Project Title: Biodiversity and habitat characteristics as indicators of community change following oyster reef restoration
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Abstract:

With ongoing declines in oyster populations due to overharvesting, climate change, and other anthropogenic drivers of ecosystem deterioration, there is a dire need to implement and monitor habitat restoration efforts to ensure they effectively allow for the persistence of these ecologically significant communities. In 2018, two oyster restoration approaches were implemented near Beaufort, NC and assessed relative to each other and to nearby cultch reefs. The two approaches (a classic shell bag (SB) method and OysterCatcher^{TM} (OC), a novel substrate) were established in a tidal creek and an exposed shoreline at Carrot Island to study changes in habitat and biodiversity over time in distinct energy environments. I used published methods to collect oyster lengths and densities on each site type in 2023 and 2024. Using published biodiversity sampling protocols, I deployed passive samplers to recruit reef-resident organisms. I predicted oyster lengths and densities to be greater for OC than SB, and higher overall in the creek than the exposed shoreline. I also expected higher biodiversity associated with more intact reefs (with more/larger oysters), and that both habitat and biodiversity metrics would be higher in 2023 than 2018. We found that oyster densities have significantly decreased over time since 2020 (with particularly heavy losses on the exposed shoreline and at shell bag reefs). Additionally, while free-living communities varied in terms of diversity over time, parasite communities largely increased in prevalence and diversity over time, thereby coinciding with previous research finding these surrogate taxa to be more effective ecosystem indicators. Thus, overall, my results highlight how substrate, location, and time can affect restoration success, and further emphasize the importance of monitoring restoration efforts over time.

Introduction:

As coastal habitats, oyster reefs are often on the frontlines of direct and indirect sources of anthropogenic ecological degradation. This impacts both the structure and function of these foundational habitats, which offer ecosystem services to the environment as well as nearby coastal communities. Restoration of these habitat-forming species offers a way to study the resilience and recovery of communities while restoring lost ecosystem function and services following disturbances. However, although some restoration methods have produced promising results, there are also concerns and documented shortcomings that affirm the need to continue to improve upon existing techniques (Buckley & Crone, 2008; Kondolf, 1998). One demonstrated issue is the type of substrate and materials selected for restoration, the significance of which has been particularly emphasized after inadvertent issues have arisen from previous efforts (Comba et al., 2023; Hennebert et al., 2014). Further restoration concerns stem from the potential for facilitated recruitment of invasive species to restored regions (Abella & Chiquoine, 2019; Jude & DeBoe, 1996), as nonindigenous species have already demonstrated an elevated rate of occurrence on artificial structures (of various composition, including seawalls and pilings in aquatic systems) when compared to natural locations (Glasby et al., 2007), suggesting notable implications for the use of structures that employ materials/species extrinsic to system being restored (Abella & Chiquoine, 2019). Moreover, even if these challenges above are avoided, restoration efforts must still withstand the test of time and function comparably to natural sites typical for the region to be considered effective. Despite general understanding that restoration efforts are needed to reinstate lost ecosystem services, there is often only short-term (1-2 years) post-project monitoring, consistent with time constraints imposed by grant-funded project timelines (often less than 3 years) (Lindenmayer & Likens, 2018). This is a trend within living shoreline monitoring specifically as well (Baker & Gittman, 2025). Without monitoring, the success and evolution (i.e., changes in habitat quality and biodiversity through time) of a restoration project is uncertain. As such, there is still a prominent and pressing need to a) ascertain the most effective approaches to restoring oyster reefs as well as b) address the lack of long-term monitoring of restoration to determine the sustainability of specific restoration techniques and improve our ability to predict future effects of restoration.

Methods:

I conducted field surveys at thirteen oyster reefs in the Rachel Carson Reserve between 2023 and 2024 and compared the data collected in the surveys to early monitoring results collected from 2018-2020 (habitat) and 2018-2019 (biodiversity). The 2023 sampling consisted of three biodiversity field dates in June, September, and October, with a habitat field date in August. In 2024, as proposed, I conducted two biodiversity field dates, although these sampling efforts took place in August and November (rather than August and October), and the habitat field day took place in November rather than August, as initially proposed. I used previously established protocols (Wellman et al., 2022) to quantify oyster lengths and densities. Oysters were given the designation of “adult” if they were >24mm, while “juveniles” were those <24mm, as defined in the initial monitoring effort (Wellman et al., 2022) and previous literature (La Peyre et al., 2014). For restored (SB and OC) sites in 2023, I removed representative portions of each reef (one shell bag from each SB reef and one 10cm section of each OC reef’s substrate frame) to process back in the lab for oyster length (mm), live/dead oysters, and dimensions of the sample. SB samples were selected by lifting off a representative shell bag from the existing reef, whereas the OC samples were more destructively removed by sawing through the substrate frame with a hacksaw at each end of a 10cm sample. At all sites, I used a Trimble

R10 Global Navigation Satellite System Receiver to record the position and elevation of the sample location (Trimble R10 MODEL 2 GNSS System Datasheet, 2021).

For lab processing the 2023 SB samples, dimensions were taken by using a ruler to measure up to three lengths, widths, and heights of the shell bag across three different portions of the sample (as the bags were often not uniformly shaped and this approach allowed an average value for each measurement to be obtained). For 2023 OC samples, dimensions were taken by using a ruler to measure up to three lengths and diameters of the sample. The number of oysters and these dimensions were used to calculate oyster densities for each sample. To do so, I divided the total number of live oysters by the calculated surface area of each sample, following the formulas for rectangular prisms and right cylinders for 2023 SB and OC samples, respectively, as these shapes best reflected the area oysters could originally recruit to at the time of initial restoration (Fig. 3; Wellman et al., 2022).

By contrast, in 2024, we adjusted our protocol to instead use a $0.0625m^2$ quadrat and count the number of oysters present at three distinct points on the reef site, as described in the progress report. These locations included the most seaward region of the reef, the middle of the reef, and the most landward region of the reef. The quadrat was haphazardly tossed into each region of these reefs to determine exact placement. After the number of live oysters was counted, the length of up to five live oysters was measured and recorded. The locations of these samples were also recorded with the Trimble, and the live oyster counts were divided by the surface area of the quadrat to achieve comparable oyster densities to the earlier samples.

As for biodiversity surveys, in addition to variations in timing, we limited the biodiversity assessment of the 2024 samples to the host level, as described in the progress report. The parasite diversity data from 2023 was still used to characterize the community changes at this level and the implications for the broader biodiversity status 5+ years post restoration, and we still collected parasite diversity data from eastern mudsnails (*Ilyanassa obsoleta*) in 2024. I deployed passive samplers ("crab condos") at all sites sampled for biodiversity about one meter seaward of the reef, and they were left to recruit organisms for ~6-8 weeks. The passive samplers were retrieved and we recorded the presence and abundance recruited organisms, including all panopeid mud crabs, stone crabs, porcelain crabs, and fish (including oyster toadfish, blennies, pinfish, belted sandfish, and skilletfish), among other mobile fauna. The eastern mudsnails were collected using active sampling methods (hand sampling and dip-netting), and processed in the lab at ECU using established protocols.

For statistical analyses, I have used generalized linear models to preliminarily quantify changes in habitat and biodiversity characteristics. I have also used multiple community diversity analyses in Primer 7, including PERMANOVAs, and primarily assessed the impacts of time post restoration and reef type on community characteristics. I initially proposed analyzing the impacts of season, habitat characteristics, and other parameters in addition to these predictors as well, and I intend to add these terms in later on in the analysis process prior to publication of these results.

Results:

For habitat characteristics, I compared previous monitoring data to samples from 2023 and 2024 to assess how oyster densities and elevation varied over time by restoration treatment. While oyster densities increased from 2018-2020 in the previous monitoring effort, the first addition of my data (up through 2023) showed a decrease in oyster densities from 2020-2023 (Fig. 1). In 2024, while Oyster Catcher reefs continued to outperform shell bag reefs, further significant decreases in oyster densities occurred across both reef types and across both shorelines at Carrot Island (Fig. 1).

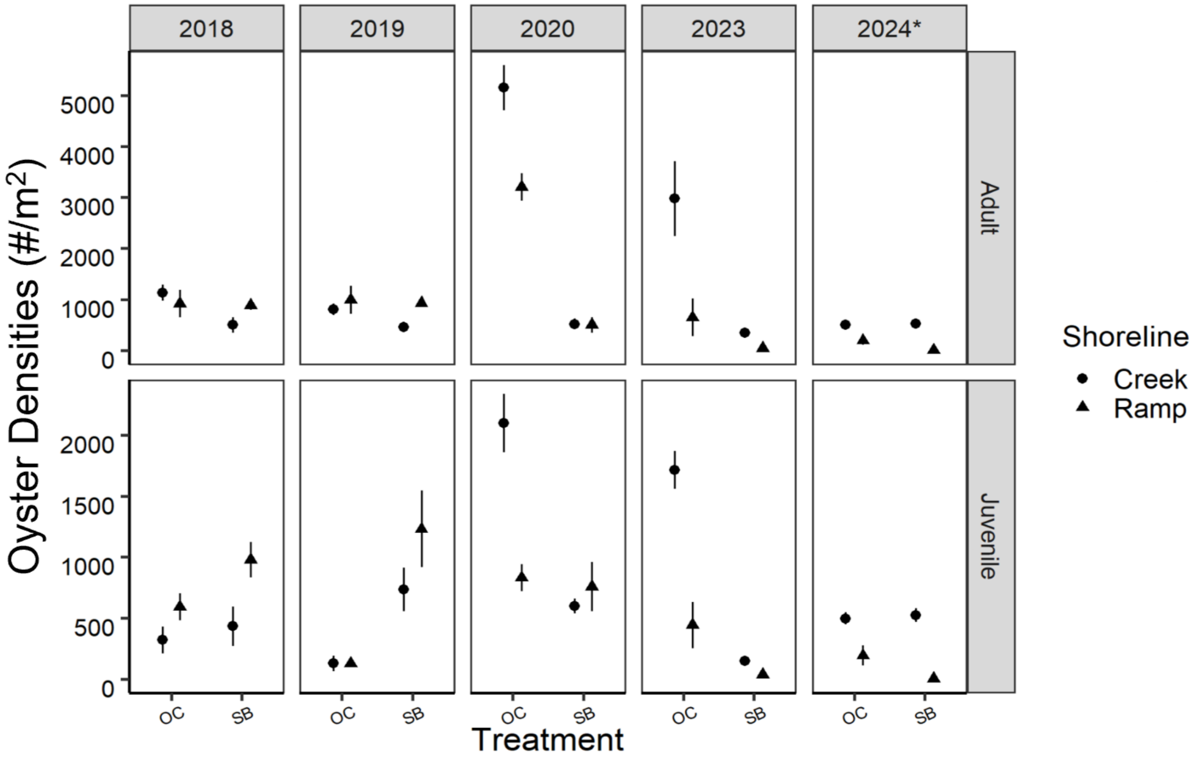

Figure 1. Oyster densities (#/m^2) by restoration treatment by year, 2018-2024. Results are also separated into size classes, with juvenile oysters (24mm or less) below adult oysters (25mm or greater). 2018-2020 data from Wellman et al. 2022.

Moreover, in 2020, the restored oyster reefs were within the "Optimal Growth Zone" (OGZ; -0.3m − -0.1m NAVD88) for intertidal oysters in central coast NC (Ridge et al., 2015), while in 2023, varying water levels resulted in all live oysters on the reefs residing above this ideal elevation range (Fig. 2.). However, by 2023 and 2024, live oysters were only found in areas that fell within or above the previously demarcated OGZ (Fig. 2). In 2024 specifically, live oysters on the ramp were only found on OC reefs within the OGZ; there were no live oysters on SB reefs on the ramp (Fig. 2). In all cases, Oyster Catcher reefs outperformed shell bag reefs in terms of oyster densities, offering some insight into the potential application of novel breakwater substrates to the issue of oyster reef restoration.

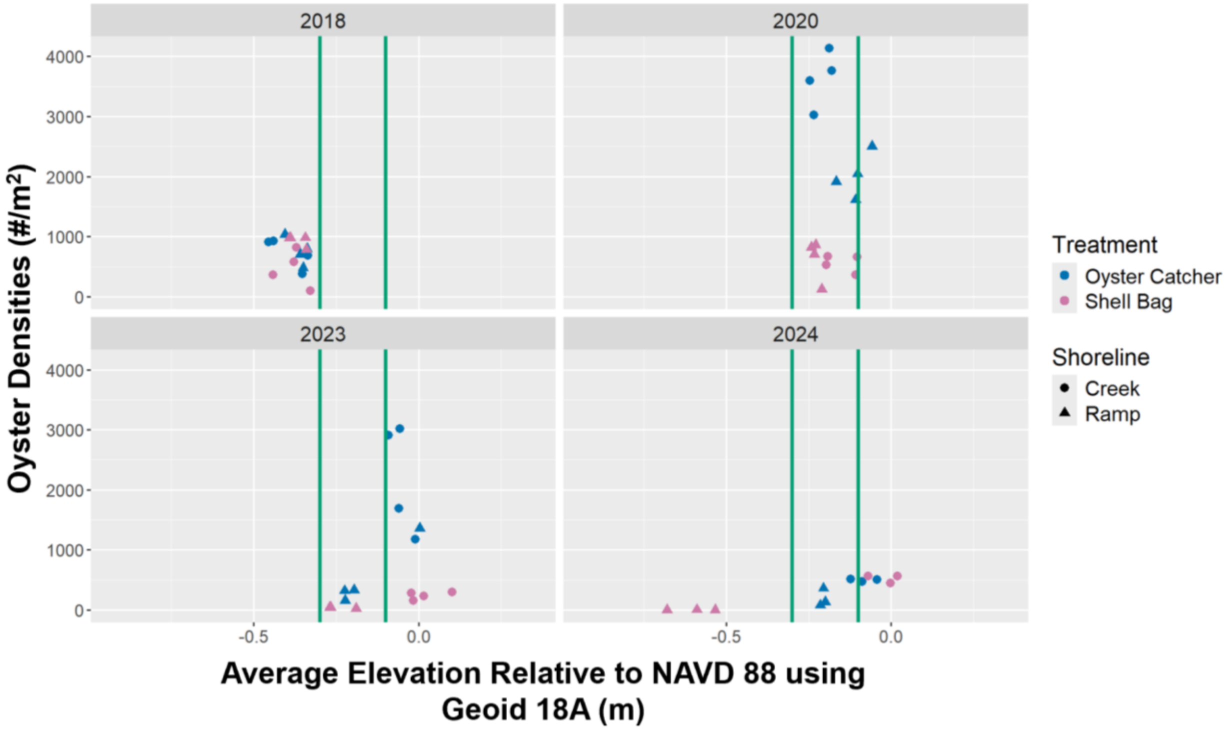

Figure 2. Average oyster density (#/m^2*) by average elevation (m) at which sample was taken for 2018, 2020, 2023, and 2024. The optimal growth zone for oysters is plotted in green (-0.3m − -0.1m NAVD88; Ridge et al., 2015). Treatment is differentiated by shape (OC reefs by circles, SB reefs by triangles).*

The following results for the biodiversity surveys (both host-level and parasite-level) for 2018-2023 only analyze changes that were analyzed along the protected shoreline at Carrot Island, as this is where we have access to early biodiversity monitoring data post-restoration (Moore et al., 2020). Free-living fish communities varied significantly with time relative to restoration (PERMANOVA, p=0.001), visible both in the early monitoring effort (with higher fish diversity represented in the one-year post-restoration sampling than in the pre-restoration fish communities; p=0.02) and with the addition of five-year post-restoration data (Fig. 3). The composition of the fish communities sampled five-years post-restoration varied more significantly from the one-year post-restoration data than the pre-restoration data (p=0.001 and p=0.003, respectively), with lower overall fish diversity affiliated with the pre- and five years post-restoration sampling. These results suggest that oyster reef restoration does not always retain the initially recruited fish community that follows restoration. Notably, species such as striped blennies, sheepshead, grey snapper, and gobies, were more abundant pre-restoration and one-year post-restoration, whereas oyster toadfish, for example, were more strongly affiliated with the five-year post-restoration fish assemblages (Fig. 3). However, given the distribution of the loose cultch reef points as well (broadly separated by year), these results may suggest a broader change in fish communities at the creek level (Fig. 3). In terms of treatment, a PERMANOVA considering time with post-hoc pair-wise tests assessing treatment revealed that none of the treatment levels (LC, SB, OC) varied significantly from one another (p>0.05). Considering this relationship with a time by treatment interaction term in the PERMANOVA was also found to be insignificant (p=0.093).

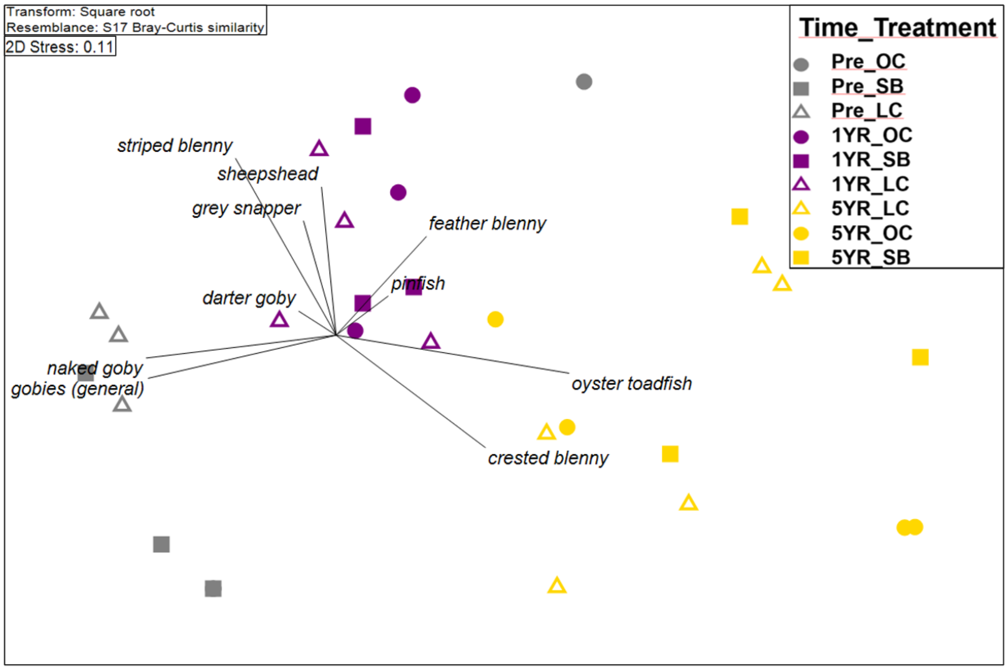

Figure 3. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index to assess the changes in free-living fish communities at Carrot Island sites by treatment (LC, OC, and SB) and time relative to the restoration effort (Pre-restoration, "Pre"; one-year post-restoration, "1YR"; and five-years post-restoration, "5YR"). The overlaid vectors represent the species that drive the distribution of the plotted samples.

The composition of free-living crab communities at Carrot Island increased in biodiversity over time; a PERMANOVA integrating the effects of time and treatment (as well as the interaction of these two variables) on the crab community composition and found time to be a significant factor (Fig. 4; p=0.003). Notably, further post-hoc pair-wise comparisons of time periods revealed five-year post-restoration crab communities to be significantly different from pre- and one-year post-restoration communities (p=0.003 and 0.006, respectively), while the earlier two sampling efforts did not differ significantly from one another (p=0.192). It is likely that porcelain crabs, as well as *P. herbstii, D. sayii,* and stone crabs are driving the differences marked in the 2023 sampling (Fig. 4). The interaction of time and treatment was not significant (p=0.842). While treatment was also found to have no statistically significant effect in the PERMANOVA (p>0.05), as with the free-living fish communities, points representing the free-living crab communities at loose cultch reefs seem to also separate by sampling year (Fig. 4).

This may once again indicate that there is a broader creek level shift in community composition at play (Fig. 4).

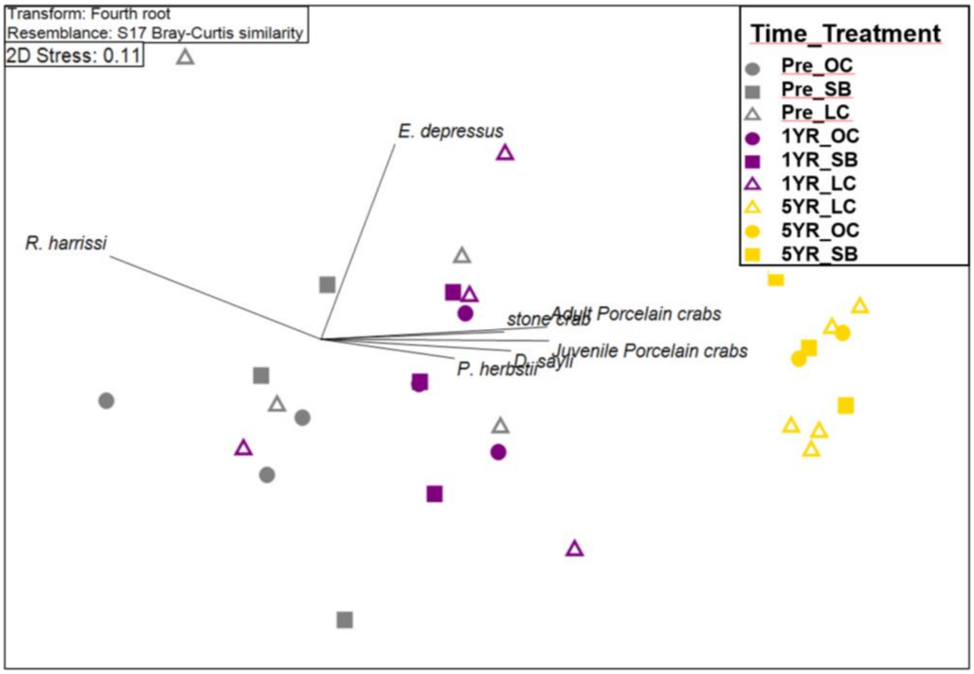

Figure 4. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index to assess the changes in free-living crab communities at Carrot Island sites by treatment (LC, OC, and SB) and time relative to the restoration effort (Pre-restoration, one-year post-restoration, and five-years post-restoration). The overlaid vectors represent the species that drive the distribution of the plotted samples.

Porcelain crabs (*P. armatus*) were not noted in the early biodiversity monitoring efforts (Moore et al., 2020; Blakeslee et al. 2024) and likely drove some of the differences seen between the 5-year post-restoration communities and those of earlier monitoring (Fig. 4). A more thorough analysis of the total porcelain crab abundances in 2023 revealed that they varied significantly by sampling month, with the highest abundances in October (Fig. 5; Kruskal-Wallis, p=6.6004e-05). A GLM considering the interactive effects of sampling month and treatment on total porcelain crab abundance revealed that OC was significantly lower than LC reefs, while LC reefs did not significantly differ from SB reefs (p=0.0005 and 0.87, respectively).

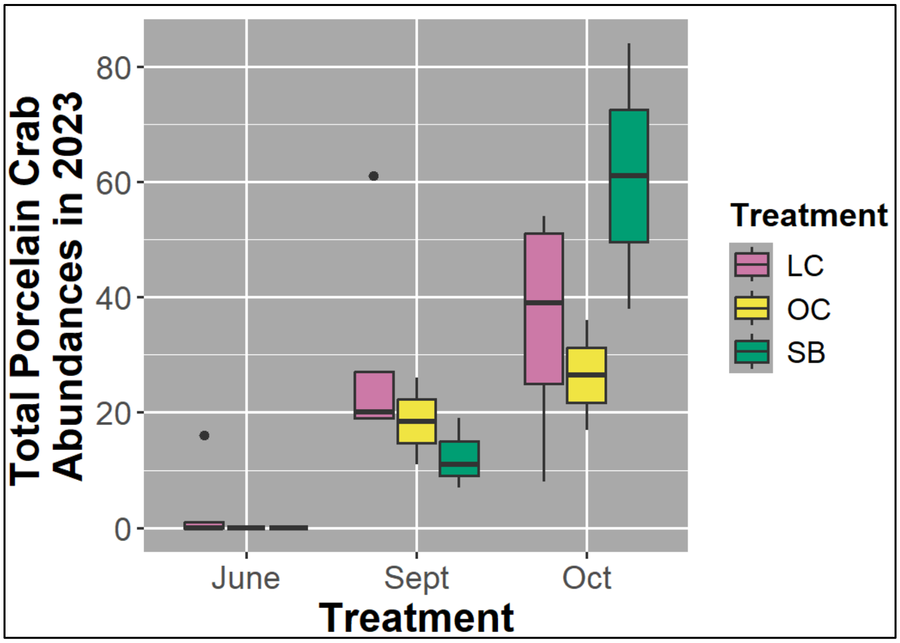

Figure 5. Total abundances of porcelain crabs in 2023 by time (months) and by treatment.

In terms of parasite diversity, initial assessment via an nMDS plot suggests markedly different fish parasite communities relative to time since restoration, with many of the "1YR" points clustered near the lower portion of the figure while the "5YR" and "Pre" points are located primarily in the top half of the plot (Fig. 6). Nematodes and cestodes appear to be driving much of the distribution of the five-year post-restoration points, and trematodes and monogeneans seem to be closely affiliated with this time point as well (in addition to the pre-restoration data) (Fig. 6). Many of the restored sites across all years (depicted as the shaded circles and squares for Oyster Catcher and Shell Bag, respectively) are also primarily clustered in the middle of Fig. 6, while the loose cultch sites (shown as unshaded triangles) are less tightly affiliated. However, PERMANOVA results integrating the effects of time and treatment suggest no significant influences of any of these factors ($p>0.05$) nor the interaction between time and treatment ($p=0.162$). Further study of these factors in PERMANOVA runs where each lone factor was included revealed a significant effect of time only ($p=0.014$), with post-hoc pair-wise tests finding significant differences between one-year post-restoration and both pre-restoration and five-years post-restoration ($p=0.001$ and 0.044, respectively).

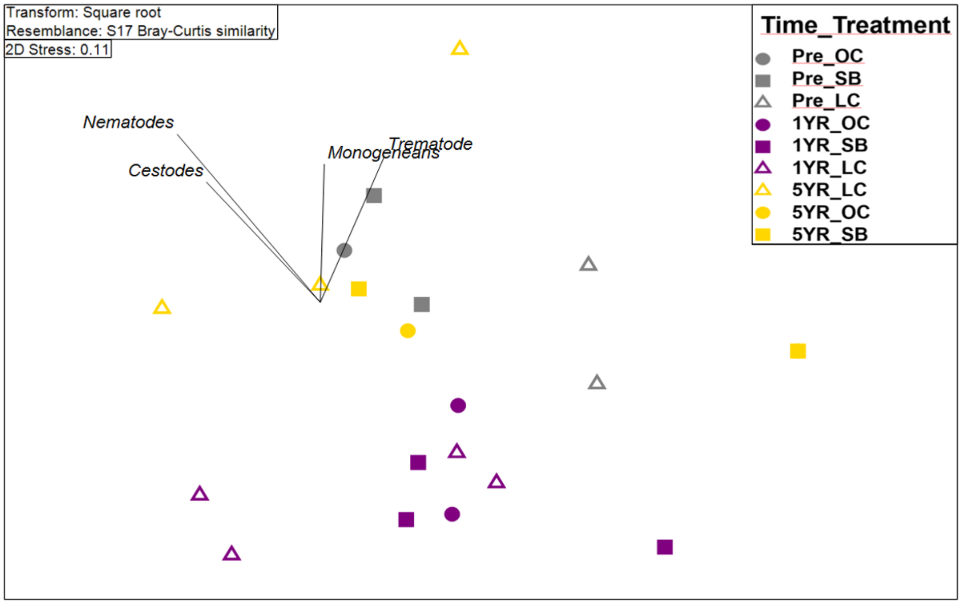

Figure 6. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index to assess the changes in fish parasite communities at Carrot Island sites by site type (loose cultch, Oyster Catcher, and Shell Bag) and time relative to the restoration effort (Pre-restoration, one-year post-restoration, and five-years post-restoration). The overlaid vectors represent the species that drive the distribution of the plotted samples.

Crab parasite communities were analyzed using a cluster analysis, and this largely supported the temporal separation of the data, as the majority of sample points clustered with points that shared a sampling year. Specifically, 2023 points in one cluster, 2020 in another, and 2018 in a third (Fig. 7). While there are some points mixed in with the pre-restoration cluster that are from later sampling efforts, the majority are LC reefs, and there is more overlap between the 1-year and 5-year post restoration clusters (Fig. 7). A PERMANOVA also highlighted time alone as a significant influence (p=0.001), but not treatment significant (p=0.253), nor the interaction of these two variables (p=0.463). Post-hoc pair-wise tests further indicated that each time period of sampling significantly differed from the rest (p=0.003 for all), further emphasizing the visual distinctions represented in Fig. 7. However, treatment appears to be more distinct when assessed via Fig. 7, particularly when studying the 1-year post-restoration sampling points, as shell bags points are closest to one another, and similar patterns are seen with the Oyster Catcher and loose cultch samples from this time period (Fig. 7).

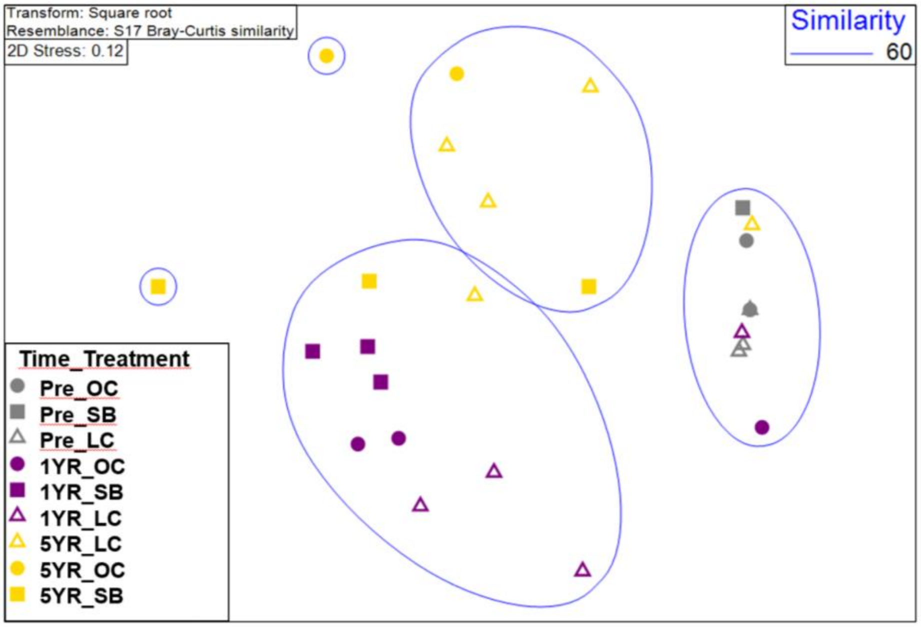

Figure 7. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index and a cluster analysis to assess the changes in crab parasite communities at Carrot Island sites by site type (loose cultch, shell bag, and Oyster Catcher) and year (pre-restoration in 2018, one-year post-restoration in 2019, and five-years post-restoration in 2023). The cluster analysis shows groupings of communities that are >60% similar to one another (shown in light blue circles).

As for snail parasite communities, a cluster analysis grouping communities with >50% similarity showed restored reef communities well-mixed with those of loose cultch reefs across all time points, and placed 5 year restored reefs quite close to the loose cultch reefs of the same time point (Fig. 8). This indicates that the restored reefs may now be more closely resembling their more-natural counterparts in terms of their snail parasite richness and abundances (Fig. 8).

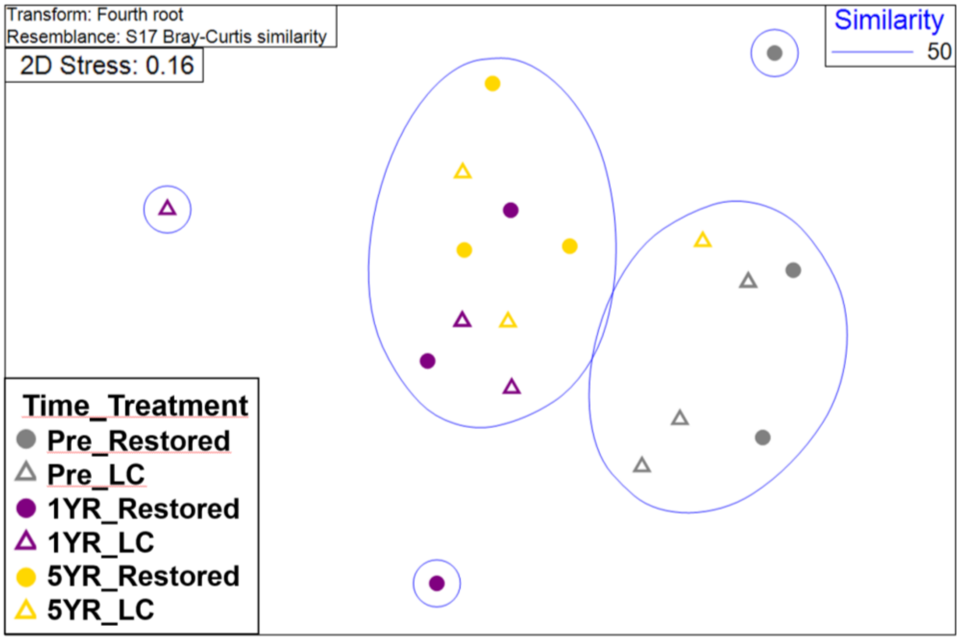

Figure 8. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index and a cluster analysis to assess the changes in snail trematode communities (in terms of prevalence and richness) at Carrot Island sites by site type (LC and restored, which includes SB and OC) and year (pre-restoration in 2018, one-year post-restoration in 2019, and five-years post-restoration in 2023). The cluster analysis reveals groupings of communities that are more than 50% similar to one another (shown in blue circles).

Additionally, as for the comparison of 2023 and 2024 data, I have thus far completed a preliminary assessment of the variation in the free-living communities at Carrot Island (Fig. 9). This plot not only highlights a clear distinction in these communities across this year, but the creek shoreline points are also more tightly clustered together in the center of the plot while the ramp plots are much more distributed around the periphery of the plot.

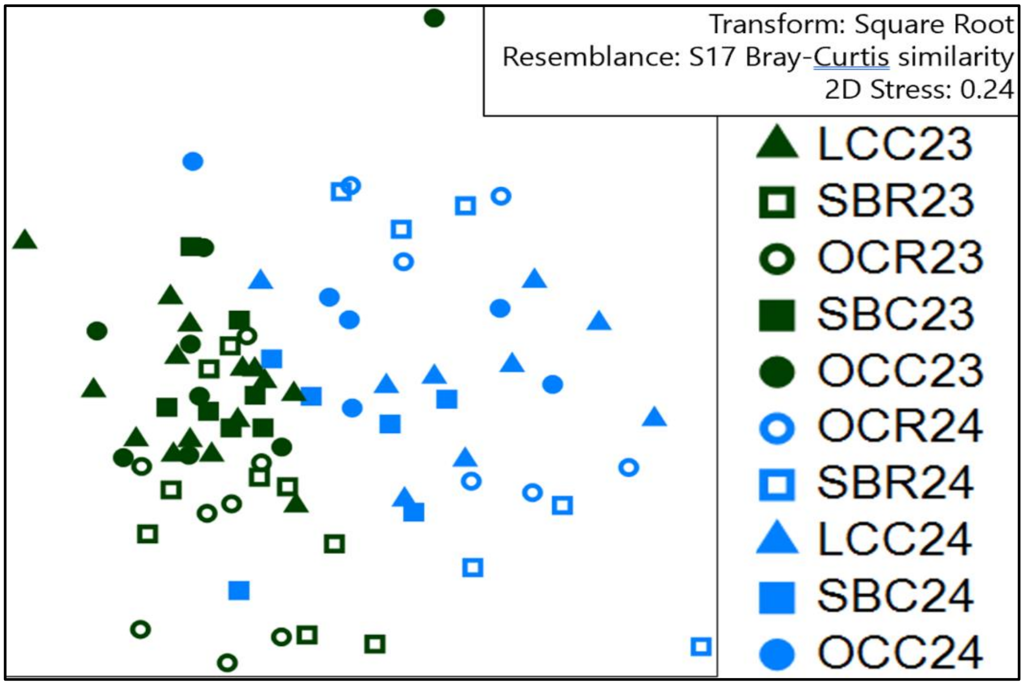

Figure 9. An nMDS plot made in Primer 7 using the Bray-Curtis similarity index and a cluster analysis to assess the changes in free-living communities at Carrot Island sites by site type (as shape; LC=loose cultch SB=shell bag, and OC=Oyster Catcher), shoreline (as shading; C=creek, R=ramp), and year (as color; 2023 and 2024).

Discussion:

My results broadly reveal that the habitat and biodiversity affiliated with oyster reef restoration plots can vary markedly with time following restoration, consistent with previous literature describing variation in biodiversity and habitat metrics of restoration assessment over both short- (Moore et al., 2020; Wellman et al., 2022; Humphries et al., 2011; Overton et al., 2024;) and long-term (Huspeni & Lafferty, 2004; Moore et al., 2023; Grizzle et al., 2021) periods of monitoring efforts. In terms of changes in habitat characteristics, previous monitoring efforts at Carrot Island initially suggested increasing oyster densities with time following restoration (Fig. 1; Wellman et al., 2022). While Oyster Catcher continued to outperform shell bag reefs in 2023 and 2024, our results also show significantly lower oyster densities relative to those recorded at the conclusion of the initial monitoring effort of 1-2 years, suggesting that what had initially looked promising may not withstand the test of time (Fig. 1; Wellman et al., 2022). These results are consistent with previous work assessing oyster reef restoration (using oyster density as an evaluation metric) over time, with a study assessing these efforts in New Hampshire recording peak oyster densities one-year post-restoration, with notable decline by the sampling period three-years post-restoration (Overton et al., 2021). Similar results were seen in subtidal oyster reefs in North Carolina as well, with high oyster densities earlier on in the restoration monitoring period followed by declines after only a few years (Puckett et al., 2018). In terms of the varying elevations of the live oysters on the reefs, the position of many of the 2023 and 2024 live oysters in the creek above the OGZ, and with lower overall densities, may be attributed to the more variable water levels in these years that could have shifted the OGZ for this area (Fig. 2; Ridge et al., 2015). As for the exposed ramp shoreline, the live oyster densities were significantly lower in 2024, though the only live oyster densities were found within the previously documented OGZ (Fig. 2).

In terms of free-living fish communities, the significant differences between both post-restoration sampling periods and the pre-restoration sampling suggest an overall positive impact of the addition of oyster habitat to the system on fish communities, and potentially a broader pattern of change in these communities over time without a specific treatment-driven effect (Fig. 3). Our results suggest that some of the species driving the results of the 1-year sampling point include striped blennies, various gobies, sheepshead, and snapper, while one of the main species affiliated with the 5-year sampling point is oyster toadfish (Fig. 3). It is worth noting that oyster toadfish have been highlighted as an ecologically significant fish species in previous research (Moore et al., 2023). Additionally, several of the species that distinguished the one-year post-restoration sampling event (including sheepshead, gray snapper, among others) are more broadly recognized as transient fishes in regards to reefs (Coen et al., 1999; Gittman et al., 2016). Thus, the stronger association of toadfish, for example, may be reflective of a stronger reef-resident community five-years post-restoration than earlier monitoring efforts documented. These results are also consistent with previous research regarding oyster reef restoration, which has noted quick recruitment of mobile (transient) nekton to restored habitat (La Peyre et al., 2014; Moore et al., 2020; Humphries et al., 2011), and thus suggest that oyster reef restoration does not always retain the initially recruited fish community that follows restoration.

For free-living crab species, the recorded differences over time appear to be largely driven by the introduction of the green porcelain crab (*P. armatus*) to the system in between the initial monitoring effort and the five-year post-restoration sampling, but three other xanthid crab species – *D. sayii, P. herbstii,* and *Menippe spp.* – also appear to be more broadly driving the visible distinction of the five-year post-restoration communities (Fig. 4; Blakeslee et al., 2024).

While treatment again showed no significant differences through time, more specific analysis considering sampling month (in 2023 only) and treatment revealed higher average porcelain crab abundances in September and October (relative to June) and in shell bag and loose cultch reefs relative to Oyster Catcher reefs (Fig. 5). The prominence of *P. armatus* in this sampling period is not without precedent, as this crab was the most abundant crab species recorded at various sampling locations in its expanded range in earlier work (Mack et al., 2019).

The variation in fish parasite communities at Carrot Island is particularly notable with time, with one-year post-restoration communities significantly different than those affiliated with the other two sampling periods (pre- and five-years post-restoration, with p=0.001 and 0.044 via PERMANOVA, respectively) (Fig. 6). The initial monitoring effort demonstrated significantly lower parasite prevalence across taxa one-year post restoration relative to pre-restoration data, and it is intriguing to see that certain species, while still not reaching their pre-restoration prevalence, are now beginning to more closely resemble these data overall in the five-year post-restoration data (Fig. 6; Moore et al., 2020). Thus, the comparison of the fish parasite data over time as well as to the free-living fish data is intriguing, as the cumulative effect may suggest a more ecologically complex and interconnected oyster reef resident community at both the host and parasite levels (Moore et al., 2023). The snail parasites assessed in this study offer an additional angle in support of the broader trend (of parasites as effective restoration indicators) suggested by the fish parasite data and previous literature (Moore et al., 2023). The utility of snail trematode parasites as an indicator in this way has been established in previous work, with one study finding significant increases in trematode prevalence at restored saltmarsh sites relative to control sites after six years (Huspeni & Lafferty, 2004). The snail parasite communities of 5 year restored reefs at Carrot Island being adjacent to the loose cultch reefs of the same time point (Fig. 8) highlight how this restoration effort may be allowing restored reefs to function similarly as loose cultch reefs in their support of this particular community. As such, this represents another point of my results in which parasites may serve as a more effective indicator of restoration success than their hosts. It is important to note, however, that the crab parasite data does not generally resemble the same trend as the fish and snail parasite data, with greater parasite diversity and prevalence one-year post restoration rather than increasing with time (Fig. 7).

Broadly, these results largely reveal that there are many ways in which the overall "success" and progressive development of a restored oyster reef system can be evaluated, and that different methods of assessment offer unique insight into the overall development of the system in response to the surrounding conditions and dynamic communities within. This is a particularly useful and important notion to consider in the context of large-scale changes underway, such as climate change and sea level rise, as it will be critical to continue restoring habitat lost to these forces and to closely monitor these restoration efforts to ensure they withstand the test of time, even in these circumstances. The significant losses in oyster densities, particularly on the exposed shoreline at Carrot Island, can be particularly useful to consider for the Reserve and future restoration practitioners in the area, as it appears that the varying water levels and location these reefs were subject to ultimately led to their severe degradation over time, despite initial location and success in the previously determined OGZ for this area. Furthermore, my work reveals that the results of short-term monitoring do not necessarily indicate the long-term results of restored oyster reefs, for better or worse (Moore et al., 2020; Wellman et al., 2022). As such, regardless of the intent of the restoration effort and the method(s) used to assess it, it is critical for oyster restoration practitioners to invest the time and resources to evaluate its condition over time rather than reaching an ultimate determination of restoration success or failure after short-term initial monitoring.

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Outreach:

Throughout the course of this project, I have presented preliminary findings as I have gathered them in a variety of venues, including conferences and public outreach events. In 2024, I presented some early results in the form of a lightning talk ("Biodiversity and habitat complexity as indicators of community assembly following oyster reef restoration") at the NC Coastal Conference in November and as a poster at the Coastal Reserve Symposium in May. I have also described the general principles guiding my work and the significance of my project in a public tag-team lecture ("Zombie Crabs: zombie crabs and other parasite invasions in our estuary") at the North Carolina Estuarium in Washington, NC on October 10th, and in a less lecture-oriented event at the same location for their "Estu-SCARE-ium" event on October 17th. I also took part in other volunteering and outreach events, including an outreach initiative with a local high school in Washington, NC, and a Career Day event at a local middle school. I have also presented the findings of this work at three conferences in 2025: the Benthic Ecology Meeting in Mobile, AL, the American Fisheries Society Tidewater Meeting in Morehead City, NC, and the Helminthological Society Meeting in Woodbridge, VA. In total, this has resulted in the distribution of my findings to an array of community members, academic researchers, non-academic professionals (i.e., restoration managers), and other significant stakeholders. As such, the use of the results by these audience members varies. From some of the questions I have received in the conference settings with primarily academic and non-academia related professionals (for example, "So, should we be transitioning from using the traditional shell bag approach to these novel breakwater structures?”), I believe my findings are providing valuable insight into the proper selection of restoration materials for both restoration’s sake along as well as a tool to study ecological assembly, knowing that these novel breakwater structures appear to have held up better with time than their traditional counterparts. I also believe these findings have been useful for the public audiences that have received them, as it provides an argument in favor of restoration, long-term monitoring, and living shoreline structures. Thus, this will hopefully provide some background for residents in areas considering restoration, either in a public area or if they are interested in installing living shorelines on their own property.

Students supported:

Sixteen undergraduate students (Ulises Mendoza, Curtasja Brooks, Trip Brooks, Louise Zango, Matt Lopez, Nordan Ludivico, Heaven Moore, India Speight, Nykeria Garvin, Ayden Campbell, Ashley Eakes, Lauren Johnson, Prisha Singh, Rachel Walker, Brianna Stanley, and Jake McBroom) were trained in a variety of field and laboratory procedures throughout the duration of this project, and provided invaluable support to the completion of the affiliated sample collections and subsequent processing. Students were invited to take part in sample collection (sorting through passive sampling devices, actively sampling reef structures, etc.), in-lab processing (measuring oysters, identifying fish and crabs to host level and collecting demographic information, dissecting fish, snails, and crabs to search for parasites, etc.). Specifically: LZ and ML worked on oyster processing; AC, CB, BS, JM, LJ, PS, and RW all worked on crab processing; HM, NL, IS, and AE all worked on both snail and crab processing; TB worked on snail processing and helped in the field; NG worked on crab processing and helped in the field; and UM worked on oyster, crab, snail, and fish processing.

These students either participated in the lab while completing undergraduate independent study credits (AC, NG, RW, BS, PS, JM, LZ, ML, and TB), with funding from a grant (HM, IS, NL, LJ), or a combination of these two options (UM, AE, CB). Many students were also given the opportunity to use the overarching project as a source of data for their own mini projects, and presented at local symposiums (ECU’s Research and Creative Activities Week, for example). This gave them the chance to practice their technical writing skills, compiling a research poster, conveying their results to the public and other academics, and answering questions from their audiences about their work.

News/Media coverage of this project:

This project has been described in two blog posts. The first was published through the ECU Water Resources Center, as I applied to join their Water Scholars program early on in my graduate studies. This achievement provided me with some initial funds to start the 2023 monitoring and gave me the chance to begin describing me work (see: Water Scholar Awardee Spring 2023: Grace Loonam | Water Resources Center | ECU). The second blog post was on the NC Reserve’s Tidal flat blog post, and describes the entirety of project more holistically (see: Comparing oyster restoration methods and parasites as ecosystem indicators | NC DEQ).

Data Management Plan Progress: (Please document that you have completed the steps outlined in your project's initial data management plan. Has your data management plan changed? If the data management plan is not finalized, please include a timeline for its completion.

At this point, all data sheets related to the field or lab processing of the data collected throughout this project are stored in a binder in the Blakeslee/Gittman lab space. These data have also all been scanned and uploaded to a shared Teams folder between the Blakeslee/Gittman labs designated for the Carrot Island project. The vast majority of this data has also been entered separately into Excel, QA/QC'd, and stored in the same Teams folder – this process is still in progress and expected to be completed by the end of August 2025.

With regards to publication and data accessibility, this data will become publicly available when my thesis is set to be published by the University. I have slightly adjusted this timeline relative to that initially proposed, as I have set a one-year embargo on this publication to allow me time to finalize some of the academic publications and submit them to peer-reviewed journals. Thus, the data should ultimately be available in open access journals within a year. The data has also been presented at several academic conferences over the last year as initially proposed, including the 2024 NC Coastal Conference, the 2025 AFS Tidewater Meeting, the 2025 Helminthological Society Meeting, and the 2025 Benthic Ecology Meeting.