A Chlorophyll a Criterion for High Rock Lake

by the

North Carolina Nutrient Criteria Science Advisory Council

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May 26, 2020

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Executive Summary

This document provides a description and technical background for a site-specific chlorophyll a (chl a) criterion for High Rock Lake, North Carolina, a freshwater reservoir in the Yadkin-Pee Dee river basin of North and South Carolina. The work by the North Carolina Science Advisory Council (SAC) to establish this criterion is part of larger effort in North Carolina to develop nutrient criteria throughout North Carolina on a site-specific basis for three separate water body types: 1) reservoirs/lakes, 2) rivers/streams and 3) estuaries. The existing numeric chl a criterion of 40 µg/L is assessed on a "not-to-exceed" basis as part of a narrative standard for lakes, sounds, estuaries, reservoirs, and other slow-moving waters not designated as trout waters. The criterion is exceeded when there is greater than a 90% statistical confidence that more than 10% of samples will exceed a 40 µg/L photic zone average concentration. The efficacy of applying a single chl a criterion to protect the wide variety of surface water habitats in North Carolina has been debated, and development of site-specific Chl a criteria have been promoted by the US EPA and authorized for North Carolina by the Environmental Management Commission.

This newly-developed, site-specific Chl a criterion has been developed according to a process that considered the designated uses (aesthetics, water supply, aquatic habitat, and recreation) of High Rock Lake. The criterion was developed to protect these designated uses. Multiple lines of evidence (e.g. literature review, water quality monitoring results, assessments of designated use attainments) were used to determine the appropriate chl a concentration, its averaging period, and the frequency of criterion exceedance that would be protective of the designated uses.

The literature review found that increases in chl a concentration decrease water clarity and correlate strongly with increasing primary production in phytoplankton dominated systems. Freshwater fisheries production generally responds positively to increases in chl a. An upper threshold exists, however, to the positive relationship between chl a and overall fisheries production. At chl a levels beyond the threshold, negative impacts of excessive algal production on water quality (e.g. dissolved oxygen concentrations, water clarity) may reduce fish production, or cause substantial shifts toward less desirable fish species. Higher chl a values may also increase risks from phytotoxins. Several genera of bloom-forming cyanobacteria can produce a potent suite of secondary metabolites that are hepatotoxic and neurotoxic and can harm aquatic life. There is not a simple relationship between chl a and toxin concentration. Despite a considerable literature on phytotoxins in lakes, given current information available, the SAC does not advise establishing chl a standards based solely on cyanotoxin risk to aquatic life.

The SAC reviewed water quality monitoring studies conducted by a number of research groups in High Rock Lake from 1973 – 2016. Designated use assessments were also reviewed. Based on nutrient and chl a concentrations, previous studies have consistently characterized High Rock Lake as a eutrophic reservoir. The lake has been considered to be like many "run-of-the-river" reservoirs that have distinct riverine, transitional, and lacustrine zones. Chl a concentrations were generally highest in the transitional zone of the lake and have frequently exceeded the existing 40 µg/L chl a standard. There are no clear long-term trends in chl a concentration.

Data on other indicators of water quality such as dissolved oxygen (DO), pH, water clarity, algal abundance, and phytotoxin concentration were also reviewed. Chl a concentrations in High Rock Lake are correlated with relatively high DO surface concentrations, mixed effects on bottom DO concentrations, and relatively high DO percent saturation values. The reservoir attains water quality criteria for DO under existing chl a conditions. The pH of surface waters in High Rock Lake (<0.2 m) was found to be highest during the months of June through September. Measured pH exceeded 9.0 in 24-38% of the measurements. Exceedances of a pH of 9.0 occurred over the entire range of chl a values, but were more common when chl a exceeded 30 µg/L. The water clarity in High Rock Lake, based upon the most recent assessment using turbidity measurements, is considered impaired in the upper riverine portion of the lake. Algal abundances and taxonomy were found to vary seasonally in a fashion typical for temperate eutrophic reservoirs with summer maxima and winter minima. In-situ phytotoxin tracking devices deployed as part of a special sampling study in 2016 indicated that microcystin, anatoxin, and cylindrospermopsin were present throughout much of the summer in High Rock Lake and were often detected simultaneously. Bulk water analysis indicated that toxin concentrations were below action limits or health advisory concentrations.

Based on assessments made by the NC Wildlife Resources Commission, current water quality conditions appear to be supportive of a sport fishery focused on largemouth bass, striped bass, and crappie, sunfish, and catfish. The largemouth fishery has been consistently evaluated as a "quality fishery" sustained by adequate recruitment and non-excessive mortality. Fish kills are uncommon in HRL, and large fish kills have only been noted during the major drought of 2002 when low flows, low water levels, high summer temperatures, and low dissolved oxygen caused major fish kills. The SAC is not aware of any aesthetic or swimming use impairment of the lake, even though chl a concentrations routinely exceed 50 µg/L.

The SAC used a literature review and the reservoir-specific water quality and use assessment observations to develop the recommended site-specific chl a criterion. The proposed chl a criterion for High Rock Lake is a seasonal geomean of 35 µg/L, not to be exceeded more than once in three years, for growing season months of April-October based on protection of all uses while maintaining the productivity of the sport fishery. In terms of spatial considerations, all monitoring data from open waters within assessment units collected during the months of April through October would be used to compute a geomean to compare with the proposed criterion. The criterion would apply to all months of the year, with attainment of the standard assessed with data from the growing season months. The SAC recommended maximum exceedance frequency is not to exceed more than one in three calculated seasonal geomean values.

The SAC also considered how lessons learned from the reservoir pilot might inform a statewide framework for deriving lake-specific chl a criteria. Such a framework should produce criteria that minimize both type I (false finding of use impairment) and type II (false finding of use attainment) errors. Several framework elements could streamline the criteria development process while still making use of both the scientific literature and lake-specific information.

These elements include: (1) using similar duration and frequency components as recommended for the lake pilot; (2) a chlorophyll a screening range to inform lake use attainment status; (3) a predetermined list of numeric and narrative indicators of use attainment; and (4) decision guidelines for translating lake evaluation results into site-specific criteria. The SAC and DEQ could revisit these concepts during the statewide criteria development phase of the NCDP.

1. Introduction

As described in the North Carolina Nutrient Criteria Development Plan (NCDP), (NCDWR 2014) and its revised version (NCDWR 2019), North Carolina is working towards developing scientifically defensible numeric nutrient criteria throughout the state on a site-specific basis. According to the plan, numeric nutrient criteria will be developed initially for one example each of three distinct water body types. The water bodies and the water body types are as follows:

1.0 High Rock Lake (reservoirs/lakes)
2.0 Central portion of the Cape Fear River (rivers/streams)
3.0 Chowan River/Albemarle Sound (estuaries)

An important component of the NCDP has been the creation of a twelve-member scientific advisory council (SAC) to advise and assist the North Carolina Division of Water Resources (NCDWR) in the development of numeric nutrient criteria. This document represents the work of the SAC done with the cooperation and assistance of the NCDWR. In this document the SAC provides a description, a rationale, and technical background for site-specific chlorophyll a (chl a) criterion for High Rock Lake, North Carolina, a freshwater reservoir in the Yadkin-Pee Dee river basin of North and South Carolina.

High Rock Lake is a freshwater reservoir in the piedmont region of North Carolina. It is a 15,180-acre reservoir with a 3,974 mi^2 drainage area located within the upper portion of the Yadkin River basin (Figure 1.1). It is the first of a chain of four lakes (High Rock, Tuckertown Badin, and Falls) that were created between 1917 and 1962 by Alcoa to provide hydroelectric power for aluminum production (Cube Hydro Carolinas 2019).

According to a 2004 review of water quality data (Tetra Tech 2004), High Rock Lake has been characterized as eutrophic since the 1970's. EPA assessed water quality conditions in 1973 in sixteen North Carolina lakes as part of a national eutrophication survey (USEPA 1975), finding High Rock Lake to be the most eutrophic of the North Carolina lakes studied. At the time, EPA noted that High Rock Lake's variable but relatively short residence time (estimated at 27 days for mean flow) produced a lake that operates more like a slow-moving river than a typical lake.

Tetra Tech summarized several additional water quality assessments in ensuing years that have each shown High Rock Lake to have relatively high levels of turbidity, nutrients, and phytoplankton abundance (i.e. high chl a concentration) (Tetra Tech 2004). High Rock Lake is currently on North Carolina's list of impaired or threatened waters as required under Section 303(d) of the Clean Water Act. Based upon the current numeric chl a criterion, the entire lake is impaired for chl a and parts of the lake are impaired for pH and turbidity. Additional information on the current numeric North Carolina chl a criterion, and policies for listing and delisting waterbodies as impaired is provided in the following section of this chapter.

The work of the SAC on a new numeric nutrient criterion for High Rock Lake has had multiple objectives. While the immediate, primary objective of the work has been to develop a site-specific criterion for High Rock Lake, a secondary objective has been to develop a methodology for criteria development that can be applied to other North Carolina lakes and reservoirs, and perhaps to other water body types within the state. The final section of this chapter describes the general approach that the SAC has used to develop a site-specific chl a criterion for High Rock Lake. One aspect of the approach is to utilize the scientific literature as a basis for the site-specific criterion. A review of the relevant literature relating important eutrophication response variables such as water clarity and chl a concentrations to relevant designated uses such as aesthetics, water supply, aquatic habitat, and recreation is provided in chapter 2. Chapter 3 then

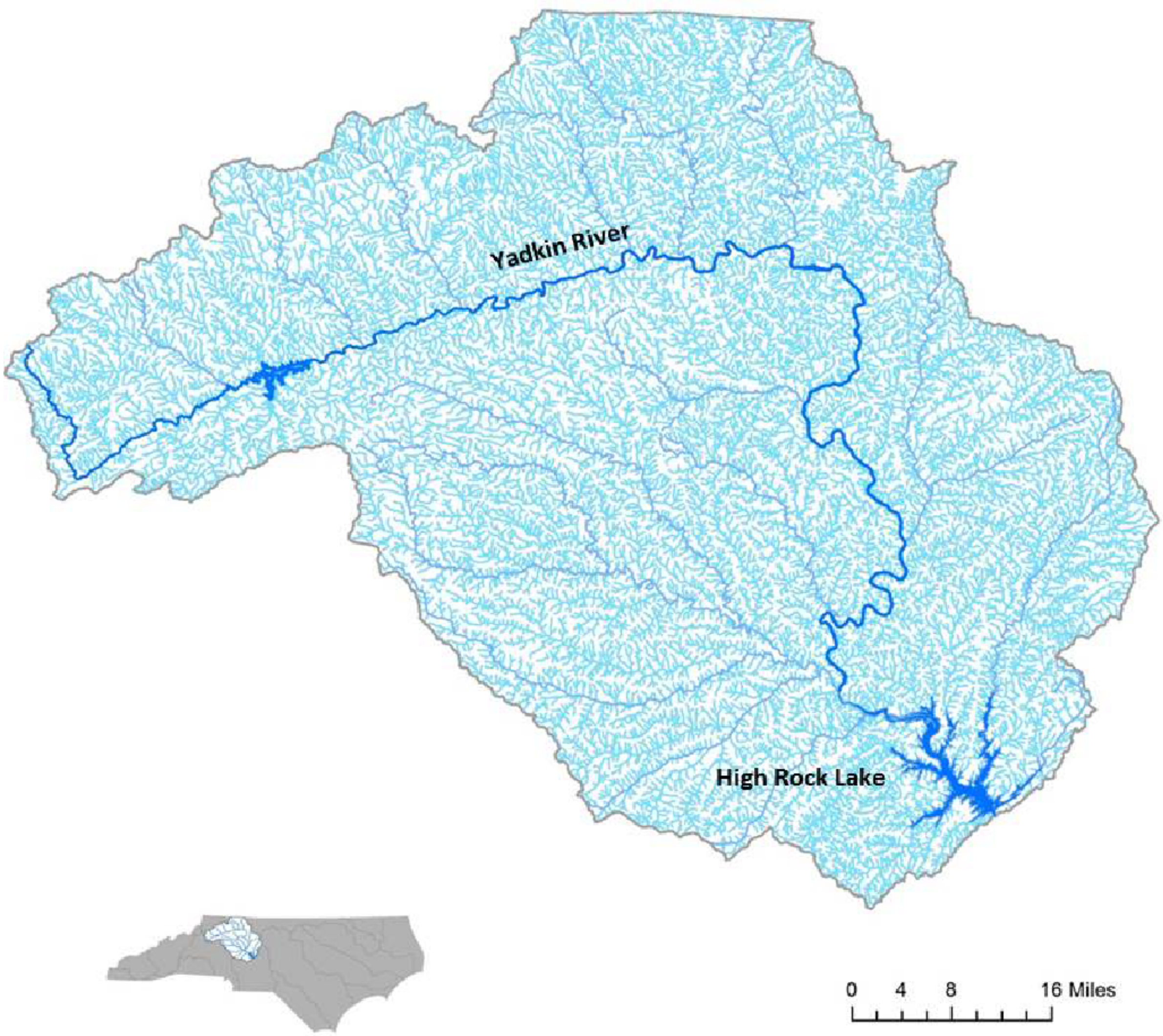


Figure 1.1. High Rock Lake and watershed. (figure taken from the North Carolina Nutrient Criteria Development Plan (NCDWR 2019))

looks specifically at the extent to which designated uses in High Rock Lake are supported given the current water quality conditions. Chapter 4 then describes the proposed site-specific numeric chl a criterion for High Rock Lake. The concluding chapter of this document (Chapter 5) then returns to the larger task of developing numeric nutrient criteria for all the lakes and reservoirs in North Carolina. The chapter proposes elements of a framework that the SAC believes could be the basis for a general approach for developing site specific nutrient criteria across the range of water body types in North Carolina.

1.1 Description of the Current North Carolina Chl a Criterion

As described in Division of Water Resources' (DWR) May 2017 chl a description document (NC Division of Water Resources 2017), the existing chl a criterion "arose through an advisory group process and was informed by lake and reservoir research including the 1976 report by Charles Weiss and Edward J. Kuenzler 'The Trophic States of North Carolina Lakes (Weiss and Kuenzler 1976) .'" The current approved regulatory text for the State's chl a criterion, located at 15A NCAC 02B .0211(4), states:

> *Chlorophyll-a (corrected):* ***not greater than 40 µg/l for lakes, reservoirs, and other waters subject to growths of macroscopic or microscopic vegetation not designated as trout waters****, and not greater than 15 µg/l for lakes, reservoirs, and other waters subject to growths of macroscopic or microscopic vegetation designated as trout waters (not applicable to lakes or reservoirs less than 10 acres in surface area). The Commission or its designee may prohibit or limit any discharge of waste into surface waters if the surface waters experience or the discharge would result in growths of microscopic or macroscopic vegetation such that the standards established pursuant to this Rule would be violated or the intended best usage of the waters would be impaired;* ***(Emphasis added)***

The 2017 Summary and 1976 report characterize 40 µg/L as the "upper range for alpha-eutrophic waters (15µg/l to 40 µg/L)." Weiss and Kuenzler indicated that the scale of quality is an interpretation that is not about whether the water should or should not be used; but rather the interpretation that some attributes of more eutrophic waters are more acceptable – plenty of fish – while others are less acceptable – swimming in algal blooms. The 2017 Summary also references excerpts from another historical document, the records provided by Mike McGhee, former EPA and NC DEM employee (McGhee 1983). The comments included in those notes point to the importance of a chl a criterion to limit point and nonpoint discharges of nutrients, including nitrogen and phosphorus.

For additional information, excerpts (shown in italics) from the NCDWR 2018 303(d) listing/delisting procedures document are included. (NCDWR 2018). The excerpts summarize how the state completes assessments of the existing chl a criterion based on collected ambient data for determining whether a waterbody should be listed on the North Carolina Section 303(d) list. The flowcharts (Fig. 1.2) for listing and delisting waters when assessing numeric criteria are also provided for reference.

ASSESSING CHLOROPHYLL-A NUMERIC CRITERIA

The following sets of evaluations will be used for the 2018 assessment for these parameters: chlorophyll-a, dissolved oxygen, MBAS, mercury, nitrate/nitrite, pH, temperature, toxic substances, and turbidity. For each parameter there is a brief discussion of the standard used for assessment of the parameter including any parameter-specific good causes for not assessing in Category 5. Note Category 5 is the 303(d) list.

The true frequency of criteria exceedances cannot be measured. It must be estimated from a set of samples, which introduces statistical uncertainty. The degree of uncertainty depends on the sample size. NC will use a nonparametric hypothesis testing approach based on the binomial distribution. The binomial method allows a quantifiable level of statistical confidence (90%) for listing decisions, which provides a 10% probability of listing an assessment unit when it should not be listed. The null hypothesis is that the overall exceedance probability is less than or equal to the 10% exceedance allowance. NC will also consider the number of excursions of criterion for newer data that have not been assessed before. For 2018 assessment, newer data are defined as data collected during calendar years 2015 and 2016.

Exceeding Criteria-Category 5

- *Sample size is greater than nine.*
- *Greater than 10% exceedance with greater than or equal to 90% confidence, or*
- *Greater than 10% exceedance, but less than 90% statistical confidence, and at least 4 excursions in newer data that have previously not been assessed.*

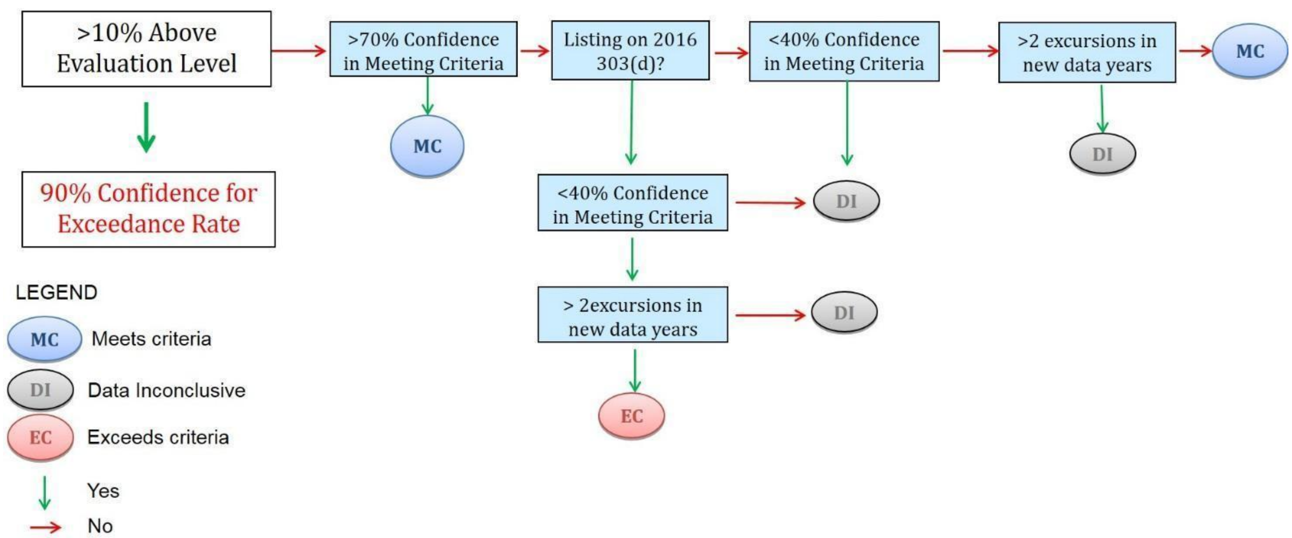
DELISTING WATERS

NC will review the final 2016 303(d) list as the starting point for the development of the 2018 303(d) list. All waters on the 2016 303(d) listing will be evaluated for appropriate inclusion on the 2018 303(d) list as defined in 40 CFR 130.2(j). NC will apply a combination of nonparametric hypotheses testing based on the binomial distribution as well as an analysis of the dates of excursions to determine if there is good cause to delist a water. An analysis of newer data that have not been previously assessed is included in the delisting procedure to allow the state to determine if criterion excursions are more recent.

For delisting waters, if the 2018 assessment results in greater than 10% exceedance rate with less than 90% statistical confidence and the water was on the 2016 303(d) list, the water will be delisted if there are less than 2 excursions of the criterion in newer data that have not been previously assessed. If the 2018 assessment results in less than 10% exceedance rate and the water was on the 2016 303(d) list, the water will be delisted if there is greater than 40%

statistical confidence that there is less than a 10% exceedance of the criterion or if there are less than 3 excursions of the criterion in newer data that have not been previously assessed.

Flow chart for listing a waterbody



Flow chart for delisting a waterbody

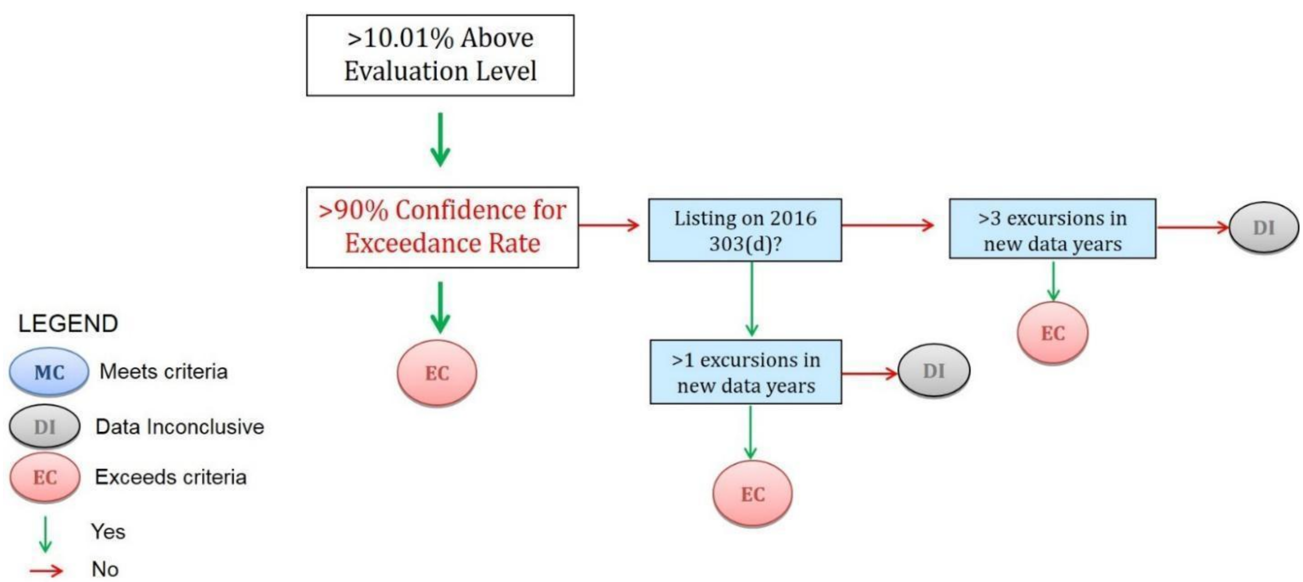


Figure 1.2. Numeric Criteria Assessment Flowcharts

1.2 Overview of Science Advisory Council Approach

The SAC was charged with recommending new numeric nutrient criteria so that High Rock Lake meets its designated uses, which include public water supply, recreation, and aquatic life. An important designated use in High Rock Lake is fisheries due to the quality of the bass fishing. The focus of the SAC discussions were around designated uses, but also included lake use protection into the future as the climate changes or if information on the lake's health changes.

The SAC has proceeded in a step-wise fashion to recommend a new chl a criterion for High Rock Lake. The first phase of the SAC's work was information gathering so that the diverse group could understand water quality standards, numeric nutrient criteria development, and learn about High Rock Lake. Information was diverse and included uses and attainments, historical water quality data, modeling, and other pertinent material collected from and about High Rock Lake.

Additional information on multiple topics, such as the relationship between lake pH or chl a values and fisheries, was developed by various SAC members from literature reviews. These data were often tabulated in a "database" that was contextualized geographically for better comparison to High Rock Lake conditions. Literature and data were shared among and discussed between SAC members.

Numerous proposals for pH and chl a criteria were then developed by various SAC members using multiple lines of evidence. Proposal discussions focused on averaging period and the frequency of criterion exceedance that would be protective of the designated uses. Once fully discussed, votes were taken on the different proposals until consensus was reached for a new recommended chl a criterion. At a two-day SAC meeting in December 2018, the group's conclusions were substantively captured and that content was used to produce the current document, including additional refinement on certain components as the document was finalized. The newly developed proposed criterion has used the best science available and multiple lines of evidence to determine the appropriate chl a criterion that is protective of the water quality standards. As with most water quality decisions, numeric outcome-points consist of both data and best scientific judgement.

Since the role of the SAC is to recommend standards protective of the water resource, the committee tried not to discuss criteria relative to their feasibility and/or attainability. A companion committee, the Criteria Implementation Committee or CIC, was formed to focus on implementation of the recommended nutrient criteria as determined by the SAC. The CIC group meets after the SAC committee proposes criteria. Their job is to refer clarifying questions back to the SAC and also make determinations relative to the feasibility that the water resource can meet these criteria. The process between the SAC and the CIC is iterative. Many of the CIC members have attended the SAC meetings to better understand the deliberations that occur around the new nutrient criteria recommendations for the High Rock Lake.

1.2 References

Cube Hydro Carolinas. (2019). "The Yadkin Project." 2019, from http://cubecarolinas.com/the-yadkin-project/. (retrieved May 25, 2020).

McGhee, R. (1983). "Experiences in developing a chlorophyll a. standard in the Southeast to protect lakes, reservoirs, and estuaries." Lake Restoration Protection and Management: 163-165.

NC Division of Water Resources (2017). Surface Water Quality Standards History Document: Chlorophyll a. Raleigh, NC, NC Division of Water Resources**:** 2.

NCDWR, N. C. D. o. W. R. (2014). North Carolina Nutrient Criteria Development Plan, June 20, 2014. Raleigh, NC, North Carolina Department of Environment and Natural Resources.

NCDWR, N. C. D. o. W. R. (2018). 2018 303(d) LISTING AND DELISTING METHODOLOGY, Approved by the North Carolina Environmental Management Commission on March 8, 2018. N. C. D. o. E. Quality. Raleigh, NC.

NCDWR, N. C. D. o. W. R. (2019). North Carolina Nutrient Criteria Development Plan, v.2, May 16, 2019. Raleigh, NC.

Tetra Tech (2004). Water Quality Data Review for High Rock Lake, North Carolina. Research Triangle Park, NC 27709.

USEPA (1975). Report on High Rock Lake, Davidson and Rowan Counties, North Carolina, EPA Region IV. National Eutrophication Survey Working Paper No. 381. Pacific Northwest Environmental Research Laboratory, Corvallis, OR.

Weiss, C. M. and E. J. Kuenzler (1976). The trophic state of North Carolina lakes, Water Resources Research Institute of the University of North Carolina.

2. Literature Review of Chl a and Use Attainment

This chapter reviews the existing literature relating chlorophyll a (chl a) concentrations and the attainment of designated uses in surface water bodies. Surface water chl a concentration is a proxy for phytoplankton biomass and correlates strongly with primary production in systems where phytoplankton are the dominant primary producers (Cloern et al. 1995). Thus, chl a is a strong indicator of trophic status. The principal function of chl a is to absorb visible sunlight within the photosynthetically active radiation band (PAR, 400 - 700 nm), and convert PAR into chemical energy needed to fuel carbon fixation. Through PAR absorption, chl a can directly impact light levels necessary for other plants (e.g., submerged aquatic vegetation) to grow and for animals including humans to see. Indirect impacts of elevated chl a on aquatic life include excessive organic matter production and subsequent water quality degradation (e.g. high/low pH, high/low dissolved oxygen), and toxicity from secondary metabolites that co-occur with chl a in phytoplankton cells (e.g. cyanotoxins).

2.1. Chl a and Water Clarity

On average, chl a in phytoplankton cause about a 0.02/m attenuation of photosynthetically active radiation (PAR) for every µg/L chl a (Koseff et al. 1993). A phytoplankton bloom of about 40 µg/L would result in a light attenuation value that approximately halves the light availability with every meter depth. The relative importance of phytoplankton chl a in attenuating light depends on the concentrations of other light attenuating substances including suspended mineral sediment, suspended organic detritus (terrestrial or aquatic), and colored dissolved organic matter (CDOM) (Biber et al. 2008).

For water bodies with submerged aquatic vegetation (SAV), the amount of light that penetrates to the bottom often determines the maximum depth that SAV can grow, and can limit SAV areal coverage of shallow, nearshore areas in waters with elevated chl a concentration. Because of the importance of SAVs for stabilizing sediments, trapping nutrients, and serving as a structured habitat for fish and invertebrate communities, maintaining SAV coverage is often an important component protecting aquatic life uses. Determining a chl a criterion that is protective of SAV coverage requires knowledge of the light requirement for SAV growth, the maximum depth of SAV beds to be protected, and the amount of background light attenuation from substances other than chl a. Light requirements for SAV growth vary modestly among species and sediment characteristics, but usually range from 10-20% of incident sunlight. Concentrations and relative importance of light-attenuating substances vary greatly across aquatic systems and result in chl a targets for SAV protection being highly site specific. For example, within different regions of Chesapeake Bay, chl a targets to maintain SAV growth at depths from 0.5 to 2 m ranged by more than an order of magnitude from 2.7 to 43 µg/L (EPA 2007). For some waters, concentrations of sediments or CDOM are so high that SAV cannot grow, even though chl a in these waters is often negligible, and otherwise suitable substrates exist (Bachmann et al. 2002). A standard of 20 µg/L was approved for Lake Winona, Minnesota to protect SAV coverage (MN PCA 2014). Although protection of SAV is often a consideration in developing chl a criteria, SAV have apparently never been established in HRL, and with high suspended sediment concentrations and widely fluctuating water levels, it is unclear whether even drastic chl a reductions would allow for SAV establishment (see Chapter 3.4.1). Decreases in water clarity associated with high chl a can also affect aquatic life uses by impacting predator/prey (Manning et al. 2013) and competitive (Stasko et al. 2015) interactions, and altering heat budgets with resultant changes in temperature and oxygen solubility (Rose et al. 2016; Heiskanen et al. 2015). These indirect effects of water clarity are becoming better understood, but at present have not been used to establish chl a thresholds for protecting aquatic life.

2.2. Fisheries Effects

In general, freshwater fisheries production responds positively to increases in chl a due to higher rates of phytoplankton based primary production (Deines et al. 2015)) that fuels production at higher trophic levels. Bachmann et al. (1996) found a clear positive relation between standing stock fish biomass and annual average chl a across 60 Florida Lakes with chl a levels ranging from 1 to about 100 µg chl a (Fig. 2.1 A). For crappie, an optimal range 20-60 µg/L has been reported (Schupp and Wilson 1993), which is slightly lower than optimal for bass and sunfish production (40-60 µg/L, Maceina and Bayne 2001). Similar results were found in a comparison of fish and chl a in Iowa reservoirs (Egerston and Downing 2004). In a meta analysis of over 700 freshwater systems worldwide, Deines et al. (2015) also found consistent positive relationships between chl a and several metrics of fish production (production, yield, catch per unit effort, and density), with coefficients of variation averaging 0.71 (95% confidence interval = 0.59-0.80). Their study also included examination of climate impacts on fisheries but measures of autotrophic production were consistently more important predictors of fish production metrics.

Across four Alabama and Georgia reservoirs, biomass and growth rates of black bass, the apex predator, were positively related to average growing season chl a across the range 2 – 27 µg/L (Bayne et al. 1994). Higher production of top predators in the eutrophic reservoirs was partly related to increased efficiency of trophic transfer that was driven by a shortened food chain. In the more eutrophic lakes, large phytoplankton were consumed directly by herbivorous shad while in the mesotrophic reservoir, crustacean zooplankton served as a more important trophic link between phytoplankton and planktivorous fish. Lower relative abundance of crustacean zooplankton in the eutrophic reservoirs was linked to lower relative abundance of *Lepomis* sunfish that prey largely on crustacean zooplankton in their early developmental stages. Thus, higher productivity may favor planktivorous fish and their predators over other guilds of fish (Bayne et al. 1994; Allen et al. 1999). There is also indication that very high productivity may increase the predominance of benthic species such as catfish and roughfish that may or may not be desirable (Egertson and Downing 2004; Michaletz et al. 2012. The types of fish communities that are desired should be considered when designing nutrient management strategies to support both fishery and water quality related uses.

In addition to causing shifts in composition of fish communities, an upper threshold to the positive relationship between chl a and overall fisheries production is expected. At chl a levels beyond the threshold, negative impacts of excessive algal production on water quality (e.g. dissolved oxygen concentrations, water clarity) may reduce fish production, or cause substantial shifts toward less desirable fish species. Yurk and Ney (1989) found that across 22 southeastern US reservoirs, chl a correlated positively with total fish abundance, but suitable habitats for desirable walleye and striped bass occurred where reduced algal production allowed hypolimnetic waters to remain oxygenated. In Westpoint Reservoir, Georgia, a 50% reduction in chl a from approximately 40 to 20 led to shifts in the dominant species of black bass (Maceina and Bayne 2001). The smaller spotted bass replaced largemouth bass with an overall increase in number of fish, but a decrease in total black bass biomass.

Boucek et al. (2017) found some evidence for an upper threshold in the relationship between largemouth bass condition (mass divided by length) and chl a such that condition improved up to a chl a level of about 80-100, but subsequently decreased at higher chl a (Fig. 2.1 B). It is worth noting that the decrease in body condition at higher chl a levels was driven only by two data points with the highest chl a. In general, evidence for declines in fisheries production at the highest chl a levels is weaker than evidence for a monotonic, positive relationship (Deines et al. 2015), and if a threshold exists it is most likely at a chl a level greater than 80 µg/L.

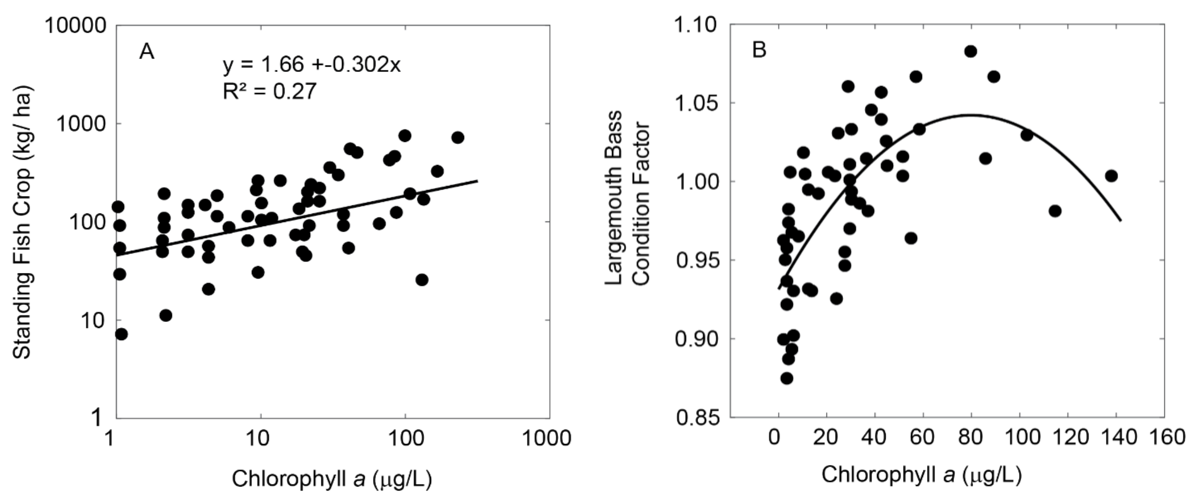


Figure 2.1. Cross lake comparisons of chl a concentration versus total fish standing crop (A) redrawn from Bachmann et al. (1996), and largemouth bass condition factor (B) redrawn from Boucek et al. (2018).

2.3. Chl a Relationships to Toxins

Several genera of bloom-forming cyanobacteria can produce a potent suite of secondary metabolites that are hepatoxic and neurotoxic and can harm aquatic life (Chorus and Bartram 1999). Some freshwater eukaryotes (e.g. *Prymnesium parvum*, Roelke 2016) also produce toxins and have caused massive fish kills in reservoirs of the southeast U.S. but these occurrences are much less common than incidences of toxic cyanobacterial blooms. Microcystins (MCYs) are the most common cyanobacterial toxins measured in freshwaters, and far more is known about MCYs than the other cyanotoxins. There are many congeners of MCYs that vary greatly in their toxicity, but all primarily affect the liver and digestive function. Direct consumption of MCY containing algal cells by feeding on toxic cyanobacteria cells, or by drinking bloom-contaminated waters are the primary exposure pathways for animals (Ibelings and Havens 2008). Acute microcystin exposure causes necrosis of the liver and death (Tencalla et al. 1994). However, the sensitivity of aquatic organisms varies significantly, and organisms from eutrophic freshwater systems where elevated microcystins are more common tend to be less affected by microcystins than those from oligotrophic systems (Malbrouck and Kestemont 2006). Toxins accumulated by zooplankton and bivalve filter feeders can be passed up the foodweb, but MCYs are not known to biomagnify at higher trophic levels (Kozlowsky-Suzuki et al. 2012; Ibelings et al. 2005). Rather, biodilution occurs, and animals at the top of freshwater aquatic food chains (e.g. predatory fish) are least likely to accumulate MCYs to levels that cause liver damage to the fish (Ibelings et al. 2005) or to humans that may eat their flesh (Hardy et al. 2015; Wilson et al. 2008). Emerging evidence indicates that MCYs may also have neurotoxic activity at concentrations lower than those known to cause liver damage. Dissolved MCY concentrations of 0.5 μg/L or prepared in food at 10 ppb have been shown to alter behaviors of fish diurnal swimming activity (Baganz et al. 1998; 2006) and refuge seeking and escape behaviors of crayfish (Clearwater et al. 2014).

Two pieces of information are needed to determine a chl a level that is protective of aquatic life from the threat imposed by MCYs. First, a toxin threshold below which negative impacts are unlikely to occur must be established. Second, a sufficiently strong linkage between chl a and MCY must be established to estimate the chl a level below which MCY concentrations remain below harmful levels. The wide range of susceptibility of aquatic organisms to impacts from MCYs, as well as uncertainties associated with impacts of low level, chronic exposures to MCYs makes establishing a safe MCY level very difficult (Bukaveckas et al. 2017). In water bodies where the dominant bloom forming phytoplankton are MCY producing cyanobacteria, strong temporal and spatial relationships between chl a and MCYs have been documented (Otten et al. 2012; Gagala et al. 2014). For these water bodies, chl a may serve as a useful indicator for toxin related risks to aquatic life (e.g. Otten et al. 2014). However, correlations of chl a with MCYs are usually weak both for studies of individual water bodies (Vaitomaa et al. 2003; Ha et al. 2009) and for intersystem comparisons (Yuan et al. 2014). The general lack of correlation between cyanotoxins and chl a is primarily due to variability in chl a driven by eukaryotes and non MCY producing cyanobacteria (Ha et al. 2009) but additional variation in MCYs relative to chl a is produced by changes in environmental growth conditions (Orr and Jones 1998), and selection of cyanobacterial strains genetically equipped for greater/lesser MCY production (Orr et al. 2004; Otten et al. 2012). Given the difficulties in establishing the necessary threshold MCY concentration for protecting aquatic life or a corresponding chl a value associated with any particular MCY level, designing a chl a criterion to be protective of cyanotoxin exposure for aquatic life would contain a very large amount of uncertainty. Therefore, given current information available, establishing a chl a criterion based on cyanotoxin risk to aquatic life is not advised.

2.4. Chl a and Potable Water Supply Use

High Rock Lake is designated as Class WS-IV (waters protected as water supplies). (See, 15A NCAC 02B .0301). In determining whether a water is suitable as a potable water supply, the physical, chemical, and bacteriological maximum contaminant levels specified by Environmental Protection Agency regulations are used as a guide. In other words, the requirements of EPA's Safe Drinking Water Act are used as a guide to determine the water quality necessary to ensure this use is protected. The North Carolina Administrative Code also provides that the suitability of water supplies are evaluated after treatment. In practice, potable water supplies are evaluated at the point of a potable water intake and take into account the treatment provided in evaluating whether uses are attained in the finished water. At a minimum, these treatment requirements include filtration and disinfection for surface water supplies. (See, 40 CFR 141.70).

The Safe Drinking Water Act establishes primary and secondary standards for contaminants in drinking water. (See, https://www.epa.gov/ground-water-and-drinking-water/national-primary-drinking-water-regulations) The Primary Drinking Water Regulations (Primary Standards) establish legally enforceable contaminant level concentrations and treatment techniques that apply to public water systems to protect public health. Primary Standards include disease-causing organisms, turbidity (an indicator of whether disease-causing organisms may be present), and various chemical substances. The Secondary Drinking Water Regulations (Secondary Standards) are non-enforceable guidelines for regulating contaminants that may cause cosmetic effects or aesthetic effects (taste, odor, and color) in drinking water, but does not prevent its use. The Secondary Standards include chemical contaminant concentrations, color, odor, and other standards.

Under the SDWA, EPA may also publish health advisories for contaminants that are not subject to any Primary Standards. In 2015, EPA developed such health advisories for two cyanotoxins, microcystins and cylindrospermopsin. (See, https://www.epa.gov/cyanohabs/epa-drinking-water-health-advisories-cyanotoxins) EPA also published guidance on managing cyanotoxins in public drinking water systems. (See, https://www.epa.gov/ground-water-and-drinking-water/managing-cyanotoxins-public-drinking-water-systems) This guidance generally discusses cyanobacteria, hazardous algal blooms (HABs) of cyanobacteria, and the potential for cyanotoxins to be present when HABs occur. The guidance notes that HABs can create taste and odor problems in drinking water. Conventional water treatment (coagulation, sedimentation, filtration, and chlorination) can generally remove cyanobacterial cells and low levels of cyanotoxins. Risks associated with HABs can also be reduced through active management of public water systems.

The chl a concentration of water does not directly affect its use as a potable water supply. Rather, chl a or the presence of algal cells would be considered in a similar fashion to secondary drinking water standards. Secondary drinking water standards apply to contaminants that are not health threatening but may affect color, taste and odor, or have other undesirable effects. Conventional potable water treatment facilities include processes to remove algal cells and their associated chl a prior to use. Consequently, even if chl a levels are elevated, adjustments in treatment can generally be made without the need for additional facilities. However, operations and maintenance (O&M) costs may be affected but this is not an impairment of the use.

Source water chl a concentration, at the point of intake to a potable water treatment system, influences the potential cost of treatment to prepare the water for potable use, but does not affect its use as a potable water supply. Treatment requirements for potable water supplies that originate from surface waters, such as lakes and rivers, are highly regulated by USEPA. Under the Safe Drinking Water Act (SDWA), the EPA Office of Water (EPA-OW) is charged with setting water quality standards and regulations to protect the public drinking water supply. These requirements impose treatment strategies at all potable water treatment facilities that are readily able to control particulates (including algal cells). The regulatory basis for these treatment strategies is presented in Attachment B of the pH criteria proposed by the SAC for HRL.

As discussed in Attachment B to the proposed pH criteria, potable water supplies, which use surface water as a source, must provide treatment to settle and filter waterborne disease-causing contaminants, and provide disinfection. The chemicals used in treatment to enhance particulate removal will remove algal cells/chlorophyll before the treated water is provided for use. Additional treatment, such as that required to minimize the formation of disinfection byproducts under the Disinfection Byproducts Rule, would typically require the use of activated carbon to reduce the amount of naturally occurring dissolved organic material. Activated carbon is also very effective in removing taste and odor-causing compounds (2-methylisoborneol (MID) and geosmin) and cyanotoxins. (EPA, 2015)

A review of the literature on chl a concentration necessary to protect drinking water uses yields a mixture of reports that confound chl a with the actual cause of concern. Several of these studies were identified during the meeting of the SAC in April, 2016. The meeting minutes and presentation slides for this meeting identified several literature references related to development of a chl a criterion to protect drinking water uses. These include the following specific references (Table 2.1).

Table 2.1. A Review of Chl a Concentrations Necessary to Protect Drinking Water Uses

Chl a Target (µg/L)	Source/Notes
30	Values above 30 µg/L increase the risk of algal-related health problems. (Heath et al., 1998)
9 – 10 15 – 20 20 – 80	Taste and Odor problems become noticeable Water supply uses impaired Consumptive uses severely impaired (Carney, 1998)
10 50	Relatively low probability of adverse health effects Moderate probability of adverse health effects (assumes cyanobacteria dominance) (Chorus and Bartram, 1999)
15	To keep geosmin < 5 ng/L. (Smith et al, 2002)

A review of these citations shows that the parameter associated with the impairment of the drinking water use is not chl a but some other parameter. Heath et al (1998) and Chorus and Bartram (1999) were primarily concerned with cyanobacteria and cyanotoxins. Carney (1998) and Smith et al (2002) focused on taste and odor issues. These are separate issues that would require a two-step process to generate a chl a criterion for the protection of drinking water (EPA, 2010). The first step involves identification of an impairment threshold for the agent causing the impairment (e.g., cyanotoxin, geosmin). Then the causative agent must be related to chl a concentration. This relationship typically results in low predictive capability.

For example, the State of Illinois prepared a literature review on taste and odor issues in potable water supplies (Lin, 1977; https://www.isws.illinois.edu/pubdoc/C/ISWSC-127.pdf). Taste and odor issues are attributed to chemical substances released by algae during the growth phase of algal cell development, with about 60 species identified as producers of substances leading to taste and odors in water. One such substance, geosmin, is produced by certain algae, including cyanobacteria. In addition, taste and odor problems may also be caused by actinomycetes. This literature review identifies other sources of taste and odor issues, various characteristics of taste and odors, as well as methods for controlling taste and odor issues.

The Kansas Department of Health and Environment (KDHE) prepared a white paper on Chlorophyll-a Criteria for Public Water Supply Lakes or Reservoirs (2011) (http://www.kdheks.gov/water/download/tech/Chlorophylla_final_Jan27.pdf). They note that excessive algal growth can have undesirable effects on drinking water supplies including taste and odor problems, increased levels of cyanotoxins, higher levels of trihalomethane precursors, and increased turbidity levels in source water. Treatment costs for dealing with issues caused by excessive algal growth can be very high. KDHE noted, for example, that the City of Wichita spent $8.5 million on an ozone facility to control taste and odor problems in the Cheney Reservoir, and massive algal blooms have triggered the shutdown of drinking water intakes at several other reservoirs. They conclude, prevention is one of the most cost-effective ways for dealing with nutrient related problems for lakes and reservoirs. Problems associated with excessive algal growth are specific to the types of algae present, but direct counting of algal communities is time-consuming and labor-intensive, while chl a measurement is a good practical alternative for assessing algal biomass. For Kansas reservoirs, taste and odor problems begin occurring once chl a values reach 10 µg/L. KDHE subsequently adopted a chl a criterion of 10 µg/L to protect domestic water supply uses (See, http://www.kdheks.gov/tmdl/download/Unofficial_Copy_SURFACE_WATER_QUALITY_STANDARDS_04.11.18.pdf).

As discussed above, Kansas adopted chl a criterion of 10 µg/L to protect drinking water supplies from taste and odor problems. Taste and odor problems are secondary drinking water standards that do not preclude the use as a potable water supply under the SDWA. This is readily apparent given that the use of High Rock Lake water as a potable water supply for a downstream municipality has not been impaired by chl a concentrations that are significantly higher. Moreover, based on modeling of High Rock Lake, it would be impossible to consistently achieve

10 µg/L as a seasonal mean concentration. Consequently, the application of this criterion to High Rock Lake is not recommended. As described by KDHE, dealing with taste and odor issues is a cost-effectiveness problem. In this case, the cost to lower chl a concentrations in the lake should be weighed against the cost of treatment to provide drinking water from this source.

KDHE also noted the relationship between chl a and the likelihood of cyanobacteria dominance, the occurrence of cyanotoxins, precursors to disinfection byproducts, and turbidity. For these parameters to serve as a basis for setting a chl a criterion, an impairment threshold for the specific condition must be identified and then related back to chl a concentration, with consideration for the removal that occurs during treatment at the water treatment plant. Since these parameters are all subject to removal at the treatment works by the currently mandated treatment processes, the analysis will become a cost-effectiveness evaluation to set an appropriate criterion.

2.5. Chlorophyll a and Recreation Use

Clearer water is valued more highly for recreation than turbid waters (Andradi et al. 2018; Smeltzer and Heiskary 1990), and therefore chl a-rich, turbid waters are generally perceived as having poorer recreational value compared to waters with less chl a (Andradi et al. 2018; Smith et al. 2015; Smeltzer and Heiskary 1990). It is important to recognize, however, that water clarity is also controlled by suspended sediment and CDOM, and it is mainly water clarity rather than chl a that relates to recreational value (Andradi et al. 2018). Waders and swimmers value water clarity because the ability to see the bottom provides increased perception of safety pertaining to physical hazards, a greater perception that the water is "clean", and an increased aesthetic appeal (Angradi et al. 2018). The aesthetic value of low chl a waters also extends to non-contact recreational activities such as boating, fishing, or just lake viewing (Andradi et al. 2018). However, other factors including surround land use (e.g. forested, cleared/ developed shorelines) and abundance of litter play equal roles in a water body's aesthetic appeal (House 1996; Andradi et al. 2018). Aesthetic values are not explicitly protected as a designated use for NC waters but implicitly are protected due to this strong relationship with recreational value. High algal biomass can also generate unsightly scums that may also produce odors, and or toxins. Increasing public recognition of toxin production by some bloom-forming phytoplankton may further strengthen the perception of the safety of recreating in clearer waters. However, as discussed in Section 2.3, the relationships between chl a and toxin production is too uncertain at this time to derive a meaningful, quantitative chl a criterion for High Rock Lake.

Although water clarity is a strong determinant of perceived recreational value, the quantitative water clarity judged by water users to be acceptable for recreation displays strong regional variation that depends on the water clarity to which recreators are accustomed (Andradi et al. 2018; Smeltzer and Heiskary 1990). In regions that generally have high water clarity with Secchi depths extending down 5-10 meters, a lake with a 2 m deep Secchi depth might be judged to have impaired recreational value. At the same time, a lake with a 2 m deep Secchi depth might be judged as having outstanding recreational value in the piedmont of NC where water clarity is generally poor due to a combination of high phytoplankton and suspended sediment. In regions with very poor water clarity, water clarity also becomes a less useful predictor of recreational value (Smeltzer and Heiskary 1990). These regional variations in user perceptions of acceptable water clarity lessen the usefulness of recreational chl a criteria outside of the region where they were developed. When translating survey results across regions, it is important that the average water clarity in the survey region matches the average clarity of the region where the criterion is being developed. Surveys of recreators on eight Texas reservoirs with water clarity similar to North Carolina reservoirs indicated that lakes with annual average chl a values between 35-40 mg/L, about 30% of respondents judged the water quality to be impaired to some degree for recreation (Glass 2006).

2.6. References

Andradi, T.R., Ringold, P.L., Hall, K. 2018. Water clarity measures as indicators of recreational benefits provided by U.S. Lakes: Swimming and aesthetics. *Ecological Indicators* 93: 1005-1019.

Allen, M.S. Greene, J.C. Snow, F.J. Maceina, M.J. DeVries, D.R. 1999. Recruitment of Largemouth Bass in Alabama Reservoirs: Relations to Trophic State and Larval Shad Occurrence. *North American Journal of Fisheries Management* 19(1):67-77.

Bachmann, R.W., Horsburgh, C.A., Hoyer, M.V., Mataraza, L.K., Canfield, D.E. Jr. 2002. Relations between trophic state indicators and plant biomass in Florida lakes. *Hydrobiologia* 470: 219-234.

Bachmann, R.W., Jones, B.L., Fox, D.D., Hoyer, M., Bull, L.A., Canfield, D.E. Jr. 1996. Relations between trophic state indicators and fish in Florida (U.S.A.) lakes. *Canadian Journal of Fisheries and Aquatic Science* 53: 842-855.

Baganz, D., Siegmund, R., Staaks, G., Pflugmacher, S., Steinberg, C.E.W. 2005. Temporal pattern in swimming activity of two fish species (*Danio rerio* and *Leucaspius delineatus*) under chemical stress conditions, *Biological Rhythm Research* 36: 263-276, DOI: 10.1080/09291010500103112

Bayne, D.R., Maceina, M.J., Reeves, W.C. 1994. Zooplankton, fish, and sport fishing quality among four Alabama and Georgia reservoirs of varying trophic status. Lake and Reservoir Management 8: 153-163.

Beganz, D., Staaks, G., Steinberg, C. 1998. Impact of the cyanobacteria toxin, microcystin-LR on behavior of zebrafish, Danio rerio. *Water Research* 32: 948-192.

Biber, P.D., Gallegost, C.L.., Kenworth, W.J. 2008. Calibration of a bio-optical model in the North River, North Carolina (Albemarle-Pamlico Sound): A tool to evaluate water quality impacts on seagrasses. *Estuaries and Coasts* 31: 177-191.

Boucek, R., Barrientos, C., Bush, M.R, Gandy, D.A., Wilson, K.L., Young, J.M. 2017. Trophic state indicators are a better predictor of Florida bass condition compared to temperature in Florida's freshwater bodies. *Environmental Biology of Fishes* 100: 1181-1192.

Bukaveckas, P.A., Lesutiene, J., Gasiunaite, Z.R., Lozys, L., Olenina, I., Pilkaityte, R., Putys, Z., Tassone, S. Wood, J. 2017. Microcystin in aquatic food webs of the Baltic and Chesapeake Bay regions. *Estuarine Coastal and Shelf Science* 191: 50-59.

Carney, C.E. 1998. A primer on lake eutrophication and related pollution problems: Kansas Department of Health and Envirnoment. Bureau of Environmental Field Services.

Chorus, I., Bartram, J. 1999. *Toxic Cyanobacteria in Water: A Guide to Their Public Health Consequences, Monitoring and Management*. London, United Kingdom. E. and F.N. Spon/Chapman and Hall.

Clearwater, S.J., Wood, S.A., Phillips, N.R., Parkyn, S.M., Van Ginkel, R., Thompson, K.J. 2014. Toxicity thresholds for juvenile freshwater mussels Echyridella menziesii and crayfish Paranephrops planifrons, after acute or chronic exposure to Microcystis sp. Environmental Toxicology 29: 487-502.

Cloern, J.E., Grenz, C., Videgar-Lucas, L. 1995. An empirical model of the phytoplankton chlorophyll: carbon ratio-the conversion factor between productivity and growth rate. *Limnology and Oceanography* 40: 1313-1321.

Deines, A.M., Bunnell, D.B., Rogers, M.W., Beard, T.D. Jr., Taylor, W.W. 2015. A review of the global relationship among freshwater fish, autotrophic activity, and regional climate. *Review in Fish Biology and Fisheries* 25: 323-336.

Egerston, C.J. Downing, J.A. 2004. Relationship of fish catch and composition to water quality in a suite of agriculturally eutrophic lakes. *Canadian Journal of Fisheries and Aquatic Science* 61: 1784-1796.

EPA. 2007. Ambient water quality criteria for dissolved oxygen, water clarity, and chlorophyll a for the Chesapeake Bay and its tidal tributaries: 2007 chlorophyll criteria addendum. U.S. Environmental Protection Agency Region III. Chesapeake Bay Program Office. Annapolis, Maryland. November 2007.

EPA. 2010. Using Stressor-Response Relationships to Derive Numeric Nutrient Criteria. https://www.epa.gov/sites/production/files/2018-10/documents/using-stressor-response-relationships-nnc.pdf

EPA. 2015. Recommendations for Public Water Systems to Manage Cyanotoxins in Drinking Water. https://www.epa.gov/sites/production/files/2018-11/documents/cyanotoxin-management-drinking-water.pdf

Gagala, I., K. Izydorczyk, T. Jurczak, J. Pawełczyk, J. Dziadek, A. Wojtal-Frankiewicz, A. Jóźwik & A. Jaskulska,and J.Mankiewicz-Boczek. 2014. Role of environmental factors and toxic genotypes in the regulation of microcystins-producing cyanobacterial blooms. *Microbial Ecology* 67:465–479.

Glass, P. W. 2006. Development of use-based chlorophyll criteria for recreational uses of reservoirs. *Proceedings of the Water Environment Federation* 2006 (8): 4038-4050.

Ha, J.H., Hidaka, T., Tsuno, H. 2009. Analysis of factors affecting the ratio of microcystin to chlorophyll-a in cyanobacterial blooms using real-time polymerase chain reaction. *Environmental Toxicology* 26: 21-228.

Hardy, F.J., Johnson, A., Hamel, K., Preece, E. 2015. Cyanotoxin bioaccumulation in freshwater fish, Washington State, USA. Environmental Monitoring and Assessment 187: 667. DOI 10.1007/s10661-015-4875-x

Heath, R.G., Steynberg, M.C., Guglielmi, R., Maritz, A.L. 1998. The implications of point source phosphorus management to potable water treatment. *Water Science and Technology 37(2): 343–350*.

Heiskanen, J.J., Mammarella, I., Ojala, A., Stepanenko, V., Erkkila, K.M., Miettinen, H., Sandstrom, H., Eugster, W., Lepparanta, M., Jarvinen, J., Vesala, T., Nordbo, A. 2015. Effects of water clarity on lake stratification and lake-atmosphere heat exchange. *Journal of Geophysical Research-Atmospheres* 120: 7412-7428.

Hollister, J.W., Kreakie, B.J. 2016. Associations between chlorophyll a and various microcystin-LR health advisory concentrations. *F1000 Research* 5:151. DOI: 10.12688/f1000research.7955.1

House, M.A. 1996. Public perception and water quality management. *Water Science and Technology* 34: 25-32.

Ibelings, B.W., Bruning, K., de Jonge, J., Wolfstein, K., Pires, L.M.D., Postma, J., Burger, T., 2005. Distribution of microcystins in a lake foodweb: No evidence for biomagnification. *Microbial Ecology* 49, 487-500.

Ibelings, B. W., Havens, K. E. 2008. Cyanobacterial toxins: a qualitative meta-analysis of concentrations, dosage and effects in freshwater estuarine and marine biota. In Hudnell H. K. (ed.), Cyanobacterial Harmful Algal Blooms: State of the Science and Research Needs, Vol. 619. *Advances in Experimental Medicine and Biology*. Springer, New York: 675–732.

Koseff, J.R., Holen, J.K., Monismith, S.G., Cloern, J.E. 1993. Couple effects of vertical mixing and benthic grazing on phytoplankton populations in shallow, turbid estuaries. *Journal of Marine Research* 51: 843-868.

Kozlowsky-Suzuki, B., Wilson, A.E., Ferrao-Filho A.S. 2012. Biomagnification or biodilution of microcystins in aquatic foodwebs? Meta-analyses of laboratory and field studies. *Harmful Algae* 18: 47-55.

Maceina, M.J., Bayne, D.R. 2001. Changes in the black bass community and fishery with oligotrophication in West Point Reservoir, Georgia. *North American Journal of Fisheries Management* 21: 745-755.

Malbrouck, C., Kestemont, P. 2006. Effects of microcystins on fish. *Environmental Toxicology and Chemistry* 25: 72-86.

Manning, N.F., Mayer, C.M., Bossenbroek, J.M., Tyson, J.T. 2013. Effects of water clarity on the length and abundance of age-0 yellow perch in the Western Basin of Lake Erie. *Journal of Great Lake Research* 39: 295-302.

MN PCA. 2014. Minnesota Pollution Control Agency. Site Specific Water Quality Standards. Winona Lake. https://www.pca.state.mn.us/water/site-specific-water-quality-standards

Orr, P.T. and G.J. Jones. 1998. Relationship between microcystin production and cell division rates in nitrogen-limited *Microcystis aeruginosa* cultures. *Limnology and Oceanography* 43:1604–1614.

Orr, P.T., G.J. Jones, and G.B. Douglas. 2004. Response of cultured *Microcystis aeruginosa* from the Swan River, Australia, to elevated salt concentration and consequences for bloom and toxin management in estuaries. *Marine and Freshwater Research* 55: 277–283.

Otten, T.G., H. Xu, B. Qin., G. Zhu, and H.W. Paerl. 2012. Spatiotemporal patterns and ecophysiology of toxigenic Microcystis blooms in Lake Taihu, China: Implications for water quality management. *Environmental Science & Technology* 46: 3480–3488.

Roelke, D.L, Barkoh, A., Brooks, B.W., Grover, J.P, Hambright, K.D., LaClaire, J.W., Moeller, P.D.R., Patino, R. 2016. A chronicle of a killer alga in the west: ecology, assessment, and management of *Prymnesium parvum* blooms. *Hydrobiologia* 764: 29-50.

Rose, K.C., Winslow, L.A., Read, J.S., Hansen, G.J.A. 2016. Climate-induced warming of lakes can be either amplified or suppressed by trends in water clarity. *Limnology and Oceanography Letters* 1: 44-53.

Schupp, D., Wilson, D. 1993. Developing lake goals for water quality and fisheries. LakeLine 13(4): 18-21.

Smeltzer, E., Heiskary, S.A. 1990. Analysis and applications of lake user survey data. *Lake and Reservoir Management* 6: 109-118.

Smith, A. J., Duffy, B.T., Novak, M.A. 2015. Observer rating of recreational use in wadeable streams of New York State, USA: implications for nutrient criteria development. *Water Research* 69: 195-209.

Smith, V. H., Sieber-Denlinger, J., deNoyelles, Jr., F., Campbell, S., Pan, S., Randtke, S.J., Blain, G.T., Strasser, A.A. 2002. Managing taste and odor problems in a eutrophic drinking water reservoir. *Lake & Reservoir Management* 18(4): 319-323.

Stasko, A.D., Johnston, T.A., Gunn, J.M., 2015. Effects of water clarity and other environmental factors on trophic niches of two sympatric piscivores. *Freshwater Biology* 60: 1459-1472.

Tencalla, F., Dietrich, D., Schlatter, C. 1994. Toxicity of *Microcystis aeruginosa* peptide toxin to yearling rainbow trout (*Oncorhynchus mykiss*). *Aquatic Toxicology* 30:215–224.

Vaitomaa J, Rantala A, Halinen K, Rouhiainen L, Tallberg P, Mokelke L, Sivonen K. 2003. Quantitative real-time PCR for determination of microcystin synthetase E copy numbers for Microcystis and Anabaena in Lakes. *Appl Environ Microbiol* 69:7289–7297.

Wilson, A.E., Gossiaux, D.C., Hook, T.O., Berry, J.P., Landrum, P.F., Dyble, J., Guildford, S.J. 2008. Evaluation of the human health threat associated with the hepatotoxin microcystin in the muscle and liver tissues of yellow perch (*Perca flavescens*). *Canadian Journal of Fisheries and Aquatic Sciences* 65: 1487-1497.

Yuan, L.L., Pollard, A.I., Pather, S., Oliver, J.L., D'Anglada, L. 2014. Managing microcystin: identifying national-scale thresholds for total nitrogen and chlorophyll *a*. *Freshwater Biology* 59: 1970-1981.

Yurk, J.J., Ney, J.J. 1989. Phosphorus-fish community biomass relationships in Southern Appalachian reservoirs: Can lakes be too clean for fish? *Lake and Reservoir Management* 5: 83-90.

3. Current Conditions in High Rock Lake

The first two sections of this chapter present chlorophyll a (chl a) conditions in High Rock Lake and relationships between chl a and other parameters of interest such as dissolved oxygen, pH, water clarity. Later sections describe High Rock Lake conditions with respect to algal abundance and species composition, and algal toxins. The final section of this chapter reviews the current state of designated use attainment in High Rock. Separate evaluations of use attainment are provided for fisheries and aquatic life, potable water supply, and aesthetics/swimming. Included with each evaluation is a discussion of how the findings were considered to indicate support or nonsupport of designated uses under High Rock Lake's prevailing chl a conditions.

High Rock Lake is one of the most studied reservoirs in North Carolina. Tetra Tech (2004) summarized the results of six separate water quality monitoring programs conducted by the EPA, NC DEM, a UNC research team, and contractors to Alcoa Power Generating Inc. that took place between 1973 and 2001. These studies had various project objectives and sampling designs, and consistently characterized High Rock Lake as a eutrophic reservoir based on nutrient and chl a concentrations. More recently, the NC Division of Water Resources (DWR) conducted two rounds (2005-2006 and 2008-2010) of intensive water quality investigations that collected "photic-zone" composites (defined as twice the Secchi depth) that were analyzed for chl a and other water quality constituents. Twelve stations (Figure 3.1) across the lake and its tributaries

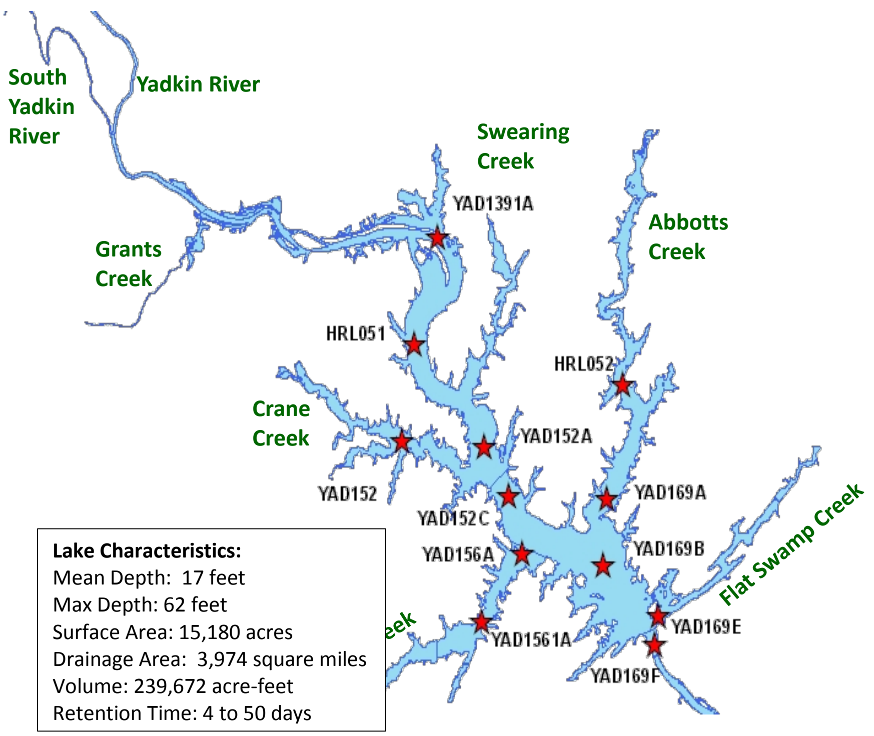


Figure 3.1. DWR Monitoring Stations in High Rock Lake during the 2005-2006, 2008-2010, and 2016 monitoring programs. Not all stations were sampled in all of the monitoring programs.

were sampled as part of the High Rock Lake Scoping Study of 2005-2006 and the 2008-2010 intensive monitoring study. As part of the nutrient criteria development process, an additional round of water quality sampling and analysis was performed the NC Department of Environmental Quality (DEQ) in 2016. Results of these sampling efforts are described in detail in the following sections.

3.1 Spatial and Temporal Patterns of Chl a Concentrations in High Rock Lake

An examination of the spatial and temporal patterns of chl a in High Rock Lake provides a foundation for understanding the algal dynamics within the reservoir. Spatially, the reservoir exhibits a consistent upstream-to-downstream pattern in relative chl a concentrations. A useful conceptual model of the lake is that it operates like many "run-of-the-river" reservoirs that have distinct riverine, transitional, and lacustrine zones. (Figure 3.2). The boundaries separating these zones can shift upstream or downstream with river discharge, and the extent of the zone can expand or contract in response to watershed runoff events, operation of the dam, and other changes within the reservoir that influence the flow and water residence time (Cooke et al. 2005).

The riverine zone is located furthest upstream from the dam where the major river flows into the lake. The riverine zone is characterized by the highest velocity and shortest hydraulic residence time. This region tends to receive relatively high levels of nutrients and particulate matter. The turbidity within this zone limits light penetration so primary production can be influenced by light limitation. The transitional zone is marked by an increase in lake width, which can cause decreased velocity and an increase in residence time. As the water slows, the suspended sediment tends to settle out of the water and deposit on the lakebed. As turbidity decreases, light penetration increases, and irradiance levels in the epilimnion increase. The transitional zone can be a more productive region of the reservoir because light limitation plays less of a role there. Bio-available nutrient concentrations decrease through the transitional zone while turbidity decreases and irradiance levels increase. Controls on phytoplankton production transition from

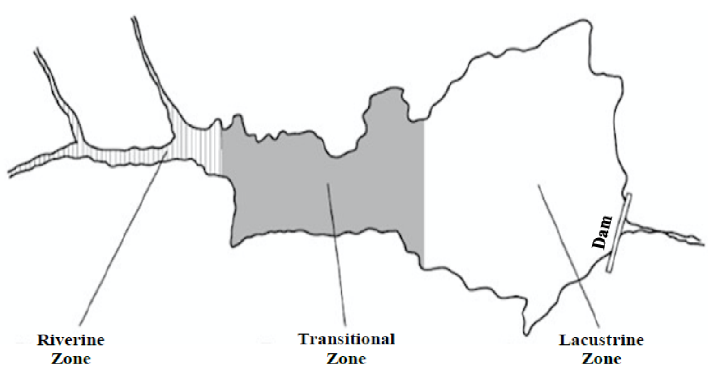


Figure 3.2. Common lake zones (riverine, transitional, and lacustrine) observed for run-of-the river reservoirs, such as High Rock Lake (modified from Cooke et al 2005).

light-limited production in the riverine zone to nutrient limited production in the downstream lacustrine zone (Rudd 2018). In addition, internal nutrient recycling can play a larger role in the transition and lacustrine zones (Cooke et al. 2005).

DWR monitoring stations are located in each of three zones within High Rock Lake (Figure 3.3). Consistent spatial differences have been seen between chl a concentrations located in different zones, for samples collected between 2008 and 2012. Two stations in the transitional zone of the lake frequently exceeded the existing 40 µg/L chl a criterion (Figure 3.4). YAD152C and YAD152 are the sites that have most frequently exceeded the 40 µg/L chl a criterion (Figure 3.5).

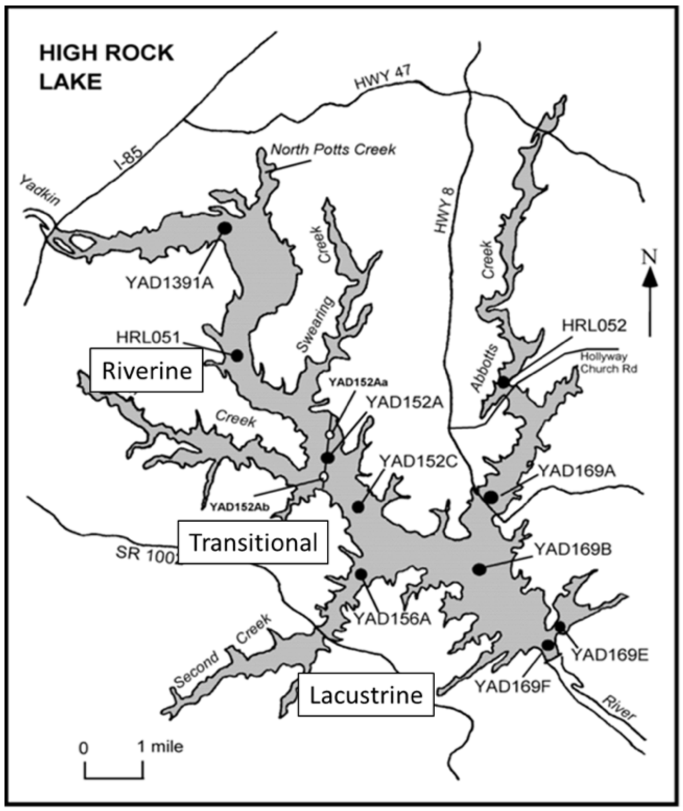


Figure 3.3. High Rock Lake monitoring station locations and lake zones.

Seasonal patterns in chl a are difficult to determine, because the majority of the samples collected over the long-term have been collected during the growing season only. Monthly sampling during 2008-2011 at station YAD152C showed that chl a concentrations were highest during July and August, but could remain relatively high even in December (Figure 3.6). The samples with the highest chl a tended to be dominated by cyanobacteria (Figure 3.6) in terms of number of cells, although other taxa still comprised significant proportions of the algal biomass or biovolume. Samples from a site located in one of the arms of the reservoir (Abbotts Creek) also tended to show higher chl a values during the summer, but in this location high values during September and October were also observed (Figure 3.7).

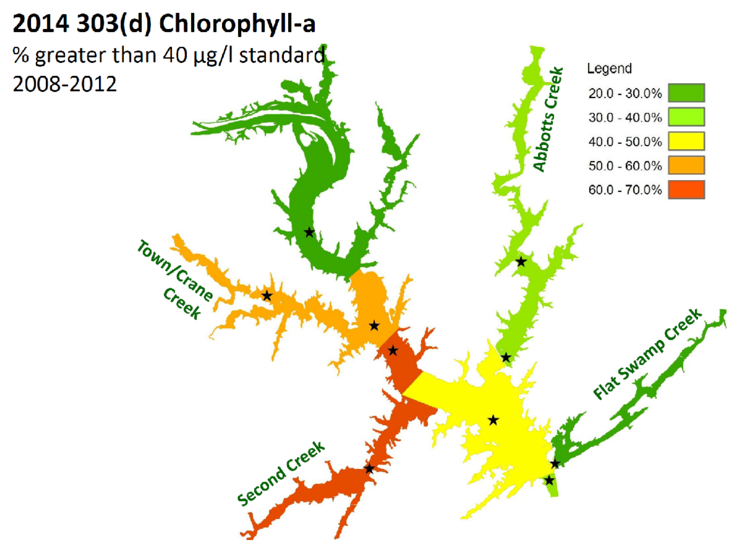


Figure 3.4. Map of percentage of water samples with chl a concentrations greater than 40 µg/L in the time period 2008-2012 in High Rock Lake (from Behm presentation to SAC May 6, 2015).

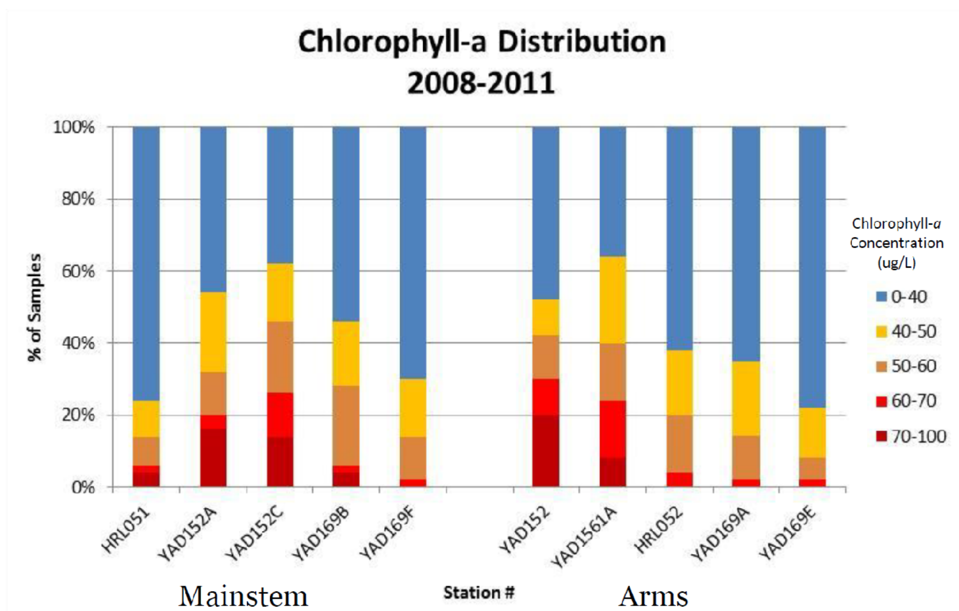


Figure 3.5. Distribution (% of samples) of chl a concentrations across different stations in High Rock Lake sampled between 2008-2011 (from Behm presentation to SAC May 6, 2015).

The variation in sampling frequency over the various High Rock Lake monitoring programs (e.g. monthly, yearly, every five years) makes it challenging to draw conclusions on temporal trends in the monitoring data. There are no clear long-term trends in chl a concentrations (Figures 3.8 and 3.9). Plots are shown for two of the sites with most data over the long-term sampling period (1980-2011). There is no statistically significant trend, examined using linear regression.

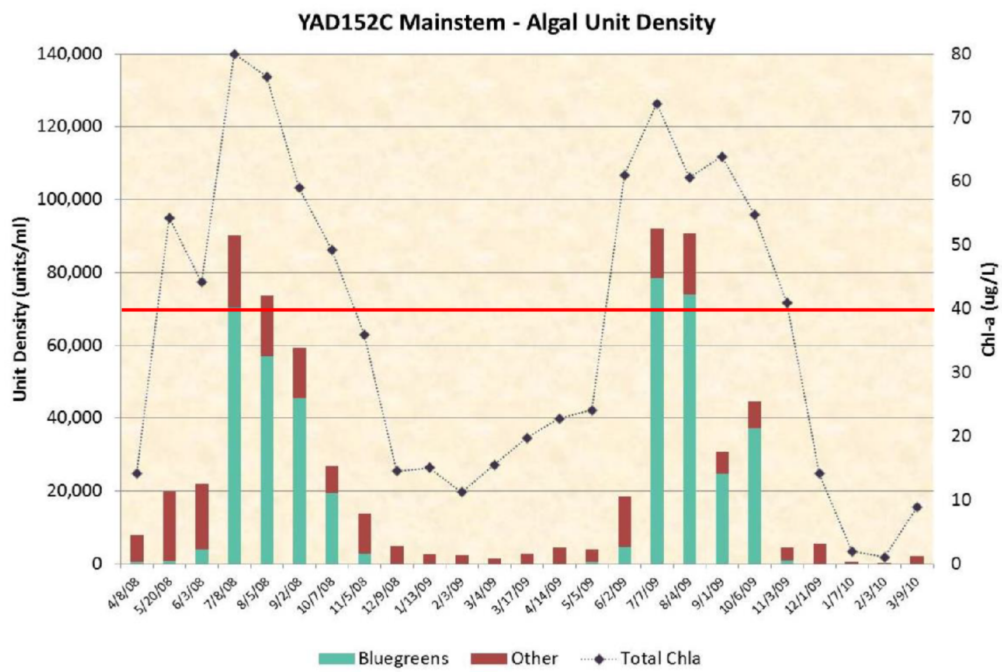


Figure 3.6. Seasonal patterns of chl a in station YAD152C High Rock Lake 2008-2010. (from Behm presentation to SAC May 6, 2015).

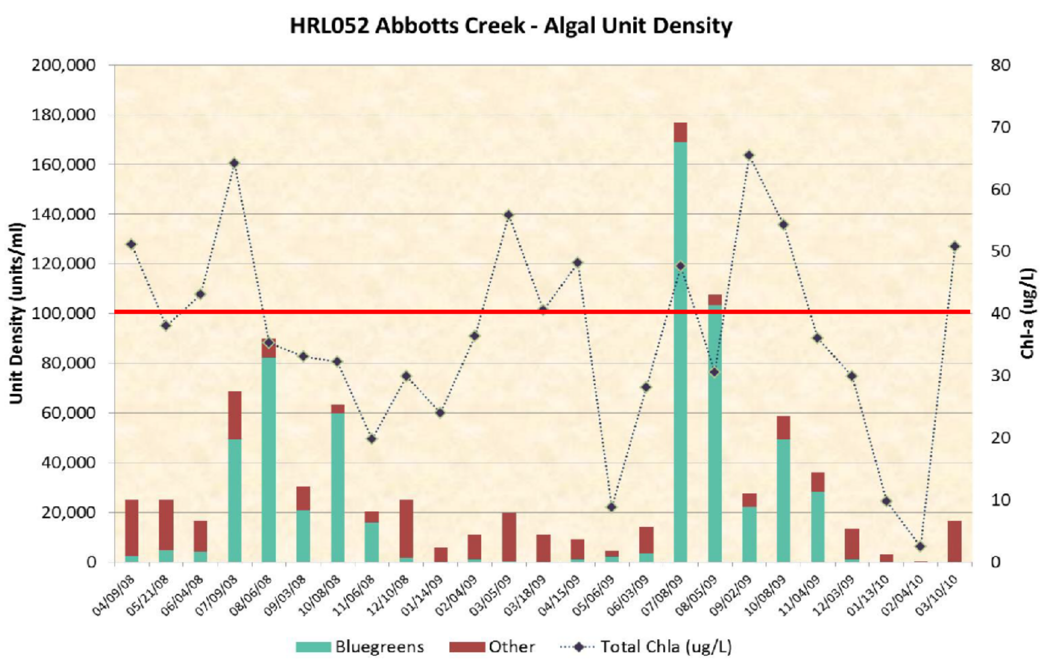


Figure 3.7. Seasonal pattern of chl a in station HRL052 in High Rock Lake 2008-2010. (from Behm presentation to SAC May 6, 2015).

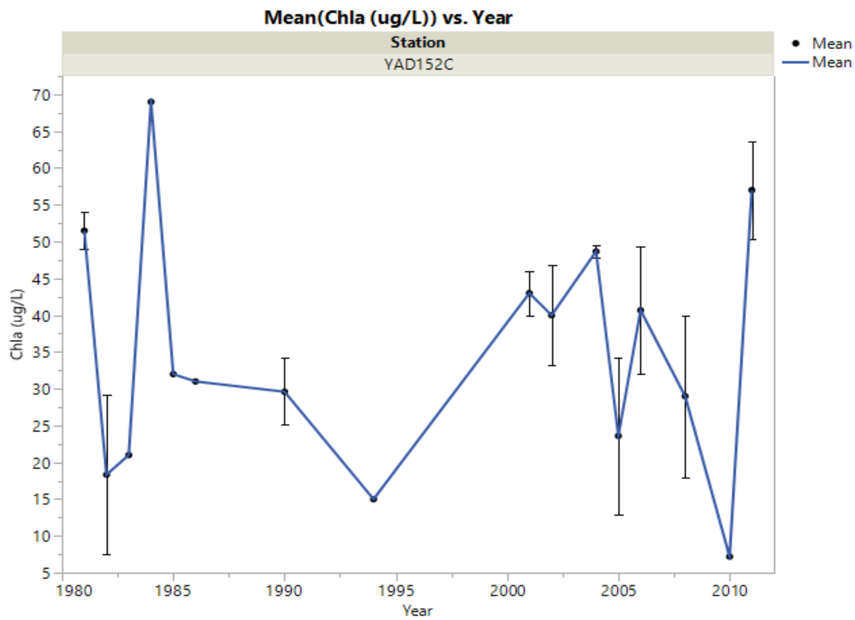


Figure 3.8. Long-term (1980-2011) chl a concentrations in YAD152C station in High Rock Lake. Years in which more than 1 sample were collected were averaged and error bars represent standard error. There has not been a clear increase or decrease in chl a concentration.

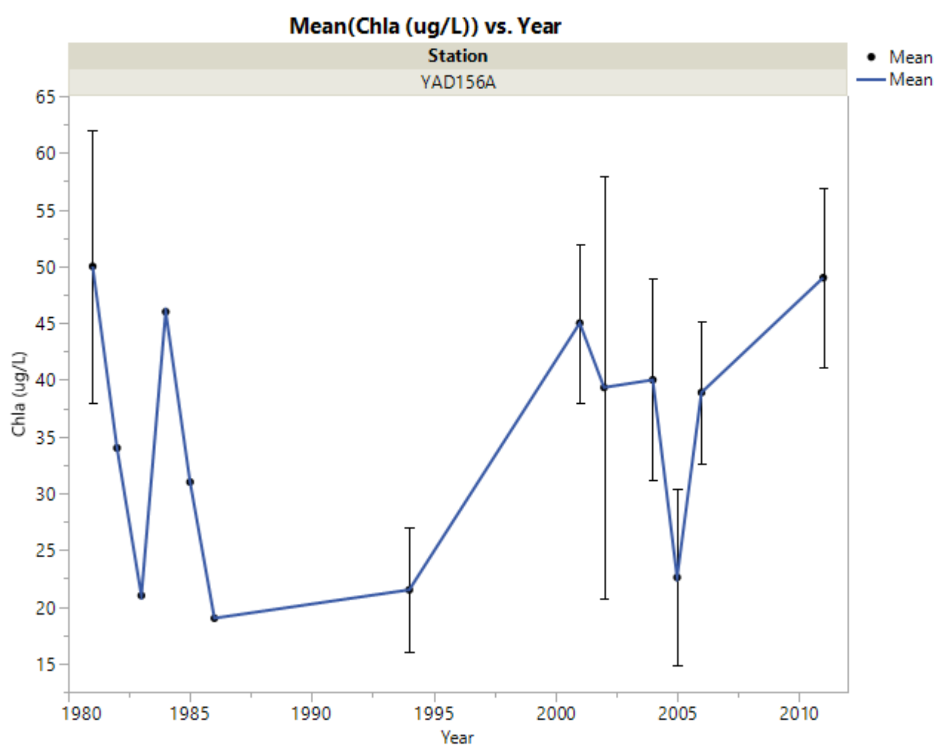


Figure 3.9. Long-term (1980-2011) chl a concentrations in YAD156A station in High Rock Lake. Years in which more than 1 sample were collected were averaged and error bars represent standard error. There has not been a clear increase or decrease in chl a concentration.

3.2 Chl a Relationships with Other Indicators

The subsections below present evaluations of the relationships of chl a with other key parameters such as dissolved oxygen, pH, and water clarity. These parameters are useful indicators of attainment of aquatic life and recreational uses, so their relationship with chl a has direct bearing on the selection of a chl a criterion for High Rock Lake. Specifically, if chl a has a strong relationship with a key parameter, it would be desired to set chl a criteria at levels at which that parameter is within use-supporting ranges, considering both magnitude and temporal aspects of the parameter goals. If a parameter lacks strong relationships with chl a, or the parameter lacks clear thresholds of attainment/non-attainment, it would have less bearing on the chl a criteria selected.

3.2.1 Dissolved Oxygen

High Rock Lake generally experiences favorable dissolved oxygen (DO) concentrations in the epilimnion and is not 303(d)-listed for this parameter. However, DO concentration is one of the most direct indicators of aquatic life support, and so the relationship between chl a and DO should be considered when setting site-specific criteria. The North Carolina Administrative Code (15A NCAC 02B) identifies the DO water quality criteria applicable to High Rock Lake based on the designated uses of the lake. The DO criteria for Class C waters (15A NCAC 02B.0211(6)) provides: for non-trout waters, not less than a daily average of 5.0 mg/l with a minimum instantaneous value of not less than 4.0 mg/l; swamp waters, lake coves, or backwaters, and lake bottom waters may have lower values if caused by natural conditions.

Lin (2015) previously evaluated the relations between chl a and DO in High Rock Lake based on the historical fixed station monitoring record. This evaluation determined that surface DO concentration and DO percent saturation was positively correlated with chl a in spring and summer, but negatively correlated with chlorophyll in the winter (Figure 3. 10). Bottom DO was negatively correlated with chl a in the winter and spring.

The positive correlation between chl a and surface DO in growing season months is expected due to algal photosynthesis, especially considering that most fixed station data were collected during daytime hours. Weaker correlations were detected between chl a and deeper DO. While some of the DO from surface algal photosynthesis can reach hypolimnetic waters by diffusion or advective mixing, increases in organic matter may also increase the decay of algal biomass, thus depleting DO in bottom waters. Some hypolimnetic oxygen depletion is considered a natural process in lakes and reservoirs (such as High Rock Lake) especially during temperature-driven stratification in warm months. For this reason, compliance with DO standards is normally assessed using surface measurements, and the present evaluation did not consider hypolimnetic DO depletion as an impairment of designated uses.

In 2016, the North Carolina Division of Water Quality (DWQ) also deployed monitoring sondes in High Rock Lake to measure short-term variations in chl a, DO, and DO percent saturation, among other variables. The sondes were deployed from July 13 to October 5, 2016. Surface and bottom sondes were deployed at station YAD152C for the entire period, whereas the other sonde

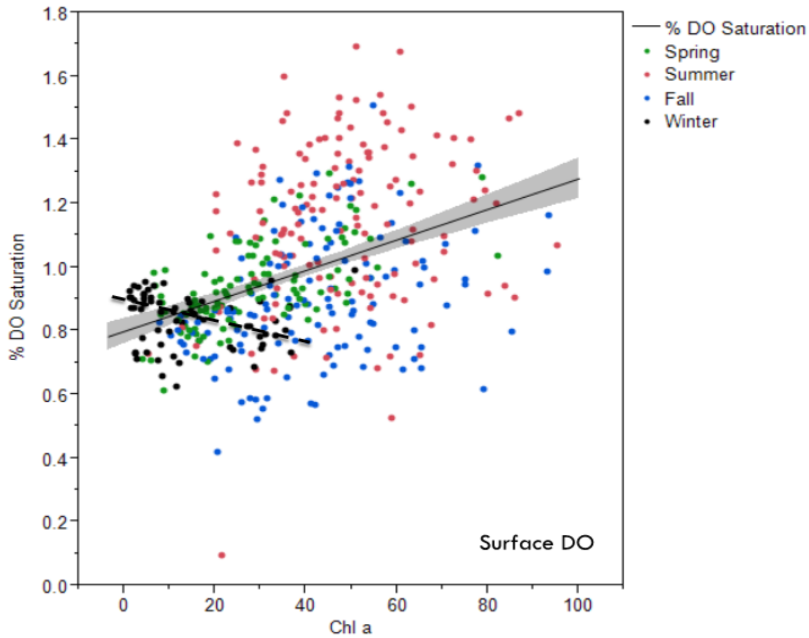


Figure 3.10. Relation between fractional DO saturation and chl a in High Rock Lake fixed station data. Source: Lin (2015).

pair was moved between three stations (YAD169A, YAD169B, and HRL051). The chl a concentrations from the sondes were not similar in magnitude to chl a concentrations measured in grab samples (extraction method), and so are of questionable reliability. However, the sonde data are still considered useful for exploring the DO conditions that High Rock Lake experiences under the prevailing chl a conditions. For reference, chl a concentrations measured in grab samples in July-October 2016 ranged from 11 to 47 µg/L station HRL051, 58 to 75 µg/L at YAD152C, and 31 to 56 µg/L at YAD169B.

The sonde data reveal generally favorable DO concentrations at the surface, with >99 percent of individual measurements above North Carolina's minimum DO criterion of 4 mg/L for Class B waters, and almost 100% percent of daily average DO measurements exceeding the daily average criterion of 5 mg/L (Table 3.1).

Table 3.1. Proportion of High Rock Lake Surface Sonde DO Measurements at or Above DO Criteria

Station	Proportion of Observations ≥ 4 mg/L	Proportion of Daily Averages ≥ 5 mg/L
YAD152C	100%	100%
YAD169A	~98%	100%
YAD169B	100%	~100%
HRL051	~100%	100%
All	>99%	~100%

The sonde data also revealed relatively high diel variability in surface DO concentration (Figure 3.11) and surface DO saturation (Figure 3.12) associated with diurnal cycles in algal photosynthesis and respiration. Table 3.2 presents a statistical summary of the sonde chl a and

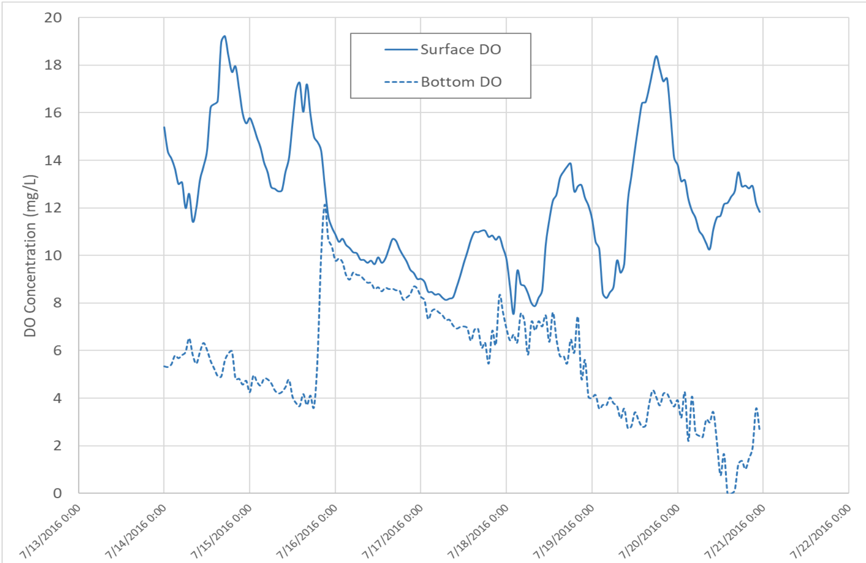


Figure 3.11. Surface and bottom DO concentrations at YAD152C during a portion of the 2016 sonde data collection period.

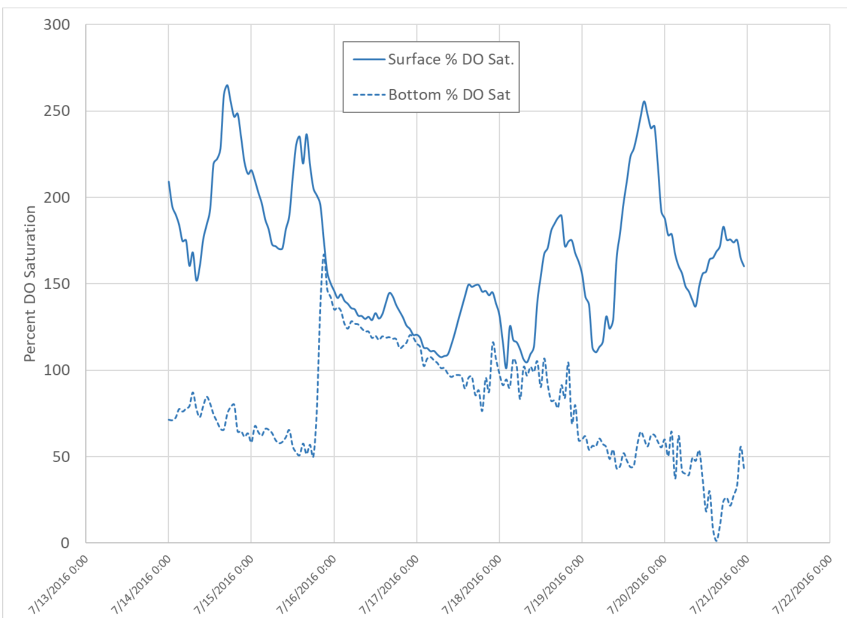


Figure 3.12. Surface and bottom DO percent saturation during a portion of the 2016 sonde data collection period.

DO data by station. The surface DO percent saturation averaged 122% for all the sonde data combined, but exceeded 175% about 10 percent of the time and was less than 71% about 10 percent of the time. The surface DO percent saturation occasionally exceeded 225%, although this occurred in only about one percent of the individual measurements. The single highest DO percent saturation measurement (265%) was observed at station YAD152C.

Chl a was positively correlated with DO concentration and DO percent saturation in both surface and bottom sonde measurements (Table 3.2). The positive correlation with bottom DO demonstrates the possibility of downward diffusion/mixing of high DO at the surface, at

Table 3.2. Spearman Rank Correlation Coefficients of Daily Average Surface Chl a vs DO Metrics [Data source: DWQ 2016 sonde data from High Rock Lake]

DO Metric	Depth Zone	Statistic	n	Spearman Rank Correlation Coefficient	p-value
DO Concentration	Surface	Daily Minimum	164	+0.303	<0.001
		Daily Average	164	+0.371	<0.001
		Daily Maximum	164	+0.404	<0.001
	Bottom	Daily Minimum	145	+0.313	<0.001
		Daily Average	145	+0.472	<0.001
		Daily Maximum	145	+0.512	<0.001
DO percent saturation	Surface	Daily Minimum	164	+0.279	<0.001
		Daily Average	164	+0.345	<0.001
		Daily Maximum	164	+0.375	<0.001
	Bottom	Daily Minimum	145	+0.322	<0.001
		Daily Average	145	+0.472	<0.001
		Daily Maximum	145	+0.504	<0.001

least under certain conditions. North Carolina does not have a water quality criterion for DO percent saturation and utilizes DO concentration criteria to protect against low DO conditions. This approach for protection against low DO conditions is consistent with federal guidance (USEPA, 1986) which states that concentration-based DO criteria are more direct and easier to administer than percent saturation-based criteria and that percent saturation-based criteria could be either over or under protective based on temperature and elevation. North Carolina does have a criterion of not more than 110 percent saturation of total dissolved gas saturation, intended to prevent over-aeration of water and subsequent gas bubble disease in aquatic life, as can occur in hydroelectric dam tailwaters. However, percent saturation of total gases cannot be directly translated to a goal for DO percent saturation, and gas bubble disease is usually caused by excess nitrogen rather than excess oxygen (Weitkamp and Katz, 1980).

The effects of oxygen supersaturation on aquatic life is not as well understood as that of total dissolved gases or nitrogen. Under most circumstances, fish can tolerate short periods of oxygen supersaturation relatively well, partly because (unlike nitrogen) oxygen can be removed from tissue via metabolic activity (Weitkamp and Katz, 1980). However, some studies have attributed gas bubble disease to oxygen supersaturation (Renfro, 1963; McKee and Wolf, 1963; Woodbury, 1942; Lassleben, 1951; Faruqui, 1975), albeit at higher percent saturation values than would apply to nitrogen or total dissolved gases. Mortality has been attributed with DO percent saturation values of 200 – 410%, depending on study. However, other authors point out that despite the frequency occurrence of oxygen supersaturation in eutrophic lakes and aquaculture facilities, fish mortality from oxygen supersaturation is very rare (Boyd and Tucker, 1998).

Chronic effects have been noted at lower DO percent saturations under laboratory conditions when the supersaturated condition was maintained for extended periods. For example, Doulos and Kindschi (1990) found signs of gas bubble disease in cutthroat trout when percent DO saturation was maintained at levels as high as 172%. Espmark and others (2010) found signs of gas bubble disease in Atlantic salmon with continuous, multi-day exposures to DO percent saturation levels of 160 – 220%, and McKee and Wolf (1963) cite a greater incidence of disease in carp exposed to 150% DO saturation, compared with carp exposed to 100-125% DO saturation. Based on these studies, a DO percent saturation of 150% is sometimes cited in the aquaculture literature as the maximum safe level for continuous, long-term exposures. It is unclear if similar chronic effects occur in the field, where conditions of >150% DO saturation tend to be more variable in space and time, and fish can migrate vertically within the epilimnion. High Rock Lake has not been observed to experience fish kills associated with gas bubble disease, and the North Carolina Wildlife Resources Commission reports no signs of gas bubble disease in fish from the reservoir (L. Dorsey, pers. comm., 18 Nov 2015).

In conclusion, the current chl a concentrations in High Rock Lake are correlated with favorable surface DO concentrations, mixed effects on bottom DO concentrations, and relatively high DO percent saturation values under some conditions. The reservoir attains water quality criteria for DO under existing chl a conditions. However, based on the limited scientific literature available, exceedances of 150% DO saturation for extended periods—or 200-250% for shorter periods—might be cited as a reason for concern. Because this parameter correlates with chl a, chl a reduction would probably also reduce the DO percent saturation values and daily variability in this parameter.

3.2.2 pH

The acidity or alkalinity of water as measured by pH is considered a eutrophication-related parameter because algal photosynthesis can elevate pH, especially during the day. North Carolina's existing pH criteria are expressed as range of 6.0 to 9.0 and lack an explicit averaging period or return frequency. North Carolina DEQ's current practice is to only use surface pH measurements to assess reservoirs for pH impairment.

For the present evaluation, variation in the measured pH of surface waters in High Rock Lake was assessed using data collected by NCDWR staff from 1981 to 2016. Monitoring typically includes multiple measurements at different depths at established ambient monitoring stations. On a seasonal basis, the pH of surface waters (<0.2 m) was highest during the months of June through September (days 150-270), and pH exceeded 9.0 in 24-38% of the measurements, depending on month (Figure 3.13, top panel). Exceedances of a pH of 9.0 occurred over the entire range of chl a values, but were more common when chl a exceeded 30 µg/L. The line in the bottom panel of Figure 3.13 connects the median pH value for each interval of 10 µg/L chl a (0-10, >10-20, etc). The median pH value was 8.6-8.9 for chl a concentration intervals greater than 30 µg/L. However, the frequency of pH values greater than 9.0 increased from 21.6% for the >30-40 µg/L chl a interval to 37.5% for the >50-60 µg/L chl a interval. The frequency of pH value greater than 9.0 for chl a intervals below 30 µg/L ranged from 4.8% to 15.2%.

The pH of waters in High Rock Lake varied with depth, consistent with the expectation that maximum rates of photosynthesis occur near the surface of the reservoir. Figure 3.14 displays depth versus pH based on 2011-2016 monitoring, with the dataset filter to only include pH observations at stations and dates on which the chl a concentration exceeded 40 µg/L. For the profiles shown, the maximum pH value occurred near the surface of the reservoir to a depth of about 3 m for some dates and locations. The majority of the water column at the open water stations had a pH below the existing criterion of 9.0 for all profiles of pH reported from the ambient monitoring. Thus, there is available habitat in the mid-depth portion of the reservoir even when the surface reading is >9.0. As part of the evaluation of the pH criterion, the Science

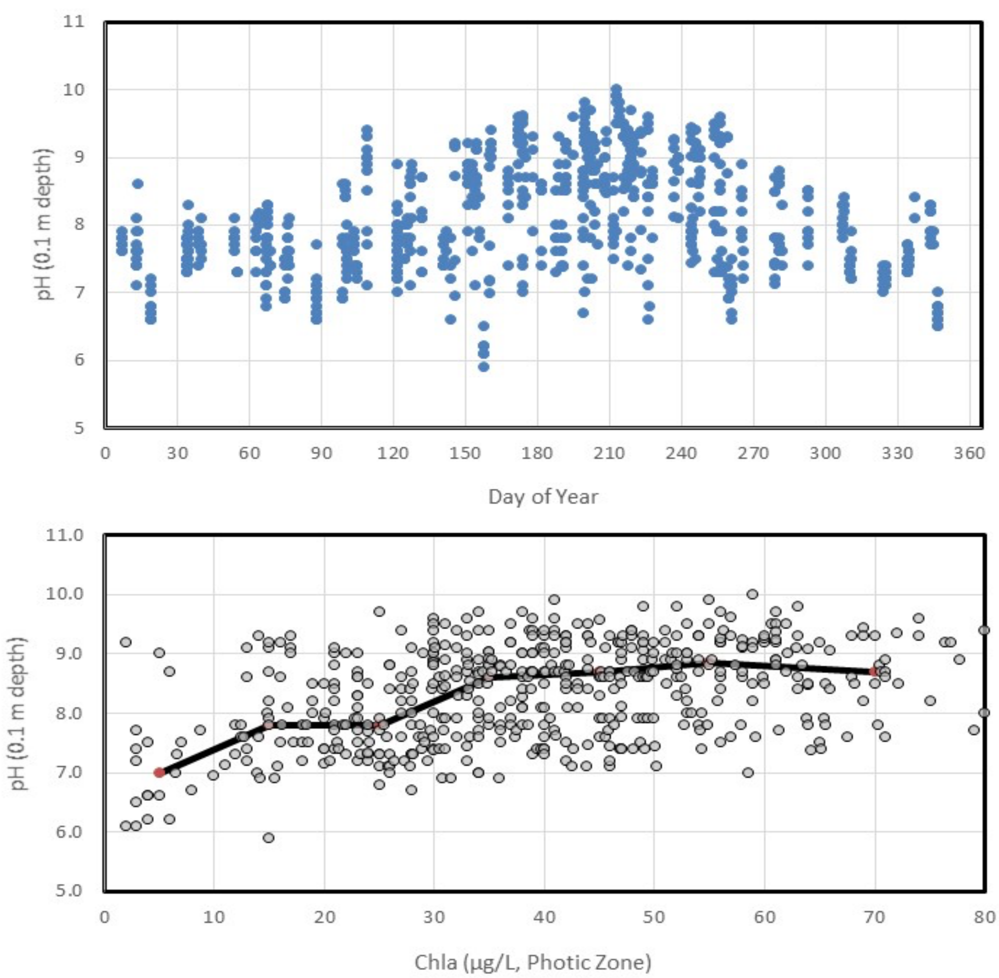


Figure 3.13. Measured pH in the surface layer for 1981-2016 by day of year and by chl a. The line in the lower panel connects the median pH by chl a intervals of 10 µg/L.

Advisory Council evaluated the availability of habitat for aquatic life where pH was below the existing criterion and DO was sufficient (>4 mg/L). Habitat meeting both the pH and DO criteria was available for all dates and locations on which NCDWR conducted ambient monitoring (SAC, 2019). This is relevant to the selection of a chl a criterion, because the oxic zone-average pH could be maintained below 9.0 at moderate to high chl a concentrations, whereas Figure 3.13 would indicate that maintaining the surface pH below 9.0 might not be practicable even with very large chl a reductions.

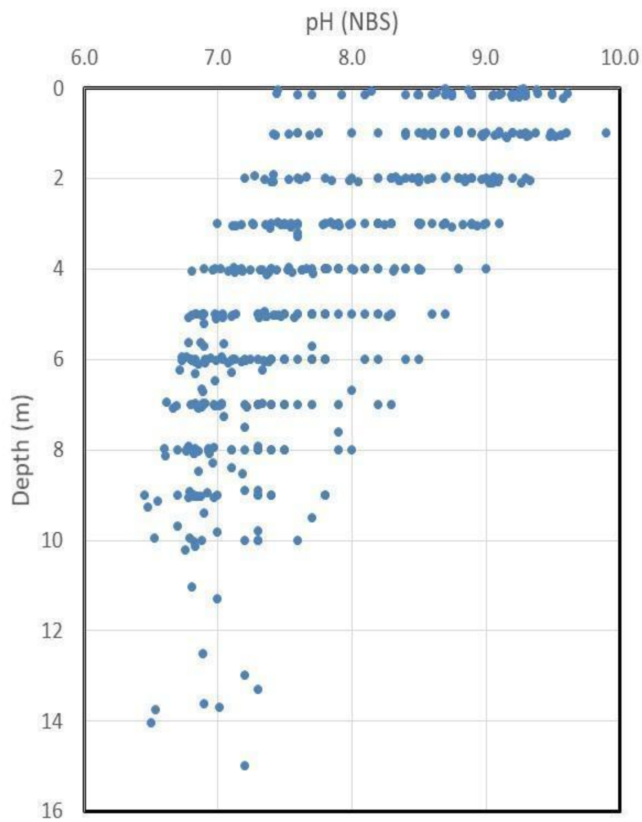


Figure 3.14. Measured pH by depth in 2011 and 2016 for stations with reported chl a > 40 µg/L in High Rock Lake.

3.2.3 Water Clarity

Water clarity is a measure of how deep into the water column light can penetrate. Suspended mineral and organic particles and dissolved organic matter can affect light attenuation in surface waters. Reduced water clarity associated with suspended sediments and algal blooms can affect lake ecosystems by reducing the visual range in water and the light available for photosynthesis. Impacts associated with poor water clarity include reduced visual range (fish feeding), reduced light availability for increased water treatment costs, diminished aesthetics and recreation value, and reduced property values (Dodds et al. 2009 and Borok, 2014). Indicators of water clarity such as turbidity or Secchi depth can be early response variables that can indicate nutrient-related changes to the system, particularly when algal growth affects light penetration. However, because these indicators are also sensitive to suspended mineral sediment, increased turbidity and decreased Secchi depth can also indicate sediment transport from the watershed upstream, particularly during wet weather conditions.

Turbidity is a metric of light scattering by suspended particles that can be used as a proxy for suspended sediment and water clarity. Secchi depth is a direct metric of visual clarity attained by quantifying the depth of transparency in the water column. A Secchi disk is lowered into the water column, and the Secchi depth is recorded as the depth at which the disk is no longer visible. Thus, Secchi depth provides an indication of the transparency of the water column. Secchi depth can be directly relevant to aesthetics, recreational uses, and fish habitat (Davies-Colley and Smith 2001). Turbidity and Secchi depth are typically inversely related, as shown for High Rock Lake (Figure 3.15). Currently, there are no Secchi depth criteria for NC lakes but there is a turbidity criterion (25 nephelometric turbidity units or NTU: https://deq.nc.gov/documents/nc-stdstable-06102019). Based on the relationship between Secchi depth (m) and turbidity in High Rock Lake (2.12 $(\text{Turbidity})^{-0.47}$), a Secchi depth value of approximately 0.47 m or 1.54 ft. would be similar to a turbidity value of 25 NTU, the NC lake turbidity standard (Figure 3.15).

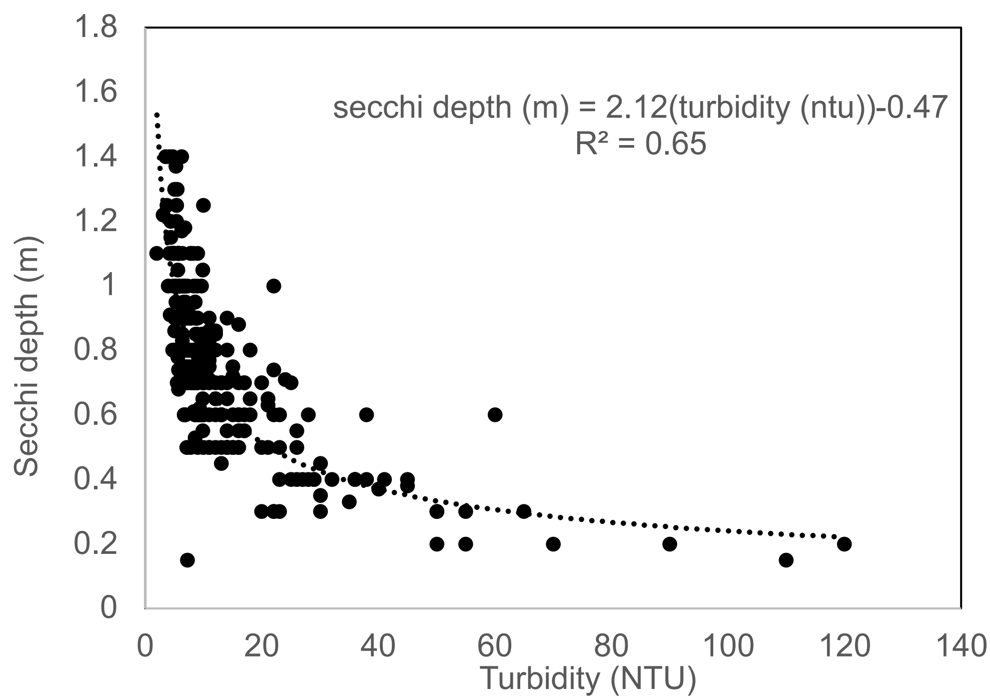


Figure 3.15. Secchi depth (m) vs. turbidity (NTU) in High Rock Lake based on the 2008-2009 and 2016 water quality sampling campaign.

Based on the most recent 2018 NC Category 5 Assessments "303(d) List" (approved by EPA May 22, 2019), the water clarity in High Rock Lake is considered impaired based on turbidity measurements in portions of the lake and its tributaries (Figure 3.16). The Yadkin River and upper portion of the lake, the lower portion of the lake to Second Creek Arm, the Abbotts Creek Arm, Second Creek, and the Yadkin River are listed as impaired for turbidity. The turbidity impairment in High Rock Lake has been partially attributed to sediment loads, although algal growth also contributes to the increased turbidity (Tetra Tech, 2012), particularly in the transitional and lacustrine (downstream) segments of the lake (Rudd 2018).

The most recent assessment of High Rock Lake was based on 2016 data and included Secchi depth and turbidity data for eight stations (HRL051, YAD152A, YAD152C, YAD156A, YAD169A, YAD169B, YAD169E, and YAD169F) with monitoring data collected on 10 dates from May 11, 2016- October 5, 2016 (NC DEQ, 2018) (Figure 3.15). The Secchi depth data for

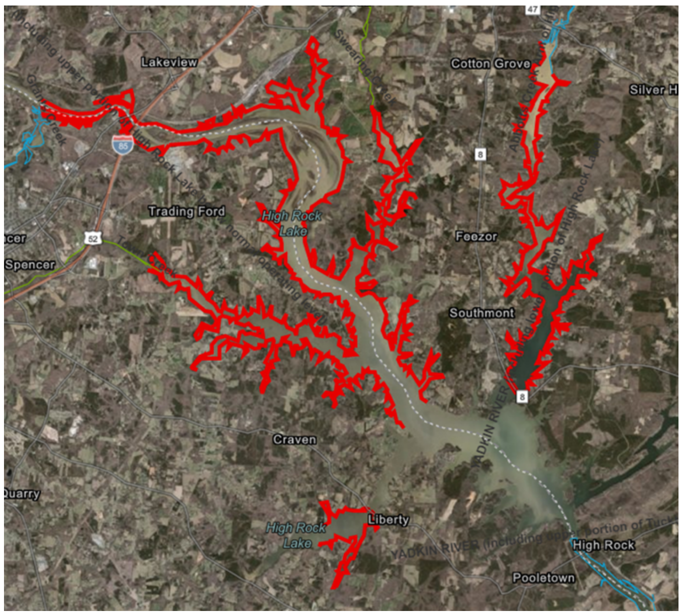


Figure 3.16. Segments of High Rock Lake that are currently listed as impaired due to elevated (> 25 NTU) turbidity based on the 2018 NC 303(d) list:
https://files.nc.gov/ncdeq/Water%20Quality/Planning/TMDL/303d/2018/2018-NC-303-d--List-Final.pdf

High Rock Lake for this period ranged from 0.2-1.3 m, indicating that the clarity of the water ranged from good to poor. The lowest Secchi depths (0.2-0.6 m) were observed at the most upstream sampling site (HRL051), in the riverine segment of the lake. At this site, turbidity averaged 44 NTU and was above the 25 NTU lake criterion for most (8 out of 9) of the sampling dates, except for May 11, 2016, when turbidity levels were 23 NTU. The report stated that the soils in the watershed are highly erodible and high sediment inputs to the lake have resulted in deposition of sediments in the upper section of the lake that have reduced lake depth and affected boat navigation (NC DEQ 2018).

In addition to the lake assessment, the Yadkin-Pee Dee River Basin Ambient Monitoring System Report (NC DEQ 2012) provided a synthesis of turbidity data collected in rivers in the watershed. The NC turbidity criterion for rivers is 50 NTU. This study found that the turbidity criterion was exceeded more than ten percent of the time at 32 of the 103 monitoring stations in the study area. Of the 103 stations monitored, only six stations had no samples that exceeded the 50 NTU threshold. They noted that episodic high turbidity values can often be associated with rainfall events (NC DEQ, 2012). The monitoring data for stations on streams draining to High Rock Lake showed that turbidity in the streams draining to the upper segments of the lake were commonly elevated above the state standard. These data and the recent synthesis by Rudd (2018) suggest that riverine sediment inputs have a large influence on lake water clarity, particularly during storm events and in the upstream segments of the lake near the HRL 051 monitoring site. The literature on run-of-the river reservoirs suggests that reservoirs often exhibit a longitudinal gradient of water clarity from the riverine inflow to the outflow at the dam, as the system transitions from riverine to lacustrine conditions. As discussed earlier, based on this gradient, reservoirs can be divided into three zones: riverine, transitional, and lacustrine (Cooke et al. 2005) (Figure 3.2). This lake zone framework could be useful to categorize High Rock Lake sampling stations (Figure 3.3) and assist with data interpretation of water clarity measurements (see section 4.4.2 for additional discussion on spatial considerations regarding chl a measurements). Longitudinal patterns in water clarity become evident when the turbidity data are plotted versus the distance upstream from the dam (Figure 3.17). The turbidity and data suggest that the uppermost stations: HRL051 and YAD1391A, are in the riverine zone. During high flows YAD152A may also be in the riverine zone. The transition zone generally occurs from YAD 152C until the YAD169A station, where the lacustrine zone begins. However, during extreme streamflow events the riverine and transition zones may extend closer to the dam.

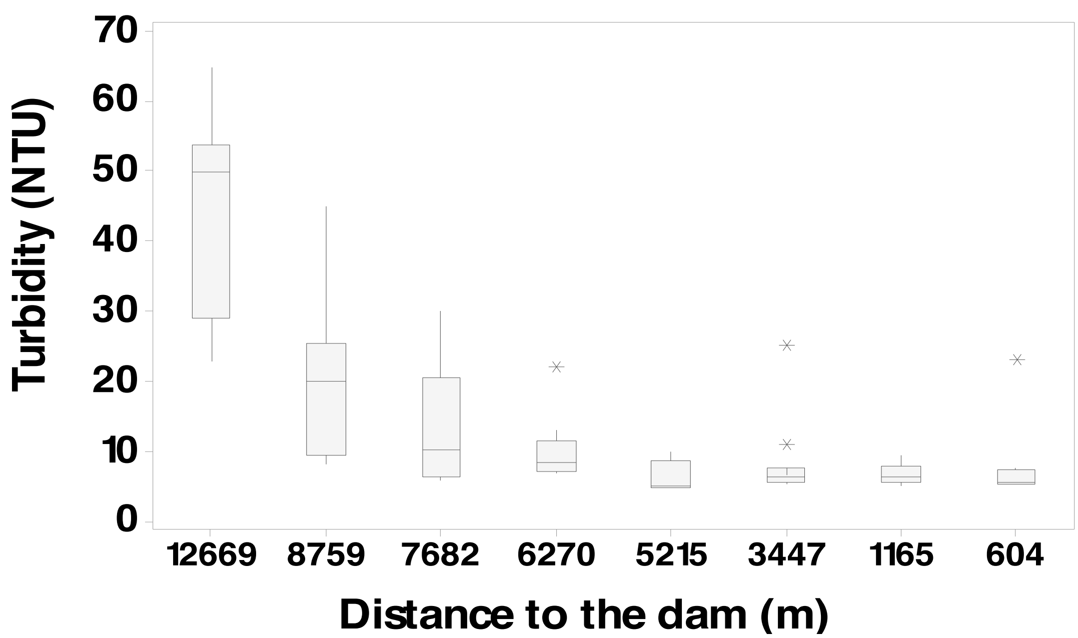


Figure 3.17. Lake turbidity vs. distance to the dam (2016 lake survey data).

In High Rock Lake, the relationship between chl a and water clarity is complex due to variations in nutrient inputs, residence time and the influence of riverine sediment inputs on clarity and light limitation in the riverine and transitional zones (Rudd 2018). The relationship between chl a and water clarity can be more direct in the transitional and lacustrine sections of the reservoir, during time periods when riverine inputs are low and the residence time is longer. For instance, a comparison between chl a concentrations and Secchi depth in the transition and lacustrine zone revealed a decline in Secchi depth with increased chl a concentrations in this zone (Figure 3.18). However, in the riverine zone an inverse relationship between Secchi depth and chl a was present presumably due to the influence of riverine sediment inputs on Secchi depth in that zone. In general, in the transition and lacustrine zones, the Secchi depth was lowest during periods when chl a was elevated, but data from some years show the opposite pattern, presumably due to higher sediment concentration reaching these zones. These data suggest that decreased nutrient concentrations and reduced chl a concentrations can result in an increased water clarity in the lake, but that the improvement potential varies based on year and hydrologic conditions. For example, reducing the chl a from the high of 73 µg/l to the current criterion of 40 µg/l would increase the Secchi depth by approximately 0.3 m, based on the chl a-Secchi depth relation observed in the 2016 303(d) assessment dataset (primarily 2011 data).

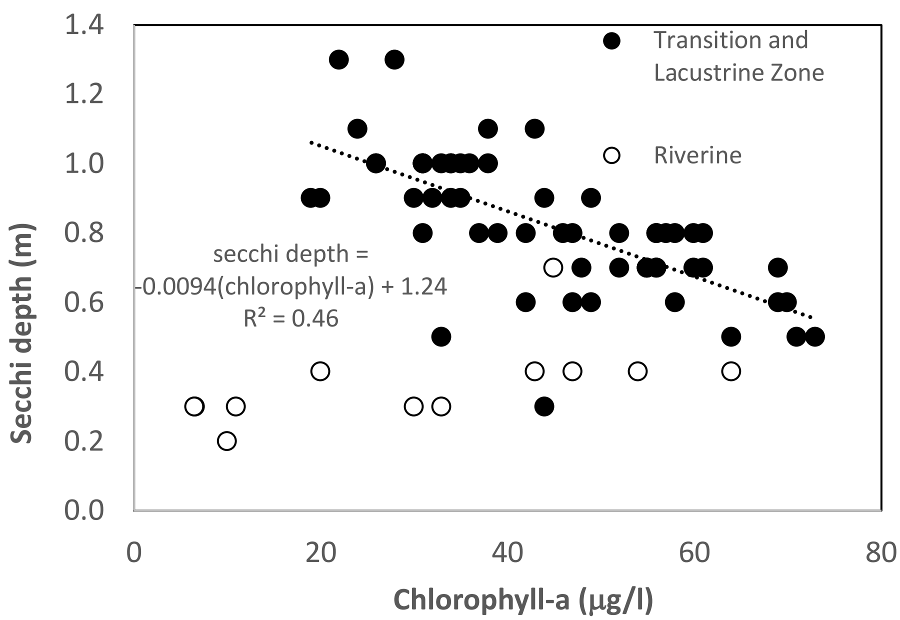


Figure 3.18. A comparison of the relationship between chl a and secchi depth for the upstream riverine zone vs. the downstream transition and lacustrine zones. The data are from the 2016 HRL assessment, which included growing season data from 2011.

Overall, these data suggest that streamflow variations have a strong influence on chl a and water clarity in the reservoir. Riverine discharge and residence time are important variables to consider when developing nutrient criteria for this and other NC reservoirs. In the future, modeling efforts may help to elucidate more of the complex inter-relationships associated with discharge, nutrient concentrations, chl a, and water clarity variability. Because of the influence of low flows on increased residences time and elevated chl a levels, it will be important to understand the role of dam operations and climate change on streamflow to the lake, residence time, and potential influences on chl a exceedances.

3.3 Algal Taxonomy

3.3.1 Background and Rationale

Algal species composition is a potential indirect indicator of use attainment in High Rock Lake. North Carolina defines biological integrity as "the ability of an ecosystem to support and maintain a balanced and indigenous community of organisms having species composition, diversity, population densities and functional organization similar to that of reference conditions" (15A NCAC 02B.02020). This definition lacks a specific meaning for an artificial reservoir for which no reference conditions are available, and North Carolina has not adopted an index of biotic integrity (IBI) for algal assemblages. However, the SAC identified biovolume and algal assemblage as one of the intermediate components of the conceptual model relating nutrients to use impairment, adopted at its February 17, 2016 meeting (Fig. 3.19) (Hall, 2018). Algal data are integral in trend analysis and in the development of NC DWR nutrient response models.

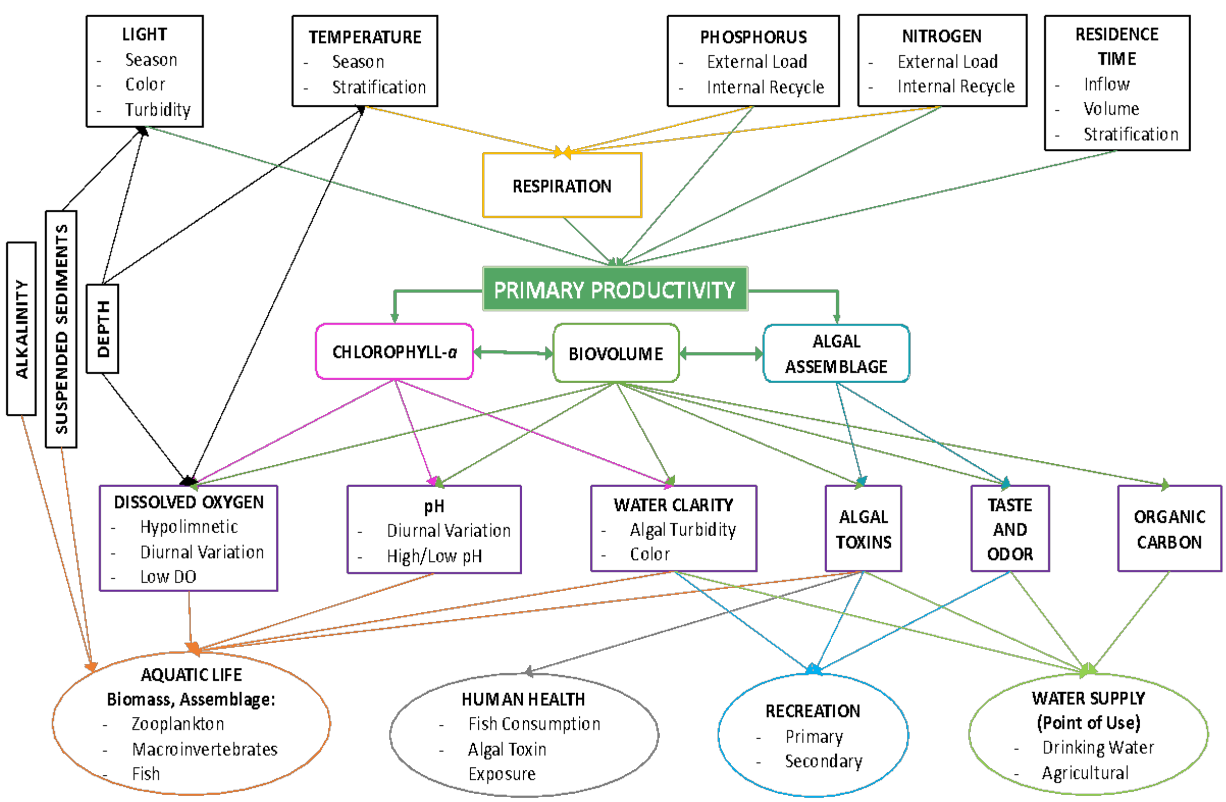


Figure 3.19. Conceptual model relating nutrients to use impairment (NC DEQ, Feb. 17, 2016)

Moreover, an understanding of the qualitative nature of algal blooms is essential for assessment of their potential toxicity (Touchette et al, 2007; Vanderborgh, 2015; Hall, 2018). This section summarizes available information on algal assemblages in High Rock Lake, and how they vary with chl a concentration.

3.3.2 Methods and Sampling Sites

Unless otherwise noted, phytoplankton analyses were performed on whole water samples collected from NC DEQ Ambient Lakes Monitoring Program designated sites on High Rock Lake (Lin, 2015b) (Fig. 3.20). Algal studies were conducted by NC DEQ in the following years: 2004, 2004-2006, 2008-2010, and 2011, encompassing a total of 181 assessments. Additionally, NC DWR staff requested a supplemental analysis of High Rock Lake samples by SAC member, Dr. Linda C. Ehrlich, of Spirogyra Diversified Environmental Services. This analysis was conducted by Dr. Linda C. Ehrlich on samples collected by NC DEQ on August 30, 2017. NC DEQ staff collected whole water phytoplankton samples (fresh and Lugol's iodine-preserved) at the following lake sites: HRL151, YAD152C, and YAD169F. Additionally, a fresh sample was collected in an un-named arm of the reservoir (N35.64.430 W80.28816). Phytoplankton samples are collected according to the standard procedure for Lake Water Sample Collection described in the NC DEQ Intensive Survey Branch SOP document (NC DEQ, 2013).

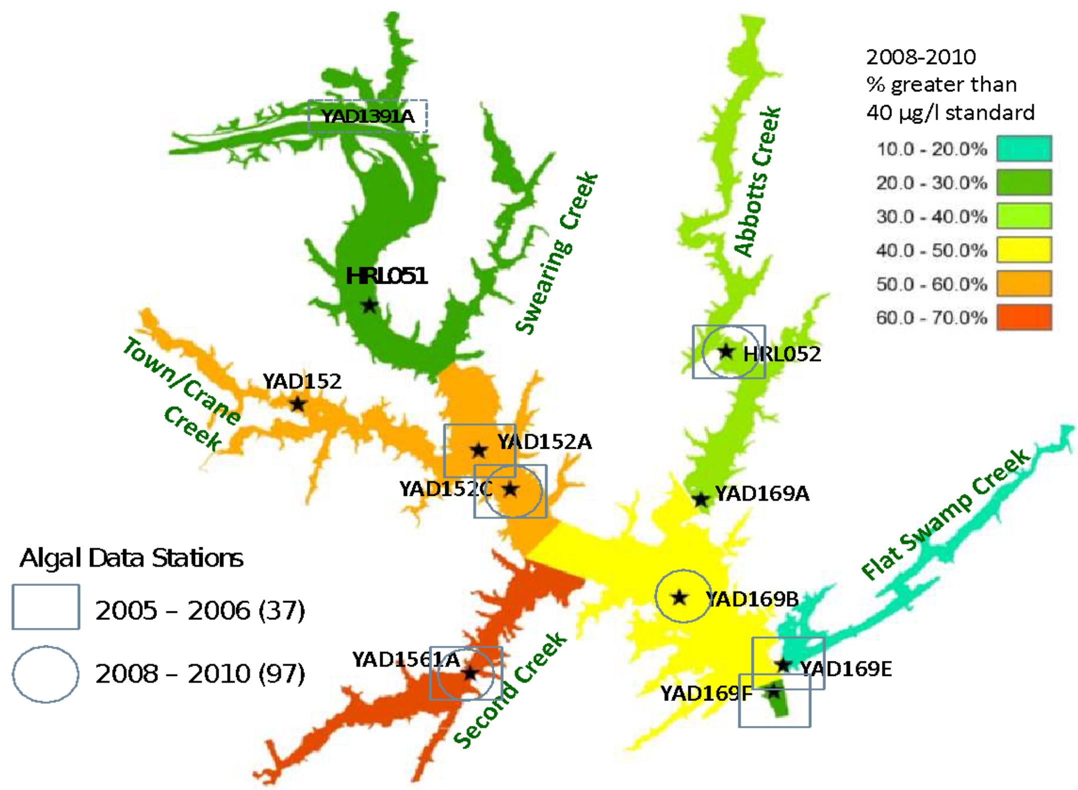
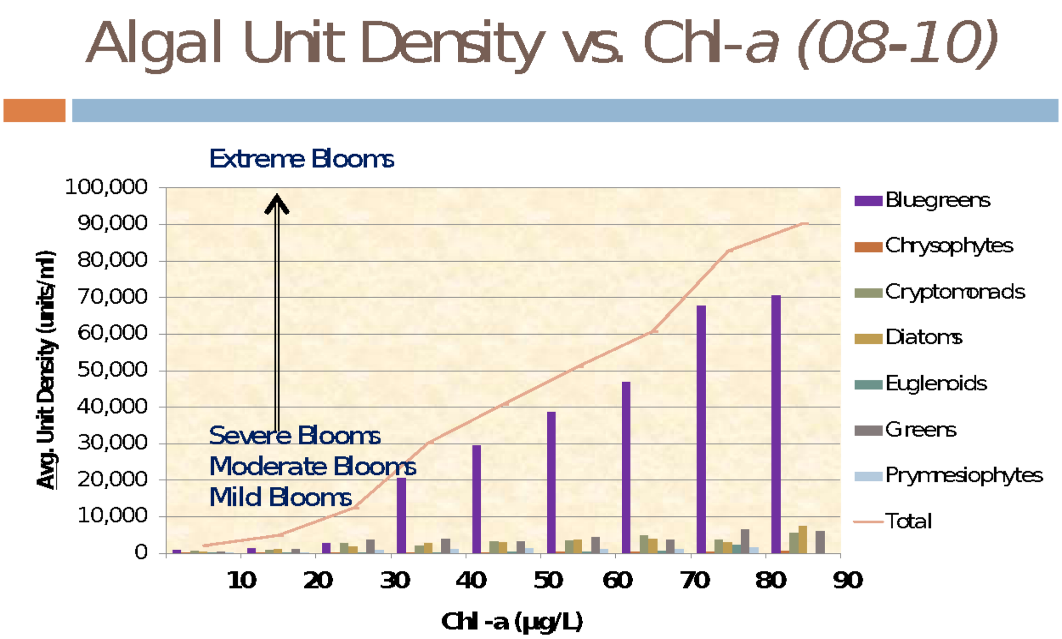


Figure 3.20. Designated NC DEQ algal sampling sites on High Rock Lake (Lin, 2015b).

3.3.3 Results, NC DEQ

Over the totality of its studies, NC DEQ taxonomists documented 140 unique taxa, identified to genus or to species when possible. Although all of the major algal phyla were represented at various levels, the three most commonly observed phyla were the Bacillariophyta, the Cryptophyta, and the overwhelmingly predominant Cyanobacteria (Fig. 3.21) (Vanderborgh, 2015; Lin, 2015b). There was seasonal variance, though summers were consistently dominated by high densities of cyanobacteria (up to 177,000 units/mL), comprising 69% - 96% of the total unit density in July-September. Through the other months, January - March, unit densities were consistently much lower (as low as 100 units/mL), and were dominated by the Cryptophyta, the Bacillariophyta, and Ochrophyta[1] (Chrysophyta), comprising 40% - 50% of total unit density. Figure 3.23 clearly reveals the positive relationship between chl a and cyanobacterial unit density versus the negative relationships for diatoms and green algae (Lin, 2015b).
The most common genera within the Cryptophyta were *Komma* and *Cryptomonas*, whereas, within the Bacillariophyta, the most common genera were centric diatoms and *Synedra*. However, the distinctly most influential genus was the cyanobacterium, *Pseudanabaena*, found in 83% of the assessments, often comprising > 60% of total unit density (Fig. 3.24).

Possible toxigenic cyanobacteria that were observed included *Pseudanabaena* (83% of samples), *Microcystis* (7% of samples), *Aphanizomenon* (17% of samples), *Anabaena* (*Dolichospermum*) (22% of samples), and *Cylindrospermopsis* (44% of samples).



[1] See www.algaebase.org for current taxonomic hierarchies.

Figure 3.21. Algal unit density of the major algal phyla in High Rock Lake, 2008-2010 (Lin, 2015b).

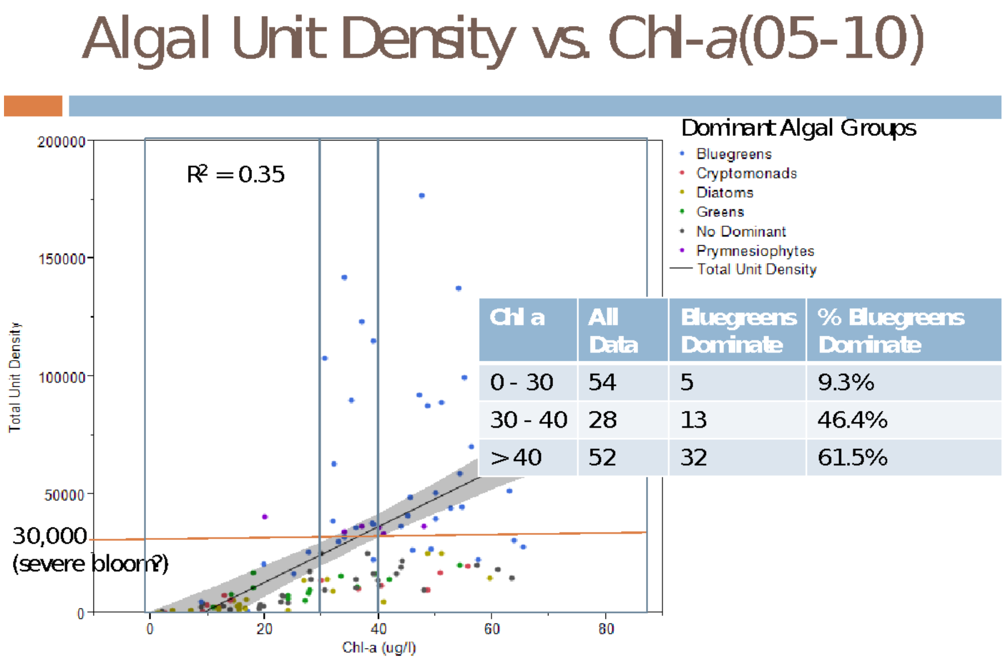


Figure 3.22. Correlation between algal unit density and chl a in High Rock Lake, 2005-2010 (Lin, 2015b).

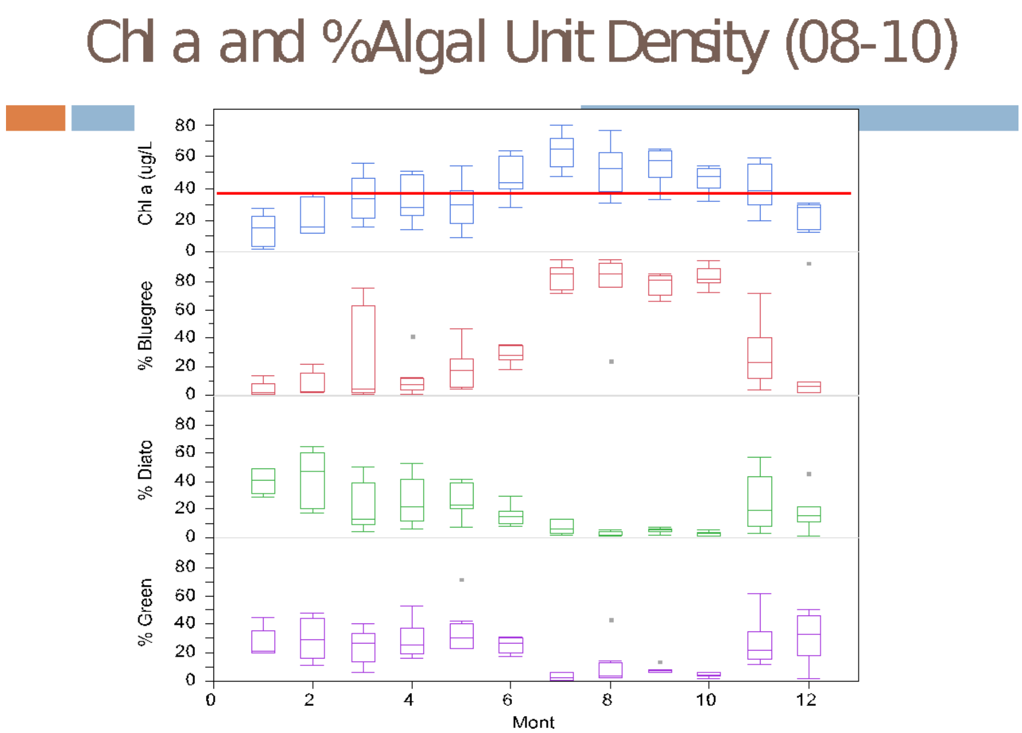


Figure 3.23. Chl a and percent algal unit density in High Rock Lake, 2008-2010 (Lin, 2015b).

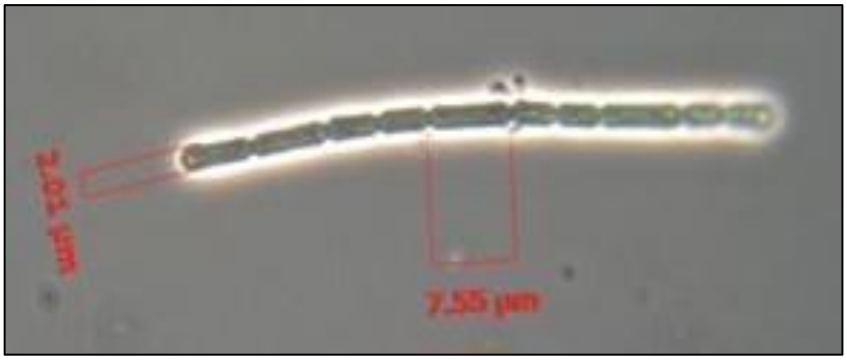


Figure 3.24. The filamentous cyanobacterium, *Pseudanabaena*, 1000X (Spirogyra Diversified Environmental Services, JP Optical).

Because cyanobacteria cells are smaller than those of most other algae taxa, cyanobacteria by density will generally be lower than by biovolume. Analysis of the 2004-2011 NC DEQ algal data by Rudd (2018) revealed that although cyanobacteria were often dominant by density in High Rock Lake, the sum of non-cyanobacteria algal taxa usually comprised the majority of the algal biovolume. Cyanobacteria were a relatively minor component of the biovolume in the samples from the riverine stations (see Fig. 3.3), but on average were over a third of the biovolume in the samples from transitional and lacustrine stations.

3.3.4 Results, Spirogyra Diversified Environmental Services

3.3.4.1 Qualitative Observations

The phytoplankton assemblages at all four sites were mixed, though there was an immediately observable dominance of the filamentous cyanobacterium, *Pseudanabaena limnetica* (Lemmerman) Komarek C, corroborating NC DEQ results (Fig. 3.25). Algal taxa representing all of the major algal groups (phyla), except for the Haptophyta (haptophyte flagellates) were observed at all four sites, including the Cyanobacteria (blue-green algae), Chlorophyta (green algae), Bacillariophyta (diatoms), Ochrophyta (Chrysophyta) (golden algae), Cryptophyta (cryptomonad flagellates), Euglenophyta (euglenoids), and Dinophyta/Pyyrhophyta (dinoflagellates). There were only minor spatial differences in the total numbers of taxa observed, even though there was considerable spatial difference in sediment content, with notably high levels of sediment in the HRL051 sample. There also appeared to be some spatial differences in physiological health of the phytoplankton. Many of the cells in the highly turbid HRL051 sample appeared small, deformed, and chlorotic (reduced green coloration); whereas, cells in the un-named arm sample appeared more robust. Other visibly important, though considerably less abundant, cyanobacterial taxa included *Komvophoron sp*. K. Anagnostidis & J. Komarek, 1988 and *Cylindrospermopsis phillippinensis* (W.R. Taylor).

3.3.4.2 Quantitative Results

There were clear spatial differences in abundance of the three taxa (Fig. 3.25). Abundance was clearly highest at YAD152C and lowest at the highly turbid HRL051. However, abundance was only slightly higher at the low turbidity dam site than at HRL051. At the dam, nutrient limitation may have become influential.

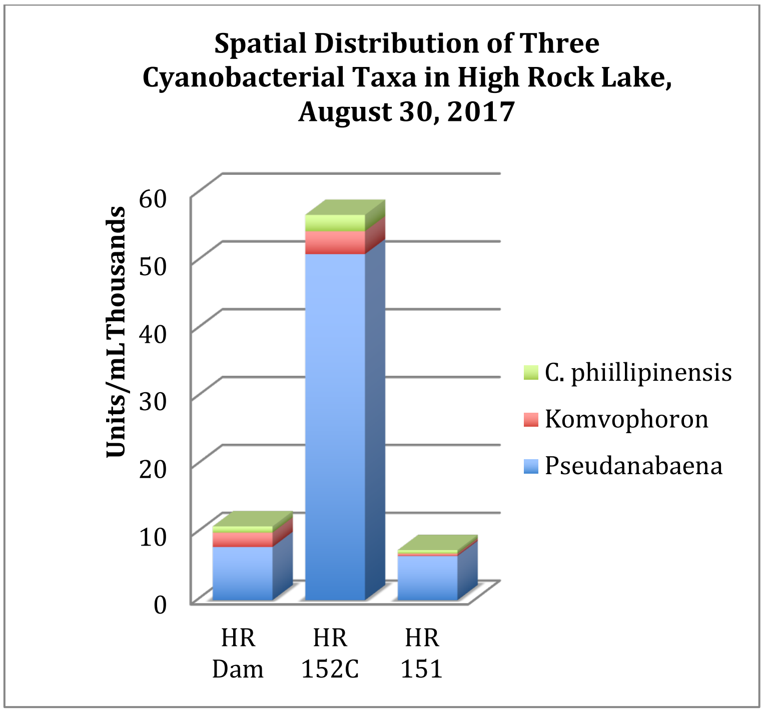


Figure 3.25. Phytoplankton community structure at three sites in High Rock Lake, August 30, 2017 (Spirogyra Diversified Env. Svcs.).

In conclusion, cyanobacteria dominated High Rock Lake's algal assemblage by cell density during the summer, but non-cyanobacteria usually comprised a majority of the biovolume. Cyanobacteria densities and dominance were positively correlated with chl a. The algal assemblage contained several potential toxin formers, and several of these were frequently detected. High cyanobacteria counts are not a direct indication of impairment. Potentially toxigenic cyanobacteria do not always produce high concentrations of toxins, and algal toxin concentrations (addressed in the following section) are a more direct measure of potential toxic effects. Similarly, there is no evidence that the prevailing algal assemblage is incapable of supporting higher tropic levels, and measures of fishery health (addressed in a following section) would be a more direct measure in that regard. Based on relationships such as that shown in Figure 3.21), it can be stated that cyanobacteria are likely to remain a significant component of High Rock Lake's algal assemblage over a wide range of chl a concentrations (30 or 40 µg/L and higher).

3.4 Algal Toxins

At least a dozen cyanobacterial genera have been implicated with toxin production, and at least eight toxin groups have been characterized, of which microcystin (MCY) has been studied most extensively (Cheung et al. 2013). ***However, cyanobacterial abundances (or chl a concentrations) are not reliable indicators for the presence of cyanotoxins since not all species within a genus produce these substances and those that can, do not do so continuously*** (e.g., Kaebernick and Neilan 2001; Loftin et al. 2016). Toxin production can be associated with specific environmental conditions but these conditions are likely species-dependent. For instance, MCY concentrations may be linked to increased dissolved inorganic nutrients (mainly N and P), or more strongly associated with temperature and light levels (Codd et al. 2005; Davis et al. 2009). A recent US-wide survey of over 1,100 lakes showed that at least one of four common cyanotoxins could be detected in 92% of the States; all of which can harm fish, livestock, pets and humans in varying ways (Loftin et al. 2016). Understanding the conditions that favor cyanobacterial growth and/or toxin production is of key importance to guarantee the safe use of freshwater systems and lakes.

For High Rock Lake, the presence and distribution of cyanotoxins was examined in a subset of the water quality sampling stations (Fig. 3.26) during the most recent water quality assessment in summer of 2016 (NC DEQ, 2018). Here, the common toxins that were investigated included MCY, cylindrospermopsin (CYL), anatoxin (ANA), N-methylamino-L-alanine (BMAA) and Saxitoxin (STX). Exposure to MCY and CYL can impair liver function and at high doses be lethal (Carmichael and Boyer 2016; Chorus 2000; Råbergh et al. 1991). ANA and STX are both neurotoxins (Cheung et al. 2013; Falconer and Humpage 2006). ANA causes an overstimulation in neuromuscular junctions, leading to respiratory failure (Falconer 2008). STX is responsible for paralytic shellfish poisoning (PSP), a condition that can cause paralysis and death in humans (Acres and Gray 1978; Kaas and Henriksen 2000). More recently, BMAA has been investigated for its connection to neurological diseases, including amyotrophic lateral sclerosis (ALS), Alzheimer's disease and Parkinson's disease (Banack et al. 2010; Murch et al. 2004).

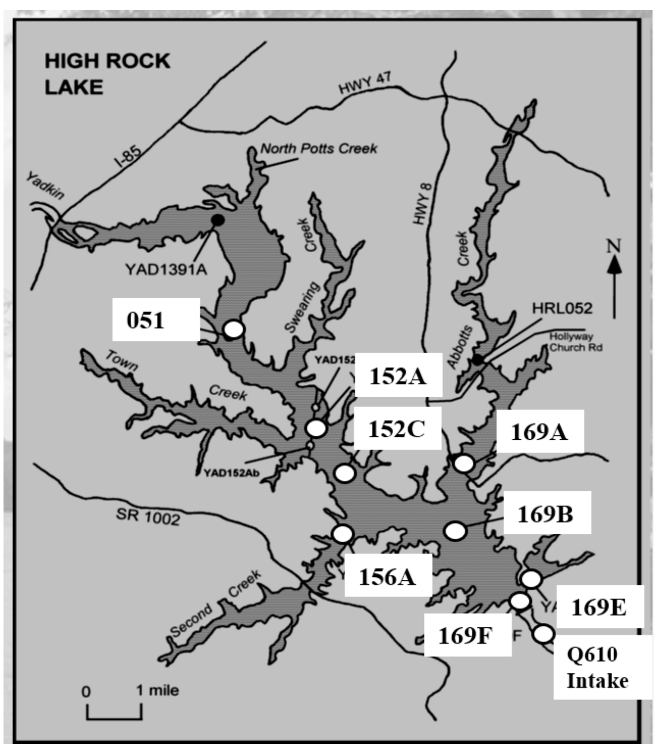


Figure 3.26: Toxins Assessment, Sampling locations.

For the assessment, a combination of *in-situ* toxin tracking devices (Solid Phase

Adsorption Toxin Tracking or SPATT; (Kudela 2011) and the collection of surface water grab samples was used. In contrast to “grabbing” a sample and analyzing for toxins in a finite volume of water at one specific time, the advantages of employing SPATTs comes from their higher sensitivity in detecting low toxin levels via a time-integrative signal. Moreover, SPATTs can be used in freshwater to marine environments, they facilitate testing for multiple toxins (depending on the resin used), and they are easily deployed and recovered (Howard et al. 2018; Kudela 2011; Wiltsie et al., 2018). The disadvantage of using SPATTS is that the method is semi-quantitative and average accumulation values cannot yet be linked to absolute concentrations and therefore health risk guidelines. All cyanotoxin analyses for SPATT extracts and dissolved samples were conducted using Enzyme-Linked ImmunoSorbent Assays or ELISAs (Abraxis Inc.,Warminster, PA, USA). Each toxin kit allows for the detection of a specific suite of congeners and has its specific lower detection limit (LDL): 1) MCY-ADDA (#520011) sensitive to MCY-LR, -YR, -LF, -RR, LW, and nodularin; LDL = 0.10 µg L^{-1}, 2) CYL (#522011) sensitive to CYL and deoxy-CYL; LDL = 0.04 µg L^{-1},) ANA (#520060); sensitive to anatoxin-a and homoanatoxin-a; LDL = 0.1 µg L^{-1}, 4) STX (#52255B; sensitive to STX and other paralytic shellfish poison [PSP] toxins; LDL = 0.015 µg L^{-1}, and (5) BMAA (#520040) sensitive to BMAA and other amino acids; limit of quantitation = 4 µg L^{-1}.

SPATTs were deployed at stations 051 (n =1), 152C (n = 5), 169A (n = 8), 169B (n = 2), 169E (n = 6), and at Q6120 (n = 7) and typically replaced on a biweekly to weekly schedule (Fig. 3.26). Q6120 was located close to the intake for the Denton Water Treatment Plant south of the dam. SPATT sampling revealed that MCY, ANA and CYL were present throughout much of the summer and often detected simultaneously (Fig. 3.27). MCY was found across the lake while CYL and ANA were observed at 4 and 3 out of 6 SPATT locations, respectively (Fig. 3.27).

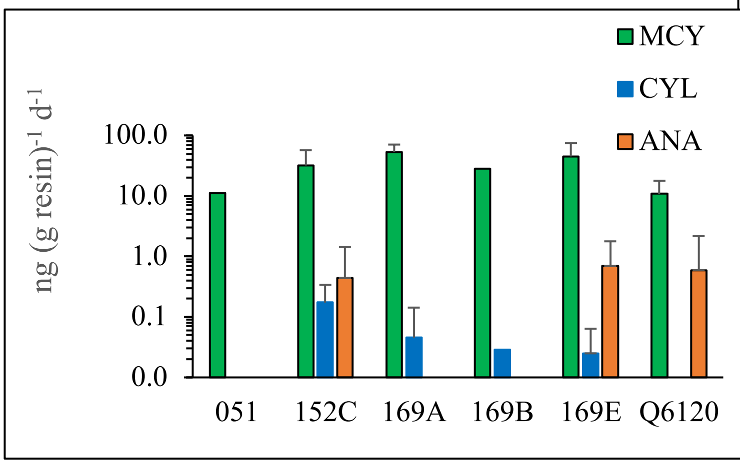


Figure 3.27. SPATT toxin values for MCY, CYL and ANA in ng toxin (g resin)$^{-1}$ d^{-1}. Averages are shown for multiple deployments throughout the summer. Standard deviations (SD) absent if less than 2 observations were made. Note: y-axes is log-transformed due to differing concentration ranges.

In addition to SPATT sampling, grab samples were analyzed for absolute dissolved and intracellular toxin concentrations at each of the stations (shown for dissolved fraction in Fig. 3.28). Running several intracellular extracts for all five toxins did not result in detectable levels for any of the substances (n = 10), despite SPATT data indicating at least the presence of MCY, ANA and CYL for several of the dates and locations. For the dissolved fraction, MCY and ANA could be confirmed at a subset of stations and sampling events (Fig 3.28) but considerable discrepancies between toxin dynamics based on SPATT versus grab samples were indicated due to detection limits. Dissolved BMAA and STX were not present during our study period based on a subset of grab samples (n = 30 across varying sites).

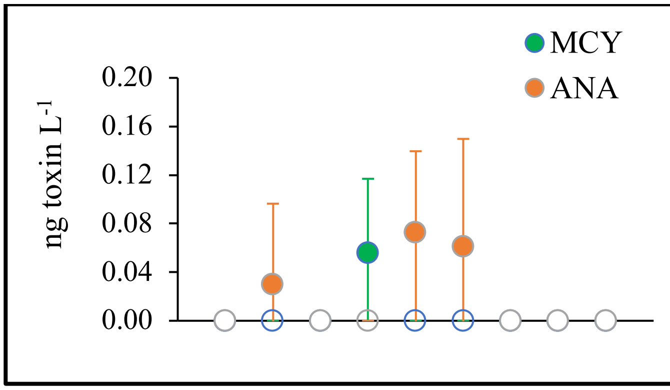


Figure 3.28. SPATT toxin values for MCY, CYL and ANA in ng toxin (g resin) $^{-1}$ d^{-1}. Averages are shown for multiple deployments throughout the summer. Standard deviations (SD) absent if less than 2 observations were made. Note: y-axes is log-transformed due to differences among concentrations for each of the toxin types.

The discrepancies between SPATT and grab sampling are partially explained by continued flow that transports algae and by the "boom and bust nature" of algal blooms since both make grab sampling a "hit or miss affair" compared to *in-situ* tracking. While the dissolved MCY and CYL concentrations (Fig. 3.28) never reached EPA recreational guidelines (https://www.epa.gov/cyanohabs), an increasing number of studies do raise questions about the risks that might be associated with recreational exposure to chronic low-level toxins (e.g., swimming, boating and wading) (Backer et al. 2010; Stewart et al. 2006). This issue together with the potential poisoning of wildlife and humans that consume toxified fish and shellfish (Ibelings and Chorus 2007; Lehman et al. 2010) has yet to be addressed in High Rock Lake.

3.5 Other Indicators of Use Attainment in High Rock Lake

Whereas section 3.2 explored relationships between chl a and specific quantitative indicators, this section examines other useful information on use support in High Rock, including available knowledge on fisheries and aquatic life, potable water supply, and recreation/aesthetics. The types of information presented in this section do not necessarily lend themselves to direct graphical or statistical comparison with chl a concentrations. However, the associated conclusions regarding use support (or lack thereof) can be considered in light of the reservoir's existing trophic status and chl a concentrations, along with other lines of evidence presented in this document. If a use currently appears to be met, it would support the conclusion that the reservoir's existing chl a concentrations are supportive of that use. Conversely, information that a use is not supported could lead to the conclusion that lower chl a concentrations would be beneficial, if a cause-effect linkage between algal biomass and the use can be reasonably assumed. While not a part of the sampling and analysis of pelagic algae presented here, benthic algae are also present in High Rock Lake. At the time of writing this document a bloom of benthic cyanobacteria, *Lyngbya wollei*, has been reported in HRL, which may warrant further assessment in the upcoming years.

3.5.1 Fisheries and Aquatic Life

In HRL, aquatic life is managed primarily to support a sport fishery focused on largemouth bass, striped bass, and crappie, though fishing for sunfish and catfish also occur. Support for the fishery includes ensuring healthy populations of fish that are also safe for human consumption. Based on assessments made by the NC Wildlife Resources Commission (NC WRC), current water quality conditions appear to be supportive of the sport fishery. Table 3.3 summarizes the findings of sportfish population assessments in HRL over the last decade. The largemouth fishery has been consistently evaluated as a "quality fishery" sustained by adequate recruitment and non-excessive mortality. Body condition of young fish has been observed to be lower than ideal but within the normal range for other Piedmont reservoirs. Crappie also showed high abundances with slightly lower than average body condition. Lower average body condition of both crappie and largemouth bass is believed due to intraspecific competition that results from high fish densities (Table 3.3), and therefore, is likely more related to fisheries management than

Table 3.3. Summary of conclusions from fisheries assessments conducted by the North Carolina Wildlife Resources Commission for High Rock Lake over the past decade.

Species (reference)	Survey Year	Fishery status	Growth/ Condition	Recruitment/Mortality
Largemouth bass (NC WRC 2007)	2006	Quality fishery	Relative weight of some year classes not ideal but within normal range for piedmont reservoirs	As expected, and no apparent negative impacts on population
Crappie (NC WRC 2008)	2006	High densities of black and white crappie	Good body condition but somewhat slow growth for black crappie, potentially due to high density and intraspecific competition	Weak recruitment during 2002 during drought
Striped bass (NC WRC 2009)	2006		Fast growth with excellent body condition	Recruitment due to stocking. Few large (> year 3) fish caught, believed due to small gill net size used
Largemouth bass (NC WRC 2011)	2009	Quality fishery	Average growth for piedmont reservoirs. Relative weight of younger fish not ideal, but at or above levels in other Piedmont reservoirs. Suspected cause intraspecific competition from higher than average density	As expected, and no apparent negative impacts on population
Crappie (NC WRC 2012)	2009	Survey catch below normal, suspected cause was high turbidity from high river inputs	Slower than average growth, suspected due to high density and intraspecific competition	
Largemouth bass (NC WRC 2013)	2012	Quality fishery	Relative weights of younger fish slightly less than expected. Suspected due to high density and intraspecific competition	Well balanced age structure. Adequate reproduction and mortality is not excessive

to water quality conditions. As in most NC piedmont reservoirs, striped bass do not reproduce in HRL due to high temperature and low hypolimnetic dissolved oxygen conditions (L. Dorsey, NC WRC personal communication). Annual stocking of 89,000 fingerlings maintain the HRL population of striped bass. The 2006 striped bass survey indicated that striped bass grow fast in HRL and maintain a high body condition for longer than average as they age compared to other piedmont reservoirs. Estimation of the number of older (> 3 year) striped bass abundance has been hampered by sampling biases. Fish kills are uncommon in HRL, and large fish kills have only been noted during the major drought of 2002 when low flows, low water levels, high summer temperatures, and low dissolved oxygen caused major fish kills (L. Dorsey, NC WRC personal communication).

As noted in chapter 2, the relationship between fishery production and chl a is generally positive between 0 and about 100 µg/L (Bachmann et al. 1996; Deines et al. 2015). Currently, chl a averages about 50 µg/L in the most production region of HRL near station YAD152C. Reducing chl a to meet a new criterion may cause some decrease in fisheries production. However, there is a huge degree of variation in the relationship between lake productivity and fisheries, and there are many examples of lakes with highly productive fisheries with chl a concentrations much lower than 40 µg/L. Studies of fisheries in Alabama and Georgia reservoirs have found that chl a concentrations of 10-15 µg/L supported fisheries that were as productive as more eutrophic lakes and also maintained high water clarity desirable for recreation (Maceina et al. 1996; Bayne et al. 1994). The SAC views the risk of a potential modest reduction in fisheries production an acceptable tradeoff for the reduction in risks associated with the current high level of phytoplankton biomass (e.g. potential for cyanobacterial blooms and toxin production).

Harmful effects on fish by cyanotoxins with subsequent consumption by fishermen is also a potential concern, particularly due to the high levels of cyanobacteria biomass. In HRL, this risk has not been fully assessed. Low resolution sampling (monthly) for total MCY (intracellular and dissolved) in summer of 2002 (Touchette et al. 2007) and for accumulated dissolved toxins using a field tracking approach in 2016 (see 3.2.5.) indicated concentrations < 1 µg MCY /L. Limited data on toxin ranges, maxima and temporal dynamics are the presumed reason for a virtual absence of a relationship between chl a and any of the cyanotoxins observed in the southeast US (Chapter 2). Any refinement of chl a criterion, established to minimize the risks posed by cyanotoxins including fish intoxication, will depend on more comprehensive measurements of toxins in lake water as well as animals.

Submerged aquatic vegetation (SAV) is an aquatic life that is commonly protected by chl a criteria. SAV, however, are not present in HRL probably due to a combination of poor water clarity and highly variable water level. High phytoplankton biomass contributes significantly to poor water clarity in HRL with Secchi disk depths rarely more than 1 m (see section 3.3.3). However, high concentrations of suspended sediment also contribute significantly to low water clarity and large fluctuations in water level would likely inhibit SAV colonization in the absence of high phytoplankton biomass due to periodic desiccation of suitable benthic habitats. Lack of existing SAV and a hydrologic regime unfavorable for their development renders a chl a criterion to protect SAV irrelevant for HRL.

3.5.2 Potable Water Supply

High Rock Lake is designated as Class WS-IV (waters protected as water supplies). (See, 15A NCAC 02B .0301). In determining the suitability of waters for use as a source of water supply for drinking, culinary or food processing purposes after approved treatment, the Commission will be guided by the physical, chemical, and bacteriological maximum contaminant levels specified by Environmental Protection Agency regulations. As noted, the suitability of water supplies are evaluated after treatment. In practice, potable water supplies are evaluated at the point of a potable water intake and take into account the treatment provided in evaluating whether uses are attained.

There are no potable water intakes in HRL. Consequently, HRL is not being used as a potable water supply. Consequently, a direct assessment of use attainment is not possible. However, there is a potable water intake located downstream. The Town of Denton Water Treatment Plant (WTP) is located downstream of the dam on HRL and takes its water supply from the Tuckertown Reservoir, the next downstream lake on the Yadkin River. The intake is located only about 0.5 mile downstream of the High Rock Lake dam, and much of the water at that location was recently released from High Rock Lake. The WTP employs conventional water treatment processes including coagulation, flocculation, settling, activated carbon filtration, and disinfection. Although chl a levels in HRL are routinely elevated during the growing season, staff at the Denton WTP do not report that the reservoir has been unavailable as a source for potable water due to chl a level. Rather, the conventional treatment processes have been capable producing a high quality potable water. The Town does report the need to carefully monitor the quality of the raw water supply—especially with regard to turbidity from high flow and seasonal turnover—and adjust treatment processes accordingly.

The chl a concentration of water does not directly affect its use as a potable water supply. Rather, chl a or the presence of algal cells would be considered in a similar fashion to secondary drinking water standards. Secondary drinking water standards apply to contaminants that are not health threatening but may affect color, taste and odor, or have other undesirable effects. Conventional potable water treatment facilities include processes to remove algal cells and their associated chl a prior to use. Consequently, even if chl a levels are elevated, adjustments can be made without the need for additional facilities. Operations and maintenance (O&M) costs may be affected.

Source water chl a concentration, at the point of intake to a potable water treatment system, influences the potential cost of treatment to prepare the water for potable use, but normally does not prevent its use as a potable water supply. Treatment requirements for potable water supplies that originate from surface waters, such as lakes and rivers, are highly regulated by USEPA. Under the Safe Drinking Water Act (SDWA) the EPA Office of Water (EPA-OW) is charged with setting water quality standards and regulations to protect the public drinking water supply. These requirements impose treatment strategies at all potable water treatment facilities that are readily able to control particulates. The regulatory basis for these treatment strategies is presented in Attachment B of the pH criteria document proposed by the SAC for HRL (NC SAC, 2018).

As discussed in Attachment B to the proposed pH criteria, potable water supplies, which use surface water as a source, must provide treatment to settle and filter waterborne disease-causing contaminants, and provide disinfection. The chemicals used in treatment to enhance particulate removal will remove chl a before the treated water is provided for use.

3.5.3 Aesthetics, Swimming

Aesthetic and swimming uses may be adversely affected by chl a concentrations due to the recreating public's perception of color, turbidity, and/or water clarity (Secchi depth) associated with specific concentrations of chl a. Information provided to the SAC suggests that public perception is highly dependent upon the experience of the population using the lake. More generally, the literature shows that public expectations of lake clarity and color have very large regional variations, based on the conditions to which users are accustomed (e.g., Burden and Malone, 1987; Smeltzer and Heiskary, 1988). It can also be reasonably expected that user perceptions would be influenced by the form of algal growth in a reservoir; i.e., highly visible scums or mats could elicit more user complaints than dispersed growths of the same biomass.

In the case of HRL, the SAC is not aware of any aesthetic or swimming use impairment of the lake, even though chl a concentrations routinely exceed 50 μg/L. Most phytoplankton growth in the reservoir is relatively dispersed rather than occurring as highly visible scums or mats, and SAC was not provided with any information to indicate that user complaints are common. In September 2019, the Davidson County Health Department investigated a complaint and confirmed the presence of a benthic cyanobacteria (*Lyngbya wollei*) in the reservoir. Information on the location and extent of the taxa was not available to the SAC, so it could not be determined whether it was restricted to a single cove area versus more widely-occurring. Regardless, because *Lyngbya* is a benthic alga, it would not be directly measured by water column chl a.

The contribution of chl a to water clarity was discussed in section 3.2.3. This section concluded that although water clarity was dominated by suspended sediment in much of the reservoir, chl a reduction from ~70 to ~40 ug/L could cause modest increases (0.1 – 0.3 m) in Secchi depth in parts of the reservoirs in some years or hydrologic conditions.

3.6 References

Acres, J., and J. Gray. 1978. Paralytic shellfish poisoning. Canadian Medical Association journal **119:** 1195-1197.

AlgaeBase.org. 2019. Global algal web database. https://www.algaebase.org/browse/taxonomy/detail/?taxonid=97243&-session=abv4:AC1F036516a2f00ABCLR8F6B32A0 accessed 9/26/19.

American Public Health Association (APHA), American Water Works Association (AWWA), and Water Environment Federation (WEF). 2017. Standard Methods for the Examination of Water and Wastewater. E.W. Rice, R.B. Baird, and A.D. Eaton (eds.). 23rd Edition, Washington, D.C.: APHA.

Bachmann, R.W., Jones, B.L., Fox, D.D., Hoyer, M.V., Bull, L.A., Canfield, D.E. Jr. 1996. Relations between trophic state indicators and fish in Florida (U.S.A.) lakes. Canadian Journal of Fisheries and Aquatic Science 53: 842-855

Backer, L. C. and others 2010. Recreational exposure to microcystins during algal blooms in two California lakes. Toxicon **55:** 909-921.

Banack, S. A., T. A. Caller, and E. W. Stommel. 2010. The cyanobacteria derived toxin Beta-N-methylamino-L-alanine and amyotrophic lateral sclerosis. Toxins (Basel) **2:** 2837-2850.

Bayne, D.R., Maceina, M.J., Reeves, W.C. 1994. Zooplankton, fish, and sport fishing quality among four Alabama and Georgia reservoirs of varying trophic status. Lake and Reservoir Management 8: 153-163.

Borok, A. 2014. Turbidity Technical Review: Summary of Sources, Effects, and Issues Related to Revising the Statewide Water Quality Standard for Turbidity. Oregon Department of Environmental Quality. Portland, OR.

Boyd, C.E., and Tucker, C.S. 1998. Pond Aquaculture Water Quality Management. Kluwer Academic Publishers, Boston MA. 711 p.

Burden, D.G., and Malone, R.F. 1987. A classification of freshwater Louisiana lakes based on water quality and user perception data. Environ Monit Assess. 1987 Sep;9(2):179-93.

Carmichael, W. W., and G. L. Boyer. 2016. Health impacts from cyanobacteria harmful algae blooms: Implications for the North American Great Lakes. Harm. Algae **54:** 194-212.

Cheung, M. Y., S. Liang, and J. Lee. 2013. Toxin-producing cyanobacteria in freshwater: A review of the problems, impact on drinking water safety, and efforts for protecting public health. Journal of Microbiology **51:** 1-10.

Chorus, I. 2000. Health risks caused by freshwater cyanobacteria in recreational waters. Journal of Toxicology and Environmental Health, Part B **3:** 323-347.

Codd, G. A., L. F. Morrison, and J. S. Metcalf. 2005. Cyanobacterial toxins: risk management for health protection. Toxicol. Appl. Pharmacol. **203:** 264-272.

Cooke, D., E.B. Welch, S.A. Peterson, & Nichols, S.A. (2005). Restoration and Management of Lakes and Reservoirs. Taylor and Francis Group. Boca Raton, FL.

Davis, T. W., D. L. Berry, G. L. Boyer, and C. J. Gobler. 2009. The effects of temperature and nutrients on the growth and dynamics of toxic and non-toxic strains of Microcystis during cyanobacteria blooms. **8:** 715-725.

Davies-Colley, R. and Smith, D. 2001. Turbidity, suspended sediment, and water clarity: A review. Journal of the American Water Resources Association 37(5):1085-1101.
Deines, A.M., Bunnell, D.B., Rogers, M.W., Beard, T.D. Jr., Taylor, W.H. 2015. A review of the global relationship among freshwater fish, autotrophic activity, and regional climate. Rev Fish Biol Fisheries 25:323-336.
Dodds, W., Bouska, W., Eitzmann, J., Pilger, T., Pitts, K., Riley, A., Schloesser, J., and Thornbrugh, A. 2009.Eutrophication of U.S.freshwaters: Analysis of potential economic damages. Environ. Sci. Technol. 43 (1): 12-19.
Doulos, S.K., and Kindschi, G.A. 1990. Effects of oxygen supersaturation on the culture of cutthroat trout, *Oncorhynchus clarki* Richardson, and rainbow trout, *Oncorhynchus mykiss* Richardson. Aquaculture Research 21 (1), p. 39-46.
Espmark, A.M., Hjelde, K., and Baeverfjord, G. 2010. Development of gas bubble disease in juvenile Atlantic salmon exposed to water supersaturated with oxygen. Aquaculture 306(1): 198-204.
Falconer, I. R. 2008. Health effects associated with controlled exposures to cyanobacterial toxins, p. 607-612. *In* H. K. Hudnell [ed.], Cyanobacterial Harmful Algal Blooms: State of the Science and Research Needs. Springer New York.
Falconer, I. R., and A. R. Humpage. 2006. Cyanobacterial (blue-green algal) toxins in water supplies: Cylindrospermopsins. Environmental Toxicology **21:** 299-304.
Faruqui, A. M. 1975. Fluctuations in oxygen concentration and occurrence of mortality of carp hatchlings in a hatchery pond at Parta Fish Farm, Bhopal. Broteria Series Trimester Ciencias Naturale 44:67-79.
Hall, W. 2018. Evaluation of Recommendations for High Rock Lake Criteria – Chl a by Clifton Bell (March 13, 2018). Technical document. Hall & Assoc., Washington, DC 9 pp.
Howard, D. A., K. Hayashi, J. Smith, R. M. Kudela, and D. Caron. 2018. Standard operating procedure for Solid Phase Adsorption Toxin Testing (SPATT) assemblage and extraction of HAB toxins.
Ibelings, B. W., and I. Chorus. 2007. Accumulation of cyanobacterial toxins in freshwater "seafood" and its consequences for public health: A review. Environ. Pollut. **150:** 177-192.
Kaas, H., and P. Henriksen. 2000. Saxitoxins (PSP toxins) in Danish lakes. Water Research **34:** 2089-2097.
Kaebernick, M., and B. A. Neilan. 2001. Ecological and molecular investigations of cyanotoxin production. FEMS Microbiol. Ecol. **35:** 1-9.
Komarek, J. and K. Anagnostidis. 2005. 'Cyanoprokaryota II. Oscillatoriales', in Süβwasserflora von Mitteleuropa, Band 19/2, Koeltz, Scientific Books, Koenigstein.
Kudela, R. M. 2011. Characterization and deployment of Solid Phase Adsorption Toxin Tracking (SPATT) resin for monitoring of microcystins in fresh and saltwater. Harm. Algae **11:** 117-125.
Lassleben, P. 1951. Is supersaturation with oxygen dangerous? Fischbauer 2, 105; Water Pollution Abstracts, 25:6.
Lehman, P. W., S. J. Teh, G. L. Boyer, M. L. Nobriga, E. Bass, and C. Hogle. 2010. Initial impacts of Microcystisaeruginosa blooms on the aquatic food web in the San Francisco Estuary. Hydrobiol. **637:** 229-248.

Lin, J. 2015. High Rock Lake Data Review. Presentation to North Carolina Nutrient Science Advisory Council, August 18, 2015. 95 p.

Lin, J. 2015a. High Rock Lake Nutrient Response Model. PowerPoint presentation given at NCDP, August 18, 2015. Division of Water Resources – Water Planning, NC Department of Environment and Natural Resources, Raleigh, 6 pp.

Lin, J. 2015b. High Rock Lake Data Review. PowerPoint presentation given at NCDP, August 18, 2015. Division of Water Resources – Water Planning, NC Department of Environment and Natural Resources, Raleigh, 31 pp.

Loftin, K. A. and others 2016. Cyanotoxins in inland lakes of the United States: Occurrence and potential recreational health risks in the EPA National Lakes Assessment 2007. Harm. Algae **56:** 77-90.

Maceina, M.J., Bayne, D.R., Hendricks, A.S., Reeves, W.C., Black, W.P., DiCenzo, V.J. 1996. Compatibility between water clarity and quality black bass and crappie fisheries in Alabama. American Fisheries Society Symposium 16: 296-305.

McKee, J. E., and H. W. Wolf. 1963. Water quality criteria, second edition. Publication No. 3-A, State Water Quality Control Board; Sacramento, CA.

Murch, S. J., P. A. Cox, S. A. Banack, J. C. Steele, and O. W. Sacks. 2004. Occurrence of β-methylamino-l-alanine (BMAA) in ALS/PDC patients from Guam. Acta Neurologica Scandinavica **110:** 267-269.

NC DEQ, Division of Water Quality. 2012. Yadkin-Pee Dee River Basin Ambient Monitoring System Report. Raleigh, North Carolina: North Carolina Department of Environment and Natural Resources.

NC DEQ. 2018. Lake and Reservoir Assessments: Yadkin-Pee Dee River Basin. North Carolina Department of Environmental Quality. Raleigh, North Carolina.

NC DEQ. 2013. Intensive Survey Branch Standard Operating Procedures Manual: Physical and Chemical Monitoring. Technical Document. NC Division of Environmental Quality, Intensive Survey Branch, Raleigh, NC 132 pp. https://files.nc.gov/ncdeq/Water%20Quality/Environmental%20Sciences/ISU/ISB%20SOP%20Version2.1%20%20FINAL.pdf *accessed 9/26/19.*

NC DEQ 2019. Algal and Aquatic Plant Program. Webpage description of the Ambient Lakes Monitoring Program, Water Sciences Section, NC DEQ, Raleigh. https://deq.nc.gov/about/divisions/water-resources/water-resources-data/water-sciences-home-page/intensive-survey-branch/ambient-lakes-monitoring accessed 9/26/19.

NC SAC. 2018. Recommendations for pH Criteria in High Rock Lake. 44 p.

NC WRC. 2009. High Rock Lake Striped Bass Survey, 2006. Thompson, T. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2009.

NC WRC. 2009. High Rock Lake Crappie Survey, 2006. Thompson, T. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2009.

NC WRC. 2007. High Rock Lake Largemouth Bass Survey, 2006. Dorsey, L. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2007.

NC WRC. 2011. High Rock Lake Largemouth Bass Survey, 2009. Dorsey, L. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2011.

NC WRC. 2012. High Rock Lake Crappie Survey, 2009. Thompson, T. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2012.

NC WRC. 2013. High Rock Lake Largemouth Bass Survey, 2012. Dorsey, L. North Carolina Wildlife Resources Commission Division of Inland Fisheries. Raleigh, NC. 2013.

Råbergh, C. M. I., G. Bylund, and J. E. Eriksson. 1991. Histopathological effects of microcystin-LR, a cyclic peptide toxin from the cyanobacterium (blue-green alga) Microcystis aeruginosa on common carp (*Cyprinus carpio L.*). Aquatic Toxicology **20:** 131-145.

Renfro, W. C. 1963. Gas-bubble mortality of fishes in Galveston Bay, Texas. Trans. Am. Fish. Soc. 92:320–322.

Rudd, M. 2018. An Evaluation of Water Quality Parameters and Flow Dynamics in High Rock Lake, North Carolina to Assist in the Development of Nutrient Criteria for Lakes and Reservoirs in the State. Masters in Environmental Management Report. Duke University, Durham, NC.

Smeltzer, E. and Heiskary, S. A. 1990. Analysis and applications of lake user survey data. Lake and Reservoir Management 6(1), 109-118.

Stewart, I. and others 2006. Epidemiology of recreational exposure to freshwater cyanobacteria – an international prospective cohort study. BMC Public Health **6:** 93.

SWCS Environmental Consultants. 2010. East Canyon Reservoir and East Canyon Creek TMDL. Report prepared for the Utah Division of Water Quality. 304 p

Tetra Tech 2012. High Rock Lake Watershed Model. Tetra Tech, Research Triangle Park, NC.

Tetra Tech 2015. Analysis Report For Classification and Exploratory Analysis of North Carolina Lakes Data for the Nutrient Scientific Technical Exchange Partnership and Support (N-STEPS). Tetra Tech, Research Triangle Park, NC.

Touchette, B.W., Burkholder, J.M., Allen, E.H., Alexander, J.L., Kinder, C.A., Brownie, C., James, J., Britton, C.H. 2007. Eutrophication and cyanobacteria blooms in run-of-river impoundments in North Carolina, U.S.A. Lake and Reservoir Management, 23: 179-192.

USEPA. 1986. Ambient Water Quality Criteria for Dissolved Oxygen. EPA 440/5-86-003. 46 p.

Weitkamp, D.E. and Katz, M. 1980. A Review of Dissolved Gas Supersaturation Literature, Transactions of the American Fisheries Society, 109:6, 659-702.

Wiltsie, D., A. Schnetzer, J. Green, M. Vander Borgh, and E. Fensin. 2018. Algal Blooms and Cyanotoxins in Jordan Lake, North Carolina. Toxins (Basel) **10:** 92.

Woodbury. L. A. 1942. A sudden mortality of fishes accompanying a supersaturation of oxygen in Lake Waubesa, Wisconsin. Transactions of the American Fisheries Society 71:112–117.

4. A Proposed Site-Specific Chlorophyll a Criterion for High Rock Lake

This section presents the SAC's recommendation for a site-specific chlorophyll a (chl a) criterion to protect the designated uses of High Rock Lake from excessive nutrient-driven enhanced primary productivity. The proposed criterion would minimize potential nutrient-driven adverse effects over short- and long-time scales, equating to impacts that are acute and chronic in this man-made reservoir (see Section 4.2.1.). Literature presented in chapter 2 and the reservoir-specific observations in chapter 3 were used to develop the recommended site-specific chl a criterion.

Water quality standards consist of designated uses, parameter-specific criteria to protect those uses, and antidegradation policies. The SAC is not recommending changes to the designated uses or antidegradation policies that currently apply to the waters of High Rock Lake; rather, the focus of this proposal is on a site-specific chl a criterion. Subsections below describe the designated uses of waters of High Rock Lake and recommendations on how the chl a criterion is expressed in terms of the temporal (e.g. duration, frequency), spatial, and magnitude components of the criterion.

4.1 Designated Uses for High Rock Lake

The waters of High Rock Lake are classified in the water quality standards regulations of North Carolina as WS-V Class B waters in upstream reaches or WS-IV Class B waters in downstream reaches (15A NCAC 02B .0309). The Class B designation requires protection of primary and secondary recreation, fishing, aquatic life including propagation and survival, and wildlife (15A NCAC 02B .0219). The water supply designations (WS-IV and WS-V) protect waters as water supplies in moderately to highly developed watersheds. The water supply designations require local programs to control nonpoint sources and stormwater discharges for WS-IV waters and may apply appropriate management requirements in WS-V waters, as deemed necessary, for the protection of downstream receiving waters per 15A NCAC 2B .0203.

The key components of the designated uses for classifications applied to High Rock Lake that may be impacted by nutrient-driven enhanced primary production are primary recreation, fishing/aquatic life, and water supply. For recreational activities, protection of primary recreation activities, which includes swimming on a frequent or organized basis, also would be protective of secondary recreation and fishing activities. Further, protection of primary recreation and aquatic life would be protective of wildlife uses around the margins of High Rock Lake. For the aquatic life use, propagation of species naturally occurring in the man-made system and the overall productivity and diversity of the sport fishery, as an indication of healthy transfer of primary production to apex predators, are the primary considerations. The use of apex predator species as an indicator of overall aquatic life protection was used in the rationale developed for protection of aquatic life in Missouri reservoirs (MDNR, 2017). For this application for High Rock Lake, the productivity and diversity of multiple trophic levels were considered in combination with available information on the site-specific fisheries described in section 4.4.2.

4.2 Temporal Components

The temporal components of how a water quality criterion is expressed include both duration (averaging period, which for chl a focused on seasonal considerations) and allowable frequency of exceedance. These components are discussed in subsections below.

4.2.1 Duration Components

Water quality studies to assess nutrient-driven enhanced productivity in natural and man-made systems have shown that both short-term acute impacts (fish kills, algal toxins, etc.) and long-term enhanced productivity, with potential shifts in the species assemblage present, can occur in different systems (e.g. USEPA, 2000). The development of the temporal component for a site-specific chl a criterion should consider how the key designated uses described above in Section 4.1 may be impacted on an acute and chronic basis. In general, acute effects can be associated with algal toxins or with depletion of dissolved oxygen due to the decay of large algal blooms. For High Rock Lake, the algal assemblage during the growing season often has a high proportion of cells contributed by species of cyanobacteria (see Section 3.2.4). A limited number of measurements to date indicate that algal toxins are present but at a relatively low concentration (see Section 3.2.5). As discussed in Chapter 3, it is important to note that these observations are mainly limited to biweekly measurements of dissolved toxins during the summer of 2016. The sample resolution may not be representative of peak bloom conditions when toxin concentrations (both dissolved and intracellular) tend to reach their maxima nor can any conclusions be drawn in regard to year to year variability. The abundance of algae during the growing season is typically high, and periods of depleted dissolved oxygen in deeper waters of the reservoir have been reported when bottom waters become isolated from surface waters due to thermal stratification (see Section 3.2.1). It is unclear, however, as to the extent that elevated levels of nutrient-driven productivity contribute to dissolved oxygen depletion compared to the thermal isolation of bottom waters during warm season conditions. Due to a lack of clear nutrient-driven acute effects in High Rock Lake, the SAC chose to focus criterion development efforts on longer-term measures of the reservoir's trophic state.

The potential long-term or chronic effects of nutrient-driven enhancement of primary production would be evaluated with a seasonal geometric mean (geomean). The objective of the criterion would be to assess the central tendency of chl a concentrations over time for stations included in each assessment unit. The use of a geomean for the proposed criterion is due to the geomean being the best measure of central tendency for log-normally distributed parameters such as chl a (USEPA, 2010). It is proposed that the geomean be calculated with data collected during the growing season (April-October), as an indication of overall algal production and representative of the time of maximum productivity in High Rock Lake, since chl a concentrations in High Rock Lake are typically higher during the growing season than in other months of the year (see Section 3.2.4). Utilizing data from the growing season is appropriate to assess reservoir trophic status and the general potential for algal-related effects. Overall, the reduction of the long-term central tendency for chl a would also reduce the frequency of elevated chl a values over time.

Use of a geomean statistic to express the proposed chl a criterion is also consistent with approved water quality criteria for chl a in other states. Examples of states that have adopted chl a criteria expressed as a geomean include Arkansas, Florida, Texas, and Virginia. While the current SAC's analysis focused on the current science supporting the development of a geomean criterion, the expression of the criterion as a geomean is also consistent with the historical discussions related to the development of the existing instantaneous chl a criteria for North Carolina.[2]

The proposed chl a criterion is intended to serve as an indicator of average algal growth during the growing season. Therefore, the SAC recommends sufficient data be collected to provide a representative average for the growing season, including samples collected in at least five different growing season months for each year of data included in the analysis. Additional discussion and SAC recommendations on the use of data from more than one year is included in the following section.

4.2.2 Frequency of Exceedance

Water quality criteria have allowable frequencies of exceedance to acknowledge natural variability and the fact that aquatic life can recover from periodic exceedances. Some states have adopted specific allowable frequencies of exceedance for chl a criteria expressed as a geometric mean (geomean). For example, Florida's criteria for lakes, reservoirs, and estuaries may not be exceeded more than once in three years. Florida adopted chl a criteria with an 20 percent probability of exceedance in any given year, and used binomial statistics to demonstrate that a 1-in-3 exceedance frequency would limit the probability of a Type I error (false finding of impairment) to 10 percent (FDEP, 2012).

Similarly, Virginia and Missouri use a version of a once in three-year exceedance frequency approach for chl a criteria in lakes and reservoirs (e.g. 9VAC25-260-187), which is based on a magnitude tied to a single year's computed mean. Minnesota has adopted multi-year average criteria for total phosphorus, chl a, and Secchi depth in lakes and reservoirs (MAR 7050.0222). Water bodies are considered impaired for phosphorus if the phosphorus criterion is exceeded and either the chl a criterion or Secchi depth criterion (or both) are exceeded. Because the criteria are expressed as long-term summer averages, values are computed by aggregating summer data collected over multiple years. Minnesota uses a period as long as ten years for assessments because it provides reasonable assurance that data will have been collected over a range of weather and flow conditions and that all seasons will be adequately represented (MPCA, 2018). All of the criteria components of these approaches have been approved by USEPA.

[2] The chair of the advisory group that recommended North Carolina's existing chl a criterion confirmed the intent of the 40/15 standards were based on "growing season" averages and not any time / any place standards (Mike McGhee, elec. comm., May 10, 2009).

The SAC considered the existing data collection efforts by NCDWR in considering potential frequency approaches for the proposed chl a criterion. For many lakes and reservoirs in North Carolina, monitoring data are collected approximately monthly during the growing season as part of the ongoing ambient monitoring program in a single year during each five-year assessment period. Limited available data with which to assess compliance with a seasonal geomean criterion for chl a presents an obvious challenge to considering a frequency component to the criterion. The most common frequencies used by states are instantaneous or a frequency based on some limited number of exceedances, which as described above, is typical for chl a criteria.

The SAC recommends data incorporated into the assessment be collected in two or more years to incorporate year-to-year variability in chl a concentrations (see Table 4.1). The SAC considered two options to evaluate compliance with the seasonal geomean criterion: (1) computing the geometric mean for each year of individual data and applying a frequency component of not more than one exceedance out of three years of data; or (2) computing a multi-year geometric mean by aggregating data from at least two years within the assessment period. The multi-year geometric mean would be considered a not-to-exceed value. The SAC's criterion discussions did not include an explicit maximum number of years to be included in a calculated multi-year geometric mean. The SAC's agreement from December 2018 cited the use of data from "the assessment period," which corresponds to an implicit maximum of five years. Some SAC members expressed concerns that if multi-year averaging periods were too long, the assessment would have a more difficult time detecting eutrophication-related problems in the reservoir. Some SAC members also discussed the fact that a three-year averaging period would have the closest statistical correspondence to a single-season, 1-in-3 year allowable exceedance approach. The recommendation from the SAC is to utilize the exceedance frequency approach, and recommended a maximum exceedance frequency of no more than one-in-three.

In cases when data are only available for a single year within an assessment period, data from previous assessment periods could be used in order to complete the assessment. This is consistent with North Carolina's existing practice for some other parameters, and the SAC would support this practice up to a total assessment period of 10 years. The SAC also recommends additional sampling be undertaken to add a third year of sampling when the data are needed to assess the maximum one-in-three exceedance frequency. The additional year of sampling would provide nearer term information regarding the current health of the lake to help conclude whether the criterion is met (i.e. only one of the <u>three</u> geometric mean year values exceed 35 µg/L) or not (i.e. two of the <u>three</u> geometric mean year values are greater than 35 µg/L). No additional sampling would be added if both existing seasonal geomean chl a values are below 35 µg/L or both existing seasonal geomean values are above 35 µg/L. This approach is recommended by the SAC in that it adds additional sampling <u>only</u> in instances when the data are needed to assess the one-in-three maximum exceedance frequency.

4.3 Criterion Magnitude

The magnitude component of the chl a criterion is more challenging to derive than for constituents that display a simple dose-response relationship with designated uses. In some settings, development of precise, quantitative relationships between chl a and indicators of designated use impairment may be possible, and a magnitude could be selected to limit identified response indicators from exceeding specific thresholds. However, the SAC's comprehensive examination of relationships between chl a and potential indicators in High Rock Lake (see Section 3) did not identify dose-response relationships upon which a chl a criterion could be based. In fact, High Rock Lake exhibits a combination of favorable indicators and indicators of potential concern. With the understanding that scientific judgment would be required, the SAC adopted the following general approach for deriving a site-specific chl a criterion magnitude for High Rock Lake:

1. An extensive review of literature was conducted to define the ranges of chl a concentration in natural and man-made systems that have been interpreted to be protective of designated uses potentially impacted by a high abundance of algae (see Section 2). This review culminated in the decisions made by the SAC at its December 2018 meeting. [3]

2. The current conditions of High Rock Lake were evaluated, with an emphasis on current chl a levels, on relationships between chl a and indicator parameters, and on evidence for algal-related impacts to designated uses (see Section 3).

3. The results of steps 1 and 2 were synthesized to develop chl a concentration range that was deemed to support designated uses in water bodies similar to High Rock Lake. At the December 2018 SAC meeting, a chl a criterion magnitude was selected from this range. [4]

4. A Monte Carlo analysis was performed to confirm that attainment of the recommended criterion would protect the reservoir's fishery and result in a low rate of exceedance of the upper end of the acceptable chl a range.

The results of steps 3 (range derivation) and 4 (Monte Carlo analysis) are provided in the following subsections along with the specification of the SAC recommended criterion magnitude.

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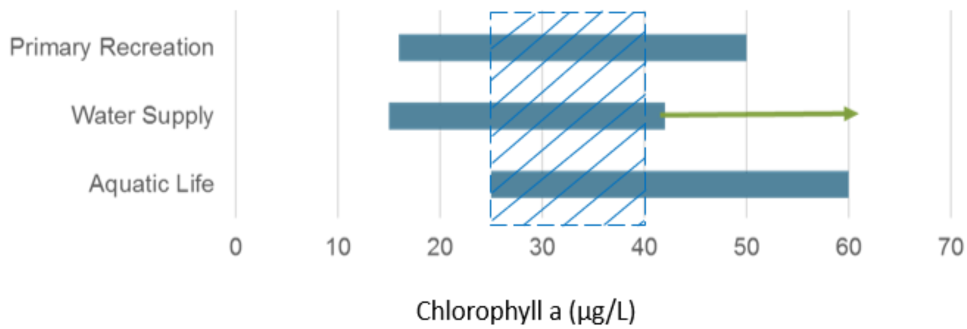
[3] The summary of the group's basis states that the "literature supports recreation, aquatic life and drinking waters uses are achieved when chla is 20-40 µg/L."

[4] The magnitude summary states "35 µg/L to support average chl a levels throughout High Rock Lake of 20-25 µg/L, derived from 25-40 µg/L range for warmwater reservoirs." The 35 µg/L was "near the upper end of the range selected due to mostly favorable use indicators."

4.3.1 Derivation of a Chl a Protective Range

The literature review identified relatively wide ranges for chl a that have been supportive of designated uses in different aquatic systems (Figure 4.1). The target range highlighted represents a range for High Rock Lake that protects the water supply use, primary recreation (if algal toxins can be presumed low; see Chapter 3), and apex predator productivity as an indication of aquatic life protection (see below). In terms of site-specific observations, the existing condition of High Rock Lake supports a thriving sport fishery for apex predators and with no surface, scum-forming algal species (see Chapter 3 for details). These observations, in combination with the literature review, were used to derive a chl a protective range of 25-40 µg/L for warmwater reservoirs similar to High Rock Lake.



Designated Use	Low	High	Notes
Aquatic Life	25	60	25-60 µg/L (abundant apex predators)
Water Supply	15	42	Recommended ideal chl a of 15 µg/L (literature); 42 µg/L (HRL data near active intake); Taste & Odor are treatable.
Primary Recreation (includes aesthetics)	16	50	Lower values from northern lakes with algal toxins issues.
HRL Measured Values	24	55	HRL Measured Values for April-Oct; geometric mean for 2006 – 2016 years; Open water stations

Figure 4.1. Proposed chl a concentration (µg/L) ranges by designated use. The green arrow for Water Supply acknowledges treatment can remove chl a at higher concentrations.

An important indicator for protection of the aquatic life designated use in High Rock Lake is the productivity of apex predators, including the forage trophic levels. Studies on the productivity of apex predators in reservoirs have shown increased abundance of apex predators, prey species, and zooplankton for chl a concentrations of 35-40 µg/L (Allen et al., 1998; Bayne et al, 1994) typically reported as growing season mean values. In terms of changes with lower nutrient levels, Maceina and Bayne (2006) showed a decrease in largemouth bass recruitment and growth rate when chl a concentration was reduced from greater than 40 µg/L to 9-17 µg/L. The lower end of the proposed range of chl a concentrations is set at 25 µg/L to provide sufficient algal production to support abundant apex predators in High Rock Lake and avoid the potential impact to the fishery noted by Maceina and Bayne (2006). Based on the literature review (see Section 2; Figure 4.1), a chl a value of 25 µg/L would be protective of the water supply and primary recreation uses, assuming the associated presence of cyanobacteria is not linked to algal toxin levels that can pose a risk to animal and human health (Chapter 3), and since observation of surface algal scums have generally been absent at High Rock Lake (see Chapter 3 for details).

The upper end of the chl a range to support aquatic life is based on reservoir research documenting abundant apex predators, prey species, and zooplankton at average chl a concentrations of 35-40 µg/L (Allen et al., 1998; Bayne et al., 1994). Overall fish production has been shown to increase even with chl a concentrations greater than 100 µg/L, although there is indication of more benthic species (e.g. carp and flathead catfish) at very high chl a levels (e.g. Egertson and Downing, 2004; Michaletz et al., 2012). The selection of 40 µg/L as the upper end of the range is to maintain a balanced overall aquatic community considering the apex predators, prey species, and zooplankton. The literature indicates overall apex predator abundance would be higher at chl a concentrations >40 µg/L, but there likely would be a shift in species toward bottom-dwelling species and the diversity of prey species and zooplankton may be affected. Further, frequent high chl a concentrations in High Rock Lake could be associated with a higher risk of toxin exposure potentially above proposed thresholds protective of human health, which was also a factor in setting the upper end of the chl a range at 40 µg/L. Literature and observations from High Rock Lake indicate primary recreation and public water supply would be supported at a chl a concentration of 40 µg/L.

4.3.2 The SAC Recommended Criterion Magnitude

The SAC recommends a criterion magnitude of 35 ug/L, from the derived range of 25-40 µg/L, expressed as a seasonal geomean. In developing the recommendation, the SAC considered proposals as low as 25 µg/L and as high as 40 µg/L. Ultimately, the criterion magnitude was set in the upper half of the potential range in acknowledgement of the favorable indicators of use attainment in High Rock Lake, such as a thriving fishery and low algal toxin levels observed in summer of 2016. The maximum value was not selected based on site-specific fisheries information presented to the SAC indicating abundant benthic species, possible overall decreased fish species diversity, and decreased catch rate of striped bass compared with other North Carolina Piedmont reservoirs. Implementation of the proposed criterion of 35 µg/L in High Rock Lake would require a reduction in the level of chl a in the reservoir from the existing condition (see Figure 4.1). Total productivity of the fishery would be expected to decrease, which may increase diversity and shift species abundance toward pelagic species.

4.4 Spatial Components

The spatial variation in biological and physical properties in man-made reservoirs follows a regular spatial pattern (see Figure 3.2). The most upstream reach reflects primarily river conditions as the river flows into the impoundment in which water level is controlled by the downstream dam structure. In the case of High Rock Lake, waters at HRL051 reflect turbid river conditions, and the average chl a is lower than in downstream waters. In the middle reach or transitional zone, water velocity slows down, mineral turbidity settles to the bottom, and a peak in algal abundance typically occurs. In the case of High Rock Lake, waters at YAD152A and YAD152C would be in the transitional zone. Waters downstream of the transitional zone in the lacustrine zone above the dam (YAD169B and YAD169F) would typically have decreased algal abundance compared with the transitional zone.

Available chl a data for monitoring stations listed above for the three reservoir zones are summarized in Table 4.1. Monitoring during the growing season was conducted approximately monthly in five individual years during the period of 2006 through 2016. Substantial variation, expressed as the coefficient of variation (COV), is evident in the river reach (HRL051), but variability is lower in the transitional and lacustrine zones. In terms of protection of uses, the chl a criterion's geometric mean calculated as the geomean of samples collected during the growing season (April-October) will normally be protective of all designated uses even though winter months are not part of the calculation, since chl a is typically lower in winter months (see Sec. 3.2.4).

Table 4.1. Growing Season (April-Oct) chl a geomean (µg/L) by sampling location (COV = coefficient of variation)

Year	HRL051	YAD152A	YAD152C	YAD169B	YAD169F
2006	27.3	51.2	59.6	38.3	34.6
2008	34.1	49.2	53.4	40.3	32.5
2009	16.9	42.1	53.0	43.4	36.0
2011	30.7	50.1	55.6	42.5	36.5
2016	20.8	52.3	58.7	44.3	36.1
Overall	24.1	47.9	55.2	42.0	34.8
COV	29.2%	8.3%	5.5%	5.8%	4.7%

It is recommended that the spatial assessment scale for the site-specific chl a criterion be consistent with the derivation of the criterion magnitude (see Section 4.4.2.) and expressed as a seasonal geomean. It is recommended that all observations for the assessment period from open waters within an assessment unit would be incorporated into the computation of the geomean of available data from the growing season months (April-October). Monitoring locations in backwaters, isolated coves, or where water depth is typically shallow (e.g. <10 feet) would be evaluated based on narrative criteria but excluded from the calculation of the chl a geomean for open waters based on the expectation that such data are not representative of the data used to develop the criterion itself. The SAC also recommends that compliance with the chl a criterion be evaluated with samples collected as photic zone composite samples (e.g. from the water surface down to twice the Secchi depth).

4.4.1 Monte Carlo Spatial Analysis

Evaluation of chl a data for High Rock Lake has shown a consistent spatial pattern with maximum values in the transition zone for the reservoir and lower values in the lacustrine zone and downstream tributaries (see Table 4.1). A Monte Carlo analysis was performed to evaluate how spatial grouping of sampling locations could affect three specific implementation scenarios relative to a seasonal geomean criterion of 35 µg/L. The Monte Carlo approach was used for the analysis to extend conditions simulated to include the five primary years in which regular monitoring was done and to include conditions that could have occurred in other years. Data from monitoring efforts during 2006-2016 were used in the analysis. The objective of the analysis was to evaluate how the seasonal geomean for chl a varies at target stations in the transitional and in the lacustrine reservoir zones relative to the selected range for protection of nutrient-sensitive uses (25-40 µg/L; see section 4.4.1) based on whether the seasonal geomean of 35 µg/L is achieved at all locations individually or for multiple locations aggregated together.

For the evaluation, the Monte Carlo approach was used to create 100 potential datasets for each of four monitoring locations evaluated based on reported chl a concentrations for the growing season for the five primary years in which regular monitoring was done (see Table 4.1). Monitoring stations simulated were HRL051 (riverine), YAD152C (transitional), YAD169B (lacustrine), and YAD169A (tributary embayment) (see Figure 3.2). Figure 4.2 plots the

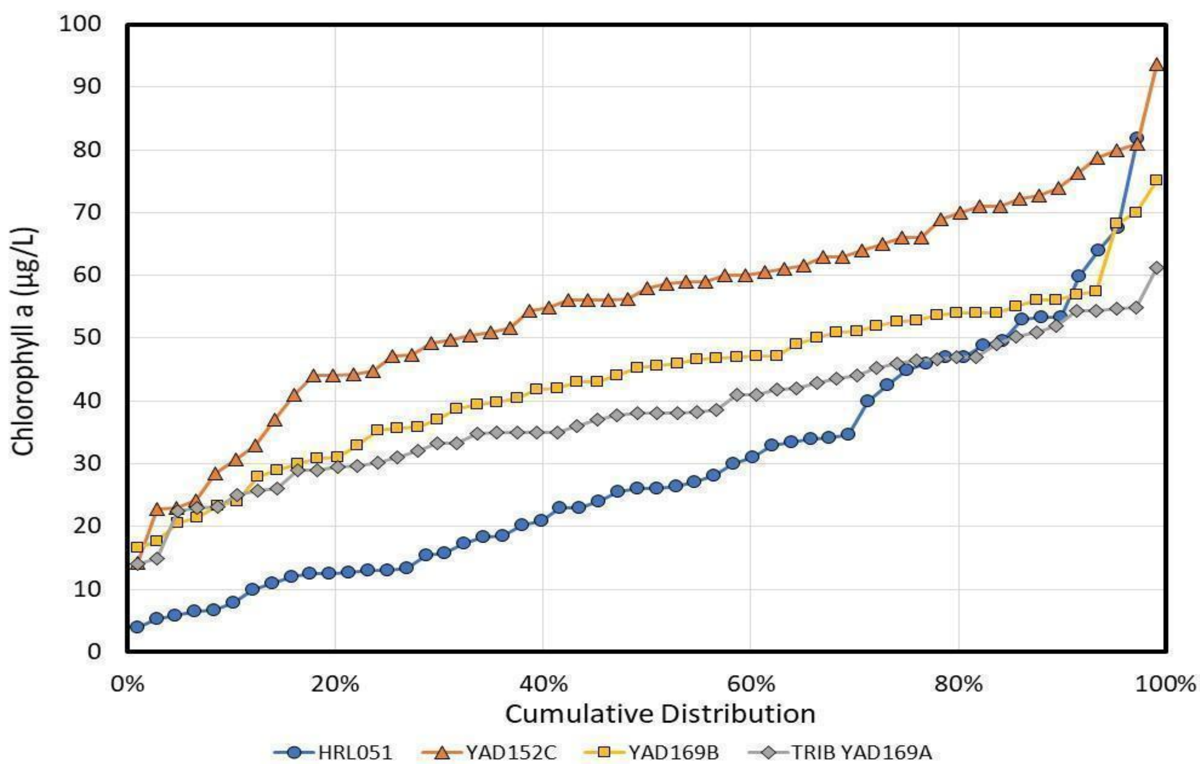


Figure 4.2. Cumulative distributions of measured data by stations utilized in Monte Carlo analysis

cumulative distribution of reported individual sampling chl a concentration values for the four simulated stations for growing season samples from the five monitoring years listed in Table 4.1.

The datasets derived through a Monte Carlo analysis for the four locations simulated were developed with a sampling design comparable to the current NCDWR ambient monitoring effort of monthly sampling during the growing season. Five monthly samples were derived from the cumulative distribution for a given location for two separate years, yielding a total of 10 data points from which to calculate the seasonal geomean. Each point was derived by selecting randomly a probability between 0 and 100%, and then converting the probability to a chl a value by linear interpolation from the respective distribution in Figure 4.2 for the location. This process of creating a dataset of 10 chl a values was performed 100 times for each location. Figure 4.3 provides the distribution of geomean values for each location based on the 100 Monte Carlo simulations for existing conditions.

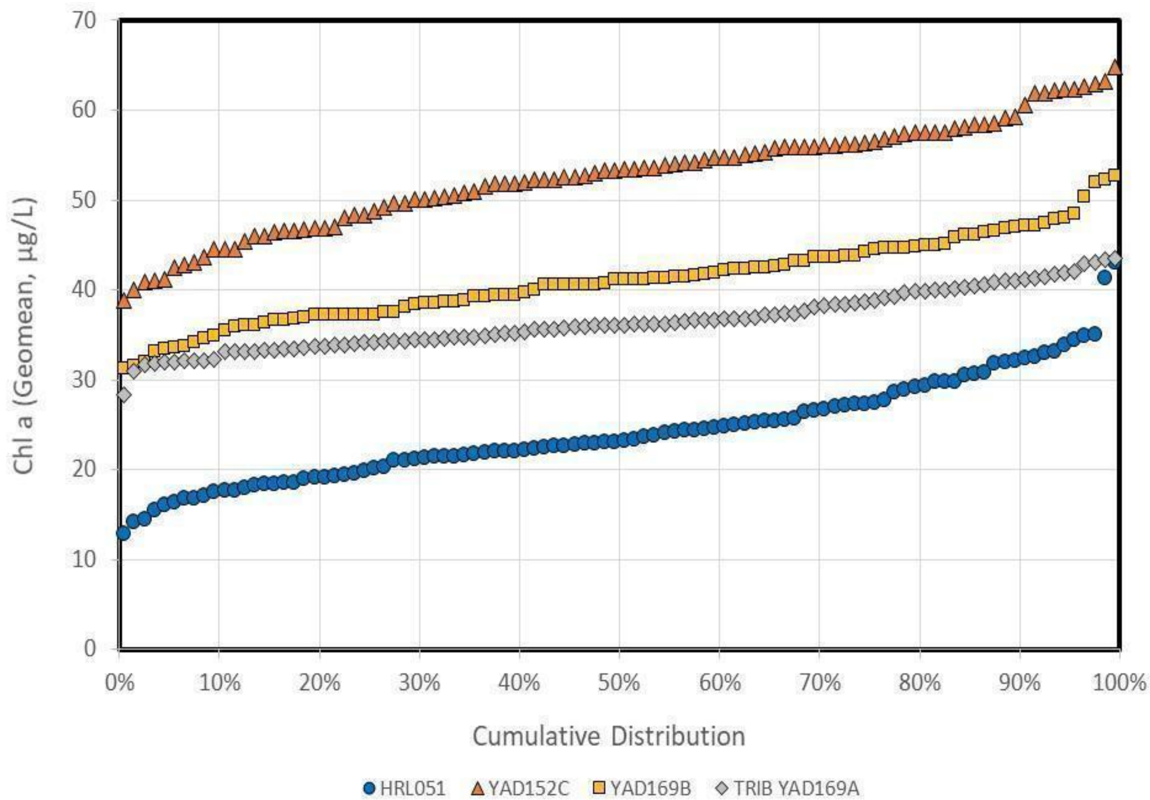


Figure 4.3. Distributions of growing season geomean chl a concentration (µg/L) by location derived from Monte Carlo analysis

The Monte Carlo simulation results were used in conjunction with a target seasonal geomean for chl a of 35 µg/L selected to be above the midpoint of the range highlighted in Figure 4.1 but below the maximum value of 40 µg/L (see Section 4.4.1). Three potential approaches to applying the target chl a geomean criterion were simulated: (1) each individual station meets the criterion as a long-term geomean; (2) each reservoir zone meets the criterion as a long-term geomean; and (3) the transitional and lacustrine zones collectively meet the criterion as a long-term geomean. A long-term geomean was used in the analysis for reduction scenarios to reduce the influence of year-to-year variation in the seasonal geomean on predicted results. The analysis to support the evaluation is summarized in Table 4.2. The reduction percentage in long-term geomean for chl a to achieve the criterion of 35 µg/L varied from 36.6% for Approach 1 based on YAD152C to 18.7% for the combined transitional and lacustrine zones approach. Approach 2 is based on reducing chl a in the transitional zone to 35 µg/L. The potential impact of each approach on the chl a levels to support the currently healthy fishery was evaluated by reducing the chl a distributions derived for YAD152C, YAD169B, and YAD169A (Tributary) by the required reduction for each approach to achieve the criterion. The analysis assumed reductions in chl a would be the same percentage throughout the reservoir stations.

Table 4.2. Influence of assessment unit approach on results

Unit	Existing Long-Term chl a Geomean (µg/L)	Range for Individual Years (see Table 4.1)	Reduction to 35 µg/L (%)
YAD152A	47.9	42.1 - 52.3	27.0%
YAD152C	55.2	53.0 - 59.6	36.6%
YAD169B	42.0	38.3 - 44.3	16.6%
YAD169F	34.8	32.5 - 36.5	N/A
Transitional	48.8	44.0 - 55.5	28.3%
Lacustrine	37.8	35.8 - 40.0	7.5%
Reservoir	43.0	41.1 - 47.3	18.7%
Notes: (1) Reduction percent is to reduce long-term geomean to 35.0 µg/L; (2) Transitional zone assessed as YAD152A and YAD152C; (3) Lacustrine zone assessed as YAD169B and YAD169F; (4) Reservoir assessed as YAD152A, YAD152C, YAD169B, and YAD169F.			

Cumulative distributions for chl a at YAD152C, YAD169B, and YAD169A for the three approaches evaluated are provided in separate panels of Figure 4.4. Evaluation of the three approaches, in terms of protection of aquatic life, was determined by the frequency of overall data points for each approach that were between 25 and 40 µg/L. The analysis also considered whether data points outside the target range were below 25 µg/L or greater than 40 µg/L. In terms of a frequency comparison with the target range, Approaches 2 and 3 were comparable at 72.7% and 73.7%, respectively, while Approach 1 had only 60.3% of data points in the target range (see Table 4.3). Data points outside the target range were primarily <25 µg/L for Approach 1 and primarily >40 µg/L for Approach 3, with data points <25 and >40 µg/L for Approach 2. Approach 1 would likely cause the seasonal geomean of portions of the reservoir to frequently fall below 25 µg/L, which could impact the valued fishery. Approach 2 provides a balance between limiting chl a values <25 µg/L, which may impact the fishery, and limiting chl a values >40 µg/L that could contribute to acute nutrient-dependent impacts in the future.

Approach 3 would likely continue seasonal geomean chl a in the transitional zone >40 µg/L on a frequent basis.

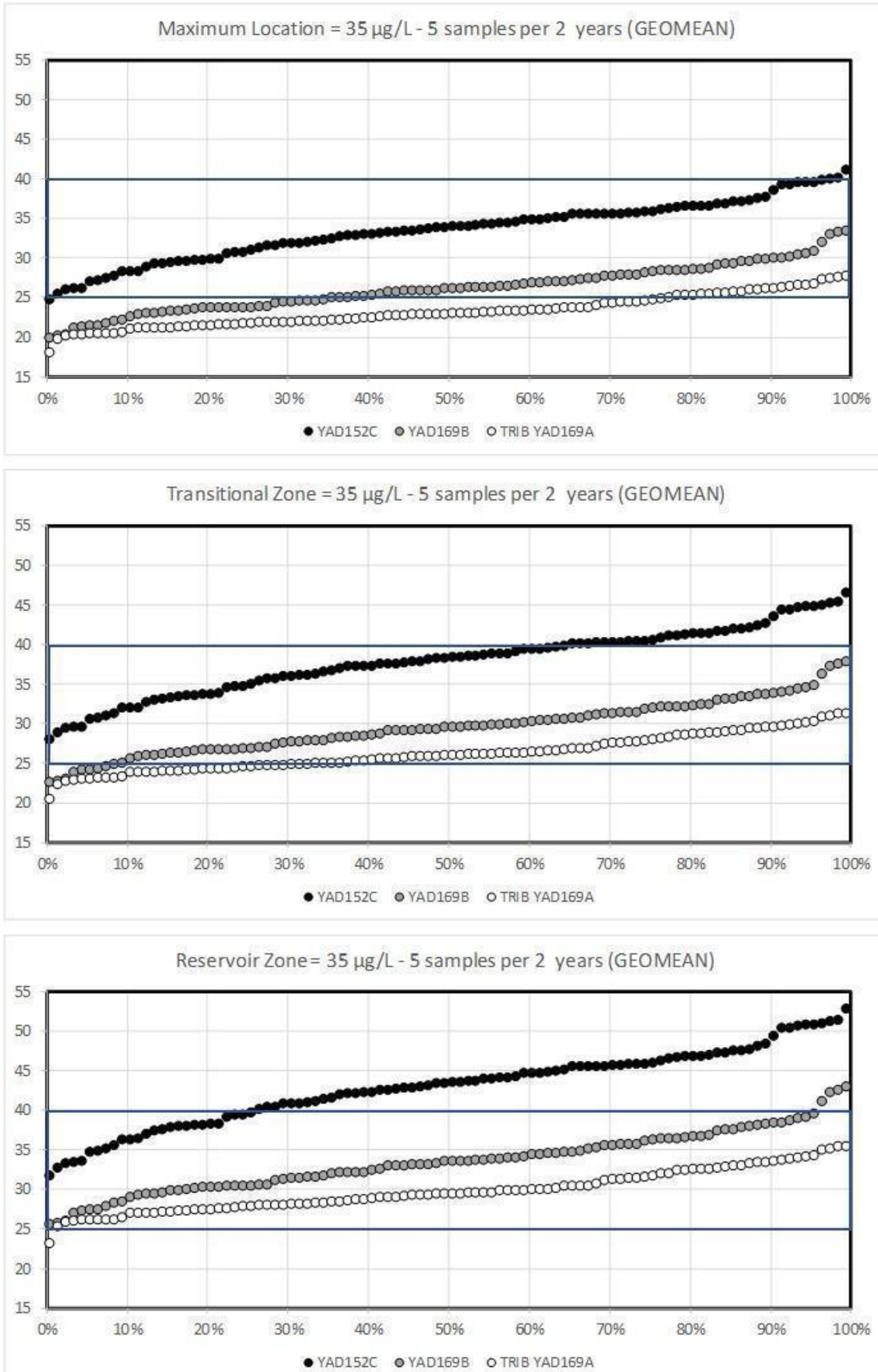


Figure 4.4. Distributions of growing season geomeans (µg/L) by location and assessment approach (top panel: each individual station meets the criterion as a long-term geomean; middle panel: each reservoir

zone meets the criterion as a long-term geomean; and bottom panel: the transitional and lacustrine zones collectively meet the criterion as a long-term geomean). The box indicates target chl a concentration range of 25-40 µg/L.

Table 4.3. Distribution of chl a geomean by spatial assessment approach.

Assessment	<25 µg/L (%)	25 – 40 µg/L (%)	>40 µg/L (%)
Approach 1	39.0	60.3	0.7
Approach 2	15.7	72.7	11.7
Approach 3	0.3	73.7	26.0

Of note is the difference between the temporal averaging used in the Monte Carlo analysis (ten randomly selected chl a values used to compute a geometric mean) and the temporal averaging in the proposed chl a criterion (all growing season chl a values from a single year used to calculate a seasonal geometric mean). The normal lake sampling plan of the NC DWR is to collect five such chl a samples each growing season. Also not included in the Monte Carlo analysis is the consideration of a maximum allowable exceedance frequency (the proposed criterion is that one-in-three seasonal geomeans may exceed the chl a criterion). It is believed that these two differences between the Monte Carlo analysis and the proposed chl a criterion offset one another, so that the analysis presented is usable as-is for comparing the implications of the three assessment unit approaches analyzed with the Monte Carlo analysis. Repeating the Monte Carlo analysis with a different set of assumptions would likely not have significantly changed the analysis outcome and would have led to an additional delay in completing the proposed High Rock Lake chl a criterion development, and was therefore not pursued.

4.4.2 Considerations for Delineating Assessment Units

In the Clean Water Act framework, an assessment unit (AU) is the basic spatial component that states use for evaluating attainment status of water bodies. States use various bases to delineate AU boundaries, including hydrography datasets, hydrologic unit codes, maps of water body names, major junctions, morphology, or limnological zones. Although assessment units can be delineated in different manners, USEPA (2005) offers the following guidance on segmentation:

> *Segmentation may reflect an a priori knowledge of factors such as flow, channel morphology, substrate, riparian condition, adjoining land uses, confluence with other waterbodies, and potential sources of pollutant loadings…Segments should… represent a relatively homogenous parcel of water (with regard to hydrology, land use influences, point and nonpoint source loadings, etc.)*

States also vary widely with regard to how chl a is assessed spatially within reservoirs, and the procedures often differ from those used for toxics. For example, Alabama uses an assessment methodology that varied based upon the size of the waterbody. Some relatively small lakes that are most easily monitored near the forebay use only this location for assessment. When a lake is considered large enough to have more than one station, separate criteria are generally applied to these separate stations (A.A.C. 335-6-10-.11). Georgia assigns specific chl a criteria to individual stations within large reservoirs. Criteria can vary between stations to recognize different expectations for different parts of the reservoir. Florida applies chl a criteria for most lakes as a lake-wide or lake segment-wide average (F.A.C. 62-302). Virginia recognizes three limnologically-defined zones within reservoirs (riverine, transitional, and lacustrine), but only applies numeric nutrient criteria to the lacustrine zone (Virginia DEQ, 2009).

Despite considerable discussion, the SAC did not come to a consensus regarding how spatial assessment units should be defined for High Rock Lake or other water bodies. However, the manner in which assessment units are spatially defined for chl a has implications for the stringency/conservativeness of the criterion, and also for how different uses or risks are balanced within a reservoir. For that reason, this section provides a general discussion of the two basic approaches discussed and considered by the SAC: (1) delineating AUs based on individual monitoring stations, similar to NC's existing or default approach; and (2) delineating AUs by three major limnological zones.

4.4.2.1 Defining Chl a Assessment Units by Individual Stations

In large reservoirs, many DWR monitoring stations are more than 1 mile apart. For example, in High Rock Lake, the distance between neighboring monitoring stations varies between 0.3 and 3.6 miles. Hence, most of the AUs delineated around individual stations in High Rock Lake are still relatively large. Compared with other approaches, the use of individual stations increases the homogeneity of water within an AU, which is an important characteristic of AUs as recommended by USEPA (2005). The single station approach also avoids averaging that can mask temporal and/or spatial changes in chl a concentration. Accordingly, an individual-station approach will generally be more sensitive to detecting chl a related changes that occur at specific locations within the reservoir. The individual station approach will also be better able to detect chl a related problems that result from changes in the spatial distribution of nutrient loading to the lake from loading hot spots or changing development patterns in the watershed.

Because the highest-chlorophyll station would tend to control a reservoir TMDL, an individual-station approach for delineating AUs will generally require higher levels of nutrient reduction than approaches that would average the chl a goal over larger segments. To this extent, the individual station approach is more environmentally conservative with respect to potential harmful effects of excess algae (e.g, toxins, bloom events, etc.). An estimate from the Monte Carlo analysis is that applying the criterion using individual stations for AU specification rather than the limnological AU specification will decrease the prevalence of chl a values above 40 µg/L from 11.7% to 0.7% (Table 4.3). Another practical advantage of the individual station approach is consistency with North Carolina's existing approach and assessment data processing procedures.

4.4.2.2 Defining Chl a Assessment Units by Limnological Zones

In contrast to delineating AUs around individual stations, this approach would define AUs using *a priori* knowledge of major reservoir zones that are functionally different and represent logical units for water quality management. The concept that reservoirs exhibit three major spatial zones (riverine, transitional, lacustrine) is well established in the scientific literature and consistent with observed water quality in High Rock lake (see section 3.1). In practice, the three-zone approach would only involve aggregating data from DWR monitoring stations that are relatively close to each other (e.g., YAD152A and YAD152C) and would not involve a dramatic change in overall segmentation, but would avoid the delineation of small segments around individual stations such as the AU currently associated with YAD152C.

A potential advantage of the limnological zone approach to AUs is the protection of current levels of fish production in High Rock Lake, as demonstrated by the Monte Carlo analysis (section 4.4.1). The limnological zone approach for AU specification raises the percentage of chl a values within the fully protective range from 60.3% to 70.2%, when compared to the individual station approach. The percentage of chl a values below the protective range also decreases from 39.0% to 15.7% (Table 4.3). Attainment of the recommended criterion will require significant chl a reductions in High Rock Lake, regardless of whether AUs are individual station or three limnological zones. The three-zone approach reduces the risk of harmful effects associated with high chl a, relative to existing levels, but provides a higher level of protection of the fishery use compared to the individual station approach.

4.5 Consideration of Statistical Confidence

The SAC discussed the concept of incorporating a statistical test of confidence that the chl a criterion had been exceeded in a given assessment period, as a potential means to reduce false findings of non-attainment (for 303d listing of water bodies) or false findings of attainment (for delisting water bodies). North Carolina currently uses a non-parametric statistical test (the binomial method) for not-to-exceed criteria. Although the binomial method is not appropriate for a seasonal geometric mean, other methods could be developed, such as the calculation of confidence limits on the geometric mean. An argument against the use of a statistical test is the primary purpose of these test is to prevent a very small number of data from controlling the listing/delisting decision, but seasonal geometric mean chl a values (calculated for at least two years) would be based on at least 10 data points. Also, if only 10 data points were available for a given assessment period, confidence limits could be relatively wide, which could make it very difficult to either list or delist water bodies. Although the SAC is not recommending a specific statistic test at this time, this topic could be re-examined at the time of statewide criteria development.

4.6 Summary of Proposed Criterion

The proposed chl a criterion for High Rock Lake is a seasonal geomean of 35 µg/L, not to be exceeded more than once in three years, for growing season months of April-October based on protection of all uses while maintaining the productivity of the sport fishery (Table 4.4). In terms of spatial considerations, all monitoring data from open waters within assessment units collected during the months of April through October would be used to compute a geomean to compare with the proposed criterion. The criterion would apply to all months of the year, with attainment of the criterion assessed with data from the growing season months. The SAC recommends the exceedance frequency assessment approach. The SAC recommended frequency is not to exceed more than one in three calculated seasonal geomean values.

The SAC recognizes that several considerations remain in establishing the site specific chl a criterion for High Rock Lake. These considerations include how much data to include and what data might be excluded during assessment, spatial aggregation of data, and whether the criterion should include a statistical confidence test (Table 4.5). Furthermore, the SAC encourages continued monitoring of cyanobacterial toxin levels paired with chl a assessments to better evaluate potential exposure risks and toxin dynamics in High Rock Lake. The SAC refers these implementation questions to the CIC for further consideration.

Table 4.4. Proposed Chl a Criterion for High Rock Lake.

Component	Selection	Notes on Selection
Magnitude	35 µg/L	None
Period/Duration	Seasonal Geomean	Calculated Geomean based on all data from growing season
Season/Duration	April-October	Include samples collected in at least five different growing season months for each year of data included in the analysis
Frequency	Maximum Exceedance Frequency of One-in-three	Compute the geometric mean for each year of individual data and apply a frequency component of not more than one exceedance out of three years of data
Spatial Considerations	Open Waters	Photic zone composite based on twice the Secchi depth; shallow waters and isolated coves to be addressed through narrative criteria; all data within each assessment unit would be incorporated into the calculated geomean

Table 4.5 SAC's Additional Topics for Specific Consideration by CIC

Component	**Alternatives or Additional Information included in this document**
Sample Size/Filtering of Monitoring Data	SAC encourages CIC to offer implementation thoughts on whether data should be collected from at least five different months within the growing season or if there are other bounds or minimums on data density that may be acceptable.
Spatial Assessment	Whether or not to include multiple stations in an assessment unit
Statistical Test of Confidence	Whether or not to consider a statistical test of confidence that the chl a criterion was exceeded in a given assessment period

4.7 References

Allen, M.S. Greene, J.C. Snow, F.J. Maceina, M.J. DeVries, D.R. 1999. Recruitment of Largemouth Bass in Alabama Reservoirs: Relations to Trophic State and Larval Shad Occurrence. *North American Journal of Fisheries Management* 19(1):67-77.

Bayne, D.R. Maceina, M.J. Reeves, W.C. 1994. Zooplankton, fish and sport fishing quality among four Alabama and Georgia reservoirs of varying trophic status. *Lake and Reservoir Management* 8(2):153-163.

Egertson, C.J. Downing, J.A. 2004. Relationship of fish catch and composition to water quality in a suite of agriculturally eutrophic lakes. *Canadian Journal of Fisheries and Aquatic Science* 61: 1784-1796.

Florida Department of Environmental Protection. 2012. Overview of Approaches for Numeric Nutrient Criteria Development in Marine Waters. 110 p.

Maceina, M.J. Bayne, D.R. 2001. Changes in the black bass community and fishery with oligotrophication in West Point Reservoir, Georgia. *North American Journal of Fisheries Management* 21(4):745-755.

Missouri Department of Natural Resources (MDNR). 2017. Rationale for Missouri Lake Numeric Nutrient Criteria. December 2017.

Michaletz, P.H. Obrecht, D.V. Jones J.R. 2012. Influence of Environmental Variables and Species Interactions on Sport Fish Communities in Small Missouri Impoundments. *North American Journal of Fisheries Management*, 32:6, 1146-1159. First Published November 1, 2012.

North Carolina Division of Water Resources. 2019. 2020 303(d) Listing and Delisting Methodology. https://files.nc.gov/ncdeq/Water%20Quality/Planning/TMDL/303d/2020/2020-Listing-Methodology-approved.pdf. 14 p.

U.S. Environmental Protection Agency (USEPA). 2000. Nutrient Criteria Technical Guidance Manual – Lakes and Reservoirs. EPA 822-B00-001. Washington, DC: USEPA, Office of Water.

USEPA. 2010. Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries: 2010 Technical Support for Criteria Assessment Protocols Addendum. EPA 903-R-10-002. 63 p.

U.S. Environmental Protection Agency. 2005. Guidance for 2006 Assessment, Listing and Reporting Requirements Pursuant to Sections 303(d), 305(b) and 314 of the Clean Water Act. Memorandum from Diane Regas to Water Division Directors. 89 p.

Virginia Department of Environmental Quality. 2009. Monitoring and Assessment of Lakes and Reservoirs. Water Guidance memo No. 09-2005. 31 p.

5. Potential Elements of a Framework for Deriving Site-Specific Criteria

North Carolina's Nutrient Criteria Development Plan (NCDP) states a commitment to develop nutrient-related criteria (causal and/or response variables) throughout the state on a site-specific basis. High Rock Lake has served as the pilot water body for reservoirs and lakes, and the chlorophyll a (chl a) criterion recommendation of this technical support document apply to that specific water body. However, the NCDP schedule calls for the adoption of nutrient-related criteria on a statewide basis during the 2023-2028 timeframe. Part of this process will be to "confirm the approach proposed during the adoption of the nutrient criteria in [High Rock Lake] with SAC involvement". The purpose of this section is to discuss how lessons learned during the reservoir pilot might apply to the future effort to derive chl a criteria for other reservoirs and lakes.

The SAC has not yet developed a detailed framework for deriving reservoir-specific chl a criteria. However, many elements of the SAC's approach for High Rock Lake would be transferable to other water bodies. At various times, the SAC also discussed potential elements of a more formal framework for site-specific criteria derivation. This section attempts to document some of those concepts in case they are useful during the future, statewide effort. Any of the framework elements discussed herein are subject to additional discussion by the SAC and DWR.

5.1. Desired Characteristics of a Framework

North Carolina's intent to develop nutrient criteria throughout the state on a site-specific basis is challenging from a scientific and regulatory perspective. Site-specific chl a criteria have the advantage of reflecting water body-specific responses to nutrient inputs, and to avoid the misallocation of resources that can result from one-size-fits-all criteria. However, the derivation of site-specific criteria can be resource-intensive because it requires evaluation of water body-specific conditions and nutrient-response relations. It is not practical for North Carolina DEQ to develop complex nutrient-response models for every water body in the state, nor to devote the level of time and resources that were devoted to the High Rock Lake pilot. Ideally, a framework for deriving site-specific criteria would be streamlined enough for practical application with datasets of moderate size, while also including enough water body-specific information to make the correct criteria decisions.

With this background, the SAC cites the following characteristics as desirable for a framework for developing site-specific chl a criteria:

1. *The framework produces site-specific chl a criteria that are protective of designated uses*. This is a minimum requirement of any criteria derivation process. All uses of the reservoir should be considered, including public water supply, recreation, and aquatic life. The site-specific nature of the desired framework is explicit in the NCDP, and is based in the understanding that different water bodies can respond to nutrient inputs in different manners.

2. *The framework should minimize assessment and management errors*. Both type I (false findings of impairment) and type II (false finding of attainment) errors are of concern and should be minimized to the extent possible. Overprotective criteria would lead to type I errors, whereas underprotective criteria would lead to type II errors. Although some degree of conservativeness is appropriate for water quality criteria, highly overprotective criteria would misdirect TMDL and implementation resources.

3. *The framework should consider both literature and reservoir-specific information*. The SAC's chl a recommendations for High Rock Lake were derived using both literature-based and reservoir-specific information. Research for the reservoir pilot revealed that targets based on the literature and reservoir-specific data can be very different. The scientific and lake management literature includes a wide range of potential chl a targets associated with different regions, reservoir/lake types, and uses. Many of the studies from the literature focus on water bodies that have experienced algal-related problems that might or might not occur in other reservoirs being considered for site-specific criteria. The literature also includes many chl a targets from higher latitudes or altitudes, many of which could be unrealistically low for southeastern lakes and reservoirs. Some literature-based chl a targets are based in concepts such as user perception, which are difficult to transfer from one region to the next.

Reservoir-specific data or models can help determine whether uses are currently being met, and also provide insights into the empirical relations between chl a and other use indicators. But like the scientific literature, reservoir-specific information also has limitations for deriving site-specific criteria. Some water bodies may have relatively few water quality data and little narrative information on use attainment (e.g., fishery status, water treatability issues, algal toxins, etc.). Even for a relatively data-rich water body such as High Rock Lake, the SAC did not find it simple to identify chl a thresholds above which specific uses were met or not met. Rather, much of the information pointed to a continuum of risk, where the concern over potential impacts increased with chl a.

Ultimately, the SAC recommended a chl a criterion from within a range of candidate values (25 – 40 µg/L), as described in section 4. That range was determined from both literature and High Rock Lake-specific information. The lower end of the range was more strongly influenced by the literature and the desire to limit potential impacts to the fishery, whereas the upper end of the range was from multiple lines of evidence that include the literature and High Rock Lake's existing chlorophyll-indicator relations. Similar consideration of both literature and reservoir-specific information is likely to be useful for a statewide framework. The framework could emphasize reservoir-specific information for water bodies with more definitive chlorophyll-use indicator relations. The literature will remain informative of the chl a concentration at which some lakes/reservoirs experience algal-related problems.

5.2. Potential Common Elements

Some elements of the proposed chl a criterion for High Rock might be directly transferred to other lakes and reservoirs without site-specific deliberations. This could be the case for criteria elements whose technical justification for High Rock Lake would apply equally to other reservoirs, or criteria elements for which it would be unnecessarily problematic to use different approaches for different water bodies during the assessment process. The basis for the following criteria elements is provided in section 4, and much of the reasoning for High Rock Lake would also apply to other reservoirs:

- Geometric mean
- April – October growing season
- A 1-in-3 year allowable exceedance frequency
- Photic zone grab sample at 2X Secchi depth

The magnitude of the chl a criterion is the element most likely to change between lakes/reservoirs. Factors to consider in the adjusting the magnitude of chl a criterion between water bodies include warmwater vs. coldwater classification, historical and recent chl a concentrations, designated uses, and various narrative and numeric indicators of use support. Following are major steps of a potential framework to derive site-specific criteria:

1. Application of a chl a screening range as the initial evaluation of impairment status.
2. Consideration of other numeric and narrative indicators.
3. Application of decision rules on impairment status.
4. Application of decision rules on site-specific criteria.

These factors are discussed in subsections below.

5.3. Chl a Screening Range Concept

With any framework for deriving site-specific criteria, one of the first steps would be to determine whether the reservoir is effectively meeting designated uses vs. experiencing tangible nutrient-related impairments. Results of this determination would be a major factor in deciding if the site-specific criteria should be lower than existing conditions. The use of readily-available water quality data such as chl a concentrations could streamline this determination. As discussed in previous sections, the SAC did not identify a one-size-fits-all chl a criteria that could be used in a pass-fail manner to answer this question. However, the SAC did consider it more practical to identify a range of chl a concentrations that was associated with increasing risk of impairment.

With this background, a potential first step of a framework could be to compare a reservoir's existing chl a concentration (seasonal geometric mean) to a screening range, with the goal of determining whether the reservoir can be categorized as likely attaining vs. likely impaired based on chl a alone. The upper end of the range would represent a value above which nutrient impairment is likely, and the lower end of the range would represent a value below which nutrient impairment is unlikely. Reservoirs in the "gray area" (i.e, within the range) would require additional narrative assessment (step 2) to determine if they experience nutrient-related impairments. Figure 5.1 illustrates the chlorophyll-based screening range with a range developed by the SAC (25-40 µg/L) during the High Rock Lake pilot.

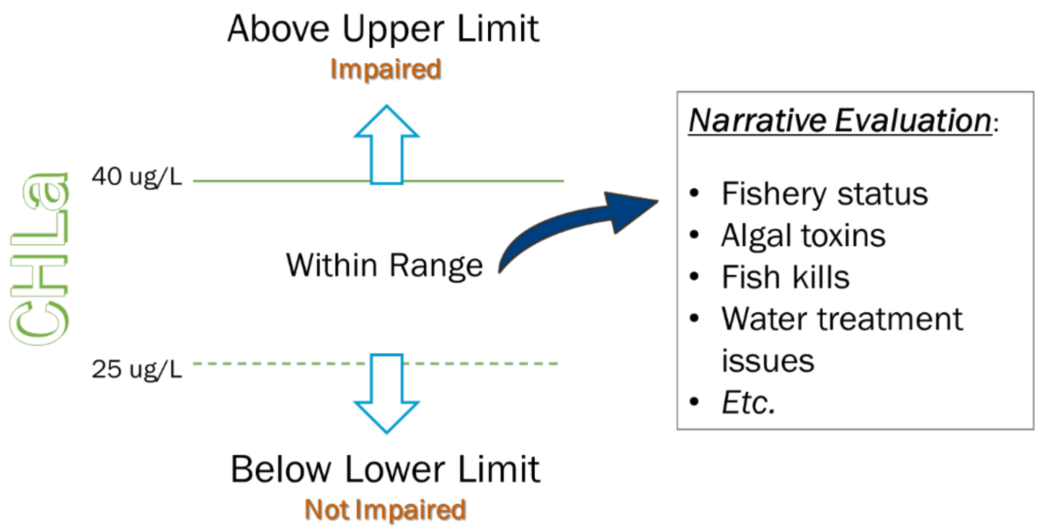


Figure 5.1 – Illustration of the chl a screening range concept.

The use of a chl a screening range is conceptually similar to an approach published by Arizona (Arizona DEQ, 2008), and is also similar to criteria recently adopted by Missouri [10 CSR 20-7.031(5)(N)1.C.(I)] and approved by USEPA. However, the screening range concept described is specifically discussed herein as a step to streamline the derivation of site-specific chl a criteria rather than a long-term assessment method.

5.4. Consideration of Narrative and Numeric Indicators

In the second step of a potential framework for deriving site-specific criteria, various other types of reservoir-specific information would be considered to support impairment categorization. Although many types of information might be considered during this step, the framework could be applied more consistently if it included a pre-defined list of useful indicators with associated thresholds. Table 5.1 provides an example of such a checklist. The list includes both narrative indicators (e.g., presence/absence of fish kills, nuisance conditions, fishery status) and numeric indicators (pH, DO, cyanotoxin concentrations). It would not necessarily be required to have information for every indicator to perform the categorization.

An important aspect of the indicator list is that indicators are categorized as either primary or secondary. Primary indicators are those that are more direct indicators of nutrient impairments, whereas secondary indicators may indicate concerns but are not direct indicators of impairments. For example, a high cyanobacteria density would be a secondary indicator, whereas persistent exceedance of cyanotoxin thresholds would be a primary indicator. This distinction is important because decision guidelines for impairment determinations would weight primary indicators more than secondary indicators.

Table 5.1: Examples of Potential Indicators for Narrative Evaluation

Use Category	Indicator	Primary or Secondary Indicator	Narrative or Numeric Indicator
Aquatic Life	DO concentration	Primary	Numeric
	DO saturation	Secondary	Numeric
	Ph	Primary	Numeric
	Algal toxins	Primary	Numeric
	%Cyanobacteria	Secondary	Numeric
	Fishery status	Primary	Narrative
	Fish kills	Primary	Narrative
	Fish abnormalities	Secondary	Narrative
Public water supply	Algal toxins	Primary	Numeric
	T&O-causing compounds	Secondary	Numeric
	Treatability challenges	Primary	Narrative
Recreation	Algal toxins	Primary	Numeric
	Secchi depth	Secondary	Numeric
	Nuisance blooms; mats or extensive scums	Primary	Narrative

Under a potential framework, each indicator could be categorized as green (full use support indicated), yellow (potential concerns), or red (strong evidence of use impairment). Associated guidance would provide numeric ranges or other guidelines for these determinations. The guidance could also include decision rules for how multiple or mixed-result indicators would be used to interpret existing use support.

If sufficient data were available, this step 2 could also involve direct examination of the relations between chl a and other indicators such as water clarity, pH, cyanotoxins, etc. Such empirical relations could lead to the selection of chlorophyll targets to achieve specific responses. Examples of chlorophyll-indicator relations for High Rock Lake are provided in Section 3 of this document.

5.5. Decision Guidelines for Site-Specific Criteria

After application of the chlorophyll-based screening range and narrative numeric evaluation, the final steps would be to make the appropriate site-specific chl a criterion. Although professional judgment would be required, DWR's decisions would be more transparent and defensible if clear decision guidelines were developed. The associated decision guidelines could be organized as a matrix based on the existing chl a concentration (below, within, or above screening range) and outcome of the narrative evaluation (narrative evidence of use attainment, non-attainment, or inconclusive). For example, if a reservoir's chl a concentration was above the screening range but the reservoir did not show clear signs of impairment from the narrative/numeric evaluation, the criterion could likely be set at or near the top of the screening range. But a reservoir within the screening range that failed the narrative/numeric evaluation might receive criteria in the lower half of the screening range. The formulation of specific decision guidelines would require additional discussion by the SAC and DEQ.

In some cases, criteria could be set to protect a reservoir's existing condition. For example, a lake with chl a in the 15-20 ug/L range (below the screening range) but with some exceedances of secondary indicators might receive a criterion of 20 ug/L to prevent impairments. If robust chlorophyll-response linkages were available, they could also be applied to set a specific chl a target during this step, and these linkages might support criteria outside of the screening range.

5.6. References

Arizona Department of Environmental Quality. 2008. Narrative Nutrient Standard Implementation Procedures for Lakes and Reservoirs. Available at https://legacy.azdeq.gov/environ/water/standards/download/draft_nutrient.pdf. 21 p.