# Restoring Resilient Oyster Reefs in Florida's Big Bend

*Peter Frederick[1], Bill Pine[1], Jennifer Seavey[1,2], Leslie Sturmer[3]*

[1]. Department of Wildlife Ecology and Conservation, University of Florida, Gainesville, FL.

[2]. Shoals Marine Laboratories, School of Marine Science and Ocean Engineering, University of New Hampshire, Durham, NH and College of Agriculture and Life Sciences, Cornell University, Ithaca, NY

[3]. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, FL.

Final Report to

The Nature Conservancy

And the

National Oceanic and Atmospheric Administration

August 31, 2015

# Executive Summary

Oyster reefs may be the world's most endangered biotic community, with entire or major losses to nearly every stock worldwide during the last 50 years. Oyster reefs in the Gulf of Mexico remain among the most widespread and viable; moreover, they support a variety of highly valued ecological services and human industries. Although Florida's Big Bend region is the most undeveloped coastline in the continental U.S., oyster reefs there have undergone a 66% decline in 30 years, possibly in response to episodic periods of reduced freshwater flow. The proposed mechanism for the observed declines is that above normal estuarine salinities resulting from reduced levels of freshwater discharge lead to high oyster mortality and reduced recruitment. Low oyster recruitment can lead to reductions in the rate of accumulation of cultch – the living and dead matrix of material that comprises healthy oyster bars. As cultch levels decline the risk of oyster bars "breaking apart" during storms and other disturbances increases; once these reefs begin to fracture the remaining cultch destabilizes and washes away, leaving a rapidly eroding sand bar. This is a one-way process – oysters do not appear to be able to recolonize these bars because the substrate is not appropriate for recruitment or survival of young oysters, thus new shell structure cannot establish. Losses of these reefs have much greater impact than simply the net loss of oyster habitat. In this region these oyster reefs serve as ecosystem engineers, and the linear coastwise directionality of reefs strongly modulates river discharge by impounding freshwater on the bay side and slowing its mixing with the ocean, resulting in estuarine conditions. As the oyster reefs decrease in elevation and area, their ability to retain freshwater and the mid-range salinities typical of estuaries declines, leading to accelerated oyster mortality, cultch loss, and limited freshwater retention via healthy oyster reef communities. It is important to note that this relationship can also function in a positive direction – higher, more contiguous reefs would presumably retain more freshwater, thereby buffering oysters and estuarine habitats to effects of low freshwater discharge while also dampening impacts to coastal marsh and habitats from rising sea-levels.

While efforts are being made to conserve upstream water resources and reduce anthropogenic impacts on frequency or severity of low-flow events, this process may take decades. In this context, an effective conservation strategy is to minimize the likelihood of further losses in oyster reefs with Suwannee Sound by adding durable substrate to the degraded reefs but using material that will persist following episodic die-offs of oysters in the future. There is no apparent shortage of larvae in this nearshore ecosystem (probably due to the redundancy in source populations from wide distribution of oyster reefs in inland marsh/tidal creek habitats), and the existence of durable substrate would allow oysters to repeatedly recolonize reefs, making them resilient in the long term to fluctuations in freshwater flow. This intervention offers a means to make reefs persistent and to potentially reverse the negative feedback loop.

This pilot project was designed to test two main assumptions of this proposed conservation action. First, can two durable materials (limerock boulders and cultch-filled aquaculture bags) be used to recruit and maintain oyster populations on what have become shifting sand bars? Second, in the longer term can this durable material result in more persistent oyster populations? Finally, to what extent do linear coastwise chains of oyster reefs detain fresh water and create estuarine conditions?

The study design involved creation of four restoration sites (21 X 21m) and four control sites on the Lone Cabbage reef, a coastwise chain of degraded oyster bars located near the mouth of the East Pass of the Suwannee River, Levy Co., Florida. Restored and control sites were arranged in pairs around inlets between individual bars. Restoration sites were covered 20 – 40

cm deep with 10 – 20 cm diameter limerock boulders, and the outline of the boulders was lined with 1.4 m$^2$ recovered clam bags from local aquaculture operations in nearby Cedar Key.  The bags were ones in which clams had largely died, and oysters and other fouling organisms had caused the bags to become difficult to move and useless for further clam aquaculture.  Recovered clam bags in a previous project contained an average of 8,000 live oysters, and have been used to construct one existing oyster bar that has had persistent oyster populations during the past 7 yr.

   Following permitting, rocks were deposited on restoration sites in September 2013, and clam bags were gradually placed over the next 19 months.  Evaluation of the restoration activities included comparing restoration sites with themselves before and after construction, comparing restoration sites with control sites before and after construction, comparing restoration sites with a suite of 38 local oyster reefs in the area, and comparing restoration sites with other restoration sites within the Gulf of Mexico.  We monitored oyster density and size using consistent protocols, monitored elevation change over time using surveying equipment, and documented changes in aquatic bird and blue crab (*Callinectes sapidus*) use of restored and control sites using time lapse trail cameras.

        We placed a total of 471 cu yards of limestone rock on the four restoration reefs at a cost of $142/yard or a total of $67,184. Although clam bags were donated, the cost of moving bags from aquaculture areas to restoration sties totaled $13,051. Overall, the construction of the reefs totaled $80,235, and the constructed reefs totaled 455 square meters, resulting in a cost of $176.34/square meter or $16.37/ square foot. Note that this tally does not include the preconstruction costs in boat time and personnel, nor the costs of contracting, billing, permitting or inspection.

        By placing durable substrate on the reef, average elevation on restoration reefs increased by 16 cm compared with the same sites pre-restoration.  Compared with control sites, there was an increase in mean oyster density on restoration sites by 2.65X on rock, 14.5X on clam bags and 9.2X overall on the restoration reefs.  Clam bags contributed approximately 25% of the surface area on the reef, but accounted for 52% of the oysters overall. Densities on restoration sites were between 89 and 125 times the densities measured at a suite of other offshore reefs in the vicinity of the Lone Cabbage complex, and between 7 and 9 times the densities on inshore marsh reefs, which have the highest densities in the area.  Densities on restored sites at Lone Cabbage exceeded all but one of the reported densities on restored oyster reefs in the Gulf of Mexico, and were in the 90$^{th}$ percentile of these sites for density. Restoration activities were also associated with a moderate increase in mean oyster size on rock (6.5mm, 23.7% increase) and a more substantial one on clam bags(12.0 mm, 44% increase) compared with control sites.

         The effects of this change in substrate and habitat appear to have had the desired outcomes.  We also found that the Lone Cabbage reef chain has lower salinities on the landward side than the seaward, strongly suggesting an effect of the reef on local salinities by an average of 3 psu over a period of 1.7 years, with particular periods when salinities on bay and sea sides differed by over 10 psu.  Modeled results suggest that the effect on local salinities will increase markedly with restored increments of reef elevation.

        This restoration action also appears to have had effects at other trophic levels.  Total bird use on restoration sites decreased by an average of 92% across species groups, with some species being found 56 times as often on the sandy control sites.  This suggests a potential tradeoff between restoring hard substrate for oyster restoration, and attracting aquatic birds.  Bird surveys

were performed before oyster populations were well established however, and it is unclear whether the same results would be obtained with a more mature restoration site. Blue crab usage sampled via trail cameras (an original application of visual censusing) on restoration sites suggested a density increase of 7.5X by comparison with control sites, and this result agreed in direction with less formal walk-through surveys. Although we have little confidence in the absolute values of the density estimates because of visibility and detection issues, we believe that this evidence is collectively suggestive of higher densities of blue crabs on the restored sites.

Rock and clam bags each had advantages as restoration materials - rock was more durable but initial costs per square meter were higher than bags. Bags showed much higher productivity of oysters but probably have a shorter lifespan as a durable reef material. We suggest that in the future clam bags may be superior for hardening edges of high tidal flow areas (inlets or cuts) between higher elevation areas. We also suggest that the contents of clam bags can be used to adjust the interstitial space of rock layers to increase suitability for oyster recruitment and survival. Although the large rock size we used (10 – 20 cm) was appropriate for the high energy nature of the site, it created large spaces between rocks. We speculate that this large interstitial space may have allowed predators of small oysters unusual access to juvenile oysters, an effect not seen in the clam bags. Although we noted some increase in elevation due to entrapment of sediments and buryment of oysters on the rocky restoration sites, we hypothesize that this will become much less of an issue as more of the sandy substrate of the entire reef is stabilized with durable material.

Several contracting lessons were learned. First, there are various industries in competition for the local labor pool (clam aquaculture, construction, tourism, oyster harvest and other fishing) and the least competition for this pool appears to occur during the summer months. Second, costs of restoration (materials, labor, energy) are all related to economic cycles and local economic bubbles, and these can change very rapidly. Since there is often a long permitting phase for this kind of work, actual and estimated construction costs can be strongly mismatched. For this reason we suggest that restoration funders contract in two phases: first, permitting, design and other preconstruction activities and secondly, later, construction and post-restoration monitoring– this allows the most up to date costs to be incorporated.

This project indicated that (1) the restoration techniques used can result in dramatic increases in oyster density and biotic function on sites that are highly degraded, and (2) increasing elevation and hardening of these degraded reefs could affect estuarine salinities of a much larger area, with an estimated 167 ha of marsh affected for every 1 ha of reef restored. Overall, this pilot project demonstrated the very strong potential for restoring reefs and reef chains in the Big Bend region, using locally sourced materials and local labor. The use of both clam bags and rock was successful in the short term. We hope in the future to be able to evaluate the longer term prediction that sites hardened with durable substrate will be better at retaining and quickly recruiting oyster populations following a variety of catastrophes including low flow events, and so make reefs and the estuaries they protect more resilient in the face of changing weather, human use of freshwater, and sea level rise.

# Table of Contents

## Introduction and background

Oyster reefs are biological communities that support considerable biodiversity, provide keystone habitat for juvenile fish, forage fish, invertebrates, and birds, and support economically important fisheries (Coen et al. 2007, Beck et al. 2011, Grabowski et al. 2012). Oyster reefs can also function as barrier islands in many areas, dampening wave action to reduce coastal erosion and protect human coastlines and coastal communities from storm damage (Scyphers et al. 2011). Oyster reefs are probably the most highly impacted marine habitat in the world, and though populations in the Gulf of Mexico (GOM) are degraded, they still constitute some of the healthiest and most viable oyster habitat globally (Beck et al. 2011).

Oyster reefs in Florida's Big Bend, the location of the project described here, exist within a region unique to the GOM because of its ecologically rich, low energy shoreline habitats (i.e., sea grass meadows, salt marshes, and oyster reefs), low human population density, and high percentage of coastline under state or federal management. Although this is the largest stretch of undeveloped coastline in the continental US, oyster reef area has been reduced by 66%-88% during the past 30 years (Seavey et al. 2011) probably because of increasing drought severity and frequency, leading to high oyster mortality and low recruitment. Because the Big Bend has such high conservation value (evident by large purchases of state and federal conservation property holdings) and otherwise excellent habitat conditions for oyster reefs and oyster reef dependent ecosystems, promoting resilience in these oyster populations is important to the ecology and economy of the GOM region as a whole.

Recent evidence by our team (Seavey et al. 2011) suggested that reductions in freshwater discharge from the Suwannee River, which feeds the Big Bend (Figure 1), triggers a cascading decline in oyster reef resilience mediated by high salinity which leads to increased disease and predation on oysters, eventual loss of oyster settlement substrate, and conversion of oyster reefs to sandbars. Once these reefs lose their covering of durable, living shell, they begin to break apart, spread out, and rapidly lose elevation (cf 7 cm/yr, on average), and eventually become low elevation sandbars. In many cases, entire chains of reefs have become truncated, with less than half the original area of intertidal reef remaining, and gaps between reefs have enlarged from 27 to 984 meters (Figure 2). This general sequence has been repeated in over 30 long-term monitoring stations in the area, and appears to be irreversible through natural dynamics. This is a considerable departure from historical resilience – these reefs are thought to be ca 3,500 yr old (Grinnell 1972). Restoration of resilience to these reefs requires that (1) suitable, durable substrate is provided as foundation, and (2) oyster shell substrate accrues through constant recruitment and natural mortality (i.e., positive shell budget).

Figure 1. Map showing the oyster reef construction area within Florida's Big Bend (inset) and within Levy County.
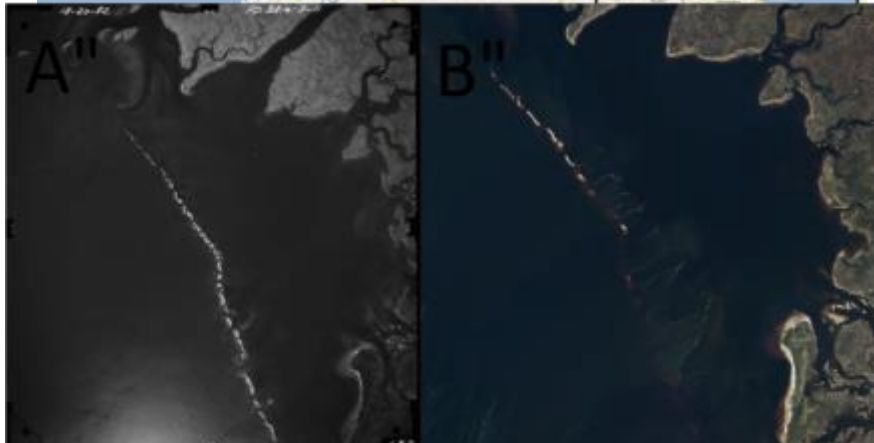


**Figure 2.** Aerial photographs of the Lone Cabbage chain of oyster reefs at low tide in 1982 (left) and 2010 (right). Note loss of entire reefs, loss of total reef area, and widening of gaps between reefs. East pass of the Suwannee River is shown at the top of both photos.

To arrive at a resilient restoration strategy, it is important to further understand the conditions resulting in loss. Decline of oyster reefs in this area has been linked to repeated episodes of reduced freshwater inputs, which result in high salinity and associated mortality events from disease and predation (Seavey et al. 2011; Petes et al. 2012). While eastern oysters (*Crassostrea virginica*) are capable of enduring periodic fluctuations in salinity (2-30 psu, practical salinity units), there are temporal limits to their physiological endurance to both high and low salinity levels (Buzan et al. 2009, Petes et al. 2012). In this region of coastline, the Suwanee River is the major source of freshwater, and precipitation patterns and water use in the larger Suwannee River and adjacent river and basins influence surface and sub-surface discharge across this coastline (Raabe and Bialkowska-Jelinska 2007). Frequency of low river discharge events in the Suwannee River (< 1 s.d. below period of record monthly means) has increased by approximately one order of magnitude during the period from 1995-2008 compared to the previous 55 years. Since annual rainfall has not changed appreciably in the drainage, human use of freshwater is most likely the driver of this change in discharge (Seavey et al. 2011).

While efforts are underway to increase river flow through water conservation and minimum-flow deliveries in the Suwannee River (Farrell et al. 2005), this change in human freshwater usage is not likely to be reversed in the near future. For this reason, any oyster restoration strategy that leads to resilience must recognize that oyster populations will periodically and markedly decline as a function of episodic periods of low freshwater discharge (Petes et al. 2012). Based on our prior assessments of oyster spatfall, the project area does not appear to be limited by available oyster larvae, many of which come from inshore populations of oysters that are more buffered from the effects of freshwater discharge (Frederick and Sturmer, *unpublished data*). Instead, recruitment appears to be limited by survival of larvae due to the lack of appropriate, stable substrate following episodic, low-freshwater discharge events.

For this reason, **the primary goal of this pilot project has been to promote resilience of Big Bend oyster populations, following the hypothesis that oyster populations are substrate limited.** Success in achieving this goal is clearly dependent upon the establishment of robust substrate that oysters can attach to during the often-long periods of normal freshwater flow in which conditions for growth and reproduction are favorable. We therefore proposed to deploy durable restoration substrates that would allow repeated and sustainable rebuilding of oyster populations following cyclical mortality events. In this way, human intervention could introduce resilience into what had become a one-way path to reef loss. The main prediction of this intervention is not that durable substrate will entirely prevent oyster death during periods of low freshwater flow – but that it will provide a mechanism for repeated recolonization following periods of low freshwater flow.

In this particular section of the Gulf coast, it is also likely that the restoration and retention of reefs may also lead to attenuation of the low inflow/mortality events. This is because there is a critical feedback mechanism between the restoration of offshore reefs and the maintenance of inshore reefs and estuarine habitat in this area. In the Big Bend, oyster reefs are not protected behind barrier islands, meaning chains of offshore oyster reefs often form the first critical line of defense for coastal habitats from wave action and storm impacts (Hine et al. 1988). In many places, these chains also form a permeable barrier to freshwater flow from adjacent watersheds and, as such, are able to retain freshwater in estuaries (Wright et al. 2005), functioning much like barrier islands in other systems. This ability to retain freshwater temporarily and thus influence inshore salinities has been shown by inshore and offshore salinity measurements and modeling (Kaplan et al. 2014) and sediment samples (Wright et al. 2005). The retention of freshwater by the linear chains of oyster reefs within the restoration area is therefore seen as a keystone biological process, estimated to sustain estuarine conditions and erosion protection to salt marsh, oyster reef, and seagrass meadow habitats. Within Suwannee sound and adjacent coastline, we estimate that the ratio of restored reef to restored estuarine habitat is approximately 1:670, a substantial return on restoration investment. This retention of freshwater is likely the single largest ecosystem service of oyster reefs in this region and, in many years, we believe is largely responsible for the estuarine nature of the coast.

Given the reduction in freshwater discharge from the Suwannee River in recent decades, retention of freshwater is of increasing importance to maintain estuarine habitats in the Big Bend (Seavey et al. 2011). This is because the presence, or absence, of offshore reefs can exaggerate both positive and negative feedback in this system, respectively. Specifically, our prediction is that restoration of these permeable "dams" is likely to make Big Bend estuaries *more* resilient to the effects of decreased freshwater discharge by retaining available freshwater in the estuary. Moreover, facilitating the restoration of these living dams through the deployment of durable substrates also has the potential to help oysters keep up with sea level rise because vertical growth rates of oysters can outpace current rates of sea level rise (Rodriguez et al. 2014). In contrast, continued loss of these reefs, which is nearly certain without immediate restoration action, is likely to make this region *less resilient* to fluctuations in river discharge and sea level rise as the erosion of these offshore chains has been linked to the salinization of inshore waters and loss of estuarine habitats, including shorelines (Kaplan et al. 2014). **Thus, a second major long term goal of restoring these reefs is to damp fluctuations in estuarine salinities in the inshore areas, increasing the resilience of these habitats to both increased variability in freshwater inputs and rising sea levels.**

Oyster reefs are known to provide a suite of ecosystem services including reduction of storm surge and wave impacts, filtering of pollutants and nutrients in the water column, sequestration of nutrients into the substrate, creation of critical nursery habitat for commercially and recreationally important species, and provision of recreational fishing habitat (Coen et al. 2007, Grabowski et al. 2012). In Florida, recreational fishing is a more than $7 billion annual industry and, in some years, accounts for one third of the global recreational fishing effort (Walters and Martell 2004). Oyster reefs in the Big Bend provide key habitat for a variety of important recreationally fished species, including spotted seatrout (*Cynoscion nebulosus*) and red drum (*Sciaenops ocellatus*). NOAA-managed and federally endangered Gulf sturgeon (*Acipenser oxyrinchus desotoi*) also are found in estuarine waters in the Big Bend and are likely to respond positively to reef restoration as juveniles of these species depend on estuarine habitats and brackish conditions. Seagrass meadows, habitats that benefit from the water-enhancement services of adjacent oyster reefs (Grabowski et al. 2012), also are widely distributed throughout the Big Bend, where they are considered Essential Fish Habitat under MSFCMA for gag (*Mycteroperca microlepis*) and red grouper (*Epinephelus morio*). Likewise, salt marshes that are adjacent to and immediately affected by ativities in the restoration area are critical habitat for the Florida Salt Marsh Vole (*Microtus pennsylvanicus dukecampbelli*), an endangered species. Additionally, oyster reefs in the Big Bend host the largest aggregation of American oystercatcher (*Haematopus palliates*) in the US, a Florida Species of Concern that forages and roosts in this habitat. Thus, oyster reefs, and the nearby habitats they support, function as critical habitat for over 60 species identified by US Fish and Wildlife Service (USFWS) Florida State Wildlife Action Plan as species of greatest conservation need. Restoration of this particular chain of oyster reefs should therefore be a top priority, given its disproportionately large effects on the production of ecologically and economically valuable resources in the Big Bend**.**

While some of the features of this coastline are unique, this pilot restoration project addresses a problem that is central to oyster reef management and restoration throughout the Gulf of Mexico and South Atlantic (USGCRP 2009). Most of the large estuaries in this region are receiving reduced freshwater inputs as a result of upstream water use and management. These reductions may intensify in the future due to climate-change-induced drought events (reduced precipitation) coupled with increased surface and subsurface water demands and sea-level rise. Because freshwater discharge has been isolated as the only major driver of oyster populations in the otherwise pristine Big Bend, it has become a test location for making oyster populations resilient in increasingly salinized estuaries. This project is therefore foundational for future work in this and other GOM locations because it tests two basic premises of restoration to promote resilience under conditions of reduced freshwater inputs and salinizing estuaries: first, that durable substrate will allow repeated recruitment of oysters following episodic low freshwater flow events, and second, that chains of oyster reefs can be used as aquascaping tools at large spatial scales to retain fresh water for inshore estuaries.

A broadly supported strategy from industry, resource managers, and scientists for restoring oyster reefs in the Big Bend region is to (1) restore chains of reefs closest to sources of freshwater to promote persistence of oysters even during low freshwater flow, and (2) provide durable substrate which will enable oysters to recruit both now and in the wake of future mortality events and therefore promote resilience of oyster reefs.

**Restoration Approach**

The basic approach for an overall restoration plan in the study area is to provide durable substrate (materials that last >10 yr and that persist in the presence of strong currents and wave action) on historic reefs, with an overall prediction that oyster populations will be more persistent in restored sites than in unrestored conditions. We used two forms of durable substrate in our restoration efforts: limerock boulders and recycled clam aquaculture bags containing shell
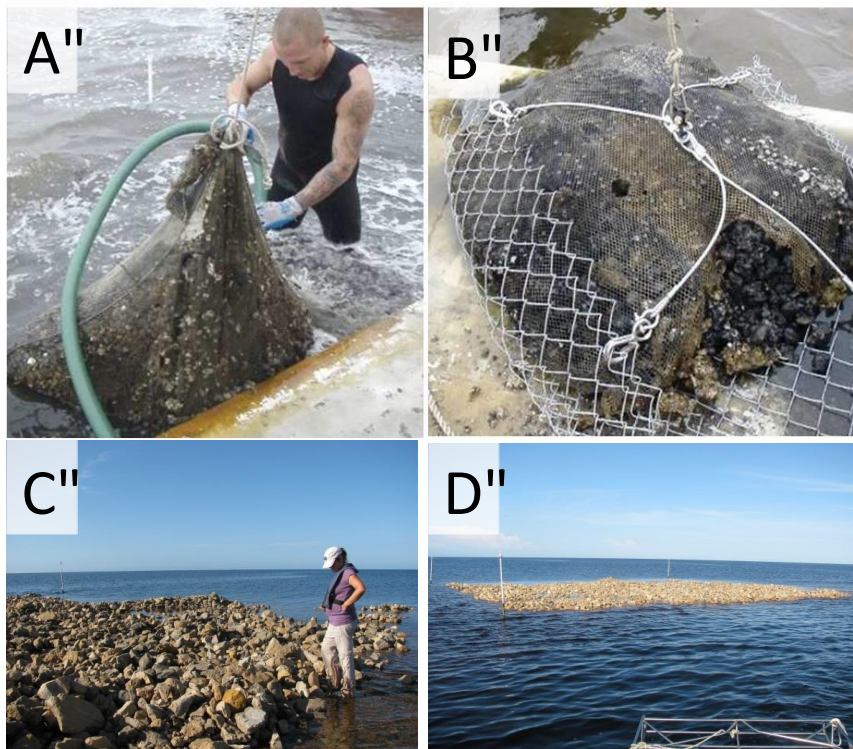


Figure 3. Aquaculture clam bags (A,B) and limestone boulder (C) materials used to construct durable oyster reefs (D) in the Lone Cabbage oyster reef chain.

cultch, living oysters, and other encrusting organisms (Figure 3). The mesh bags were used to grow cultured hard clams (*Mercenaria mercenaria*) on lands leased from the State of Florida by commercial clam farmers that are adjacent (cf 11 km) to the Lone Cabbage Reef complex. During storm or extreme weather events, many of the clams die and their shell becomes colonized by oysters and other species – an average of 8,000 living oysters per bag has been recorded. These bags are often then abandoned on the lease by the clam industry because it is cheaper and easier to use new bags than to reclaim these older, encrusted clam bags for re-use in clam farming. We estimate over 20,000 encrusted bags are currently available to be relocated for oyster bar restoration. These bags and their living contents (cf 35 – 135 kg) are durable, heavy, and represent ready-made oyster reef building blocks that are resistant to wave action and strong currents. We employed local watermen to remove and transport the bags for use in oyster restoration. Thus, the removal and relocation of the clam bags can benefit the clam industry and employ fishermen, while the reuse of this material can form a cheap, locally sourced, and effective construction product for oyster restoration.

Our second restoration substrate material was locally-sourced limestone boulders (Figure 3). The west coast of Florida is a karst limestone platform and exposed limestone is a common feature in submerged habitat. Oysters commonly grow on limestone and Hine et al. (1988) identified these rock outcroppings as key bases for the development of some oyster reefs. Local examples indicate that natural aggregations of limestone persist even in areas of high wave

action. In Suwannee Sound, however, oyster reefs are based primarily on riverine sediments (Wright et al. 2005) that are much less stable, and, without an intact shell base, do not provide the persistent nucleation sites for oyster settlement provided by limestone (Hine et al. 1988). Our restoration design combined both materials using a mix of clam bags and limestone to create stable structure and persistent nucleation sites for oyster settlement, growth, and reef formation.

These techniques have a track record of success. Although rock has never been used before in oyster restoration on this section of coast, rock mostly in the form of limestone was found to be much more successful than shell in recruiting oysters to restoration sites (LaPeyre et al. 2014). In 2007-2009, a 0.17-acre oyster reef was built using 4,200 clam bags to help limit erosion of cultural sites at Atsena Otie Key. This oyster reef has proved to be both ecologically viable with a full suite of oyster reef-associated flora and fauna, as well as robust growth of oysters, and resilient in a high wave action and high tidal flow zone (Figures 4 and 5). Oyster densities on this reef also remained high during and following a region-wide, drought-induced mortality event from 2011-2013, thus fulfilling a key prediction of persistence following periods of low flow and high mortality.



Figure 4. Constructed oyster reef composed entirely of repurposed clam bags on Atsena Otie Key near the town of Cedar Key, FL.



Figure 5. Oysters growing through mesh clam bags on Atsena Otie reef.

We restored oyster reefs through the addition of durable substrate, which would (a) provide structure for the retention and recruitment of oysters, (b) provide a living breakwater to

help make the coastline more resilient to wave action and storm surge, and (c) provide a self-sustaining and self-reinforcing means to retain increasingly scarce freshwater for maintaining estuarine salinities on the shoreward side.  While restoring an entire chain of reefs is clearly the most straightforward way to achieve these benefits, that objective would take very large amounts of money and effort.  In this pilot project we explored the efficacy and cost-efficiency of using durable substrates, and the potential effects that these methods would have on oyster density and size, salinity regimes, blue crab densities, and aquatic bird use. To understand the effect of restoration efforts, we (1) compared specific restoration sites to themselves before and after restoration, (2) compared restoration sites to nearby control sites in the same reef chain at the same time, and (3) compared restoration sites to an array of oyster reefs that we had studied in 2010 – 2012.

## Methods

**Study area**

Our restoration site was located in the Gulf of Mexico, along Florida's Big Bend, in Levy County approximately 3 km south of the mouth of the Suwannee River and 14 km north of Cedar Key, Florida (Lat: 29°14'53.86"N, Long: 83° 6'1.21"W, see Figure 1). The approximate location of construction sites within the Lone Cabbage chain of oyster bars in the Suwannee Sound and the arrangement of restored reefs at inlets are illustrated in Figure 6. Lone Cabbage reef is at least 3,500 years old (Wright et al. 2005), has lost more than 80% of the historical oyster populations (Seavey et al 2011), and compared to many reefs in the area, is close to a major freshwater source (East Pass of the Suwannee River). While most of Lone Cabbage reef is within an area currently designated as conditionally closed to oyster harvest due to fecal coliform contamination from the Suwannee River, it was a significant area for commercial harvest as recently as the 1970s.

Some features of the recent history of the reef have been documented by aerial photographs (Seavey et al. 2011, see Figure 2).  As recently as the early 1990s, the chain extended from East Pass south to Buck Island (6.2 km) with as many as 38 individual bars.  By comparison, by 2010 the reef extended only 3.3 miles, with only 13 islands.  Further, the size and depth of the inlets had changed markedly, with expansion in width from 27 to 984 meters on average.  The elevation of bars prior to 2010 is unknown, but post-2010 trends in elevation (-7.2 cm/yr), elevations of oysters on less degraded inshore bars, and local oral histories suggest that the Lone Cabbage reef elevation may have been reduced by up to a meter in some places since the early 1980s. While the reef chain has been closed to harvest since the late 1970s, local fishermen indicate that harvest may have been exceptionally heavy in some years when closures in the much larger Apalachicola fishery forced large numbers of oystermen onto these beds.  In addition, there may have been some illegal fishing pressure since the closure which has varied with oyster dock prices and local availability. Thus while harvest pressure has probably not been an important forcing function for oyster populations on this reef in the past two decades, it is possible that episodic harvest events in the past may have been important to the trajectory of oyster populations on this reef.

**Restoration methods**

We initially recognized that the inlet/bar arrangement of the Lone Cabbage chain resulted in important flow characteristics that could have feedbacks to oysters and other reef faunal dynamics. Generally the reefs force flow into the intervening inlets at some point as water recedes below the level of the bars, and flow rates of up to 2.5 m/sec have been witnessed on strong outgoing tides. Because of this relationship, we considered one inlet formed by the ends of two bars should be the experimental unit, rather than individual bars. We randomly designated the first major inlet at the north end of the chain as a restoration site, and then alternated restoration with control sites moving south. Edges of reefs adjoining these inlets were then numbered so that the northernmost pair (restoration) was 1 & 2, the next pair (control) was 3&4, and so on. We created four total inlet treatments, with sites1&2 and 5&6 being restoration sites, and 3&4 and 7&8 being the intervening control sites (Figure 6).
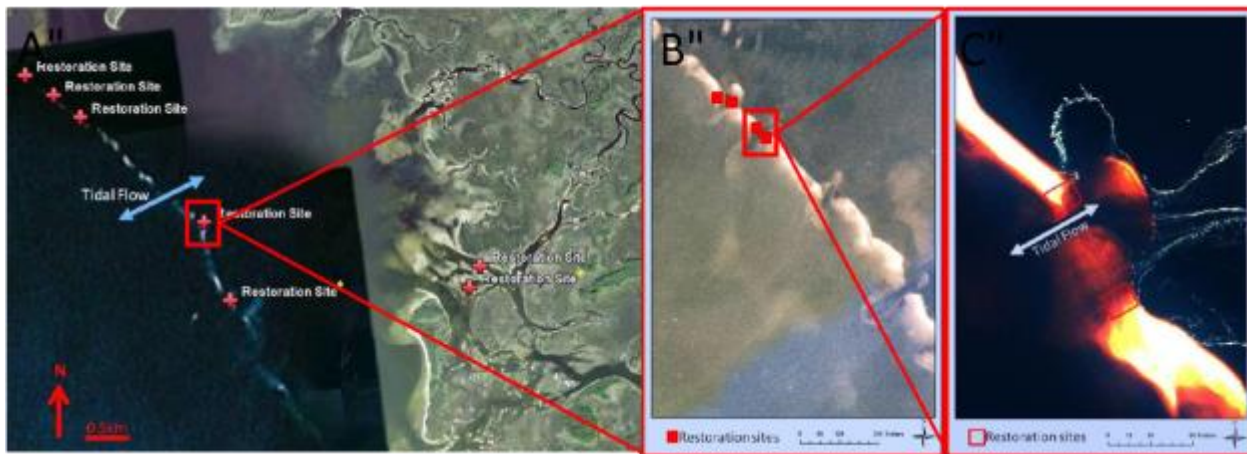


Figure 6. Maps showing approximate location of construction sites within the Lone Cabbage chain of bars in the Suwannee Sound (A), and the arrangement of restored reefs at inlets at the landscape (B) and local (C) spatial scales.

All sites were mapped as squares 21.3 meters on a side, with one side oriented normal to the direction of flow through the inlet; all squares were oriented the same relative to one another. The middle of each square at each site was determined through true elevation, with squares centered on -0.425 m ortho elevation (see Elevation Determinations below). Corners were marked with 10 cm diameter PVC poles that were approximately 2 meters in length driven into the substrate.

Prior to construction, we measured elevation, oyster density and oyster size on each of the restored and control sites. Following this pre-construction assessment we cleared all live oysters from the restored sites and scattered them to the northern side of the construction area. These oysters were then redistributed by random throwing onto the construction footprint once construction was complete. We carried out the same activity in a mock manner (picking up and moving all oysters) at all control sites.

Restoration sites were then covered with limerock boulders (10 – 20 cm diameter) to an approximate depth of 0.3 meters. Rock was staged in the town of Suwannee, and barged to the site using a 23 m long barge equipped with spuds and moved by pushboat. The rock was placed onsite using a 13.7 m articulated hydraulic arm with bucket during September 2013. In practice, the hydraulic arm was unable to achieve a completely even surface, and there was considerable difference in even-ness of the surface created (see *Results*). The use of the arm also created linear "arcs" or rows of higher and lower elevation surface on restored sites. These were not considered undesirable, since natural reefs also have considerable variation in elevation.

Following rock placement, the four edges of all restoration sites were lined with clam aquaculture bags that contained live oysters, shell, and associated fauna. These clam bags were taken from clam aquaculture leases ("Gulf Jackson") approximately 10 km south of the middle of the Lone Cabbage chain, usually removed from the lease and placed on Lone Cabbage on the same tidal cycle. These bags (0.3 X 0.3 X 0.2 m) were full to varying degrees, with some packed tight, and others no more than half full. Contents of bags varied too, with most containing predominantly dead clam shell, encrusted to varying degrees with live oysters and other fouling organisms. Bags were laid against the edge of the rock next to one another, and tied to their neighboring bags at each corner using large cable ties to ensure a cohesive mass. In practice, we found approximately 25 bags were necessary to complete each side of a square.

**Monitoring Methods**

*Elevation Determinations.*

Benchmarks were created for each site by first establishing a local area benchmark using a base-rover pair of survey-grade gps units (Magellan Mark V). The base was allowed to record over the closest surveyed USGS benchmark approximately 8 km distant for 3.2 hours while the rover recorded over the benchmark on the reef; the solution for the reef benchmark showed a mean estimated vertical error of 1.8 cm. From this benchmark on the reef, we then used a different base-rover pair (Topcon HiPer Lite+ GPS receiver, TDS Nomad data collector) in bluetooth enabled real-time kinematic mode to establish benchmarks on the other reefs in the chain. At each benchmark, we drove a 15.2 cm OD X 91 cm PVC pipe into the ground using a sledge hammer so that its end was nearly level with the ground surface, and poured concrete aggregate mix into the pipe. A capped 12 mm diameter X 20 cm copper pipe or piece of iron rebar was then set into the concrete to form the measurement point of the benchmark.

Elevations of other points on each reef site were determined using a laser level with marked pole. We measured elevations relative to the reef benchmark, the four corners of each site, and every 5 m along each of the transects established (above).

*Oyster density and size.*

Oyster density and size sampling was performed at each site during low tides pre-construction (low tides on April 23 – 24, 2013) and at two times post-construction in fall/winter 2014/5 (October 27 and 28, November 11, 5 December 2014, January 21 2015) and early summer 2015 (May 16 – 18, May 31 - June 1). Post-construction comparisons below refer to a comparison between preconstruction and early summer 2015.

Oyster density: on each sampling date at each site, we counted live and dead oysters within temporary belt transects marked using stakes and string. Each transect was 15.4 cm wide and as long as the site footprint (generally 21 m). We counted within three transects that were spaced evenly across the square (ends were 5.34, 10.68 and 16.02 meters from northeast and

southeast corners), and were oriented parallel to the east and west sides of the squares, which is also along a gradient moving away from the inlet.  Transects were formed by placing vertical "H" shaped iron stakes (providing attachment points for strings)  at each end of a transect, and running tightened nylon string between the corners so that a 15.4 cm wide belt transect was formed approximately 5 – 10 cm above the substrate.  Metric tape measures were laid along these transects with 0 at the farthest point from each inlet.  Using click-counters, live and dead oysters were then counted within each 2.5 m section of the transect.  Dead oysters were defined as having two valves that were clearly open with no evidence of a living oyster within.  All size classes were counted including live and dead spat. Only oysters that could be distinguished visually from above or to the side were counted: oysters on the undersides of clumps were not counted, and clumps were not picked up.  The temporary belt transects were removed after each sampling event.

Oyster size:  Using the same transects above, we measured sizes of oysters within randomly placed 0.0625 m2 portable quadrats (25X25 cm within the quadrat shape) formed out of pvc pipe. We generated random numbers (with replacement) for the location of quadrat placement both along and normal to the belt transects.  Distance along transects varied from 0 – 21 m in 1.0 m increments and distance from the transect varied 0 – 4 m, in 0.1 m increments. Quadrats were placed with the corner closest to the transect on the outside edge of the measured distance, and were oriented with the far edge closest to the inlet.  All live and dead oysters within the quadrat were measured using either Vernier calipers or a simple sewing gauge to the nearest mm, from umbo to point of longest dimension.  Once 50 oysters had been measured within any quadrat, all other oysters were counted but not measured so that a total density estimate could also be derived from the quadrats. We placed enough quadrats to equal either 50 total oysters measured, or 20 total quadrats for each transect on each site.

*Sampling oysters in clam bags*:  Clam bags from aquaculture leases are usually highly fouled, and it is nearly impossible to count or measure oysters in the bags without cutting the bags open. For this reason, clam bags were not sampled for density or size on the regular transects that we describe above.  On the last sampling (late May/early June 2015) we sampled oyster density and size within clam bags on bags that had been on the reef for at least 9 months. We cut open the upward facing side so that a 0.0625 m2 quadrat could be placed on top of the contents.  We used the same methods for counting density and measuring size as above under *Oyster Size*.  We chose clam bags to sample by randomly selecting a side (1 – 4 with replacement), and then randomly selecting a number of clam bags (0 – 25 with replacement) from each corner to sample.  Clam bags were stitched closed following sampling using plastic cable ties.

## Bird sampling methods

We used trail cameras in time-lapse mode to estimate usage of each of the sites by aquatic birds. Cameras were placed 2.5 m above the substrate within a weatherproof housing mounted on 15.2 cm diameter PVC poles driven into the substrate (Figure 7). Cameras were deployed in fall/winter 2013/4 following placement of the rock, between 22 October and 17 December. This period coincides with the maximum period of both fall migration and of winter residence for most of the aquatic birds in Gulf coastal Florida.

Figure 7. Trail camera mounted on PVC pole aimed at restoration site 2 on Lone Cabbage Reef, December 2013.

Each of 8 cameras (1 per site) deployed simultaneously was oriented to face either northeast or southwest along the axis of the reef chain depending on the site to be viewed; all cameras were oriented to face towards the inlet in question. Camera poles were placed far enough away from the edge of each study site to be able to clearly see all four corners of each site. We used Bushnell Trophycam 119467 trail cameras which were outfitted with solar charging panels. Cameras were set to take pictures once every 5 minutes with motion detection turned off, and daytime operation only. Cameras were variously reliable, with usable pictures of birds from each camera from between 1 and 68 days during this time period depending on the site and camera. Batteries and SD cards were changed once during this period.

We only counted images that had some substrate showing as the tide receded, and did not count images that were too dark to detect birds or for which the lense was occluded by precipitation or fog. We visually scanned both for images of birds, and for apparent movement as we toggled among adjacent images. Birds were identified to species if possible, and if not were lumped into categories of pelicans, gulls, cormorants, shorebirds (small, medium and large), terns, ducks and raptors. We expressed usage as numbers or proportions of frames the birds were noted in.

## Crab sampling methods.

The study site is generally estuarine in character and provides good habitat conditions for blue crabs (*Callinectes sapidus*, Figure 8). We often saw blue crabs on the oyster restoration sites and noted that the embayment landward of Lone Cabbage reef has a high density of blue crab traps. Since blue crabs are a typical predator on oyster reefs, we measured crab density on restored and control sites.

We used trail cameras to estimate density of blue crabs. Up to ten trail cameras (Bushnell Trophycam 119467) were placed at random locations within the footprint of each site studied at

2.3 meters above the substrate on 5.4 cm diameter aluminum poles set into 10.1 cm diameter pvc pipe driven into the substrate (Figure 9). Cameras were placed on falling tides as the water



Figure 8. Blue crab (*Callinectes sapidus*) in refuge between two clam bags.

receded beginning at approximately 0.3 m depth over the reefs on 31 May (sites 2, 3) and 1 June 2015 (sites 1, 4) at two restoration sites (1, 2) and two controls (3, 4). Counting of crabs ceased when the water coverage in the frame decreased to 20%. Cameras were set on each reef for one low tide in late afternoon/evening.



Figure 9. Cameras mounted on PVC poles approximately 1 meter above the reef for estimating crab densities.

Cameras were mounted to point downwards parallel to the substrate, and were set to take 10-s long videos once per minute. The area seen within the frame of the trail cameras averaged 0.997 m$^2$.

Recordings were analyzed by playing each clip and scanning for crab outlines and crab movement. All evidence of crabs of any species was recorded, and we distinguished between blue crabs, stone crabs, and fiddler crabs. Although we did not do a sensitivity analysis, we suspect strong bias towards seeing larger crabs (>3 cm) and bias towards seeing moving crabs.

**Salinity sampling**.

Because we were interested in the potential effect of the reef on salinities we deployed six sensors in landward and seaward pairs (Figure 10).

Figure 10. Locations of salinity recording instruments seaward and landward sides of Lone Cabbage reef.

Each station consisted of a conductivity-temperature-depth (CTD) sensor (CTD-Diver, Schlumberger Water Services, Tucson, AZ) logging at 15-minute intervals. Sensors were inserted into a slotted PVC housing mounted via rebar to a cement block and placed directly on the benthic surface. Stations were placed to keep sensors submerged at low tide (30-45 cm above the benthic surface) and were located between 5 and 30 m from the reef edge; the seaward reef slope is shallower than the landward side, requiring station placement at a greater distance. Data were collected from November 2013 though June 2015; sensors were downloaded and housings cleaned every 2-4 months. Daily, tide-filtered discharge data were downloaded from the USGS Gopher River station on the Suwannee River (station ID: 02323592).
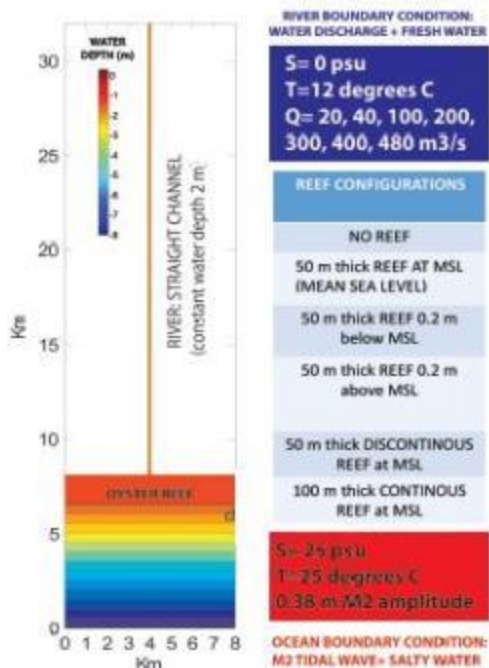


Figure 11. Schematics of domain configurations (left) and topographical/hydrographical conditions modeled to understand the effect of reef chains on detention of freshwater in Suwanee Sound.

We also modeled the potential effect of the Lone Cabbage chain of reefs on freshwater detention within Suwannee Sound by modeling salinities. We used the 3-D Regional Ocean Modeling System (ROMS) which is a numerical, hydro-dynamic model that solves finite-difference approximations of the Reynolds-Averaged Navier-Stokes (RANS) equations using the hydrostatic and Boussinesq assumptions (Chassignet et al., 2000; Haidvogel et al., 2000), with an explicit time stepping algorithm (Shchepetkin and McWilliams, 2005; Haidvogel et al., 2008). Domain configurations (Figure 11, left) represented an idealized morphology based on Suwannee Sound. We compared hydrodynamics resulting from the interaction of different water discharges (Q) and oyster reef configurations (Figure 11 right). The basic idea behind this relatively crude modeling effort was to see if there was any support for the hypothesis that restored elevations would lead to greater detention of fresh water.

## Results

**Permitting**

We obtained permits for restoration construction work on the Lone Cabbage reef from U.S. Army Corps of Engineers, Florida Department of Environmental Protection, and Florida Fish and Wildlife Conservation Commission (Special Activity License). In addition, we also received a permit for the installation of a private aid to navigation from the U.S. Coast Guard in order to post signage marking the restored oyster reef sites.

The application to USACE and FDEP was joint under the Consolidated Environmental Resource Permit process for generalized construction, and was first submitted in September 2011, with the Suwannee River Water Management District as the applicant. Soon thereafter, we received many comments requiring revision from FDEP, and the Suwannee River Water Management District pulled back support for the project. These two actions were unrelated – the lack of support from the SRWMD was due to statewide policy changes in the Water Management District. While responding to the FDEP comments, we realized that our elevation surveys were inaccurate, particularly in the vertical element, requiring complete resurvey effort using different methods. For this reason, on the advice of FDEP the permit request was withdrawn, and resubmitted in September 2012 to both FDEP and USACE. A permit was granted by FDEP in January 2013 (6 months from resubmission).

The USACE permit request was more laborious because of federally listed species that could be impacted by the project – specifically sea turtles (Kemp's Ridley *Lepidochelys kempii*, Loggerhead *Caretta caretta* and Green *Chelonia mydas*) that make heavy use of seagrass meadows and oyster reefs in nearshore waters, and the Suwannee River population of the Gulf Sturgeon *Acipenser oxyrinchus desotoi*. The mouths of the Suwannee River and the entire area of the Lone Cabbage restoration area are in fact designated as critical habitat for the Gulf Sturgeon. This triggered partial ESA review by the National Atmospheric and Oceanic Administration, which delayed the granting of the permit by many months. In the end, there was a determination of no negative effect, and likely positive effect of oyster reef restoration for these species. The USACE permit was granted on 30 August 2013 (13 months from resubmission).

The Special Activity License from the Florida Fish and Wildlife Conservation Commission was necessary as authorization to move living marine resources (oysters and associated fauna within clam bags) from clam aquaculture leases to the Lone Cabbage site.

**Construction: costs, progress and effectiveness**

        Preconstruction activities included surveying reefs, mapping out and marking construction and control sites, measuring preconstruction oyster densities and reef elevations, placing construction and warning signs, and removing live oysters from construction sites. These activities were carried out in November 2012 – April 2013, with oyster density estimation during April 2013 (Figure 12). Preconstruction reef characteristics can be found below under Effects of Restoration Activities. In general, oyster densities on all sites preconstruction were very low. We removed 15, 4, 0, and 2 nineteen-liter pickle buckets full of oysters from sites 1, 2, 5, and 6 on 10 September 2013. These oysters came from the footprints of the sites, suggesting 0 - .65 liters/$m^2$ on these sites preconstruction.



Figure 12. Restoration site 5 looking south, April 2013 prior to restoration activities. Note the complete absence of typical oyster reef structure, and very low density of oysters.

        Elevation surveys were carried out in spring and summer 2012 using long term recording with a single survey grade gps unit at benchmark sites and solutions derived from permanent Continuously Operating Reference Stations maintained by USGS. While this method works well for sites that are within 20 km of CORS sites, we found the nearest CORS stations to Lone Cabbage were 20 – 45 km distant. Consequently even with long recording times (3+ hours), we had generally inconsistent solutions and poor confidence in the vertical component of positions (eg, 8 – 15 cm). For this reason, we re-surveyed all benchmarks in spring and summer 2013 using long term (2 – 3 hour) recording with the Base unit on a known USGS monument (Levy 45), approximately 8 km from our site, and the rover unit on the benchmark. These solutions were much more reliable and accurate, with an estimated average error for the vertical component of 0.5 – 1.5 cm.

During spring 2013 we also placed signage on the site based on permit requirements and USCG marking permits.  At each constructed site we placed one aluminum sign advising boaters and construction crew to be aware of the possible presence of manatees, and one aluminum sign with the words "Danger – Oyster Bar" in large letters.  These were pounded into the reef substrate to a depth of 1 meter.  All were blown down or knocked down by wave action during fall 2013.  All signs were then placed on longer sections of pvc pipe and pounded to 2.5 meters in the substrate. All signs are still up at the time of writing.

*Rock placement:* To move rock to the site, we employed D&L Construction in Suwannee Florida.  This firm specializes in marine construction and had constructed several offshore fishing reefs in the past. This firm also was the only one on this sparsely inhabited coast that had a barge and other equipment necessary for moving large volumes of rock – there were no contractors in Cedar Key or Yankeetown with these capabilities and no firms in the closest metropolitan area (Tampa) willing to move a barge to the area (distance approximately 430 km). We used limerock ("surge rock") obtained from a local quarry that was 10 – 20 cm in



Figure 13. Limerock boulders at low tide on restoration site 1 in the Lone Cabbage reef, approximately 1 week after placement.

diameter (Figures 13 – 15), with no fine material.  Because we had some short term funding external to this project to devote to this, we purchased the rock and paid for trucking to, and storage at the town of Suwannee in December 2012. Since this was well prior to obtaining construction permits, the material was stored in Suwannee from December 2012 through September 2013.  The rock was then moved by barge to the construction site in late September 2013 after obtaining all construction permits. The barge was moved into place at high tide, and stabilized in place using hydraulic spuds that

Figure 14. View of restoration site #1 from south looking north, 3 months following rock placement.

anchored it to the bottom. Rock was offloaded using a hydraulic arm and bucket. Although a reasonably uniform distribution of rock was achieved within the footprint, there were obvious high and low spots, often in lines or arcs parallel with the radius of the hydraulic arm. All rock deposition took place over the course of two weeks in late September 2013 (see Figure 3). Rock protruded above the parent sediment by 10 – 40 cm at that time.

    We placed a total of 471 yards limestone rock on the four restoration reefs at a cost of $142/yard or a total of $67,184. This cost is somewhat idiosyncratic for several reasons. First, there was only a single contractor in this region that routinely conducted this type of marine construction work so no other bids were provided from experienced vendors. We did receive one other bid for the job from a contractor that did not typically do marine work - their quote was $238,420 or nearly four

Figure 15. Rock on Lone Cabbage reef restoration site 1 approximately 1 month following deposition.



times the cost we obtained. Second, because of delays in permits, we were forced to pay for the rock installation in two phases: rock purchase, trucking and storage (Phase I) and rock placement on the reef via barge (Phase II). Because of the delay in permitting we had to store the rock at the contractor's site for 9 months. Because trucking, rock purchase and storage were all bundled into one price it was difficult to derive the cost of delivery to the reef without storage.

    The cost of rock placement was much higher than we had originally estimated (70% more), largely due to the fact that no one had ever barged material in these quantities in this

region, and the true costs of such work were mostly speculative.  We were forced to pay for the unexpected increase in cost by sacrificing all salary dollars in the budget.

Despite the categorical cost overrun, the quality of the job was very good and the work was done in a timely fashion, allowing the rock to be in the water during the typical time of fall spat set.

*Clam bag placement:* We contracted with the Cedar Key Aquaculture Association (CKAA) to move derelict clam bags from aquaculture sites on the Gulf Jackson site west of Cedar Key to the Lone Cabbage sites.  Total cost of moving and placing each bag was set in the purchase order at $32/bag.  The mover also received an additional $3/bag from the leaseholder to remove each bag – so movers received a total of $35/bag.  For comparison, a leaseholder selling the contents of a fully grown-out crop of clams typically receives about $100/bag for the product, and still must pay the costs of seed, planting, harvest, boat and fuel costs and labor.  The mover must only pay labor and fuel. The $35/bag was generally perceived in the local aquaculture community as being an attractive incentive for moving bags.

Removing and transporting a derelict bag is not easy, since the bag may weigh up to 135 kg, the bag itself may be partly rotten, and is probably at least halfway buried in mud and sand.  Past experience has shown that bag movers need to have the proper equipment, including a large enough boat, an experienced crew, a roller rig or boom to get the bags off the bottom and a high volume pump to hose out the large amounts of mud and sediment typically in half-buried bags.  For partially decomposed bags, the bag must be rolled onto a wire basket or pulled up using a hydraulic grabbing arm.  Individual movers must also be trusted by individual lease holders, since there may otherwise be damage to live crop or removal of the wrong bags.  For these reasons, the CKAA decided in fall 2013 to move forward only with approved bag movers from another earlier project.

The first approved mover was equipped with a large barge-like boat capable of carrying 40 – 50 bags at a time, and an airboat equipped with spuds and a hydraulic arm for grabbing clam bags.  While this appeared to be a well-equipped and trained operator, this contractor only worked during fall 2013 and moved only 78 bags, and despite many promises did not ever resume work. Reasons for work stoppage seemed to be personal rather than related to the work itself.

The second approved mover began work in summer 2014, and moved over 200 bags during summer 2014.  However, this contractor began having progressive health issues and despite many promises finally abandoned the work entirely during winter 2014/15.

At this point all other approved movers had either moved out of town, did not have the appropriate equipment or were unwilling to do the work. During winter 2014/5, oyster prices had nearly doubled compared to the year before, and clam prices increased as well. Nearly all of the local fishermen and aquaculturists were therefore incentivized to work for themselves, and we were not able to locate another mover until the end of oyster season in May 2015.  This last contractor moved the rest of the bags, but many of the bags moved in June had not been derelict for more than 6 months, and so were not encrusted and contained mostly dead clam shell.  As a result, many of these bags were less than 1/16th full, and did not have enough material to keep them on the reef during wave action.   This resulted from a miscommunication – movers had not

been given a minimum size or weight for bags, and we had assumed they would only move full bags. The mover eventually agreed to coalesce the bags into a smaller number, but we were also forced to pay for more bags to make up the difference in coverage.

For these reasons, we paid for many more bags than originally estimated (407 vs. 278 estimated), which drove the total costs up considerably ($13,051 vs. $8,901). The difference was made up from both the salary category in this grant, and from discretionary University of Florida monies under the supervision of the PI. Ultimately this difference is one that arose from mis-estimating how much space the clam bags actually take up – we had calculated based on completely full bags, and the reality was smaller and oddly shaped bags took up smaller space (46% less). Our previous experiences with clam bags at the Atsena Otie site had focused on making vertical structure out of clam bags rather than linear, and in general, we had also found much fuller derelict clam bags during that time (2005 – 2007).

Overall, the construction of the reefs totaled $80,235, and the constructed reefs totaled 455 square meters, resulting in a cost of $176.25/square meter or $16.37/ square foot. Note that this includes the cost of rock materials but not the estimated match cost of clam bag materials ($80,753 or $198/bag). This summary also does not include the preconstruction costs in boat time and personnel, nor the costs of contracting, billing, permitting or inspection.

**Effects of restoration on oyster reefs and associated fauna.**

*Reef colonization and pattern of oyster growth*: The rock portion of the reef was installed in late September 2013, and was rapidly colonized by a near-monoculture of barnacles (Figure 16).



Figure 16. Limestone boulder covered with barnacles, fall 2013 on the Lone Cabbage reef.

The barnacle spat set at that time was remarked upon in Cedar Key by aquaculturists and hatchery personnel, and the set may have been a particularly large or synchronous one. Though we did not measure densities, the rocks appeared to have been covered 100% by barnacles, and there was literally little or no space for oysters to colonize. This situation persisted throughout the winter of 2013/2014.



Figure 17. Small oysters growing together over a gap between rocks. Note that the seam between rocks has become invisible.

The winter/spring of 2013/2014 was relatively wet, with the Suwannee River exceeding flood stage for several months. Discharge from the Suwannee River kept the reef under nearly continuously fresh conditions for major parts of each day for weeks at a time (Figure 18). As a result, nearly all of the barnacles had died by late spring 2014 though their shells encrusted most of the surface area of the rock. In addition, we began seeing oyster spat on the rocks, though the distribution seemed very spotty spatially, and consistently zoned in the vertical plane. By mid-summer 2014, we saw many small oysters on the tops of the rock, and many larger oysters at the very base of the rocks, with low numbers in between. We hypothesized that this pattern was created by the high densities of crabs that had also colonized the site (see below) – the crabs were presumably not able to feed on the lowest oysters because they were protected in rock crevices, and were not able to feed on the high oysters because it would expose the crabs to predation during higher water. In between, the crabs may have been relatively protected and could have devoured most of the oyster spat in that zone.

Sampling in fall 2014 suggested a growing population of mostly small oysters (<30 mm) on the rocky portion of the restored reefs, with many oyster beginning to span and even cement-over the spaces between rocks (Figure 17).

The winter of 2014/15 was normal to wet with persistent rainfall and discharge keeping the reef under fresh to estuarine conditions for nearly the entire time, and periods of weeks when the average PSU was 5 or less (Figure 19).
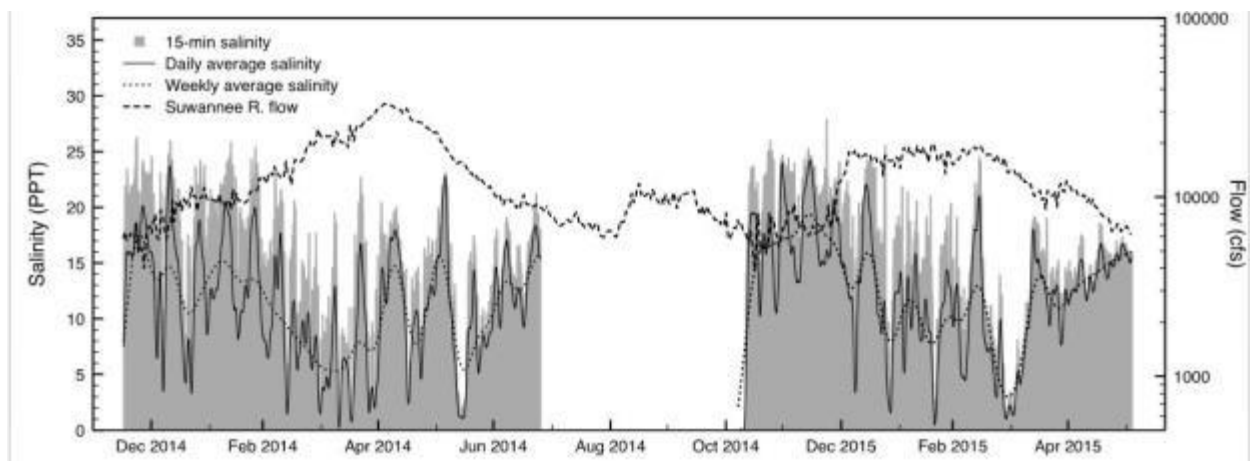
Figure 18. Graph of Suwannee River Discharge (dashed line, Gopher River USGS station ID: 02323592) and average weekly and daily salinities at a station on the seaward side of restoration site 2 on Lone Cabbage reef. The salinity sensor was not in the water during July – October 2014.

*Oyster density and size:*

Figure 19 and Table 1 show total densities of oysters counted on rock on belt transects across the eight sites preconstruction (April 2013) and post-construction (May/June 2015). In general, there was an increase in oyster density on the control oyster reefs (increase of $10 - 43$ oysters/m$^2$ or $0 - 64X$ increase) between these dates, probably because the salinity conditions had improved compared with relatively saline conditions in $2011 - 2013$. However, oyster densities on rock on restoration sites increased much more, with absolute increases of $107 - 199$ oysters/m$^2$ or $15 - 157X$.
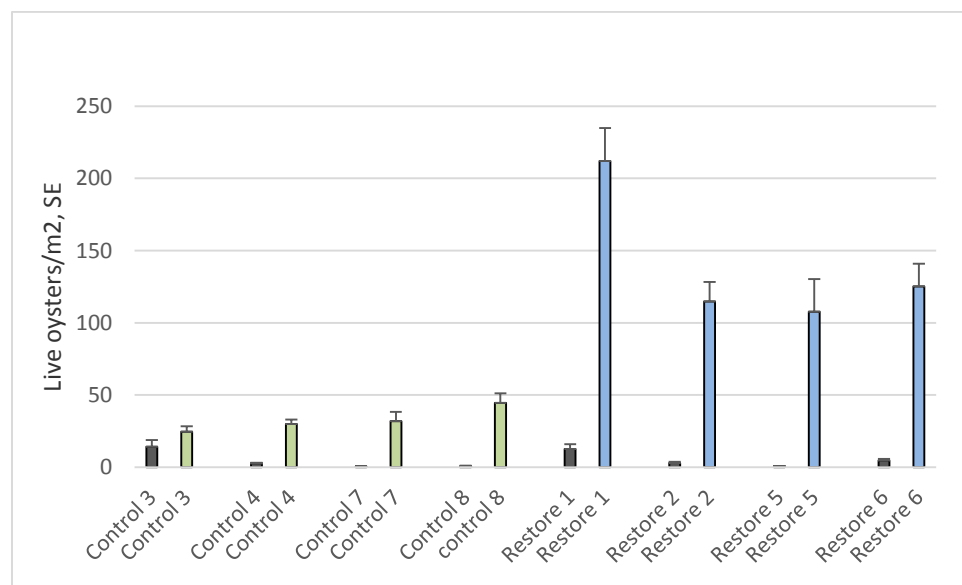


Figure 19. Densities of live oysters + SE counted in transects preconstruction (black, mostly not visible) and postconstruction on eight restored (blue) or control (green) oyster reefs.

Table 1.  Oyster densities on rock substrate on control and restoration sites 5 months before (April 2013) and 19 months after (May/June 2015) construction activities.

| Sample date | site | oysters/m2 | s.d. | SE |
|---|---|---|---|---|
| 4/24/2013 | Control 3 | 14.2 | 24.21 | 4.66 |
| May/June 2015 | Control 3 | 24.7 | 19.54 | 3.76 |
| | | | | |
| 4/24/2013 | Control 4 | 2.6 | 3.19 | 0.61 |
| May/June 2015 | Control 4 | 29.8 | 16.59 | 3.19 |
| | | | | |
| 4/24/2013 | Control 7 | 0.5 | 1.44 | 0.28 |
| May/June 2015 | Control 7 | 31.9 | 33.78 | 6.50 |
| | | | | |
| 4/24/2013 | Control 8 | 0.8 | 1.40 | 0.27 |
| May/June 2015 | control 8 | 44.6 | 34.18 | 6.58 |
| | | | | |
| 4/24/2013 | Restore 1 | 12.7 | 17.36 | 3.34 |
| May/June 2015 | Restore 1 | 212.0 | 118.52 | 22.81 |
| | | | | |
| 4/24/2013 | Restore 2 | 3.0 | 4.39 | 0.84 |
| May/June 2015 | Restore 2 | 114.7 | 70.72 | 13.61 |
| | | | | |
| 4/24/2013 | Restore 5 | 0.7 | 1.53 | 0.30 |
| May/June 2015 | Restore 5 | 107.8 | 116.68 | 22.46 |
| | | | | |
| 4/24/2013 | Restore 6 | 4.8 | 5.39 | 1.04 |
| May/June 2015 | Restore 6 | 125.2 | 81.41 | 15.67 |
| | | | | |
| May/June 2015 | clam bags | 474.0 | | 48.31 |

We also measured oyster densities and sizes in a total of 43 clam bags that had been in place for at least 12 months, and found an average of 474 oysters/m² (s.d.= 19.8).  For sizes of oysters in clam bags see below under *Oyster Size*.  Although there were some live oysters in the clam bags when installed, we believe the majority of the oysters were recruited in place – they were surprisingly uniform in size and shape suggesting a cohort effect, and oysters generally had not been evident during installation of the bags (Figures 20,21).



Figure 20.  Photo of clam bag cut open to reveal high densities of large oysters.  Compare densities with rock immediately behind.

Figure 21. Clam bags as placed on the edge of rock on restoration sites, Lone Cabbage reef.

Clam bags had consistently higher densities of oysters than did rock on restored sites. We estimated total oysters on rock and clam bags reefs based on average densities and areas (Figure 22). Reefs had on average 25 bags per side, or 100 bags total, with an estimated bag surface area of 1.46 m$^2$. The rock on restoration sites was estimated to contain between 49,076 and 96,514 oysters total, while the clam bags on a single reef were estimated to contain 70,159 oysters. If this math is correct, then using an average of the rock on the four sites (63,701 oysters), clam bags contributed 52.4% of the individual oysters on the reef.

Estimated total densities (rock + bags, calculated from total oyster estimates) averaged 303.5 oysters/m$^2$, while oysters on control sites in June 2015 averaged 32.8 oyster/m$^2$. Thus oyster densities on restoration sites were 9.26x densities on control sites.
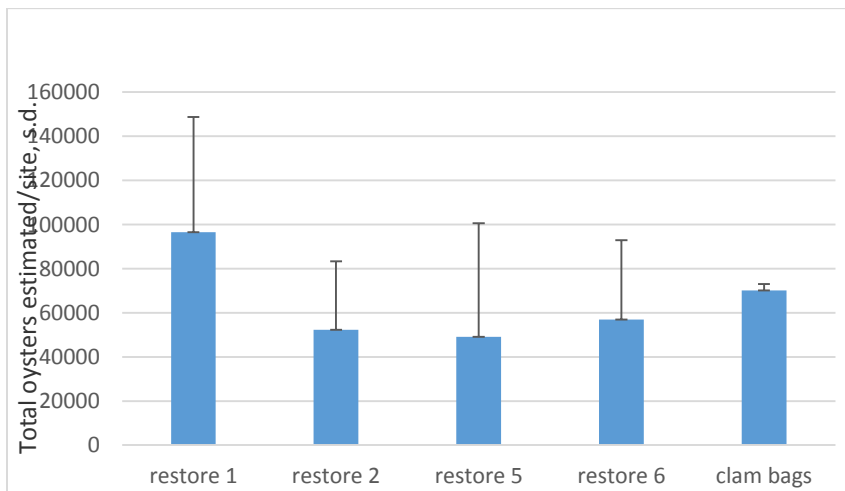


Figure 22. Total oysters estimated on each site for rock (restore 1, 2, 5, 6) and clam bags that had been in place for at least 12 months. Clam bags contributed 52.4% of the oysters on restored sites that had at least 12-month old clam bags on them.
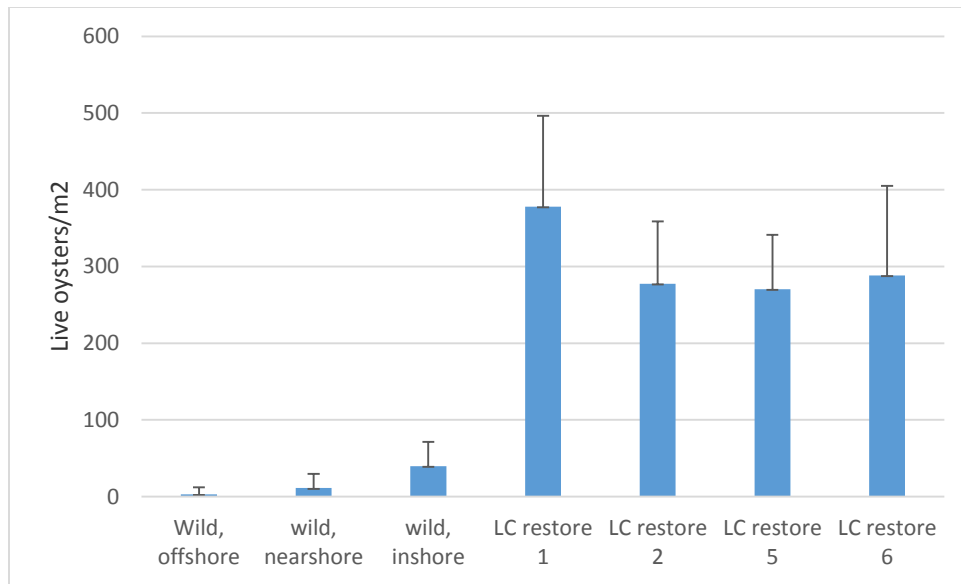
Figure 23. Mean oyster densities on restored sites at the Lone Cabbage reef with densities collected from 38 oyster reefs in the area between Wacasassa Bay and Horseshoe Beach collected in 2010, representing a gradient of conditions from offshore to inshore locations.

Figure 23 compares densities on the restored sites at Lone Cabbage, with mean densities at wild oyster reefs in the surrounding area (Wacassassa Bay to Horseshoe Beach). Inshore reef densities are highest ($39.9/m^2$), and offshore reef densities lowest ($3.01/m^2$, Seavey et al. 2011), with offshore reefs being edaphically most comparable to the Lone Cabbage sites. Densities on restored sections of the LC reef were between 89 and 125 times higher than the other offshore sites, and were also between 7 and 9 times higher than the inshore reef densities.

Figure 24 compares these densities with those at other restored and natural sites around the Gulf of Mexico (LaPeyre et al. 2014). Oyster densities in clam bags on restored sites at Lone Cabbage were higher than all but one of the reported sites (a natural site in Texas). Oyster densities on any of the Lone Cabbage restored sites was higher than 89% of all the sites reported. These densities are not directly comparable since the LaPeyre sites were all subtidal and measured using diver-recovered quadrats counted in the lab, whereas the LC sites were counted in situ without disturbance and we only counted oysters we could see from above. Further, the LaPeyre sites were of various ages post-restoration whereas the LC sites were 19 months post-restoration. Presumably, the LaPeyre counting method is likely to produce an upwards bias in density compared with the method used at Lone Cabbage because more of the oysters are unearthed and counted. It is unclear whether one should expect higher densities at sites with longer postrestoration histories or not, since weather events and harvest could easily depopulate those sites over time. Further, early colonization sites like Lone Cabbage may have generally small, crowded oysters, and competition may result in lower densities as oysters grow. Nonetheless, the densities found at LC appeared to fulfill a designation of high densities of oysters compared with other restoration sites in the Gulf of Mexico, and many natural sites in the Big Bend study area.
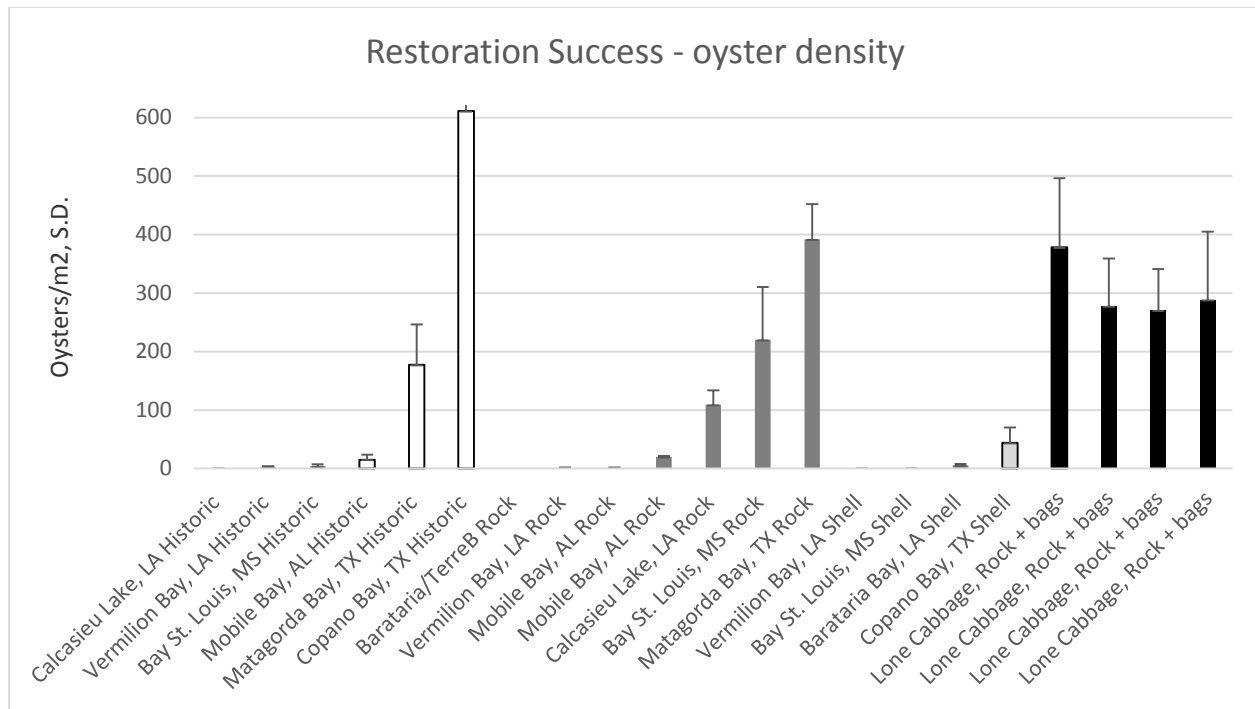
Figure 24. Oyster densities at restored and historic sites in the Gulf of Mexico (from LaPeyre et al. 2014). Historic sites are shown in white bars, sites restored with rock are shown in dark gray, sites restored with shell are shown in light gray, and the four Lone Cabbage restoration reef sites are shown in black.

*Oyster size:* We found less consistent differences by treatment and time in sizes of oysters measured at the sites at Lone Cabbage. At one of the restoration sites, only a single live oyster fell within our quadrats in the preconstruction and so we were unable to derive an average size. In the three restoration sites at which comparisons could be made pre-and post-construction, all three sites had significantly larger oysters post-restoration. Of the control sites, two were not usable due to small sample size in pre- or post-construction condition. One of the two remaining sites one (control 3) had significantly larger mean oyster size preconstruction, and the other (Control 4) had significantly smaller oysters post-construction (Table 2, Figure 25).

During the final sampling in May/June 2015, rock on restoration sites had significantly larger oysters (mean 33.9 mm, n = 819, s.d. = 12.67, p<.01) than did control sites (mean 27.4 mm, n = 361, s.d. = 8.46, p<.01), though the difference in mean size was not large (6.5mm, Table 2). Clam bags also had the largest mean oyster size of any treatment or substrate (mean 39.43mm, s.d. 13.94, n = 1,306) and were significantly larger than oysters on control sites (t-test, p<<.001).

Table 2. Mean sizes of oysters by treatment and date sampled at Lone Cabbage Reef.

| Date Sampled | Site and treatment | Mean (mm) | S.D. | N oysters | SE |
|---|---|---|---|---|---|
| 4/23/2013 | control 3 | 36.2 | 11.89 | 20.0 | 2.66 |
| 5/16/2015 | control 3 | 28.9 | 9.79 | 100.0 | 0.98 |
| | | | | | |
| 4/23/2013 | control 4 | 34.5 | 6.66 | 6.0 | 2.72 |
| 5/17/2015 | control 4 | 27.2 | 8.53 | 123.0 | 0.77 |
| | | | | | |
| 4/24/2013 | control 7 | 23.1 | 0.00 | 1.0 | 0.00 |
| 5/31/2015 | control 7 | 25.9 | 6.82 | 138.0 | 0.58 |
| | | | | | |
| 5/31/2015 | control 8 | 27.7 | 6.02 | 143.0 | 0.50 |
| | | | | | |
| 4/23/2013 | restore 1 | 32.2 | 13.02 | 20.0 | 2.91 |
| 5/17/2015 | restore 1 | 38.8 | 14.40 | 156.0 | 1.15 |
| | | | | | |
| 4/23/2013 | restore 2 | 27.5 | 8.51 | 9.0 | 2.84 |
| 5/16/2015 | restore 2 | 34.8 | 11.75 | 149.0 | 0.96 |
| | | | | | |
| 4/24/2013 | restore 5 | 37.7 | 0.00 | 1.0 | 0.00 |
| 5/18/2015 | restore 5 | 30.0 | 9.32 | 155.0 | 0.75 |
| | | | | | |
| 4/24/2013 | restore 6 | 29.7 | 7.16 | 4.0 | 3.58 |
| 6/1/2015 | restore 6 | 36.3 | 14.59 | 227 | 0.97 |
| | | | | | |
| 5/18/2015 | clam bags | 39.1 | 13.94 | 1,306 | 0.39 |
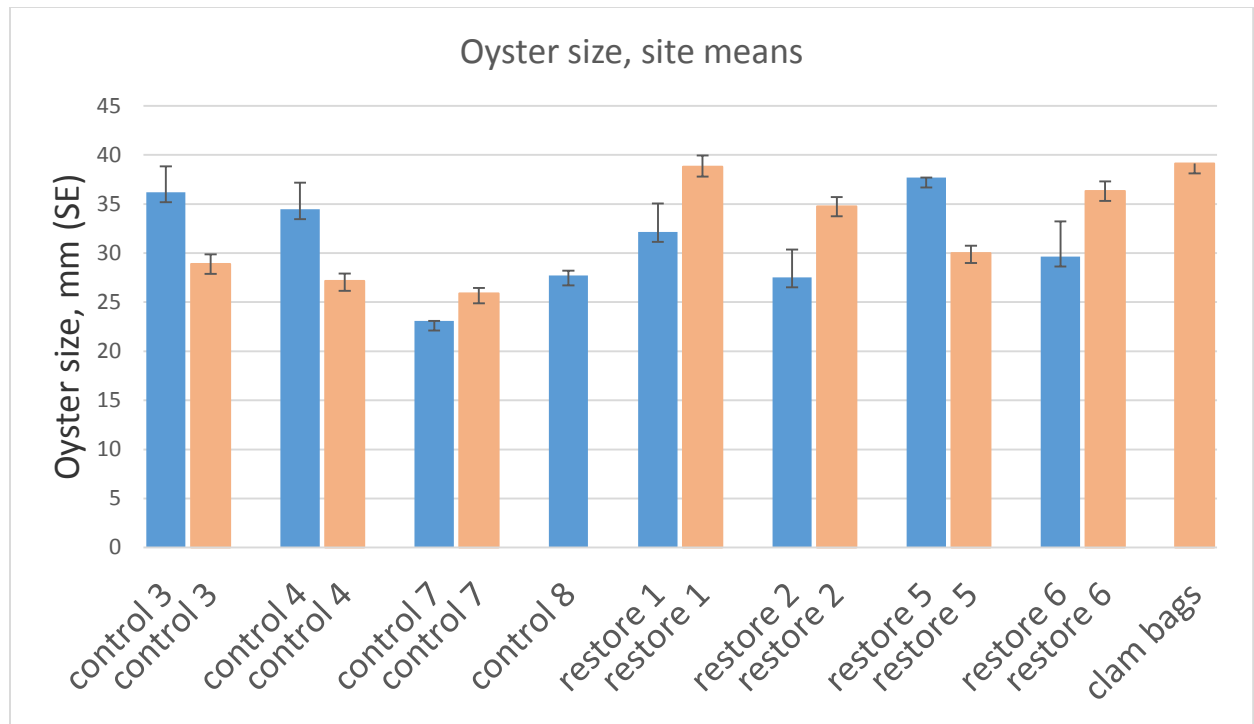
Figure 25. Mean +SE sizes of oysters by site found on control and restoration sites before (blue, April 2014) and after (orange bars, May/June 2015) restoration activities.

*Effects of restoration activities on reef elevation:*
Placement of rock and bags on the restoration sites by definition increased elevation on those sites initially but it was unclear whether rock and bags would settle into the substrate. Figure 26 shows before and after elevations on the sites that were restored or not. Restoration sites showed average absolute increases in elevation across the reef of 16.05 cm (range -.51 – 32.6 cm) whereas mean elevation change on control sites was 3.1 cm (range -18.7 – 14.8). It is difficult to tell how much of this difference was due to placement of rock (possibly followed by some settling), and how much was accretion of sediment onto the rock. Patterns of sediment accretion were difficult to portray with the relatively coarse grid of measurements we took, but it appeared that sediment either accreted slightly or generally decreased in elevation on control sites, and accreted considerably on the restored sites (see Figure 27). This was confirmed to some extent by our difficulty in finding two of our benchmarks on the restored reefs (buried under 8 – 10 cm sediment), whereas the benchmarks on the control sites were all clearly elevated above the surface of the sediment.
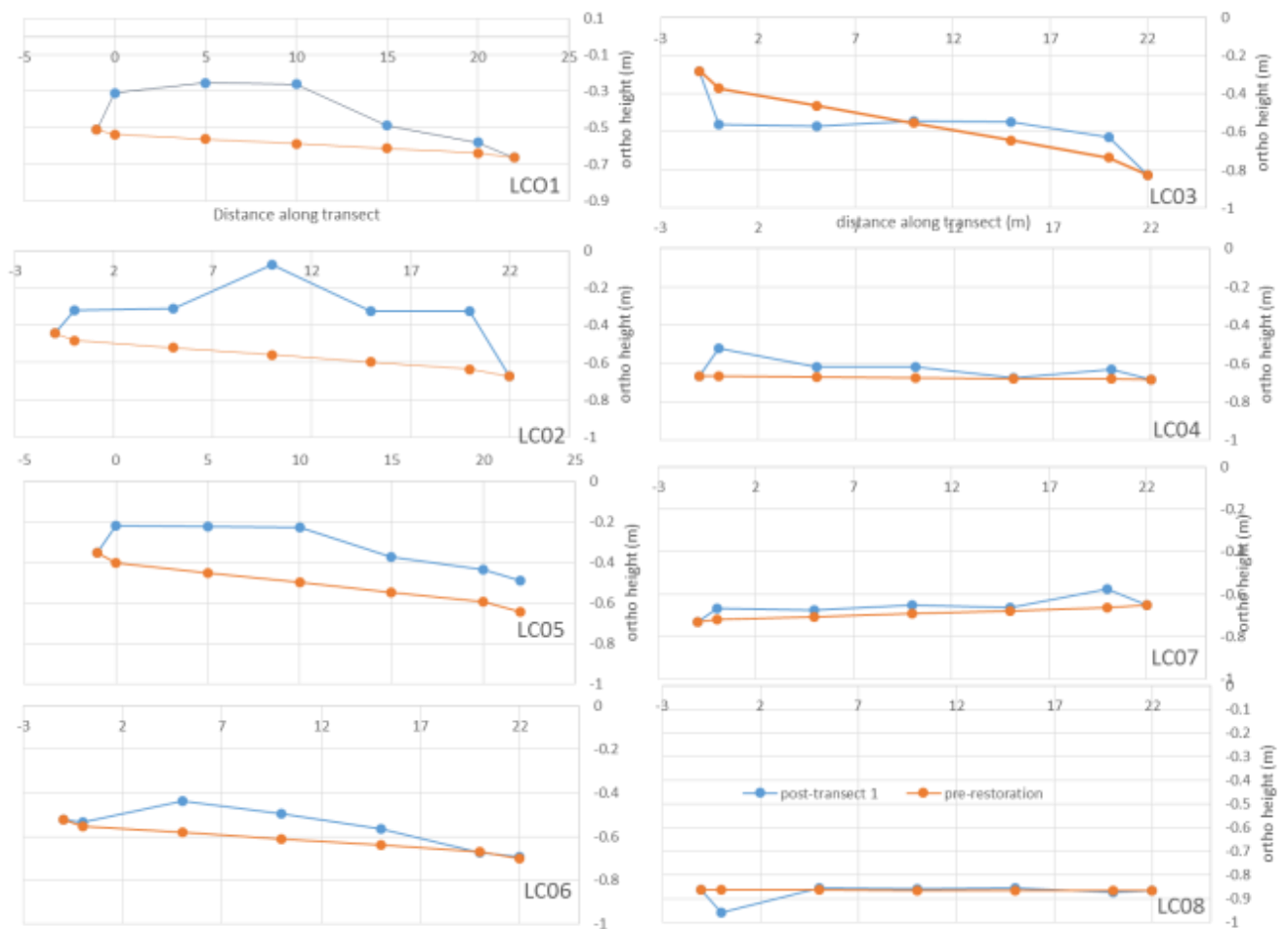
Figure 26. Elevation profiles of restoration (left) and control (right) reefs showing before (red line) and after restoration (blue line). Horizontal axes show decreasing distance away from each inlet at each site.



Figure 27. Aerial photo of site 5 on July 1 2015 on the Lone Cabbage reef showing considerable coverage of rock by sediment.

*Effects of restoration activities on avian community composition and usage of Lone Cabbage Reef.*

After excluding frames in which the reef was covered by tide or that had poor visibility, we recorded a total of 163 hours of potential avian observation on control sites, and 530 hours on restoration sites (Table 3). The large difference in observation time was due to a) the higher elevation restoration sites being above tide for longer, and b) camera failures that were concentrated on the control sites. Birds (all species groups combined) were observed in an average of 32.6 frames/hr on control sites and 17 frames/hr on restoration sites (Table 3), or about 1.9 times as often in the control sites.

Because many of the birds could not be identified to species, we lumped bird observations into species or species-groups. "Large shorebirds" were those clearly as large or larger than a Ruddy Turnstone (*Arenaria interpres*), and generally were Willets (*Catoptrophorus semipalmatus*), Marbled Godwits (*Limosa fedoa*) or Whimbrels (*Numenius phaeopus*). "Small shorebirds" included Semipalmated Sandpipers (*Calidris pusilla*), Western Sandpipers (*Calidris*

Table 3. Results of camera trap surveys of birds on restoration and control oyster bars at Lone Cabbage Reef, Oct 2013 – December 2013. "Frames" were snapshots taken once every 5 minutes. "Bird-frames" are numbers of frames X average numbers of birds/frame.

| | Mean birds per hour | | Lower C.I. | | Upper C.I. | |
|---|---|---|---|---|---|---|
| | control | restored | control | restored | control | restored |
| Osprey | 1.04 | 0.38 | 0.87 | 0.33 | 1.22 | 0.44 |
| lg. shorebird | 0.39 | 0.37 | 0.21 | 0.31 | 0.57 | 0.42 |
| sm.shorebird | 2.81 | 0.05 | 1.77 | 0.02 | 3.84 | 0.08 |
| gulls | 8.35 | 3.61 | 7.02 | 3.09 | 9.68 | 4.13 |
| cormorants | 15.04 | 11.13 | 10.98 | 8.95 | 19.10 | 13.32 |
| bald eagles | 0.25 | 0.51 | 0.16 | 0.44 | 0.33 | 0.58 |
| pelican | 2.63 | 0.22 | 1.96 | 0.16 | 3.30 | 0.28 |
| terns | 2.09 | 0.68 | 0.94 | 0.52 | 3.23 | 0.83 |
| totals | 32.59 | 16.95 | | | | |

*mauri*), Least Sandpipers (*Calidris minutilla*), plovers and other small "peeps". While the vast majority of gulls were Ring-billed (*Larus delawarensis*) and Laughing Gulls (*Leucophaeus atricilla*), we lumped all species into one category because they were difficult to distinguish when roosting together. We also lumped White Pelicans (*Pelecanus erythrorhynchos*) and Brown Pelicans (*Pelecanus occidentalis)* since they were sometimes silhouetted and difficult to distinguish in roosting flocks.

Nearly all species were more often found on the sandy control sites, ranging from close to parity for large shorebirds to 56 times as often for the small shorebirds. This stark contrast for

shorebirds is probably a direct reflection of foraging habitat preference – large shorebirds like Willets and Ruddy Turnstones actively foraged among rocks. All of the smaller sandpipers and plovers typically forage by probing in the substrate, which was largely impossible in rocky substrate. The only species preferring the rocky restoration sites were Bald Eagles, which used the rocks as perches.
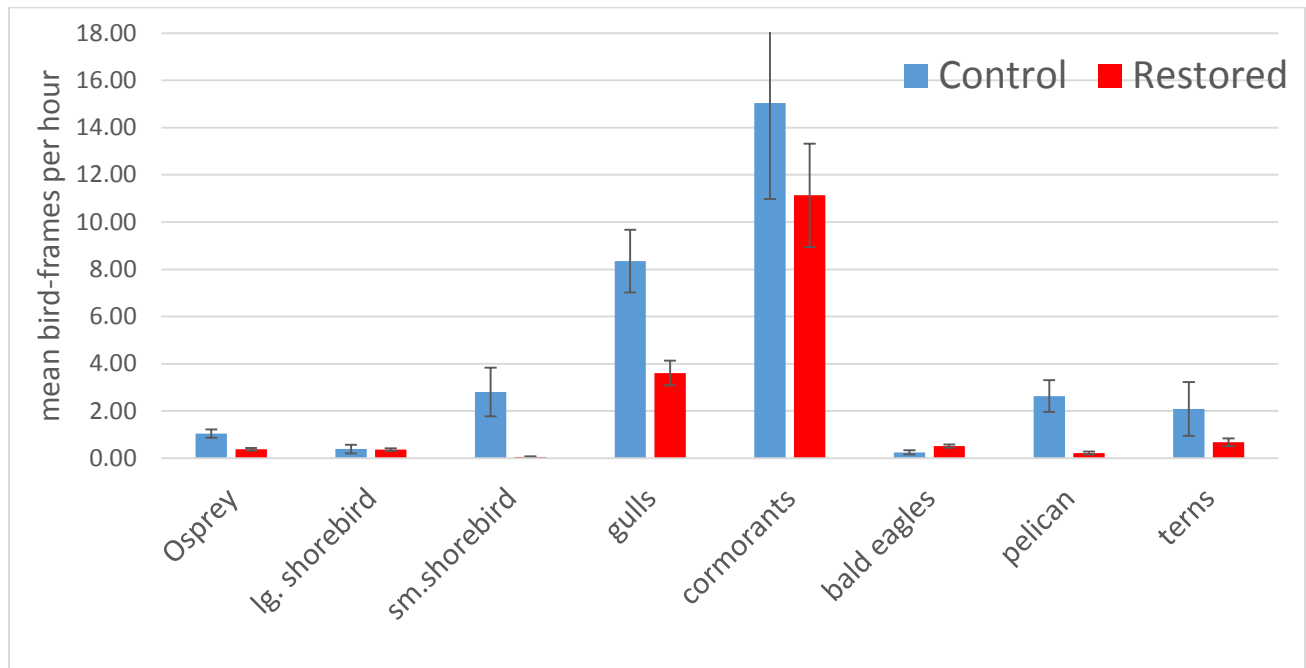


Figure 28. Total number of bird-frames per hour in which different species groups were observed by treatment (blue = control, red = restoration) at Lone Cabbage Reef.

*Effects of restoration activities on blue crab densities on Lone Cabbage*. We recorded and analyzed a total of 2,253 10-s video clips, 732 on restoration sites (sites 1, 2) and 1,167 on control sites (sites 3, 4). Since videos were taken every minute, these numbers also describe the total number of minutes of time the cameras were set up. Since videos were 10s each, this amounts to a total of 375 minutes sampling time, 122 at restoration sites, and 195 at control sites. We found 0.196 crabs/video in control sites, and 0.071 crabs/video in restoration sites, or 2.76 times more total crabs on the control sites. Since each camera's field of view approximated one square meter, these figures can also be used as estimates of areal density (eg, 0.196 crabs/m2 per ten second video).

However, the vast majority of the crabs in control sites were hermit crabs, identifiable by the shell carried as refuge, and typical of the open sand/shell substrate of the control sites. When

focusing only on blue crabs, we found 0.0063 crabs/video on restoration sites and 0.00086 per video on control sites, or 7.52 times more blue crabs on restored sites. Although we could not measure the crabs, the vast majority of blue crabs appeared to be juvenile (<7 cm tip to tip carapace length).

The finding of more blue crabs on restoration sites agreed with our anecdotal observations, made while wading in shallow water over both control and restoration sites (= walkthrough surveys). We waded extensively around these reefs during 15 different trips to the reef during 15 months pre-restoration and the approximately 19-month period postrestoration. We rarely saw blue crabs on any of the reefs pre-restoration, or on control reefs. On restoration sites, we always saw some blue crabs, and during the period April – November typically saw large numbers fleeing our approach (cf 3 – 4 individuals at any one time). These observations suggest both that our sampling efforts reflected true patterns (more crabs on restoration sites), but that the camera sampling was not very accurate in terms of quantity of crabs. For example, if the density of blue crabs is scaled up to an entire site, we would have expected to see a total of 2.8 crabs on the entire plot at any one time. This suggests a gross underestimate by the camera method compared to the walk-through method. The walk-through method has the advantage of disturbing crabs from spaces between rocks and so may make more of the crabs present visible. In addition, the rocky substrate is dark, and the blue crabs can be quite cryptic in that environment, leading to an underestimate of true densities. The control sites, in contrast, have generally light colored sediments against which blue crabs are obvious, presenting a probable bias towards finding more blue crabs in control sites. While the absolute densities are therefore likely to be both biased and inaccurate for the restored sites, the direction of bias and the concordance with walkthrough surveys suggests that the result of more blue crabs on restoration sites is likely to be a robust finding.

*Effects of Lone Cabbage reef on salinities.*
Salinity regime: We recorded salinities at six stations over the course of two years. Several of the stations were either lost or did not have complete records; and all but one were pulled out for service between July 2014 and early November 2014. Thus the picture of salinity is relatively complete, though there are some holes in the data that do not allow complete comparisons.

Salinity dynamics at Lone Cabbage are a product of 1) tidal flow, 2) river discharge, and 3) the physical structure of the reef. On many days, salinities on both sides of the reef may vary from near-ocean strength to near-zero, presenting a sometimes challenging physiological environment for oysters. During the period post-construction, we found generally estuarine conditions at the reef when averaged over the entire period (Figure 29).

While these averages were important for characterizing the overall conditions for oyster growth, they belie very drastic changes at both daily and weekly scales (Figure 30). For example, salinities at the same sites varied from near-zero to 25 ppt over the course of 24-hours, and mean weekly salinities could vary by as much as 18.4 ppt.
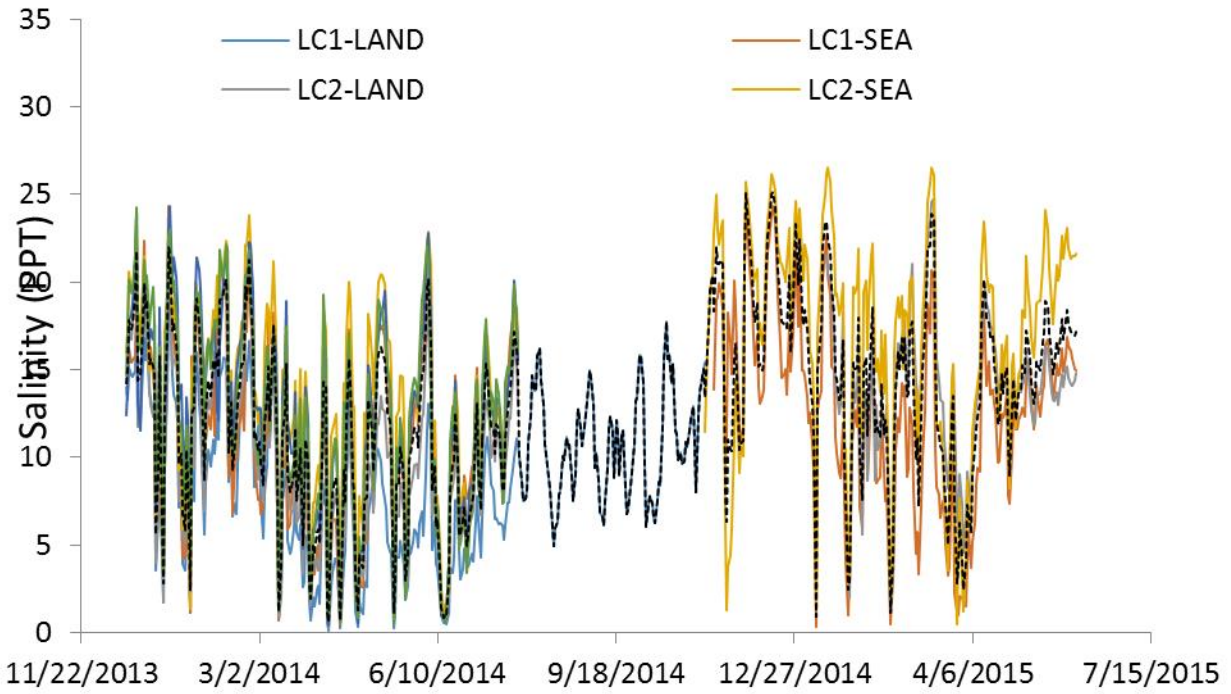
Figure 29. Mean daily salinities on Lone Cabbage reef from sensors on landward and seaward sides of the reef, as averaged November 2013 – June 2015.
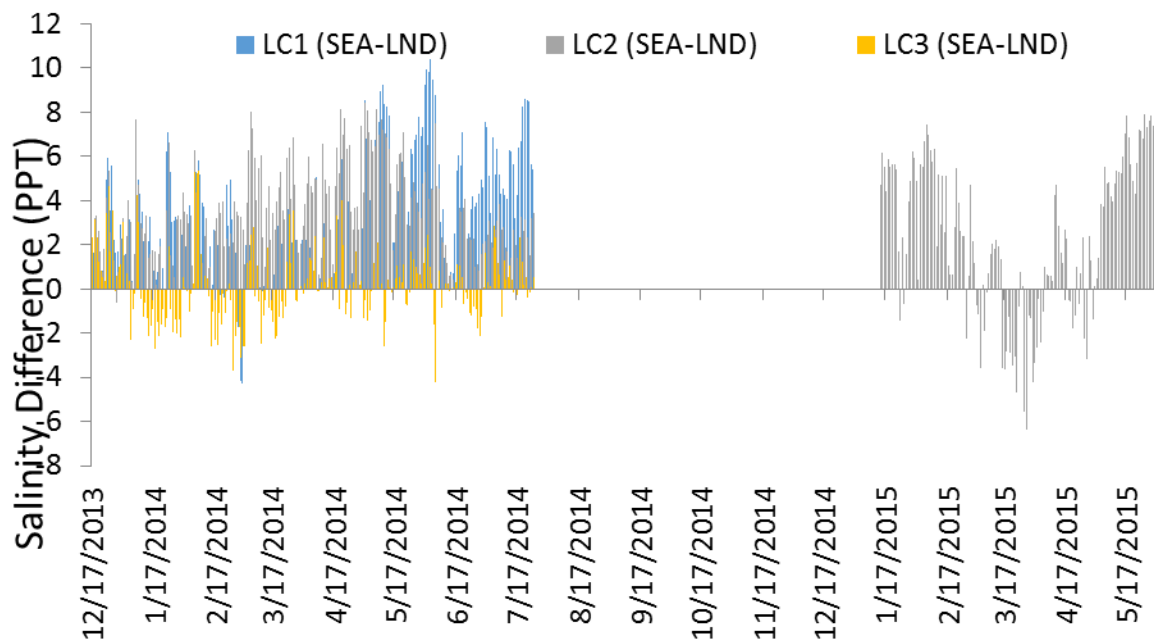
Figure 30.  Instantaneous salinity differences between landward and seaward salinity recording stations at Lone Cabbage reef.  Positive values indicate higher salinities at the seaward station. All stations with full sea/land pairs were included.
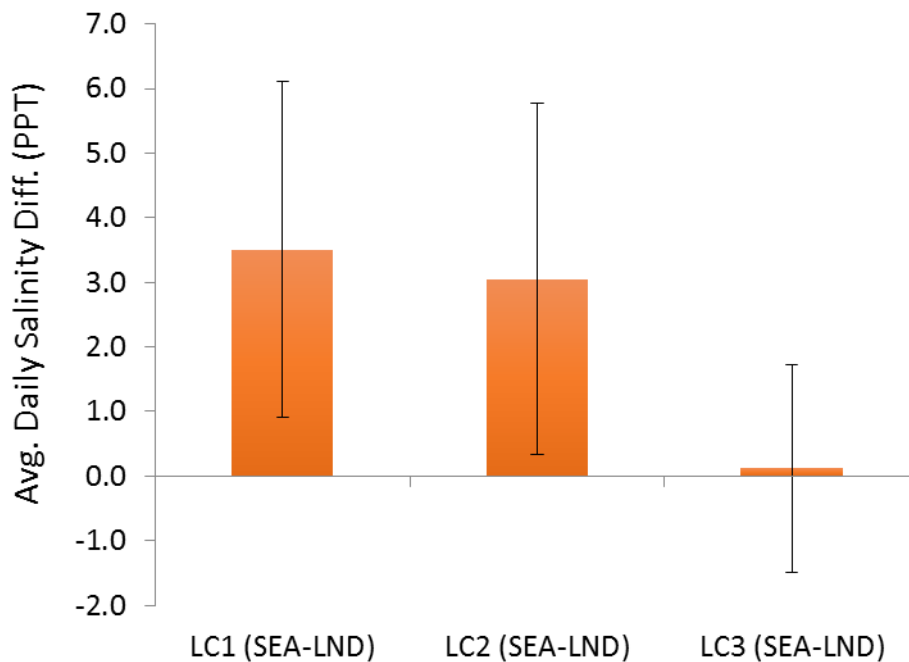


Figure 31.  Average salinity differences between seaward and landward pairs of salinity meters along the Lone Cabbage reef, as averaged over 221 – 425 days depending on station pair. Positive values indicate higher salinities on the seaward side. Mean differences were significant at LC1 and LC2 (t-test, $p<.001$) but not at LC3 ($p = 0.805$).

We also found that there were consistent instantaneous differences between landward and seaward salinities on the reef (Figure 31).  Using the three station pairs that had relatively complete data, we found that seaward stations had consistently higher average salinities than landward stations ranging from 0.12 – 3.5 ppt.

*Modeled salinity response*: We also modeled salinity in relation to both existence of, and elevation of, a hypothetical reef similar in size and elevation to the Lone Cabbage reef (see Figure 32).  Although the model we used was relatively crude (see methods), there was confirmation from this exercise that:

1. Maximum estuarine freshwater extent occurs at low tide, and maximum saline water intrusion occurs at high tide across all simulations.
2. With reef in place, estuarine salinity variability is reduced; freshwater extension towards the sea is also reduced.
3. Instead, the freshwater plume is dispersed laterally within the shallow estuarine zone.

4. This effect is exacerbated with increasing reef height. High reefs lower estuarine salinities on the bay side and reduce overall salinity variability on both the landward and seaward sides of the reef.



NO OYSTER REEF: Q=40 m3/s
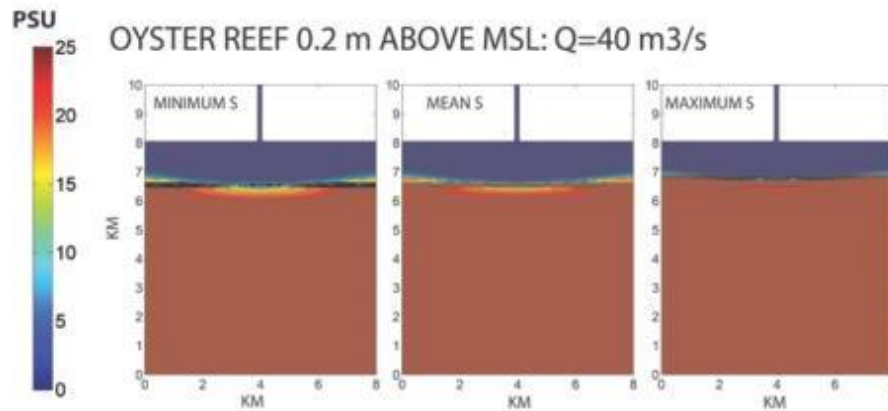
OYSTER REEF AT MSL: Q=40 m3/s
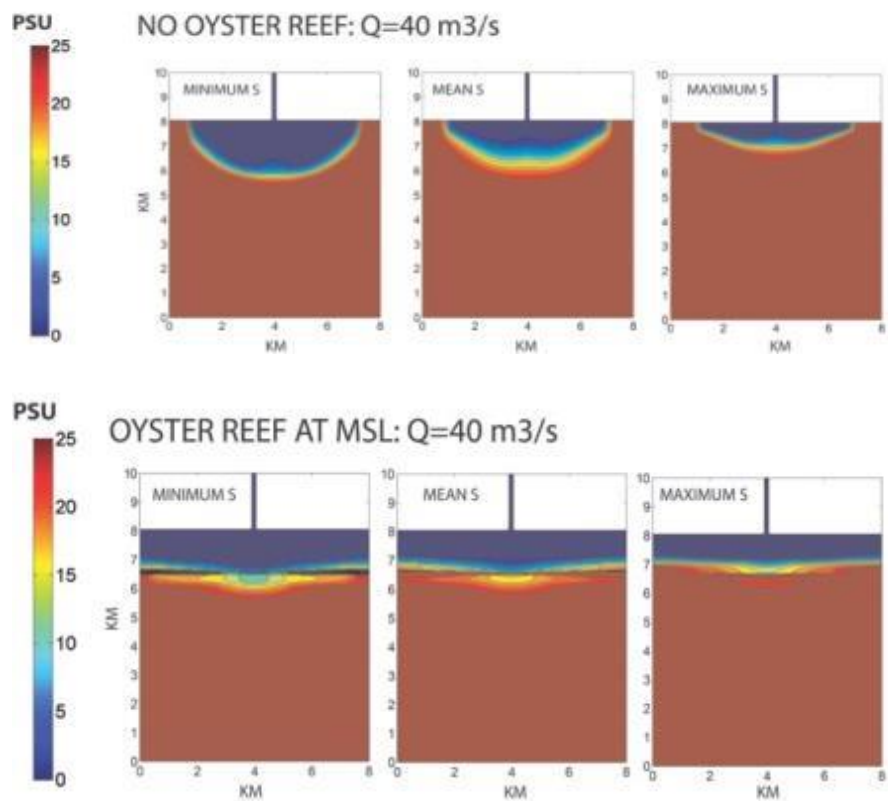
Figure 32. Pictorials of modeled salinities in hypothetical topographies, with no oyster reef, an oyster reef chain with top elevation at mean sea level, and an oyster reef at 20 cm above MSL. Vertical blue line at top of each picture represents the incoming river as seen from above, and the modeled oyster reef is at 6 – 7 km on the vertical axis.

## Discussion

The Lone Cabbage pilot restoration project has been largely successful by all measures we used.  By placing durable substrate on the reef, we have been able to increase elevation by an average of 16 cm, and increase mean oyster density on restoration sites by 2.65X on rock, 14.5X on clam bags and 9.2X overall on the restoration reefs.  Restoration activities were also associated with a moderate increase in mean oyster size on rock (6.5mm, 23.7% increase) and a more substantial one on clam bags (12.0 mm, 44% increase) compared with control sites.

The effects of this change in substrate and habitat also seem to have been felt at higher trophic levels.  Total bird use on restoration sites decreased by an average of over 92% across species groups, with some species over 56 times as numerous on the control sites.  This appears to be due to foraging and roosting habitat preference, and suggests a direct tradeoff of restoring reefs using rock – fewer total birds may be attracted.

Blue crab usage appeared to be much higher on restored sites, though the absolute value of the increase was not accurately measured. Finally, we have shown empirical evidence that the Lone Cabbage reef chain has lower salinities on the landward side than the seaward, strongly suggesting an effect of the reef on local salinities by an average of 3 ppt over a period of 1.7 years.  Modeled results suggest that the effect on local salinities will increase markedly with restored increments of reef elevation.

The effects on oyster density are perhaps the most consistent and striking of these effects, with absolute densities greater than the 90th percentile of restored sites within the Gulf of Mexico.  While there are important differences in the methodologies used to estimate densities between this study and those of LaPerre et al. (2014), the effect of the bias would be in the direction of finding fewer oysters on the LC sites.  For this reason, the finding of higher or

equivalent densities seems robust. We believe the size differences between restoration and control sites will also become more exacerbated as time goes on and the oysters on the LC site mature.

Performance of the rock substrate was generally good, with an increase in oyster density over nearby control sites of 2.65X, suggesting more rapid recruitment and survival under generally favorable salinity regimes seen during the period of study. There was also apparently good resistance to movement, and to sinking into the substrate. For this site, the size of rock (10 – 20 cm diameter) seemed appropriate and smaller sizes would probably have performed less well because of wave action and strong currents. Oysters also cemented themselves across the cracks between some rocks or to one another across those gaps, further stabilizing the rocks and creating a cohesive mass.

Where not physically touching, the rock sizes we used created interstitial spaces of 1 – 15 cm. These spaces were large enough to harbor fish and adult blue crabs. As noted in Results, we initially saw high densities of oysters on the top and the very bottom of rocks, with an apparent "killing zone" in between. We suggest that this pattern is created by predators like crabs and fish (molluscan predators were never seen on the reef during this study), which would themselves be exposed to predation at the tops of the rock at higher tide levels, and would not be able to pick off the spat at the very bottom edge of the rock where it meets the substrate. Thus the zone to forage in without incurring predation costs would be in the mid-rock range. If this is the source of the pattern, then it would mean that there are costs in predation to establishing an oyster reef on this rock size, suggesting a smaller interstitial space may be more efficient. One way to decrease interstitial space without using a smaller rock size would be to lay shell cultch on top of the rock layer, effectively filling in the largest gaps between the rock, but leaving small enough gaps between shells for spat to colonize and grow.

Since clam bags were only partially installed for much of the period of study, the rock seems to have been the primary substrate structuring the restoration reefs. The increased elevations we demonstrated on the restoration reefs was mostly due to the placement of rock on top of the parent substrate – but some of the increase in elevation was also due to accumulation of sand and shell sediments. This makes sense, since the water moving over the rough rocky surface must have slowed down by comparison with the flat control sites, allowing sediments to fall out in eddy zones. This is a two-edged sword – entrapment of sediments clearly reverses the general erosion of elevation that we have measured on Lone Cabbage in the past, but it also leads to buryment of the rock substrate and any oyster reef attached to it. The Suwannee River is generally seen as carrying very little sediment, and the coastal environment as being sediment starved (Hine et al. 1988). This notion is supported by the fact that the Lone Cabbage reef is eroding rapidly and not receiving sediments. This leads to the conclusion that if the entire reef was to be stabilized by restoration of oyster reefs, there would be little buryment potential. In this light, the sediment accumulation we saw is likely an artifact of partial, rather than complete restoration of the reef chain, and seems unlikely to be an issue when sediments are stabilized.

Clam bags also performed well, and were much better than the rock at producing oysters (14.5x oyster density vs. 2.65x on rock, when compared with controls). This is not surprising – many clam bags arrived on site with live oysters, and the mesh bags within which the material was contained excluded most of the large predators of oysters, including oyster drills, crabs

larger than 2 cm, and nearly all fish and elasmobranchs. Thus one of the main reasons for higher densities in clam bags may have been higher survival of existing oysters or those recruited as spat post-construction. This is an important consideration, especially when comparing with rock substrate where (as above) predation on spat may have been high. However, oyster growth within clam bags may also be eventually limited by competition of densely crowded oysters, and reduction of flow created by the mesh of the bags (Bouchillon 2015). One strategy for releasing oysters from growth limitation within bags may be to cut partial slits in bags to increase flow and therefore access to food. However, if done too early this may result in release of unconsolidated contents, and erosion and spreading of the oysters. The durability of the bags themselves is limited to 6 – 10 years and eventually the contents of the bags will not be contained. If composed largely of living oysters, however, the contents may be reasonably well cemented together by that time.

This strategy would of course make the clam bags less competent and more vulnerable to breakup from wave action. One of the original reasons for using clam bags in this high energy site is that they would serve as revetment for the rock, and the placement was designed to keep the rock within a sort of square corral. The bags were indeed durable, and did serve to keep sediments within the corral that might otherwise have washed away. However, the rock itself did not prove to be prone to movement without the bags, and we believe that this function of the clam bags may not necessarily be needed. However, there are other reasons to keep clam bags in the restoration toolkit. At least in the early stages post-restoration the bags are contributing disproportionately (52% of oysters despite being less than 25% of the area of the restored reef) of the oysters to the restored sites because of their protected, high growth environment. So the bags are clearly an important tool in restoring reefs, and provide the vast majority of the ecological benefits such as water filtration and sequestration of nutrients that live oysters reef community can provide (Kellog et al. 2013).

The first prediction that we tested was that local oyster populations on this reef are limited in the short term by substrate – ie, both attachment surface and refugia were in short supply. This pilot study provides evidence that the addition of appropriate substrate does result in an over 9X increase in oyster recruitment to the reef. Since this study provided local and temporal controls, the conclusion that substrate limits local recruitment to this population seems well supported.

The second prediction could not be tested during this pilot study – that restoration of the reef will make the reef and the reef ecosystem (including bay and salt marsh) more resilient to future stressors. This prediction requires at a local scale a comparison of control and restored sites in the growing seasons following a major die-off of oysters, which we did not experience during this study. For the larger implication of the predictions, we would also need to measure salinities, and characteristics of salt marsh and seagrass communities over a longer term study.

**Lessons learned:**
*Choice of restoration substrate materials*: One of the most obvious conclusions from this study is that the construction of durable substrate is expensive and time consuming. There are several related lessons that have to do with materials, cost and contracting. The use of rock in this study was ultimately appropriate, since it was resilient to wave action and provided an immediate, durable increase in elevation that is an important first step towards establishing healthy oyster

populations. However, the cost per square meter ($176) means that the restoration of the entire reef chain would be extremely expensive. The current reef chain that is actually intertidal is approximately 19,350 m2. At these prices the use of rock alone alone would cost more than $3.5 million, and it would only complete about two thirds of the historic reef chain. It may be that there are other materials that are cheaper to use. We have repeatedly seen polyethylene mesh cover netting used in clam aquaculture encrusted densely with large oysters. This netting could be a very cheap source of anchored oysters that could be spread out over large areas of oyster reef at extremely low cost. The current barrier to use of this material is that the use of cover netting is strongly discouraged by the Big Bend Aquatic Preserve, within which the LC site (and most other oyster resources in the area) exist. The perception is that this plastic netting would break loose, float, and become an entanglement hazard and hard plastic pollution in the area. This perception may have flaws which can only be revealed with field tests. Another candidate inexpensive material is derelict crab traps, which are numerous in the area, and which apparently can grow very dense blocks of oysters. Crab traps might be used to rapidly create vertical elevation on a reef, and appear to be very resilient to wave action and movement. Use of this material is currently being explored.

Clam bags also appear to have strong potential as a restoration material in this area. They are donated material, and reasonably cheap to transport, and the total cost in this study was less than the rock on a per-area basis (16% of total costs, 25% of area). Further the clam bags contributed over half of the living oysters on the reef, which is also very disproportionate to area. The cost per ecological function therefore seems well in favor of the use of clam bags compared with rock. However, the two substrates may turn out to have different functions long-term. Rock is likely to be very durable over periods of decades (unless, as above it becomes buried and therefore dysfunctional as oyster substrate). Clam bags are expected to last no more than 10 yr, and by that time their contents may or may not have been self-replicating depending on whether significant low-flow events or major storms have occurred.

Since the "breakwater" function of clam bags is clearly not needed if using large size rock, we should ask whether there is some different usage of materials that might be optimal. The use of rocks alone has merit as breakwater and ranks highly as long-lasting materials, a high priority for restoring these reef chains. However, this option is relatively expensive per area, and only contributes about half the oysters that a combination of materials might. Similarly, the use of clam bags alone also has problems. Although clam bags grow significantly more oysters than rock per unit area, they are not as durable a substrate, and at some point in the future after clam bags disintegrate, an extended cycle of high salinity and oyster mortality will point the reef's trajectory back towards erosion and negative feedback.

One possible alignment of the two materials could be to cover rocks with a layer of clam bag contents (without bags). This would be a good solution to the problem of large interstitial space within rocks – the shell within bags would presumably greatly reduce that interstitial space and also increase net surface area. However, taking oysters out of bags at least partially void the "nursery" function of clam bags – protecting spat that have settled on bag materials from predators. There was evidence of a real difference between survival on rock and survival in bags, with rock having an obvious killing zone in between low and highest elevations, and extremely dense populations of oysters being supported within intact clam bags. In some

situations, clam bag cultch as a top dressing may be a very inefficient use of the live oyster material within bags if the oysters are small because of exposure to predation; as above the opposite would be true for larger oysters.  For this reason, it would be optimal to allow bags with small oysters to grow out within the bag before spreading them on the rock, or to use those bags in places where revetment is needed.

Such a combination would also make excellent use of bags that are less than competent. Although there are more than 20,000 bags available on leases in the area, many of those are now approaching the end of their useful lives at 7 – 10 yr of age. We encountered significant numbers on leases that disintegrated when picked up, or that had large holes needing stitching before they could be used, resulting in variation in quality of the end product on the reef.

While clam bags are clearly less durable in the long run than rock, they are probably more cost efficient as revetment in the short run. Rock must be piled several decimeters deep in order to completely cover the surface of the sand on degraded bars, and even deeper in places with stronger currents and more mobile sediments. In contrast, clam bags can be tied together to form a cohesive mass and can form a complete hard surface without gaps.  Full, competent clam bags may therefore be more effective and cost-efficient to establish hard surface at the edges of gaps between bars where currents are strong and sediment is most active. Clam bags can also be used to create modular structure and can be stacked on top of one another to rapidly create vertical elevation.  While vertical elevation can also be created with rock, the resulting pile of rock is always pyramidal in cross section, and increasing elevation requires exponentially more rock to create.  So in cases where vertical elevation is a priority, clam bags have advantages.

The extremely rough flow surface created by uneven mounds of rock has advantages and disadvantages.  On the one hand the variation in topographic relief may mimic natural oyster bars and provides a full range of intertidal edaphic variation for oyster reef fauna to colonize and occupy.  Further, the topography and rough surface provide huge surface area for oyster reef fauna, though as above, the interstitial spaces may not be optimal for oyster recruitment. Topography and rough surface also slow currents down and create a diversity of eddies and flows within the reef, which may also mimic a natural oyster reef.  The only real downside is that by slowing currents down, rock apparently also causes deposition of lighter sediments and eventual burying of rock and oysters.  This problem can probably be overcome with increasing scale of restoration – the more surface area covered on the entire chain of reefs, the less sediment available to move and bury the reef materials. It is unclear whether clam bags have some advantage for this sediment deposition problem – the surface of packed clam bags is much smoother than that of rock, and may therefore be less likely to cause sediment fall-out.  The Atsena Otie reef created in 2007 entirely of clam bags shows no signs of silting in, though this reef is in an entirely different flow and sediment environment than Lone Cabbage.  However, at some point oysters will grow through the clam bags as the bags disintegrate, and it is unclear in general whether thesmoother surface of the bag is a net advantage in reducing sediment deposition.  One option would be to add new clam bags as old ones disintegrate when and if erosion/deposition forces become apparent.

*Contracting issues:* This problem of variation in competence of bags became a contracting and logistical issue during this project.  Bag movers were given the task of moving bags on a per-

piece basis, with the assumption that they would install full, competent bags. But on most leases, the movers encountered bags that were not competent or were not full. The goal for the lease owner is to get all derelict bags removed regardless of condition, and this puts the bag mover at some odds with the leaseholder. Effort to move disintegrating bags or to combine bags to make full ones is much higher than picking up competent, full bags. Further, if competent bags are the only thing to be delivered to the restoration site, what is to be done with the incompetent bags? The latter become dead freight. The solution is 1) pay more per bag or yard of cultch moved to accommodate the variation in effort, and 2) make use of all cultch, whether in a bag or not. Thus using cultch as a "top-dressing" would fit quite well with the cost and effort efficiency of processing and moving clam bags, and would remove the potentially corrosive disparity in goals between leaseholder and mover.

A second contracting lesson is that the labor needed to move clam bags is clearly competing with other local employment such as clam aquaculture and the wild oyster fishery. Relative attractiveness of these kinds of work is strongly related to product price – the marked increase in oyster prices in the winter of 2014/5 coupled with a selloff in clam product late in 2014 fomented a noticeable shift towards oystering. One lesson from this experience is that clam bag moving (and other kinds of manpower-intensive restoration work) should be targeted for the summer months when oyster beds are closed. A second lesson is that the labor price that incentivizes clam bag movers may fluctuate considerably with competing industries as well as fuel price fluctuations. Budgets for future restoration efforts should therefore be constructed to incorporate flexibility in labor prices of 15 – 30%. This need for flexibility in pricing and costs extends to rock moving – costs of rock, fuel and labor will fluctuate markedly with the larger construction industry that ultimately drives each of these costs. Thus restoration funding agencies must have contracting mechanisms that allow for price fluctuation, and restoration contractors need to have as up-to-date pricing as possible when writing budgets. The unpredictable and generally long periods of permitting involved in major restoration projects contributes strongly to a temporal mismatch between estimated and actual pricing for restoration. Thus two-phased budgets would best insulate funding agencies – a relatively inexpensive permitting phase followed by a construction phase budget. The key concept is that the construction phase budget is determined AFTER permits are obtained. While this means more contract negotiation, the accuracy it allows for pricing will ultimately be a strong determinant of cost efficiency and project effectiveness.

Another hard lesson learned is that surveying is best done by a surveyor. Despite having good equipment, we spent nearly two years deriving accurate solutions for our benchmarks. While the geographic isolation of the construction site presented some real challenges for accurate elevation determination, a competent surveyor would probably have overcome these in a period of weeks rather than years.

We also emphasize that the true costs of restoring reef are not embodied in the figures presented here, for several reasons. First, we present only the contracted construction costs, and have not included actual costs of preconstruction permitting, surveying and methods development, and monitoring. In the end, we had to sacrifice all budget for personnel in this project to cover construction costs. So the true costs in personnel were heavily subsidized by volunteer labor both by students and faculty from University of Florida. Second, the costs we

present are those that reflect all the mistakes and missteps typical of a first-time pilot project – those "development costs" would not be predicted to recur in future efforts because of lessons learned. These include learning about surveying, increases in efficiency in contracting for moving rock and oysters, development of better material configurations, and a general increase in efficiency of working on this logistically challenging section of coast. For these reasons, the construction costs are probably misleading as true costs, but the future construction costs would almost certainly be less in future projects than those experienced in this pilot work.

## Literature Cited

Baggett, L.P., S.P. Powers, R. Brumbaugh, L.D. Coen, B. DeAngelis, J. Green, B. Hancock, and S. Morlock, 2014. Oyster habitat restoration monitoring and assessment handbook. The Nature Conservancy, Arlington, VA, USA, 96pp.

Beck, M.W., R.D. Brumbaugh, L. Airoldi, A. Carranza, L.D. Coen, C. Crawford, O. Defeo, G.J. Edgar, B. Hancock, M.C. Kay, H.S. Lenihan, M.W. Luckenbach, C.L. Toropova, G.F. Zhang, and X.M. Guo. 2011. Oyster reefs at risk and recommendations for conservation, restoration, and management. Bioscience 61: 107-116.

Berrigan, M., T. Candies, J. Cirino, R. Dugas, C. Dyer, J. Gray, T. Herrington, W. Keithly, R. Leard, J.R. Nelson, and M. van Hoose. 1991. The oyster fishery of the Gulf of Mexico, United States: a regional management plan. Gulf States Marine Fisheries Commission. Ocean Springs, Mississippi.

Bouchillon, R. 2015 Characterizing Eastern Oyster (*Crassostrea virginica*) growth in response to environmental and seasonal factors in the Big Bend Region of Florida. Unpublished MS thesis, University of Florida.

Buzan, D., Lee, W., Culbertson, J., Kuhn, N., Robertson, L. 2009. Positive relationship between freshwater inflow and oyster abundance in Galveston Bay, Texas. Estuaries and Coasts 32: 206-212.

Chassignet, E.P., H.G. Arango, Dietrich, D., Ezer, T., Ghil, M., Haidvogel, D.B., Ma, C.-C., Mehra, A., Paiva, A.M. & Sirkes, Z., 2000. DAMEE-NAB: The Base Experiments, Dynamics of Atmospheres and Oceans, 32, 155-183.

Coen, L.D., R.D. Brumbaugh, D. Bushek, R. Grizzle, M.W. Luckenbach, M.H. Posey, S.P. Powers, and S.G. Tolley. 2007. Ecosystem services related to oyster restoration. Marine Ecology Progress Series 341: 303-307.

Farrell, M. D., Good, J., Hornsby, D., Janicki, A., Mattson, R., Upchurch, S., ... & Batchelder, P. (2005). Technical report: MFL establishment for the lower suwannee river and estuary, little fanning, fanning, and manatee springs. Water Resource Associates, Inc., Tampa, Florida.

Fish and Wildlife Economics and Statistics. 2015. Opportunities for growth: a summary of economic analysis of Florida's Big Bend Region. The Conservation Fund and Southwick Associates.

Grabowski, J.H., Brumbaugh, R.D., Conrad, R.F., Keeler, A.G., Opaluch, J.J., Peterson, C.H., Piehler, M.F., Powers, S.P., Smyth, A.R. 2012. Economic valuation of ecosystem services provided by oyster reefs. BioScience 62: 900-909.

Greenberg, S. and T. Godsin. 2015. Angling effort data from remote camera images. Fisheries 40:276-287.

Grinnell, Jr., R.S. 1972. Structure and development of oyster reefs on the Suwannee River delta, Florida. Ph.D. Dissertation, State University New York, Binghampton, New York.

Haidvogel, D. B., Arango, H.G., Budgell, W. P., Cornuelle, B. D., Curchitser, E., DiLorenzo, E., Fennel,K., Geyer, W.R., Hermann, A. J., Lanerolle, L., Levin, J., McWilliams, J. C., Miller, A. J., Moore, A. M., Powell, T. M., Shchepetkin, A. F., Sherwood, C. R., Signell, R. P., Warner, J. C. & Wilkin, J.,2008. Regional Ocean Forecasting in Terrain-following Coordinates: Model Formulation and Skill Assessment. Journal of Computational Physics 227 3595-3624.

Haidvogel, D.B., H.G. Arango, K. Hedstrom, A. Beckmann, P. Malanooe-Rizzoli, and A.F. Shchepetkin, 2000. Model Evaluation Experiments in the North Atlantic Basin: Simulations in Nonlinear Terrain-Following. Dynamics of Atmospheres and Oceans, 32, 239–281.

Hine, A.C., D.F. Belknap, J.G. Hutton, E.B. Osking, and M.W. Evans. 1988. Recent geological history and modern sedimentary processes along an incipient, low-energy, epicontinental-sea coastline: Northwest Florida. Journal of Sedimentary Petrology 58:567–579.

Kaplan, D., Olabarietta, M., Frederick, P., Valle-Levinson, A., Seavey, J. 2014. Estuarine Freshwater Entrainment By Oyster Reefs: Quantifying A Keystone Ecosystem Service. Poster at the American Geophysical Union Fall Meeting, December 2014, San Francisco, CA.

Kellogg, M.L., J.C. Cornwell, M. S. Owens, and K.T. Paynter. 2013. Denitrification and nutrient assimilation on a restored oyster reef. Marine Ecology Progress Series 480:1–19

La Peyre M., J. Furlong, L. A. Brown, B.P. Piazza and K. Brown. 2014. Oyster reef restoration in the northern Gulf of Mexico: Extent, methods and outcomes. Ocean & Coastal Management 89:20-28.

Petes, L.E., A.J. Brown, and C.R. Knight. 2012. Impacts of upstream drought and water withdrawals on the health and survival of downstream estuarine oyster populations. Ecology and Evolution 2(7): 1712-1724.

Raabe, E. A., & Bialkowska-Jelinska, E. (2007). Temperature anomalies in the Lower Suwannee River and tidal creeks, Florida, 2005 (No. 2007-1311). US Geological Survey.Rodriguez, A.B., F.J. Fodrie, J.T. Ridge, N.L. Lindquist, E.J. Theuerkauf, S.E. Coleman, J.H. Grabowski, M.C. Brodeur, R.K. Gittman, D.A. Keller, and M.D. Kenworthy. 2014. Oyster reefs can outpace sea-level rise. Nature Climate Change 4(6): 493-497.

Scyphers, S.B., S.P. Powers, K.L. Heck, Jr., and D. Byron. 2011. Oyster reefs as natural breakwaters mitigate shoreline loss and facilitate fisheries. PLoS ONE 6: e22396.

Seavey, J.R., W.E. Pine III, P. Frederick, L. Sturmer, and M. Berrigan. 2011. Decadal changes in oyster reefs in the Big Bend of Florida's Gulf Coast. Ecosphere 2: 114.

Shchepetkin, A.F. and McWilliams, J.C., 2005. The regional ocean modeling system (ROMS): a split-explicit, free-surface, topography-following-coordinates ocean model. Ocean Modelling 9, 347-404.

USGCRP (2009). Global Climate Change Impacts in the United States . Karl, T.R. J.M. Melillo, and T.C. Peterson (eds.). United States Global Change Research Program. Cambridge University Press, New York, NY, USA.Walters, C.J. and S.J. Martell. 2004. *Fisheries ecology and management*. Princeton University Press.Wright, E.E., A.C. Hine, S.L.

Goodbred, and S.D. Locker. 2005. The effect of sea-level and climate change on the development of a mixed siliciclastic-carbonate, deltaic coastline: Suwannee River, Florida, USA. Journal of Sedimentary Research 75:621– 635.