Florida International University Research Summary
2011-2012 funding period
Dr. Craig Layman – November 11, 2012

This document summarizes the recent research conducted by Florida International University on the Loxahatchee River, focusing on specific studies that were carried out during 2011-2012. Since 2005, my lab at FIU has maintained an active and diverse research program on the river. We have been highly adaptive, taking advantage of emerging research opportunities (e.g., the sudden appearance of lionfish and the construction of a large-scale oyster restoration reef). While most of our work focuses the role that human disturbance plays in estuarine ecosystems, we are more broadly interested in understanding how various ecological interactions (both natural and human driven) affect overall estuarine health. Many of our findings have the potential to inform adaptive management decisions on the river. The specific areas of research discussed in this report include: 1) Community ecology of natural and restored oyster reefs, 2) Estimating the filtration capacity of dock piling fouling communities, 3) Tracking the lionfish invasion, and 4) Other fish ecology studies.

1. Oyster reef studies in the Loxahatchee River

Over the past century, oyster reefs throughout North America have experienced significant declines as a result of overharvest, degraded water quality, altered salinity patterns, and disease. As the ecological and economic importance of oyster reefs has become more widely recognized, habitat restoration is increasingly being used to combat these declines. While some oyster reef restorations are designed specifically to increase oyster production for commercial purposes, a more common goal is to restore multiple ecosystem services associated with intact natural oyster reef communities. As a result, the restoration of living oyster reefs has the potential to enhance populations of many organisms, including commercially and recreationally valuable species (Peterson et al., 2003).

Establishing baseline values representative of healthy oyster reef communities, accounting for both temporal and spatial variability, is critical to success of oyster reef restoration efforts. The success of an oyster reef restoration should be measured not only by the recovery of a population of living oysters, but also by the reestablishment of fauna that inhabit reefs, and the associated function of this reef community (Coen and Luckenbach, 2000). Where long-term data sets are lacking for natural oyster reef communities, it can be difficult to determine an appropriate restoration target. Additionally, baseline values facilitate comparisons between natural and human-made reefs over time, making it possible to assess the overall success of a restoration project while tracking potential convergence between natural and restored communities.

The initial goal of our oyster reef research was to utilize a long-term dataset to characterize the structure of oyster reef faunal communities (e.g., small benthic crustaceans, mollusks, and demersal fishes) in the Loxahatchee River. Specifically, we identified spatial (i.e., upstream-to-downstream) and temporal (i.e., wet season vs. dry season) patterns in biomass, abundance, and community composition of organisms from natural oyster reefs, creating baselines for comparison between natural and human-made reefs (*Section 1A*). We then used these baseline values to assess and track the development of oyster reef communities at a 2.36 hectare (5.84 acre) restoration reef over time, in order to determine if (and how long it takes for) a restored reef to resemble a nearby natural reef (*Section 1B*). Additionally, we tested the hypothesis that increased habitat complexity (i.e., greater vertical relief) in a restored reef would lead to increased biomass of benthic organisms and a more rapid convergence with a natural oyster reef community (*Section 1C*).

1A. Long-term Oyster Reef Monitoring – Establishing Baselines to Track Oyster Reef Health and Assess the Success of Restoration Efforts

Understanding how natural oyster reefs in the Loxahatchee River function over time has important management and conservation implications. Continuous long-term monitoring allows for early detection of potentially harmful changes in the ecologically and economically valuable oyster reef communities in the river, allowing for a proactive response in the event of natural or anthropogenic disturbance. For this portion of our research, we identified long-term patterns of community dynamics for oyster reef-associated organisms on natural oyster reefs at multiple locations within the river. We were particularly interested in understanding how communities of reef-associated organisms varied spatially (from upstream to downstream) as well as temporally (from season to season, and year to year).

Since May 2007, we have conducted bimonthly sampling of oyster reef-associated organisms at three natural oyster reef sites in the Loxahatchee River (Figure 1). These sites were located between 6.5 and 9.5 km upstream from the ocean, spanning the entire upstream-to-downstream range of present-day oyster reef development in the river. To sample benthic invertebrates and demersal fishes, we deployed four replicate benthic sampling tray traps at ~2-10 m intervals (based on size of reef) at each site. These sampling units (Figure 2) are 64 x 52 x 10 cm plastic bakery trays with polyethylene mesh shade cloth securely attached to the tray bottom (Rodney and Paynter, 2006). Each tray trap was filled with 19 l of clean, dry oyster shell, obtained from local restaurants. We placed the tray traps into shallow depressions that were excavated into the natural oyster reef substrate at each site, such that the top surface of the material in the tray trap was flush with the surrounding live oyster matrix. This allowed organisms to move laterally across the benthos and into the tray traps. To collect organisms, traps were lifted vertically, allowing water to run through the mesh cloth on the tray bottom, trapping benthic invertebrates and small demersal fishes within the tray. All fishes and invertebrates were collected by hand, kept on ice in the field, and returned to the laboratory for later processing (identification to lowest possible taxonomic level, counting, measuring wet mass). After the trays were sampled, they were refilled with shell and returned to their original location in the oyster reef. The organisms collected in these traps were used to characterize seasonal (wet season vs. dry season) and spatial (upstream vs. downstream) patterns in oyster reef-associated organisms,

Figure 1. The three long-term natural oyster reef monitoring sites where samples have been collected bimonthly since March 2007. BS = Boy Scout Camp, OI = Oyster Island, and SD = Seventh Dock. REST is the location of site #14 of the NOAA/Martin County/LRD restoration.

Figure 2. A benthic sampling tray filled with oyster shell (left) and a deployed tray (visible at low tide) located at one of the river's natural oyster reefs (right).

providing baseline values for natural oyster reef communities in the river. This benthic tray trap methodology was used for all of our oyster reef community analyses (See below).

Between May 2007 and May 2012, we collected and identified nearly 27,000 individual organisms representing 11 fish and 20 invertebrate taxa from natural oyster reefs in the Loxahatchee River. The most dominant organisms in these natural oyster reef communities were mud crabs, gobies, snapping shrimp, and porcelain crabs (Table 1). Our updated data set further reinforces the importance of seasonal and spatial patterns in structuring the overall biomass of oyster reef-associated communities in the river. Although there was some year-to-year and site-to-site variability, we found that biomass values on natural oyster reefs in the Loxahatchee River were typically greatest during our May or July sampling dates (Figure 3). This time frame corresponds to the end of the dry season or the beginning of the wet season. Annual minimum biomass values were less consistent among years, but usually occurred some time between November and March. When averaged across all three natural reef sites over the course of the five-year study, mean biomass values peaked in July and were lowest in November, with greater variability in biomass values occurring during the wet season than the dry season. We have speculated that these observed fluctuations in biomass were likely triggered by changes in salinity caused by varying levels of freshwater inflow. However, it is also possible that we are simply observing seasonal shifts that might be caused by changes in day length, temperature, or some other environmental parameters, and that the changes in biomass are correlated with, but not directly caused by, changes in freshwater inflow. A manuscript in preparation explicitly quantifies and explores these two mechanisms that may drive observed patterns. This manuscript will be part of the 2012-2013 report to LRD.

Table 1. Ten most common taxa (based on biomass) across five years of sampling on natural oyster reefs in the Loxahatchee River.

Species	**Common Name**	**% of total biomass**
Panopeus herbstii	black-fingered mud crab	24.5
Eurypanopeus depressus	depressed mud crab	16.5
Lophogobius cyprinoides	crested goby	15.8
Eurypanopeus spp.	xanthid crab <9 mm	13.3
Alpheus spp.	snapping shrimp	8.9
Petrolisthes armatus	green porcelain crab	7.6
Bathygobius soporator	frillfin goby	5.4
Nassarius sp.	nassa snail	2.2
Lupinoblennius nicholsi	highfin blenny	1.4
Panopeidae spp.	unidentified xanthid crab	1.1

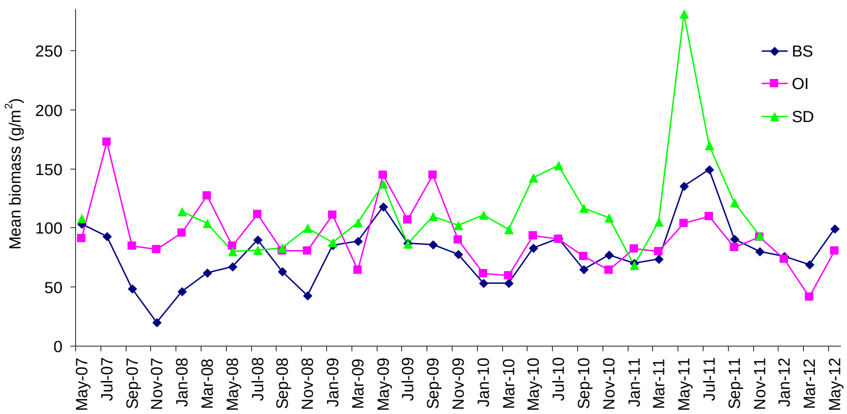

Figure 3. Seasonal biomass patterns for three natural oyster reef sites over the course of the five-year study. Although there is some variability in timing and magnitude from year-to-year, biomass typically peaks in May or July, with annual minima occurring between November and March.

In additional to seasonal variability, long-term mean biomass of oyster-reef associated organisms in the Loxahatchee River showed considerable spatial variability, with values increasing along an upstream-to-downstream gradient (Figure 4). Lowest long-term mean biomass values (averaged across all months) were recorded at the upstream natural oyster reef site, Boy Scout Camp (78.7 g/m^2) (Figure 5). This site represents the upstream limit of oyster reef growth in the system. Highest long-term mean biomass values were recorded 3 km further

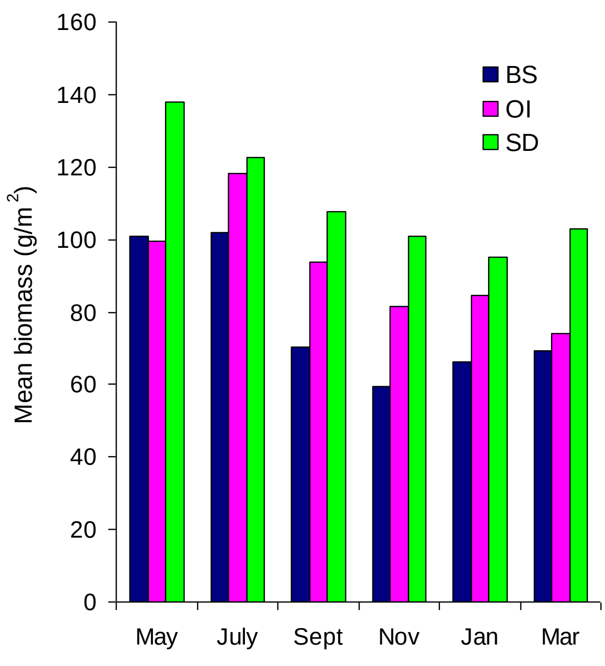

Figure 4. Spatial and temporal patterns in natural oyster reef biomass, averaged across 5 years. Boy Scout Camp (BS) is the upstream study site, and Seventh Dock (SD) is the downstream study site. Oyster Island (OI) is located between the other two sites, and is closest to Restoration Site

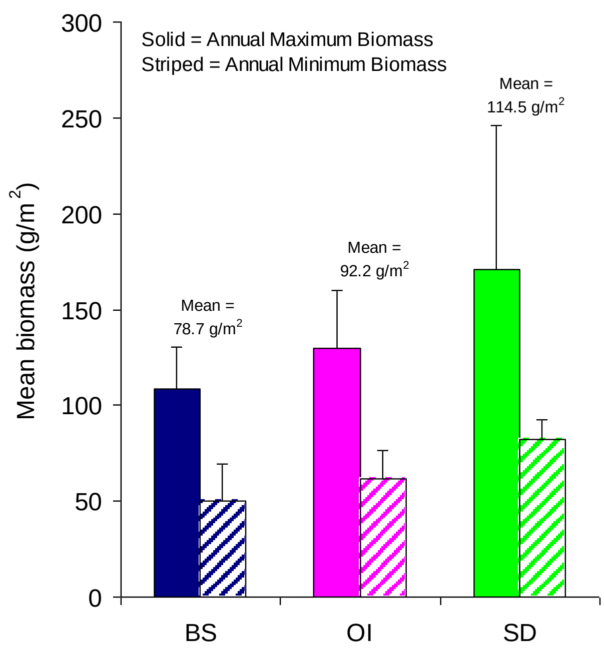

Figure 5. Maximum (solid) and minimum (striped) annual biomass for all three natural oyster reef sites (BS – Boy Scout Camp, OI – Oyster Island, SD – Seventh Dock) averaged over 5 years. Biomass increased along an upstream-to-downstream gradient (BS to SD). At all three sites, annual maximum biomass (near the end of the dry season) was 2.1 times greater than annual minimum biomass (during the wet season).

downstream at Seventh Dock (114.5 g/m^2). Intermediate biomass values occurred at Oyster Island (92.2 g/m^2), which was located between the other two natural sites, and was closest to the oyster restoration reef discussed below. This upstream-to-downstream pattern was present during all sampling months. When spatial and seasonal patterns were examined simultaneously (averaged across five years), it became apparent that the shift between maximum and minimum seasonal biomass values occurred more rapidly at the upstream site than at the downstream site (Figure 4). At Boy Scout Camp (upstream), biomass typically peaked in July and reached annual minimum values four months later in November, while at Seventh Dock (downstream), biomass declined over a seven month period, with peak biomass occurring earlier (May) and minimum values occurring later (January). It is possible that the more gradual shifts in biomass that we observed between sampling dates at the downstream site were due to the fact that this site is somewhat buffered from environmental variability (e.g., salinity, temperature) relative to the upstream site, owing to its proximity to the ocean.

To minimize the effects of interannual variability in the timing of biomass highpoints and lowpoints, we calculated the typical annual maximum biomass and the typical annual minimum biomass for each site. Over the course of the five-year study, average maximum annual biomass (at the end of the dry season, or early wet season) ranged from 108.3 g/m^2 at the upstream site (Boy Scout Camp) to 171.1 g/m^2 at the downstream site (Seventh Dock) (Figure 5). Average minimum biomass values recorded during the winter wet season ranged from 49.8 g/m^2 at the upstream site to 81.9 g/m^2 at the downstream site. At all three sites, maximum biomass values were ~2.1 times greater than minimum biomass values. These maximum and minimum biomass values will help us compare overall biomass at restored oyster reefs to natural oyster reefs, taking into account interannual variability in the timing of peaks and troughs. The full five years of natural oyster reef community data were then used to create a multivariate ordination to allow us to track community-level changes at the restored reef over time (see Restoration section below).

1B. Oyster Reef Restoration – Tracking the Success of the Loxahatchee River's Oyster Restoration Reefs

In the Loxahatchee River, oyster reefs have been significantly degraded, largely as a result of anthropogenic alteration of freshwater inflow and associated salinity changes. Freshwater flow into the estuary has decreased over time due to flood control measures, while marine contributions increased following the widening and stabilization of Jupiter Inlet in the 1940's, resulting in a shift in the optimal salinity zone for oysters from its historical location. Present-day optimal salinity levels are found several kilometers upriver from optimal larval settlement habitats (i.e., remnants of historical oyster reefs in the lower estuary, ~1.5 km from the ocean, where salinities are now too high for oyster reef development), in an area that is substrate limited. Construction of a restoration reef in this substrate-limited part of the river would provide carbonate substrate for settlement and growth of living oysters. In addition to colonization by oysters, the reef would facilitate the recruitment of numerous other oyster reef-associated organisms, leading to an eventual transformation into something functionally analogous to a natural oyster reef. An important component of restoration efforts is selecting an appropriate ecological endpoint or target, and assessing if and when plant and animal communities that develop as a result of the restoration reach that endpoint. For this portion of our oyster reef research, we used long-term baseline values that we obtained from natural reefs (see above) to assess the success of a large-scale oyster reef restoration project in the Loxahatchee River.

In July 2010, 2.36 hectares (5.84 acres) of oyster restoration reef were constructed in the Loxahatchee River as part of the NOAA-funded Martin County Oyster Reef Restoration Project, in partnership with the Loxahatchee River District (referred to as the NOAA Restoration hereafter). Because of the large scale of the reef construction project, barges and excavators were used to spread a continuous 15 cm (6") layer of limestone and sandstone rock and mollusk shells (5-20 cm in diameter, obtained as a byproduct of a beach nourishment project) across the river bottom. Prior to restoration, the benthos in this section of the estuary consisted of sand and coarse sediment. The reef was constructed in three sections (1.93 hectares, 0.38 hectares, and 0.05 hectares). Our research focused on the largest of these reefs, NOAA Restoration Site 14, which was located immediately adjacent to the Oyster Island natural oyster reef site (Figure 1).

In January 2010, 6 months prior to the start of oyster reef construction, we added four new benthic tray units to the future site of the restoration reef. Holes were dug into the sand/coarse sediment substrate (at ~4 m intervals), and the tray traps were filled with 19 l of the excavated substrate (rather than oyster shell, as described above). Trays were then placed into the resulting holes, flush with the surrounding river bottom. By initiating sampling 6 months prior to reef construction, we hoped to establish a pre-construction baseline for benthic community structure at the site. Following reef construction in July 2010, the four tray traps were redeployed at the site. Each tray trap was filled with 19 l of the loose limestone/sandstone rock and mollusk shell aggregate that was used to build the reef. For the remainder of the study, these trays were sampled at the same bimonthly frequency (using the same methodology) as the natural reef monitoring trays. As we began to detect differences in biomass values between high- and low-relief sections of the oyster reef (see below), we added four additional trays to a high-relief section of the reef. These high-relief tray traps were subsequently added to our bimonthly sampling protocol. By maintaining both high- and low-relief tray traps as part of our bimonthly monitoring program, we will be better able to track the effects of vertical relief (i.e., reef thickness) on oyster reef community structure as the reef continues to mature.

Between March 2010 and May 2012, we collected ~4,000 organisms from our bimonthly sampling at Restoration Site 14, representing 21 invertebrate and 6 fish taxa. Prior to restoration, biomass at the future restoration site was only 9-15% of what would be expected at a natural oyster reef (based on the long-term baselines). Biomass values at the restored reef increased slowly for the first 6 months following the completion of the restoration project; however, between months 6 and 8, biomass began to increase rapidly (Figure 6). By month 20, biomass values at the restored reef had converged with natural reef values, at which point they were characterized by the predicted seasonal (wet season vs. dry season) shifts that have been observed in our long-term monitoring.

Examining biomass alone fails to account for differences in community composition that occur during the months following reef construction. By creating a non-metric multidimensional scaling (MDS) ordination, we can visualize community composition among the three natural oyster reef sites and the restoration reef site across all 31 sampling dates. MDS creates a 2-dimensional ordination that facilitates visual comparisons of communities by representing relative similarity (or dissimilarity) by the relative distance between data points. The closer two data points are in the ordination plot, the more similar the overall community structure (relative biomass of each taxon present) is between those points. Community structure of oyster reef-associated organisms varies among the three natural oyster reef sites, but is similar within each site (Figure 7). The overall natural oyster-reef associated community is represented by a single cluster of blue and green points. Within this larger cluster, each natural site separates out into a distinct sub-cluster. A 1-way analysis of similarities (ANOSIM) revealed significant differences in community structure among the three natural sites. Samples were most dissimilar between the two sites that were situated furthest apart, Boy Scout Camp and Seventh Dock. The two data points representing pre-restoration communities (red in Figure 7) were significantly different than any of the post-restoration communities, as well as all natural reef communities. This was predicted, since the restoration site was dominated by sandy and silty benthic habitats prior to restoration, and these habitats typically support different communities than structurally complex oyster reefs.

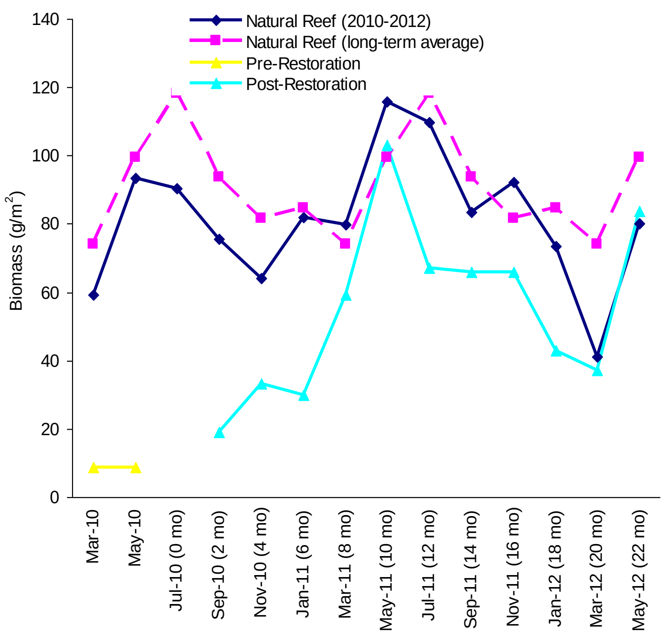

Figure 6. Plot of mean organismal biomass (g/m^2) at natural and restored oyster reef sites in the Loxahatchee River. The natural oyster reef site (Oyster Island) was located approximately 100m from the restoration reef and was used as a control to compare community structure between natural and restored reefs over time. The dark blue line represents actual biomass measurements at the natural reef site taken between March 2010 and May 2012. The dashed purple line represents biomass values at the natural site, averaged across the 5-year monitoring dataset. The gap in the restoration reef data at July 2010 represents the reef construction period

The post-restoration points in Figure 7 show a gradual convergence between restored and natural reef communities over time. Cluster analysis revealed that the communities present at the restoration site in March and May 2012 (20 and 22 months post-construction) clustered with communities at Oyster Island, the closest natural reef site to Restoration Site 14. The restoration community present on November 2011 clustered with Seventh Dock, the downstream natural site. All of the earlier restoration site communities form a single unique cluster that is distinct from the three natural sites. Based on these community-level analyses, it appears that the restored oyster reef community began to closely resemble a natural oyster reef community after ~20 months – directly paralleling the univariate biomass patterns.

The community-level changes that we observed following construction of the restoration reef were driven by several key taxa. While depressed mud crabs, small unidentified mud crabs, snapping shrimp, and juvenile naked gobies appeared at the restoration site two months after the reef was constructed, other oyster-reef associated taxa were absent from the site for considerable periods of time following restoration. From the time the reef was constructed, black-fingered mud crabs did not appear in appreciable numbers for eight months, crested gobies were completely absent until month 14, and frillfin gobies did not make their first appearance until month 18. All three of these species were present in large numbers at nearby natural oyster reefs, suggesting that the gradual development of a full natural oyster reef community at a restoration reef may be driven by a complex interaction between habitat quality, specific settlement cues, and the presence of previous plant and animal colonists. We will continue to track the changes in restoration reef community structure into the future as the reef matures.

Since our bimonthly sampling efforts were spatially confined to one section of the restoration reef, we felt that it was important to verify that our findings were representative of other sections of the reef. In April 2012, we deployed an additional 24 sampling trays at three new locations within the reef to look for spatial patterns in community composition. At each site, we deployed four trays within a high-relief plot and four trays within an adjacent low-relief plot. We sampled these trays in July 2012, at which time biomass and community composition values were compared to the long-term monitoring trays that had been deployed within the restoration reef for two years. During this one-day intensive sampling event, we collected more than 5,000 individual organisms from high-relief and low-relief sampling trays throughout Restoration Site 14. Our findings showed that for each level of reef thickness (high relief or low relief), biomass values were relatively similar across the restoration reef. This suggests that the biomass values we collected from high- and low-relief tray traps during bimonthly sampling at the restoration reef were indicative of biomass values in other sections of the reef. However, at the time of this sampling, community composition differed among sites within the restoration reef. It is possible that habitat variability (e.g., distance to mangroves, distance to channel, etc.) within the reef is driving differences in community structure.

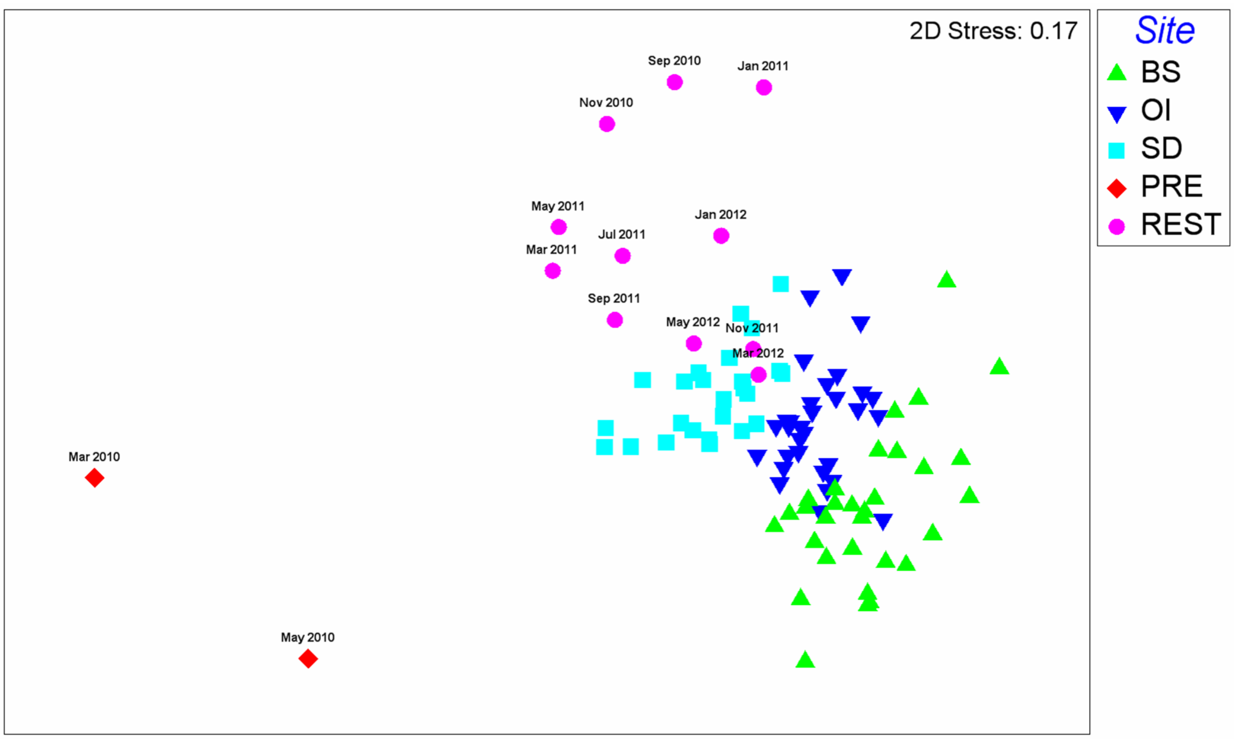

Figure 7. Multiple-dimensional scaling ordination representing differences in community structure at the restoration and three natural oyster reef sites. The two pre-restoration sampling dates (red) are clearly differentiated from the natural reef communities. Following restoration (pink), community composition becomes progressively more similar to natural reef communities over time. Each point represents a single site on one date.

1C. Testing the Effects of Habitat Complexity on Restored Oyster Reef Communities

Habitat complexity has been shown to affect species richness and community composition in a diverse range of ecosystems. To test effects of habitat complexity on community assembly in a restored oyster reef, we created experimental sites representing two levels of bottom relief within the larger restoration reef matrix at NOAA Restoration Site 14. Here, we use the term 'relief' to refer to submeter-scale changes in elevation within the reef matrix (i.e., higher "piles" of shell within the shallow ref matrix are referred to as high relief). During the construction of the reef, we used an excavator (as well as hand tools) to create three parallel high-relief plots within the restoration reef matrix. These plots were 10 m x 2 m, and 30 cm thick (the greatest height allowed by our research permit). For each high-relief plot, we created a paired low-relief plot (10 m x 4 m, 15 cm thick) in the adjacent reef matrix, utilizing the same volume of rock and shell. Since the restoration reef was constructed as a homogeneous 15 cm thick layer of rock and shell, the low-relief experimental plots served as controls for the remainder of the reef. The three paired experimental blocks were located near the center of the restoration reef, ~100 m from a mangrove shoreline to the north and 50-100 m from the primary river channel to the east. The experimental blocks were separated by ~25 m, and each high-relief and low-relief plot was surrounded by a ~1 m wide perimeter of sand. The long axis of each plot ran parallel to the direction of river/tidal flow. Since each pair of trays (high relief/low relief) within a block are <5 m apart, they are exposed to same environmental and physical conditions (e.g., current velocity and direction, distance to mangroves, etc.). To assure that the high-relief plots remained subtidal (a requirement of our permit), we constructed the experimental blocks in the deepest section of the reef. As such, all three blocks had approximately the same initial elevation. Within each high/low experimental block, 14 benthic tray traps were filled with 19 l of rock and shell and placed in rows ~1 m apart (7 trays per high-relief plot, and 7 trays per low-relief plot). A total of 42 benthic tray traps were deployed across the three experimental blocks in August 2010 (see Figure 10 below).

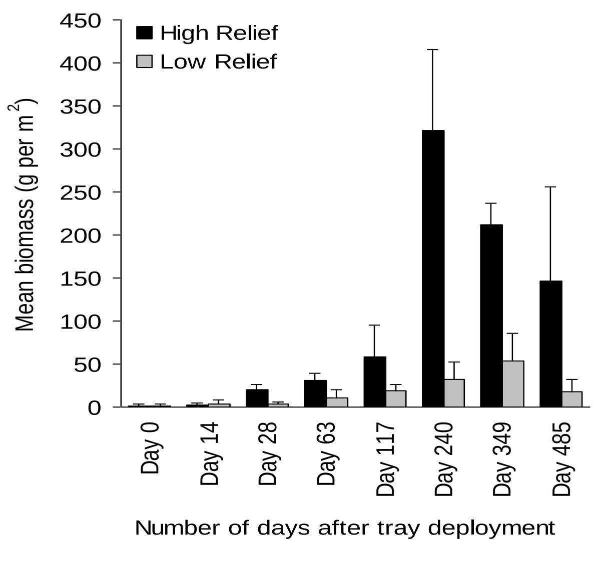

Figure 8. Changes in organismal biomass over time at high- and low-relief sections of the restoration reef. On all sampling dates, abundance was greater at high-relief experimental ridges than at adjacent low-relief plots. Biomass peaked on day 240 – April 3, 2011 (high-relief ridges) and day 349 – July 21, 2011 (low-relief plots), both near the end of the dry season. This matches the temporal pattern we have observed in our long-term monitoring dataset

Rather than sampling these trays at a fixed bimonthly time interval, we chose *a priori* to sample at approximately day 0, 14, 30, 60, 120, 240, 365, and 485, where day 0 (the start of our sampling) was one week after construction of the restoration reef. On each sampling date, one randomly selected pair of trays (high/low) was removed from each experimental block and processed (six trays per sampling date). Unlike the bimonthly monitoring trays, these were left undisturbed from the time of deployment to the time of sampling, at which point they were removed from the river. At the time of our final sampling, we counted the number of live oysters in each tray trap, and used a bent wire transect to estimate rugosity.

Between day 0 (one week after reef construction was completed) and day 485, we collected more than 3,000 organisms from the experimental high- and low-relief treatments. During the first 8 months of the study, biomass increased at both treatment levels, however the rate of increase was much greater at high-relief sites (Figure 8). Between month 8 (April) and month 12 (July), biomass values at the high-relief plots slowly began to decrease. The timing of this decrease corresponded to seasonal patterns in our long-term natural reef baselines. Low-relief plots had a similar decline in biomass, but the decrease occurred 3 months later (July) than for the high-relief plots. High-relief biomass may have begun to decrease earlier in the wet season because it takes less freshwater inflow to expose these taller reefs to the surface layer of low-salinity water. Throughout the study, biomass at high-relief plots was substantially greater than at low-relief plots. When high-relief biomass peaked on day 240, mean biomass at these plots was 10 times greater than at adjacent low-relief plots. On this sampling date, we recorded a biomass of 388 g/m^2, the highest oyster reef biomass value ever detected in the system.

Over time, the difference between high- and low-relief biomass values became smaller. By the end of the experimental high-relief/low-relief study (day 485), high-relief biomass was 8 times higher than low-relief biomass. Since we continued to sample high- and low- relief sections of the reef as part of our bimonthly protocol, we were able to track these patterns after the experimental phase of the study ended. On our most recent sampling date (May 2012), low-relief biomass (84 g/m^2) was 36% lower than high-relief biomass (130 g/m^2). However, it is important to note that these tray traps were sampled (i.e., emptied and refilled) every two months, possibly resulting in different biomass patterns than our experimental tray traps, which were left undisturbed for the duration of the study (up to 485 days for the final set of trays).

This gradual convergence between high- and low-relief biomass values provides some insight regarding the mechanisms that drive relief-based differences in biomass in the

Figure 9. Low-relief and high-relief benthic tray traps after 485 days in the water at the restoration site. In the low-relief tray (left), the original limestone rock and mollusk shell aggregate material is still visible. Below this surface layer, the tray was densely packed with fine sediment (not visible in photo). The surface of the high-relief tray (right) is almost entirely covered with living oysters,

Loxahatchee River's restored oyster reefs. Structurally complex high-relief oyster reefs are often exposed to tidal currents and wave action, particularly in intertidal or immediately subtidal settings. High-relief reefs have been found to experience increased current flow velocities and decreased sedimentation rates when compared to low relief reefs (Lenihan, 1999), both of which favor survival and growth of oysters (Schulte et al., 2009). Reduced sedimentation and compaction rates can also lead to greater rugosity and increased interstitial space in high-relief reefs, creating refuge for numerous reef-dwelling organisms. Low-relief restoration reefs often experience hypoxic conditions (Lenihan, 1999) that could potentially harm oysters and associated benthic communities. Additionally, habitat complexity can affect community composition on oyster restoration reefs as a result of altered predator-prey interactions (Grabowski et al., 2008; Grabowski and Powers, 2004; Hughes and Grabowski, 2006). In some cases, highly complex high-relief habitats provide increased shelter for prey species, reducing overall predation rates.

While many possible mechanisms could explain the differences in biomass and abundance we detected between high- and low-relief sites, our observations suggest that sedimentation and its related impact on live oyster growth may be the primary driver. By day 485, many of the low-relief tray traps were partially filled with densely packed fine sediment. Sedimentation was never observed in the high-relief tray traps, despite just a 15 cm difference in vertical relief. Oyster growth and rugosity were also greater in these sediment-free high-relief tray traps. By day 485, high-relief tray traps had on average 419 live oysters per m^2, while nearby low-relief tray traps only had 207 live oysters per m^2. This difference in live oyster growth between high- and low-relief treatments led to 35% higher rugosity in the high-relief treatments by day 485 (Figure 9). Early in the post-restoration phase, before live oysters had begun growing, sedimentation in the low-relief treatments likely reduced the amount of interstitial space available for organismal colonization. However, as oysters began to grow in low-relief areas, the negative effects of sedimentation on organismal biomass may have been slightly reduced, potentially explaining the gradual convergence between high- and low-relief biomass values over time.

These findings may also explain why observed biomass differences between high- and low-relief treatments were smaller for trays that are emptied every other month (i.e., our bimonthly monitoring trays) than those that were left undisturbed for many months (i.e., the experimental relief manipulation). By emptying and refilling the bimonthly sampling trays on a regular basis, we may have reduced the effects of sedimentation. Overall, our findings emphasize the importance of incorporating vertical relief into future oyster restoration projects. Higher relief leads to more rapid increases in the biomass of oyster reef-associated organisms, as well as increased growth of live oysters. These factors likely facilitate a more rapid convergence between restored and natural reef communities. Live oyster growth, in particular, is essential for the long-term sustained success of any oyster restoration reef.

Based on our findings that biomass within the NOAA restoration reef varied significantly with vertical relief, we felt that it was important to create an accurate bathymetric model of Restoration Site 14, the largest of the three restoration sites (Figure 10). We used a single-beam boat-mounted sonar system coupled with a GPS receiver to map vertical elevation within the reef. Depth measurements were taken at one second intervals as we drove the boat in a grid pattern over the reef. All depth measurements were tide corrected to a nearby NAVD88 reference benchmark before being added to the model. In Figure 10, depth is indicated by color, with the deepest sections of the reef marked in shades of blue and the shallowest areas marked in red and orange. Sudden changes in color reveal rapid changes in depth within the reef. Small areas of yellow and red embedded in larger areas of green and blue indicate potential high-relief patches of reef.

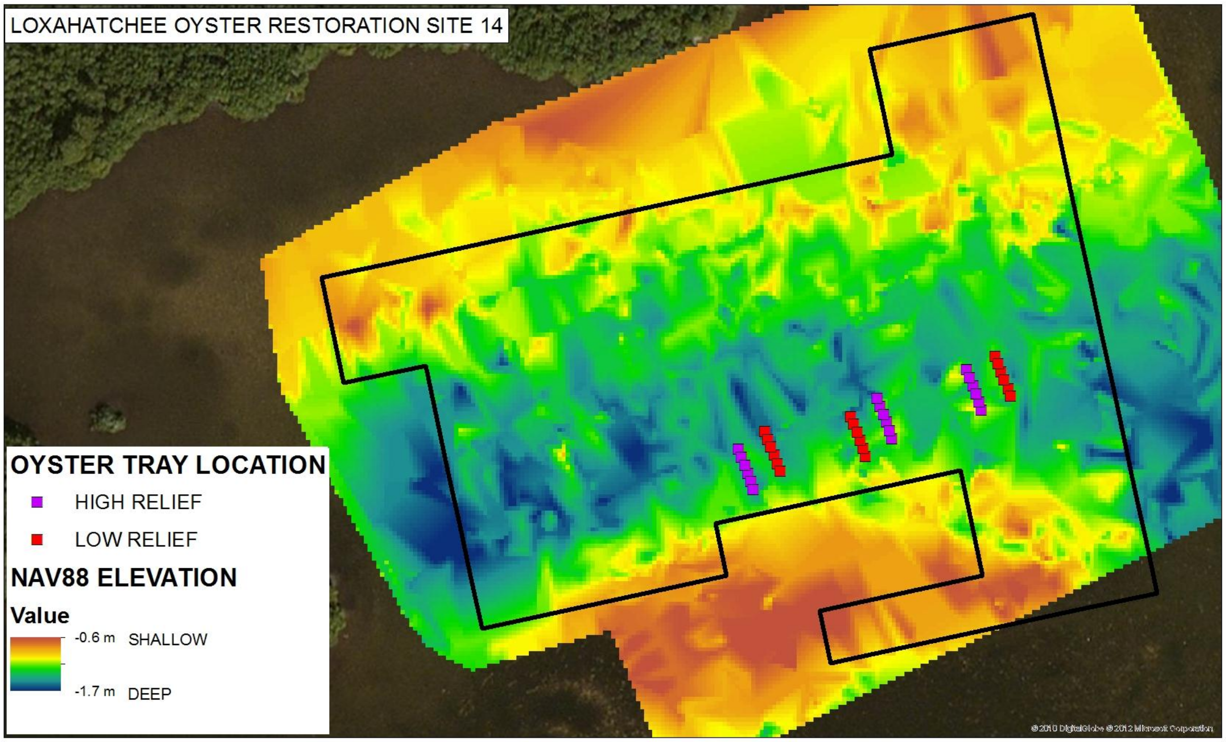

Figure 10. Initial digital elevation model of the NOAA Restoration Site #14. Elevations shown represent depths below NAVD88, obtained from a nearby reference benchmark. Sudden changes in color reveal rapid changes in depth. Small areas of yellow and red embedded in larger areas of green and blue indicate potential high-relief patches of reef. Purple and red squares indicate the location of high- and low-relief sampling trays (not to scale – 7 trays per row) (Map credit: D. Sabin and B. Howard, LRD).

This elevation model allowed us to quantify the area covered by high- and low-relief habitats within the reef. We were then able to use these area values, combined with mean biomass values from both high- and low-relief sections of the reef, to estimate total organismal biomass that is supported by the restoration project. Using our most recent mean biomass values obtained through bimonthly sampling in May 2012 (high relief = 130 g/m^2, low relief = 84 g/m^2), we estimate that the current total biomass contribution from all three sections of the restoration reef is at least 2000 kg. Although high-relief habitats make up less than 3% of the total area of Restoration Site 14, these areas contributed disproportionately to overall reef biomass, particularly early in the reef colonization process. For example, when we used earlier biomass estimates from the high-relief/low-relief experimental manipulation obtained at the end of the dry season in April 2011, ~8 months after the reef was built (high relief = 321 g/m^2, low relief = 32 g/m^2), high-relief sections contributed 169 kg of biomass, while low-relief areas contributed 600 kg. At that time, high-relief habitats accounted for nearly 22% of the biomass at the site, despite representing only 3% of the reef's total area. Since biomass values from high- and low-relief areas have slowly started to converge as the reef matures, this disparity has diminished somewhat. However, vertical relief and habitat complexity should still be taken into account during the construction of oyster restoration reefs, as high-relief areas seem to experience the most rapid increases in biomass of oyster-reef associated organisms.

Our observation of highly variable biomass between high- and low-relief sections of the Loxahatchee River oyster reef warrants additional research to improve our understanding of which specific parameters are the most important drivers of this variability. Future oyster restoration projects provide an excellent opportunity to study specific factors that could maximize the rate of maturity of new restoration reefs. Based on our findings, the following is a summary of several possible factors that may affect community structure and biomass in future restoration projects:

1. Vertical relief / large-scale rugosity – The overall complexity and variation of the surface of a restored oyster reef are important factors for oyster settlement and organism utilization. A more complex three-dimensional reef design (e.g., mounds, piles, or ridges of material, rather than a smooth veneer) likely alters small-scale water flow, thereby improving spat settlement and feeding opportunities. It may also reduce sedimentation rates, the opposite of what would be expected from laminar flows over a more smooth-surfaced restoration reef.
2. Small-scale rugosity – The cultch material used to create a restored oyster reef may play a large role in the rate that the reef is colonized by oysters and benthic organisms. Rougher, more textured, materials may provide greater surface area for organismal colonization than smooth materials. Additionally, more irregular cultch materials provide greater interstitial space for organisms to inhabit, and are less likely to be negatively affected by sedimentation and compaction.
3. Cultch layer thickness – The construction specification for the majority of this restoration project was to deploy a 15 cm thick layer of rock and shell cultch. However, high-relief ridges that experienced the most rapid increases in biomass following restoration were 30 cm thick. Perhaps the cultch is settling into the fine sands and the thicker layer provides additional interstitial space for organisms to inhabit. If true, this could have important consequences for the volume of cultch material used in restoration projects, or the manner in which the cultch is deployed (see point 1 above – thick mounds and ridges as opposed to a thin veneer).
4. Location – The placement of the restoration site relative to prevailing current flow, eddies, and existing (and historical) oyster reefs may influence oyster settlement and organism utilization.
5. Elevation / Water depth – Previous studies have shown that there is substantial variation in oyster settlement and organismal utilization from intertidal to subtidal zones. While all sections of this restoration project were subtidal, future studies should include intertidal restoration reef plots to test the magnitude of these differences. Identifying the optimal elevation for the settlement and growth of oysters and other benthic organisms may improve the success of restoration projects.

It is likely that some combination of the above factors account for variation in biomass within restored oyster reefs. Improving our understanding of these factors and how they affect oyster reef-related organisms could improve the quality and success of future restoration work.

2. Dock Piling Fouling Communities – Contributions from Human-Made Structures to River-Wide Filtration Capacity

One of the reasons why coastal habitats, such as oyster reefs and mangroves, are critically important is due to the “ecosystem services” they provide. For example, vast volumes of water are filtered by oyster reef-associated communities, thereby improving water quality in coastal estuaries. But many coastal systems – especially in South Florida – have been drastically altered, resulting in declines of natural habitat types; declines that are usually associated with a direct loss ecosystem services that would have been provided by these habitats. However, the potential role that human-made structures may play in ameliorating the effects of habitat loss is often ignored. For example, docks and seawalls can support high densities of filter-feeding organisms such as oysters, barnacles, and sponges. In this project, we aim to provide the first estuarine-wide quantification of the filtering capacity that is supported by communities associated with human-made structures in coastal ecosystems. Concomitantly, these data will provide specific recommendations to local managers as to what type of artificial structure (e.g., wood vs. cement dock pilings) may support the most beneficial filtering communities for the system.

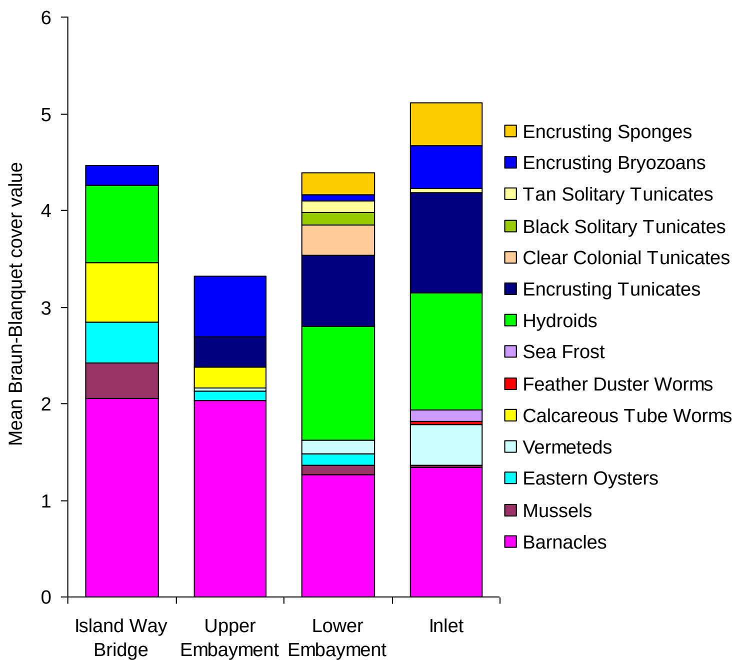

Figure 11. Braun-Blanquet (B-B) values (based on percent cover) for the 14 most abundant filter-feeding functional groups found on wooden dock pilings in the Loxahatchee River, arranged spatially from upstream (left) to downstream (right). See text for the B-B coding system and description of the river sections surveyed.

Preliminary sampling for this project began during fall 2011 and winter 2012. The goal of this initial sampling was to identify filter-feeding species that grow on dock pilings in the Loxahatchee River, and to spatially divide the river up into sections based on community composition. We selected 37 docks in four sections of the river (termed inlet, lower main embayment, upper main embayment, and Island Way Bridge). The inlet section, which was largely marine in nature, ran from the first dock west of the north inlet jetty to the entrance to the Indian River Lagoon. The lower and upper main embayments, wide bay-like sections of the estuary, spanned from the railroad bridge to Oyster Island. Continuing further upstream, the Island Way Bridge section (immediately downstream and upstream of the bridge) was heavily influenced by freshwater inflow during the wet season and represented the upstream limit for marine fouling communities in the river.

At each dock, we sampled one side of a single wooden piling using a series of 30 x 30 cm quadrats lined up vertically from the maximum high tide mark to the benthos. Species were identified, and assigned a Braun-Blanquet value based on percent cover (<5% - one organism = 0.1; <5% - several organisms = 0.5; <5% - many organisms = 1; 5-25% = 2; 25-50% = 3; 50-75% = 4; 75-100% = 5). Throughout the estuary, we identified 54 filter-feeding species that utilize dock pilings. These were combined into 16 functional groups (e.g., all barnacle species were combined for form a barnacle functional group). Fouling communities appear to vary spatially in the Loxahatchee River (Figure 11). Total percent cover was greatest at the inlet, while functional group diversity was higher in the lower main embayment. Percent cover for encrusting sponges and tunicates increased along an upstream-to-downstream gradient, while barnacles exhibited the opposite spatial pattern. Based on these coarse spatial patterns of community structure, we were able to further divide the river into seven sections based on the relative composition of the dock piling fouling communities. While we only sampled wooden dock pilings during the preliminary phase of this study, we anecdotally observed substantial differences in community structure among the other types of pilings, suggesting that piling material may play a role in filtration capacity on a river-wide scale.

Following our preliminary study, summer 2012 was devoted to more extensive sampling of dock piling communities throughout the entire extent of the main fork of the Loxahatchee River. Every dock in the Northwest Fork of the river was visited and the number of pilings quantified (n = 12,860 that support significant fouling communities). The fouling communities of 240 pilings in seven distinct sections of the river were surveyed (Figure 12), and all individual filter-feeding organisms were identified and quantified. Dock pilings in the Loxahatchee River are constructed from four different materials. These include wooden pilings, wood pilings wrapped in a layer of plastic sheeting to reduce fouling (specifically from wood-boring organisms), round polyvinyl chloride (PVC) pipe filled with concrete, and square steel-reinforced concrete beams. In each section of the river, we randomly selected 5 to 22 docks of each piling type. At each dock, a single piling was haphazardly chosen for sampling. Our selection of specific pilings was not based on water depth or position within the dock, although we did not sample pilings in less than 0.3 m of water at low tide, since fouling was virtually absent from these.

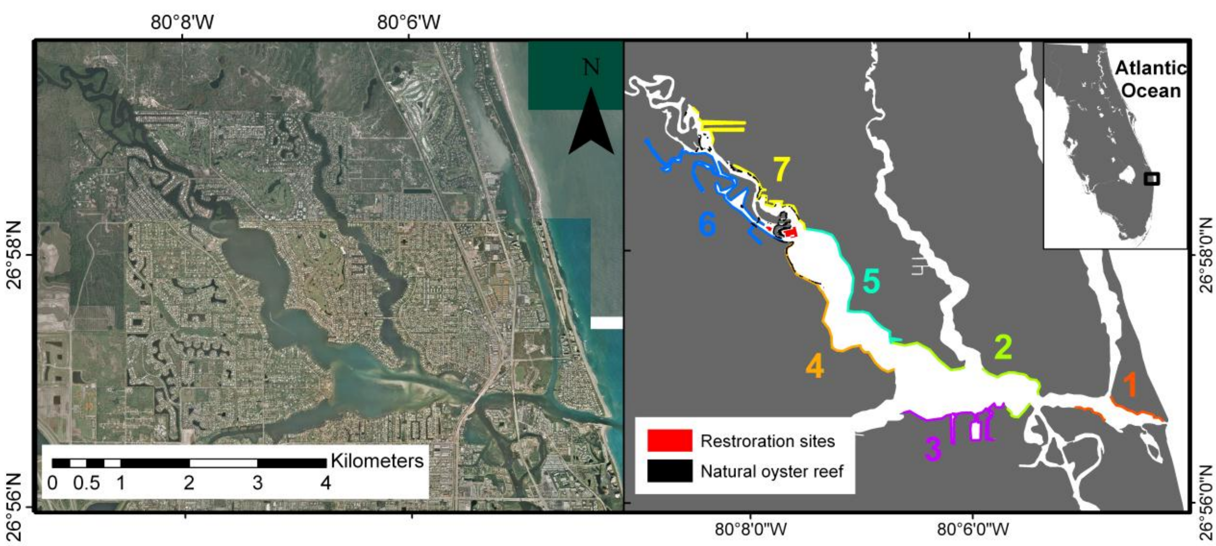

Figure 12. Map of the seven distinct zones used in the dock piling fouling community study.

For each piling, total abundance or cover estimates were made for all sessile filter-feeding organisms. Whenever possible, organisms were identified to the species level; however, certain taxa were combined into functional groups (e.g., barnacles, hydroids, encrusting sponges, encrusting tunicates) to simplify our analyses. For encrusting organisms (e.g., encrusting sponges, tunicates, bryozoans), we estimated total area covered (cm^2 per piling) for each functional group. In the case of solitary taxa, we counted the number of individual organisms present on each piling. When abundance values for a taxon were too large to readily count (i.e., >250 individuals), we estimated per-piling abundance. Solitary taxa that varied considerably in size were divided into multiple size classes. For certain colonial organisms, we utilized different metrics to estimate abundance. For example, we counted the number of stalks (per piling) for hydroids, gorgonians, and erect bryozoans, and zooids for colonial tunicates.

Since most clearance rates for filter feeding organisms in the literature are reported as a rate per g dry mass (i.e., $ml \cdot g^{-1}$ dry $mass \cdot hour^{-1}$) or a rate per polyp (i.e., $ml \cdot polyp^{-1} \cdot hour^{-1}$), we converted our field measurements (counts, area covered) into relevant units. First, we collected replicate samples from each species or functional group from multiple pilings throughout the river. For encrusting organisms, we used a woodworking chisel to remove square tissue samples of a known area. These samples were dried at 60° c until a stable mass was reached. Samples were weighed to the nearest 0.01 g, and mean mass per cm^2 was calculated for each encrusting functional group. Solitary organisms were sorted by size class, dried and weighed as above (live tissue only for mollusks and barnacles), and a mean mass per organism was calculated. For organisms with polyps (e.g., hydroids, gorgonions) or zooids (bryozoans), we used a microscope to count the number of polyps/zooids per stalk or per cm^2 of encrusting tissue. These values allowed us to use field measurements of cover and abundance to estimate total per-piling filtration rates for each piling type in each river section.

Using these data and a nested sampling design at the river scale, we will be able to estimate the biomass of entire dock piling fouling community throughout the river. Undergraduates at FIU were trained to survey the scientific literature and compile a comprehensive database of all previously published filtration rate values for filter feeding organisms that might be found on Loxahatchee River dock pilings. Together, these data sets will be used to estimate the total volume of water filtered per day by dock piling fouling communities – an effort that has not been completed for any other estuarine system. This is a novel approach, and will likely garner wide attention in the scientific and restoration communities once complete.

To assess the ecological value of dock piling fouling communities, we will compare overall filtration rates among dock pilings (made from various materials), natural oyster reefs, and restored oyster reefs at the scale of the entire estuary. Additionally, we will attempt to use archival data to reconstruct filtration rates from the large oyster reefs that were historically found in the lower portion of the estuary. Our model will allow us to see whether the added filtration capacity contributed by human-made structures (docks, artificial oyster reefs) is enough to equal or exceed the filtration capacity that was lost when historic oyster reefs died off due to changing salinity regimes. This project will be a major focus during the upcoming year. In addition to producing a publication in a high-profile scientific journal, we hope to inform the community about the importance of dock piling construction and material selection, so that the ubiquitous docks that line the shoreline of the Loxahatchee River are contributing the greatest possible benefit to the river's health.

3. Lionfish Invasion in the Loxahatchee River

The lionfish invasion of the Caribbean region has emerged as one of the most high profile global environmental issues. Our program in the Loxahatchee River has positioned our team as one of the leading lionfish research groups in the world. We were the first team to document a lionfish invasion of an estuarine ecosystem. We have continued to make observations about the estuarine invasion of lionfish in the river, including a recent discovery of a lionfish 6.6 km (4.1 m) upstream from the ocean at oyster reef Restoration Site 14, where bottom salinity was only 8‰ (Figure 13). This is the furthest documented upriver intrusion by lionfish to date, highlighting the invasion potential of lionfish in other Florida and Caribbean estuaries. Our work has shown that reduced salinities commonly encountered in estuaries are little barrier to the establishment of lionfish, as documented by our discovery of numerous individuals in the river during the extreme freshwater discharge event that followed the passage of Tropical Storm Isaac.

Our experiences working with lionfish on the Loxahatchee River have facilitated numerous other collaborative projects regarding the basic ecology of impacts of lionfish in their invasive range. Locally, we have worked with FWC and other agencies to help develop the Indian River Lagoon Lionfish Megatransect, and will help support initiation of a lionfish removal experiment with the University of South Florida. We also will continue to promote educational activities regarding the invasion and track the status of the lionfish population within the estuary. During the upcoming year, we will take advantage of all possible opportunities to present our highly popular and educational public outreach programs throughout the region. We will consider specific new additional research projects and collaborations as individual opportunities arise.

Links to our scientific publications on lionfish are now posted on the Loxahatchee River District web page: http://www.loxahatcheeriver.org/reports.php. In addition, funding from LRD has helped support the website "The Abaco Scientist": http://absci.fiu.edu/. This website contains up to the minute lionfish information in South Florida and beyond. Posts regarding lionfish in the Loxahatchee have received thousands of internet visits, highlighting the collaboration of FIU and LRD with respect to the lionfish invasion.

Figure 13. A 13 cm (5") lionfish hovering over a restored oyster reef (Restoration Site 14) in the Loxahatchee River. In the background, a benthic tray trap is visible (see oyster reef sections above). This represents the furthest documented upriver intrusion by the species to date, and provides important insight regarding the invasion potential of lionfish in other estuaries.

4. Other Fish Ecology Studies

Support from LRD has provided a basis to develop a multi-faceted study of the ecology of economically and ecologically important fish species in the river (e.g., gray snapper, snook). We have examined a variety of factors associated with fish ecology, including fish movements associated with altered freshwater inflow patters, fish habitat preference in response to oyster reef restoration, and fish feeding interactions along a salinity gradient. One paper has been published from this research (Yeager et al. 2010, link available on the Loxahatchee River District website), and a second submitted for review (Yeager et al. In review). We also have continued to collaborate with other research groups (Florida Fish and Wildlife Conservation Commission, University of Florida). We have maintained an array of underwater acoustic listening devices in the river to track the movements of fishes (and other marine organisms) tagged by research groups throughout Florida and in other Atlantic coast states. Through our regular downloading and servicing of these underwater receivers, we have helped other researchers identify important fish movements, including long-distance migrations into the Loxahatchee River by fish tagged outside of Florida.

References Cited

Coen, L.D. and Luckenbach, M.W. 2000. Developing success criteria and goals for evaluating oyster reef restoration: Ecological function or resource exploitation? *Ecological Engineering* 15, 323-343.

Lenihan, H.S., 1999. Physical-biological coupling on oyster reefs: how habitat structure influences individual performance. *Ecological Monographs* 69, 251-275.

Peterson, C.H., Grabowski, J.H. and Powers, S.P. 2003. Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. *Marine Ecology Progress Series* 264, 249-264.

Rodney, W.S. and Paynter, K.T. 2006. Comparisons of macrofaunal assemblages on restored and non-restored oyster reefs in mesohaline regions of Chesapeake Bay in Maryland. *Journal of Experimental Marine Biology and Ecology* 335, 39-51.

Schulte, D.M., Burke, R.P. and Lipcius, R.N. 2009. Unprecedented Restoration of a Native Oyster Metapopulation. *Science* 325, 1124-1128.

Yeager, L.A. and Layman. C.A. 2011. Energy flow to consumers in a sub-tropical oyster reef food web. *Aquatic Ecology* 45(2):267-277.

Yeager, L.A., Layman, C.A. and Hammerschlag-Peyer, C.M. In review. Foraging trade-offs for a generalist fish predator across a landscape gradient. Submitted to *Canadian Journal of Fisheries and Aquatic Sciences*.