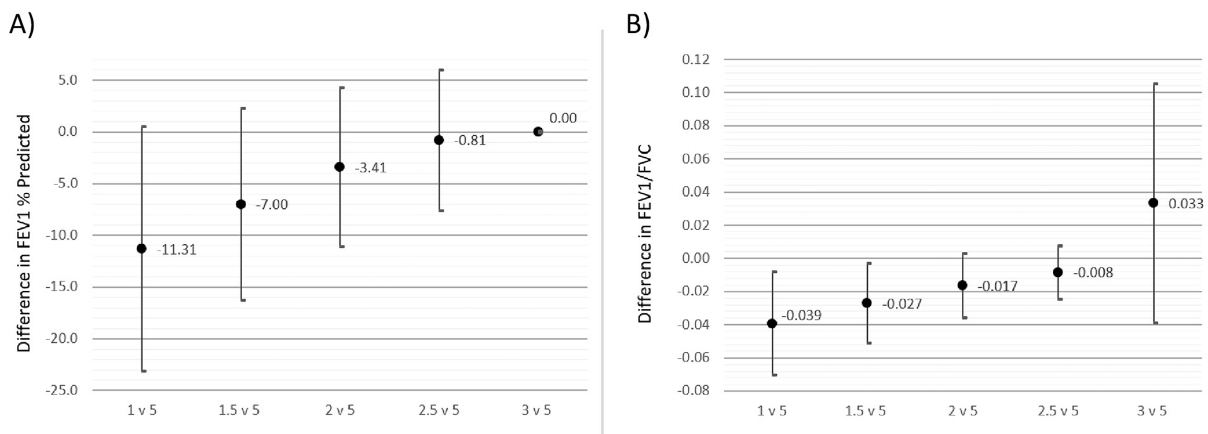

Fig. 4. Results of linear regression assessing (A) FEV1% predicted and (B) FEV1/FVC ratio by restricted cubic spline of residential distance to the nearest CAFO. Residential distances of 1, 1.5, 2, 2.5 and 3 miles (1.6, 2.4, 3.4, 4.0, 4.8 km) from a CAFO were compared with a residential distances of 5 miles (8.0 km) from a CAFO. Models are adjusted for gender, age, poverty to income ratio, education, BMI, smoking status, pet ownership, height, and physical activity.

particularly in Germany and Netherlands, where proximity cut points are typically at 500 m (0.31 miles) or 1000 m (0.62 miles). Several factors may contribute to this. For example, European confined livestock farms are generally smaller than in the U.S., densely clustered, and located in areas of higher population density. Thus, shorter distance cut points and livestock farm counts within 500 or 1000 m are more appropriate. Borlée et al. (2015, 2017) is one of the few studies to assess nonlinear associations using cubic splines of CAFO proximity and nasal allergies, finding inverse results to those seen in this study. Borlée et al. (2015) and Smit et al. (2014) both found inverse associations with doctor diagnosed asthma and allergies using EMR data in the Netherlands. Hooiveld et al. (2016), another Netherlands study which used EMR data found null results, but did not use individually measured exposure data as seen in the other two Netherlands studies. (Radon et al., 2007a) found self-reported asthma and nasal allergies were associated with increased livestock farm odor in Germany, but the number of animal houses near the home was not a predictor of allergies or specific sensitization. (Schulze et al., 2011) is one of the few European studies to find those exposed to higher ammonia levels from livestock farms to be 4.2 times more likely to be sensitized against ubiquitous allergens.

Findings from European studies largely suggest livestock farms provide a protective effect, if any, and support the hygiene hypothesis, specifically with allergy endpoints. The most comprehensive studies dedicated to disentangling the various factors of the protection against allergy provided by farming, such as ALEX, GABRIEL Advanced Surveys, and PASTURE, have been performed in European regions where dairy production is the main activity and where farming is not industrialized; (Alfvén et al., 2006; Genuneit et al., 2011; Riedler et al., 2001) rather in mid-mountain-altitude and among small cheese farms in areas like the Alps (Lis et al., 2008; Roque et al., 2016). In the ALEX and GABRIEL studies, the overall farm effect has been explained by specific and diverse exposure to types of livestock, crops, straw, fodder storage, manure, and unpasteurized milk (Vuitton et al., 2014; Vuitton and Dalphin, 2017). However, the industrialization of farming is thought to have decreased the microbial diversity and increased the abundance of specific bacterial genera which may induce inflammatory response (Kong et al., 2012; Powers et al., 2015; Schaeffer et al., 2017). This is further supported by studies showing household dust and the nasal microbiota from farm children to have higher alpha and beta diversity than those found from nonfarm children, and lower nasal microbiota diversity to be associated with asthma prevalence (Depner et al., 2015; Pekkanen et al., 2018).

However, protective or null effects have also been seen among adults living near non-traditional, industrialized confined livestock operations in Europe, which are generally smaller in size than CAFOs seen in the United States. (Borlée et al., 2015, 2017; Hooiveld et al., 2016; Radon et al., 2007b; Smit et al., 2014; Michalopoulos et al., 2016) This suggests that the dose of exposure to microbes, in combination with particulate matter, gases, and vapors emitted from livestock operations, may also play a role in the respiratory health effects seen among nearby residents. While it appears both the dose and type of exposure to microbial agents from livestock farms may be of importance, additional research is needed with attempts to identify etiological agents from livestock agents. Differences in the size and management practices of the livestock farms themselves, the microbial diversity emitted, the regulations imposed on them or the populations living near them are all factors which may have contributed to the different results seen in the European studies when compared to the U.S.

Discrepancy in findings across studies in Europe and the U.S. could also be due to varying ways in which asthma and allergies are diagnosed, or defined. Asthma diagnoses are often made based on symptoms and treatment based on severity of symptoms. However, asthma is a heterogeneous disease that manifests differently in different people, symptoms can vary over time and change day to day within the same person, and therefore diagnoses may vary by individual, doctor, or region. (Jacob et al., 2017) Previous studies have showed the challenges to accurately diagnosing asthma have resulted in over- or underdiagnosis of asthma. (Jacob et al., 2017) Furthermore, distinctions between allergic and non-allergic asthma can often not be made without a serological test. All these factors may also contribute to discrepancies in results across the literature.

Lung function was positively associated with proximity to a CAFO, with lung function improving as distance from a CAFO increased. The effect sizes, although most non-significant, were similar to results from European studies of adults in Germany and the Netherlands (Schulze et al., 2011; Radon et al., 2007a, 2007b). A distance of 1.5 miles was associated with −7.0% predicted FEV1 when compared with a distance of 5 miles from a CAFO. Schulze et al. (2011) found a −8.19% predicted FEV1 among those with average ammonia concentration greater than or equal to 19.71 $\mu g/m^3$ when compared to those with levels below. Similarly Radon et al. (2007a, 2007b) reported a −7.4% predicted FEV1 among those more than twelve animals houses within 500 m of home. While definitions of exposure to CAFO varied, the fact that all three studies found very similar results suggests residential proximity to a CAFO, or many AFOs, is likely associated with decreased lung function.

As one of the first studies in the U.S. to use a randomly selected statewide, population-based sample of rural adult residents to assess multiple respiratory health effects among people living in proximity to CAFOs, this study has numerous strengths. Prior U.S. studies have tended to rely on grouped exposures, removing individually variability among the exposure (Mirabelli et al., 2006; Rasmussen et al., 2017; Sigurdarson and Kline, 2006; Wing and Wolf, 2000). Our study was able to report on the nonlinear association between proximity to the nearest CAFO and respiratory health outcomes in the U.S., providing an important link between dispersion modeling of CAFO emissions and human health effects.

While utilizing a randomly selected statewide sample is a strength of this study, it is also a limitation. Rare exposures, such as living near a CAFO in the U.S., can result in low power and are best studied with cohort studies where subjects are selected by exposure status. Low power may have resulted in our inability to detect interaction with proximity to a CAFO and smoking status. Though we carefully controlled for multiple confounding factors, residual confounding or confounding by other unmeasured factors may affect estimated associations including individuals with potential higher livestock exposures via occupation. However, the number of subjects reporting livestock exposure was small and not sufficient to examine as a separate subpopulation. Similarly, residents in urban areas were not included to reduce bias and reduce potential unmeasured confounding introduced by air pollution sources unique to urban areas. The cross sectional nature of this study also limits conclusions regarding the temporal association between exposure and respiratory outcomes, particularly self-reported asthma prevalence. Self-report is not ideal and can lead to recall bias, however asthmatic and allergic symptoms may go clinically underreported in rural areas, where people may be less likely to seek medical care due to inconvenience, cost, or lack of insurance. While objective and self-report data on asthma was available, this study relied on self-report of allergies. Therefore, results cannot definitively tease out allergic and non-allergic asthma, something that would have strengthened the study and increased comparability with other studies. Furthermore, the lack of allergic sensitization data limits comparisons with other studies.

We were able to acquire retrospective CAFO data and ensure CAFOs linked to participant residences were in existence prior and during their study participation. However, the farm size and type could not be validated from this data. Additionally, we were unable to account for proximity to non-CAFO livestock farms. The assumption being made here is that the distribution of smaller farms is random throughout the study sample, resulting in non-differential misclassification bias. This assumption results in estimates biased towards the null.

5. Conclusion

In summary, residential proximity to a CAFO among individuals from a randomly sampled general population health survey was positively associated with self-reported nasal and lung allergies, asthmatic outcomes, and objectively measured lung function. This study provides evidence for respiratory health effects among residents living near dairy CAFOs. CAFOs may be an important source to regulate as current evidence suggests that large livestock farms may contribute to health disparities among rural residents. Building on findings from this observational study, future research should consider longitudinal study designs, more refined estimates of exposure source-apportioned air constituents in nearby homes, and more systematic tracking and validation of outcomes. More research is also needed to understand the mixtures of airborne agents from nearby livestock facilities in order to identify any etiological agents which may be associated with asthma, allergies, or lung function in residents living near large livestock facilities. Passive air pollution monitoring using filters or dust collection in homes would be useful to collect in order to better understand composition of air particles and how they may change over time. A cohort study which selects study participants by residential proximity and monitors respiratory health symptoms across multiple seasons should also be considered. Alternatively, a case-control study that recruits from hospitals and clinics in areas with a large concentration of CAFOs and could follow-up self-reported symptomology overtime could overcome some existing limitations of this work.

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Declaration of Competing Interest

None.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2019.104911.

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ATTACHMENT 77

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Spatial and temporal variability in excessive soil phosphorus levels in eastern North Carolina

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Abstract

Numerous studies have shown that accumulation of excessive soil phosphorus raises the potential for phosphorus export and eutrophication of adjacent surface waters. Soil phosphorus data from the North Carolina Agronomy Division's database were analyzed for two-year periods spanning the decades of the 1980s and 1990s for 39 eastern North Carolina counties. Eastern North Carolina supported extensive row crop agriculture, rapidly growing intensive livestock industries, and a growing human population during these decades. Excessive soil phosphorus levels, defined as having a soil phosphorus index (P-I, based on Mehlich III testing) $>$ 100, occurred in over 40% of almost a million samples reported for the three two-year periods analyzed. Excessive soil P-I levels were most frequent in central eastern North Carolina, declined in the 1980s and rose again in the 1990s. The distribution of row crop area with excessive soil P-I levels was very similar in time and space. Increases in the area harvested for cotton (+635%) and pasture (+523%) with excessive soil P-I levels were particularly large during the 1990s, when crop areas harvested associated with excessive soil P-I levels for other major crops (corn, tobacco, peanuts) declined. Residential and recreational land uses were associated with similarly high frequencies of excessive soil P-I levels, but these land uses were relatively unimportant ($<$5% area) compared to agricultural land use (~34%) in the region. Recent increases in fertilizer shipments (approximately twofold in the late 1990s) likely reflected increased cotton production. Rapid growth in concentrated animal production (almost twofold increase in total animal units (AU) between 1992 and 2001), with accompanying land application of wastes, accounted for increases in soil P-I values in pasturelands in the 1990s, particularly in central eastern North Carolina, where these activities were concentrated. The potential threat to water quality from export of excessive soil phosphorus is therefore greatest in this region. North Carolina is currently developing a Phosphorus Loss Assessment Tool (PLAT) in an attempt to manage the challenge posed by excessive soil phosphorus levels.

Introduction

Phosphorus is generally considered the macronutrient most frequently responsible for eutrophication problems in freshwater and, to a lesser extent, in estuarine ecosystems (Vollenweider 1976; Hecky and Kilham 1988; Sundareshwar et al. 2003). Excess phosphorus loadings to aquatic ecosystems can drive algae blooms, stimulate bacterial growth, increase biological oxygen demand, and alter aquatic community structure (Correll 1998; Carlsson and Caron 2001; Mallin et al. 2001).

The importance of phosphorus loading from non-point sources, in addition to more well-regulated point source discharges, is widely recognized, e.g., Wanielista and Yousef (1993), Carpenter et al. (1998). Among non-point sources, agricultural sources of phosphorus, which include conventional fertilizers and land-applied animal manures, have received considerable attention, e.g., Sharpley et al. (1994), Daniel et al. (1998), Parry (1998), Sims et al. (1998), Sharpley and Tunney (2000). The relatively recent spread of intensive livestock production using concentrated animal feeding operations (CAFOs) has also led to research on the environmental effects of land-applied manures, with much research focused on phosphorus loading to aquatic ecosystems from land application of wastes from livestock in various countries (Austin et al. 1996; Liu et al. 1997; Sharpley 1997; Sims et al. 1998; Martinez and Peu 2000; Novak et al. 2000; Sauer et al. 2000; Schoumans and Groenendijk 2000; McDowell et al. 2001a, b, c).

Assessment of the risk posed to water quality by phosphorus application to the landscape is a complicated task (Sharpley et al. 1994; Edwards et al. 2000; Nash and Halliwell 2000; McDowell et al. 2001a, b, c). Phosphorus tends to associate strongly with soils, in contrast to more readily soluble nitrogen compounds. Soil erosion has, therefore, been considered a primary pathway of phosphate export. However, other mechanisms of phosphorus transport from landscapes into aquatic ecosystems have been documented, including surface transport of unincorporated materials containing phosphorus, loss of dissolved phosphorus in surface runoff, and leaching of phosphorus through the soil (Austin et al. 1996; Sharpley 1997; Sims et al. 1998; Leinweber et al. 1999; Turner and Haygarth 1999; Turtola and Yli-Halla 1999; Djodjic et al. 2000; Martinez and Peu 2000; Novak et al. 2000).

Soil phosphorus concentration is frequently assessed as a measure of the risk of phosphorus export from landscapes (Pote et al. 1996, 1999; Liu et al. 1997; Sharpley 1997; Sims et al. 1998; McDowell et al. 200la, b, c; Sharpley and Tunney 2000). Although other factors, including distance from water, precipitation/irrigation duration and intensity, water table depth, soil type, phosphorus sorption capacity, slope, tillage, phosphorus application rate and form, and plant cover, affect phosphorus export (Sharpley et al. 1994; Hooda et al. 2000), the concentration of soil phosphorus in excess of plant needs usually determines the potential for phosphorus export. Studies have shown that the amounts and concentrations of phosphorus in surface runoff and in water leaching downward increase significantly as soil phosphorus concentrations rise (Sims et al. 1998; Turner and Haygarth 1999; Novak et al. 2000). Recent studies have also shown that phosphorus export rises dramatically beyond a threshold soil phosphorus concentration or 'break point', which may vary among soil types (McDowell et al. 2001b). Guidelines for interpretation of soil test data on phosphorus concentrations have therefore highlighted the increased risk of phosphorus export above certain soil phosphorus levels, although actual export rates can vary with the factors listed above (Sharpley and Tunney 2000).

Excessively high soil phosphorus levels have been shown to arise from long-term application of commercial fertilizers and, more recently, from widespread land application of fecal wastes, both human and animal. The application of animal wastes at agronomic rates based on nitrogen content and the typically low nitrogen:phosphorus ratio in fecal wastes result in a high potential for accumulation of excess soil phosphorus (Carlile and Phillips 1976; Westerman and King 1983; Barker and Zublena 1995; Jongboeld and Lenis 1998). Regions where row-crop agriculture and/or CAFOs are particularly concentrated have shown significant increases in soil phosphorus concentrations to potentially problematic levels, and regulatory agencies have responded with various management measures (Schoumans and Groenendijk 2000; Sharpley and Tunney 2000).

Non-agricultural land uses, such as residential development (Schueler 1994) and recreational uses, e.g., golf courses (Mallin et al. 2002), can also contribute to phosphorus loading to aquatic ecosystems through fertilizer application, land disturbance, and changes in drainage. As shifts in the relative proportions of agricultural and other land uses occur with changing patterns of human population growth, it is important to consider the relative potential of these different land uses to cause impacts on phosphorus loading and impacts on aquatic ecosystems, particularly in coastal regions with extensive development activities and population shifts.

Eastern North Carolina now supports all of the activities potentially associated with increasing soil phosphorus concentrations. This low-lying, mostly flat portion of the coastal plain has extensive riverine and estuarine systems vulnerable to nutrient loading (Figure 1; Bricker et al. 1999). Eastern North Carolina has supported significant row crop production, particularly tobacco, corn, soybeans and small grains, for many years. Eastern North Carolina has also become a major center of intensive livestock production, especially swine and turkeys, in the last two decades (Cahoon et al. 1999). These agricultural activities have been particularly concentrated in eastern

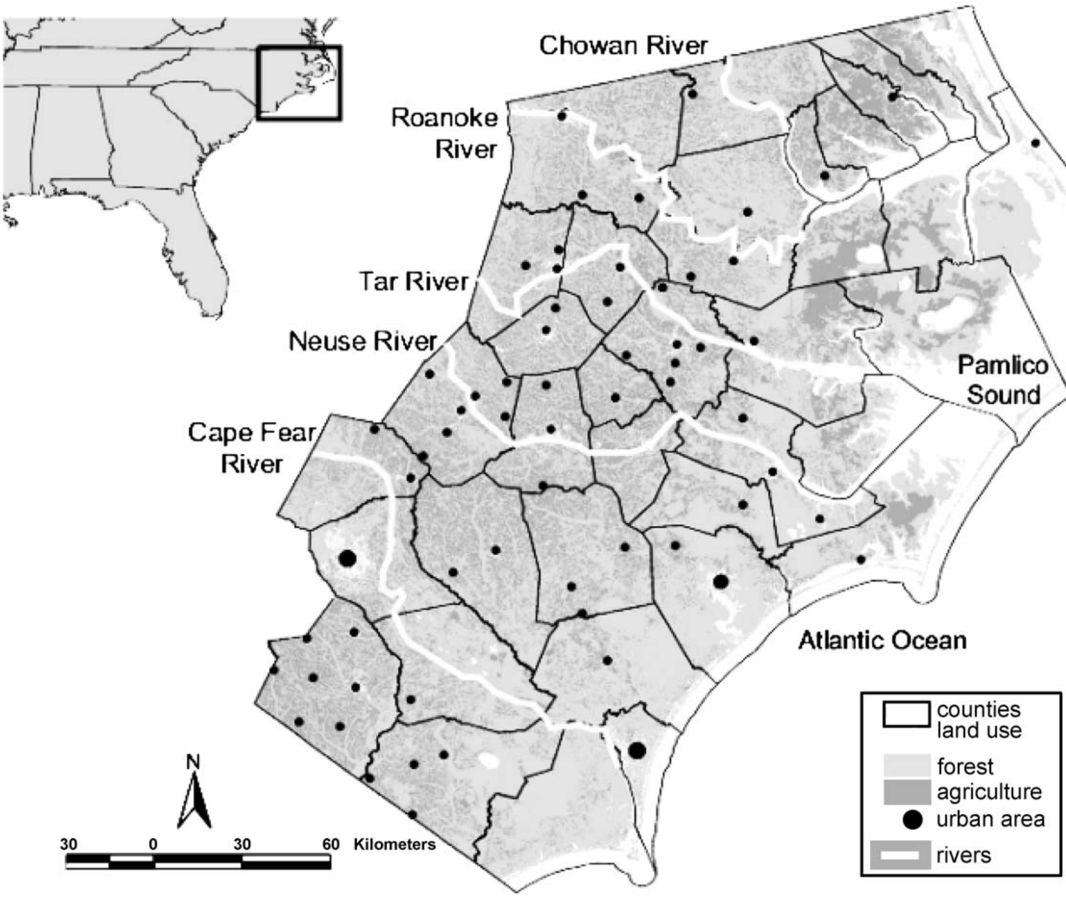

Figure 1. Map of the 39 counties of eastern North Carolina, illustrating county boundaries, major water bodies, and forested, agricultural, and urban areas. Large dots denote urban areas with populations $>50{,}000$, smaller dots denote towns with smaller populations. Land use data from U.S. Geological Survey (2000).

North Carolina. Finally, North Carolina experienced population growth of 21.4% during the decade 1990-2000, with even higher growth rates in coastal counties. Other states and countries, e.g., Maryland and the Netherlands, have experienced serious water quality problems arising from similar trends. The objective of this investigation was to determine the patterns of occurrence of excessive soil phosphorus levels in eastern North Carolina during the last two decades in relation to the major crop types and land uses.

Materials and methods

The North Carolina Department of Agriculture's Agronomy Division provides free soil testing services for state residents and maintains a publicly accessible database of soil test results. Results are compiled for each state fiscal year (FY, July 1 – June 30) with summary soil test data grouped by result ranges for each crop or land use type for each county. Results of individual sample submissions with numerical soil test data and accompanying information have also been posted on the Division's web page since 1998.

The N.C. Agronomy Division employs the Mehlich III extraction method followed by atomic absorption spectrometry to measure soil-P levels (Tucker et al. 1997). The Mehlich III method is considered to extract 'plant-available' phosphorus compounds ('M3P', primarily orthophosphate), providing a useful measure of soil phosphorus concentrations with respect to crop needs (N.C. Coop. Ext. Serv. 1997). Results are reported using a Phosphorus Index (P-I) system, which scales soil-P levels in terms of plant needs. P-I values can be converted into concentration units, e.g., kg P ha^{-1}, using appropriate factors. P-I values in the range 0-10 are considered very low, 11-25 low, 26-50 moderate, 51-100 high, and >100 very high (Tucker et al. 1997). Fertilization generally drives a positive plant response at P-I values <50. P-I values >100 ($=240$ kg P ha^{-1}) are considered excessive and are thought to indicate an increased risk of P export and water pollution (N.C. Coop. Ext. Serv. 1997).

Soil P-I data for 39 counties in eastern North Carolina were obtained for the years (FY) 1980 and 1981, 1990 and 1991, and 2000 and 2001, broken down into P-I ranges by crop or land use type, from the N.C. Agronomy Division. Data from two-year periods were averaged in order to provide larger data sets to analyze patterns over each decade and to mitigate year-to-year variability in soil testing. The limitations and assumptions involved in the soil P-I data summarized here must be considered before evaluating their full importance. North Carolina's soil test data were derived from samples collected by many individuals at different frequencies, times of year, and different densities per unit area across different land uses. The use of large data sets and two-year intervals may have reduced some of the variability otherwise expected, but sampling biases cannot be completely ruled out. Soil samples are supposed to be collected from the top 10 cm of soil and so represent only surface concentrations of phosphorus. Most phosphorus added to soils, i.e., as fertilizer or animal waste, is found in the top portion of the soil (Sims et al. 1998; Martinez and Peu 2000; Novak et al. 2000), so soil P-I data probably represented the bulk of the soil P content, certainly the fraction most labile to runoff and erosion losses (Liu et al. 1997). However, soil P-I data represent an underestimate of total P in the soil when incorporation into or leaching to deeper horizons is considered (Sims et al. 1998; Turner and Haygarth 1999; Novak et al. 2000), and because soil P-I levels determined by the M3P method are considered to indicate only extractable, 'plant available' P (N.C. Coop. Ext. Serv. 1997). The categorization of soil P-I data sets into classes, particularly the class considered here, 'P-I >100', likely also understates the magnitude of excessive soil P-I levels.

Harvested crop acreage, fertilizer shipment and livestock census data were obtained from the N.C. Department of Agriculture's annual statistics yearbooks and converted as appropriate into metric units (hectares and metric tons).

The frequency of excessive soil P-I values as a percentage of total row crop area per county was calculated using soil P-I data for crop types and data from the N.C. Department of Agriculture's annual statistics yearbooks (N.C. D.A. 1982, 1992, 2001) on harvested area for each crop type in each county for 1980, 1990, and 2000. Areas associated with 'home grounds' and 'turf' land uses were not assessed in these yearbooks. 'Miscellaneous' crop type samples were apportioned to each row crop type (according to area harvested data for those crop types in counties for which the summary soil reports did not report soil test results for that crop type) and used to calculate total area and area with excessive soil P-I levels for each crop type.

The intensity of fertilizer use was analyzed by comparing crop area harvested and fertilizer shipments using data from the N.C. Department of Agriculture's Agricultural Statistics yearbooks for the period 1980-1998 (Figure 10; N.C. D.A. 1981–1998). North Carolina's legislature abolished the requirement to report fertilizer shipments annually by county after 1997, so data for the eastern 39 counties were unavailable after that year.

Individual numerical soil test data from swine production operations for the period 1998-2001 were obtained using the N.C. Division of Water Quality's lists of permitted operations and the Agronomy Division's public web site. Soil P-I data were obtained from 1,670 soil reports with 14,125 individual samples. Approximately 500 of these reports and 4100 of the individual samples were from swine production operations owned by corporate integrators and the remainder from individual swine production operations contracting with an integrator.

The effects of residential and recreational land uses on the distribution of excessive soil P levels were assessed by estimating the frequency of excessive soil P levels for the 'home grounds' and 'turf' crop type designations in the N.C. Agronomy Division's database. 'Home grounds' included lawns, gardens, and other residential land uses. 'Turf' corresponded primarily to golf courses and turf farms. Owing to lack of these data for many counties in the period FY80-FY81, only data for the periods FY90-FY91 and FY00-FY01 were plotted for the eastern counties.

Results

Agricultural land uses dominated in eastern North Carolina, with harvested cropland averaging approximately 1.3 million ha between 1980 and 2000 (Figure 1). Total 'land in farms' was 34% of total land area of the 39 eastern North Carolina counties in 1997 (N.C.D.A. 1999). Forests, wetlands, and other undeveloped lands accounted for over 50% of the land area in eastern North Carolina, while 'urban' land cover was less than 5% of land area and golf courses accounted for much less than 1% of total land area (N.C. D.W.Q. 1996, 1997a, b, c, 1999a, b, c, 2000). Although silviculture is important in eastern North Carolina, there were very few soil test results reported for 'forestry' crop types and those values were very low.

Distribution of excessive soil P-I values through time

The N.C. Agronomy Division analyzed over 226,000 soil samples from 39 eastern counties in FY80+FY81 (Table 1). Sample numbers rose approximately 30% a decade later and then another 50% to approximately 450,000 by FY00+FY01. Larger counties and counties with extensive agriculture operations, e.g., Sampson, generated more samples than smaller counties and counties with little agriculture, e.g., Dare.

The frequency of excessive soil P-I values for all land use types was plotted by county for each of the three time periods assessed (Figure 2). The frequency of excessive soil P-I values generally declined (in 34 of 39 counties) between FY80-FY81 and FY90-FY91, but generally increased (in 35 of 39 counties) by FY00-FY01, with considerable variation among counties (Table 2). The highest frequencies of excessive soil P-I values ($>50\%$ of all samples) were consistently found in the central portion of eastern North Carolina at each time interval (Figure 2). The counties bordering estuarine or oceanic waters generally had lower frequencies of excessive soil P-I values (mean = 35% in FY00-FY01). One notable pattern was the declining frequency of excessive soil P-I values between FY80-FY81 and FY90-FY91 in the central region, followed by an increase during the interval between FY90-FY91 and FY00-FY01 in many of those counties, particularly Duplin, Greene, Lenoir, and Sampson counties (Figure 2).

Analysis of the percentages of samples with excessive soil P-I by land use type during the three time intervals revealed several patterns (Table 3). There was a general decrease in the percentages of excessive soil P-I values between FY80-FY81 and FY90-FY91 with an increase by FY00-FY01, except that values for 'pasture' increased and values for 'home grounds' declined steadily. Most ($>80\%$) excessive soil P-I values were associated with row crops, which may partially reflect greater use of soil testing

Table 1. Numbers of soil sample data reported by N.C. Agronomy Division in FY80-FY81, FY90-FY91, and FY00-FY01 for 39 eastern N.C. counties. County names in bold denote counties directly adjoining estuarine or ocean waters (Figure 2).

County	FY80+FY81 n	FY90+FY91 n	FY00+FY01 n
Beaufort	8501	10336	23248
Bertie	7559	7921	16057
Bladen	4498	3456	6001
Brunswick	3123	2860	4120
Camden	2975	5891	9784
Carteret	2342	4933	4607
Chowan	4472	6730	9690
Columbus	4394	3943	4603
Craven	5092	7370	12852
Cumberland	5888	8558	7991
Currituck	2294	2733	4395
Dare	370	659	644
Duplin	5968	7508	13793
Edgecombe	9844	12741	22355
Gates	2939	7757	11849
Greene	6068	9184	11855
Halifax	13353	13215	14819
Harnett	2501	2799	4220
Hertford	5432	7010	9871
Hyde	3729	5986	12000
Johnston	9093	10606	16710
Jones	4427	5799	8250
Lenoir	7130	8555	13539
Martin	8811	7694	13880
Nash	5955	11292	17032
New Hanover	2712	5033	5089
Northampton	12964	15244	15246
Onslow	4355	5070	6465
Pamlico	4505	6212	7517
Pasquotank	5154	6046	9331
Pender	2773	3290	3665
Perquimans	3843	5060	9957
Pitt	11026	11104	20213
Robeson	7774	8450	9397
Sampson	11769	16414	25416
Tyrrell	1654	5964	7465
Washington	4426	6831	8540
Wayne	9429	15348	24161
Wilson	6996	10874	23109
Totals	226138	296476	449736

Table 2. Weighted mean and range of values for frequency of excessive soil P-I levels in eastern North Carolina for two-year periods, FY80-FY81, FY90-FY91, and FY00-FY01.

Period	Mean %	Low value		High value	
	P-I > 100	%	County	%	County
FY80-FY81	53.2	11.0	Hyde	76.4	Greene
FY90-FY91	40.3	5.4	Hyde	67.6	Greene
FY00-FY01	49.0	10.5	Carteret	75.4	Lenoir

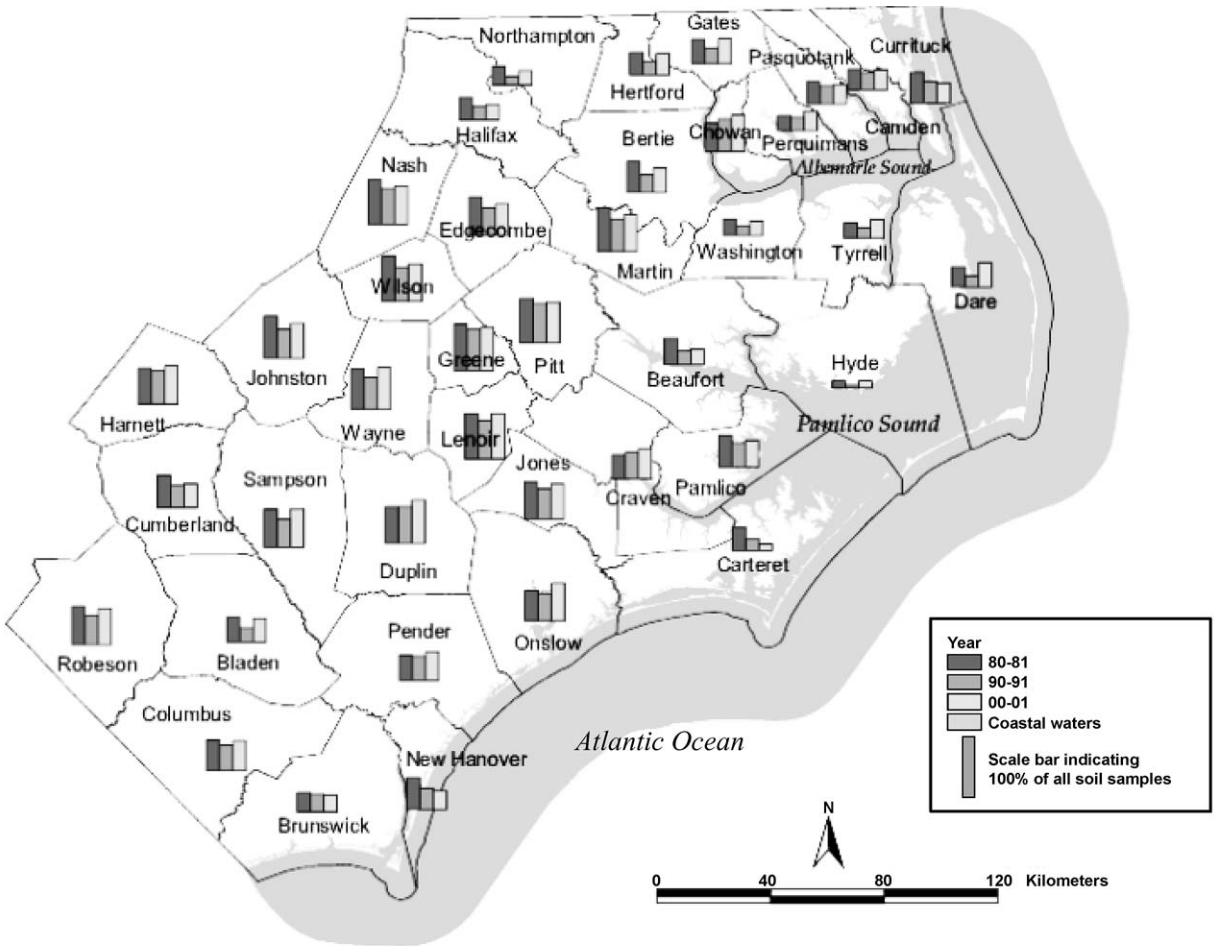

Figure 2. Map showing summary data for frequency of excessive soil P-I levels for the periods FY80-FY81, FY90-FY91, and FY00-FY01 for the 39 counties of eastern North Carolina for which soil P-I data and other information were obtained.

services by commercial farmers. A large proportion (>40%) of the samples with excessive soil P-I levels in FY00-FY01 was associated with cotton production, although the area of harvested cotton was only about 25% of total harvested area (Table 3; N.C. D.A. 2001). This observation suggested that the numbers of soil samples per crop ha harvested varied considerably among crop types.

Patterns of excessive soil P-I values by crop type

The percentage of row crop area with excessive soil P-I values was highest in central eastern North Carolina during each time period (Figure 3), with patterns very similar to those in the summary data set (Figure 2). The counties adjoining estuarine or oceanic waters had generally low frequencies of row crop area

Table 3. Total number (1,000) of soil samples reported and percent of samples with excessive soil P-I levels by land use type in eastern North Carolina for two-year periods, FY80-FY81, FY90-FY91, and FY00-FY01.

	FY80-FY81		FY90-FY91		FY00-FY01	
Land use type	Number	%	Number	%	Number	%
Corn	73.8	49.0	75.5	30.7	71.0	35.0
Cotton	2.5	53.4	46.1	40.4	182.6	50.5
Home grounds	16.9	58.4	27.8	44.5	28.9	43.1
Pasture	0.3	12.8	4.3	37.6	18.2	63.4
Peanuts	23.2	49.9	22.2	31.9	15.9	44.8
Potatoes	1.1	80.3	3.7	47.4	6.0	50.5
Small grains	16.3	46.1	19.6	36.8	32.5	48.7
Soybeans	39.5	43.8	42.3	36.6	48.7	48.2
Sweet potatoes	2.2	77.2	7.7	61.1	6.9	70.4
Tobacco	19.2	86.7	20.1	70.4	13.0	75.7
Turf	0.6	22.7	4.3	34.7	4.1	40.8

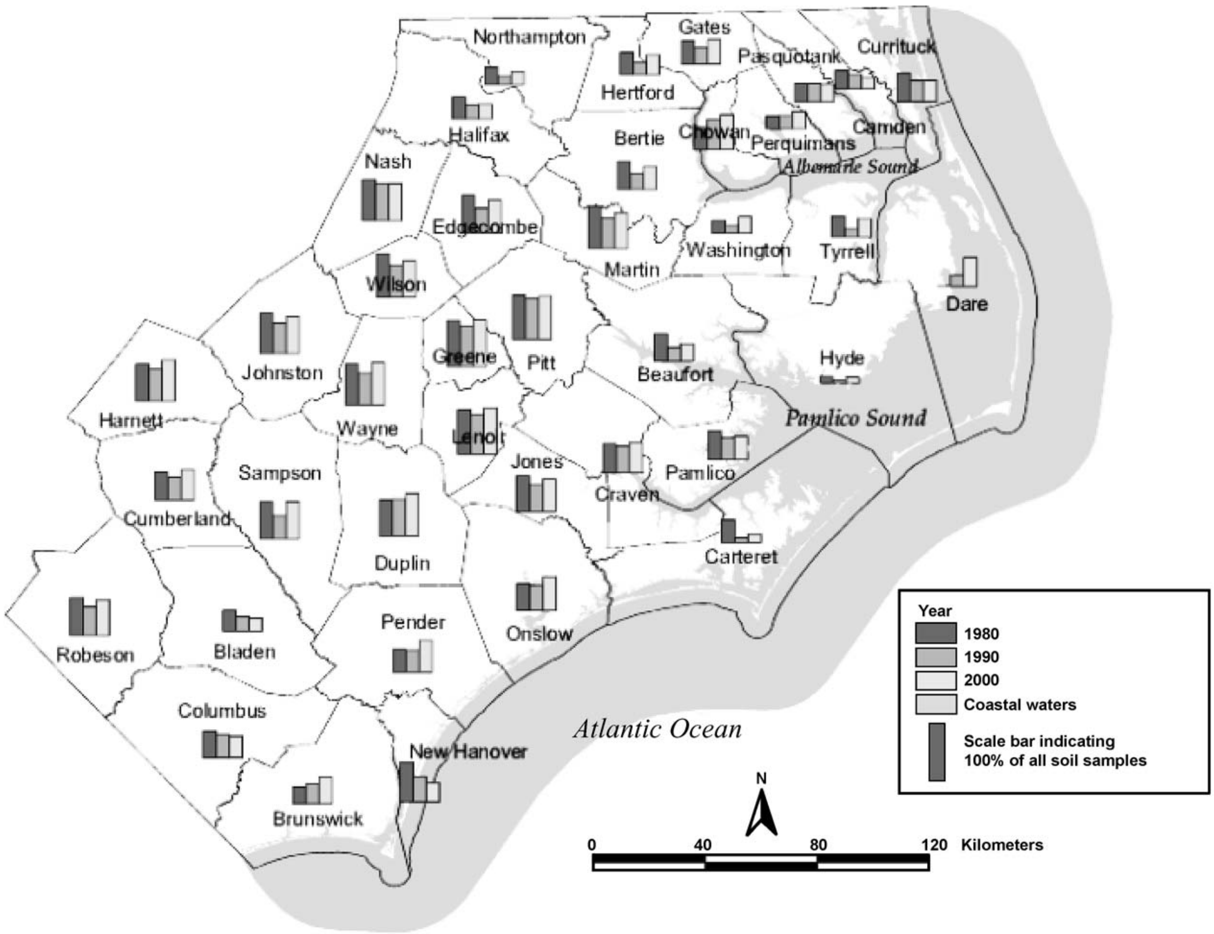

Figure 3. Percentages of row crop area in 1980, 1990, and 2000 with excessive soil P-I levels in FY 1980, FY 1990, and FY00, respectively, for each of 39 counties in eastern North Carolina.

with excessive soil P-I values, but sometimes, particularly in 1990, had little or no row crop area

Table 4. Crop area harvested (1,000 ha) and percent with excessive soil P-I levels for major crop types in eastern North Carolina for 1980, 1990, and 2000.

	1980		1990		2000	
Crop type	ha	%	ha	%	ha	%
Corn	559	51.7	371	34.3	220	41.4
Cotton	17	54.4	73	36.5	341	49.8
Pasture	17	52.7	16	38.2	52	61.8
Peanuts	67	53.3	66	36.5	50	44.7
Potatoes	5	71.8	7	51.4	7	64.1
Small grains	65	55.0	155	40.5	169	44.4
Soybeans	559	46.0	420	38.3	445	49.0
Sweet potatoes	14	73.3	13	64.6	14	74.9
Tobacco	92	84.4	67	71.6	44	78.5

harvested. The temporal patterns of row crop area with excessive soil P-I values were also very similar to those of the summary data set, with declines in 35 of 39 counties between 1980 and 1990 and increases in 34 of 39 counties between 1990 and 2000.

The changing frequencies and crop-specific patterns of excessive soil P-I levels associated with row crop agriculture during 1980, 1990 and 2000 were further examined by calculating the percent areas of each row crop type with excessive soil P-I levels in each corresponding FY (Table 4). The total area of row crops with excessive soil P-I levels in eastern North Carolina decreased about 30% during the 1980s from approximately 0.7 million ha, and then increased in the 1990s from about 0.48 million to over 0.64 million ha. The percentages of each row crop type with excessive soil P-I levels generally decreased between 1980 and 1990, but increased again by 2000. The crop types associated with excessive soil P-I levels changed significantly in their relative contributions to the total area figures in the intervals FY80 to FY90 to FY00. The largest change was the increase in cotton crop area harvested with excessive soil P-I levels (+279% between FY80 and FY 90, +635% between FY90 and FY00, to over 160,000 ha in FY 00), accounting for almost all the total increase. A smaller, but similarly impressive relative increase in crop area with excessive soil P-I occurred for pasture (− 30% between FY80 and FY90, but +523% between FY90 and FY00, to over 30,000 ha in FY00), which included various forage grasses planted on animal manure application fields. Soybeans were also associated with large areas with excessive soil P-I levels (at least 160,000 ha each year). On the other hand, the area with excessive soil P-I levels declined markedly over the interval FY80 to FY00 for peanuts, tobacco, and corn (from just over 0.4 million ha for the three crops in FY80 to just over 0.2 million ha in FY90 to approximately 0.14 million ha in FY00), with modest changes for other crop types.

Shifts in the crop types and acreages harvested associated with excessive soil P-I levels paralleled changes in total acreage harvested for those crops in eastern North Carolina during the intervals 1980 to 1990 and 1990 to 2000 (Table 4). Total crop area harvested declined from about 1.4 million ha in 1980 to 1.19 million ha in 1990, then increased to about 1.34 million ha in 2000. Area harvested for corn, tobacco, and peanuts declined significantly over this period, with a smaller decline in area harvested for soybeans. Area harvested for small grains increased somewhat. The largest increases in area harvested during the period 1980–2000 were for hay (equivalent to ‘pasture’ in soil reports; up over 300%) and cotton (up almost 20-fold).

Crop area harvested and fertilizer shipments declined generally during the 1980s and rose beginning in the early 1990s (Figure 4). These data showed an average ratio of fertilizer shipped per FY to ha harvested per year, e.g., FY82 and 1982, of 0.13 mt total fertilizer/harvested ha during the 14 years, 1982–1995. There was an increase in fertilizer shipments from almost 1 million mt per year as late as FY 1995 to over 1.8 million mt per year by FY97, yielding an average ratio of mt fertilizer shipped per harvested ha of 0.24 for 1996–1997. Fertilizer shipments to 35 of 39 eastern N.C. counties increased between FY95 and FY97 (N.C. D.A. 1996, 1998)

Excessive soil P-I values and CAFO distribution

Livestock inventories increased dramatically during the 1980s and 1990s, with the most dramatic increase driven by the growth of the swine industry in the 1990s, stabilizing at an inventory of over 8 million head in the 39 eastern counties of North Carolina when a legislative moratorium on new swine production facilities was enacted in 1997 (Mallin and Cahoon 2003). When expressed as ‘animal units’ (1 beef cow = 2.5 swine = 30 broiler chickens = 55 turkeys), total animal units in the eastern 39 counties of North Carolina increased rapidly during the 1990s, until the 1997 swine moratorium (Figure 5; N.C. D.A. 1993–2001). Data on turkey production, a significant fraction of animal unit totals, were reported only from 1992 onward. During the period 1992–2001 swine

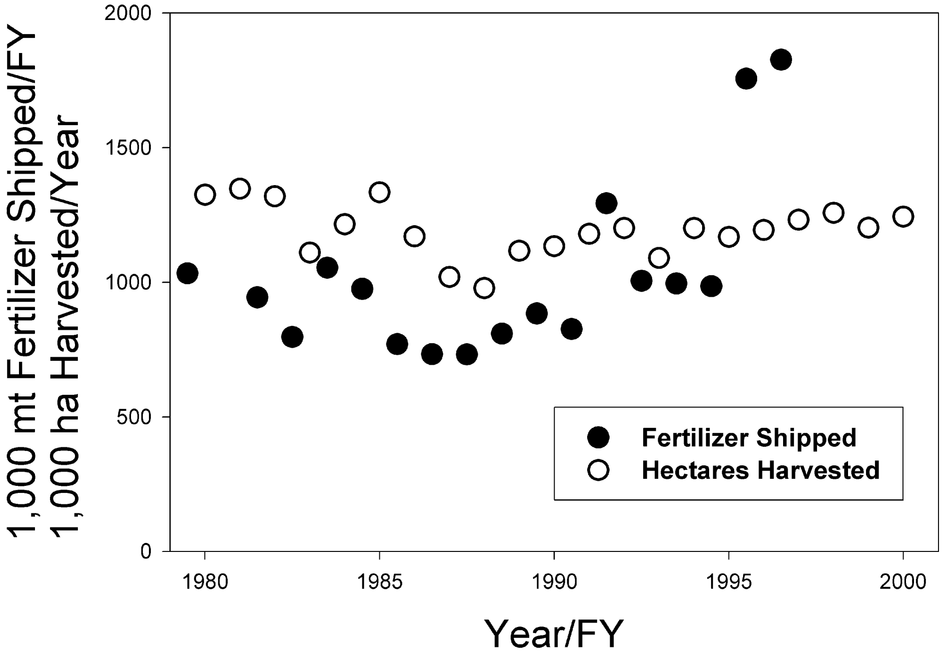

Figure 4. Fertilizer shipments (by FY) and crop area harvested (by calendar year) in eastern North Carolina, 1980–2000.

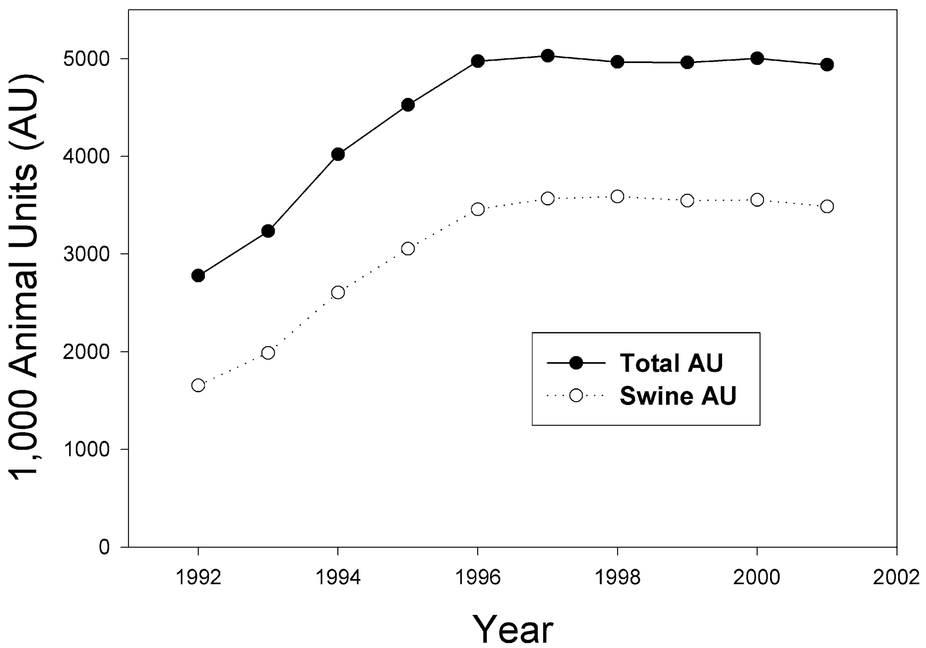

Figure 5. Changes in total animal units (beef cattle + swine + turkeys + broiler chickens) and swine animal units in eastern North Carolina, 1992–2001.

inventory increased from 60% of total animal units in the eastern 39 counties (1,655,000 AU in 1992) to approximately 71% of the total (3,490,000 AU in 2001). Broiler chickens and turkeys accounted for most of the remainder, with beef cattle accounting for only 76,000–92,000 animal units during the period.

Mean soil P-I values from swine CAFOs were 167 (400 kg P ha^{-1}) for all operations, 153 (368 kg P ha^{-1})

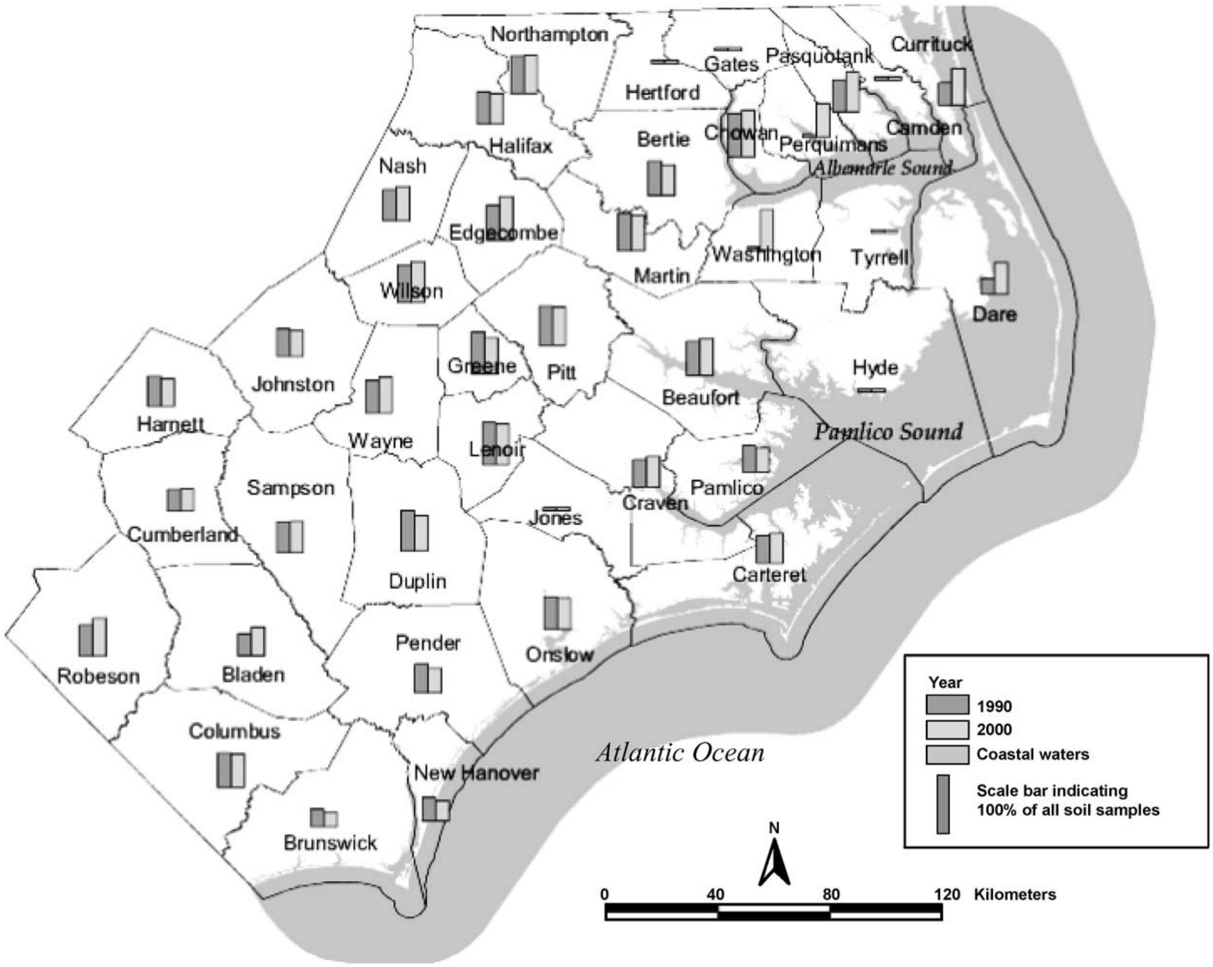

Figure 6. Percentage of soil reports from 'home grounds' and 'turf' with excessive soil P-I levels for each county in eastern North Carolina for FY90-FY91 and FY00-FY01.

for integrator-owned operations, and 172 (413 kg P ha^{-1}) for contracting operations. Liquid swine waste is generally land-applied to various cover crops, particularly coastal Bermudagrass, which has a high nitrogen absorption capacity (N.C. Coop. Ext. Serv. 1997). Data from 1,549 individual samples from coastal Bermudagrass fields associated with swine production operations yielded an overall mean soil P-I value of 188 (451 kg P ha^{-1}), 208 (500 kg P ha^{-1}) for integrator-owned operations, and 170 (408 kg P ha^{-1}) for contracting operations. Bermudagrass is included in 'hay' and 'pasture' categories for data on crop acres harvested and soil P-I values, respectively.

Excessive soil P-I values and non-agricultural land uses

The highest frequencies of excessive soil P-I associated with residential and recreational land use occurred in the central coastal plain and some northeastern counties, with no major shifts during the 1990s (Figure 6). Counties adjoining estuarine and oceanic waters had generally lower frequencies of excessive soil P-I levels, although some of these counties reported no data for each period. Mean frequencies of excessive soil P-I values associated with home grounds and turf were slightly higher than for the summary soil data (Table 2) and variation among counties was high (Table 5).

Table 5. Mean and range of values for frequency of excessive soil P-I levels associated with home grounds and turf in eastern North Carolina for two-year periods, FY90-FY91, and FY00-FY01.

Period	Mean %	Low value		High value	
	P-I > 100	%	County	%	County
FY90-FY91	48.3	21.7	Dare	68.1	Chowan
FY00-FY01	52.8	20.1	Brunswick	69.7	Edgecombe

Discussion

Excessive soil P-I levels occurred in 43% of almost a million soil samples reported from the periods assessed, spanning two decades, in eastern North Carolina. Excessive soil P-I levels occurred in large portions of soil sample results for many land use types across much of the region. These widely distributed and recently increasing levels of excessive soil phosphorus represent a significant potential threat to water quality in the rivers and estuaries of coastal North Carolina.

Agricultural activities (row crop production and intensive livestock production) were clearly associated with a significant portion of the excessive soil P-I levels in eastern North Carolina. The highest frequencies of excessive soil P-I levels were found in the heavily agricultural counties of central eastern North Carolina (Figures 1 and 2). Temporal changes in the frequency of excessive soil P-I levels paralleled changes in crop area harvested, fertilizer shipments, and growth of animal production.

Residential and recreational land uses (‘home grounds’ and ‘turf’) also contributed to the distribution of excessive soil P-I levels, with frequencies of excessive soil P-I levels that were similar to those reported for the summary data (Tables 2 and 3). Although it is difficult to compare these land uses to agricultural land uses, owing to lack of comparable areal data and likely differences in soil testing frequencies, several points suggest that residential and recreational land uses accounted for a generally smaller, if sometimes locally significant, portion of the excessive soil P-I levels in eastern North Carolina than agricultural land uses. As stated above, agricultural land uses dominated in eastern North Carolina, except in some of the counties along the ocean. The large and growing contribution of phosphorus-rich wastes from North Carolina’s massive animal production industries, particularly during the 1990s, must be considered an agricultural cause of excessive soil P-I levels in addition to row crop fertilization practices. The frequencies of excessive soil P-I associated with ‘home grounds’ and ‘turf’ in the coastal counties were generally lower than inland; the highest frequencies were found in the same central eastern North Carolina counties where the highest overall and row crop-associated frequencies of excessive soil P-I levels were found (Figures 2, 3 and 6). It should be noted that soil samples from ‘home grounds’ included samples from vegetable gardens and that soil samples from ‘turf’ likely included samples from turf farms, both forms of agricultural land use, and not strictly residential or recreational land uses. Finally, soil P-I data for New Hanover County, the most heavily urbanized (population: 146,000 in 1997; New Hanover County Planning Dept. 1997) and least agricultural (< 500 ha harvested in 1997, N.C. D.A. 1998) county in eastern North Carolina, had declining frequencies of excessive soil P-I levels throughout the period considered here: 52.7% in FY80-FY81, 35.0% in FY90-FY91, and 30.8% in FY00-FY01. Although residential application was the main use of fertilizer in New Hanover County (Cahoon 2002), excessive soil P-I levels in that urban, coastal county had a very different trend than the inland counties, suggesting that residential fertilizer applications were less likely to create excessive soil P-I levels than agricultural uses.

The increasing frequency of excessive soil P-I levels in eastern North Carolina between the period FY90-FY91 and FY00-FY01 was associated with several trends. Fertilizer shipments rose dramatically in the last two reporting years (FY96 and FY97), out of proportion with crop area harvested (Figure 4). Fertilizer shipments to eastern North Carolina during these two years accounted for 67% of total fertilizer shipments to the whole state (N.C. D.A. 1997, 1998). Total fertilizer shipped for the entire state rose again in FY98 (N.C. D.A. 1999), then declined to levels comparable to the late 1980s by FY00 (N.C. D.A. 2001). Declines in fertilizer shipments and crop area harvested in the 1980s may have allowed the declines in excessive soil P-I levels observed during the interval FY80-FY81 to FY90-FY91, so it is reasonable to assume that increases in these measures in the late 1990s would drive increases in soil P-I levels as well.

Although the actual nutrient contents of fertilizers were not reported by the N.C. Department of Agriculture, an approximate estimate of the phosphorus contents of fertilizers can be made. The fertilizer totals reported above included other materials, such as lime and land plaster, which left about 60% of the total as nutrient-containing fertilizer materials. If phosphorus as P_2O_5 was about 6% of nutrient-containing fertilizers, then phosphorus ($=0.436 \times P_2O_5$) would constitute approximately 1.56% of total fertilizer shipped, or approximately 28,500 metric tons of phosphorus shipped to eastern North Carolina in FY97. Similarly, the excess phosphorus present in soils in eastern North Carolina may be conservatively estimated as the difference between a P-I level of 50 (120 kg ha^{-1}) and 100 (240 kg ha^{-1}) for the 0.64 million hectares with excessive soil P-I levels in 2000, yielding 76,800 metric tons of phosphorus. Clearly, accumulation of this minimal estimate of excess phosphorus must have taken years of fertilization.

It is difficult to attribute the large increase in fertilizer shipments to eastern North Carolina in FY96 and FY 97 to any one crop or land use, partly because different crops may have been grown on the same fields at different times. However, the rapid increase in cotton production, area harvested for cotton, and excessive soil P-I levels associated with cotton production between 1990 and 2000 (Table 4) strongly suggested that heavy fertilization of increasingly widespread cotton crops drove up soil P-I levels widely or that cotton was rotated into already overloaded fields.

The contribution by land application of animal manures to increasing soil P-I levels must also be considered. Eastern North Carolina is a regional center of intensive livestock production, with a swine inventory ($>$8.7 million in 2001) alone exceeding the entire state's human population (~7.6 million in 2000), plus significant populations of broiler chickens, turkeys, and cattle. Much of the growth in these animal industries occurred during the decades of the 1980s and 1990s (Figure 5; Cahoon et al. 1999), so the contribution of animal wastes to soil phosphorus levels also increased proportionately. Mallin and Cahoon (2003) calculated that animal waste produced in the coastal plain of North Carolina (38 eastern counties) represented an annual contribution of 29,000 metric tons of phosphorus as of 2001. Cahoon et al. (2001) showed that in portions of eastern North Carolina where animal production was particularly intense, animal manures were the single most important source of nutrients, exceeding commercial fertilizers and human waste. Furthermore, most of the nutrients in animal manures were imported from out of state, thus representing 'new' nutrients (Cahoon et al. 1999), also likely true for much of the inorganic fertilizer. Soil P-I levels from swine production operations averaged well above 100, and even higher for coastal Bermudagrass fields, which are commonly used to receive land-applied liquid swine waste. Swine CAFOs owned by corporate integrators and by other private contractors exhibited similar patterns of excessive soil P-I levels, indicating that standard industry practices led to these conditions. Increases in crop area of 'hay' (which includes coastal Bermudagrass), excessive soil P-I levels from 'pasture', and crop area of 'pasture' with excessive soil P-I levels (Table 4) during the 1990s were almost certainly driven by increasing swine and poultry production (Figure 5) and land application of wastes, not to use of pasture for beef cattle production, which did not change significantly during that period. Examination of individual soil reports from swine production operations also showed that swine waste was applied to a variety of other crops, including cotton. Therefore, animal waste production, particularly by swine, has contributed significantly to increasing soil P-I levels in the region. This effect was most noticeable in central eastern North Carolina, where animal production was concentrated and where soil P-I levels rose most significantly in the 1990s (Figure 2). Moreover, the 'organic fertilizers' produced by animals did not significantly displace commercial fertilizer use, which increased substantially in the late 1990s (Figure 4).

The frequency of excessive soil P-I levels was highest and increased most significantly in the 1990s in central eastern North Carolina. The well documented relationship between soil phosphorus levels and the potential for phosphorus export (Sims et al. 1998; Turner and Haygarth 1999; Novak et al. 2000; McDowell et al. 2001a, b, c) therefore suggests that the rivers and streams of eastern North Carolina are at greater risk of eutrophication problems than previously. Stow et al. (2001) have shown relationships between agricultural and point sources and nutrient loads in the Neuse River in eastern North Carolina, although relationships vary within the basin. Mallin et al. (2001) have shown that phosphorus inputs to blackwater streams, a common feature of eastern

North Carolina, can drive significant increases in heterotrophic production and Biochemical Oxygen Demand (BOD). Aerial and ground observations show that many of these streams adjacent to agricultural areas have extensive coverage of a wide variety of aquatic macrophytes (Cahoon, personal observation). Mallin et al. (1999) have demonstrated springtime phosphorus stimulation of phytoplankton production in the Cape Fear River Estuary, which drains major animal production and row-crop agricultural areas. Phosphorus-driven eutrophication of North Carolina's coastal rivers during the 1980s led to a ban on residential use of phosphate-rich detergents in 1989, with significant benefits (N.C. D.E.M. 1991). Buildup of excessive soil P-I levels may reverse the progress made by that measure. However, the greatest accumulations of excessive soil P have occurred relatively recently upstream of the region's estuarine waters most vulnerable to eutrophication. The relatively slow transport of phosphorus through erosion, surface runoff, wind-blown dust, and groundwater flow will likely take years to exert their full impact.

North Carolina has recognized the potential threat to water quality posed by the buildup of soil P-I levels in recent years, and is developing a management approach to the problem (a 'Phosphorus Loss Assessment Tool' or PLAT) in response to federal directives (U.S. Natural Resource Conservation Service's Nutrient Management Standard 590, adopted in 1999). The PLAT considers soil P-I levels, soil type, slope, distance to water, land cover, rainfall, soil loss rates and other factors to estimate relative risk of phosphorus export and identify the most effective remedial measures, similar to the P indexing system described by Sharpley et al. (1994). However, the PLAT has not (as of this writing) been formally adopted, may need further revision, and will only be voluntarily implemented when it is finalized and adopted. It remains to be seen if this approach will mitigate the threat to water quality posed by eastern North Carolina's widespread and increasing frequencies of excessive soil P-I levels.

Conclusions

1. Excessive levels of soil phosphorus are widespread across eastern North Carolina. Much of the excess soil phosphorus is associated with row crop agriculture, but this may be partly an artifact of sampling effort.
2. Both excessive use of commercial inorganic fertilizers and widespread land application of animal manures contributed to increases in excessive soil phosphorus levels in the 1990s. Fertilizer shipments increased even as row crop acres declined and animal waste production increased in the 1990s.
3. Increasing levels of soil phosphorus and the widespread distribution of soils with excessive soil phosphorus pose a potentially significant threat to water quality in this coastal region.

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Integration of swine manure anaerobic digestion and digestate nutrients removal/recovery under a circular economy concept

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ABSTRACT

The application of the circular economy concept should utilize the cycles of nature to preserve materials, energy and nutrients for economic use. A full-scale pig farm plant was developed and validated, showing how it is possible to integrate a circular economy concept into a wastewater treatment system capable of recovering energy, nutrients and enabling water reuse. A low-cost swine wastewater treatment system consisting of several treatment modules such as solid-liquid separation, anaerobic digestion, biological nitrogen removal by nitrification/denitrification and physicochemical phosphorus removal and recovery was able to generate 1880.6 ± 1858.5 kWh d^{-1} of energy, remove 98.6% of nitrogen and 89.7% of phosphorus present in the swine manure. In addition, it was possible to produce enough fertilizer to fertilize 350 ha per year, considering phosphorus and potassium. In addition, the effluent after the chemical phosphorus removal can be safely used in farm cleaning processes or disposed of in water bodies. Thus, the proposed process has proven to be an environmentally superior swine waste management technology, with a positive impact on water quality and ensuring environmental sustainability in intensive swine production.

1. Introduction

Pork meat has been reported as the most widely consumed meat worldwide, with an increase of 20% from 2008 to 2018, producing more than 4 millions of tons in 2018 (FAOSTAT, 2018; Sporchia et al., 2021). To supply these increasing demands for animal protein, the Confined Animal Feeding Operations (CAFOs) model have been applied, also resulting in the generation of high volume of waste in small geographical areas. Generally, wastewater from swine production systems is stored and stabilized in anaerobic lagoons before application as biofertilizer on cropland. In cases of geographical areas with excess swine wastewater application to the soil, due to high organic matter, nitrogen and phosphorus surplus, environmental impacts must be considered, causing water, soil and air pollution besides unpleasant odors, water eutrophication and nutrient accumulation in soil (Ferreira et al., 2018; Kunz et al., 2009a, 2009b; López-Pacheco et al., 2021).

Due to the high pollutant potential of swine manure, the implementation of better management systems on farms must be considered and often swine wastewater treatment is an alternative when the soil support capacity is exceeded. As an important component of a treatment process, anaerobic digestion (AD) can be applied as a cost-effective solution for animal manure treatment, removing the biodegradable organic matter present in the effluent (López-Pacheco et al., 2021; Tan et al., 2021). Furthermore, the major benefit of AD is the biogas generation that can be used as a renewable fuel (*e.g.,* thermal or electrical applications). However, such treatment is not able to solve the problems with nutrients (nitrogen and phosphorus), pathogens and heavy metals present in the digestate. Therefore, treatment technologies are necessary to achieve the effluent environmental standards (González-García et al., 2021; Vanotti et al., 2018; Zubair et al., 2020).

Beyond the main biological processes applied to nitrogen removal, the process based on nitrification/denitrification could be used when a

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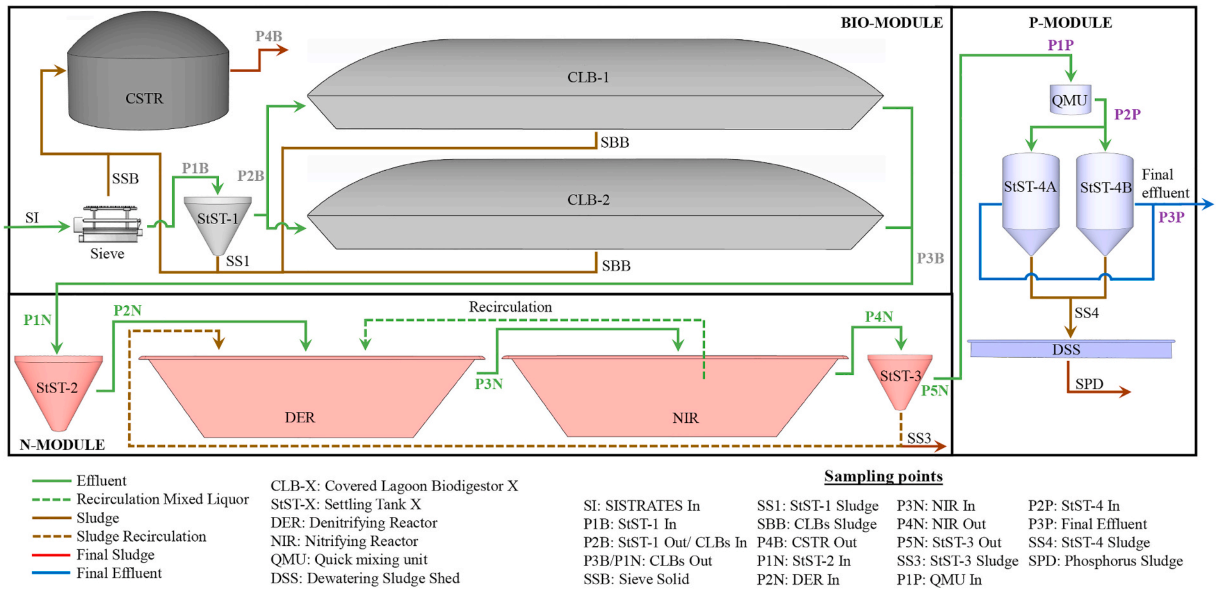

Fig. 1. The configuration of SISTRATES® system. *Bio-module*: composed of the receiving unit, sieving, settling tank-1, CLB (liquid fraction anaerobic digestion) and CSTR biodigester (treatment of sludges fraction). *N-module*: settling tank-2, followed by a denitrification and nitrification tank, and settling tank-3. *P-module*: composed of a quick mixing unit and two settling tanks (4a and 4 b) and sludge drying unit.

pre-treatment for carbon abatement as AD is applied (Bonassa et al., 2021). Through this process the ammoniacal nitrogen is oxidized to nitrate by nitrification and subsequently reduced the nitrite and gaseous nitrogen by denitrification, using an organic substrate as electrons acceptor (He et al., 2018; Liu et al., 2018). It has been proved that denitrification/nitrification process could be applied to treat wastewater with high loads of organic carbon and ammoniacal nitrogen with high removal efficiency (Hollas et al., 2019). The configuration of the process is an important point and the Modified Ludzack-Ettinger System (MLE) meets the necessities of swine manure, that has high concentrations of carbon, enabling its application to wastewater with high solids concentrations, and demonstrating yours robustness to withstand variations in operational feeding rates, but is not efficient in terms of phosphorus removal and recovery (Bortoli et al., 2019; Hollas et al., 2019).

For phosphorus removal and recovery, several treatment methodologies can be applied, as physical separation, biological treatment, adsorption and chemical precipitation (Fernandes et al., 2012; Li et al., 2019; Magrí et al., 2020; Peng et al., 2018; Sperlich et al., 2010; Suzin et al., 2018; Wang et al., 2018). The physic-chemical process for P removal that consists in precipitation using calcium hydroxide ($Ca(OH)_2$) is a great alternative for swine effluent, once hydrated lime is cheaper than the other reagents. It consists in the addition of a $Ca(OH)_2$ suspension up to pH 9,0, followed by settling after precipitation process to achieve a removal percentage of $P_{soluble}$ and P_{total} at least of 96% (Fernandes et al., 2012; Suzin et al., 2018). To ensure satisfactory performance of this process, a pre-treatment for removal of ammonia, organic carbon, and alkalinity is recommended (*e.g.,* AD and nitrogen removal processes), because these parameters have a significant influence on P removal efficiency and purity of the P recovered. High carbon concentrations will cause the co-precipitation of organic matter generating a impure sludge, that can infeasible the process (Peng et al., 2018; Suzin et al., 2018; Szogi and Vanotti, 2009). The inactivation of pathogens is another favorable point in this treatment technology, which makes it applicable following the precepts of circular economy for water reuse (Viancelli et al., 2015).

Considering wastewater management and treatment processes, a big challenge is to connect different processes keeping efficiency and reducing costs. In this way, inside the manure management system, the resources and wastes from the treatment plant can be converted into value-added products based on the cascading principle. For example, the swine manure treatment system could consist of: (i) solid-liquid separation (SLS) by sieving and settling (Amaral et al., 2016); followed by (ii) AD at different reactor configurations (Lins et al., 2020); (iii) digestate treatment for ammonia removal (Bortoli et al., 2019; Hollas et al., 2019); and (iv) phosphorus recovery (Fernandes et al., 2012; Suzin et al., 2018; Viancelli et al., 2015). Following the above principle of cascading idea, the integrated management system would provide: generation of biofertilizer (steps i. ii and iii); bioenergy and biogas (ii); second generation phosphorus (iv); and water for reuse (iii and iv).

In this sense, the SISTRATES® (Kunz et al., 2021) (a Portuguese acronym that means Swine Wastewater Treatment System) is an on-farm technology composed by: (i) SLS; (ii) AD in continuous stirred tank reactor (CSTR) and covered lagoon biodigesters (CLBs); nitrogen removal by nitrification and denitrification; and phosphorus recovery by chemical precipitation, designed to efficiently integrate these technological routes and maximize the environmental benefits of the treatment system. Thus, the aim of this study was to evaluate a full-scale swine manure treatment system where it was possible to perform the removal efficiencies, mass and energy balances, characterization, and evaluation of each step. The correlations between each unitary process were also studied to process control and to establish the technical coefficients. All component systems were evaluated during cold and warm periods.

2. Materials and methods

2.1. Site location

The SISTRATES® was implemented on a farm located in Videira, Santa Catarina - Brazil (27°02′38.8″S 51°05′35.7″W), in a farrow-to-wean piglet producing unit with 6655 ± 201 sows, with average daily flow rate of $192 \pm 45\ m^3\ d^{-1}$ of swine manure. Water consumption and swine manure generation were monitored during two years for conversion calculations.

2.2. Wastewater treatment system description

The treatment system - SISTRATES® aims to treat the swine wastewater in a complete way and therefore it is divided in tree modules. Bio-Module, to reduce total solids (TS) and total organic carbon (TOC) and recovery of biogas, N-Module, to reduce nitrogen and at last the P-

Module to reduce phosphorous and release a high-quality treated effluent. The system configuration, as well as the sampling points (which will be described in the next items), are presented in Fig. 1.

This research was conducted during the period of January 2019 until December 2020. or statistical analysis the periods were subdivided in **Period I** (January, February, March, April, May 2019), **Period II** (June, July, August, September 2019), **Period III** (October, November, December 2019 and January, February, March, April, May 2020), **Period IV** (June, July, August, September 2020) and **Period V** (October, November, December 2020).

2.2.1. Bio-Module: anaerobic digestion setup and operation

This module consists in a SLS process followed by AD biological treatment. The Bio Module was fed in a semi-continuous regime at a daily flow rate of 191.10 ± 44.98 m^3 d^{-1}. The SLS process is composed of a brush-roller screen (2 mm mesh), followed by settling tank 1 (StST-1,40 m^3). The liquid fraction flow rate is split and sent to two CLBs (2500 m^3 each one) without heating or mixing systems. The solid/sludge fractions (0.005 Qin_{screen} + 0.07 Qin_{StST-1} + 0.02 Qin_{CLBs}) were sent to a CSTR (700 m^3) operated under mechanical mixing and mesophilic conditions.

The biogas generated in the process (CLBs and CSTR) was measured using thermal mass flowmeters (Contech, model FT2, Brazil). The biogas composition was also simultaneously checked for methane (CH_4), carbon dioxide (CO_2), oxygen gas (O_2) and hydrogen sulfide (H_2S) (Awite, model Awiflex, Germany). Biogas H_2S concentration was reduced by microaeration (Awite, model Awidesulf 300/500) previously to a combined Heat and Power (CHP) unit.

For the AD treatment system, composite liquid samples were collected, from eight points: (SI) SISTRATES® in; (P1B) input of StST-1; (P2B) input from CLBs (supernatant from StST-1); (P3B/P1N) digestate from CLBs; (SS1) settled sludge (sludge of StST-1); (SBB) sludge from CLBs; (SSB) solid retained in sieve; (P4B) digestate from CSTR.

Samples were collected monthly and submitted to analysis of, alkalinity, pH, VFA/TA ratio, TOC, total nitrogen (TN), total ammoniacal nitrogen (TAN), TS, Volatile Solids (VS) and Fixed Solids (FS) and total phosphorus (TP). Biodigesters temperature and biogas production were continuously measured.

2.2.2. N-module: nitrogen removal setup and operation

For the nitrogen removal process, a MLE system was implemented (Bortoli et al., 2019). The effluent from CLBs flows to settling tank 2 (StST-2,40 m^3), previous to the nitrification/denitrification process, aiming to reduce TOC and solids concentration specially during wintertime. The liquid fraction flows in series mode through an anoxic tank for denitrification (983 m^3) an aerated reactor for nitrification (1272 m^3). Another settling tank, the settling tank 3 (StST-3,30 m^3), was installed after the nitrification tank, to separate and discharge biological sludge (Hollas et al., 2019). To start-up the nitrification process, the reactor was previously filled with water and inoculated wit 30 m^3 of nitrifying sludge acclimated to achieve high activity prior to starting the complete system ($0.01351 g_N\ g^{-1}{}_{VSS}\ h^{-1}$) (Antes et al., 2020).

The aeration inn nitrified tank was supplied with an air grid system (B&F DIAS, model B&F Air Grid, Brazil) using tubular membrane diffusers. The dissolved oxygen (DO) was kept around 2.5 ± 0.5 $mgO_2\ L^{-1}$ and monitored online (Hach, model SC200, USA). The N-module feeding flow rate (Q_{in}) consisted of a fraction of the discharge effluent from the CLBs (which did or did not pass through the StST-2 (as it will be discussed afterwards), at an average flow rate of 147.32 ± 19.50 m^3 d^{-1}. Nitrification reactor and StST-3 sludge (from nitrification) were constantly recirculated to denitrification reactor at a flow rate of 5.5 $Qin_{N\text{-}module}$ and 1.0Q_{in} $in_{N\text{-}module}$, respectively according to described by Bortoli et al., 2019. The exceeded sludge is discarded and sent to fertigation.

The MLE process was weekly monitored through analysis of samples collected from the following points: (P3B/P1N) in digestate from CLBs, (P2N) denitrifying reactor in, (P3N) denitrifying reactor out, (P4N) nitrifying reactor out, (P5N/P1P) StST-3 out and (SS3) sludge from StST-3. The samples were submitted to determination of alkalinity, TOC, TAN, nitrate ($NO_3{}^-$-N), nitrite ($NO_2{}^-$-N), Total Suspended Solids (TSS), Volatile Suspended Solids (VSS), Fixed Suspended Solids (FSS) and TP. The daily monitoring consists in determination of temperature, pH and DO that were measured in each reactor.

2.2.3. P module: phosphorus removal setup and operation

The effluent from N-Module flows to P-Module (138.3 ± 20.19 m^3 d^{-1}). Firstly, it fed a quick mixing unit (QMU, 0.186 m^2), were the effluent receives a suspension of $Ca(OH)_2$ 10% (m v^{-1}), until pH 9. The pH set point is determined using a pH controller (Digimed, model TH-48, Brazil) (Suzin et al., 2018).

After QMU the mixture flows by gravity to two parallel settling tanks (StST-4,35 m^3 each), with a hydraulic retention time (HRT) of 12.5 ± 2.9 h. The StST-4 liquid fraction is discharged to a water body and the P sludge, rich in calcium phosphate, is diverted for dewatering in polypropylene bags or directly sent to agricultural use.

The P-module effluent was also monitored weekly during a six-month period, from three points: (P5N/P1P) QMU in; (SD4) sludge out of StST-4 (4a and 4 b); (P3P) final effluent.

Samples were collected weekly and submitted to determination of alkalinity, TOC, TAN, $NO_2{}^-$-N, $NO_3{}^-$-N and TP (liquid fraction), to verify if the final effluent meets the quality requirement by Brazilian regulation (CONAMA 430, 2011) and TS, VS, FS, TP, TOC, potassium (K) and calcium (Ca) (solid fraction). Microbiological analyses (*Escherichia coli*, *Salmonella* spp. and *Porcine circovirus* type 2 - PCV2) were performed in the final effluent (P3P) in warm and cold periods corresponding to the months of August/2020 and January/2021, respectively.

2.3. Analytical methods

2.3.1. Physical-chemical parameters

TOC, total carbon (TC), TN, TAN (TAN = $NH_4{}^+$-N + NH_3–N, expressed as NH_3–N), $NO_3{}^-$-N, $NO_2{}^-$-N, alkalinity (expressed as $mgCaCO_3\ L^{-1}$), VFA/TA ratio, VSS, FSS and TSS, VS, SF and TS and TP, were determined according to Standard Methods for the Examination of Water and Wastewater (Rice et al., 2017). Analyses were performed in triplicate following standardized protocols based on good laboratory practices and quality assurance policy.

TOC and TN were determined with a TOC analyzer (Shimadzu, TOC-LCPH/CPN, Japan). TC was measured by CNHS elemental analyzer (Thermo Fisher Scientific, model Flash, 2000; USA); both following the manufacturer recommendations, NH_3–N (modified Berthelot's reaction colorimetric method), $NO_2{}^-$-N (colorimetric method at pH 2.0–2.5 using sulfanilamide with N-(1-naphthyl)ethylenediamine) and $NO_3{}^-$-N (same method of $NO_2{}^-$-N, but by its reduction in a cadmium column) were measured in a flow injection analysis system (Fialab Instruments, model 2500, USA). Alkalinity and VFA/TA ratio determination were performed in an automatic titrator (Metrohm, model 848 Titrino plus, Switzerland). TP was quantified by spectrophotometric molybdovanadate method using a spectrophotometer (Varian, Cary® 50 UV–Vis, USA). The samples were dried at 105 °C for the determination of TS and TSS, and calcined at 550 °C for VS VSS, FS and FSS determination. Samples were also analyzed for determination of Ca and K. The K was quantified by flame photometry Micronal, model B-462 Brazil), and Ca was quantified by flame Atomic Absorption Spectrometry (Varian, Spectr AA 220, USA), according to Standard Methods for the Examination of Water and Wastewater (Rice et al., 2017).

2.3.2. Microbiological parameters

E. coli was quantified using a Chromocult Coliform Agar (Merck, Germany) following manufacturer's instructions, and results expressed in the form of Colony Forming Units (CFU). *Salmonella* was quantified in deoxycholate-lysine-xylose agar (Merck, Germany), as described by

Michael et al. (2003), and also expressed in CFU. PCV2 determination was performed using VetMAX™ Porcine PCV2 Quant Kit (Applied Biosystems®, Foster City, USA), according to the manufacturer's protocol.

2.4. Technical coefficients

Based on the monitoring of the treatment system over a period of two years it was possible to determine several technical coefficients related to the treatment processes employed. The technical coefficients used for the system were calculated according to the Equations shown in Table 1S, in the supplementary material.

2.5. Statistical analysis and machine learning

Data management and analysis of variance (ANOVA) for data validation were performed with software STATISTICA 8.0. Analysis and predictions of the wastewater treatment system, based on the known past events, was done with the WEKA® Software. The algorithm used for the decision tree to perform the data mining task was J48, operated in the WEKA® computer program, based on the C4.5 tree training algorithm (Quinlan, 1993).

3. Results and discussion

3.1. Swine manure characterization

The daily manure flow rate that fed the SISTRATES® during the two years of monitoring was 192 ± 45 $m^3{}_{manure}\ d^{-1}$ and presented TS and VS concentrations of 15.63 ± 9.71 g L^{-1} and 10.70 ± 7.13 g L^{-1}, respectively with TC concentration of 6489 ± 4514 mg L^{-1}. The nutrient concentrations in the influent wastewater were for TN 1817 ± 708 $mg_N\ L^{-1}$, TAN 1221 ± 457 $mg_N\ L^{-1}$, TP 453 ± 368 mg L^{-1} and K 757 ± 338 mg L^{-1}.

The high variability of these parameters is justified due to the influence of seasonality. According to Cárdenas et al. (2021) the main factor that impacts the swine manure characteristics is the water consumption in productive system, which tends to be higher in the summer, affecting the concentration of manure. The averages of environmental parameters such as temperature and precipitation are presented in Table 2S, in addition to the volume of water consumed in the farms and swine manure generation, according to the warm and cold periods, where in general the highest water consumption was for the warm periods.

3.2. - SISTRATES® performance

The complexity of the swine wastewater points out that the appropriate way to manage these effluents is a decisive key for the success of the treatment. For this reason, SISTRATES® is divided in incremental modules that aim to reduce and stabilize distinct parameters and fractions of swine wastewater. This assumption will base the discussion afterwards.

3.2.1. Bio-Module

The first SISTRATES® module is the Bio-Module consisting in a SLS unit, followed by AD biodigesters (CLB and CSTR). The SLS unit liquid fraction fed the CLB, and the solids and sludge fraction were sent to a sidestream CSTR. This module enable the separation of solids fraction and reduction of organic matter beyond the biogas generation in the biodigesters, that is sent to a CHP unit for energy production.

3.2.1.1. SLS. The SLS process aims to prepare the swine wastewater fractions that will be directed to each type of reactor while this step begins in the brush-roller screen (2 mm mesh), which has the purpose of separating the coarse solids followed by a StST-1 (HRT 1.8 ± 0.7 h).

Table 1
CLB removal efficiency considering the seasonality effect in Bio-Module of SISTRATES® (n = 14).

Parameter	Seasonality				
	Period I	Period II	Period III	Period IV	Period V
VS removal (%)	76.5[a] ± 4.0	58.4[b] ± 1.7	81.0[a] ± 0.6	63.9[ab] ± 5.8	72.8[a] ± 2.6
FS out (%)	61.0 ± 4.0	71.3 ± 1.1	53.3 ± 5.5	72.2 ± 6.9	66.7 ± 7.3
TC removal (%)	74.6[a] ± 7.0	55.3[b] ± 7.0	70.9[a] ± 7.8	61.0[ab] ± 7.4	72.8[a] ± 2.5
TOC removal (%)	79.3[a] ± 4.8	55.9[b] ± 11.7	81.9[a] ± 5.4	67.2[ab] ± 6.9	79.3[a] ± 3.0
TN removal (%)	15.3[a] ± 8.9	3.9[a] ± 3.2	5.22[a] ± 3	12.3[a] ± 6	5.3[a] ± 2.8
P removal (%)	68.1[a] ± 3.3	58.4[a] ± 14.6	47[a] ± 18	62[a] ± 2	65.8[a] ± 6.3
K removal (%)	11.2[a] ± 6.4	8.2[a] ± 3.6	8.5[a] ± 1.4	7.3[a] ± 3.8	5.9[a] ± 5

Where: VS_{in}: 9.3 ± 1.2 g L^{-1}; FS_{in}: 3.9 ± 0.4 g L^{-1}; TC_{in}: 4963 ± 1157 mg kg^{-1}; TOC_{in}: 3818 ± 1296 mg L^{-1}; TP_{in}: 338.0 ± 97.8 mg L^{-1}; K_{in}: 665.4 ± 91.8 mg L^{-1}; NT_{in}: 1641 ± 453 mg kg^{-1}.
Evaluated seasonality: **Period I** (warm) - correspond to the months of January, February, March, April and May 2019; **Period II** (Cold) - correspond to the months of June, July, August and September 2019; **Period III** (warm) - correspond to the months of October, November, December 2019 and January, February, March, April and May 2020; **Period IV** (Cold) - correspond to the months of June, July, August and September 2020; and **Period V** (warm) - correspond to the months of October, November and December 2020.
*Means followed by the same lowercase letters on the lines of the same parameter do not differ significantly by the Tukey test ($p < 0.05$).

This step presented a TS removal of 7.4 ± 6.9% in the brush-roller screen, and 32.90 ± 18.9% in StST-1 (Tables 3S and 4S), which will ensure the efficiency of the next operational units to be performed in the treatment system. According to Amaral et al. (2016) the sieving step has an average efficiency in the reduction of VS that varies around 6–20.5%, depending on the type and age of effluent.

The concentrations of TC and TP showed removal efficiencies of 7.2 ± 6.7% and 3.8 ± 3.6%, respectively, in sieving, and 34.1 ± 22.3% for TC and 42.9 ± 33.45% for TP after StST-1. Nutrients such as TN and K, on the other hand, did not have such significant interference in the SLS process, due to the solubility of these elements, with average efficiencies of 1.6 ± 1.5% for TN, and 0.6 ± 0.5% for K, in the sieving; and of 12.7 ± 3.9% for TN and 15.1 ± 5.7% for K in the StST-1.

The success of this separation step is extremely important to obtain better results in AD, besides prolonging the life of the CLBs reactors because it avoids silting, since after SLS the solid and liquid fraction are conducted to distinct AD reactors, as described below.

3.2.1.2. CLB. The use of CLB to treat swine manure has intensified in Brazil in view of easy operation and low cost. CLBs receive the StST-1 supernatant, with an average concentration of VS at 11.0 ± 2.5 $g_{VS}\ L^{-1}$, and an organic loading rate (OLR) of 0.35 ± 0.16 $kg_{VS}\ m^{-3}d^{-1}$ (Tápparo et al., 2021). The values are according to this type of reactor, working better for liquid manure with less than 3% (w v^{-1}) of TS and low OLR (approximately 0.5 $kg_{VS}\ m^{-3}d^{-1}$) (Wu, 2013).

The average biogas productivity was 0.18 ± 0.05 $Nm^3{}_{biogas}\ m^{3}{}^{-1}{}_{reactor}\ d^{-1}$ and biogas yield was 0.56 ± 0.27 $Nm^3{}_{biogas}\ kg_{VSadd}{}^{-1}$ (Tápparo et al., 2021). The biogas productivity and yield variation in this reactor design is predictable, due to seasonal temperature fluctuation. A biogas productivity decreases of 30% was observed during winter and consequently the TOC and VS removal was reduced (Table 1). CLBs are left operate at ambient temperature, due to impracticality in temperature control, consequently biogas productivity and carbon removal decrease at low temperature was expected (Yu and Schanbacher, 2010). The decrease of biogas productivity was similar as observed by Schmidt et al. (2019) that indicated a 30% reduction in biogas production when

Table 2
Concentration of nutrients, total carbon and solid removal efficiency of CSTR in Bio-Module of SISTRATES® (n = 10).

Parameter	Average ± S.D.
TC removal (%)	55.9 ± 9.7
TS removal (%)	53.8 ± 9.2
VS removal (%)	64.4 ± 5.6
TP_{out} (mg L^{-1})	1295.4 ± 378.6
K_{out} (mg L^{-1})	807.2 ± 138.6
TN_{out} (mg L^{-1})	3183.4 ± 642.2

Where: TC_{in}: 21.9 ± 5.3; TS_{in}: 55.4 ± 18.7; VS_{in}: 50.7 ± 7.2; TP_{in}: 1935.9 ± 733.6; K_{in}: 709.0 ± 58.1; TN_{in}: 1653.0 ± 188.4.

the temperature decreased 7 °C in a CLB operated at lab scale. The CLB was operated in a way to have a residual TOC (1040 ± 482 mg L^{-1}), due to the denitrification step, where carbon is used to convert nitrate and nitrite to gaseous nitrogen.

The SLS unit proved to be an excellent strategy to increase the useful life of CLB, considering that only 3.94 ± 0.45 g_{FS} L^{-1} entered the reactor, with an output of 70 ± 9%, so we can assume that approximately only 30% was settled at the reactor bottom. Furthermore, this system has a daily discard of CLB sludge. Usually, there is no physical pre-treatment before CLB, then periodic cleaning is necessary due to FS accumulation (Yu and Schanbacher, 2010). The solid accumulation can decrease the useful volume of the reactor, HRT and consequently the biogas recovery and digestate quality (Cantrell et al., 2008).

In addition to process preservation due to the lifetime of the reactor, SLS enables an increase in gas production as reported by Amaral et al. (2016) who also used SLS as a strategy to increase biogas yield, observing an approximately 2-fold increase. As reported by Hollas et al. (2021), that studied the effect of SLS on biogas production and implemented the same separation techniques described here, an increase of 4.5 kWh of electricity for each m^3 could be obtained, proving the importance of the implemented configuration, corroborating in practical terms the useful life of the system as well as technically benefiting the increase in biogas production.

3.2.1.3. CSTR. As previously described, the CSTR feeding consists in a mixture of three solid fractions of SLS step, solid retained in a brush-roller screen, sludge from StST-1 and sludge from CLBs. The flow rate was 20 ± 5 m^3 with a VS concentration of 60.6 ± 10.8 mg L^{-1} and OLR of 1.69 ± 0.34 kg_{VSadd} m^3 $reactor^{-1}$ d^{-1} (Tápparo et al., 2021).

The CSTR, differently from CLBs, has a heating system that maintains the temperature control thus room temperature has no influence directly on the process performance, remaining between 35 ± 2 °C (because this the global results were presented). The biogas productivity was 0.65 ± 0.23 $Nm^3{}_{biogas}$ m^3 $^{-1}{}_{reactor}$ d^{-1} and biogas yield was 0.38 ± 0.14 $Nm^3{}_{biogas}$ $kg_{VSadd}{}^{-1}$ (Tápparo et al., 2021). With the combination of temperature control and the CSTR design, an increase pf biogas productivity over 10-fold was expected in comparison to CLB (Cantrell et al., 2008). Furthermore, due to high the process stability observed during all operation period, the OLR could be increased, using other substrates available on the farm, increasing biogas production and energy conversion.

The variability of nutrients on digestate was a consequence of manure variation, the inherent fluctuations of the production process (Amaral et al., 2016; Kunz et al., 2009b). The TP out in CSTR is until 5 times higher than CLB digestate, probably through P aggregation on the solid fraction (Chini et al., 2021; Deng et al., 2014). The concentration of the NPK elements and removals of TC and VS are presented in Table 2. Moreover, the nutrients are concentrated in the CSTR digestate, compared to swine manure, which increases the digestate value as fertilizer, reducing the costs with transportation and digestate spread (Deng et al., 2014).

The configuration adopted for the digestion of swine waste according to the solid matrix of the wastewater proved to be an efficient strategy for the management of swine waste maximizing energy recovery and technically ensuring the operation of the treatment plant. Environmentally, AD is favorable for reducing the impacts associated with the production of pigs, since it replaces the treatment lagoons, allowing the reduction of greenhouse gas emissions combined with energy recovery, which can be considered one of the biggest future challenges, especially for developing countries. Thus, the AD of animal waste can be one of the most viable strategies both in economic and environmental terms to promote changes in the global energy matrix (Vanegas Cantarero, 2020).

3.2.2. N-module

The digestate from the Bio-module has high concentration of TAN, due to swine manure organic nitrogen mineralization. Thus, digestate treatment strategies are necessary when the direct use as fertilizer is not possible. The N-Module is responsible for nitrogen removal from CLBs reactors using nitrification and denitrification based on the MLE process (Bortoli et al., 2019; Hollas et al., 2019).

The system took 56 days to achieve high TAN removal efficiency, which means that the final effluent achieved concentrations below 20 mg_N L^{-1} (Fig. 2). The concentration of TAN_{in} of the N-module had fluctuations over the period studied, with an average concentration of 1317 ± 222 mg_N L^{-1}. However, even with such variation, TAN at the

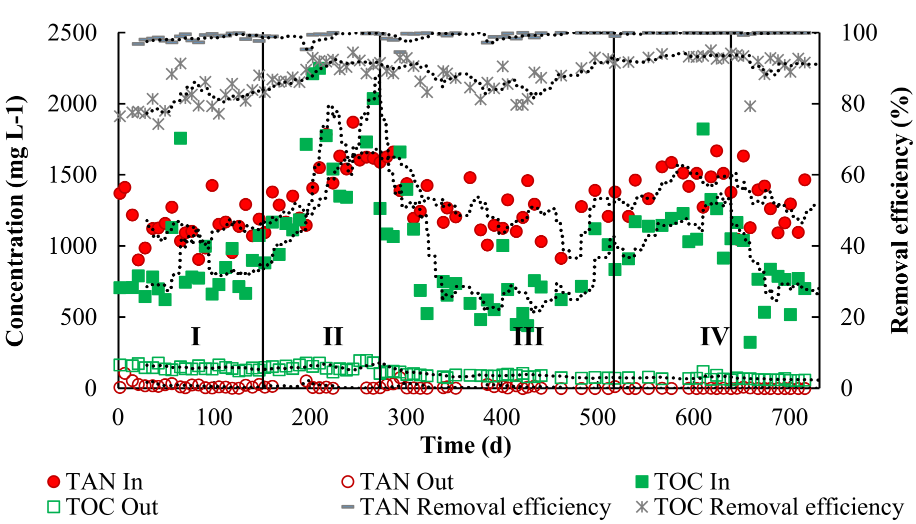

Fig. 2. Total Ammoniacal Nitrogen (TAN) and Total Organic Carbon (TOC) in and out with the respective removal efficiencies in N-Module. Evaluated seasonality: **Period I** (warm) - correspond to the months of January, February, March, April and May 2019; **Period II** (Cold) - correspond to the months of June, July, August and September 2019; **Period III** (warm) - correspond to the months of October, November, December 2019 and January, February, March, April and May 2020; **Period IV** (Cold) - correspond to the months of June, July, August and September 2020; and **Period V** (warm) - correspond to the months of October, November and December 2020. The dashed line corresponds to the moving average for a period of 30 days.

Table 3
N-Module global efficiencies considering the seasonality effect.

Parameters	Seasonality				
	Period I	Period II	Period III	Period IV	Period V
C/N	0.77[b] ± 0.27	1.08[a] ± 0.29	0.63[b] ± 0.21	0.80[b] ± 0.20	0.63[b] ± 0.21
Alkalinity consumption (%)	97.23[a] ± 0.87	97.68[a] ± 2.99	96.58[a] ± 3.37	92.17[b] ± 2.59	92.70[b] ± 3.38
TN removal (%)	73.43[d] ± 4.60	75.84[cd] ± 8.77	79.85[bc] ± 8.63	90.39[a] ± 4.81	85.33[ab] ± 4.08
TOC consumption (%)	81.58[d] ± 4.45	89.65[bc] ± 3.02	87.30[c] ± 4.04	94.66[a] ± 3.03	91.70[ab] ± 1.80
TOC loading rate (kg $m^{-3}d^{-1}$)	0.13[c] ± 0.04	0.23[a] ± 0.08	0.12[c] ± 0.05	0.18[ab] ± 0.05	0.13[bc] ± 0.05
TOC remove rate (kg $m^{-3}d^{-1}$)	0.10[c] ± 0.04	0.21[a] ± 0.08	0.11[c] ± 0.05	0.17[ab] ± 0.04	0.12[bc] ± 0.04
TAN loading rate (kg $m^{-3}d^{-1}$)	0.13[c] ± 0.01	0.16[ab] ± 0.03	0.15[bc] ± 0.03	0.18[a] ± 0.02	0.16[ab] ± 0.03
TAN remove rate (kg $m^{-3}d^{-1}$)	0.12[c] ± 0.01	0.16[ab] ± 0.03	0.14[bc] ± 0.03	0.18[a] ± 0.02	0.16[ab] ± 0.03
TAN daily load (kg d^{-1})	160.71[c] ± 26.26	211.21[ab] ± 39.38	185.23[bc] ± 36.55	230.71[a] ± 30.65	202.01[ab] ± 38.24

Note: Evaluated seasonality: **Period I** (Warm) - correspond to the months of January, February, March, April and May 2019; **Period II** (Cold) - correspond to the months of June, July, August and September 2019; **Period III** (Warm) - correspond to the months of October, November, December 2019 and January, February, March, April and May 2020; **Period IV** (Cold) - correspond to the months of June, July, August and September 2020; and **Period V** (Warm) - correspond to the months of October, November and December 2020. The dashed line corresponds to the moving average for a period of 30 days.
*Means followed by the same lowercase letters on the lines of the same parameter do not differ significantly by the Tukey test ($p < 0.05$).

final effluent and the process efficiency was maintained at levels that supply the standards into water bodies discharge based on Brazilian national regulation (CONAMA 430, 2011).

The average TAN removal efficiency obtained was 98.6 ± 2.1%, similar to the obtained by Vanotti et al. (2018), that verified 96.6% of TAN removal efficiency, with a treatment system composed by a SLS unit followed by nitrification/denitrification. For TOC, the variations were relatively high in the evaluated period making necessary to activate the StST-2 many times in wintertime (June, July and August) to reduce the concentration. The concentration of TOC in the affluent was 1039 ± 482 mg L^{-1}, with a removal efficiency of 88.0 ± 5.7%. The variations of TAN and TOC observed in the system input were due to the CLBs biomass temperature decrease during the cold periods reaching 20.2 ± 2.1 °C with an average of 25.6 ± 2.4 °C in the warm periods.

The TOC loading rates in the N-Module varied from 0.10 ± 0.04 kg m^{-3} d^{-1} to 0.23 ± 0.08 kg m^{-3} d^{-1} between seasonal periods, while TAN loading rates varied from 0.13 ± 0.01 kg m^{-3} d^{-1} to 0.18 ± 0.02 kg m^{-3} d^{-1} (Table 5S). The C/N ratio showed significant differences in the second period, corroborating with the discussion at topic 3.2.1 (Bio-Module), as well as that the variation in the applied TOC (Table 3).

As described in Tables 3 and 6S, it is possible to observe that all the parameters (except alkalinity consumption and TN removal) showed significant statistical differences between periods 1 and 2. This behavior may be linked to the fact that the first two monitoring periods are part of the initial period of N-Module, where the reactors showed instability behavior due to the variations in loads in the module, and adaptation of the microbial community. This can be observed if compared to periods 3, 4 and 5, that did not show significant difference, even in different seasons, which shows that the nitrification and denitrification processes were highly active and suffered less from the pressures exerted by

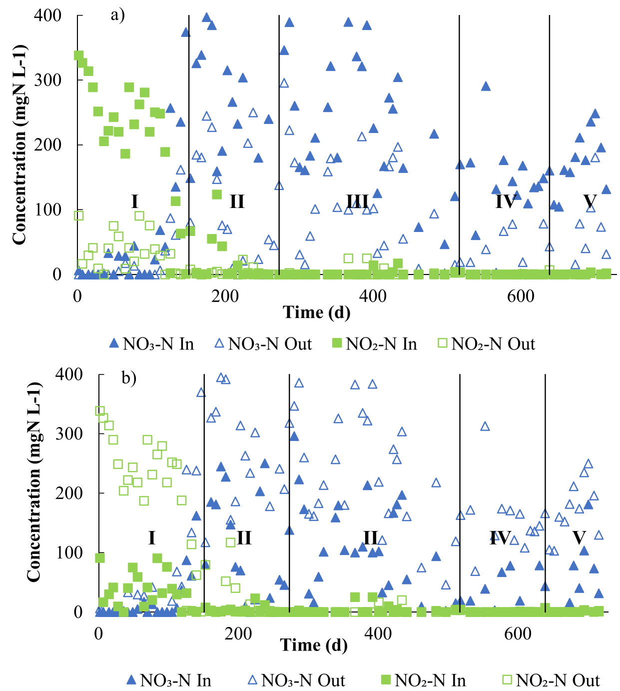

Fig. 3. Nitrite and Nitrate species concentration in N-Module denitrifying (a) and nitrifying (b) reactors. Evaluated seasonality: **Period I** (Warm) - correspond to the months of January, February, March, April and May 2019; **Period II** (Cold) - correspond to the months of June, July, August and September 2019; **Period III** (Warm) - correspond to the months of October, November, December 2019 and January, February, March, April and May 2020; **Period IV** (Cold) - correspond to the months of June, July, August and September 2020; and **Period V** (Warm) - correspond to the months of October, November and December 2020.

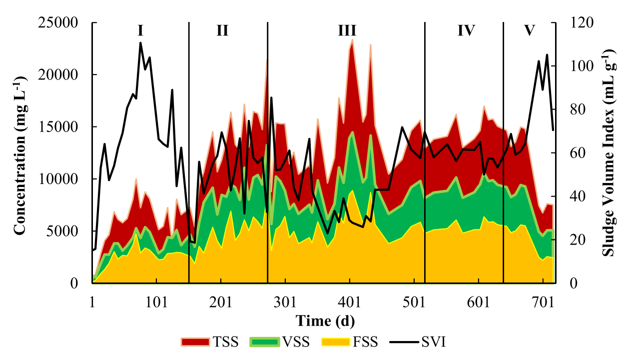

Fig. 4. Suspended solids profile inside the nitrifying reactor during the experimental period. (TSS - total suspended solids; VSS - volatile suspended solids; FSS - fixed suspended solids; SVI – sludge volume index). Evaluated seasonality: **Period I** (Warm) - correspond to the months of January, February, March, April and May 2019; **Period II** (Cold) - correspond to the months of June, July, August and September 2019; **Period III** (Warm) - correspond to the months of October, November, December 2019 and January, February, March, April and May 2020; **Period IV** (Cold) - correspond to the months of June, July, August and September 2020; and **Period V** (Warm) - correspond to the months of October, November and December 2020.

seasonal variations. Zhang et al. (2020) showed that the adaptation of the microbial community is a key element for the stability of the process, even after a load shock, raising it from 0.553 kg m^{-3} d^{-1}, to 0.790 kg m^{-3} d^{-1}. In this work the nitrogen removal system, based on nitrification, anammox and denitrification, maintained high N removal efficiency (96%), which the authors attributed to the long system operation time (246 days) and adaptation of the reactor community.

According to nitrogen species concentration in the nitrifying and denitrifying reactors (Fig. 3, and Tables 7S and 8S), it is possible to verify that in the first 120 days the denitrifying reactor (Fig. 3a) presented a high concentration of NO_2^--N in the influent (253.97 ± 104.03 mg_{NO2^--N} L^{-1}) and effluent (38.86 ± 29.01 mg_{NO2^--N} L^{-1}). This supports the high nitrite values at the nitrifying reactor effluent (253.97 ± 104.03 mg_{NO2^--N} L^{-1}) (Fig. 3b), due to the establishment and stabilization of the nitrogen removal process. After that, the NO_2^--N output and input decreased and it was observed an increase of NO_3^--N concentration, for both N-reactors. For nitrifying reactor effluent, the concentration of NO_3^--N of 219.4 ± 99.6 mg_{NO3^--N} L^{-1} could be attributed to the establishment of complete nitrification.

The content of suspended solids inside the nitrifying reactor also varied considerably (Fig. 4). During the evaluated period, the average concentration of VSS in the nitrifying reactor was 6943 ± 3000 mg_{VSS} L^{-1}, with a volatile and total suspended solids ratio (VSS/TSS) of 0.63 ± 0.06. Von Sperling (2007) shows that the ideal volatile suspended solids concentration in prolonged aeration systems is between 2000 and 4000 mg_{VSS} L^{-1}. However, Hollas et al. (2019) studied the behavior of a modified MLE system for nitrogen removal from swine manure under high suspended solid concentration (7800 mg_{VSS} L^{-1},12,500 mg_{VSS} L^{-1} and 18,500 mg_{VSS} L^{-1}) reaching good efficiency of TAN removal (99.6%, 99.1% and 96.5%, respectively) pointing out that it is possible to work at higher TSS in modified MLE systems than previously reported in literature.

During the two years of monitoring, sludge discharge from N-Module was 9.0 ± 5.0 m^3_{sludge} d^{-1}, which corresponds to an average of 2–15% of the reactor flow rate (Table 9S). The high solids concentration in the nitrifying system increases the energy consumption, since it requires higher aeration to provide the ideal mass transfer and oxygen consumption by the aerobic heterotrophic microorganisms that will act on solids degradation, raising oxygen requirement. On the other hand, the higher sludge residence time in the system reduces costs with the management of excess sludge, a challenge found in several treatment systems (Kim et al., 2020).

During the evaluated period the N-Module showed an energetic expenditure of 11,832 ± 2930 kWh $month^{-1}$. Admitting the TAN removal of 5746 ± 1213 kg_{TAN} $month^{-1}$, the energetic expenditure was 2.04 ± 0.26 kW kg^{-1}_{TAN}, resulting in 2.65 ± 0.57 kWh $m^{-3}_{treated\ effluent}$. For calculation purposes, based on the year 2020 and the price paid by the SISTRATES® project user, the kWh was US$0.1. Thus, the cost of energy for TAN removal was US$ 0.2 ± 0.026 kg^{-1}_{TAN} or US$ 0.28 ± 0.070 $m^{-3}_{treated\ effluent}$ in N-Module. It is important to highlight that the energy consumption of this module represents 23 ± 8% of the energy generated by the Bio-Module, and the N-Module does not have O_2 control, which could generate savings in energy expenditure.

Although the current trend is directed towards the recovery of resources, transforming linear chains into circular, with the insertion of waste as valuable substrates ensuring future activities, the removal of nitrogen, as presented in this study, is still configured as a fundamental tool for reducing environmental damage associated with the pig activity (Zhao et al., 2020). The environmental impacts caused by the improper release of N in the environment are known and include the eutrophication of freshwater and marine dead zones, as well as damage to human health. Therefore, the proposed configuration presented here was able to efficiently mitigate the nitrogen present in the swine waste, ensuring the viability of the activity and associated with other technologies that add to the benefits generated (Zubair et al., 2020). Although the N-Module presents a P removal (35.3 ± 39.2%), the concentration of this element on the effluent of this step still remains as potential pollutant if released without discrimination to the environment, and the addition of one more module to the system is necessary to remove this pollutant.

3.2.3. P-module

The scarcity of phosphorus resource added to the potential environmental damage that P can cause, makes its recovery an essential practice to ensure the sustainability of future activities (Jupp et al., 2021; Peng et al., 2018). Thus, the last SISTRATES® module aims the removal and recovery of P from swine digestate, by chemical precipitation with $Ca(OH)_2$.

Table 4
P-module removal efficiency considering in and out (liquid fraction) of this module compared to the whole system.

Parameter	Input SISTRATES® Concentration	Final effluent Concentration	SISTRATES® Global removal (%)
	Average ± S.D.	Average ± S.D.	Average ± S.D.
Alkalinity (mg_{CaCO3} L^{-1})	3744 ± 871	340.0 ± 117.0	99.6 ± 3.5
TAN (mg L^{-1})	1221 ± 457	2.4 ± 4.2	99.8 ± 0.3
TP (mg L^{-1})	453 ± 368	3.9 ± 2.6	99.2 ± 0.6
K (mg L^{-1})	757 ± 338	649.8 ± 178.0	20.7 ± 13.8
Ca (mg L^{-1})	–	64.2 ± 26.2	–

The start-up of P-Module was in May 2019, with the addition of an amount of 10% (m v^{-1}) $Ca(OH)_2$ to the effluent that fed the P reactor, until it reach pH of 9.0. This resulted in a consumption of 4.74 ± 2.13 $L_{suspension}$ $m^{-3}_{effluent}$ of reactant suspension. The Ca:P ratio was 1.98 ± 0.89, slightly above the theoretical molar ratio for the precipitation reaction which is 1.5 (Fernandes et al., 2012). The higher Ca:P ratio in the SISTRATES® full scale plant could be attributed to the operational drawbacks, as lack of pH sensor calibration, effluent alkalinity or organic carbon in the effluent from N-module due to seasonality effect (Karunanithi et al., 2015; Vanotti et al., 2003).

The effectiveness of using nitrification before P precipitation was observed by Szogi and Vanotti (2009) on 10 diverse North Carolina swine production farms. Chemical P precipitation was able to remove >90% of initial TP. Our results showed that the liquid fraction, that was the final effluent of the system presented TP concentration of 7.89 ± 11.85 mg L^{-1}) resulted in removal efficiency 89.6 ± 9.7% (Table 4). Similar results were found by Suzin et al. (2018) that obtained 90% of phosphorus removal at pH 9.0 promotes and (Fernandes et al., 2012) which observed that pH above 8.5 could remove >96% of the soluble phosphorus. This treatment allows the use of phosphorus sludge, rich in calcium phosphate, as a renewable and valuable fertilizer, and enables the effluent reuse or discharge into a water body.

The recovered P obtained in this module could be applied in the soil, as a fertilizer, or as a phosphorous source for animal feed. The solids concentration (dry matter) in the sludge was, on average, 1.8 ± 0.9% (w v^{-1}). Regarding the purity of the sludge, the results obtained for some elements were 1.7 ± 1.1% (w w^{-1}) for N, 5.8 ± 2.1% (w w^{-1}) for P, 2.1 ± 1.1% (w w^{-1}) for K and 21.4 ± 6.0% (w w^{-1}) for Ca.

It has been already proved that chemical precipitation is a good technique to P recovery from swine digestate and concomitantly inactivate *E. coli*, *Salmonella enterica* serovar *typhimurium* and PCV2. Vanotti et al. (2005) studied the inactivation of *Salmonella* and microbial indicators of fecal contamination (total coliforms, fecal coliforms, and enterococci) in a multi-step system composed of biological and chemical treatment of swine manure, reporting after P removal (precipitation with $Ca(OH)_2$) no colonies to count at the upper threshold limit value of <2 CFU mL^{-1}. Similar results were found in the present study, once the biomarkers *Salmonella*, *E. coli* and PCV2 were not observed in the final effluent. The sanitization of final effluent may be important for biosecurity reasons, impacting on environmental, animal, and human health, the three pillars of "One Health" concept.

The results are in line with the World Health Organization (WHO) and Environmental Protection Agency (EPA) recommendations (EPA, 2012). The WHO Guidelines (2006) recommend various levels of wastewater treatment according the irrigation category. For example, to water reuse in drip irrigation of high-growing crops must contain less than 10^3 in 100 mL of *E. coli* with 4 log reductions during the treatment. Thus, the water from the P-Module can be safely used for washing the swine houses or discharge in water bodies.

So far, there is few literatures available on water reuse for washing swine installations. According to Australian Pork (2016) the water consumption for cleaning the swine houses is on average 26% of the total water consumption. Considering the farrow-to-wean piglet producing unit described in the item 2.1, the estimated water consumption for washing and cleaning was 66.6 $m^3\ d^{-1}$. The SISTRATES® system recovers 124.5 ± 27.5 $m^3\ d^{-1}$ in liquid form. This effluent could be reused for cleaning the swine houses as one of its applications. Considering the water consumption per sow (38.5 L $sow^{-1}\ d^{-1}$), with the final effluent generated in P-Module, the results would bring water saving of 48.6% from farm water consumption.

The water scarcity affects more than 40% of the world population and this index tends to increase with the global temperature increase (Cantelle et al., 2018). According to the WWAP (2014) 70% of the total water consumption is from agriculture, including irrigation, livestock and aquaculture. Brazil plans to remain one of the main suppliers for the food and agricultural markets, a fact that highlights the importance of wastewater treatment and water reuse (Cantelle et al., 2018).

3.3. Technical coefficients, mass and energy balances

To be able to estimate the masses that were removed and recovered with SISTRATES®, to know if the recovered energy supplies the demand for energy consumed by the system, mass and energy balances for the three modules were calculated. The flow rates considered for mass balance calculations are presented in Table 10S.

Evaluating the Bio-Module mass balance, the CLBs are fed with 1151.5 ± 553.9 $kg_{VS}\ d^{-1}$, being 55.4 ± 28.9% of this daily load converted into biogas. The mass balance showed that 29.1 kg d^{-1} of FS is retained in these reactors, which represent 0.0003% of the CLBs useful volume. The CSTR reactor on the other hand was fed with 689.6 ± 316.9 $kg_{VS}\ d^{-1}$, and the conversion of VS into biogas in this reactor reaches up to 58.6 ± 32.8% while the FS accumulation is 72.6 $kg_{FS}\ d^{-1}$, which represents 0.0052% of the CSTR useful volume.

Khanh Nguyen et al. (2021) highlighted the importance of the manure pretreatment for AD, pointing out that the substrate becomes more bioavailable, optimizing the methanogenic potential of the feedstock, increasing the degree of degradation, and consequently decreasing the amount of sludge to be discharged, prolonging the useful life of these reactors.

The mass from the P-Module sludge (second generation phosphorus) results in 33.79 ± 19.40 kg d^{-1} of TP and 11.81 ± 6.19 kg d^{-1} of K, and the residual P in the liquid effluent fraction was 0.45 ± 0.34 kg P d^{-1}. In function of P_2O_5 recovery (considering the global average, according to Drangert et al., 2018) obtained with the system during this study, it would be possible to fertilize 350 ha per year, in need to supplement N. The completely nutrients balance of the system is shown in Fig. 2S.

According to FAO (2018) a ton of phosphate rock, that is used for fertilization, costs US$ 110.30, The nutrient mass balance of SISTRATES® showed that per year the recovery of P_2O_5 in average is 28 ton. This would result in savings of approximately US$ 3100 per year. However, a noble use of this material could be suggested, like the study developed by Shim et al. (2019) that evaluated the use of phosphorus recovered from swine manure applied in animal feed. The study analyzed the in vivo toxicity and in vitro solubility and concluded that it is possible use the phosphorus recovered as a P source in animal feed, representing a future application possibility for the nutrient recovered by SISTRATES®.

For energy generation considering the year of 2020 reached average values of 1881 ± 1859 kW d^{-1}, which resulted in conversion data of an average of 2.32 ± 2.4 kW $m^{-3}_{Nbiogás}$. According to the energy balance (Fig. 3S), SISTRATES® has an energy requirement of 15,925 ± 2201 kWh $month^{-1}$, which is 30 ± 11% of the total energy generated by the system.

The increasing demand for electricity worldwide confirms the importance of an effluent treatment system where energy can be recovered from agro-industrial wastes. The renewable energy from biomass represents 9.2% of Brazil in energy matrix, within the framework of renewable energy (46%) of the Brazilian energy matrix (EPE, 2018). According to CIAS (2021), the number of swine sows housed in 2019 was 2,017,645 in Brazil. With the data from this work, each sow provides daily 0.28 kWh d^{-1}, consequently the capacity of electricity generation of the Brazilian pig sector would be 565 MWh d^{-1}, emphasizing the importance of proper management of animal waste for the viability of livestock activities.

The environmental damages associated with animal waste are widely known and its efficient management is a challenge (Zubair et al., 2020). Thus, SISTRATES® can be considered an efficient model applicable to

Table 5
Technical coefficients raised in the SISTRATES® project per housed sow and per m^{-3}_{manure}.

Equation	considering sow	Average ± S.D.
01	WCS ($L_{water}\ sow^{-1}\ d^{-1}$)	38.5 ± 9.9
02	WGS ($L_{manure}\ sow^{-1}\ d^{-1}$)	28.7 ± 6.8
03	SGS_{Bio} ($m^3_{sludge}\ sow^{-1}\ d^{-1}$)	0.0027 ± 0.0007
04	SGS_{Bio+N} ($m^3_{sludge}\ sow^{-1}\ d^{-1}$)	0.0038 ± 0.014
05	$SGS_{Bio+N+P}$ ($m^3_{sludge}\ sow^{-1}\ d^{-1}$)	0.0059 ± 0.0012
06	BPS (N $m^3_{biogas}\ sow^{-1}\ d^{-1}$)	0.21 ± 0.11
07	EGS ($KWh_{generated}\ sow^{-1}\ d^{-1}$)	0.28 ± 0.27
08	ECS_{Bio} ($KWh_{consumed}\ sow^{-1}\ d^{-1}$)	0.018 ± 0.003
09	$ECS_{Bio+N+P}$ ($KWh_{consumed}\ sow^{-1}\ d^{-1}$)	0.077 ± 0.009
Equation	**considering m^{-3}_{manure}**	**Average ± S.D.**
10	SGW_{Bio} ($m^3_{slugde}\ m^{-3}_{manure}\ d^{-1}$)	0.099 ± 0.043
11	SGW_{Bio+N} ($m^3_{slugde}\ m^{-3}_{manure}\ d^{-1}$)	0.14 ± 0.07
12	$SGW_{Bio+N+P}$ ($m^3_{slugde}\ m^{-3}_{manure}\ d^{-1}$)	0.22 ± 0.09
13	BPW (N $m^3_{biogas}\ m^{-3}_{manure}\ d^{-1}$)	7.65 ± 5.56
14	EGW ($KWh_{generated}\ m^{-3}_{manure}\ d^{-1}$)	10.34 ± 11.14
15	ECW_{Bio} ($KWh_{consumed}\ m^{-3}_{manure}\ d^{-1}$)	0.68 ± 0.24
16	$ECW_{Bio+N+P}$ ($KW_{consumed}\ m^{-3}_{manure}\ d^{-1}$)	2.91 ± 1.14

Daily data considering the years of 2019–2020.
Note: S.D.: standard deviation; WCS: water consumption per sow; WGS: waste generation per sow; SGS: sludge generation per sow; BPS: biogas production per sow; EGS: electric energy generation per sow; ECS: electric energy consumption per sow; SGW: sludge generation per m^3 of treated row manure; BPW: biogas production per m^3 of treated row manure; EGW: electric energy generation per m^3 of treated row manure; ECW: electric energy consumption per m^3 of treated row manure; Bio: Bio-Module; N: N-Module; P: P-Module.

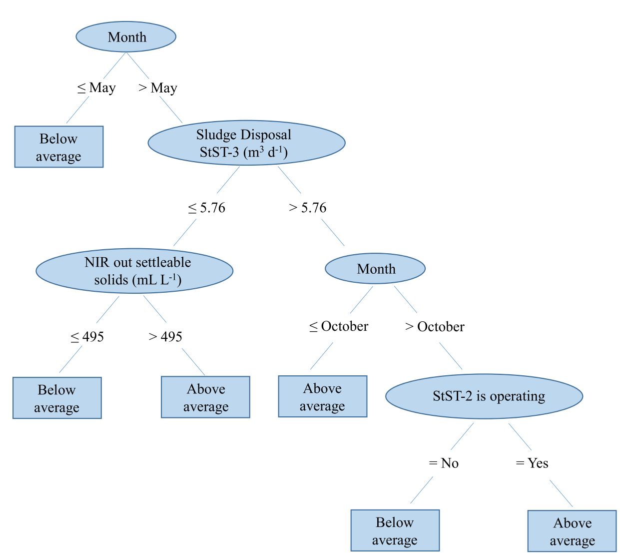

Fig. 5. Diagram of the decision tree for average electricity consumption, in N-module. NIR: Nitrifying reactor. Rectangular shape means the answer; ellipse shape means the decision criterion; the connections are the cutoff values.

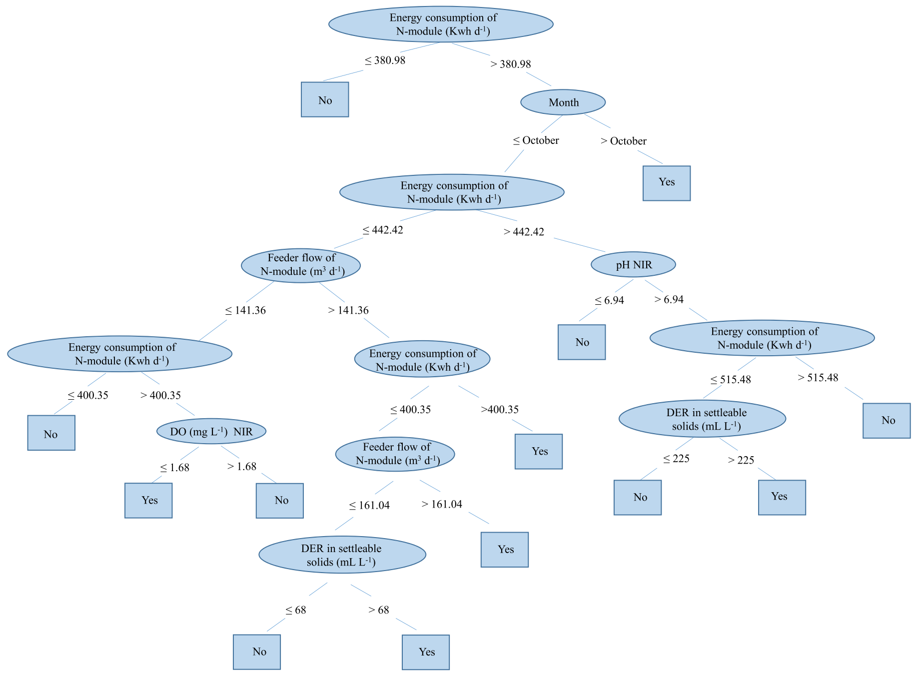

Fig. 6. Diagram of the decision tree for decision making to use StST-2 or direct from CLBs to the denitrifying reactor. NIR: Nitrifying reactor; DER: Denitrifying reactor; DO: Dissolved oxygen; CLB: Covered Lagoon Biodigester. Rectangular shape means the answer; ellipse shape means the decision criterion; the connections are the cutoff values.

the management of swine waste. In this sense, Table 5 summarizes the technical coefficients calculated to extrapolate the unit where SISTRATES® is currently able. Thus, the treatment system can be used in different production cycles, farm sizes and configurations since there is the possibility of modular implementation.

3.4. Machine learning

Decision trees are supervised learning algorithms that divide a large data set into smaller sets with certain decision rules. In WEKA®, this is accomplished by the J48 algorithm, which is an open-source Java implementation of the C4.5 algorithm (Rossi et al., 2020). In this sense, artificial intelligence approaches are important for analysis and decision making in treatment systems (Asami et al., 2021; Oprea, 2018).

Based on the large data set collected throughout the SISTRATES® monitoring, the use of data mining is important for machine training, making it possible to determine possible existing correlations not perceptible in the interpretation of the data individually, or with simple correlations. Considering this, decision trees were built, to predict behaviors in the treatment plant, considering data that can be monitored in the field to predict other parameters. The first decision tree is related to the prediction of energy expenditure in N-module. Considering the average energy spent in the process, the size of the tree was 11 and the number of leaves (tips) was 6, which have the attributes (variables) (Fig. 5). It was possible to correctly classify 100% of the instances (731), indicating a high predictive power (100% correct), with a low mean absolute error (0.0024).

The month was one of the influencing factors on energy consumption, as already discussed. It presented two nodes, being the first, marked by the end of summer and beginning of winter (January/May = summer and June/October = winter) and the second between the end of winter and beginning of summer (June/October = winter and November/December = summer). The warm months tend to present a lower energy consumption than the average, mainly because of the settleable solids present in the system as already discussed. The other nodes were directly related to solids, returning important information about the concentration of settleable solids in the nitrifying reactor, which should be below 495 mL L^{-1} to not have an unnecessary energy expenditure.

Another decision tree elaborated, predicts the need to use StST-2 to reduce the solids loading of the CLBs output. The size of this tree was 25 and the number of leaves (tips) was 13, larger than the previous one, *i.e.,* the decision making considers more factors (Fig. 6). It was possible to correctly classify 98.5% of the instances (715), also indicating high predictive power, with low mean absolute error (0.0259).

Following the trend discussed above, of the influence on energy expenditure as a function of system settleable solids, this was also a determining point for the decision making presented in the tree in Fig. 5, as well as the seasonal issue, with the increment of information on settleable solids in input of denitrifying reactor. In addition, more specific information from the nitrifying/denitrifying reactors is required, since the environmental conditions determine the performance of the N-module. Caglar Gencosman and Eker Sanli (2021) studied machine learning techniques, including the generation of decision trees, to predict the removal efficiency of polycyclic aromatic hydrocarbons from wastewater treatment sludge about the initial levels of these. The authors found that the decision trees showed an average performance of 78.0%, with high specificity and accuracy, not only for the variable of interest but being able to predict other parameters. Asami et al. (2021) also studied the use of decision trees for data prediction of biological processes in a wastewater treatment plant and found high coefficients of determination (0.83–0.90) in the trees created for the parameters BOD_5, COD and TSS, concluding that this approach is a good tool to describe and analyze the behavior of the data.

Another important prediction is to determine which sludge discharge flow rate, in StST-3 must be applied to the N-module without compromising the process based on simple analyses that can be quickly performed in the treatment plant by the operator. With this, by means of the linear regression between the sludge height of StST-3 and the concentration of settleable solids in the decanter supernatant. It was possible to verify with a correlation coefficient of 0.5052, that the sludge height cannot be higher than 1.31 m, otherwise, the concentration of settleable solids in the supernatant fraction increases indicating that the system has sludge accumulation.

Also, by means of linear regression, between the C/N ratio and the ammonia concentration at the output of the N-module, it was verified that the C/N ratio must be higher than 0.72, so that the system efficiency is higher than the average and the concentration at the output is lower than 20 mg L^{-1}, with a correlation coefficient of 0.4188. Although the regressions do not present such high fits, they describe well the behavior of the system and help in future decision making.

The surveyed data provides a valuable decision support resource, and from the increment of deeper knowledge of existing relationships, not always exploited for the operation of SISTRATES®. Data mining combined with machine learning has proven to be an important tool for simplifying the interpretation of easily measured variables in the field.

4. Conclusions

From the results presented here, it is evident the potential that swine waste presents as a precursor to changes in the production chain, in terms of taking advantage of the benefits that can be derived from its adequate management. Thus, SISTRATES® proved to be an efficient model applicable to the management of swine waste, due to the simultaneous nutrients and energy recovery reducing the main environmental contaminants. In addition to enabling the reuse of water in production facilities, a great benefit considering the world's water shortages. There are still challenges in the management of this waste, so future work should focus on improving the proposed system, such as the maximization of energy use, nutrient recovery, and water reuse, towards circularity in the pig chain.

Credit autorship contribution statement

Daniela Cândido: Conceptualization; Investigation; Data curation; Writing – original draft. **Alice Chiapetti Bolsan**: Investigation. **Camila Ester Hollas**: Conceptualization; Software; Investigation; Writing – review & editing. **Bruno Venturin**: Software; Formal analysis; Investigation; Data curation.**Deisi Cristina Tápparo**: Conceptualization; Investigation; Writing – review & editing. **Gabriela Bonassa**: Investigation; Writing – review & editing. **Fabiane Goldschmidt Antes**: Investigation. **Ricardo Luís Radis Steinmetz**: Investigation. **Marcelo Bortoli**: Writing – review & editing. **Airton Kunz**: Conceptualization; Writing – review & editing; Supervision; Project administration; Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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