

Shoreline stabilization and habitat restoration at Helen Wood Park, AL

Kenneth L Heck, Jr., Sean P. Powers, Steven B. Scyphers and Dorothy Byron

Final Report to the Mobile Bay National Estuary Program

September 30, 2010

Introduction

Estuaries along the Northern Gulf of Mexico (GOM) support a broad spectrum of marine life and have historically served as grounds for recreational fishing and commercial harvest of numerous species of fishes, crustaceans, and shellfish (Gunter 1967). Of the species harvested commercially in the GOM, approximately 98% (by weight in 1983) have been categorized as having an estuarine-dependent life cycle (Chambers 1991). An economic analysis of ecosystem goods and services contributed by estuarine and vegetated nearshore habitats, which comprise 0.7 % of global biomes, yields a yearly contribution of \$7.9 trillion dollars or approximately 24% of total global ecosystem services (Costanza et al. 1997). These include disturbance resistance, nutrient cycling, habitat, food production, and recreation. Thus, these are critical contributions that extend well beyond the coastal zone.

Unfortunately, coastal and estuarine shorelines are one of the most degraded and threatened habitats in the world because of their sensitivity to sea level rise, storms, and increased utilization by man (Vitousek et al. 1997, Syvitski et al. 2005). Impacts of nearshore habitat degradation and historical over-harvesting of many coastal fishes and shellfish have substantial economic and ecological consequences (Chambers 1991, Jackson et al. 2001). As a result, efforts to re-establish or enhance ecological function to damaged and destroyed habitats have increased substantially (Vitousek et al. 1997, Minton 1999, Thayer et al. 2003).

Previous efforts to protect shorelines have involved the introduction of hardened structures, such as seawalls, rock jetties, or bulkheads to dampen or reflect wave energy. Human alteration of the coastal zone, including shoreline armoring and construction, is one of the most detrimental impacts on estuarine and coastal shores (Vitousek et al. 1997, Kennish 2002). Located on the northern Gulf coast, Mobile Bay is an exemplary case of how shoreline armoring is directly related to population expansion and migration to coastal regions. Mobile Bay is one of the best examples of a classic estuary (Gunter 1967), and like many other coastal areas, is highly developed with a large and increasing proportion of the shorelines armored by bulkheads and seawalls (Douglass & Pickel 1999). At last analysis in 1997, Douglass and Pickel estimated that over 30% of the bay's available coastline was armored with over 10-20 acres of intertidal habitat lost, a high percentage in this microtidal bay. A major concern in implementing bulkheads and seawalls for coastal property protection is reflection of erosive

wave energies back into the bay, instead of absorbing or dampening the wave energy. This subjects adjacent shorelines to even greater wave energy and can cause vertical erosion down the barrier with subsequent loss of intertidal habitats (Douglass & Pickel 1999). The continuous progression of armoring is mainly due to a lack of practical and ecologically responsible alternatives.

Recently, shoreline protection efforts have shifted towards the implementation of biogenic reefs or “living shorelines” (NRC 2007). Living shorelines usually involve the planting or restoration of naturally occurring coastal plants or shellfish. Living shorelines, especially when utilizing bivalve shells, appear to have numerous benefits in addition to providing a buffer for estuarine and coastal shores. Given adequate larval supply and survival, oyster reefs are a promising technique for protection and sustainability, as compared to hardened vertical structures (Piazza et al. 2005). Oyster reefs are often found seaward of marshes and have been shown to mitigate erosive wave energies, stabilize sediments, and reduce marsh retreat (Dame & Patten 1981, Meyer et al. 1997). Beyond shoreline stabilization, living oyster reefs may provide several additional ecosystem services that include: 1) seston filtration, 2) benthic-pelagic coupling, 3) refuge from predation, 4) creation of feeding habitat, and 5) nesting habitat (Coen et al. 1999, Coen et al. 2007). On a larger scale these ecological benefits could facilitate the enhancement or even expansion of other habitats such as seagrass meadows or tidal marshes (Coen et al. 2007).

Natural oyster reefs provide habitat for numerous species of infauna and epifauna, including fishes, worms, and other crustaceans that serve as important prey items for larger transient and demersal fishes (Zimmerman et al. 1989, Coen et al. 1999, Glancy et al. 2003). Restored oyster reefs have been demonstrated to function at an equivalent or greater level than natural reefs with decapods, crustaceans, and fish quickly colonizing the available substrate (Meyer & Townsend 2000). Among the species with enhanced recruitment benefits from oyster reefs are several economically-important species including: stone crabs (*Menippe mercenaria*), gray snapper (*Lutjanus griseus*), gag grouper (*Mycterperca microlepis*), and southern flounder (*Paralichthys lethostigma*) (Peterson et al. 2003). These structurally-complex habitats are thought to minimize interspecific and intraspecific competition among predators at intermediate densities due to increased foraging efficiency (Grabowski & Powers 2004). Considering all of the ecosystem services provided by these living reefs, oyster restoration appears to be a more beneficial avenue than prior methods utilizing concrete bulkheads or similar structures to reduce wave energy.

Objectives

The objectives of this study were to examine the habitat value and effectiveness of two different breakwater reefs designed for stabilizing shorelines at a scale applicable to a coastal

property-owner. To evaluate the effectiveness of these reefs, we documented shoreline changes, quantified ichthyofauna and invertebrate communities adjacent to each treatment, and conducted monitored oyster recruitment. This final report describes the construction and monitoring of breakwater reefs at Helen Wood Park on the western shore of Mobile Bay, AL.

Site Description and Experimental Design

Helen Wood Park (HWP) is located just north of the mouth of Dog River in Mobile Bay. The park is heavily armored by a bulkhead and concrete bridge rubble and is located adjacent to a substantially eroded shoreline. The breakwater reefs at HWP are designed to experimentally test two breakwater reefs that could potentially serve as “living shoreline” alternatives to traditional armoring (Figure 1).

The first configuration (HWP SR, HWP NR) consists of 123 (3 rows of 41) Reef Ball Foundation “Lo-Pro” (hereafter “reef balls”) modules deployed parallel to the eroding shoreline, with adjacent modules touching each other. Each reef ball is a hollow dome structure with several circular openings in the sides and top (Figure 2). Each module is 61-cm in diameter at the base and 45.7-cm tall. The total footprint of this breakwater reef configuration measures 2-m by 25-m long.

The second configuration (HWP SO, HWP NO) was built to the same specifications as the first, but is comprised of bagged oyster shell as opposed to reef balls. This pyramid-shaped reef has a footprint of 2-m wide and 25-m long, narrowing to 1-m wide at the crest. Each bag, approximately 2,000 per reef, contains approximately 0.644-m³ of shucked oyster shell. These bags were deployed by small boats and volunteers, and all bags were placed parallel to the shoreline for the entire reef. This method was chosen due to the location of this study site and known wind patterns for this region. The majority of wind-driven waves at this site are derived from southward winds in spring and summer months, shifting to a more northern orientation in winter months (Schroeder & Wiseman Jr. 1985).

Control sites were designated (HWP SC, HWP NC) to allow comparisons with a non-restored condition (Figure 1). The controls were marked with PVC just as the experimental reefs, but lacked any other structure. The study site was divided into north and south zones separated by 100-m and treatments were randomly assigned within each zone.

Construction

All restoration activities were conducted in May 2008 and staging occurred in Helen Wood Park. The deployment of the reef ball modules, and bagging and deployment of the oyster shell took seven full days of labor and over 30 different volunteers. On average, there were ten or more volunteers on each day of preparation and deployment. The cumulative

efforts of the project personnel and volunteers total 600 labor hours during the shell bagging and deployment phase of the project. The shell bagging and deployment demanded five times more labor than the reef ball low-pro modules (Figure 3a). The cost of materials and freight for reef ball however was considerably more expensive than the cost of loose oyster shell and bagging materials (Figure 3b). The target depth for reef deployment was 1-m at mean lower low water. This was not achievable in close proximity to the retreated shoreline at HWP, therefore the treatments had to be established in 0.75-m depth MLLW and located approximately 60-m from shore.

Monitoring Methods

Shoreline Change

To gauge the effectiveness of each breakwater design we conducted GPS transects, measured relative water motion, and utilized aerial photographs of the study site. GPS transects were conducted twice yearly and consisted of walking the offshore boundary of the living vegetation line with a CEEDUCER dGPS that continuously recorded latitude and longitude with sub-meter accuracy. When no vegetation was present on the shoreline (i.e. the northern extent of the study site), transects followed the vertical drop-off of the shore. Relative water motion was measure over a 72 hour period in fall 2009 by gypsum flow clogs. Gypsum flow clogs provide an effective and economical way of measuring relative water flow between sites of interest (Yund et al. 1991). The hemisphere-shaped clogs were deployed 5 and 25-m shoreward of each treatment. Each hemisphere was mounted to a plastic base and suspended 10-cm above the benthos. After retrieval, each hemisphere was dried to a constant weight and compared to the pre-deployment weight.

Oyster Recruitment

Oyster counts were conducted at 6, 12, and 18 months post deployments. For reef balls, 9 modules were randomly selected and completely inspected for oyster recruitments. For bagged shell reefs, 6 bags were haphazardly chosen to be removed from the reef. The contents of the bags were emptied and each shell was checked for juvenile and adult oysters, mussels and oyster drills. All counts were reported as density estimates (m^{-2}).

Water Quality

To measure water clarity and the abundance of phytoplankton, we collected water samples at each sampling station monthly from May to November and bi-monthly from December to March. The samples were placed in one liter plastic bottles, stored in the dark on ice and

transported to the laboratory. Water quality, using the amount of total suspended solids in a known volume of water as a proxy, was determined by filtering between 100 -200ml onto a pre-weighed Whatman® 47-mm glass microfiber filter (GF/F). The filters were then dried at 60°C for 48 hours and reweighed to calculate the mass of total suspended solids in the water column (mg DW L⁻¹). Phytoplankton abundance (µg L⁻¹) was measured as water-column chlorophyll *a* concentration by filtering triplicate 20 ml aliquots from each 1-L sample onto a Whatman® 25-mm glass microfiber filter (GF/F). Filters were frozen at -80°C until chlorophyll *a* analysis could be performed.

Chlorophyll *a* was extracted from filters using approximately 5 ml of a 2:3 mixture of dimethyl sulfoxide (DMSO):90% acetone. The chlorophyll *a* content (µg L⁻¹) was determined fluorometrically (Turner Designs® TD-700) using the Welschmeyer method, which is designed to be minimally sensitive to chlorophyll *b* and chlorophyll degradation products (Welschmeyer 1994).

To examine how differences in suspended solids and phytoplankton concentrations affect water-column clarity, we measured the light intensity (with a LI-COR 1400 datalogger and underwater spherical sensor) above the water, just below the water's surface and at sediment surface waters at each station between June 2009 and November 2009.

Fishes and Mobile Invertebrates

To quantify the value of these reefs for finfish and shellfish, a wide range of sampling techniques was used to survey the communities surrounding the reefs. To quantify larger, transient fishes on breakwater reefs and controls, experimental gillnets were deployed (n=2 for PP and AP, n=1 for HWP) per treatment at each site. Each 30 m experimental gillnet was comprised of two different mesh sizes (5 cm and 10 cm) to broaden the size range of animals captured. The nets were fished during the crepuscular period when many demersal fishes should be foraging. A single net was deployed diagonal to shore on the offshore side of the treatment, beginning at a randomly selected end of the reef. The nets were retrieved in the same order they were deployed, and soak time was recorded as the time from when the net was first deployed until the time retrieval began. This time was approximately 2 hours (1-hour before and after sunrise or sunset), but was occasionally longer due to extended retrievals. All specimens captured were collected and returned to the lab for identification, measurement, and weighing.

To quantify smaller fishes and invertebrates, a 6 m wide bag seine with 6.25 mm mesh was used. Seining was conducted along the inshore and offshore sides of each reef or control site. Each seine was pulled directly adjacent to the reef and directed into a 4 m block net. Seine distances were recorded for catch per unit effort (CPUE) calculations. All captured fishes and

mobile invertebrates were placed in bags and returned to the lab for identification, measurement, and enumeration. In the lab, each seine sample is identified to the lowest taxonomic designation possible (Family or lower). A representative sample of twenty individuals per species or family was measured for total length (TL), and a total group biomass recorded.

Statistical Analyses

Water clarity metrics, gillnet and seine catches were analyzed using 2-factor analysis of variance with treatment (bagged oyster, reef ball, control) and season as independent factors. Analyses were run separately on 5-cm, 10-cm, and seine results for total catch and demersal fishes. Environment classifications for fishes, such as demersal, were established using FISHBASE (www.fishbase.org) and are listed in Table 1.

Results

Shoreline Change

GPS surveys of the shoreline at Helen Wood Park did not indicate differences in erosion between reef ball, bagged oyster shell, or control treatments (Figure 4). Gypsum dissolution among the 5-m measurements determined that water flow was greatest on the control sites (Figure 5). For 25-m measurements, water flow was similar for control and bagged oyster treatments and both were greater than reef ball treatments (Figure 5).

Oyster Recruitment

During the 6 and 12-month sampling periods, only 2 live oysters were observed on reef ball structures. Less than 10 live or dead oysters were found on bagged oyster shell from all sampling periods. During the 18 month sampling period, only 1 live oyster was observed in the bagged oyster shell reefs and no live oysters were found on any reef ball structures. The bagged oyster shell reefs were primarily comprised of mussels (Figure 6) with densities as high as 1962 mussels m^{-2} on the South reef. Mussels were also found on the reef balls but in much lower numbers with the highest density being 81 mussels m^{-2} .

Water Quality

There were no apparent seasonal trends in either the total suspended solids or primary producer measurements. Total suspended solids ranged from 6 to 602 mg L⁻¹, with a mean value of 58.66 mg L⁻¹ at the control site, from 12.5 to 729.5 mg L⁻¹, with a mean value of 60.06 mg L⁻¹ at the bagged shell site and from 7 and 451 mg L⁻¹ with a mean value of 61.95 mg L⁻¹ at the reef ball sites (Figure 7). Such large ranges in measurements were due to local weather conditions (ie. wind speed and direction; data not shown). Measurements were generally

highest along the shore transect at all treatment sites with the highest values for all stations occurring during the October 2009 sampling event. There was no significant difference detected between treatments ($p=0.913$) or treatments x date ($p=1.0$); however there was a significant difference detected between dates sampled ($p<0.001$).

Phytoplankton abundance measured by water column chlorophyll *a* concentration also showed little seasonal variation across sampling dates. Chlorophyll *a* concentration ranged from 1.43 and 55.38 $\mu\text{g L}^{-1}$ with a mean value of 13.77 $\mu\text{g L}^{-1}$ at the control sites, from 1.22 and 46.99 $\mu\text{g L}^{-1}$ with a mean value of 13.48 $\mu\text{g L}^{-1}$ at the bagged shell sites, from 1.50 and 64.74 $\mu\text{g L}^{-1}$ with a mean value of 13.54 $\mu\text{g L}^{-1}$ at the reef ball sites (Figure 8). Lowest values were recorded during the July 2008 and November 2008 sampling events and the highest values were measured during the August 2008 sampling event. Although the northern Gulf Coast was influenced by two tropical systems (H. Gustav and H. Ike) during 2008, the high measurements recorded at the treatments occurred before these storms influenced the area; therefore, the high variation in phytoplankton abundance was most likely due to local weather conditions (ie. wind speed and direction and cloud cover; data not shown). Again, there was no significant difference detected between treatments ($p=0.835$) or treatments x date ($p=0.611$); however there was a significant difference detected between dates sampled ($p<0.001$).

Photosynthetically active radiation (PAR) available on the benthos dictates whether the habitat can sustain submerged aquatic vegetation. This available light is directly influenced by total suspended solids. Since we did not observe any differences between treatments for measurements of total suspended solids, it is expected that we would also not see any significant differences in light reaching the bottom between treatments. The mean values of PAR on the benthos were 63, 62 and 58 $\mu\text{Em}^{-2}\text{s}^{-1}$, for the control treatment, the bagged shell treatment and the reef ball treatment, respectively (Figure 9). Although there were significant differences between sampling dates ($p = 0.003$), neither treatment nor the interaction of date and treatment were significant ($p=0.637$ and $p=0.226$, respectively).

Fishes and Mobile Invertebrates

Gillnet samples captured 30 different species of fishes. Seining for smaller fishes and invertebrates resulted in 35 different species. Total catch for all species or taxa that represented greater than one percent of the total is shown in Table 1. For the 5-cm and 10-cm gillnet catch, the abundance of larger fishes appears to be similar across all treatments (Figure 10). Results from seine samples show control sites with a lower abundance than bagged oyster or reef ball sites, which appear to be similar (Figure 10). Two-factor analysis of variance found season to be highly significant for total catch and gear types (Table 2a). Treatment was only significant for total catch of seine samples. For demersal fishes, the only significant result was a seasonal effect in the 5-cm gillnet samples (Figure 11, Table 2b). It appears that total

abundance is highest in early and late summer (Figures 12). Differences between breakwater treatments and controls appear to be increasingly evident during the peaks that occur May/June and September/October (Figure 12).

Species richness appears to show similar trends to abundance (Figures 13 and 14). There were no obvious differences in species richness for gillnet samples, but seine results showed control sites had fewer species than reef ball or bagged oyster treatments. Furthermore, there were four species that were captured by seine on both breakwater treatments, but were never captured on control sites (Table 1).

Discussion

The shoreline at Helen Wood Park has been retreating over the past decade, which could be largely due to the increased armoring of adjacent shores. Moreover, the area of HWP receives frequent boat-driven wake due to the Mobile shipping channel and its close proximity to marinas and yacht clubs. The two primary goals of this project were: 1) evaluate the effectiveness of each breakwater reef as a means of mitigating coastal erosion, and 2) quantify the habitat value of each design. Maximizing the effect of shoreline protection would be best achieved by constructing breakwaters within close proximity to shore. Maximizing the habitat provided by the reefs would be greatest if the reefs were far enough from shore to be consistently submerged. This submersion could also enhance the suitability of the reefs for oyster recruitment. Given this logic and the fact that the shallows of HWP extend far from shore, it was quite difficult to obtain the best of both attributes.

The effectiveness of breakwater reefs at protecting coastal property relies upon several factors including design, sustainability, and location. The design of the bagged oyster shell was developed based upon previous breakwater projects of the University of South Alabama and the Dauphin Island Sea Lab. Insight gained from those projects led us to bag the shell as opposed to using the loose-shell mounds of previous efforts. This design appears to be more sustainable as there was no flattening or subsiding of the reefs evident during the eighteen month monitoring period. Several projects along the Florida gulf coast have found Reef Ball Lo-pro modules to be a successful means of protecting shorelines. This design also appears to be very sustainable as it also remained intact. More evidence supporting longevity for both reef designs came in the late summer and fall 2008, as tropical cyclones Gustav and Ike made landfall along the Alabama gulf coast. The area near Dog River and Helen Wood Park took a great deal of storm-driven waves and storm surges. The reefs themselves suffered zero damage from the storms, with the only losses being the PVC markers that outlined them.

It is important to recognize that site-specific attributes such as wave climate or adjacent shoreline types could have confounding effects on the success of such projects. As previously

mentioned, the wave climate at HWP can be rather intense and the shoreline is heavily armored in both directions from the study sites. The most probable area of improvement for the reef design and deployment would be reduced distance from shore. If the primary goal is shoreline protection, with less focus on the habitat provided, we recommend moving the breakwater structures to within 20-m from the shoreline if the same size reefs were to be used.

Oyster recruitment was largely absent on the reefs during the study. This could be a result of several factors. First, the upper west portion of Mobile Bay appears to be on the lower extreme of larval supply for oysters (Kim et al. In Review). Also, the bagged reefs and reef balls were considerably fouled by barnacles and turf algae, potential competitors for space. It is likely that the combination of these factors considerably impact oyster survival and explain the extremely low abundances observed.

We did not observe any difference of reef type on water clarity nor did we see any difference between control sites and breakwater reef sites. This could be a consequence of the relative large distance of the reefs to the shoreline. Additionally, the shallow nature of the sites and the fine particles that constitute the majority of the sediment could play a role in the lack of enhanced water quality as the fine particles are easily re-suspended. Although sampling dates were significantly different, they did not follow an expected seasonal pattern and the high variability was more attributed to short-term, localized weather events.

Our fisheries results revealed limited enhancement of fishes and mobile invertebrates. While the abundance of larger fishes did not appear to be enhanced, smaller fishes and invertebrates were more abundant near reefs than controls. There was also a more diverse assemblage of small fishes and invertebrates around the reefs. Furthermore, there were certain groups of fishes that were enhanced more by breakwater structures than others. Juveniles of the drum family Sciaenidae were found in substantially higher abundances in the structured treatments than controls (Table 1). Sciaenids includes many fishes of high recreational and economic value, including spotted seatrout, white trout, and croakers (Table 1). Constructing breakwater reefs could not only mitigate habitat losses caused by traditional armoring, but enhance certain economically-valuable stocks.

The lack of differences among larger fishes could reflect the small distances between our treatments. Ideally, this distance should have been larger, but the adjacent armoring limited the width of our treatment and spacing between them. Our reefs were 25-m long, with 50-m gaps between treatments. It is very likely that many of the fishes captured in gillnets travel and forage over much greater distances than those separating the treatments.

Utilizing oyster reefs as natural breakwaters for estuarine shores is continuing to gain support. Awareness and consensus of the detrimental impacts of shoreline armoring has

increased in recent years, but movement towards more ecologically-responsible methods has been limited by the lack of cost-effective alternatives. Our study experimentally tested two breakwater methods of “living-shoreline” restoration that could be promising alternatives. Both reef balls and bagged oyster shell reefs appear sustainable through even moderate tropical storms, but further research will help determine the optimal deployment configuration to maximize protection. We believe the results of this project results provide a framework for future coastal restoration studies to build upon.

Literature Cited

- Chambers JR (1991) Coastal Degradation and Fish Population Losses. In: Stroud R (ed) Stemming the Tide of Coastal Fish Habitat Loss. National Coalition for Marine Conservation, p 45-51
- Coen LD, Brumbaugh RD, Bushek D, Grizzle R, Luckenbach MW, Posey MH, Powers SP, Tolley G (2007) AS WE SEE IT: Ecosystem Services related to oyster restoration. *Marine Ecology Progress Series* 341:303-307
- Coen LD, Luckenbach MW, Breitburg DL (1999) The role of oyster reefs as essential fish habitat: a review of current knowledge and some new perspectives. *American Fisheries Society Symposium* 22:438-454
- Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M (1997) The value of the world's ecosystem services and natural capital. *Nature* 387:253-261
- Dame RF, Patten BC (1981) Analysis of energy flows in an intertidal oyster reef. *Marine Ecology Progress Series* 5:115-124
- Douglass SL, Pickel BH (1999) The tide doesn't go out anymore - the effect of bulkheads on urban shorelines. *Shore and Beach* 67:19-25
- Glancy TP, Frazer TK, Cichra CE, Lindberg WJ (2003) Comparative patterns of occupancy by decapod crustaceans in seagrass, oyster, and marsh-edge habitats in a Northeast Gulf of Mexico estuary. *Estuaries* 26:1291-1301
- Grabowski JH, Powers SP (2004) Habitat complexity mitigates trophic transfer on oyster reefs. *Marine Ecology Progress Series* 277:291-295
- Gunter G (1967) Some Relationships of Estuaries to the Fisheries of the Gulf of Mexico. In: Lauff GH (ed) *Estuaries*, Vol 83. American Association for the Advancement of Science, Washington, D.C., p 621-638
- Jackson JBC, Kirby MX, Berger WH, Bjorndal KA, Botsford LW, Bourque BJ, Bradbury RH, Cooke R, Erlandson J, Estes JA, Hughes TP, Kidwell S, Lange CB, Lenihan HS, Pandolfi JM, Peterson CH, Steneck RS, Tegner MJ, Warner RR (2001) Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293:629-638
- Kennish MJ (2002) Environmental threats and environmental future of estuaries. *Environmental Conservation* 29:78-107
- Meyer DL, Townsend EC (2000) Faunal utilization of created intertidal eastern oyster (*Crassostrea virginica*) reefs in the southeastern United States. *Estuaries* 23:34-35
- Meyer DL, Townsend EC, Thayer GW (1997) Stabilization and Erosion Control Value of Oyster Cultch for Intertidal Marsh. *Restoration Ecology* 5:93-99
- Minton MD (1999) Coastal Wetland Restoration and Its Potential Impact on Fishery Resources in the Northeastern United States. *American Fisheries Society Symposium* 22:405-420
- NRC (2007) Mitigating shore erosion along sheltered coasts., Vol. National Academies Press, Washington, D.C.
- Peterson CH, Grabowski JH, Powers SP (2003) Estimated enhancement of fish production resulting from restoring oyster reef habitat: quantitative valuation. *Marine Ecology Progress Series* 264:249-264
- Piazza BP, Banks PD, La Peyre MK (2005) The potential for created oyster shell reefs as a sustainable shoreline protection strategy in Louisiana. *Restoration Ecology* 13:499-506
- Schroeder WW, Wiseman Jr. WJ (1985) An Analysis of the Winds (1974-1984) and Sea Level Elevations (1973-1983) in Coastal Alabama. Mississippi-Alabama Sea Grant Consortium Publication MASGP-84-024:i-102

- Syvitski JPM, Vorosmarty CJ, Kettner AJ, Green P (2005) Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science* 308:376-380
- Thayer GW, McTigue TA, Bellmer RJ, Burrows FM, Merkey DH, Nickens AD, Lozano SJ, Gayaldo PF, Polmateer PJ, Pinit PT (2003) Science-based restoration monitoring of coastal habitats, Volume One: A framework for monitoring plans under the Estuaries and Clean Waters Act of 2000 (Public Law 160-457). NOAA Coastal Ocean Program Decision Analysis Series No 23:35 plus appendices
- Vitousek PM, Mooney HA, Lubchenco J, Melillo JM (1997) Human domination of Earth's ecosystems. *Science* 277
- Yund PO, Gaines SD, Bertness MD (1991) Cylindrical tube traps for larval sampling. *Limnology and Oceanography* 36:1167-1177
- Zimmerman RJ, Minello TJ, Baumer TJ, Castiglione MC (1989) Oyster reef as habitat for estuarine macrofauna

Figure 1. Map of the experimental layout at Helen Wood Park (HWP). Treatments are coded by direction (N= North, S= South) and reef type (O= Bagged Oyster Shell, R = Reef Ball modules, C= Control). All sites are approximately 60-m from the shore.

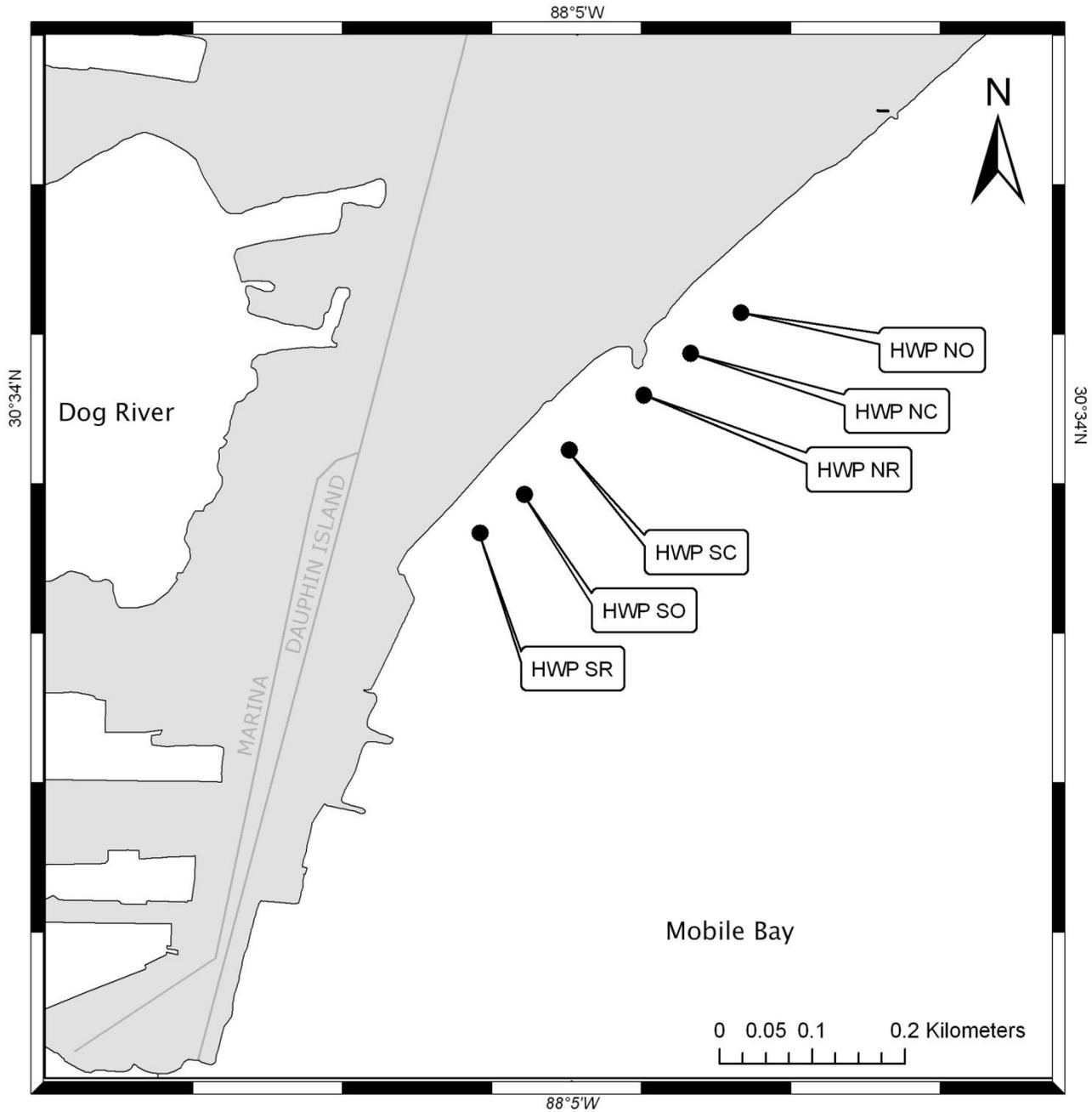


Figure 2. Graphical depiction of a Reef Ball Lo-pro module. The hollow concrete dome is constructed of marine-grade concrete and large holes provide refuge for many species.



Figure 3a. Labor hours required to bag and deploy oyster shell versus the deployment time of the pre-fabricated Reef Balls.

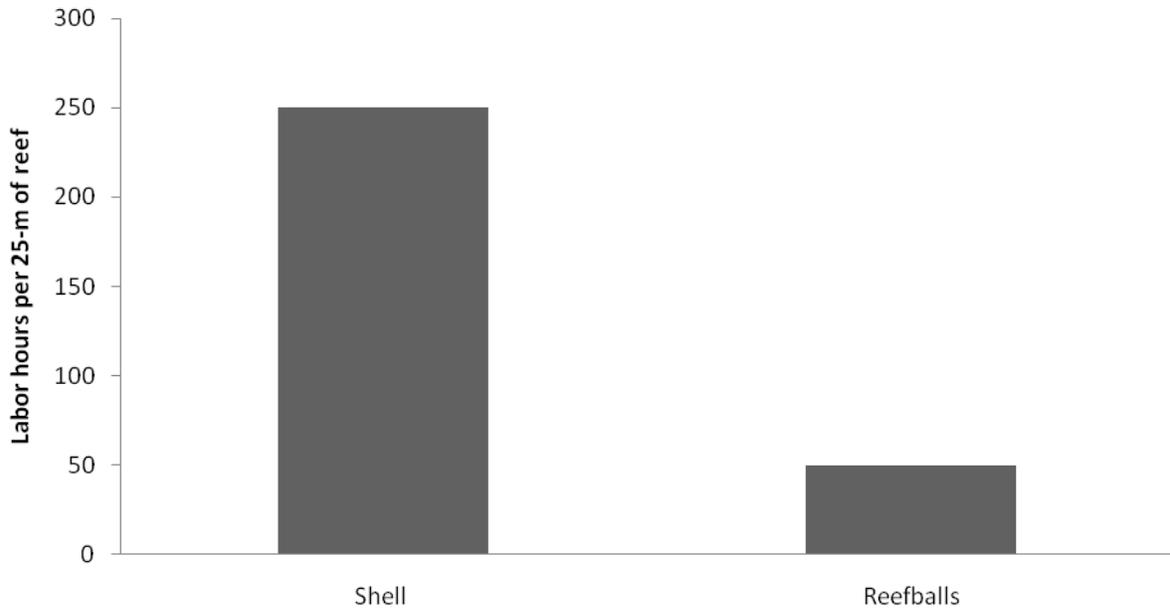


Figure 4b. Cost of materials and delivery for Reef Ball modules versus loose oyster shell and bagging supplies. No labor costs are reflected in the figure.

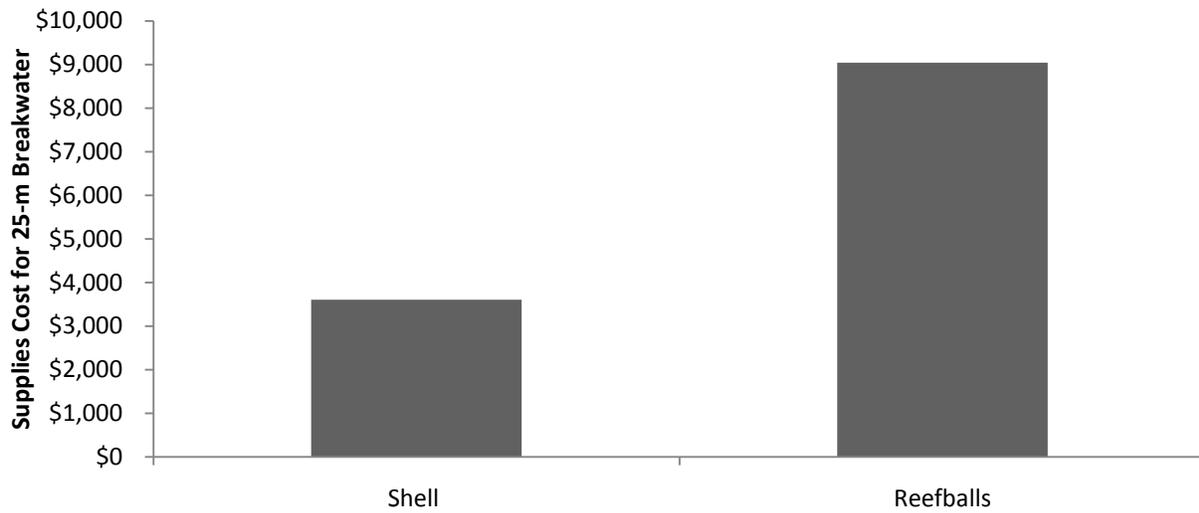


Figure 5. Aerial of Helen Wood Park breakwaters provided by ADCNR-State Lands Division. Treatments visible from North to South are North Oyster, North Reef Ball, South Oyster, South Reef Ball. North control is located between the visible breakwaters, and South Control is just North of the two southernmost breakwaters.



Figure 6. Percent loss in mass of gypsum spheres deployed to measure relative water flow. Bars represent standard error.

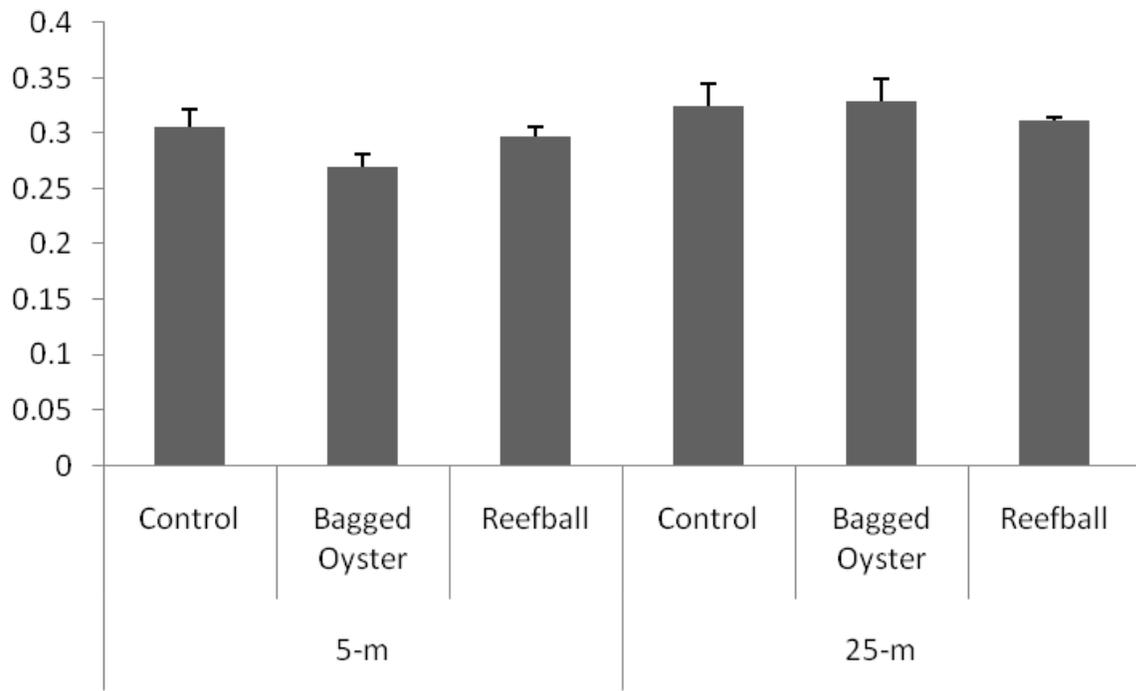


Figure 6. Density of Mussels found on each reef type at the 18 month sampling event.

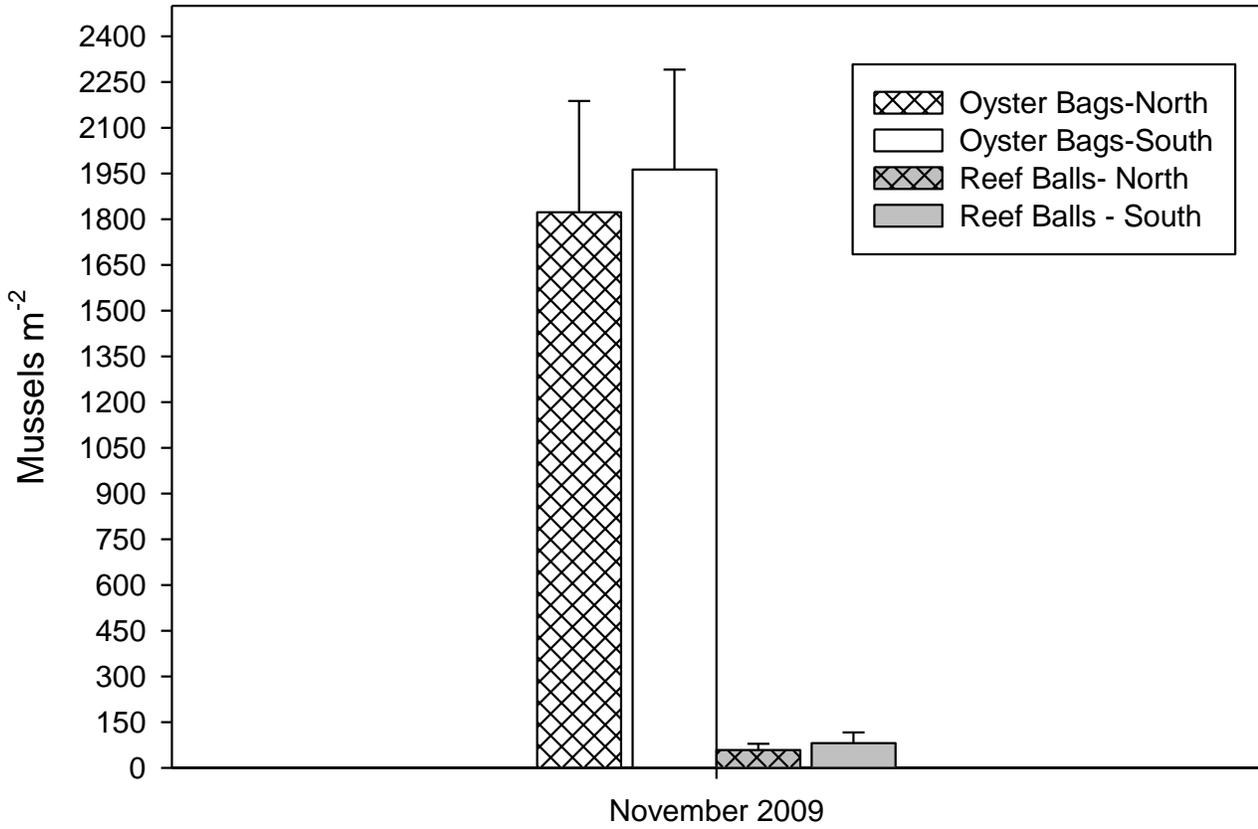


Figure 7. Average water column total suspended solids for the control areas (dark grey triangles), the bagged shell breakwater reefs (black diamonds) and reef ball breakwater reefs (grey square).

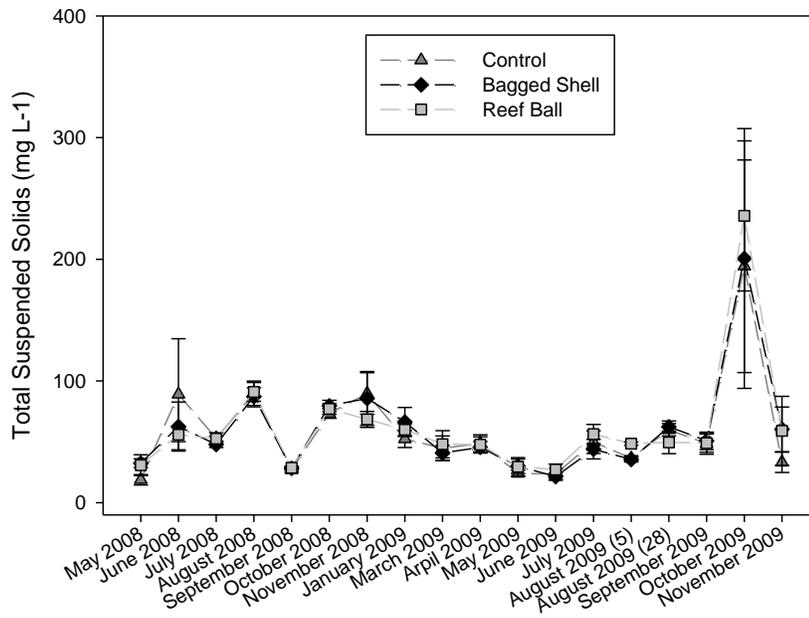


Figure 8: Phytoplankton abundance, measured by chlorophyll a concentration in the water column for the control areas (dark grey triangles), the bagged shell breakwater reefs (black diamonds) and reef ball breakwater reefs (grey square).

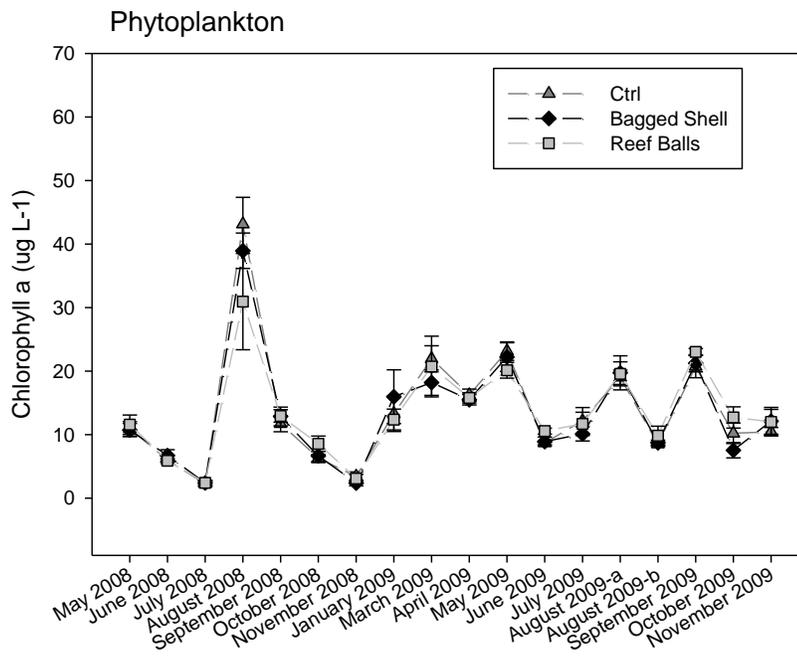


Figure 9: Percent of Photosynthetically active radiation (PAR) at the mid station for each treatment. Circles represent PAR on the benthos relative to the amount of light just below the water's surface, triangles represent PAR on the benthos relative to the incident light.

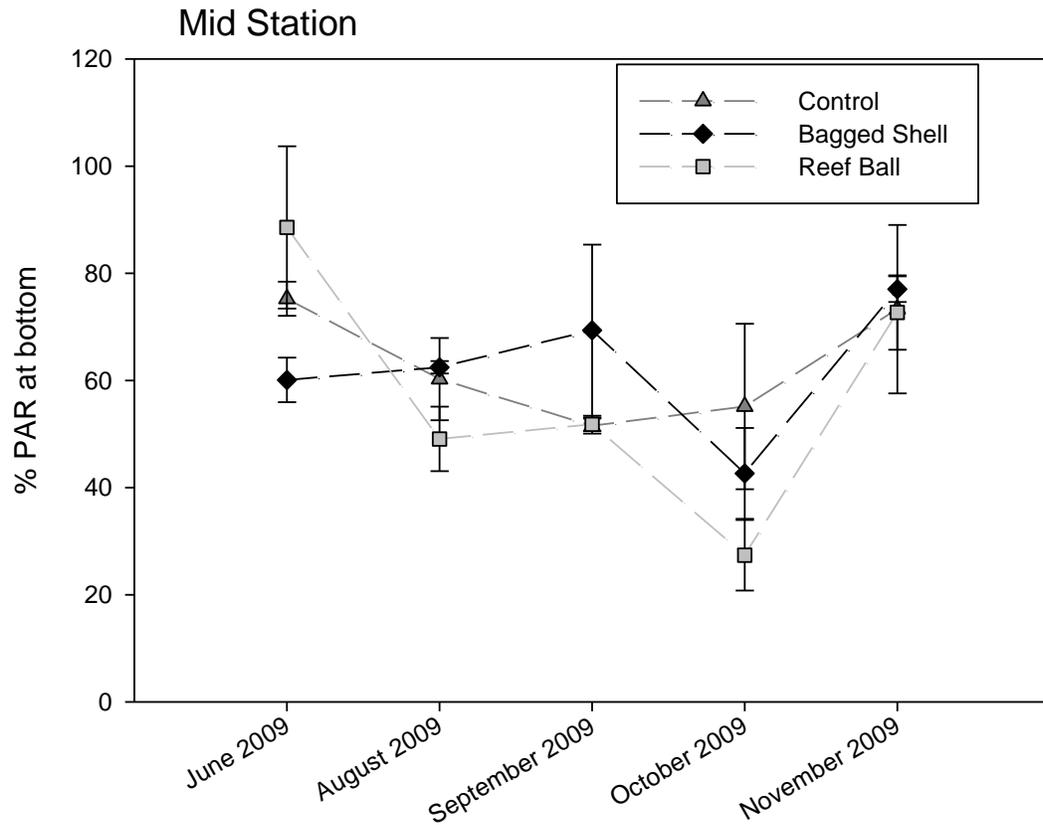


Figure 10. CPUE abundance for each treatment by gear type. Gillnet catch is presented by 5-cm and 10-cm mesh.

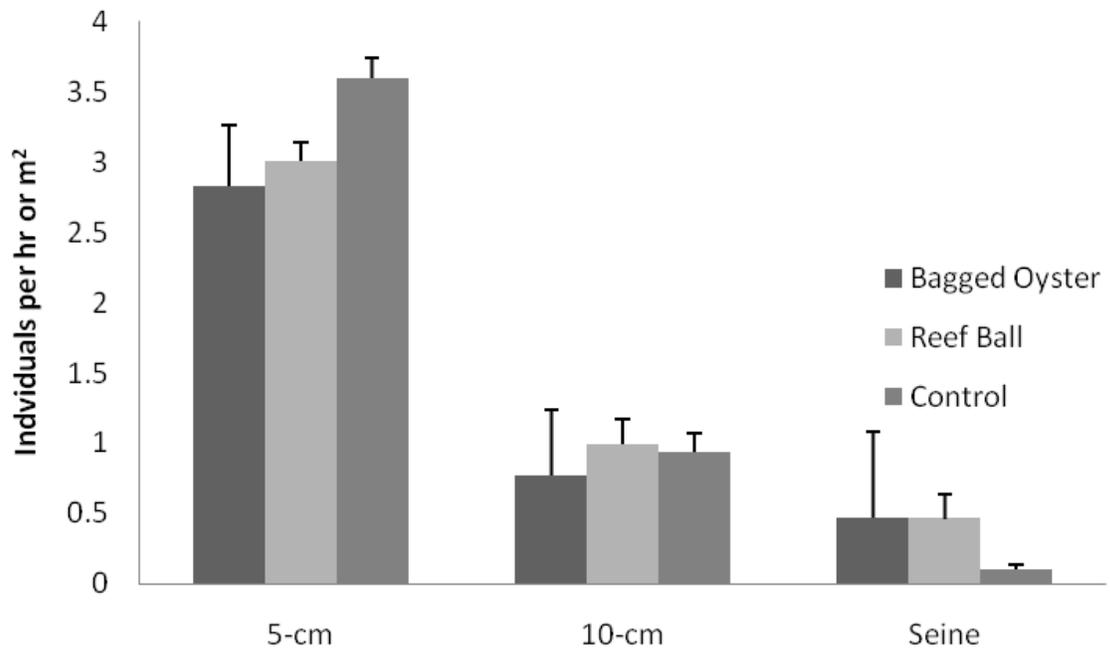


Figure 11. CPUE abundance for demersal fishes.

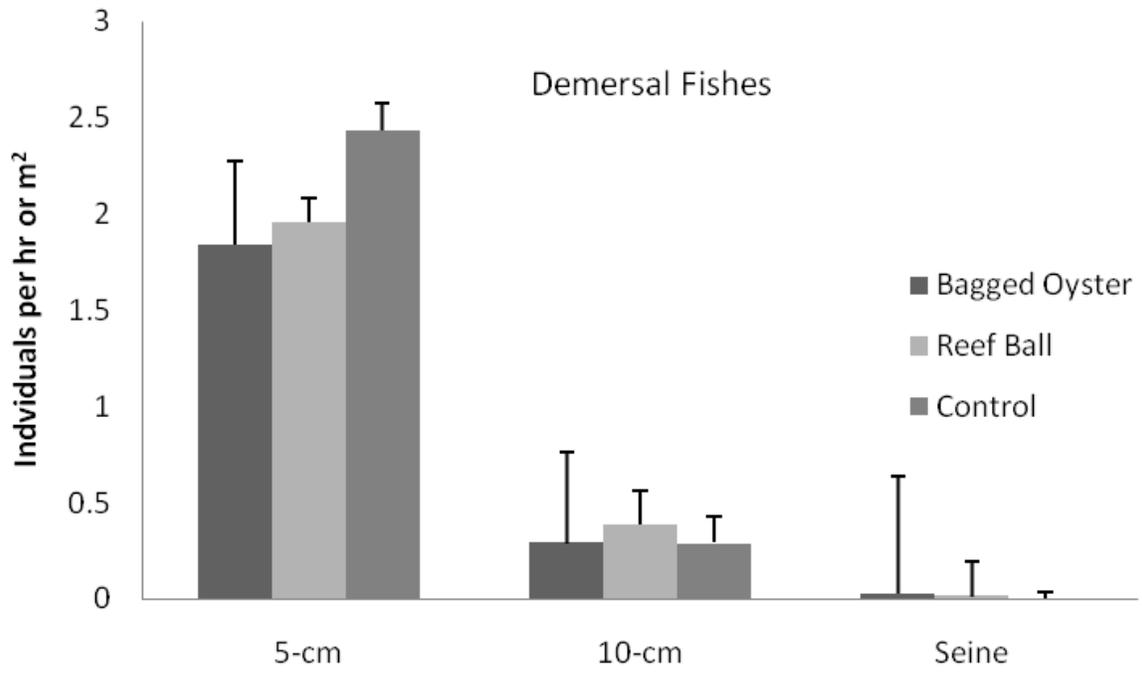


Figure 12. CPUE abundance results of seining by sampling event for each treatment.

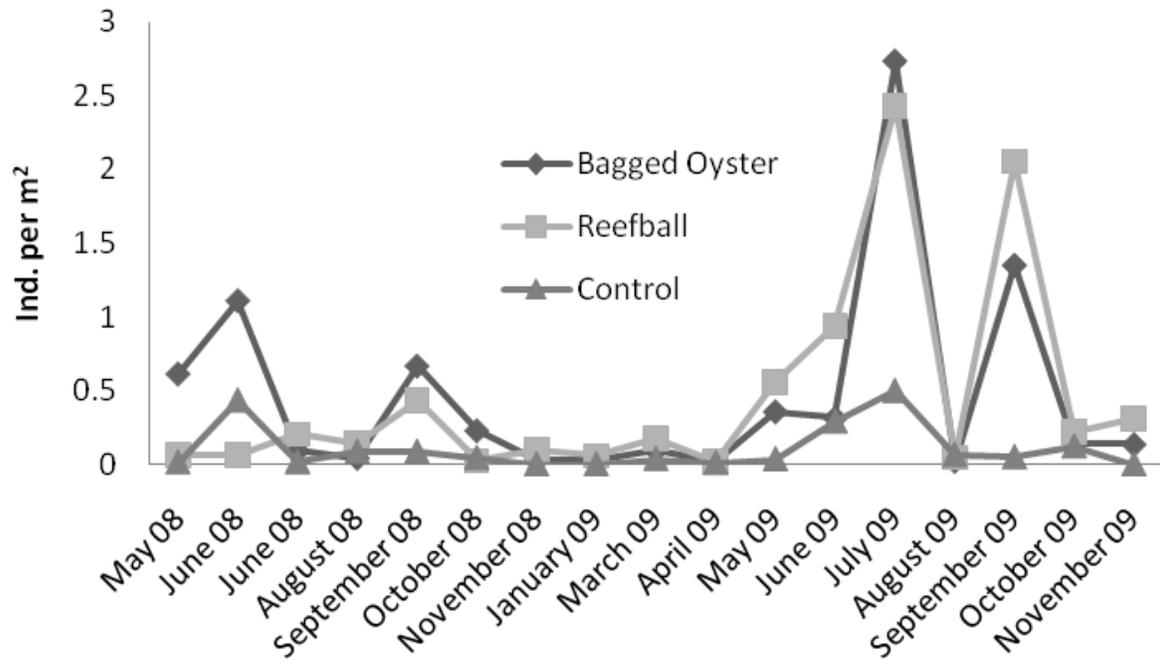


Figure 13. Species richness by gear type for each treatment.

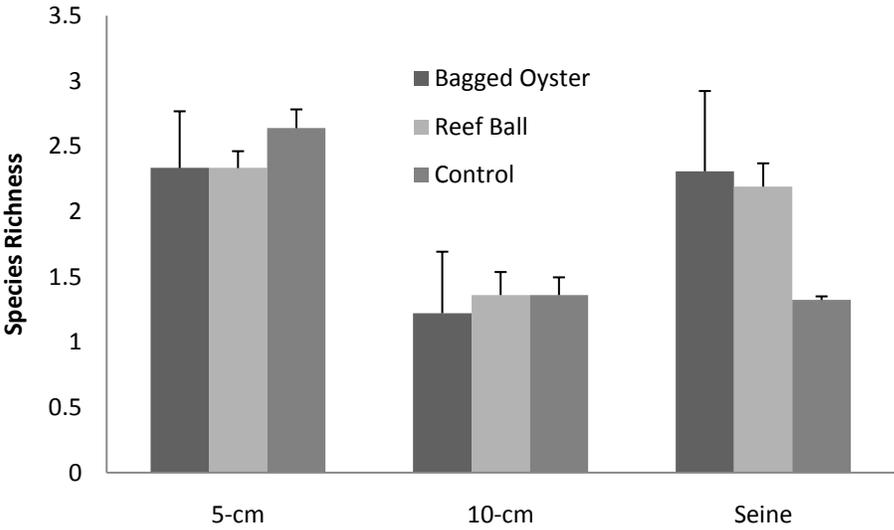


Figure14. Species richness of seining results for each treatment throughout the sampling period.

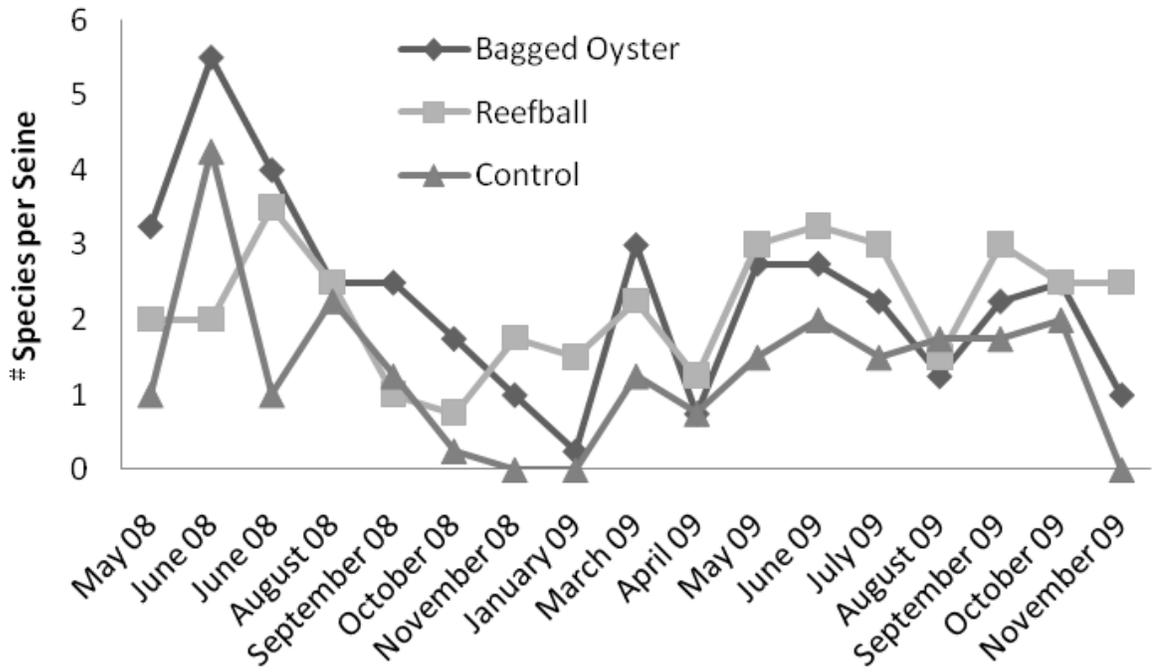


Table 1. All fishes and mobile invertebrate taxa that represent greater than one percent of the total catch for that gear. Environment category was designated by FISHBASE.

	Common Name	Scientific Name	Family	Environment	5-cm			10-cm			Seine						
					Control	Oyster	Reef Ball	Control	Oyster	Reef Ball	Control	Oyster	Reef Ball				
<i>Fishes</i>	Anchovy	Anchoa sp.	Engraulidae	Pelagic													
	Black drum	Pogonias cromis	Sciaenidae	Demersal			1	3	2	2							
	Croaker	Micropogonias undulatus	Sciaenidae	Demersal	124	102	113	10	5	5			7			16	
	Flounder	Paralichthys sp.	Paralichthyidae	Demersal		1	1	3	3	1							
	Gizzard shad	Dorosoma cepedianum	Clupeidae	Pelagic	1			3	2	5							
	Hardhead catfish	Ariopsis felis	Ariidae	Reef-associated	5	6	13	14	12	7			12				45
	Ladyfish	Elops saurus	Elopidae	Reef-associated	4	5	3		1	1							
	Menhaden	Brevoortia patronus	Clupeidae	Pelagic	46	27	33	25	13	22			2				169
	Mullet	Mugil sp.	Mugilidae	Pelagic	15	20	19	5	5	4			3	25			18
	Pinfish	Lagodon rhomboides	Sparidae	Demersal	4	1							7	17			8
	Red drum	Sciaenops ocellatus	Sciaenidae	Demersal	1	3	2	2	9	8							
	Sheepshead	Archosargus probatocephalus	Sparidae	Reef-associated				3	2	5							1
	Silver perch	Bairdiella chrysoura	Sciaenidae	Demersal	14	13	6						21	54			26
	Skipjack herring	Alosa chrysochloris	Clupeidae	Pelagic	11	15	13										
	Southern kingfish	Menticirrhus americanus	Sciaenidae	Demersal	9	2			1	7							
	Spadefish	Chaetodipterus faber	Ephippidae	Reef-associated		1		1									
	Spanish mackerel	Scomberomorus maculatus	Scombridae	Reef-associated	2			1									
	Speckled trout	Cynoscion nebulosus	Sciaenidae	Demersal	8	5	5	5	1	9							
	Spot	Leiostomus xanthurus	Sciaenidae	Demersal	26	11	24	4	1				2	74			
	Threadfin shad	Dorosoma petenense	Clupeidae	Pelagic	3	3	3	1	3	1			4	4			13
White trout	Cynoscion arenarius	Sciaenidae	Demersal	1	3	2		2					2			9	
Juvenile Sciaenids		Sciaenidae	Demersal									3	18			41	
Silversides	Menidia sp.	Atherinopsidae	Pelagic									49	169			140	
Juvenile Clupeids		Clupeidae	Pelagic									2	3			4	
<i>Invertebrates</i>	Caridean shrimp			Decapod								26	115			26	
	Penaeid shrimp			Decapod								10	26			19	
	Total Catch				277	221	238	82	63	78		742	3320			3308	

Table 2a. Two-factor ANOVA table for total catch. Treatment and season were fixed-factor independent variables.

<i>Total Catch</i>	5-cm					10-cm					Seine				
	df	SS	MS	F	P	df	SS	MS	F	P	df	SS	MS	F	P
Treatment	2	13.39	6.69	0.77	0.46	2	1.56	0.78	0.79	0.46	2	0.55	0.27	3.10	0.05
Season	3	96.89	32.3	3.73	0.01 *	3	12.59	4.20	4.23	0.01 **	3	0.91	0.30	3.44	0.02 *
Treatment:Season	6	31.87	5.31	0.61	0.72	6	5.05	0.84	0.85	0.54	6	0.13	0.02	0.25	0.96
Error	101	874.76	8.66			101	100.13	0.99			60	5.31	0.09		

Table 2b. Two-factor ANOVA table for demersal fishes. Treatment and season were fixed-factor independent variables.

<i>Demersal Fishes</i>	5-cm					10-cm					Seine				
	df	SS	MS	F	P	df	SS	MS	F	P	df	SS	MS	F	P
Treatment	2	7.17	3.58	0.99	0.38	2	0.23	0.11	0.37	0.69	2	0.00	0.00	1.47	0.24
Season	3	43.92	14.64	4.03	0.01 **	3	0.43	0.14	0.46	0.71	3	0.00	0.00	0.91	0.45
Treatment:Season	6	9.68	1.61	0.44	0.85	6	1.43	0.24	0.76	0.60	6	0.01	0.00	0.78	0.59
Error	96	349.06	3.64			96	29.96	0.31			39	0.05	0.00		