

INFLUENCE OF SHORELINE STABILIZATION STRUCTURES ON NEKTON AND
AVIAN ASSEMBLAGES IN SALT MARSH HABITATS

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ABSTRACT

Erika Lynn Young: Influence of Shoreline Stabilization Structures on Nekton and Avian Assemblages in Salt Marsh Habitats
(Under the direction of Charles H. Peterson)

Bulkheads represent a hardened shoreline stabilization structure designed to inhibit shoreline erosion and damage to coastal property. As sea level continues to rise, natural marshes can be sustained by transgression landward. The presence of a bulkhead, rip-rap revetment, or hybrid of those two, however, prevents transgression by fixing the location of the upper marsh. Simultaneously, as sea levels rise, wind-driven waves from storms and boat wakes erode the lower edge of the marsh, inducing slumping of the marsh platform into the sound. So marsh width, and thereby area, is declining in the squeeze between the fixed bulkhead and the rising waters, ultimately leading to its disappearance. We quantify and compare use by nekton (fish and crustaceans) and birds of marshes in front of bulkheads but differing in width from bulkhead to shoreline edge, and use width as an inverse proxy for time since marsh establishment to infer temporal habitat changes. Within three geographic areas of North Carolina, meteorologically driven Pamlico Sound in Kitty Hawk Bay, a strongly astronomically driven southern region along the Intracoastal Waterway around Wilmington, and a central region in Bogue Sound with mixed tidal forcing, we established five bulkheaded marsh sites with varying marsh widths of 0 to 40 m plus 1 non-bulkheaded marsh. Intensity of marsh use (abundance in fyke nets) for both fish and crustaceans exhibited similar responses of linear increase with marsh width over three geographic areas, two seasons, and two years. This repeated pattern implies that nekton use in marshes below shoreline stabilization structures declines over time as marsh width is reduced.

Nekton density per unit marsh area sharply declined with increasing marsh width, implying that even the smallest marsh remnants have important habitat functions and deserve continued protection. Community compositions of birds using the shorelines varied significantly among the three regions, which differ dramatically in tidal excursion. We conclude that coastal marsh condition influences important aspects of bird use and marsh itself is critical to sustaining a diversity of bird guilds. The continued installation of bulkheads will promote marsh loss as sea level rises, ultimately degrading avian ecosystem services of coastal marshes.

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LIST OF ABBREVIATIONS

| | |
|-----------|--|
| ANOVA | analysis of variance |
| BMM | bulkhead medium marsh |
| BMM2 | bulkhead medium marsh 2 |
| BNM | bulkhead only |
| BSM | bulkhead small marsh |
| BWM | bulkhead wide marsh |
| CICEET | Cooperative Institute for Coastal and Estuarine and Environmental Technology |
| CPUE | catch-per-unit-effort |
| CWA | Clean Water Act |
| KH | Kitty Hawk Bay |
| NM | natural marsh |
| nMDS | non-metric multi-dimensional scaling |
| PERMANOVA | permutational analysis of variance |
| PKS | Pine Knoll Shores |
| SLR | sea-level rise |
| WILM | Wilmington and Oak Island |

CHAPTER 1. NEKTON UTILIZATION OF MARSHES IN FRONT OF BULKHEADS AS A FUNCTION OF MARSH WIDTH.

Introduction

Coastal marsh has long been identified as one of the most important habitats in provision of valuable ecosystem services (e.g. MEA 2005). Accordingly, coastal marsh benefits from explicit protections afforded by federal legislation in the U.S.: the Clean Water Act (CWA) of 1972 mandates preservation of coastal marsh by requiring that any development project first avoid, then minimize where avoidance is impossible, and finally mitigate for any coastal marsh loss. Despite the Clean Water Act protections, owners of estuarine shoreline real estate are allowed to construct various shoreline protection devices to prevent shoreline erosion and damage to construction on land. The permitted position for bulkheads, revetments, and hybrid combinations falls at the top of regularly flooded elevation in the marsh, often in higher salinity locations indicated by the transition between *Spartina* marsh lower on shore and *Juncus* marsh on the high shore. Siting such shoreline protection structures above the low marsh implies little or no damage to that habitat from construction itself. However, under rising sea levels and shoreline erosion associated with global climate change, the presence of a fixed structure above the low marsh will act to inhibit the landward transgression of coastal marsh habitat as sea levels rise (Titus 1998), which is how marsh is naturally sustained when water levels rise globally. Consequently, the construction of fixed shoreline protection devices, even when sited according to the CWA, can lead over time to marsh habitat loss via coastal “squeeze” of the habitable zone for intertidal vegetation. Here we employ a “space-for-time” substitution design to enable an assessment of how well one of the important ecosystem services of coastal marsh, habitat

provision to fish and crustaceans, may be affected by the combination of sea level rise and prevention of transgression.

Northeastern North Carolina ranks third, after the Mississippi Delta region of Louisiana and the Everglades in South Florida, in land area at risk to inundation through sea-level rise (area with elevation under 1.5 m: Titus and Richman 2001). Virtually all of this area at high risk in North Carolina is coastal marsh habitat, especially common along the Pamlico Sound shorelines. Current linear rates of shoreline erosion are reported as high as 0.3 m to 12 m per year and increasing along various Pamlico Sound shores (Riggs 2001). Gittman et al. (2015b) conducted a synthesis of all hardened shoreline treatments (vertical bulkheads, sea walls, sloped rip-rap revetments, groins, jetties, or breakwaters) up through 2014 along exposed ocean and sheltered estuarine shores of the U.S. by county, finding that 14% of Atlantic sheltered shores were already hardened by shoreline stabilization structures. Analysis of possible drivers of increased hardening of Atlantic sheltered coasts revealed that 41% of the variability among counties is explained by housing density (Gittman et al. 2015). This relationship implies that property protection against erosion, flooding, and storm damage is a major motivation for shoreline hardening, ultimately leading to coastal marsh loss. Along Maryland (Titus 1998) and North Carolina (Currin et al. 2010) coasts, bulkheads are often the shoreline protection structure of choice for individual homeowners, motivated by the ease of obtaining permits, perceived affordability, presumed effectiveness, and low costs of long-term maintenance. Over an estimated 9000 miles of estuarine shoreline in North Carolina (Currin et al. 2010), from 1984 to 2000 the state issued permits for approximately 457 miles of bulkheading (Deaton et al. 2010).

Here we report results of sampling fish and crustaceans using coastal marsh habitat at high tide during day and night for two seasons (spring and fall) in three dramatically contrasting

and geographically separated coastal marshes. At all three locations, we compare this nekton use among bulkheaded marshes of differing widths, inferred to be indicative of expected marsh thinning over time under conditions of rising sea level. We include at each location a bulkheaded shore lacking marsh, the ultimate outcome of persistent sea-level rise, and a natural marsh lacking any shoreline stabilization structure, the condition reflecting natural marsh dynamics. The new insights we provide into how nekton utilization varies in bulkheaded marshes of varying width permit a quantitative prediction of effects of bulkheading on nekton utilization of coastal marsh as a function of time since construction of the shoreline stabilization structure. On the basis of these predictions, we compute the quantitative losses in this important ecosystem service of nursery habitat provision that flow from over varying but unquantified periods of time allowing bulkhead construction to block transgression of marsh up-slope with sea-level rise.

Materials and Methods

Nektonic fishes and crustaceans were quantitatively sampled in 2009 and 2010 from six sites in each of three geographic regions (Fig. 1) of coastal North Carolina: in the northern region, Kitty Hawk Bay (KH), in the central region Bogue Sound at Pine Knoll Shores (PKS), and in the southern region, located along the Intracoastal Waterway in Wilmington on Harbor Island and Oak Island (WILM). Water-level fluctuations in the northern region are meteorologically driven by winds and the marsh in KH is dominated by *Juncus roemerianus* and *Phragmites australis*, with water surface thickly covered by *Myriophyllum spicatum*. Water levels in the central region are driven jointly by meteorological and astronomical forcings (Voss et al. 2013), with an astronomical tide of about 1 m and marsh dominated by *Spartina alterniflora* along the PKS shoreline. The southern region is characterized by 2-m astronomical tides, with a marsh shore dominated by *S. alterniflora*.

For each geographically constrained study area (KH, PKS, and WILM), six sites were chosen with similar wind-wave exposure and similar fetch, differing by the absence (1 site: “natural marsh”) or presence of a continuous bulkhead (5 sites) and differing in marsh width (defined as the vertical extent). Thus providing: 1 site with “no marsh”; 1 site with “narrow” (3.5-4.3 m), even discontinuous, marsh; 2 marsh sites of “medium” width (10.7-16.0 m); and 1 site with “wide” (18.3-22.8 m) marsh. Natural marsh sites ranged in width from 15.0-39.1 m.

Sampling of nekton utilizing the marsh (or the area that marsh would have once occupied for the bulkheaded no-marsh site) was achieved by setting a fyke net at high tide and collecting the catch at low tide 6 h later after marshes had maximally drained. Fyke nets (Fig. 2) were custom-made 3-m long using 3 square aluminum frames measuring 0.9 m², 0.8 m², and 0.8 m², varying only slightly with distance away from the marsh, and covered with 3.1 mm delta, knotless, nylon mesh treated with a water-based UV inhibitor coat. The two mesh wings of 5 m each attached to either side of the front frame. The terminal end of each wing had a sleeve of mesh that fit over a PVC pole, the position of which was anchored into the marsh substrate by a rebar stake inserted within the PVC. An additional pocket of mesh extended into the third frame to form a 10.2 cm mouth that led to the cod-end bag held taut by an erect PVC pole inserted into the sediment. The fyke nets were placed at the marsh edge with the wings extended during peak high tide, where they remained until low tide allowing them to trap nekton while waters receded and nekton passed through the throat in the third frame and into the cod-end bag.

Sampling of nekton occupying unvegetated ground seaward of the marsh was achieved by deploying research gill nets (Fig. 3) that were 40 m long with six, 6.6-m monofilament panels, each of a different mesh size: 2.5 cm, 5 cm, 7.6 cm, 10 cm, 12.7 cm, and 15 cm. Gill nets were installed at peak high tide running parallel to the marsh edge at approximately 3-10 m distance

and remained deployed for a 2-hour soak time to limit risk of by-catch entanglement with mammals or reptiles.

Only the central-region was sampled in replicate years (2009 and 2010) to assess potential interannual variability. The northern and southern regions were sampled only in 2010. During each year of sampling, the complete set of sites was sampled in two separate seasons, corresponding crudely to spring and fall. Furthermore, each site was sampled for 2 replicate days and 2 replicate nights during the sampling period. Sampling was conducted in the central sites for two years on June 25-29 and October 20-26 in 2009 and June 10-16, and October 3-7, 2010. The northern sites were sampled on June 21-25 and September 24-29, 2010. The southern sites were sampled from May 26- June 4 and October 11-20, 2010. All dates were chosen within seasonal limits based on predicted persistence of suitable weather and (wind-)tidal patterns. The more challenging night-time sampling was included because sampling only during daylight often underestimates species diversity and/or abundance of nekton (Stoner 1991). By using 3 fyke nets and 3 gill nets, each of the 6 sites being sampled simultaneously received either a gill net or fyke net during each sampling period. The pattern of net type deployment alternated spatially. For example, on day 1 of sampling, sites 1, 3, and 5 received a gill net and sites 2, 4, and 6 a fyke net. This same deployment pattern was repeated for night 1 of sampling. On day/night 2, net types were switched among sites so that sites 1, 3, and 5 received a fyke net, while sites 2, 4, and 6 received a gill net. This process was repeated for each day-night pair until 2 replicate day/night cycles were sampled.

Trophic levels were determined for nekton by reviewing species-specific literature and cross referencing with FishBase (Froese and Pauly 2009). For species not thoroughly described

by FishBase, the trophic level was calculated by hand using methods described by Christensen and Pauly (1992) using the following formula:

$$TL_i = 1 + \sum_j (TL_j \cdot DC_{ij}) \quad (\text{Eq. 1.})$$

TL is the trophic level of the prey j and DC_{ij} represents the percentage of the diet that prey item makes up. The weighted average trophic level for each species was then calculated and trophic levels of nekton were compared by net type.

Upon collection, all nektonic organisms captured in the gill and fyke nets were immediately placed on ice and stored in a 0° C freezer. Labeled samples were returned to UNC-IMS for identification and enumeration. Carapace length and sex were determined for decapod crustaceans. Total length, fork length, standard length, and total weight was measured for each fish accordingly.

To calculate average total nekton densities in each marsh type, we defined the area that each net was sampling. The area was measured as the length (m) of the fyke net extended to the front of the bulkhead multiplied by the distance (m) of the fyke net wing to the bulkhead. This calculation allowed us to use the 0 m marsh width sites. For example, the narrow marsh width in PKS was 3 m and the length of the fyke net was 11 m, so the area sampled for density is 33 m².

Statistical Analyses

Nekton abundance values are reported (Appendix 1) as CPUE (catch-per-unit-effort). CPUE was determined by dividing the total nekton catch by 6 hours for fyke nets and 2 hours for gill nets to represent the number of hours of soak time for each net type. Linear and nonlinear regressions were conducted to relate abundance of total nekton as well as of fish and crustaceans separately to marsh width by appropriate combinations of net type, geographic region, season,

and day/night. For the only geographic region (central) in which sampling covered multiple years, we also employed linear regressions of total catch (separately for both spring and fall seasons) against marsh width to test for possible interannual variation.

To test whether nekton catch-per-unit-effort (CPUE) differs between night and day, we conducted paired t-tests that contrasted every independent pair of day-time and night-time CPUEs, which consisted of two pairs of day-night sampling results for each of the 8 separate samplings defined by year, geographic site, and season. Separate tests were conducted of fyke and gill net CPUEs, each with 16 independent day-night pairs. We conducted analogous t-tests to assess whether numbers of separate nektonic taxa (all nekton as well as fishes and crustaceans separately) differed between day and night in fyke net and gill net catches.

One-way ANOVAs were performed separately for $\log(x+1)$ transformed fyke and gill net CPUEs to test for differences among regions in nekton abundances (total plus fish and crustaceans separately). For the one-way ANOVAs, 2009 sampling data from PKS were excluded to hold year of sampling (2010) constant among the 3 geographic regions.

Regressions, t-tests, and one-way ANOVAs were performed in SigmaPlot 11 (Systat Software Inc.) and JMP[®] (Version 10.0.1 SAS Institute Inc., Cary, NC, 1989-2007).

Results

Pooled catches from our three geographic regions across coastal North Carolina, two (WILM and KH) of which were sampled in 2010 and one (PKS) in both 2009 and 2010, two seasons (spring and fall), equal day and night sampling effort, and two net types (fyke in marsh and scientific gill in waters below the marsh) totaled 7745 nektonic organisms representing 80 species. A total of 5858 fish from 70 species, 1887 crustaceans from 7 species. Three species of other taxa were also collected: 1 ctenophore (*Beroe sp.*), 1 brief squid (*Lolliguncula brevis*), and 1 green sea turtle (*Chelonia mydas*) (Appendix 1). We excluded ctenophores from analysis

because they are poorly sampled by nets and the single green sea turtle, which belongs to neither category of nekton, and we treated the one species of squid as a fish.

Total nekton CPUE in fyke net samples from the marsh, summing over day and night and the 2 seasons, exhibited a significant increase with marsh width in linear regressions, both with ($p=0.0007$) and without ($p=0.034$) inclusion of the natural marsh sites (Fig. 4A). These analyses used means of the two years of sampling in the central region (PKS) to avoid the overweighting that would result if each year's data were included separately.

Analogous linear regressions of fish only CPUE (Fig. 5A) and of crustacean only CPUE abundance (Fig. 6A) against marsh width revealed increasing abundances as marsh width increased, but at lower rates (slope for fishes about 30% and for crustaceans about 70% of the slope for total nekton). The relationship of abundance of fishes alone (Fig. 5A) and crustaceans alone (Fig. 6A) to marsh width exhibited statistical significance ($p=0.012$) and ($p=0.006$), respectively, when natural marsh was included and when natural marsh was not included for fishes ($p=0.029$) for fishes but not for crustaceans ($p=0.166$).

The average total CPUE nekton density (Fig. 4B) in the narrow-marsh sites, was 1.05 nekton m^{-2} of marsh, as compared to only 0.37 nekton m^{-2} in the wide-marsh sites. Nonlinear regressions of abundance (CPUE) per unit area of marsh, revealed that nekton ($R^2=0.29$)(Fig. 4B), fish ($R^2=0.50$)(Fig. 5B) and crustacean ($R^2=0.07$)(Fig. 6B) densities declined with increasing marsh width.

Regression analyses of total nekton abundance done separately by season revealed significant ($p=0.0005$ including the natural marsh sites) and marginally significant ($p=0.065$ excluding natural marsh sites) increases with marsh width during spring (Fig. 7A) but not during fall ($p=0.82$ and $p=0.21$)(Fig. 7B).

By comparing nekton catches between years (2009 vs. 2010) for the PKS data set, the only geographic region (central) with multiple-year sampling, we assess interannual variation in relationships of nekton abundance with marsh width. For fall and spring separately, we computed linear regression plots of total nekton against marsh width for PKS, both including and excluding natural marsh treatments, and comparing the outcomes between 2009 and 2010. Fyke net sampling of nekton exhibited generally non-significant patterns of increases in nekton CPUE abundance with marsh width that were similar in both years but with higher slope for spring (Fig. 8A) than in fall samples (Fig. 8B).

Nekton abundances from research gill nets sampling their flux from high tide to 2 hours later in the waters below the marsh edge failed to exhibit any statistically significant or even marginally significant relationship to width of the marsh shoreward of the sampling area, using analytical designs and methods identical to those used for fyke net catches. Regressions from gill net data of total nekton CPUE abundance against marsh width, with and without natural marsh ($p=0.704$, $R^2= 0.009$ with natural marshes included, $p=0.39$, $R^2= 0.06$ without natural marshes)(Fig. 9), on fish alone and crustaceans alone (data not shown), and on total nekton separated by season (data not shown) also failed to exhibit any suggestive pattern or marginal statistical significance. Gill net CPUE of total nekton in PKS samplings of 2 seasons in 2 successive years exhibited a similar absence of response to width of the landward marsh in both years and the response graphs were similarly uninteresting between years for both fall (Fig. 10A) and spring (Fig. 10B).

Based on sampling separately but with identical methods in both night and day at each study site throughout this entire study, formal statistical comparisons of catches between night and day can provide powerful tests of potential sampling biases that may emerge from choosing

only one temporal state. Paired t-tests (one for fyke and one for gill nets) analyzing differences between night and day in total CPUE nekton abundances (Table 1) reveal that sampling marsh nekton at night by fyke nets produced a statistically significant ($p < 0.048$) 75% more individuals than the identical procedure conducted during day and that sampling nekton below the marsh with research gill nets produced a statistically significant ($p < 0.0006$) 97% more individuals at night.

The one-way ANOVA performed to explore differences in regional CPUE abundances of nekton, as indicated by catches in fyke and gill nets, revealed a significant effect ($p < 0.021$) of geography for the fyke net samples of marsh nekton but not for the gill net catches (Fig. 11). Holm-Sidak pairwise contrasts demonstrated that the nekton catch by fyke net was lower in the central region than in the northern or southern regions, which did not differ.

We were able to determine the trophic level of 68 of our 80 species of nekton. Those that we were unable to identify due to our lack of confidence in species identification of juvenile specimens. Other species were not assigned a trophic level due to indeterminate diet and lack of substantial research. Trophic level analyses revealed differences in trophic levels between fyke and gill nets with a higher average trophic level (3.11) in gill nets than in fyke nets (2.87).

Appendix 1 presents CPUE abundance data by net type on nekton catches (broken down by fish vs. crustaceans) separately for each of the five different combinations (with only medium-width marsh replicated in each geographic area) of shoreline treatment (natural vs. bulkheaded) and marsh width (none, narrow, medium, and wide).

Salinity values as reported by O'Meara (2014) from seasonal sampling (all four seasons) at these sites in 2010 show an average salinity in the KH sites of 4.60, in PKS an average of 32.5,

and in the WILM sites an average of 30.6. The lower salinity ranges observed in the KH sites are due to the distance from the nearest tidal inlet located approximately 20 miles south.

Discussion

Our investigation into the future status of the salt marsh ecosystem service of supporting growth and production of nekton relies on taking a space-for-time substitution approach to portray how this particular ecosystem service is expected to change on shorelines fortified by massive engineered structures during a future of persistent climate change and sea-level rise. We do not have a basis on which to predict when the nekton support service will decline by any given percentage, but we can project the consequences of the temporal progression in marsh thinning as it is driven by the habitat squeeze between the fixed bulkhead barrier to transgression and the rising waters and eroding waves at the estuarine edge. Using four states of marsh width, we project nekton utilization, and presumably production to vary over time from conditions depicted in the wide marsh treatment to those of the medium-width marsh some time into the future, then later to the thin marsh conditions, and finally to the no-marsh conditions. From our regression analyses, we project that nekton use will decline linearly with reductions in marsh width. The rate at which nekton declines with marsh width is such that moving in time from one stage to the next, the density of nekton using the remaining marsh will increase. For the thinnest marsh, nekton abundance is the lowest among all other marsh width treatments but density of nekton per unit area of remaining marsh is the highest. This process is presumably a consequence of the habitat selection documented by Minello et al. (2012) and Rozas et al. (2015) in which nekton more often select marsh close to the edge of the estuary or to marsh channels as they move into the marsh with the rising tide. This behavior produces a declining density gradient with distance from the edges, reflecting growing risks of more travel time, stranding by falling tides, varying food supplies, and other fitness-influencing processes. By squeezing the

marsh over time, only the most heavily utilized distance stratum is left and nekton density is highest. Interestingly, when marsh disappears entirely, based on our sampling results, the nekton density does not fall to zero; however, if an entire estuarine shoreline were to be blocked by a bulkhead under conditions of high enough water levels, then one wonders where the food will come from to fuel growth of these out-of-marsh nekton: surely their density would be expected to decline system-wide.

This application of our spatial data to project temporal sequences of change with rising sea levels is based upon patterns in total nekton abundance and density. Neither of the two component taxonomic groups (fish and crustacea) of nekton alone exhibited the statistically significant decline in abundance with declining marsh width, although this response was shown strongly, significantly, and repeatedly by the sum total nekton. This difference could in part be caused by greater statistical power to detect patterns in the pooled, much larger database. Alternatively, the absence of pattern in either of the component groups despite compelling patterns of total abundance with declining marsh width in the pooled nekton suggests a strong possibility of spatial segregation and complementarity between crustaceans and fish. Given that these groups commonly interact when together – predatory fish, such as juvenile pinfish, preying upon crustaceans such as grass shrimp, for example – behavioral development of complementary habitat selection within the marsh should not be a surprise. Indeed pinfish represent the fish of the greatest biomass in our marsh fyke net data set, while grass shrimp is the crustacean of greatest abundance and biomass, so this example is not hypothetical. These two species do indeed exhibit an inverse relationship between their abundances in our fyke net data.

Our initial expectation for geographic patterns in nekton abundance fluxing out of the coastal marsh during ebbing tides relied on a simple biophysical construct. We hypothesized

that because these mobile high-tide residents can benefit energetically from riding the falling tidal pulse out of the marsh during ebbing tides that the geographic region with the highest tidal flux would support the largest numbers of these high-water occupants of the coastal marsh. The southern region with a tidal range of 2 m did indeed exhibit significantly higher nekton abundances than the central region of 1 m tides. We also hypothesized that the more regular, and thus more predictably reliable, astronomical tidal forcing would be exploited more commonly by nekton than the less predictable and irregular meteorologically forced water movements on and off the marsh of the northern region. Surprisingly, total nekton abundance in the northern region was not only significantly higher than that of the central region, but was not detectably lower than the southern region nekton densities. In this northern region of Kitty Hawk marshes, the vegetation was notably different from that of the central and southern regions in the form of massive biomass of the invasive European milfoil plant (*Myriophyllum spicatum* L.) clearly providing shading, certainly oxygenating the waters, and likely augmenting diets of nektonic herbivores. A plant reviled by conservationists may need to be celebrated by fisheries managers. Nevertheless, the role of the milfoil needs to be more deeply explored. Clearly, the environment and the fish community in the northern marshes were very different from the other two sites. O'Meara et al. (2014) reported annual mean salinities from water sampling in the marshes at all northern sites of 4.6, at all central sites of 32.5, and at all southern sites of 30.6 ppm. The taxonomic composition of fishes in the northern region clearly reflects a largely freshwater assemblage, quite distinct from the marine and estuarine fishes of the other two regions. The low salinity in the Kitty Hawk marsh sites is a reflection of limited oceanic water influx.

Most previous nekton sampling results are complicated by the possibility of dramatic diurnal differences (e.g. Summerson and Peterson 1984). Sampling at night carries intrinsic risks

to those field researchers wading along shorelines in the dark when marsh use by nekton appears from our data to be about double daytime use, yet if sampling were restricted to daylight hours, results would grossly underestimate the actual species richness and abundance (Stoner 1991, Rountree and Able 1993, Guest et al. 2003). Failure to recognize this difference between night and day sampling would bias quantitative estimates of nekton use. Our sampling results comparing abundances in day and night samples confirmed dramatic diel differences in net catches of both types, fyke and gill, showing that night-time sampling yielded nearly twice the numbers of nekton (75-97% respectively) than identically done day-time sampling (Table 1). This disparity could be driven by behavioral differences of nekton between day and night in foraging, reproduction, or predator avoidance. Alternatively, abundant nektonic species may be expressing gear avoidance behaviors that are more effective in daylight. Many penaeid shrimp species are often absent from collections made during daytime samples as they remain buried during daylight hours (Stoner 1991, Guest et al. 2003). We found a similar trend in our fyke net samples, with lower numbers of penaeid shrimp captured during the day (n=548) than during the night samples (n=1373). Although we provide no new test of whether greater night-time abundances of nekton represent a consequence of greater ability to avoid sampling gear in the daylight or an indication of predator avoidance behavior, our results when combined with those of Stoner (1991), Rountree and Able (1993), and Guest et al. (2003), provide a compelling indication that nekton catches differ between day and night in a consistent pattern of higher nocturnal catches. Each of these independent studies quantifying nekton use in structured nearshore habitats, coastal marsh or seagrass, confirmed higher catches at night and the percent increase from day to night ranged from 188% to 335% more individuals. This may be a small

enough difference to use the mean as a standard correction factor to convert analogous data only collected in day or only collected in night.

Our examinations of gut contents from our sampled fishes, the scientific literature, and FishBase helped characterize the diets, and therefore trophic levels, on which the nekton of the marsh, caught by fyke nets, and of the more continuously flooded waters below the marsh, sampled by gill nets, feed reveals an important difference. Those nektonic species feeding in the marsh itself and sampled by fyke net tended to occupy the secondary consumer trophic level, feeding on benthic invertebrates in and on the sediments. In contrast, the trophic level of the nekton caught and sampled by the gill nets in waters beneath the marsh tended to be characterized by fish feeding on higher trophic levels, including tertiary consumers. These fish actually consume many of the nekton, especially the shrimps and other crustaceans, which comprise the high-tide marsh resident nekton sampled by our fyke nets. One of the most compelling demonstrations of this trophic relationship is the bonnet-head shark that we found in the waters below the marsh awaiting tidal transport of blue crabs and other crustaceans to feed upon. We did not expect the nekton below the marsh to exhibit any relationship with the salt marsh structure and analyses of our nekton CPUE abundances confirmed that expectation. These higher-order predators are more mobile and move along the marsh shoreline rather than moving up into the marsh for prey, which is probably why there was no relationship between marsh width and CPUE abundance for the nekton caught by gill nets. Conserving coastal marsh habitat and its valuable ecosystem services in the face of rising sea levels and the associated direct and indirect threats to coastal marsh habitat represents a critical challenge to policy makers, managers, and the private property owners (e.g., Titus and Richman 2001, Peterson et al. 2008, Tornquist and Meffert 2008, Rahmsdorf 2010, Scyphers et al. 2011). Both coastal regulators and

estuarine waterfront property owners appear to consider engineered hard shoreline stabilization structures, especially bulkheads, to offer the most reliable protection against erosion and damage during storms, motivating high demand and rapid approval of permits for such structures (Scyphers et al. 2011). Nevertheless, evidence exists (Thieler and Young 1991) to show that bulkheads are vulnerable to damage during major storms and bulkheads in North Carolina sounds were shown to fail during Hurricane Irene more readily than other less invasive options to inhibit shoreline erosion (Gittman et al. 2014).

Even if bulkheads can be restored after storm damage, their consequent persistence in a fixed position at the top edge of the regularly flooded marsh, commonly the upper margin of *Spartina alterniflora* marsh in estuaries and sounds on the east coast of the U.S. with regular astronomical tidal forcing, prevents transgression of the coastal marsh up-slope as sea levels continue to rise. Together with increasing flooding associated with rising sea levels and enhanced erosion along the lower margin of the marsh from increasingly frequent intense storms (Peterson et al. 2008, Titus and Craghan 2009), the coastal marsh below bulkheads becomes increasingly squeezed and progressively narrower until it disappears (Peterson et al. 2008, Titus and Craghan. 2009). Ironically, one of the valuable ecosystem services generated by the presence of the structural barrier created by dense, tall coastal marsh plants across the width of the marsh platform is buffering and dissipation of wave energy and thus inhibition of flooding and wave damage higher on shore (Luettich et al. 2014). Gittman et al. (2014) documented no damage to wide natural marshes during the same Hurricane Irene that damaged so many bulkheads.

In addition to concern over the desired performance of engineered versus natural shoreline protection structures during storms and floods, recent concerns have been widely raised

about their potential impacts on the ecosystem services associated with coastal marshes (National Research Council 2007). The provision of habitat serving as nursery for fish and crustaceans is an especially important ecosystem service because of the central role played by coastal wetlands in estuaries in supporting a large fraction of the economically valued coastal production of fish, crabs, and other exploited species. Furthermore, coastal marshes support dense populations of wading birds dependent on nektonic prey so wildlife managers share concerns over sustaining wildlife dependent on the nektonic production in coastal marshes. In perhaps the first study of implications of inserting large engineered structures at the top of the regularly flooded coastal marsh, Seitz et al. (2006) sampled benthic invertebrates, which commonly serve as prey of crabs and demersal fish, examining occupation of the sedimentary estuary floor in the shallow subtidal zone below natural marsh, bulkheads and rip-rap revetments on a lower Chesapeake Bay tributary. Sampling revealed lower density, biomass, and diversity below bulkheads than below natural *Spartina alterniflora* marsh, with revetments falling in between. Bilkovic and Roggero (2008) are the first to systematically sample the nekton along estuarine shores with different levels of development and differing in type of shoreline protection. They documented high diversity of fishes, inclusive of the tidal marsh species, on natural marsh and rip-rap revetment shores with low development as opposed to low diversity of a few generalists on highly developed or bulkheaded shores.

An alternative shoreline protection option in front of coastal marshes is a marsh sill, a low-profile rock or oyster shell breakwater installed just slightly deeper than the marsh edge. Sills are designed to induce breaking of waves, thereby dissipating energy and inducing sediment deposition in the marsh behind. Early tests of sill impacts on nekton in the marshes behind them revealed higher abundance and diversity than in bulkheaded marshes and even in natural marshes

(Gittman et al. 2015a), suggesting that this form of a “living shoreline” may hold promise as an alternative to more invasive bulkheads and revetments. The capacity of the marsh sill to induce greater sedimentation in the landward marshes behind them makes them an especially attractive shoreline stabilization structure for wide consideration in North Carolina and other locations, where studies revealed that elevation of the natural marsh floor is not increasing fast enough even to match the current local rate of sea-level rise (Voss et al. 2013). By employing “marsh organs” to manipulate plant inundation, Voss et al. (2013) demonstrated that the fate of *Spartina alterniflora* and *Juncus roemerianus* is drowning when inundation exceeds a certain tolerance threshold. As sea level continues to rise with ongoing climate change, inevitable marsh habitat loss may be prevented by installation of marsh sills.

We used marshes of differing widths a proxy for time to infer the effects of rising sea level in eroding salt marsh habitat in front of engineered shoreline protection devices that prevent transgression of the marsh. The range of salt marsh widths sampled- defines the scope our space- for- time assessment, but we have no viable means of converting marsh width to proxy date-of bulkhead construction. Hence, our time scale is relative, yielding no estimate of degree of decline in nekton use with absolute time. This gap in our ability to predict absolute rate of loss in nekton use could be plugged by long-term monitoring of newly installed bulkheads and rock revetments. Even in the absence of the ability to scale the declining marshes to absolute time, the documented loss of salt marsh habitat and observed declines in nekton use has important implication for commercially and recreationally important fish that use the marsh habitat directly or prey upon nektonic forage species that benefit from feeding and sheltering in the coastal marsh habitat. More than 95% of commercial and recreational seafood species in North Carolina are to be wetland-dependent at some point during their life cycle (Deaton et al.

2010). Consequently, developing shoreline protections options that do not lead to marsh habitat loss represents an important challenge, to which marsh sills and oyster reef breakwaters may prove to be viable responses.

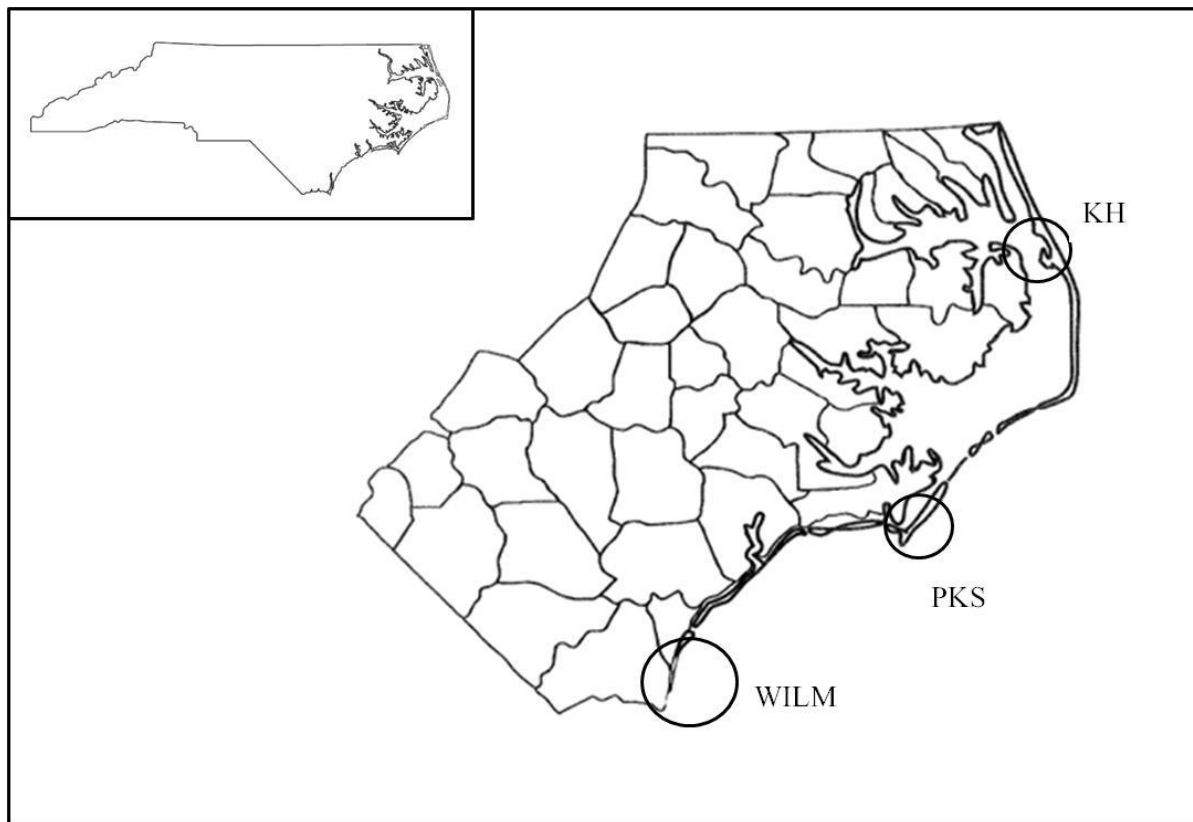


Figure 1. Locations of three study regions along coastal North Carolina. Six study sites were nested within each region for a total of 18 sampling locations. KH = Kitty Hawk; PKS = Pine Knoll Shores; WILM = Wilmington and Oak Island.

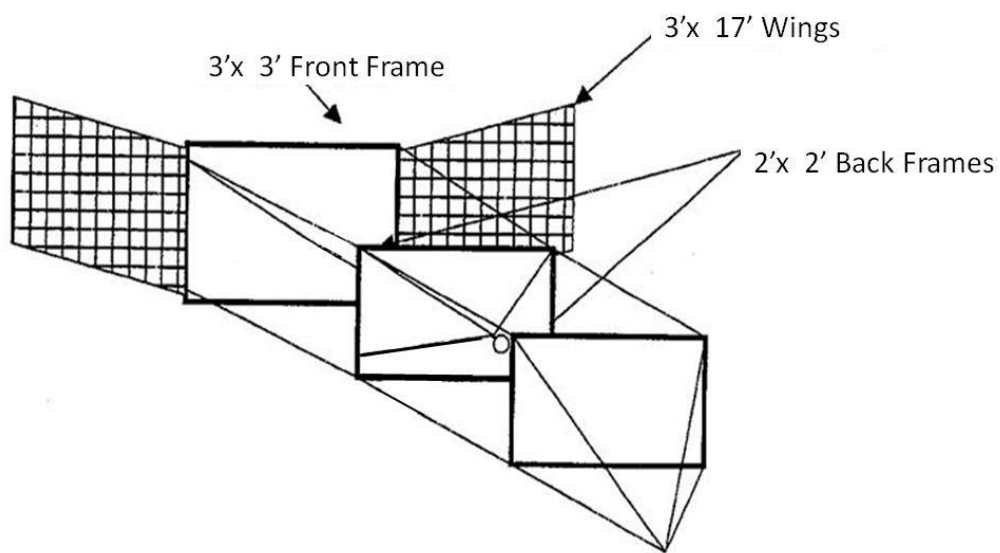


Figure 2. Diagram of the fyke net showing measurements. The wings were positioned facing the marsh and placed during high tide. Figure courtesy of Christiansen Nets.

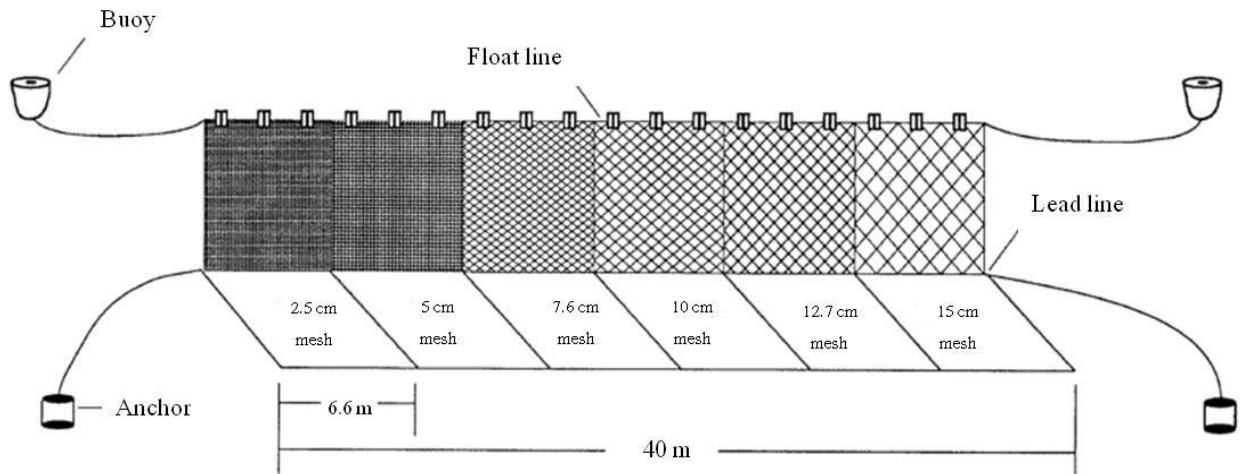


Figure 3. Diagram of 40 m custom research gill net with six, 6.6 m monofilament panels ranged in stretched mesh size from 2.5 cm to 15 cm.

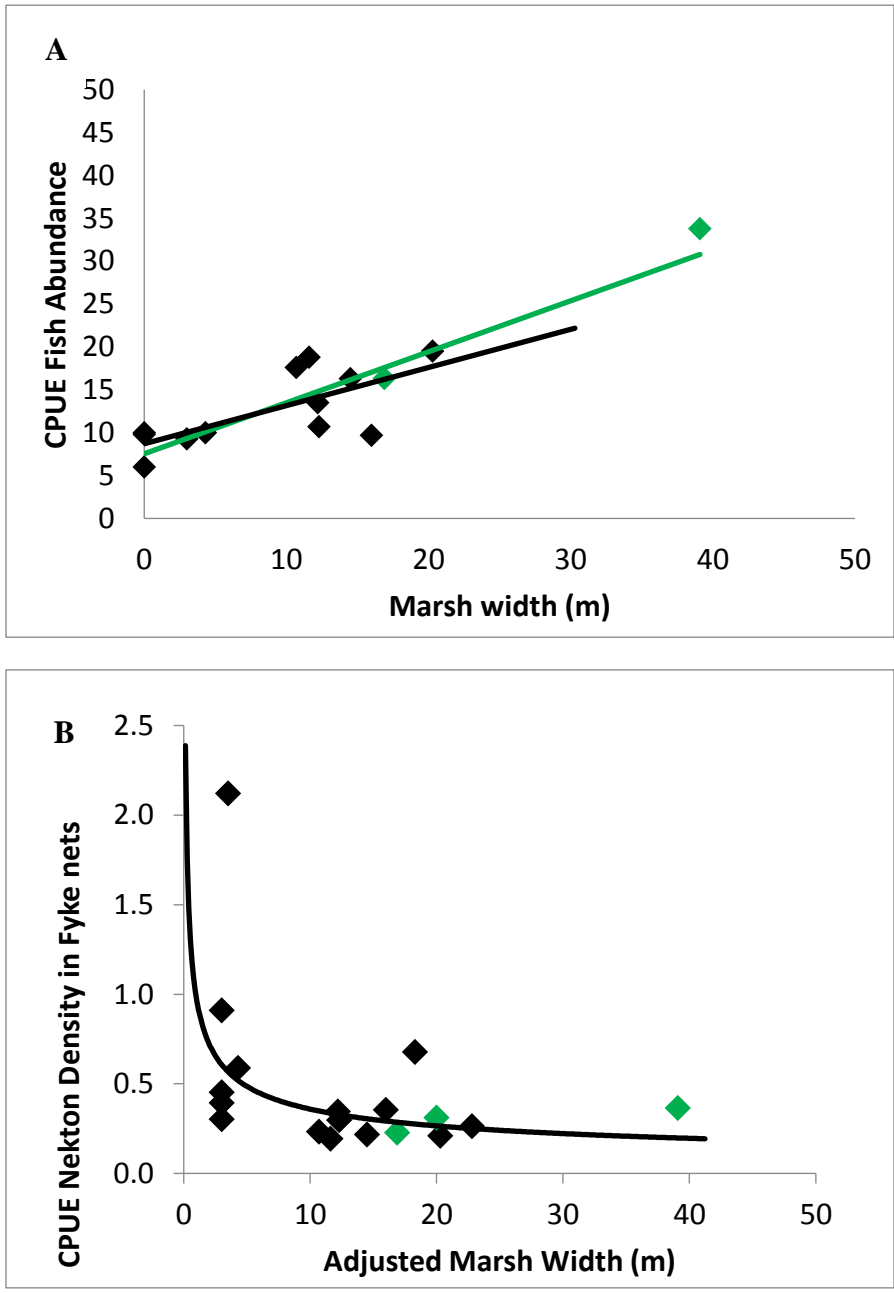


Figure 4. Catch per unit effort of total nekton (A) and density (B) in fyke net samples as a function of marsh width (m) within each geographic region (18 total) with seasons, and day/night samples pooled. Green diamonds indicate the 3 natural marshes from each region which were included (green regression line) and excluded (black regression line) from statistical regression analysis. (A) CPUE of nekton with natural marshes included ($p < 0.0007$, $R^2 = 0.74$) and excluded ($p < 0.034$, $R^2 = 0.72$). (B) Density CPUE of nekton/ m^2 per unit marsh area with natural marshes included ($R^2 = 0.2911$).

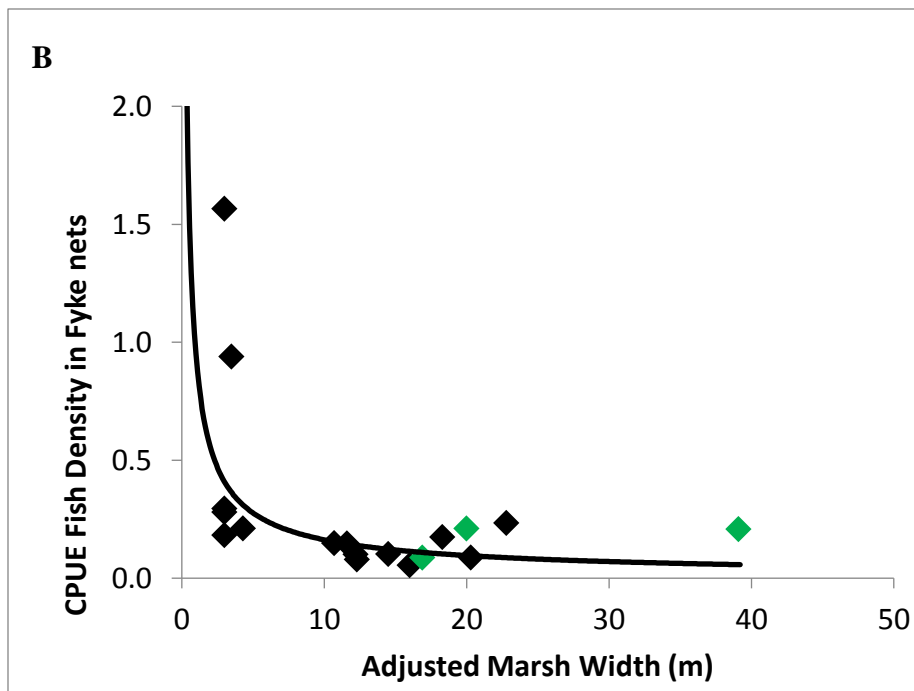
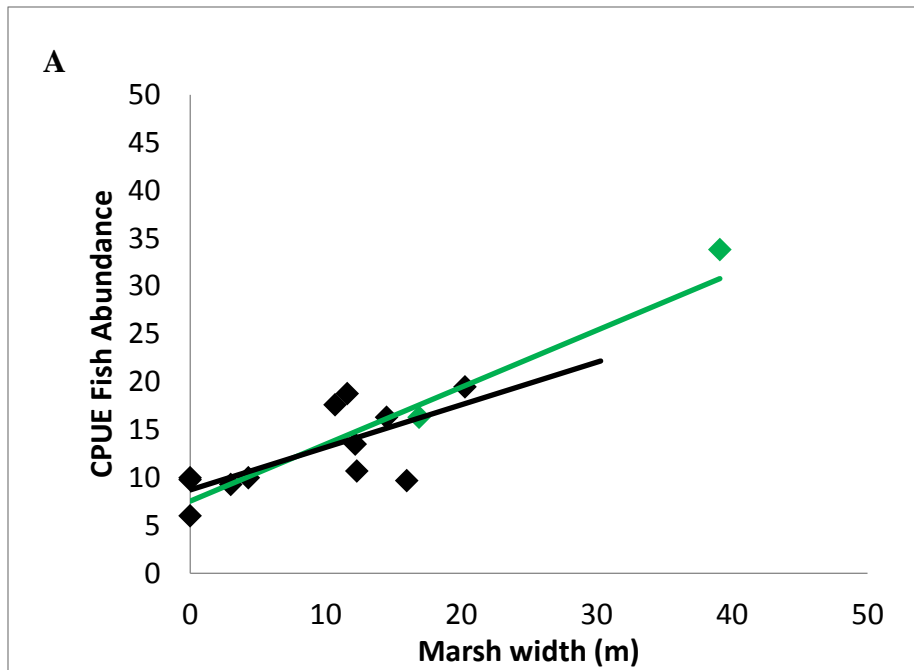


Figure 5. Catch per unit effort (A) and density (B) of fish only in fyke net samples. Green diamonds indicate the 3 natural marshes from each region which were included (green regression line) and excluded (black regression line) from statistical regression analysis. (A) CPUE of fish only with natural marshes included ($p < 0.012$, $R^2 = 0.50$) and excluded ($p < 0.029$, $R^2 = 0.79$). (B) Density CPUE of fish/m² per unit marsh area with natural marshes included ($R^2 = 0.467$).

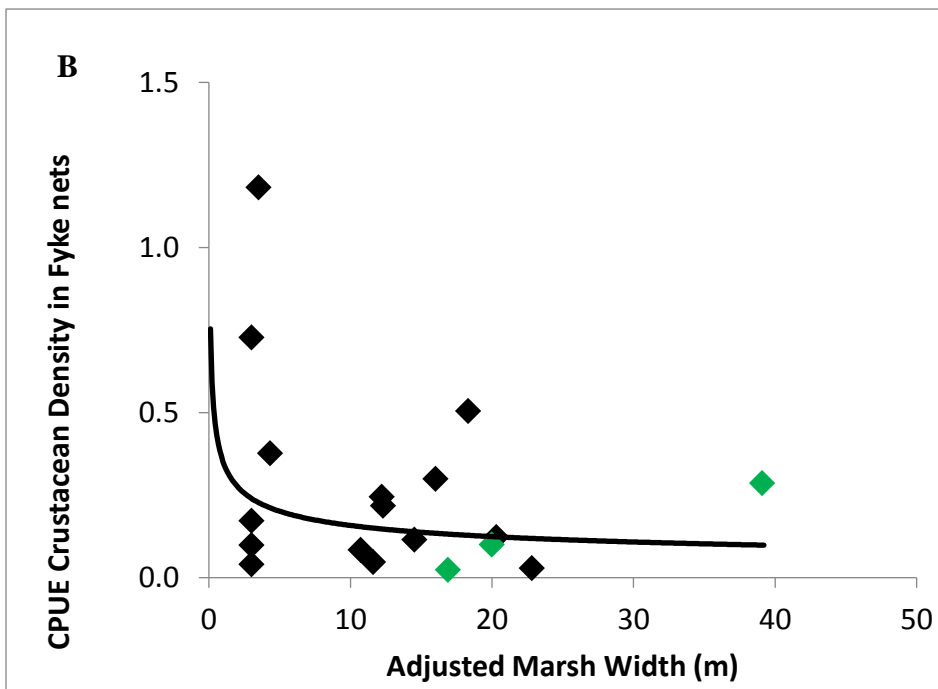
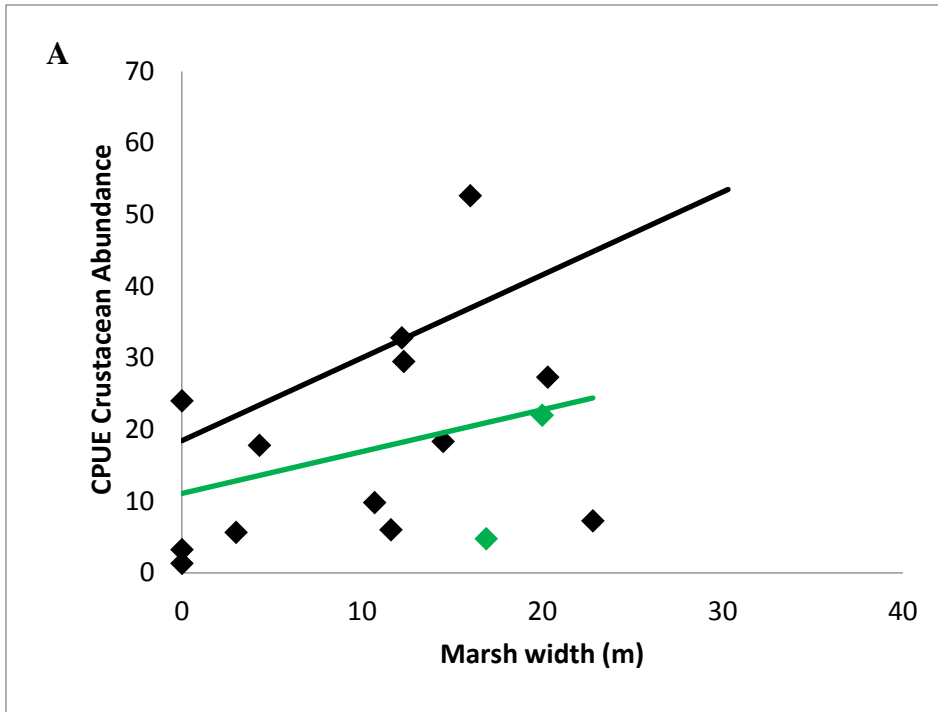


Figure 6. Catch per unit effort (A) and density (B) of crustaceans only in fyke net samples. Green diamonds indicate the 3 natural marshes from each region which were included (green regression line) and excluded (black regression line) from statistical regression analysis. (A) CPUE of crustaceans only with natural marshes included ($p < 0.006$, $R^2 = 0.08$) and excluded ($p < 0.17$, $R^2 = 0.103$). (B) Density CPUE of fish/m² per unit marsh area with natural marshes included ($R^2 = 0.07$).

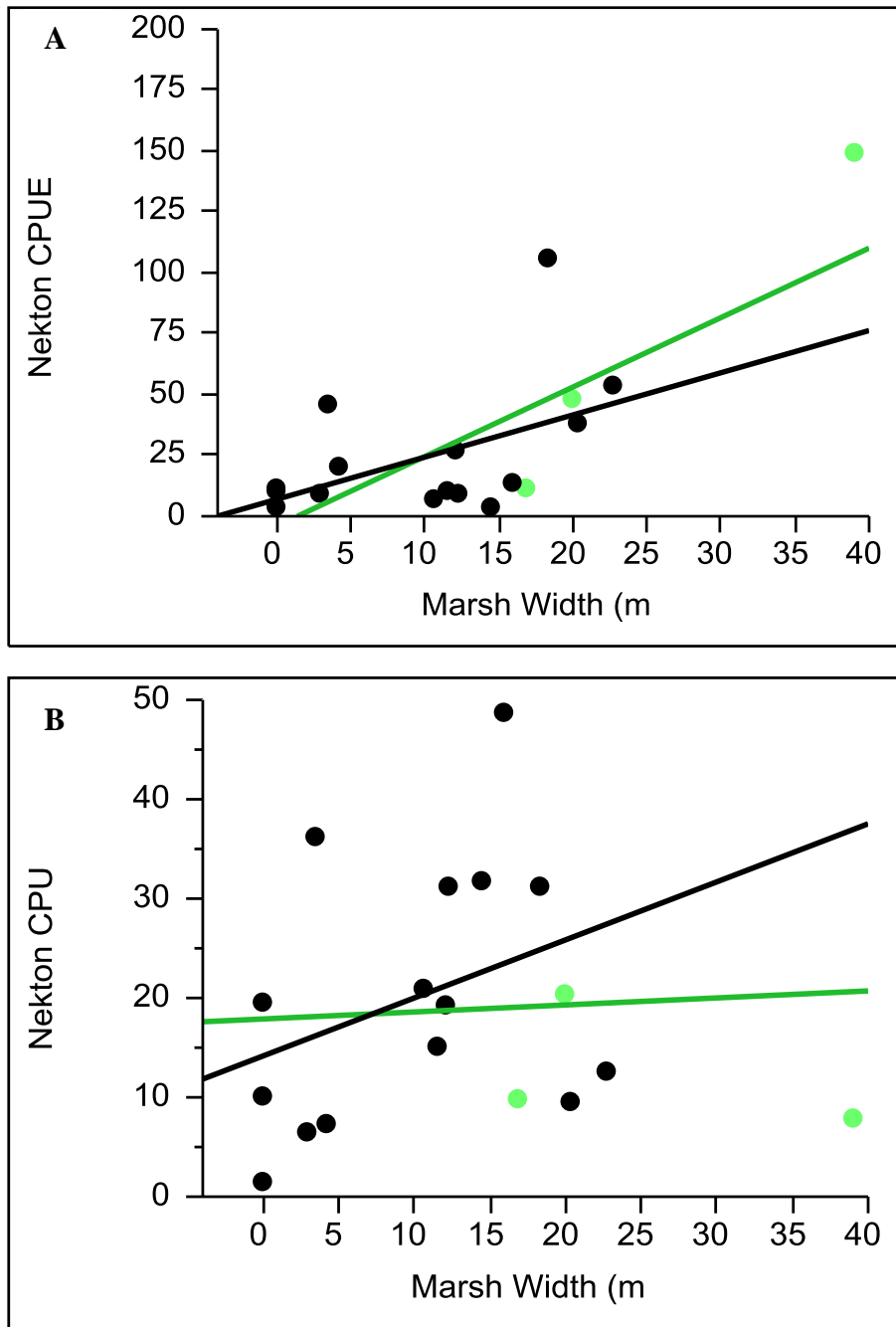


Figure 7. Catch per unit effort of nekton in spring (A) and fall (B) in fyke net samples. Green circles indicate the 3 natural marshes from each region which were included (green regression line) and excluded (black regression line) from statistical regression analysis. (A) CPUE of nekton in spring samples with natural marshes included ($p < 0.0005$, $R^2 = 0.54$) and excluded ($p < 0.06$, $R^2 = 0.24$). (B) CPUE of nekton in fall samples with natural marshes included ($p < 0.82$, $R^2 = 0.003$) and excluded ($p < 0.21$, $R^2 = 0.12$).

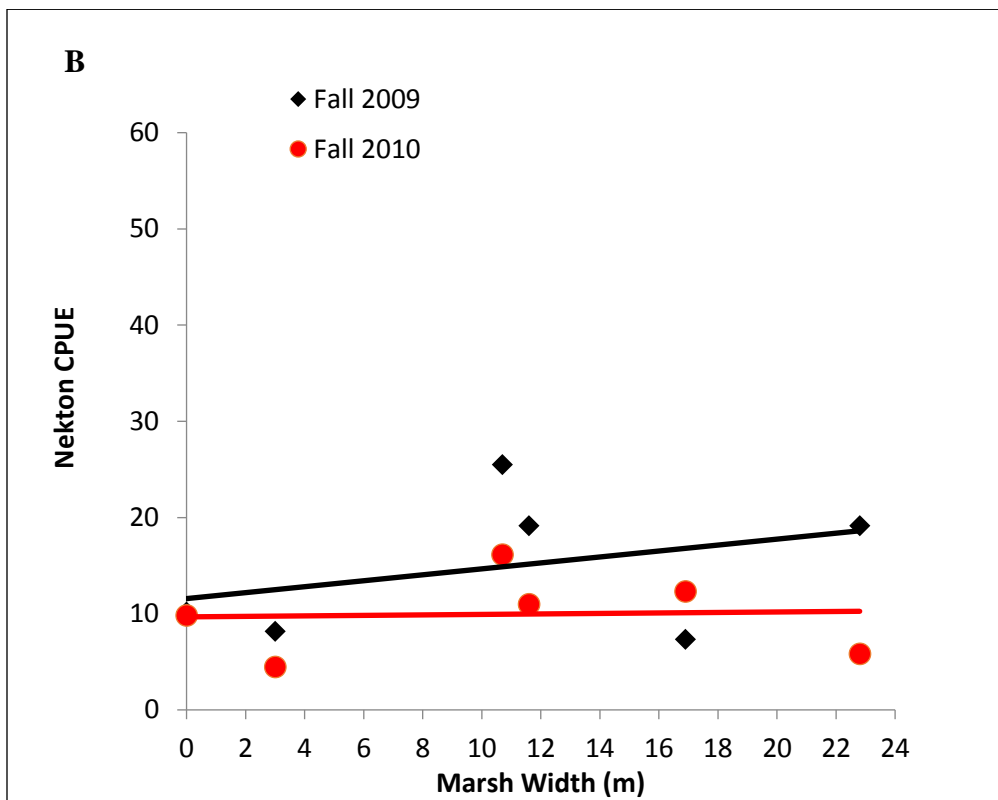
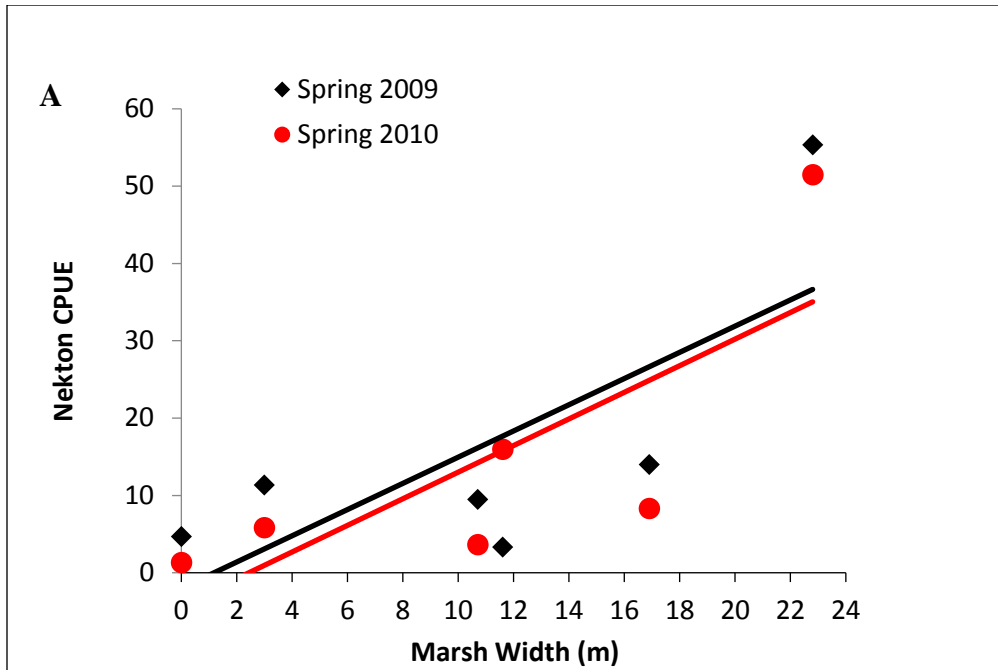


Figure 8. Catch per unit effort of nekton in PKS (A) spring 2009/2010 and (B) fall 2009/2010 in fyke net samples. CPUE of nekton in spring samples with natural marshes included in 2009 ($p < 0.1$, $R^2 = 0.54$) and 2010 ($p < 0.07$, $R^2 = 0.6$). CPUE of nekton in fall samples with natural marshes included in 2009 ($p < 0.49$, $R^2 = 0.13$) and 2010 ($p < 0.93$, $R^2 = 0.002$).

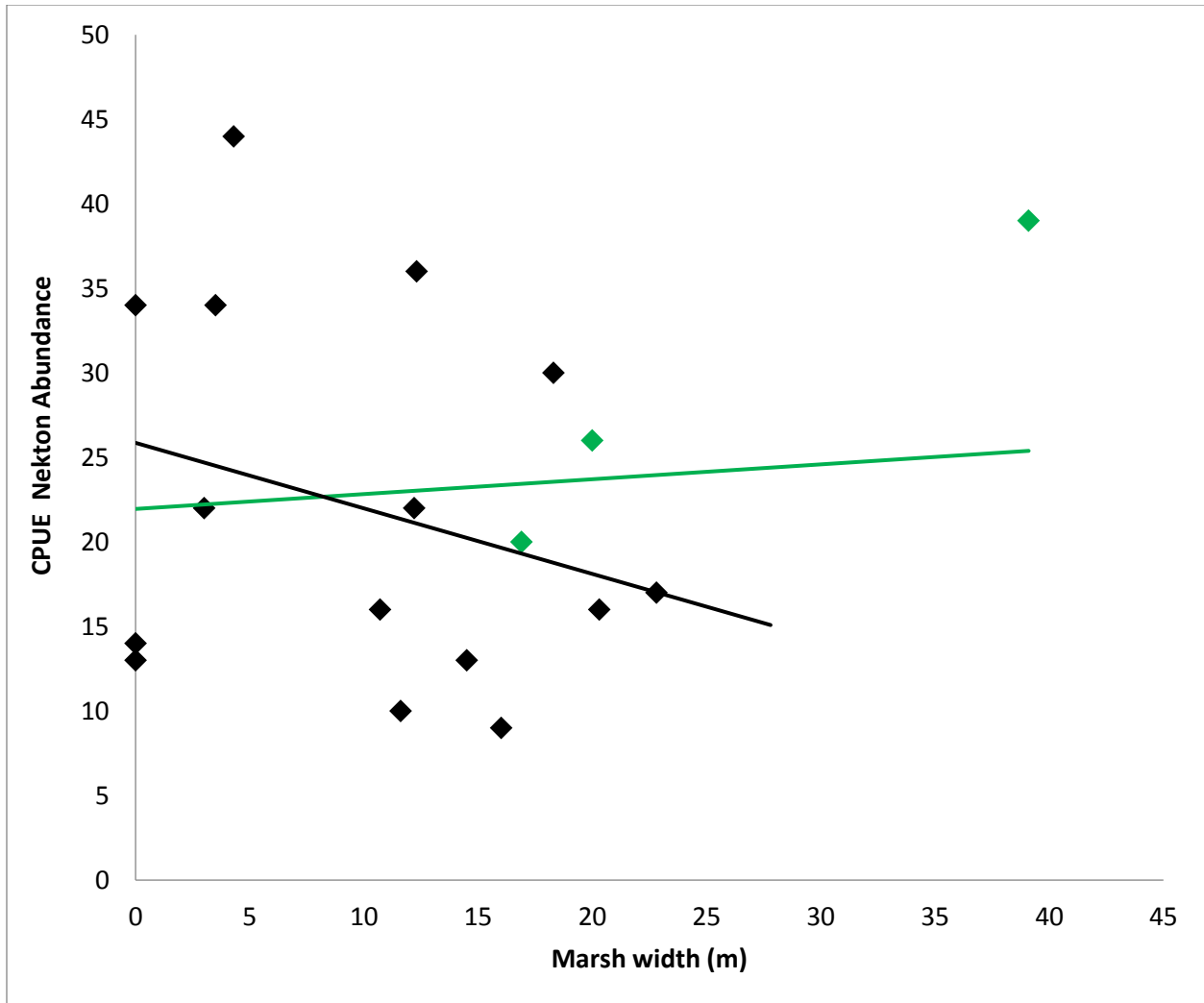


Figure 9. Catch per unit effort of total nekton in gill net samples as a function of marsh width (m) within each geographic region (18 total) with seasons, and day/night samples pooled. Green diamonds indicate the 3 natural marshes from each region which were included (green regression line) ($p < 0.704$, $R^2 = 0.009$) and excluded (black regression line) ($p < 0.39$, $R^2 = 0.06$) from statistical regression analysis.

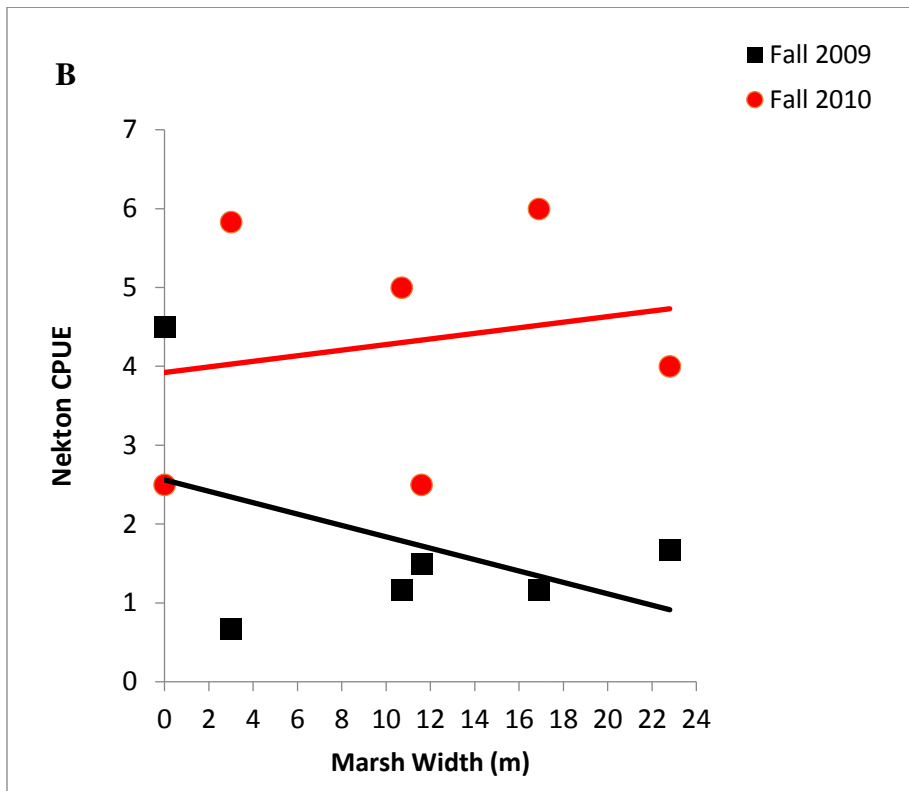
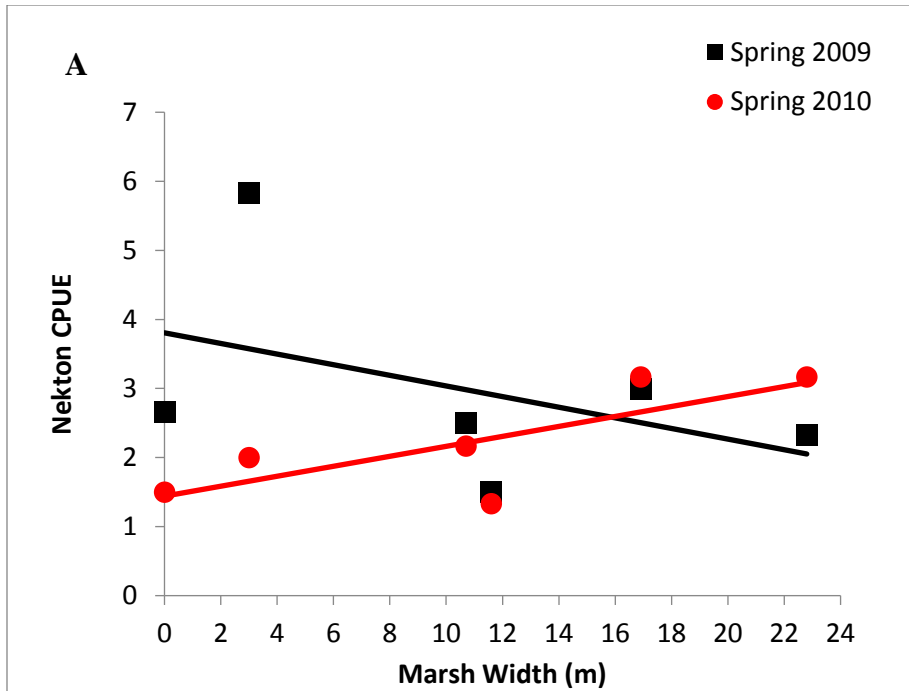


Figure 10. Catch per unit effort of nekton in PKS (A) spring 2009/2010 and (B) fall 2009/2010 in gill net samples.

CPUE of nekton in spring samples with natural marshes included in 2009 ($p < 0.38$, $R^2 = 0.19$) and 2010 ($p < 0.07$, $R^2 = 0.59$). CPUE of nekton in fall samples with natural marshes included in 2009 ($p < 0.38$, $R^2 = 0.2$) and 2010 ($p < 0.71$, $R^2 = 0.037$).

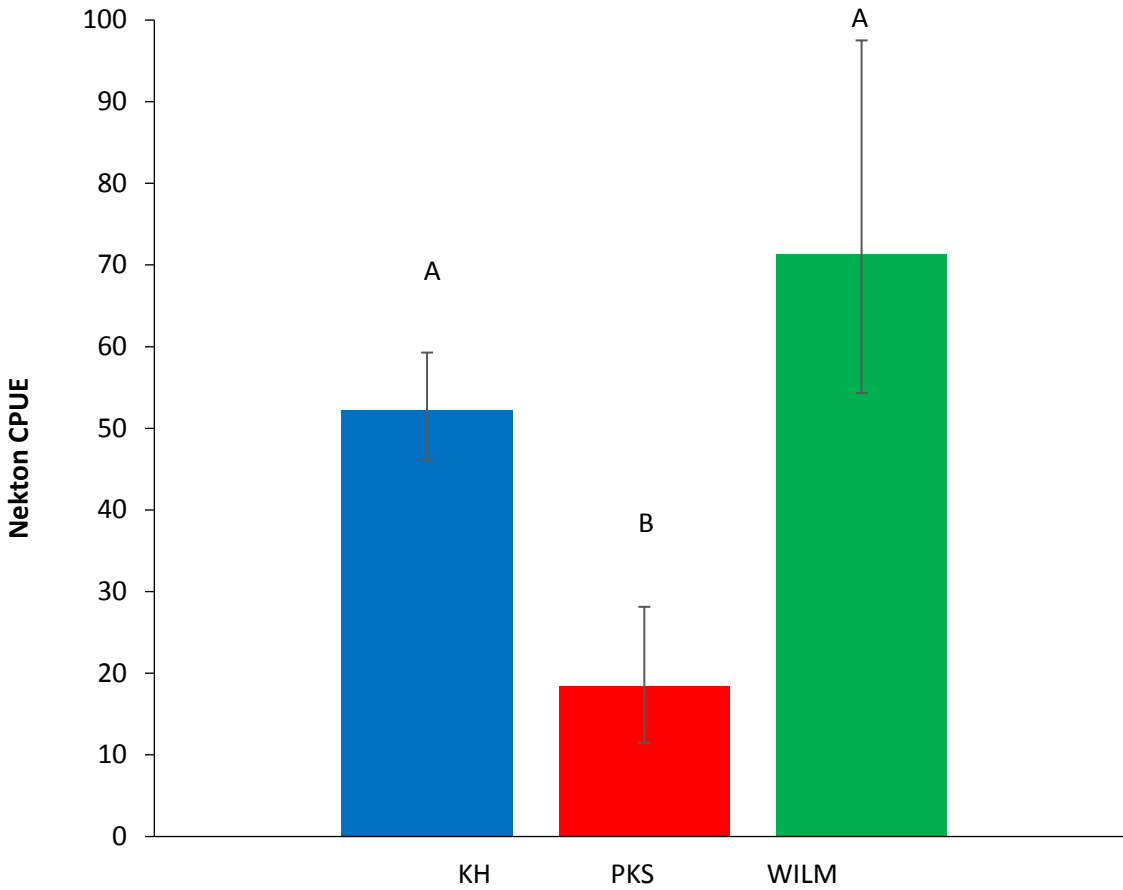


Figure 11. Regional variation in nekton abundance in fyke nets. A one-way ANOVA revealed a significant difference ($p < 0.021$) in CPUE by geographic location. Holm-Sidak pairwise contrasts indicate significance between KH and PKS ($p < 0.017$) and WILM and PKS ($p < 0.025$). Data were \ln transformed for normality and error bars are back transformed data of the SE.

Table 1. Paired t-test on day/night CPUE nekton abundances by net types

| Net | N | Mean | St. Dev | St. Error | df | t | <i>p</i> |
|------|----|------|---------|-----------|----|-------|----------|
| Fyke | 48 | 7.05 | 24.11 | 3.5 | 47 | 2.025 | 0.048 |
| Gill | 48 | 3.51 | 6.62 | 0.96 | 47 | 3.669 | 0.0006 |

Table 2. Catch per unit effort of nekton by net type and time of day.

| | | Day | | Night | |
|----------------------------------|-----------------------|-------|-------|--------|-------|
| | | fyke | gill | fyke | gill |
| Crustacea | | | | | |
| <i>Tozeuma carolinense</i> | arrow shrimp | 0.00 | 0.00 | 1.33 | 0.00 |
| <i>Callinectes sapidus</i> | blue crab | 58.00 | 59.50 | 89.33 | 83.00 |
| <i>Hippolyte species</i> | grass shrimp | 90.83 | 1.50 | 223.17 | 17.00 |
| <i>Rhithropanopeus harrisi</i> | mud crab | 0.50 | 0.00 | 0.17 | 0.00 |
| <i>Alpheidae spp.</i> | pistol shrimp | 0.33 | 0.00 | 0.17 | 0.00 |
| <i>Menippe mercenaria</i> | stone crab | 0.00 | 0.00 | 0.17 | 2.00 |
| <i>Penaeus spp.</i> | shrimp | 28.67 | 2.00 | 76.50 | 6.00 |
| <i>Libinia spp.</i> | spider crab | 0.33 | 0.00 | 0.00 | 0.00 |
| Fish | | | | | |
| <i>Anguilla rostrata</i> | American eel | 0.17 | 0.00 | 0.17 | 0.00 |
| <i>Anchoa spp.</i> | anchovy | 0.17 | 0.50 | 0.00 | 0.00 |
| <i>Strongylura marina</i> | Atlantic needlefish | 0.17 | 2.00 | 0.50 | 0.50 |
| <i>Menidia menidia</i> | Atlantic silverside | 0.17 | 0.00 | 15.83 | 2.00 |
| <i>Dasyatis sabina</i> | Atlantic stingray | 0.17 | 0.50 | 0.00 | 1.00 |
| <i>Fundulus diaphanus</i> | banded killifish | 2.67 | 0.00 | 1.17 | 0.00 |
| <i>Sphyrna spp.</i> | barracuda | 0.17 | 0.00 | 0.00 | 0.00 |
| <i>Anchoa mitchilli</i> | bay anchovy | 18.00 | 0.00 | 6.33 | 0.00 |
| <i>Citharichthys spilopterus</i> | bay whiff | 0.00 | 0.00 | 0.17 | 0.00 |
| <i>Anchoa lamprotaenia</i> | bigeye anchovy | 0.00 | 0.50 | 0.00 | 0.00 |
| <i>Prionotus tribulus</i> | bighead searobin | 0.00 | 0.00 | 0.00 | 0.50 |
| <i>Pogonias cromis</i> | black drum | 0.00 | 0.00 | 0.00 | 1.50 |
| <i>Symphurus plagiusa</i> | blackcheek tonguefish | 2.67 | 0.00 | 1.33 | 5.50 |
| <i>Pomatomus saltatrix</i> | bluefish | 0.00 | 1.50 | 0.00 | 4.00 |
| <i>Dasyatis sayi</i> | bluntnose ray | 0.00 | 2.00 | 0.00 | 0.50 |
| <i>Amia calva</i> | bowfin | 0.00 | 0.00 | 0.00 | 0.50 |
| <i>Ameiurus spp.</i> | bullhead catfish | 0.50 | 0.50 | 0.83 | 0.50 |
| <i>Rachycentron canadum</i> | cobia | 0.00 | 0.00 | 0.00 | 1.00 |
| <i>Cyprinus carpio</i> | common carp | 0.00 | 0.00 | 0.17 | 3.50 |
| <i>Micropogonias undulatus</i> | croaker | 0.17 | 0.50 | 0.50 | 3.50 |
| <i>Paralichthys spp.</i> | flounder | 0.50 | 1.00 | 0.33 | 0.50 |
| <i>Paralichthys oblongus</i> | fourspot flounder | 0.00 | 0.00 | 0.17 | 0.00 |
| <i>Dorosoma cepedianum</i> | gizzard shad | 0.00 | 0.00 | 0.00 | 3.00 |
| <i>Hyporhamphus unifasciatus</i> | halfbeak | 0.00 | 0.00 | 0.67 | 0.00 |
| <i>Arius felis</i> | hardhead catfish | 0.00 | 0.00 | 0.00 | 0.50 |
| <i>Tylosurus crocodilus</i> | houndfish | 0.00 | 0.00 | 0.33 | 0.00 |
| <i>Menidia beryllina</i> | inland silverside | 5.33 | 0.00 | 8.00 | 0.00 |
| <i>Synodus foetens</i> | inshore lizardfish | 0.83 | 2.00 | 0.17 | 0.50 |
| <i>Cyprinodontiformes spp.</i> | killifish | 0.00 | 0.00 | 2.50 | 0.00 |
| <i>Micropterus salmoides</i> | largemouth bass | 0.00 | 0.50 | 0.00 | 1.50 |
| <i>Lepisosteus osseus</i> | longnose gar | 0.00 | 0.00 | 0.17 | 2.00 |
| <i>Selene vomer</i> | lookdown | 0.00 | 0.00 | 0.17 | 0.00 |
| <i>Brevoortia smithi</i> | menhaden | 4.00 | 9.00 | 1.83 | 21.50 |
| <i>Gambusia holbrooki</i> | mosquito fish | 0.00 | 0.50 | 8.50 | 0.00 |
| <i>Mugil spp.</i> | mullet | 0.33 | 0.00 | 1.67 | 0.00 |

| | | | | | |
|------------------------------------|-------------------------|-------|-------|--------|-------|
| <i>Fundulus heteroclitus</i> | mummichog | 9.33 | 1.50 | 21.83 | 2.00 |
| <i>Gobiosoma boscii</i> | naked goby | 0.33 | 0.00 | 2.17 | 0.00 |
| <i>Strongylura marina</i> | needlefish | 0.00 | 0.00 | 0.83 | 1.50 |
| <i>Prionotus carolinus</i> | northern searobin | 0.50 | 0.00 | 0.00 | 0.00 |
| <i>Sphyræna borealis</i> | northern sennet | 0.17 | 0.00 | 0.17 | 0.00 |
| <i>Opsanus tau</i> | oyster toadfish | 0.50 | 0.00 | 0.00 | 0.00 |
| <i>Trachinotus falcatus</i> | permit | 0.00 | 1.50 | 0.00 | 0.00 |
| <i>Orthopristis chrysoptera</i> | pigfish | 1.33 | 3.00 | 10.83 | 4.50 |
| <i>Lagodon rhomboides</i> | pinfish | 97.67 | 12.50 | 114.67 | 11.50 |
| <i>Syngnathus species</i> | pipefish | 0.17 | 0.00 | 0.00 | 0.00 |
| <i>Monacanthus setifer</i> | pygmy filefish | 0.33 | 0.50 | 0.00 | 0.00 |
| <i>Lucania parva</i> | rainwater killifish | 19.00 | 0.50 | 26.00 | 0.00 |
| <i>Sciaenops ocellatus</i> | red drum | 0.33 | 0.50 | 1.17 | 7.00 |
| <i>Archosargus probatocephalus</i> | sheepshead | 0.00 | 0.50 | 0.00 | 1.50 |
| <i>Cyprinodon variegatus</i> | sheepshead minnow | 16.67 | 0.00 | 15.50 | 0.00 |
| <i>Bairdiella chrysoura</i> | silver perch | 2.33 | 2.50 | 7.17 | 22.00 |
| <i>Atheriniformes spp.</i> | silverside | 7.00 | 0.00 | 8.00 | 0.00 |
| <i>Gymnura micrura</i> | smooth butterfly ray | 0.00 | 4.00 | 0.00 | 3.00 |
| <i>Mustelus canis</i> | smooth dogfish | 0.00 | 0.00 | 0.00 | 1.50 |
| <i>Paralichthys lethostigma</i> | southern flounder | 2.33 | 4.00 | 0.67 | 12.00 |
| <i>Menticirrhus americanus</i> | southern kingfish | 0.00 | 0.00 | 0.00 | 0.50 |
| <i>Chaetodipterus faber</i> | spadefish | 0.00 | 0.00 | 0.00 | 0.50 |
| <i>Myrophis punctatus</i> | speckled worm eel | 0.00 | 0.00 | 0.17 | 0.00 |
| <i>Leiostomus xanthurus</i> | spot | 20.50 | 29.50 | 16.33 | 42.00 |
| <i>Eucinostomus argenteus</i> | spotfin mojarra | 4.17 | 0.00 | 7.67 | 5.50 |
| <i>Cynoscion nebulosus</i> | spotted sea trout | 0.33 | 0.00 | 0.67 | 1.50 |
| <i>Citharichthys macrops</i> | spotted whiff | 0.17 | 0.00 | 0.00 | 0.00 |
| <i>Anchoa hepsetus</i> | striped anchovy | 0.00 | 0.00 | 0.17 | 0.00 |
| <i>Fundulus majalis</i> | striped killifish | 3.83 | 0.00 | 9.00 | 1.00 |
| <i>Mugil cephalus</i> | striped mullet | 1.83 | 11.00 | 20.50 | 50.50 |
| <i>Paralichthys dentatus</i> | summer flounder | 1.83 | 3.00 | 0.83 | 2.00 |
| <i>Opisthonema oglinum</i> | thread herring | 0.00 | 3.50 | 0.00 | 1.50 |
| <i>Gasterosteus aculeatus</i> | three spine stickleback | 0.17 | 0.00 | 0.00 | 0.00 |
| <i>Symphurus spp.</i> | tongue sole | 0.17 | 0.00 | 0.67 | 0.50 |
| <i>Mugil curema</i> | white mullet | 0.33 | 0.00 | 0.83 | 0.00 |
| <i>Morone americana</i> | white perch | 0.00 | 4.50 | 0.50 | 8.50 |
| <i>Perca flavescens</i> | yellow perch | 0.00 | 0.00 | 0.17 | 0.00 |
| Other | | | | | |
| <i>Lolliguncula brevis</i> | squid | 0.17 | 0.00 | 2.83 | 0.00 |

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CHAPTER 2. INFLUENCE OF SHORELINE STABILIZATION STRUCTURES ON BIRD ASSEMBLAGES IN SALT MARSH HABITATS.

Introduction

Where the land at the back of the coastal marsh gradually slopes upwards, a coastal marsh can transgress shoreward in response to rising sea level. However, the presence of shoreline stabilization structures, such as bulkheads, prevents transgression of the marsh habitat, which becomes squeezed between the fixed structure and the rising water (Titus 1998, Peterson et al. 2008). Additionally when the marsh surface elevation does not increase at the rate of sea-level rise (SLR), the marsh may become flooded and eventually transform into an open water habitat (Orson et al 1985). Much of the current literature concurs that if salt marsh habitat is prevented from transgressing naturally, tidal wetlands and salt marsh habitats will be lost (Donnelly and Bertness 2001, Wilson et al 2007, Brittain and Craft 2012).

The ecosystem services provided by salt marshes have been extensively studied (MEA 2005). Important fish stocks use the salt marsh for nursery habitat, feeding grounds, and refuge from predation (Titus 1998). Birds also play important functional roles in marshes as they disperse seeds of marsh plants, fertilize marsh plants through guano deposition, and prey on fish and other marsh invertebrates (Weller 1999, Brittain and Craft 2012). Several waders, shorebirds, and terrestrial birds choose marsh habitat as breeding, nesting, and chick rearing grounds (Wilson *et al.* 2007). Several species of migratory birds use salt marsh habitats to consume high-energy foods to fuel their energy demands of migration and breeding in northern latitudes. Understanding how shoreline management interventions may affect bird usage of

marsh habitat thus has value to managers who are motivated to sustain the ecosystem services of marshes to coastal birds.

Positive correlations exist between bird species abundance (Naugle et al 1999) and richness (Findlay and Houlihan 1997) to salt marsh area. The biology of many birds requires large expanses of habitat for nest seclusion and reduced predation (Johnson and Temple 1990) and, in the absence of wide areas, many species decline. Larger expanses of habitat often support more species of birds (Watts 1992, Burke and Nol 1998, Craig 2008, Wilson *et al.* 2007), rarer species, higher abundances of food sources, and support greater levels of primary and secondary production (Erwin *et al.* 2006). Continued loss of species-specific minimum habitat requirements has pushed species to extinction. For example, the dusky seaside sparrow *Ammodramus maritimus nigrescens*, has been extinct since 1987 (Estes 1992). Clapper rails and semipalmated sandpipers are species recorded in our study that are under special conservation concern, or “yellow listed” by the National Audubon Society.

Not only are birds indicators of food availability in the lower trophic levels, they are also the target of recreational birdwatchers and have an avid fan base. Bird watching is a frequent use of salt marshes and such coastal habitats support high species richness and abundance of birds (Wilson and Brinker 2007).

Study Area

We conducted our censuses of birds in coastal marshes at six local sites within three regions of coastal North Carolina: a northern region in Kitty Hawk Bay (KH), a central region in Bogue Sound at Pine Knoll Shores (PKS), and a southern region along the Intracoastal Waterway in Wilmington on Harbor Island and Oak Island (WILM). Observations were made during August 18-24 and October 13-15, 2010 for WILM, July 27-August 2 and October 4-8, 2010 for PKS, and August 15-20 and September 24-28, 2010 for KH.

The six coastal marsh sites were chosen in each region within shoreline stretches of similar wave energy (Currin 2008). Other geochemical characteristics of these sites including: salinity, temperature, and denitrification rates can be found in O’Meare *et al.* (2014). Each bulkheaded marsh had a bulkhead landward of the upper marsh constructed of wood, concrete, and/or metal. These five sites included a no marsh or bulkhead only site with no vegetation present, a narrow marsh site 3-5 m wide, two medium width marsh sites 10-20 m wide, and a wide marsh site > 20 m. The natural marsh sites 19-40 m wide did not contain shoreline stabilization structures.

Methods

Observations were conducted, using Leica 8x42 Ultravid HD #40293 binoculars and were initiated at sunrise each day and repeated for six consecutive days. Visual surveys were conducted at each site for 10 minutes from a neighboring yard to ensure that our presence did not disturb birds in the site-specific area. Each region was sampled once in the summer 2010 and again in the fall 2010 to estimate abundances of all bird species in the site-specific marsh and surrounding habitats. All six sites were observed each day in rotation so that each day, a different site was observed first at sunrise, to reduce the effects of site-specific anthropogenic disturbances that may occur at later. All birds physically in the site marsh vegetation or within 10 m of the site marsh edge were identified and counted, as well as birds less than 100 m from the site marsh. In addition, we identified birds greater than 100 m from the marsh and flyovers. We notated general behaviors of all birds observed. Separate analyses were performed for birds in the marsh from those observed greater than 10 m or flyovers. Combined analyses were also performed for total bird observations for any regional effects on species abundance and/or composition.

The above sequence enabled us to record observations during several tidal flooding stages as different species of birds utilize the marsh habitat differently based on foraging regimes (Recher 1966). Because some sites were public and some were privately owned, we could not control outside disturbances such as boats, dog visits, homeowner activities, and road traffic. These non-avian activities were recorded as well as general weather conditions, such as wind speed, wind direction, and sky conditions.

Bird species were divided into trophic guilds based on foraging behavior and food preferences: 1) terrestrial birds (barn swallows, sparrows, grackles, red-winged blackbirds, etc.); 2) wading birds (egrets, ibis, herons, etc.); 3) piscivorous birds (raptors, pelicans, kingfishers, cormorants, terns, etc.); 4) shorebirds (sandpipers, dunlin, sora, etc.); 5) rails (clapper rails); 6) dabblers (mallards) and; 7) scavengers (fish crows, gulls, etc.). Terrestrial birds may use the marsh for foraging but are not obligate marsh species. Wading birds forage in shallow open water as well as in *Spartina* stands. Piscivorous birds were often observed flying overhead and then foraging in open water. Shorebirds are represented by a few obligate marsh species that breed and nest in the upper reaches of the marsh. Dabblers and scavengers were observed foraging in the open water of the site marsh and foraging within the marsh.

To detect patterns in species assemblages among regions and marsh types, we used a non-metric multi-dimensional scaling (nMDS) ordination plots based on Bray-Curtis similarities and performed a permutational analysis of variance (PERMANOVA) on square root transformed data of birds in the site marsh and a separate analysis on all bird observations that was $\log(x+1)$ transformed. nMDS and PERMANOVA tests were performed using PRIMER v. 6.0 (Clarke and Gorley 2006).

A one-way analysis of variance (ANOVA) was performed to determine if differences in bird abundances and marsh widths among sites. If significance was indicated, then pairwise multiple comparisons (Holm-Sidak method) were used to test for significant differences among the sites. We eliminated a flock of 22 semipalmated sandpipers that landed and began foraging on mats of Eurasian milfoil (*Myriophyllum spicatum*) in a bulkhead only site from this test. This was considered an outlier because if the milfoil was not present, the birds would not have been able to land as this site was constantly inundated throughout the study. The one-way ANOVA was performed in SigmaPlot 11 (Systat Software Inc.).

Results

Bird species assemblages

A total of 4004 birds representing 43 species was identified from the three regions in study. We considered 172 birds representing 24 species, to be actively using the site marsh for foraging, resting, etc. for our analyses. We also identified birds as flyovers and within 100 m of site marsh (n=1928), and birds beyond 100 m of the salt marsh edge (n=1904).

When using the subset of birds (n=172) observed in our site marshes, the natural marsh sites (NM) revealed the greatest species diversity, represented by 15 of the 24 total species, and the highest abundance (n=69) (Table 3). A PERMANOVA, (permutations=4999), of bird species in the marsh, showed significant differences (p=0.043) in species composition between sites (Fig. 12). Pair-wise comparisons between the sites give a significant difference (p=0.048) in bird species composition between BWM (bulkheaded wide marsh) and NM (natural marsh). Pair-wise comparison of regions detected a significant difference (p=0.019) in species composition between PKS and WILM.

When all bird observations are considered, including those beyond 100 m (n=4004), a PERMANOVA showed a significant difference among regions (p=0.0002) with pair-wise

significant differences in abundances within the three different regions ($p=0.0004$ KH, PKS) ($p=0.0014$ KH, WILM) ($p=0.0026$ PKS, WILM) and no significant differences between sites (Fig. 13).

Species richness of birds in our site marshes (Table 3) showed no statistically significant difference among marsh sites ($p<0.148$) but abundances and species richness of marsh birds differed in each region. Species richness was greater in the natural marsh site in WILM than in the other regions with nine species (Table 3). In PKS, the seven species were identified in the site marsh from the bulkhead small marsh site. Species richness in the KH region was equal in the bulkhead small marsh and the natural marsh site, with five species (Table 3).

Bird abundance

Total abundance of birds placed into trophic guilds (Fig. 14) indicated a higher percentage (40%) of wading birds observed from the natural marsh sites. Shorebird abundance within the site marsh, was greatest at the bulkhead no marsh sites, in the KH region solely due to one flock ($n=22$) of semipalmated sandpipers (*Calidris pusilla*) that were able to land on floating mats of Eurasian milfoil (*Myriophyllum spicatum*). This observation was considered an outlier and eliminated from further analyses. Shorebirds were also abundant at the bulkhead small marsh sites, most notably in PKS. A one-way ANOVA detected statistical significance between bird abundances by marsh width ($p<0.019$) on square root transformed data. Pairwise analysis of the site types (Holm-Sidak method) indicated one significant difference ($p<0.002$) (Fig. 15) between bird abundances in natural marsh sites and bulkhead only sites when the outlier from the bulkhead small marsh site in KH was removed.

Discussion

Several studies have focused on bird use in salt marsh habitats that have been compromised, whether by created marshes (Darnell and Smith 2004), restored marshes (Brawley et al. 1998), rising sea level (Erwin et al. 2006), or the addition of long piers (Banning et al. 2007). Ours is one of the first studies to observe bird abundances, densities, and species compositions between bulkheaded and non-bulkheaded marshes of varying widths. Positive correlations between bird species abundance and the area of suitable habitat have been observed in several salt marsh ecosystems (Naugle et al. 1999, Benoit and Askins 2002, and Wilson et al. 2007). Brush et al. (1986) found that marsh passerine abundances were lower due to the removal of marsh vegetation and Darnell and Smith (2004) reported wetland habitats were rarely used by wading birds if the amount of habitat did not reach a foraging depth defined by that species.

Our study did support our hypothesis that higher numbers of birds would be associated with natural marshes, but we must also acknowledge that many shorebird species are often associated with bare substrate (Darnell and Smith 2004). Piscivores are aerial fishers and cormorants are deep divers. We would not expect their densities to be correlated directly to salt marsh widths, even though salt marsh area could influence the availability of prey items for piscivores or the amount of habitat required for nesting.

The total amount of marsh area does influence bird communities. Benoit and Askins (2002) state minimum habitat areas for several salt marsh obligate species such as salt marsh sharp-tailed sparrows (10 ha), seaside sparrow species (67 ha), marsh wrens (8 ha), and swamp sparrows (55 ha). The larger expanses of habitat required by these species could, explain the absence of these species in our study. In addition to minimum marsh area requirements, marsh passerines are notoriously shy and difficult to identify with any confidence in this study, as many of our site marshes were located near disturbed habitats such as roadside areas and private yards

where other sparrow species could have easily been misidentified. Saltmarsh sharp-tailed sparrows frequent our coasts in the winter and this could also account for their absence in this study. The cryptic behavior of certain species birds could have also influenced detection, especially soras and clapper rails. Often clapper rails were not seen but heard in marshes of the central region, where they were piqued for response when a call was played on an external playback device.

Differences in species richness between sites in the bulkhead narrow marsh site in PKS could be due to the proximity of a wider, more secluded marsh adjacent to this site often with wading birds observed walking the shoreline from this area to our site marsh. The KH region shared many species found in both PKS and WILM regions but also had two species, sora (*Porzana carolina*) and semipalmated sandpiper (*Calidris pusilla*), that were only observed in this region during this study (Table 3).

Plant species were identified from each region. The central and southern region was characterized by *Spartina alterniflora*, *S. patens*, and *Phragmites australis*. The northern site was characterized by *Phragmites australis* and *Juncus roemerianus* but also Eurasian milfoil (*Myriophyllum spicatum*). Milfoil is an invasive flowering plant that dominates many freshwater and coastal ecosystems of North America (Martin and Valentine 2011, Duffy and Baltz 1998). The presence of milfoil has been shown to increase the value of salt marsh habitats by serving as a proxy for habitat when natural salt marsh plant species are not present (Martin and Valentine 2011). During one summer observation period in KH, twenty-two semipalmated sandpipers (*Calidris pusilla*) were recorded at the no marsh, bulkhead only site. The semipalmated sandpiper is a relatively small, lightweight shorebird and was observed flying from a small sandy beach 20 m north and landing on the dense milfoil mats where they began foraging. This was

the only occurrence of any bird species documented in the no marsh, bulkhead only sites from each region over the entire study. This site as well as the bulkhead only site in the PKS region was constantly inundated, even at low tide, which may account for the lack of observations as many shorebirds prefer to forage on exposed sediments. The milfoil provided structure and habitat for smaller nekton, which serve as a food source for many species of shorebirds, including the semipalmated sandpiper. The presence of milfoil and the higher abundances of bird species in the KH sites, support the contention that while invasive species may competitively exclude native species, they may serve as useful substitutes by providing specific services by offering refuge and structure (Martin and Valentine 2011).

This study focused on coastal marsh erosion and the effects on the avian community by using bulkheaded marshes as a space- for- time substitution. We were able to deduce that thinner marshes could be representative of a bulkhead or other shoreline stabilization structure, and subjected to SLR for a longer period of time than a medium or a wider marsh site. The current literature regarding SLR and various species, communities, and populations is growing and the consensus remains unchanged. If marsh transgression is prevented by hardened structures and the sea continues to rise, our tidal wetlands become squeezed between the structure and the rising waters. The mechanism of marsh decline is largely wave erosion at the marsh edge, and perhaps erosion at the bulkhead foot by waves reflected off the bulkhead itself. With our current rates of SLR, (1-2 mm yr⁻¹; IPCC 2013) coastal areas are likely to continue to lose critical marsh habitat. Greenberg (2006) describes tidal wetland areas to be home to 25 vertebrate species and subspecies that are considered endemic to tidal marshes. The preservation of our wetland habitats is vital to these species survival. Our tidal marshes face a multitude of problems, in addition to SLR, including development, pollution, salinity changes exacerbated by

SLR, and invasive species (Greenberg 2006). If we are to preserve our coastal marsh habitats then we must provide sustainable alternatives to bulkheads and other shoreline structures that eventually eliminate our marshes. Our research efforts should move forward to the development of these alternatives and further quantification of bird use patterns in our coastal marshes.

Table 3. Abundance and species composition of birds from each site observed from each region.

| Region | Species | Site | | | | | | |
|-----------|--------------------------|------------------------------|-----|-----|------|-----|----|---|
| | | BNM | BSM | BMM | BMM2 | BWM | NM | |
| KH | | | | | | | | |
| | Wading Birds | | | | | | | |
| | Great Blue Heron | <i>Ardea herodias</i> | — | — | 2 | 1 | — | — |
| | Great Egret | <i>Ardea alba</i> | — | — | — | 1 | — | — |
| | Snowy Egret | <i>Egretta thula</i> | — | — | 4 | — | — | — |
| | Shorebirds | | | | | | | |
| | Least Sandpiper | <i>Calidris minutilla</i> | — | 18 | — | — | — | 7 |
| | Semipalmated Sandpiper | <i>Calidris pusilla</i> | 22 | 3 | — | — | — | — |
| | Spotted Sandpiper | <i>Actitis macularius</i> | — | — | — | 1 | — | — |
| | Sora | <i>Porzana carolina</i> | — | 1 | — | — | — | — |
| | Piscivorous Birds | | | | | | | |
| | Royal Tern | <i>Thalasseus maximus</i> | — | — | — | — | 1 | — |
| | Double-crested Cormorant | <i>Phalacrocorax auritus</i> | — | — | — | 1 | — | — |
| | Belted Kingfisher | <i>Ceryle alcyon</i> | — | — | — | — | — | 1 |
| | Dabblers | | | | | | | |
| | Mallard | <i>Anas platyrhynchos</i> | — | 2 | — | — | — | — |
| | Terrestrial Birds | | | | | | | |
| | Red-winged Blackbird | <i>Agelaius phoeniceus</i> | — | 5 | — | — | — | 2 |

| | | | | | | | | |
|------------|--------------------------|------------------------------|---|---|---|---|---|----|
| | Sparrow spp. | | — | — | — | — | — | 10 |
| | Boat-tailed Grackle | <i>Quiscalus major</i> | — | — | — | — | — | 2 |
| PKS | | | | | | | | |
| | Wading Birds | | | | | | | |
| | Reddish Egret | <i>Egretta rufescens</i> | — | 1 | — | — | — | — |
| | Tricolored Heron | <i>Egretta tricolor</i> | — | — | 1 | — | — | 1 |
| | Shorebirds | | | | | | | |
| | Spotted Sandpiper | <i>Actitis macularius</i> | — | 1 | — | — | — | — |
| | Clapper Rail | <i>Rallus longirostris</i> | — | — | — | — | — | 1 |
| | Dunlin | <i>Calidris alpina</i> | — | 1 | — | — | — | — |
| | Piscivorous Birds | | | | | | | |
| | Belted Kingfisher | <i>Ceryle alcyon</i> | — | — | — | 1 | — | 1 |
| | Osprey | <i>Pandion haliaetus</i> | — | — | — | — | — | 2 |
| | Double-crested Cormorant | <i>Phalacrocorax auritus</i> | — | 1 | — | 1 | — | — |
| | Dabblers | | | | | | | |
| | Mallard | <i>Anas platyrhynchos</i> | — | — | — | 4 | — | 4 |
| | Scavengers | | | | | | | |
| | Laughing Gull | <i>Larus atricilla</i> | — | — | — | — | 1 | — |
| | Terrestrial Birds | | | | | | | |
| | Red-winged Blackbird | <i>Agelaius phoeniceus</i> | — | 2 | — | — | — | — |

| | | | | | | | |
|--------------|--|---|---|---|---|---|---|
| Sparrow spp. | | — | 4 | — | — | — | — |
|--------------|--|---|---|---|---|---|---|

| | | | | | | | |
|---------------------|------------------------|---|---|---|---|---|---|
| Boat-tailed Grackle | <i>Quiscalus major</i> | — | 4 | 2 | 5 | — | — |
|---------------------|------------------------|---|---|---|---|---|---|

WILM

Wading Birds

| | | | | | | | |
|------------------|-----------------------|---|---|---|---|---|---|
| Great Blue Heron | <i>Ardea herodias</i> | — | 1 | — | 1 | — | 6 |
|------------------|-----------------------|---|---|---|---|---|---|

| | | | | | | | |
|-------------|-------------------|---|---|---|---|---|---|
| Great Egret | <i>Ardea alba</i> | — | — | — | — | 2 | 7 |
|-------------|-------------------|---|---|---|---|---|---|

| | | | | | | | |
|-------------|----------------------|---|---|---|---|---|---|
| Snowy Egret | <i>Egretta thula</i> | — | — | — | — | — | 1 |
|-------------|----------------------|---|---|---|---|---|---|

| | | | | | | | |
|------------------|-------------------------|---|---|---|---|---|---|
| Tricolored Heron | <i>Egretta tricolor</i> | — | — | 1 | — | — | 6 |
|------------------|-------------------------|---|---|---|---|---|---|

| | | | | | | | |
|-------------|----------------------------|---|---|---|---|---|---|
| Green Heron | <i>Butorides virescens</i> | — | 1 | — | — | — | 1 |
|-------------|----------------------------|---|---|---|---|---|---|

| | | | | | | | |
|------------|------------------------|---|---|---|---|---|---|
| White Ibis | <i>Eudocimus albus</i> | — | — | — | — | — | 7 |
|------------|------------------------|---|---|---|---|---|---|

Shorebirds

| | | | | | | | |
|--------------|----------------------------|---|---|---|---|---|---|
| Clapper Rail | <i>Rallus longirostris</i> | — | — | — | — | 1 | 4 |
|--------------|----------------------------|---|---|---|---|---|---|

Piscivorous Birds

| | | | | | | | |
|------------|---------------------------|---|---|---|---|---|---|
| Royal Tern | <i>Thalasseus maximus</i> | — | — | — | — | 2 | — |
|------------|---------------------------|---|---|---|---|---|---|

| | | | | | | | |
|-------------------|----------------------|---|---|---|---|---|---|
| Belted Kingfisher | <i>Ceryle alcyon</i> | — | 1 | — | — | — | 5 |
|-------------------|----------------------|---|---|---|---|---|---|

| | | | | | | | |
|---------------|-------------------------------|---|---|---|---|---|---|
| Brown Pelican | <i>Pelecanus occidentalis</i> | — | — | — | — | 2 | — |
|---------------|-------------------------------|---|---|---|---|---|---|

Terrestrial Birds

| | | | | | | | |
|--------------|------------------------|---|---|---|---|---|---|
| Barn Swallow | <i>Hirundo rustica</i> | — | — | — | — | — | 1 |
|--------------|------------------------|---|---|---|---|---|---|

KH (Kitty Hawk), PKS (Pine Knoll Shores), WILM (Wilmington and Oak Island). BNM (Bulkhead only), BSM (Bulkhead small marsh), BMM (Bulkhead medium marsh), BMM2 (Bulkhead medium marsh 2), BWM (Bulkhead wide marsh), NM (Natural marsh, no bulkhead).

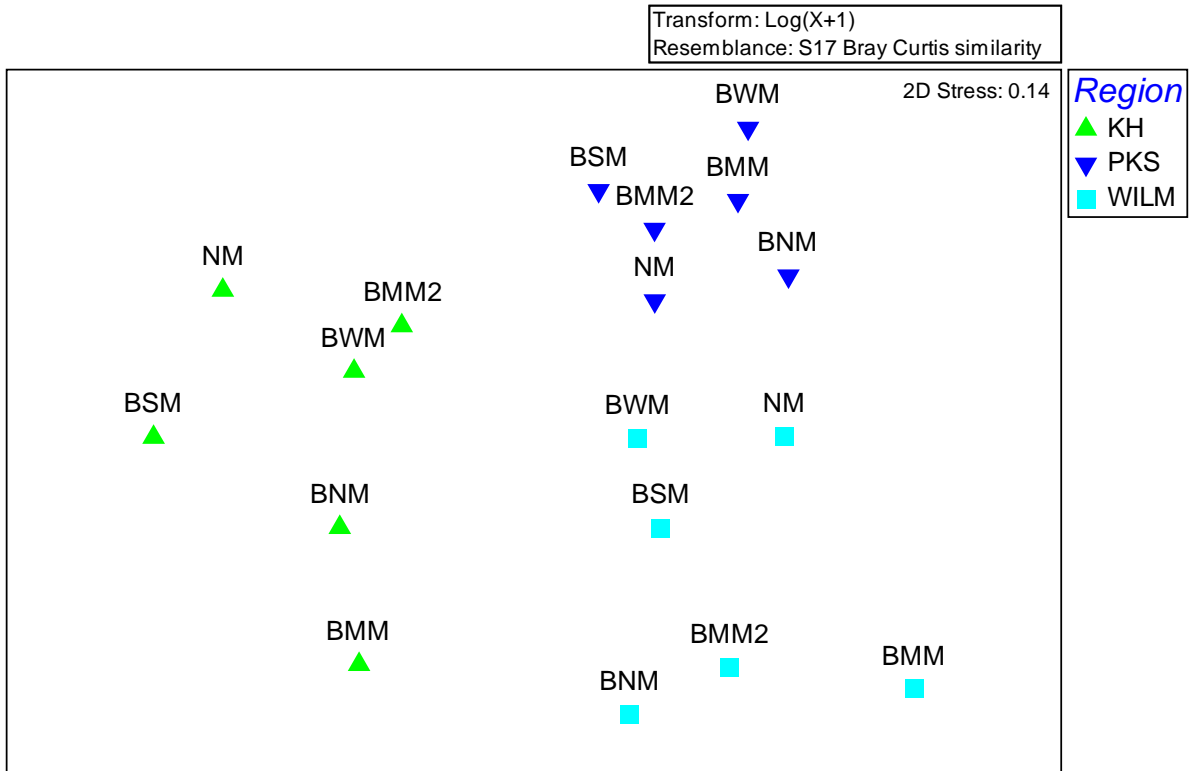


Figure 12. MDS plot of bird community composition of all birds observed from each region, performed as a Bray-Curtis similarity index. D stress = 0.05. Data were log(x+1) transformed.

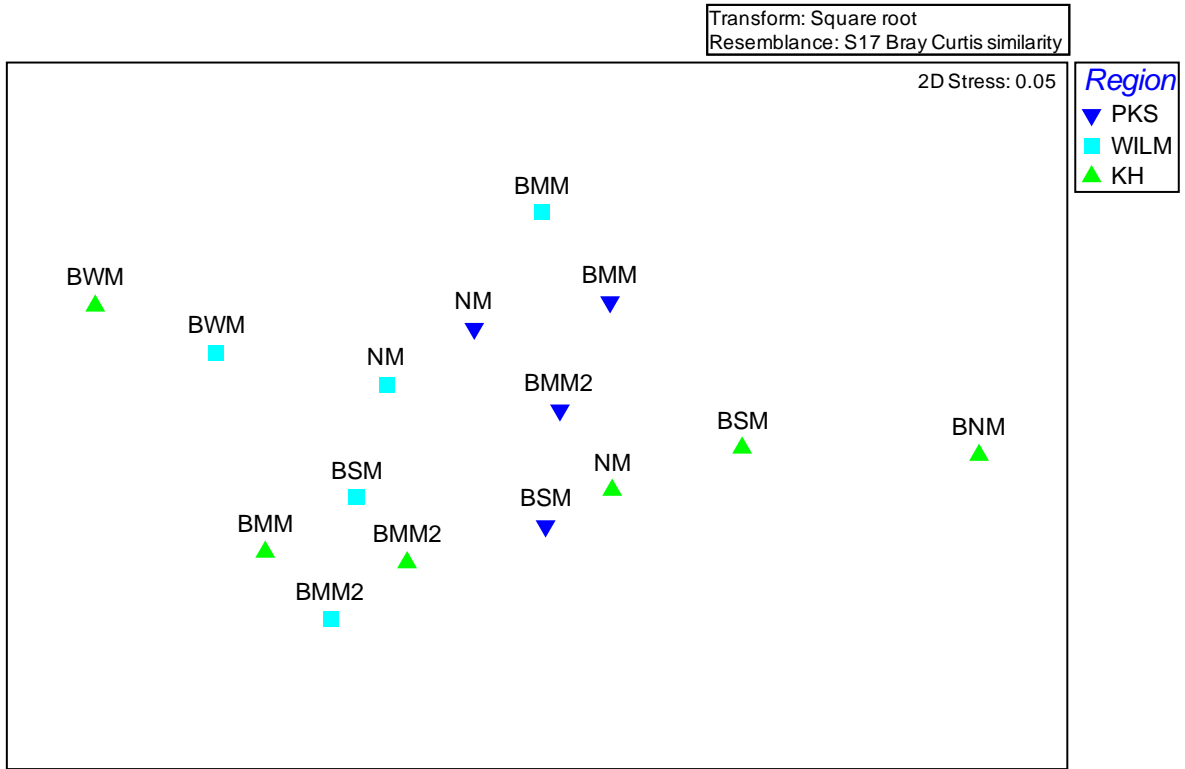


Figure 13. MDS plot of bird community compositions of birds described as in the target zone for the three regions, performed as a Bray-Curtis similarity index. D stress = 0.05. Data were square root transformed.

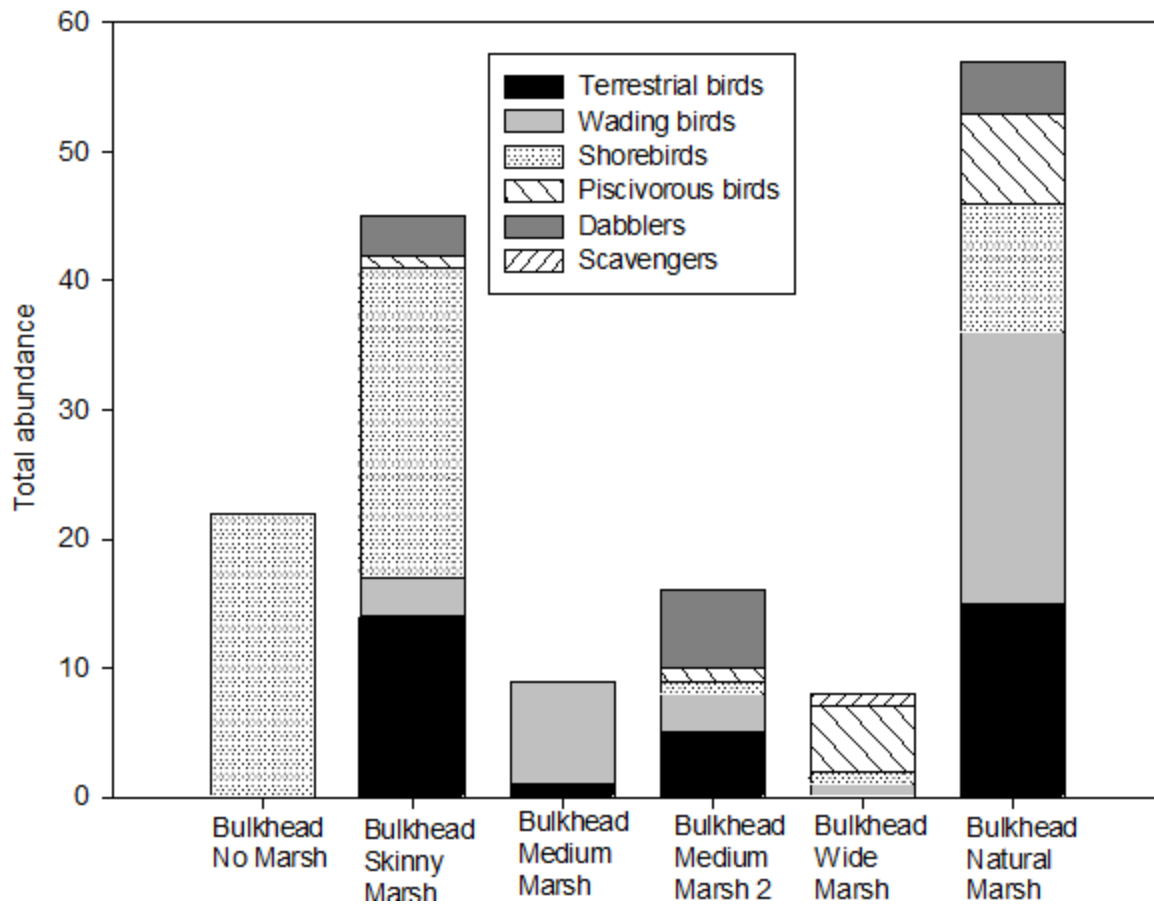


Figure 14. Total abundance of bird species by dominant group. Marsh types were combined from each region.

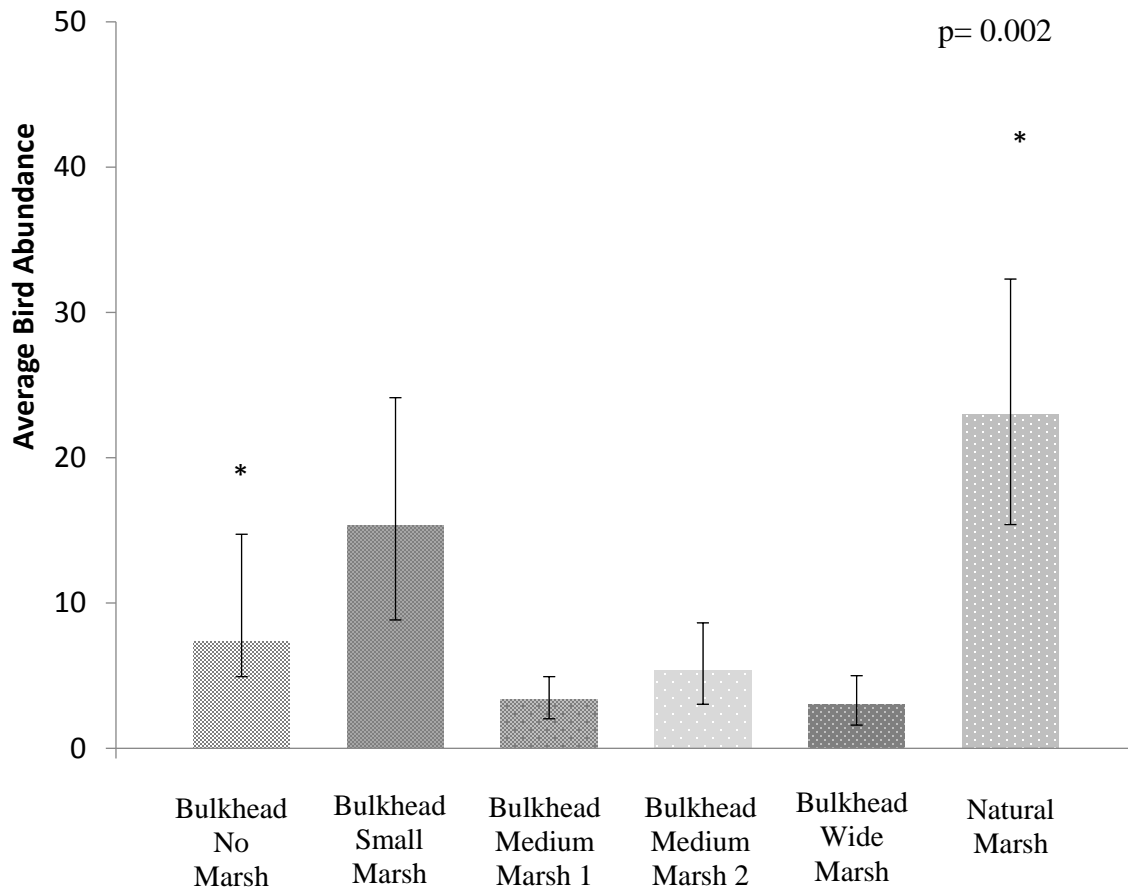


Figure 15. The average bird abundance observed in each marsh type combined from each region. There is a significant difference ($p=0.002$) from abundances in the Natural Marsh site and the Bulkhead No Marsh site. Data were square root transformed.

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CHAPTER 3. ROLES OF OYSTER FEEDING AND BIODEPOSITION IN AFFECTING COASTAL MARSH FOOD WEBS.

Introduction

One of the major human transformations that has affected estuarine ecosystems worldwide is the overharvesting, depletion, and destruction of oysters and hence oyster reef habitat (Rothschild et al. 1994, Lenihan and Peterson 1998, Jackson et al. 2001, Lotze et al. 2006). This transformation has resulted in dramatic declines in oyster fisheries with consequent impacts on fishermen and coastal economies. In response to the recognition that oyster stocks were severely depleted, oyster restoration has been practiced by several states on the Atlantic and Gulf coasts for at least two decades. Until about a decade ago, this restoration effort was almost entirely directed towards restoring the fishery, and typically consisted of planting oyster shell or some other hard analog as cultch, and then allowing “put-and-take” fishing on the oysters that settled and grew to harvestable sizes. However, within the past few years, the value of oysters as habitat and as an ecosystem engineer that provides valuable ecosystem services to the estuary has been recognized (Grabowski and Peterson 2007), transforming and broadening the purposes of oyster restoration.

The ecosystem services provided by oyster reefs are numerous and can be extremely valuable (Lenihan and Peterson 1998, Luckenbach et al. 1998, Coen et al. 2007, Grabowski and Peterson 2007, Piehler and Smyth 2011, Grabowski et al. 2012). Oysters have been documented to offer stable hard substratum in a system of otherwise mobile sediments, dramatically enhancing the diversity of associated benthic invertebrates (Wells 1961). The reefs that oyster’s form and the associated benthic organisms offer prey and habitat for mobile crustaceans and

fishes (Meyer and Townsend 2000, Lenihan et al. 2001). Through their filtration, oysters lower turbidity in the water column and can thereby enhance growth of SAV (submerged aquatic vegetation) habitat in nearby shallow waters (Dame 1996, Newell et al. 2002). Suspension feeders like oysters and other bivalves, package particulates into larger, aggregated pellets, some as pseudofeces that are rejected before ingestion, and the rest as feces representing undigested materials (Wootton and Malmqvist 2001, Haven and Morales-Alamo 1966). The deposition of these biodeposits induces net denitrification in the surface sediments around oysters, (Piehler and Smyth 2011), thereby converting nitrogen nutrients into relatively unreactive N₂ gas (Newell et al. 2002). However, there is another unexplored ecosystem service associated with biodeposits. Biodeposits represent packages of organic-rich foods for deposit-and bottom-feeding invertebrates and many anecdotal observations have suggested that several crustaceans, including grass shrimp and penaeid shrimps, aggregate in oyster deposits, benefiting from the now concentrated food resources. Predatory macroinvertebrates then indirectly benefit from biodeposits by feeding on these associated prey items. Biodeposits influence the benthic community by adding organic matter and nutrients to the sediment, thereby fertilizing and enhancing submerged aquatic vegetation (Peterson and Heck 1999). Nutrient enhancement also increases the benthic microbial and microalgal communities (Radziejewska 1986, Grabowski et al. 2012) and so on up the trophic web extending to trophic level 4 or 5 and even beyond (Fig. 16). Freshwater systems also benefit from the creation of a biodeposition-based food web, as demonstrated by documenting feeding by invasive zebra mussels (*Dreissena polymorpha*) (Gregs and Rothhaupt 2008).

We include analyses with meiofauna because they can serve as an intermediate trophic level between microphytobenthos and microbenthic infauna. Questions about the trophic

dynamics and food-web importance of estuarine meiofauna have been explored (McIntyre and Murison 1973, Smith and Coull 1987), but much of the literature simplifies the complexities often associated with meiofauna and their role in food webs, often considered a trophic dead end (Schrijvers et al. 1998). However, many studies have emphasized the potential of meiofauna as an important source of food for higher trophic levels (Bell and Coull 1978, Bell 1980, Watzin 1983, Smith and Coull 1987). Bivalve beds may also provide physical habitat to meiofauna as they are irregularly shaped and routinely covered with fine sediments by biodeposition and physical particle settlement. (Norling and Kautsky 2007). The primary purpose of the study was to investigate the currently underexplored ecosystem service of the benefits of oyster biodeposition and to assess its quantitative importance to various taxonomic and functional groups of estuarine organisms that can be logically expected to benefit from oysters. We identified the following groups as candidates that could potentially benefit from added biodeposition by oysters: microphytobenthos, epifaunal invertebrates such as snails and amphipods, meiofauna, and grass, and penaeid shrimps. Measuring the survival of these groups may augment understanding of oyster's trophic contributions, expanding the scope of their known ecosystem services.

Materials and Methods

To investigate the role of oyster feeding and biodeposition in stimulating higher trophic level production in marsh food webs, we conducted a mesocosm experiment. Experiments were run at the University of North Carolina, Institute of Marine Sciences in Morehead City, NC between July 10 and October 13, 2009. A mesocosm array (Fig. 17), consisting of twelve round, plastic pools, each 90 cm in diameter and 20 cm deep, received unfiltered seawater (32-34ppt) pumped from Bogue Sound, NC. Six replicate pools were used to determine the effects of living oyster clumps (*Crassostrea virginica*), hereafter known as live pools and six pools served as a

control treatment with an equivalent volume of dead oyster shells added, hereafter known as sham oyster pools. Eighty six oysters were divided among the live pools with an average length of 5.2 cm and an average greatest width of 2.5 cm. Ninety four clumped oyster shells were divided among the sham pools with an average length of 5.5 cm and average greatest width of 3.0 cm. All oysters smaller than 1 cm were removed from the live pool oysters and living oysters and sham shells were scrubbed clean of epibionts to ensure that no other macro-organisms were present. Approximately 0.04 m³ of azoic sediment was added to each pool to maintain a depth of 5 cm and mesh screens, 1 mm², were fastened over pools to prevent or inhibit escape of organisms and provide protection against bird predation. Mesocosms were also drilled with twelve 2 cm diameter holes also covered with mesh screen to allow outflow of the sea-water from the mesocosms.

In addition to oysters, 10 white shrimp (*Penaeus setiferus*), 10 grass shrimp (*Palaemonetes pugio.*), 30 mud snails (*Ilyanassa sp.*), 30 amphipods (*Gammarus sp.*), and an aliquot of meiofauna, retained on a 63 µm sieve collected from 1 L of surface sediment, was added to each of the 12 pools. All other organisms were collected from Hoop Pole Creek in Bogue Sound in Carteret County, North Carolina. After 13 weeks, all organisms present in the pools were re-collected for identification, enumeration, and measurement. Shrimp carapace lengths were measured before and after the experiment and wet weight measurements were also taken at the conclusion of the experiment. Sham and living oysters were inspected for new oyster spat. Dry weight (g) of oyster tissue, length and greatest width (cm), of all living oysters was measured at the conclusion of the experiment. Mud snail total length, aperture length, was measured before and after the experiment. Mud snail wet and dry weight was measured at the end of the experiment. Amphipods and infauna was extracted from the sediment before

enumeration by rinsing and sieving all shells and sediments on a 500 μm sieve. Meiofauna were also collected by randomly extracting 5 cores from the top 2 cm of each pool. Meiofaunal samples were processed by MgCl_2 decantation (Gregg and Fleeger 1998), stained with rose bengal, and identifications made to the lowest taxon possible. Sediments may settle on oyster shells due to their irregular shape and texture. Therefore, to assess meiofaunal abundance on oyster shells, oyster shells were also processed for meiofauna as described the MgCl_2 decantation combined with totals from sediment cores.

Benthic chlorophyll- *a* samples were taken at the experiment's beginning and end to represent primary productivity of microphytobenthos. Three 1 cm^3 of sediment cores were taken from each pool and were analyzed by SHIMADZU spectrophotometry to determine pigment concentrations and using optical density equations set forth by Jeffery and Humphrey (1975).

Statistical Analyses

T-tests were performed on benthic chl- *a* concentrations (mg/m^2), abundances of oyster spat, penaeid shrimp, white shrimp, mud snails, amphipods, and meiofauna before and after the experiment in both treatments. Benthic chl- *a* data were \ln transformed for normality in both treatments for the after data set only. T-tests were performed in SigmaPlot 11 (Systat Software Inc.).

Meiofauna community compositions was also analyzed with non-metric multi-dimensional scaling (nMDS) ordination plots based on Bray Curtis similarities and we also performed a permutational analysis of variance (PERMANOVA) on $\log(x+1)$ transformed data. Non-metric MDS and PERMANOVA tests were conducted using PRIMER v. 6.0 (Clarke and Gorley 2006).

Results

Sham pools had a lower concentration of chl- *a*, 35.89 mg/m² than live pools, 43.06 mg/m² in the beginning of the experiment but these were not significantly different. Chl- *a* concentrations were significantly higher in the live pools than in the sham pools at the end of the experiment ($p < 0.003$) (Fig. 18). The average chl- *a* concentration was 85.47 mg/m² in the sham pools and 150 mg/m² in the live pools.

Counting new oyster spat on sham oyster shells revealed an average addition of 34 live oysters. Subtracting the original number of living oysters (86) from the number at the end of the experiment reveals an addition of 115 new oysters attached to live oysters in the live pools. A *t*-test detected a significant difference ($p < 0.003$) in the abundance of oyster spat found in the live pools vs. the sham pools.

Shrimp carapace growth differences between oyster treatments were compared for grass shrimp (Fig. 20A) and white shrimp (Fig. 20B), but no significant differences were detected between the live and sham pools. We did experience die off of grass and penaeid shrimp, largely due to the caridoid escape reaction, a flicking of the telson, often allowing them to slip between the mesocosm and the mesh screen cover.

A marginally significant difference was found in the mud snail (Fig. 20C) population ($p < 0.051$) between the sham and live mesocosms. Snail abundance decreased in the sham mesocosms by 0.65% and we were able to retrieve the same number of snails ($n=180$) from the live mesocosms before and after the experiment. Though no significant differences were observed in amphipod abundance (Fig. 20D), amphipods were more numerous in the sham pools after the experiment. The *n*-MDS analysis (Fig. 21) of meiofaunal communities did not indicate any significant differences in composition or abundance (Fig. 22) between the live and sham pools.

Discussion

Our mesocosm experiment was designed to evaluate experimentally whether feeding and biodeposition of oysters induces detectable benefits to common marsh organisms at higher trophic levels. Results revealed evidence that the combination of oyster filtration and biodeposition stimulated greater production of microphytobenthos on the sediment surface. This response could be driven by greater water clarity in the pools with live, filtering large oysters or by enhanced fertilization through deposition of nutrient-rich biodeposits, or the combined effects of both processes.

We observed no significant differences in the abundance of: meiofauna, amphipods, or shrimp in the live and sham pools. Increased mud snail abundance was marginally significant ($p < 0.051$) in the live pools. Our observations also show that oyster biodeposition facilitated the growth and production of microphytobenthos in the surface sediments of the pools. Both live and sham pools had similar concentrations of benthic chl- *a* upon the initiation of the experiment; and both treatments displayed a significant increase in the amount (mg/m^2) of microphytobenthos present in the soil over the 3- month experimental period. The live pools with biodeposition had significantly higher ($p < 0.003$) chl- *a* abundances than those with sham oysters, indicating that oysters enhanced the microphytobenthic production.

Live pools also had significantly higher levels of oyster recruitment. Because individual oysters used in the live pools were counted and measured, and all oyster spat smaller than 1 cm were removed from the sham pools at the beginning of the experiment, we are able to deduce that all new oyster recruits were introduced from the unfiltered seawater and most likely not from within-pool spawning events. Spawning tends to occur between temperatures of 18-25°C (Stanley and Sellers 1986, Ingle 1951), and due to the higher water temperature of the mesocosms, spawning may have been re-initiated. Our data show that the live pools with the

lowest amount of oyster spat recruitment is still significantly higher than the amount of oyster spat recruitment in the sham pools implying that larval recruits rely on some trigger emitted by living oysters. The exact mechanism of how pediveligers settling is still to be explained however, most studies offer chemical cues or scent plumes as the most likely rationale. There are other possibilities, currently being evaluated including the possibility of pre-recorded oyster reef sounds (Lillis et al. 2013).

Though amphipod abundance was not significantly different in the live and sham pools at the end of the experiment, we did experience a 30% increase in amphipod abundance in the live pools and a 156% increase in amphipod abundance in the sham pools. This result is not consistent with other studies that show increased amphipod abundances when bivalve biodeposition is available (Gergs and Rothhaupt 2008). Gammarid species may occupy several feeding niches (Duffy 1990, Zimmerman et al. 1979, Williams and Bynum 1972) and we expected higher abundances in the live pools. Casual observations of filamentous algae, another food source for amphipods, imply that the mesh screens covering both pool types were equally inhabited.

Although we retrieved the exact same number of snails as initially added to the pools, this was due to the overall count and not the same snails as there were size class differences in new recruits. We observed a 7% reduction in snail abundance in the sham pools. Total length and aperture measurements (cm) taken of snails before and after the experiment revealed that snails grew over the course of the experiment, 29% in total length 43% in aperture width for live pools and 29% in total length and 36% in aperture width for sham pools, but the difference in size was not significant.

This experiment allowed a glimpse into the complex of relationships between oyster biodeposition of feces and pseudofeces and their enhancement of trophic interactions beginning with microphytobenthos and moving up the food web to higher consumers. Our hypothesis for this experiment used only a subsample of the biodiversity seen in natural oyster reef habitats, and our design focused on a simplistic trophic cascade where white shrimp were the highest level consumers. Though trophic interactions from altered food webs may not provide the best indication of ecosystem structure (Heck and Valentine 2007), we still observed the importance of oyster biodeposition on the small scale of our mesocosms.

Though our experiment did not include higher-level consumers such as small fishes and crabs, a repeated mesocosm experiment that includes these consumers may ultimately reflect the enhancement of trophic web scenarios through bivalve biodeposits.

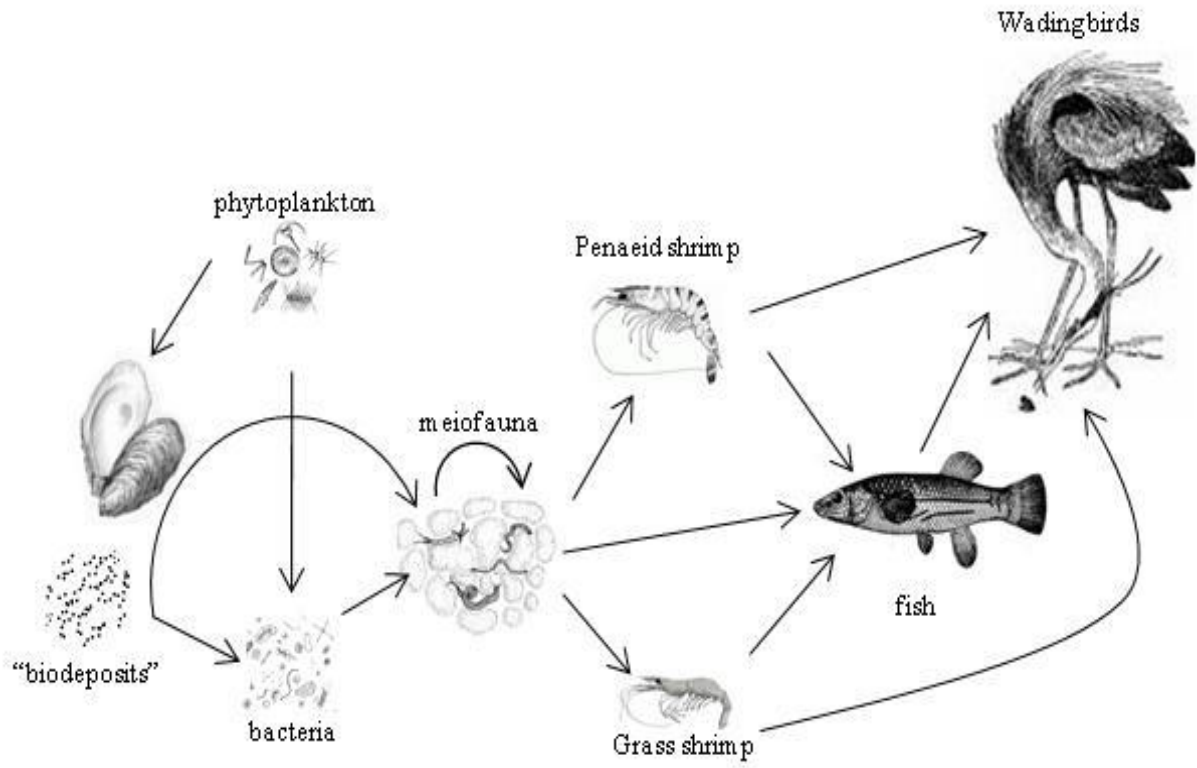


Figure 16. Probable estuarine food web based on oyster biodeposition.



Figure 17. Mesocosm array of 12 plastic pools receiving unfiltered seawater and covered with mesh screen.

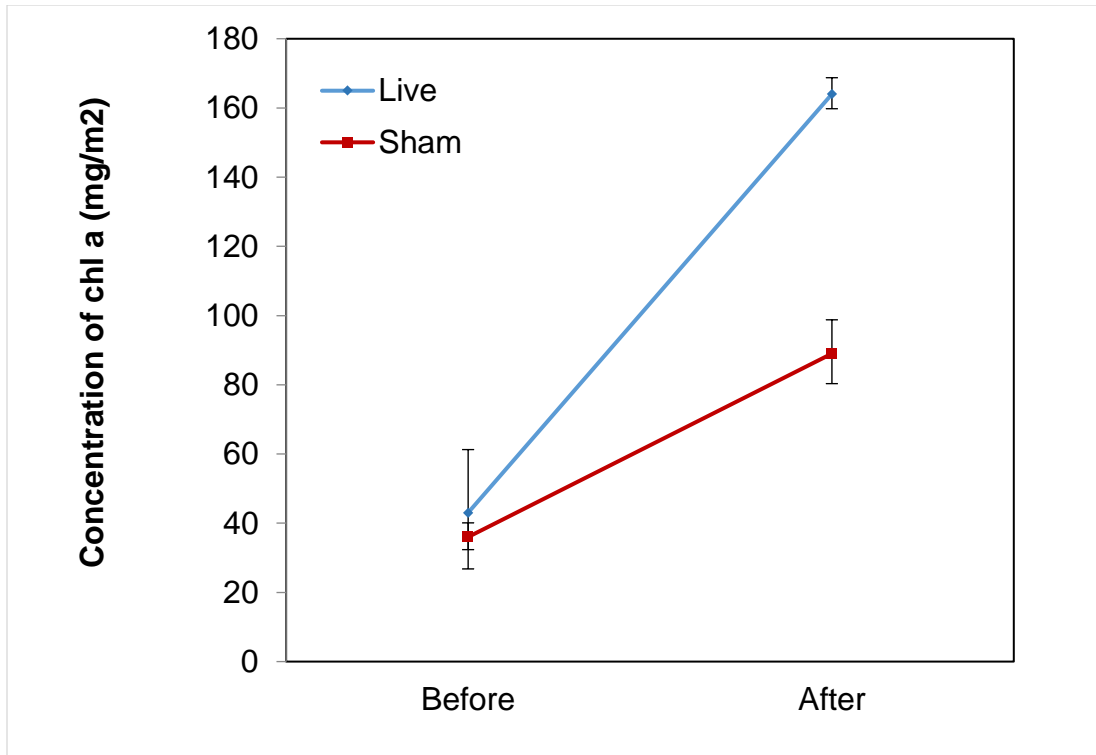


Figure 18. Benthic chlorophyll a concentrations (mg/m^2) of live and sham pools before and after the experiment. A 2-way full factorial ANOVA of oyster treatment versus time with mesocosms nested within the interaction showed significant differences between time ($p < 0.001$), treatment ($p < 0.0032$), and the treatment X time interaction ($p < 0.0518$). Data were transformed using the Box-Cox transformation to reduce heterogeneity of the residuals. Error bars are back transformed Standard Error.

Table 4. Nested ANOVA for chlorophyll- *a* concentrations (mg/m²) of mesocosms.

| | SS | DF | F Ratio | P |
|----------------------|-----------|-----------|----------------|-------------------|
| Time | 73912.9 | 1 | 84.9793 | <0.0001 |
| Treatment | 9693.78 | 1 | 11.1452 | 0.0032 |
| Treatment * Time | 37.09.68 | 1 | 4.2651 | 0.0518 |
| Pool [time, oysters] | 17403.7 | 20 | 1.0451 | 0.4341 |

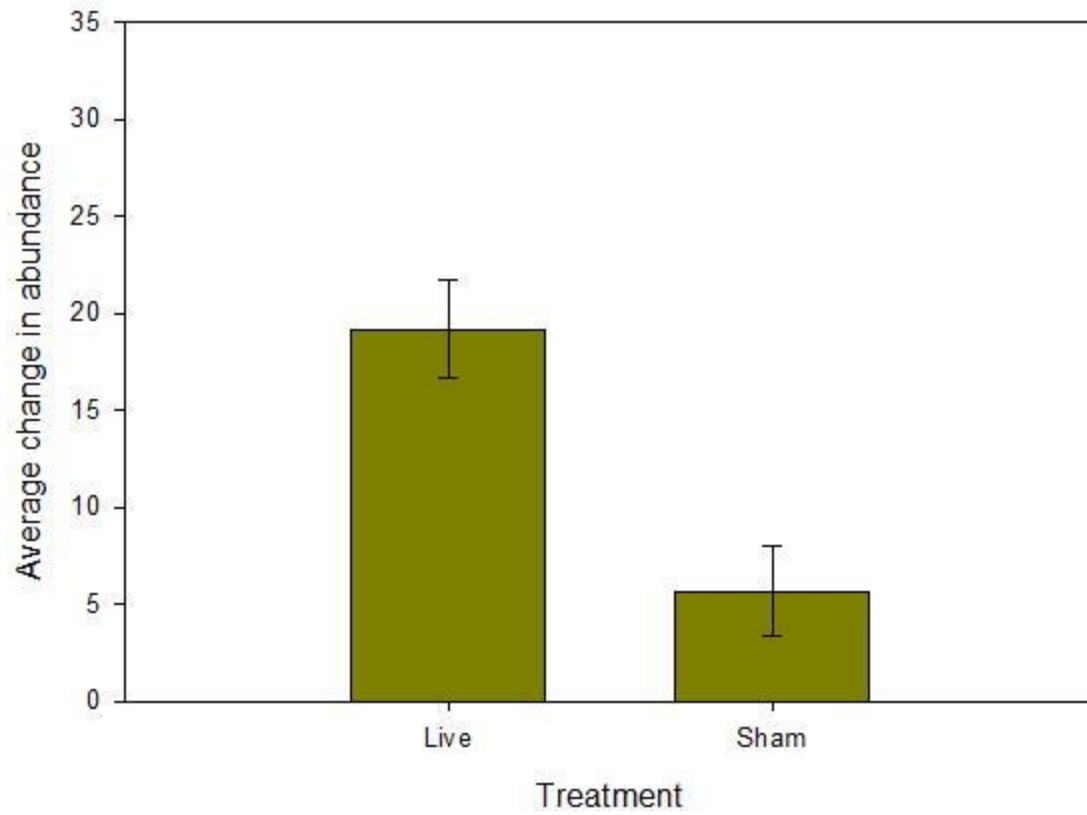


Figure 19. Difference in oyster spat abundance between live and sham pools. A t-test revealed a significantly ($p < 0.003$) higher number of oyster recruits settling on living oysters than sham shells.

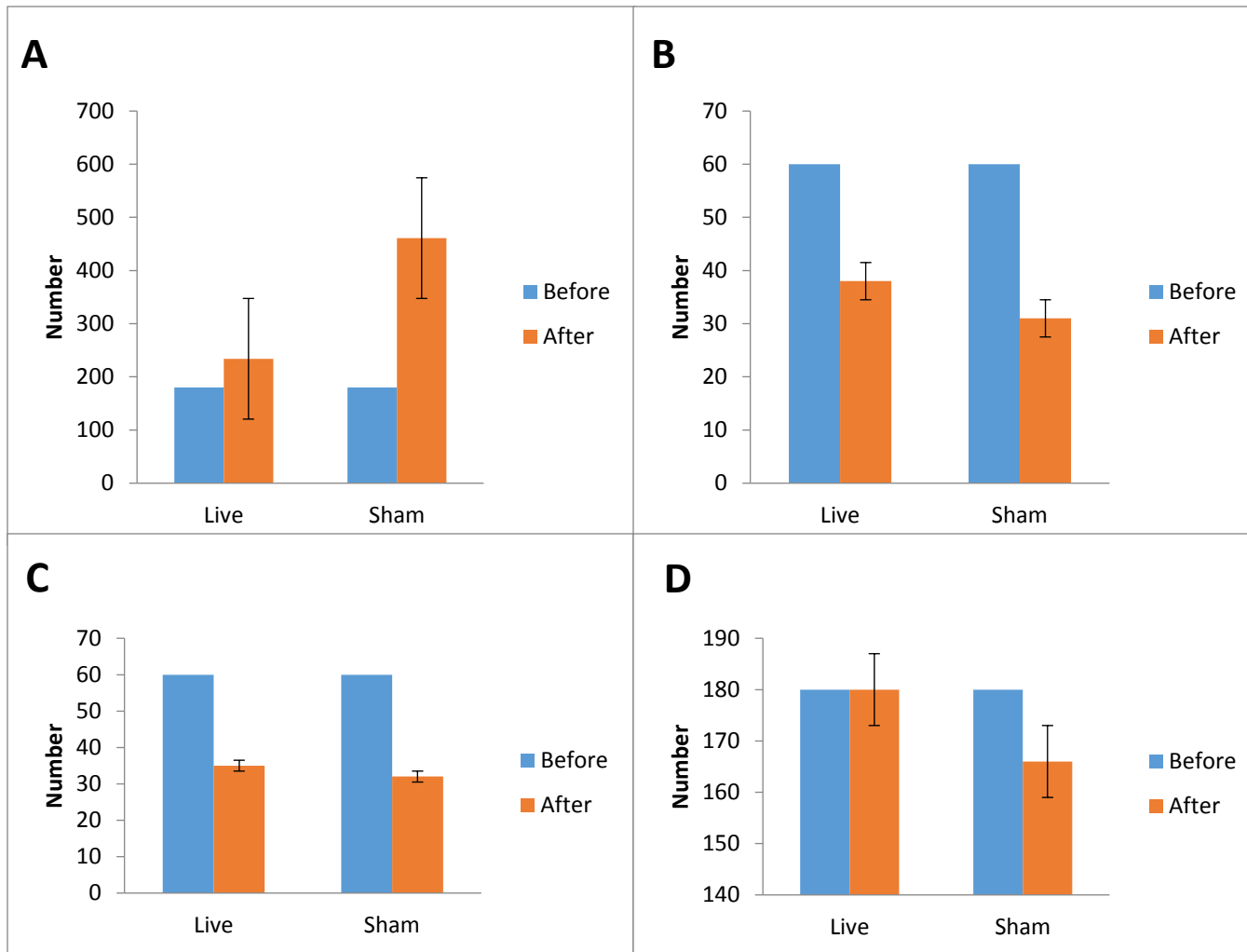


Figure 20. Change in abundance for amphipods (A), grass shrimp (B), white shrimp (C) and mud snails (D) before and after experiment in live and sham pools. Mean \pm SE are shown. Mud snail abundance was marginally significant ($p < 0.051$) in the live pools after the experiment.

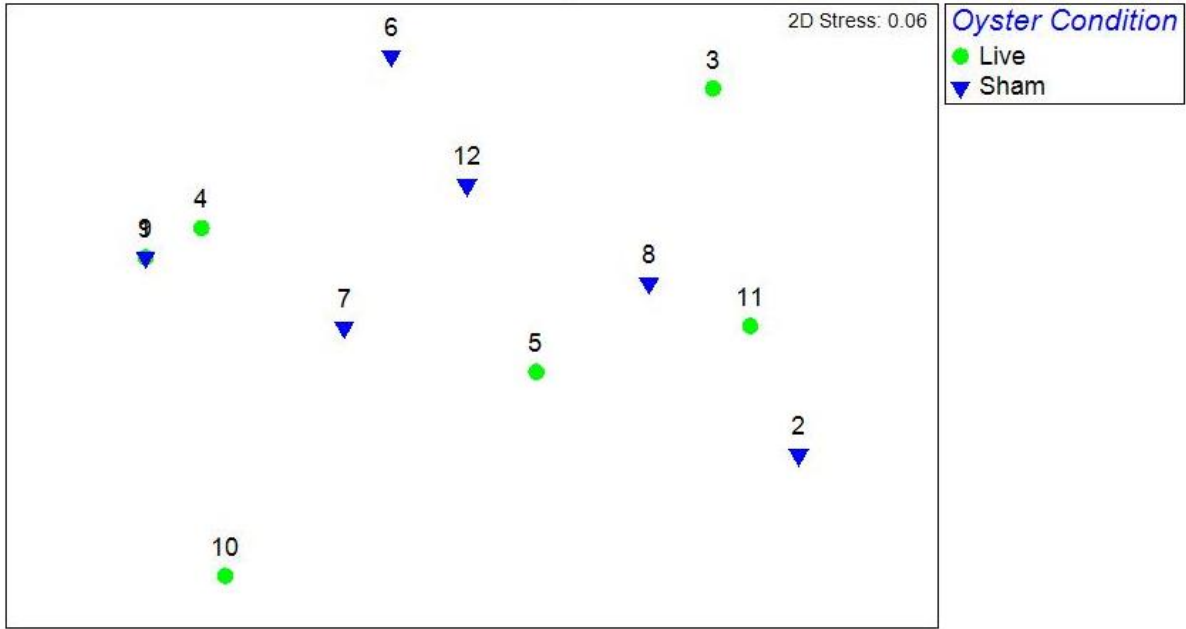


Figure 21. Non-metric multidimensional plot of meiofauna. Numbers represent live and sham pool assignments. No significant differences were observed of meiofaunal assemblages between live and sham pools.

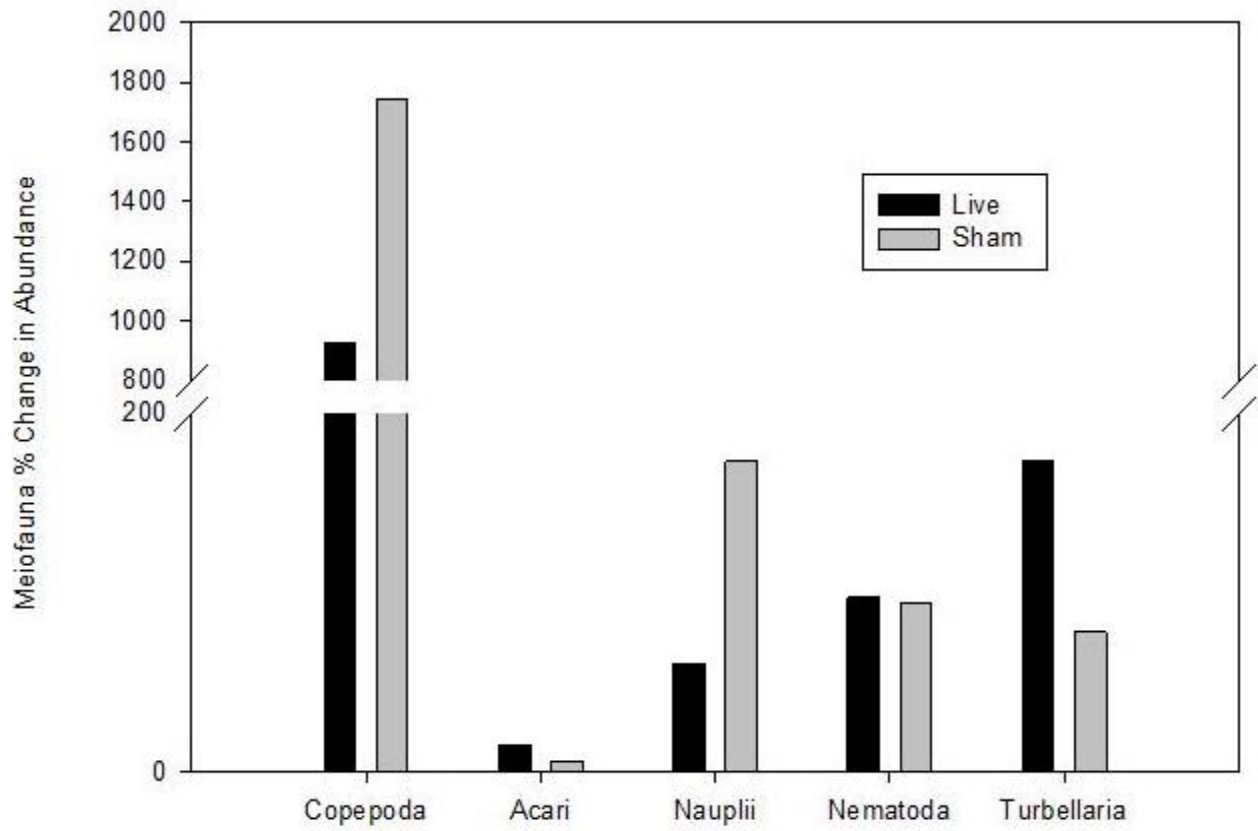


Figure 22. Percent change in abundance of meiofauna between live and sham pools.

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APPENDIX

| Fyke | | CPUE | | | | | |
|-----------------------|---------------------------------|-------------|-----------------|-----------------|-------------------|---------------|------------------|
| | | SITE | | | | | |
| | | No marsh | Narrow Marsh | Medium Marsh | Medium Marsh 2 | Wide Marsh | Natural Marsh |
| Crustacea | | | | | | | |
| grass shrimp | <i>Hippolyte species</i> | 13.17 | 33.50 | 42.83 | 53.33 | 89.00 | 82.17 |
| blue crab | <i>Callinectes sapidus</i> | 13.83 | 28.83 | 15.17 | 26.17 | 36.17 | 27.17 |
| shrimp | <i>Penaeus spp.</i> | 4.67 | 12.00 | 9.50 | 16.17 | 17.67 | 45.17 |
| arrow shrimp | <i>Tozeuma carolinense</i> | 0.00 | 0.00 | 0.00 | 1.33 | 0.00 | 0.00 |
| mud crab | <i>Rhithropanopeus harrisi</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.50 | 0.00 |
| spider crab | <i>Libinia spp.</i> | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.17 |
| stone crab | <i>Menippe mercenaria</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| Actinopterygii | | | | | | | |
| pinfish | <i>Lagodon rhomboides</i> | 52.00 | 8.67 | 17.67 | 24.17 | 88.17 | 21.67 |
| rainwater killifish | <i>Lucania parva</i> | 1.00 | 8.50 | 4.50 | 5.33 | 10.17 | 15.50 |
| spot | <i>Leiostomus xanthurus</i> | 4.17 | 4.67 | 2.67 | 1.67 | 17.83 | 5.83 |
| sheepshead minnow | <i>Cyprinodon variegatus</i> | 0.33 | 2.67 | 0.17 | 2.17 | 0.50 | 26.33 |
| mummichog | <i>Fundulus heteroclitus</i> | 0.50 | 4.67 | 1.83 | 0.67 | 7.83 | 15.67 |
| bay anchovy | <i>Anchoa mitchilli</i> | 11.17 | 4.00 | 1.50 | 4.83 | 1.17 | 1.67 |
| striped mullet | <i>Mugil cephalus</i> | 0.33 | 0.17 | 1.00 | 1.00 | 16.33 | 3.50 |
| Atlantic silverside | <i>Menidia menidia</i> | 0.33 | 0.83 | 10.17 | 1.33 | 0.50 | 2.83 |
| silverside | <i>Atheriniformes spp.</i> | 0.83 | 1.50 | 1.83 | 0.83 | 8.67 | 1.33 |
| inland silverside | <i>Menidia beryllina</i> | 2.17 | 5.67 | 0.67 | 4.33 | 0.17 | 0.33 |
| striped killifish | <i>Fundulus majalis</i> | 0.00 | 1.33 | 5.83 | 0.33 | 1.17 | 4.17 |
| pigfish | <i>Orthopristis chrysoptera</i> | 0.67 | 5.67 | 0.50 | 1.00 | 3.67 | 0.67 |
| spotfin mojarra | <i>Eucinostomus argenteus</i> | 0.17 | 1.00 | 5.33 | 2.33 | 1.33 | 1.67 |
| silver perch | <i>Bairdiella chrysoura</i> | 1.67 | 1.00 | 2.50 | 1.83 | 0.33 | 2.17 |
| mosquito fish | <i>Gambusia holbrooki</i> | 0.00 | 2.33 | 0.00 | 0.00 | 4.00 | 2.17 |
| menhaden | <i>Brevoortia smithi</i> | 0.33 | 3.67 | 0.33 | 0.33 | 1.17 | 0.00 |
| blackcheek tonguefish | <i>Symphurus plagiusa</i> | 0.00 | 0.67 | 0.83 | 1.00 | 1.17 | 0.33 |

| | | | | | | | |
|---------------------|----------------------------------|------|------|------|------|------|------|
| banded killifish | <i>Fundulus diaphanus</i> | 0.00 | 0.50 | 0.00 | 2.83 | 0.50 | 0.00 |
| Summer flounder | <i>Paralichthys dentatus</i> | 0.00 | 0.50 | 0.17 | 0.00 | 1.83 | 0.67 |
| southern flounder | <i>Paralichthys lethostigma</i> | 0.00 | 0.17 | 0.50 | 0.33 | 1.67 | 0.33 |
| killifish | <i>Cyprinodontiformes spp.</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 2.50 |
| naked goby | <i>Gobiosoma bosci</i> | 0.17 | 1.00 | 0.83 | 0.33 | 0.17 | 0.00 |
| mullet | <i>Mugil spp.</i> | 0.00 | 0.83 | 0.00 | 0.00 | 0.67 | 0.50 |
| red drum | <i>Sciaenops ocellatus</i> | 0.17 | 0.50 | 0.33 | 0.17 | 0.33 | 0.00 |
| bullhead catfish | <i>Ameiurus spp.</i> | 0.00 | 1.17 | 0.17 | 0.00 | 0.00 | 0.00 |
| white mullet | <i>Mugil curema</i> | 0.00 | 0.50 | 0.33 | 0.33 | 0.00 | 0.00 |
| inshore lizardfish | <i>Synodus foetens</i> | 0.17 | 0.17 | 0.00 | 0.50 | 0.00 | 0.17 |
| spotted seatrout | <i>Cynoscion nebulosus</i> | 0.00 | 0.00 | 0.67 | 0.00 | 0.17 | 0.17 |
| black drum | <i>Pogonias cromis</i> | 0.00 | 0.17 | 0.17 | 0.50 | 0.00 | 0.00 |
| needlefish | <i>Strongylura marina</i> | 0.00 | 0.00 | 0.50 | 0.33 | 0.00 | 0.00 |
| tongue sole | <i>Symphurus spp.</i> | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 | 0.67 |
| Atlantic needlefish | <i>Strongylura marina</i> | 0.00 | 0.00 | 0.17 | 0.33 | 0.00 | 0.17 |
| croaker | <i>Micropogonias undulatus</i> | 0.17 | 0.17 | 0.00 | 0.00 | 0.17 | 0.17 |
| halfbeak | <i>Hyporhamphus unifasciatus</i> | 0.00 | 0.00 | 0.33 | 0.33 | 0.00 | 0.00 |
| northern searobin | <i>Prionotus carolinus</i> | 0.00 | 0.00 | 0.17 | 0.00 | 0.33 | 0.00 |
| oyster toadfish | <i>Opsanus tau</i> | 0.17 | 0.17 | 0.00 | 0.17 | 0.00 | 0.00 |
| pistol shrimp | <i>Alpheidae spp.</i> | 0.17 | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 |
| white perch | <i>Morone americana</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.17 |
| American eel | <i>Anguilla rostrata</i> | 0.17 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| flounder | <i>Paralichthys spp.</i> | 0.00 | 0.00 | 0.33 | 0.00 | 0.00 | 0.00 |
| houndfish | <i>Tylosurus crocodilus</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 |
| northern sennet | <i>Sphyaena borealis</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.17 | 0.00 |
| pygmy filefish | <i>Monacanthus setifer</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 |
| anchovy | <i>Anchoa spp.</i> | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| Atlantic stingray | <i>Dasyatis sabina</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| barracuda | <i>Sphyaena spp.</i> | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| bay whiff | <i>Citharichthys spilopterus</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| common carp | <i>Cyprinus carpio</i> | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |

| | | | | | | | |
|----------------------------|-------------------------------|------|------|------|------|------|------|
| fourspot flounder | <i>Paralichthys oblongus</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| longnose gar | <i>Lepisosteus osseus</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 |
| lookdown | <i>Selene vomer</i> | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 | 0.00 |
| pipefish | <i>Syngnathus species</i> | 0.00 | 0.00 | 0.00 | 0.00 | 0.17 | 0.00 |
| speckled worm eel | <i>Myrophis punctatus</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| spotted whiff | <i>Citharichthys macrops</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| striped anchovy | <i>Anchoa hepsetus</i> | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| three spine stickleback | <i>Gasterosteus aculeatus</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| yellow perch | <i>Perca flavescens</i> | 0.00 | 0.17 | 0.00 | 0.00 | 0.00 | 0.00 |
| Other squid | <i>Lolliguncula brevis</i> | 0.33 | 0.33 | 0.00 | 0.67 | 0.83 | 0.83 |

Gill

CPUE

SITE

| | | No marsh | Narrow Marsh | Medium Marsh | Medium Marsh 2 | Wide Marsh | Natural Marsh |
|-----------------------|---------------------------------|-------------|-----------------|-----------------|-------------------|---------------|------------------|
| Crustacea | | | | | | | |
| blue crab | <i>Callinectes sapidus</i> | 15.5 | 29.5 | 20.0 | 9.0 | 25.0 | 43.5 |
| grass shrimp | <i>Hippolyte species</i> | 0.0 | 17.0 | 1.5 | 0.0 | 0.0 | 0.0 |
| stone crab | <i>Menippe mercenaria</i> | 0.5 | 1.0 | 0.0 | 0.0 | 0.0 | 0.5 |
| mud crab | <i>Rhithropanopeus harrisii</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| shrimp | <i>Penaeus spp.</i> | 0.5 | 2.0 | 4.0 | 0.0 | 0.5 | 1.0 |
| pistol shrimp | <i>Alpheidae spp.</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| spider crab | <i>Libinia spp.</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Actinopterygii | | | | | | | |
| spot | <i>Leiostomus xanthurus</i> | 8.5 | 17.5 | 14.0 | 11.5 | 10.0 | 10.0 |
| striped mullet | <i>Mugil cephalus</i> | 17.0 | 5.0 | 11.5 | 13.5 | 7.5 | 7.0 |
| menhaden | <i>Brevoortia smithi</i> | 6.0 | 9.5 | 2.0 | 1.0 | 7.5 | 4.5 |
| silver perch | <i>Bairdiella chrysoura</i> | 3.5 | 3.0 | 3.0 | 1.5 | 4.5 | 9.0 |
| pinfish | <i>Lagodon rhomboides</i> | 4.0 | 5.5 | 3.5 | 1.5 | 5.0 | 4.5 |
| southern flounder | <i>Paralichthys lethostigma</i> | 1.0 | 3.5 | 3.5 | 2.0 | 3.5 | 2.5 |
| white perch | <i>Morone americana</i> | 2.0 | 1.5 | 3.5 | 1.0 | 2.5 | 2.5 |

| | | | | | | | |
|---|--|-----|-----|-----|-----|-----|-----|
| pigfish | <i>Orthopristis chrysoptera</i> | 1.5 | 1.0 | 0.5 | 1.0 | 2.5 | 1.0 |
| red drum | <i>Sciaenops ocellatus</i> | 1.0 | 2.5 | 1.5 | 0.5 | 0.5 | 1.5 |
| smooth butterfly ray | <i>Gymnura micrura</i> | 1.0 | 0.5 | 1.0 | 0.5 | 2.5 | 1.5 |
| Summer flounder blackcheek tonguefish | <i>Paralichthys dentatus</i> | 0.0 | 0.5 | 1.5 | 1.5 | 2.0 | 1.0 |
| bluefish | <i>Symphurus plagiusa</i> | 0.0 | 5.5 | 0.0 | 0.0 | 0.0 | 0.0 |
| spotfin mojarra | <i>Pomatomus saltatrix</i> | 0.5 | 1.5 | 0.5 | 0.5 | 1.0 | 1.5 |
| thread herring | <i>Eucinostomus argenteus</i> | 0.0 | 3.0 | 1.0 | 1.0 | 0.0 | 0.5 |
| croaker | <i>Opisthonema oglinum</i> | 5.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| common carp | <i>Micropogonias undulatus</i> | 0.5 | 1.0 | 0.5 | 1.0 | 0.5 | 0.5 |
| mummichog | <i>Cyprinus carpio</i> | 0.5 | 3.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| gizzard shad | <i>Fundulus heteroclitus</i> | 0.5 | 0.5 | 1.0 | 0.0 | 0.0 | 1.5 |
| Atlantic needlefish | <i>Dorosoma cepedianum</i> | 0.5 | 0.0 | 1.5 | 0.5 | 0.0 | 0.5 |
| bluntnose ray | <i>Strongylura marina</i> | 0.0 | 0.5 | 0.0 | 0.0 | 0.0 | 2.0 |
| inshore lizardfish | <i>Dasyatis sayi</i> | 0.0 | 0.0 | 1.0 | 0.0 | 0.5 | 1.0 |
| Atlantic silverside | <i>Synodus foetens</i> | 0.5 | 0.0 | 0.0 | 0.5 | 0.5 | 1.0 |
| largemouth bass | <i>Menidia menidia</i> | 0.0 | 0.0 | 2.0 | 0.0 | 0.0 | 0.0 |
| longnose gar | <i>Micropterus salmoides</i> | 0.0 | 2.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| sheepshead | <i>Lepisosteus osseus</i> | 0.0 | 0.0 | 1.0 | 0.5 | 0.0 | 0.5 |
| Atlantic stingray | <i>Archosargus probatocephalus</i> | 0.5 | 0.5 | 0.5 | 0.0 | 0.5 | 0.0 |
| black drum | <i>Dasyatis sabina</i> | 1.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 |
| needlefish | <i>Pogonias cromis</i> | 0.5 | 0.5 | 0.0 | 0.0 | 0.5 | 0.0 |
| permit | <i>Strongylura marina</i> | 0.5 | 0.5 | 0.0 | 0.5 | 0.0 | 0.0 |
| smooth dogfish | <i>Trachinotus falcatus</i> | 0.0 | 1.0 | 0.0 | 0.0 | 0.5 | 0.0 |
| spotted seatrout | <i>Mustelus canis</i> | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 | 0.5 |
| bullhead catfish | <i>Cynoscion nebulosus</i> | 0.5 | 0.5 | 0.0 | 0.0 | 0.0 | 0.5 |
| cobia | <i>Ameiurus spp.</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 |
| striped killifish | <i>Rachycentron canadum</i> | 0.5 | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 |
| anchovy | <i>Fundulus majalis</i> | 0.0 | 1.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| | <i>Anchoa spp.</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 |

| | | | | | | | |
|---------------------|--------------------------------|-----|-----|-----|-----|-----|-----|
| bigeye anchovy | <i>Anchoa lamprotaenia</i> | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 |
| bighead searobin | <i>Prionotus tribulus</i> | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 | 0.0 |
| bowfin | <i>Amia calva</i> | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 | 0.0 |
| hardhead catfish | <i>Arius felis</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 |
| mosquito fish | <i>Gambusia holbrooki</i> | 0.5 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| pygmy filefish | <i>Monacanthus setifer</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 |
| rainwater killifish | <i>Lucania parva</i> | 0.5 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| southern kingfish | <i>Menticirrhus americanus</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 |
| spadefish | <i>Chaetodipterus faber</i> | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 |
| tongue sole | <i>Symphurus spp.</i> | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 |
| Other | | | | | | | |
| comb jelly | <i>Beroe spp.</i> | 5.5 | 0.0 | 0.5 | 0.0 | 0.0 | 1.0 |
| green sea turtle | <i>Chelonia mydas</i> | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 |

Nekton species and abundance by CPUE observed during gill and fyke net collections from each site during the 2009-2010 sampling period. No marsh sites have a bulkhead only and no marsh plants present (0 m), Narrow marsh sites are bulkheaded with marsh widths ranging from 3.5-4.3 m, medium marsh sites are bulkheaded with marsh widths ranging from 10.7-16 m, wide marsh sites are bulkhead with marsh widths ranging from 18.3-22.8 m, and natural marsh sites do not have a bulkhead present and marsh widths ranging from 15.0-39.1 m.