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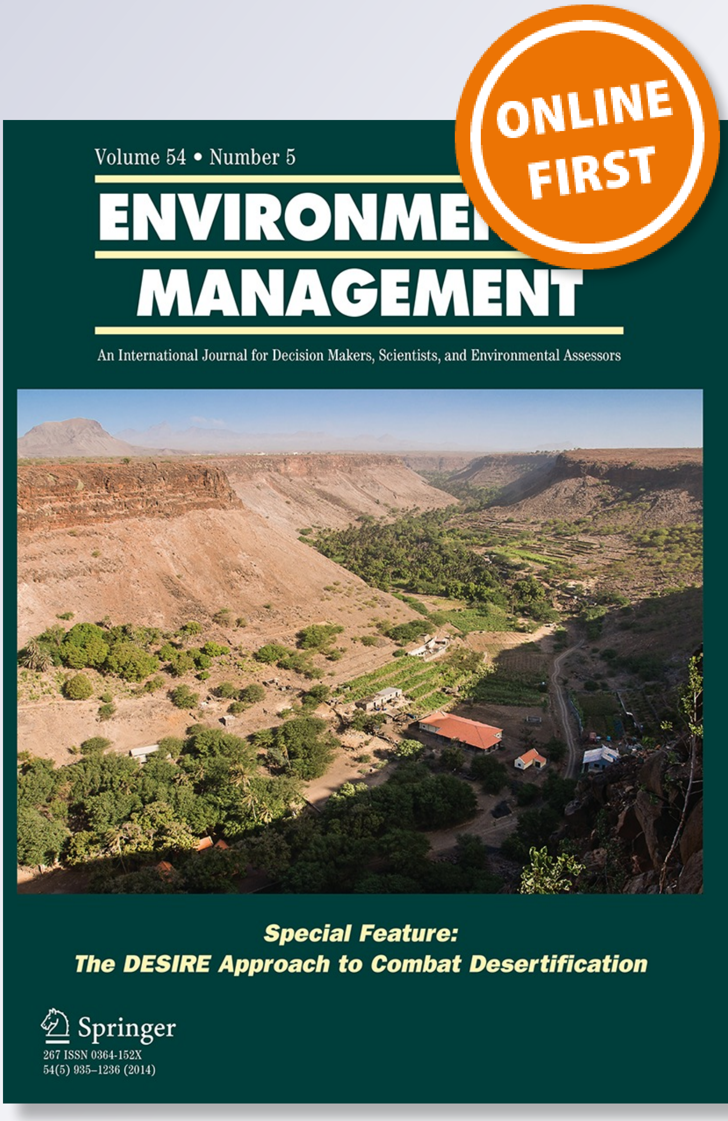

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Abstract Estuarine shorelines have been degraded since humans arrived in the coastal zone. In recent history, a major cause of habitat degradation has been the armoring of shorelines with vertical walls to protect property from erosive wave energy; however, a lack of practical alternatives that maintain or enhance ecological function has limited the options of waterfront residents and coastal zone managers. We experimentally investigated the habitat value of two configurations of submerged breakwaters constructed along an eroding shoreline in northwest Mobile Bay, AL (USA). Breakwaters comprised of bagged oyster shell or Reef Ball™ concrete domes were built by a community-based restoration effort. Post-deployment monitoring found that: bagged oyster breakwaters supported much higher densities of live ribbed mussels than Reef Ball breakwaters; both breakwater configurations supported increased species richness of juvenile and smaller fishes compared to controls; and that larger fishes did not appear to be affected by breakwater presence. Our study demonstrates that ecologically degraded shorelines can be augmented with small-scale breakwaters at reasonable cost and that these complex structures can serve as habitat for filter-feeding bivalves, mobile invertebrates, and young fishes. Understanding the degree to which these structures mitigate erosive wave energy and protect uplands will require a longer time frame than our 2-year-long study.

Keywords Community-based restoration · Ecosystem engineers · Fisheries · Living shorelines · Oyster reef · Participatory management

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Introduction

The shorelines of coastal and estuarine ecosystems have been increasingly transformed to meet the desires of the dense human populations that reside along them (Halpern et al. 2008; Pilkey and Wright 1988; Vitousek et al. 1997). Along many sheltered and densely populated coastlines, residential development has often been followed by increased shoreline erosion and the construction of vertical barriers such as seawalls (NRC 2007), which can detrimentally affect estuarine hydrodynamics, water quality, sediment transport, and essential fish habitats (Bilkovic and Roggero 2008; Douglass and Pickel 1999; Mallin et al. 2000; Syvitski et al. 2005). In recent decades, a wider variety of alternative approaches to coastal protection have been developed (NRC 2007), but very few studies have assessed their ecological impacts on scales relevant to coastal residents (Pilkey and Cooper 2012).

In Mobile Bay, Alabama (USA), shoreline armoring has increased by approximately 0.5 % per year since 1955

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(Douglass and Pickel 1999), with 38.4 % of the bay's shorelines armored as of 2009 (Jones et al. 2009). Vertical walls and bulkheads characterize more than 25 % of the bay's shorelines (Jones et al. 2009) and have been considered the most ecologically damaging approach to shoreline stabilization (e.g., Douglass and Pickel 1999; NRC 2007; Bilkovic and Roggero 2008). While slightly more than 60 % of the bay's shorelines still remain natural or vegetated, more than 90 % of the bay is currently experiencing some erosion (Jones et al. 2009). Although coastal engineering studies have indicated that much of Mobile Bay experiences wave climates higher than critical limits at which native *Spartina alterniflora* could survive, submerged breakwaters have been suggested as a potential means for reducing wave energy and enhancing marsh viability (Roland and Douglass 2005).

Submerged breakwaters, which are structures deployed to reduce wave energy at the shoreline, can be constructed from a variety of materials including rock, granite, concrete (Borsje et al. 2011; NRC 2007), or oyster shell (Allen and Webb 2011; Scyphers et al. 2011). In heavily impacted systems, breakwaters may provide complex, structured habitat in otherwise featureless settings where native habitats have been lost to erosion (Borsje et al. 2011). When designed to incorporate or support the recruitment of oysters and other filter-feeding bivalves, these structures may provide an even broader array of ecological functions and services. For instance, the ecosystem services provided by oyster reefs include the provision of essential habitat and foraging grounds for numerous species of fishes and mobile invertebrates (e.g., Coen et al. 1999; Grabowski et al. 2005; Humphries et al. 2011; La Peyre et al. 2014; Peterson et al. 2003), water filtration, benthic-pelagic coupling, and enhanced denitrification (e.g., Beseres Pollack et al. 2013; Dame 1996; Newell 2004; Piehler and Smyth 2011), as well as wave attenuation and erosion control (Meyer et al. 1997; Piazza et al. 2005; Scyphers et al. 2011).

The majority of studies on the ecological value and efficacy of breakwaters have been on scales that are logistically or financially prohibitive for many coastal property owners (NRC 2007; Pilkey and Cooper 2012). To experimentally evaluate breakwaters on a scale relevant to coastal residents, we conducted a manipulative field experiment and examined the habitat value of residential scale breakwaters constructed of Reef Ball™ Low-Pro modules (Reef Balls) and bagged oyster shell (*Crassostrea virginica*) (Fig. 1). We hypothesized that breakwater treatments would: (1) provide substrate for bivalve recruitment and (2) support higher densities and species richness of fishes and mobile macro-invertebrates than mudflat control treatments. We also initially hypothesized that the presence of breakwaters would mitigate shoreline vegetation retreat compared to unaltered control plots; however, several unexpected events prevented us from effectively testing this hypothesis.

Materials and Methods

Site Description and Experimental Design

We conducted the experiment along an eroding shoreline in northwest Mobile Bay, Alabama (Fig. 2). The 0.63 km shoreline along the study site at Helen Wood Park (HWP) is comprised of retreating marsh (*Spartina alterniflora* and *Phragmites australis*) just north of the mouth of Dog River. The park is bordered by a large bulkhead and concrete bridge rubble on opposite ends. The majority of wind-driven waves at this site are derived from the south during spring and summer months, and from a more northerly direction during winter (Roland and Douglass 2005; Schroeder and Wiseman 1985). Three years of hydrographic data (2008–2011) from a monitoring station 16 km southeast of the study site indicated that mean annual temperature was 22.0 °C, salinity was 12.3 PSU, and dissolved oxygen was 7.5 mg/L^3.

Concrete domes and bagged oyster shell have both been utilized to stabilize shorelines and mitigate habitat losses (Harris 2003; Meyer et al. 1997; Piazza et al. 2005). The "Reef Ball" breakwaters contained 3 rows of 41 Reef Ball™ Lo-Pro modules deployed parallel to the retreating marsh shoreline without spacing between adjacent modules (Fig. 1). Reef Ball™ Lo-Pro modules are hollow concrete domes with several circular openings in the sides and top. Each module was 0.6 m in diameter at the base and 0.5 m tall with an external surface area of 1.5 m^2. The base footprint of each Reef Ball breakwater measured 2 m by 25 m. The "bagged oyster shell" breakwaters were comprised of approximately 2,000 bags filled with 0.025 m^3 of oyster shell purchased from a local seafood processor. The shell bagging material was comprised of polyethylene mesh with 1.6 cm diamond shaped openings (Atlantic Aquaculture, Inc.). When placed on the reef, each shell bag was approximately 0.2 m wide and 0.75 m long, with an exposed surface area of approximately 0.15 m^2. Each breakwater was constructed as a 0.5 m tall pyramid-shaped reef with a base footprint of 2 m wide and 25 m long, narrowing to 1 m wide at the crest. Reef Ball Low-Pro modules were prefabricated by an authorized contractor and transported to the study site for deployment. Loose oyster shell was purchased from a local seafood processor and delivered to the site by dump trucks where community volunteers conducted bagging and deployment. The Reef Balls and oyster shell bags were deployed by small boats and volunteers and placed parallel to the shoreline in May

Fig. 1 Photographs of project staging and deployment of (**a**, **c**) Reef Ball and (**b**, **d**) bagged oyster shell breakwaters

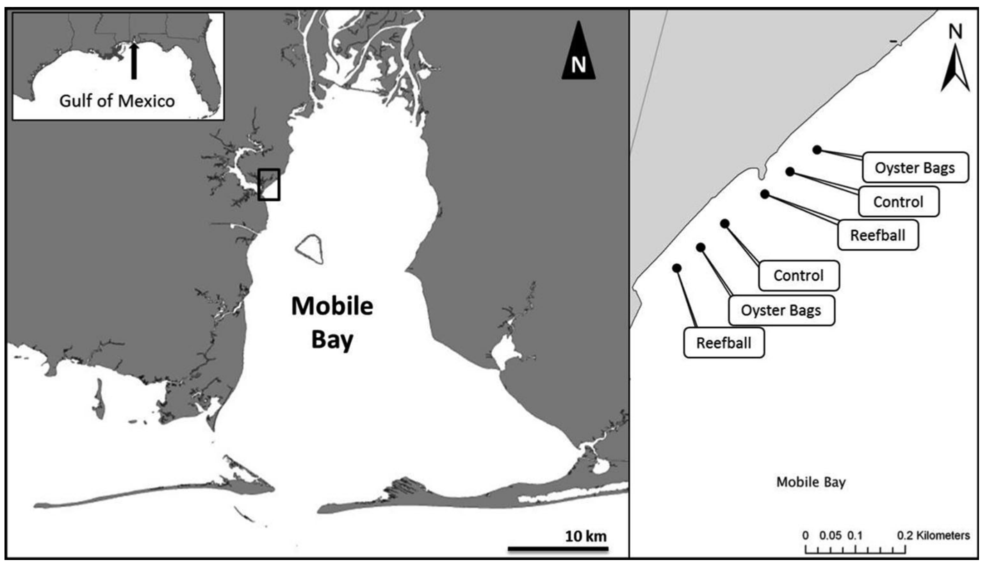

Fig. 2 Map of the experimental design at Helen Wood Park (HWP). All treatments are approximately 60 m from the shore

2008. Identical areas of mudflat were designated as "control" treatments and were marked with PVC pipe but lacked any other structure. The study site was divided into north and south zones separated by a 100 m buffer, and treatments were randomly assigned within each zone (Fig. 2). The reefs were set at 0.75 m depth (Mean Lower Low Water), which was located within 60 m from shore.

Bivalve Recruitment

To measure oyster and other bivalve recruitment, we quantitatively sampled the breakwaters three times over a 2-year-period following deployment. For Reef Balls, nine modules were randomly selected, removed from the water, and all external surfaces were completely inspected. For bagged shell breakwaters, nine bags from the external surface of each breakwater were haphazardly chosen and removed. The contents of the bags were emptied into a bucket, and all shells were examined for live oysters (*Crassostrea virginica*) and ribbed mussels (*Geukensia demissa*). After sampling, the oyster shell was re-bagged and all components were returned to the breakwaters. Bivalve densities were standardized with respect to the exposed surface area of each module or bag and are reported as individuals per square meter.

Fishes and Mobile Invertebrates

To quantify the habitat value of the breakwaters, we used multiple sampling techniques to survey finfish and macroinvertebrate communities monthly from May 2008 until November 2009. We utilized 30 m experimental gillnets to collect a diverse assemblage of larger fishes. Each 30 m gillnet was comprised of two different mesh sizes (5 and 10 cm stretched openings, 15 m segment length) and was deployed directly offshore (<1 m) and diagonal to each treatment beginning at a randomly selected end of the treatment. The nets were fished for approximately 2 h (1 h before and after sunrise or sunset) and were retrieved in the same order they were deployed. Soak times were calculated as the total time from which each net was first deployed until retrieval began. We used a 6-m-wide bag seine with 6.25 mm mesh to quantify smaller finfishes and macro-invertebrates. Seining was conducted adjacent to the inshore and offshore sides of each breakwater or control treatment and terminated into a 4-m-wide block net. Seine distances were always 12.5 m, and all specimens captured were placed on ice and returned to the lab where they were identified to Family or lower taxonomic level and enumerated. Catch per unit effort (CPUE) was calculated as individuals collected per hour for gillnet data and individuals collected per m^2 for the seine data.

Statistical Analyses

We used univariate statistics to assess the habitat value of breakwaters for bivalves, mobile invertebrates, and fishes. Bivalve densities on each breakwater treatment were compared using *t* tests. Gillnet and seine catches were analyzed using univariate permutational analysis of variance (PERMANOVA) to examine the effects of treatment (bagged oyster, Reef Ball, control) and season on total abundance (CPUE) and species richness for each gear type. All PERMANOVA tests were run on Euclidean distances matrices (with 4,999 permutations). Univariate PERMANOVA is similar to traditional analysis of variances (ANOVA) as it allows for two factor designs and considers the interaction of factors, but differs as it is not sensitive to non-normal distributions (Anderson 2001a, b).

Results

Bivalve Recruitment

We sampled bivalve densities three times over 2 years following deployment. A total of 20 oysters were documented on bagged oyster breakwaters, and only two live oysters were found on Reef Balls. Mussel densities were higher on both breakwater treatments. Standardized mean mussel densities were much higher on the bagged oyster breakwaters with mean densities greater than 2,500 m^{-2}, while densities on Reef Balls were 13.7 m^{-2} (Fig. 3; df = 10, $t = 5.602$, $P < 0.001$).

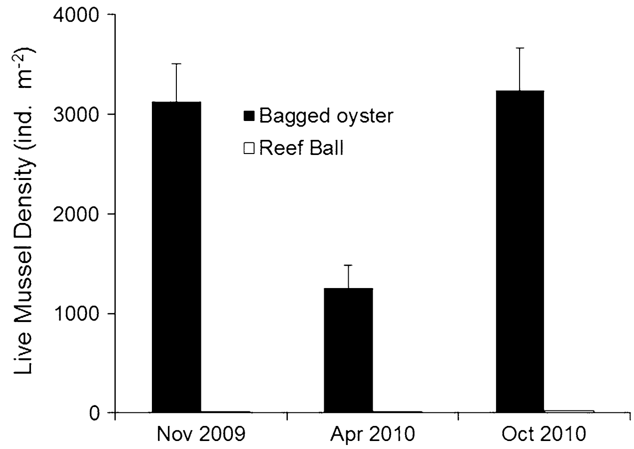

Fig. 3 Density of ribbed mussels (*Geukensia demissa*) for each breakwater treatment

Fishes and Mobile Invertebrates

The multiple sampling gears effectively captured diverse fish and invertebrate assemblages adjacent to the breakwaters and control plots. Gillnets captured more than 30 different species of fishes, and seining for smaller fishes and invertebrates resulted in more than 35 species (Table 1). Four species were captured by seines on both breakwater treatments that were never captured on control sites, and all species captured by seines in control areas were also captured adjacent to one of the breakwater treatments. Approximately 66 % of the smaller mesh (5 cm) catch was comprised of demersal fishes, while pelagic species accounted for approximately 25 % (Table 1). Of the larger species captured by 10 cm mesh, pelagic and demersal species were the most common fishes. Pelagic species dominated more than 89 % of seine catches across all treatments, while demersal fishes accounted for less than 6 %. Fishes and crustaceans of significant recreational or commercial importance in the Gulf of Mexico (NMFS 2009) accounted for more than 33 % of non-pelagic catches across all gear types and treatments, and the dominant species were Atlantic Croaker (*Micropogonias undulatus*), Gulf Menhaden (*Brevoortia patronus*), and Spot (*Leiostomus xanthurus*) (Table 1).

The effects of treatment and season on the total abundance and species richness of animals captured was assessed using two-factor PERMANOVA. Treatment was only a significant factor for the species richness of seine catches (Table 2), which was higher on each breakwater treatment than control plots (Fig. 4a, c). The influence of season was significant for both total abundance and species richness for all three gear types (Table 2). Spring gillnet catches generally displayed the lowest CPUE values of the

Table 1 All fishes and mobile invertebrate taxa captured during seine and gillnet sampling separated by treatment: control (C), bagged oyster (O), and Reef Ball (RB)

Common name	Scientific name			5-cm			10-cm			Seine		
		Group	EI	C	O	RB	C	O	RB	C	O	RB
Alabama Shad	*Alosa alabamae*	P					2	1	1			
Skipjack Herring	*Alosa chrysochloris*	P		11	15	13						
Anchovies	*Anchoa* sp.	P								603	2,759	2,734
Sheepshead	*Archosargus probatocephalus*	RA	R				3	2	5			1
Hardhead Catfish	*Ariopsis felis*	RA		5	6	13	14	12	7		12	45
Silver Perch	*Bairdiella chrysoura*	D		14	13	6				21	54	26
Gulf Menhaden	*Brevoortia patronus*	P	C	46	27	33	25	13	22	2		169
Jack Crevalle	*Caranx hippos*	RA	R	1								1
Spadefish	*Chaetodipterus faber*	RA			1		1					
Atlantic Bumper	*Chloroscombrus chrysurus*	P										9
Sand Seatrout	*Cynoscion arenarius*	D	R	1	3	2		2			2	9
Spotted Seatrout	*Cynoscion nebulosus*	D	R	8	5	5	5	1	9			
Gizzard Shad	*Dorosoma cepedianum*	P		1			3	2	5			
Threadfin Shad	*Dorosoma petenense*	P		3	3	3	1	3	1	4	4	13
Ladyfish	*Elops saurus*	RA		4	5	3		1	1			
Scaled Sardine	*Harengula jaguana*	RA			2					1	12	13
Pinfish	*Lagodon rhomboides*	D		4	1					7	17	8
Spot	*Leiostomus xanthurus*	D		26	11	24	4	1		2	74	
Spotted Gar	*Lepisosteus oculatus*	D		1								
Silversides	*Menidia* sp.	P								49	169	140
Southern Kingfish	*Menticirrhus americanus*	D	R	9	2			1	7			
Atlantic Croaker	*Micropogonias undulatus*	D	R	124	102	113	10	5	5		7	16
Striped Mullet	*Mugil cephalus*	P	R	11	9	11	1	5	4		6	12
White Mullet	*Mugil curema*	RA	R	4	3	5	4				3	2
Mullet	*Mugil* sp.	P	R		8	3				3	16	4
Leatherjacket	*Oligoplites saurus*	RA								3	6	5
Flounder	*Paralichtys* sp.	D	R/C		1	1	3	3	1			
Harvestfish	*Peprilus alepidotus*	P										2
Black Drum	*Pogonias cromis*	D				1	3	2	2			
Bluefish	*Pomatomus saltatrix*	P	R		1							
Red Drum	*Sciaenops ocellatus*	D	R	1	3	2	2	9	8			
Spanish Mackerel	*Scomberomorus maculatus*	RA	R/C	2			1					
Puffers	*Sphoeroides* sp.	D									1	
Needlefish	*Strongylura marina*	RA									3	
Tonguefish	*Symphurus* sp.	D									1	
Pipefish	*Syngnathus* sp.	RA									4	
Inshore Lizardfish	*Synodus foetens*	RA									1	
Hogchoker	*Trinectes maculatus*	D		1								1
Juvenile Clupeids		P								2	3	4
Juvenile Sciaenids		D	R							3	18	41
Blue crab	*Callinectes sapidus*		R/C							6	7	8
Caridean shrimp										26	115	26
Penaeid shrimp			R/C							10	26	19
Total Catch				277	221	238	82	63	78	742	3,320	3,308
Effort				457.2	457.2	457.2	457.2	457.2	457.2	15,300	15,300	15,300

Gillnet effort is presented as total hours of soak time, and seine effort is total m^2. Environment groups (Group) of demersal (D), pelagic (P), and reef-associate (RA) were designated by FISHBASE. Economic importance (EI) for commercial (C) and recreational (R) fisheries were determined using the National Marine Fisheries Services Fisheries Economics Report of 2009

Table 2 Results of univariate PERMANOVA tests on total abundance and species richness for each gear type

	df	5-cm F^*	5-cm $P^\dagger$	10-cm F^*	10-cm $P^\dagger$	df	Seine F^*	Seine $P^\dagger$
Total abundance								
Treatment	2	1.057	0.3476	0.669	0.5122	2	0.813	0.4536
Season	3	3.699	0.0154*	4.795	0.0040**	3	4.147	0.0126*
Treatment:season	6	0.529	0.7778	0.896	0.4996	6	0.178	0.9812
Error	96					90		
Species richness								
Treatment	2	0.424	0.6494	0.090	0.9174	2	3.478	0.0410*
Season	3	6.404	0.0004***	5.050	0.0032**	3	5.090	0.0022*
Treatment:season	6	1.042	0.4020	0.458	0.8370	6	0.438	0.8476
Error	96					90		

* $P \leq 0.05$, ** $P \leq 0.01$, *** $P \leq 0.001$

The test statistic (F^*) is a pseudo-F value and the probability values ($P^\dagger$) are computed by the PERMANOVA routine

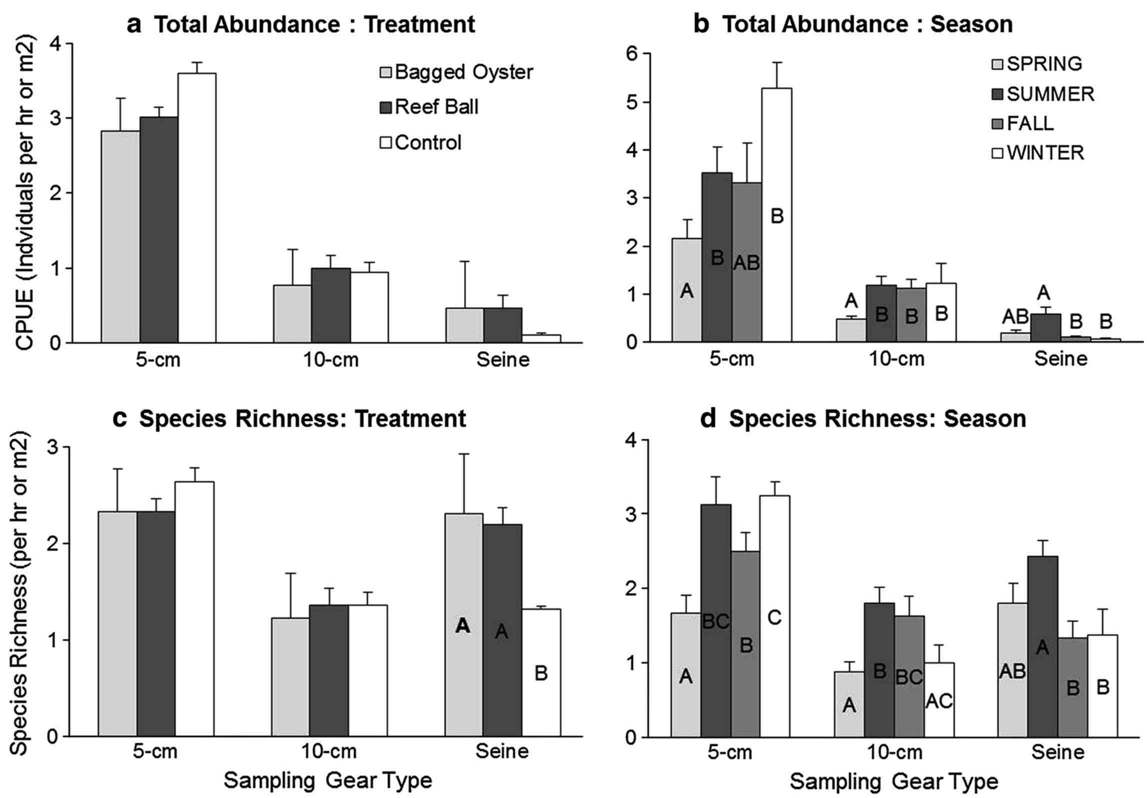

Fig. 4 CPUE abundance (**a**, **b**) and species richness (**c**, **d**) across treatments and seasons by gear type. Gillnet catch is separated by 5 and 10 cm mesh. *Different letters* indicate statistically different treatments from pairwise comparisons

year, although fall and spring were similar for the 5 cm mesh size (Fig. 4b). Seine catch CPUE was highest in summer, but was not significantly different from spring collections. For gillnet catches, species richness was also lowest during spring, which was indistinguishable from fall for 10 cm mesh (Fig. 4d). Seine catch species richness was highest in the summer, but was not significantly different from spring.

Discussion

In many coastal bays and estuaries residential properties comprise a large and increasing proportion of the shoreline, demonstrating the critical importance of coastal development strategies that balance stakeholder desires and environmental outcomes (NRC 2007; Scyphers and Lerman 2014). Here we experimentally assessed the ecological value of submerged breakwaters at scales relevant for typical waterfront property owners. We found that relatively small-scale breakwaters can provide habitat for filter-feeding bivalves, mobile invertebrates, and young fishes, but their value for protecting upland properties was unresolved and requires a longer time frame than the 2 years of our study. Our study also demonstrated that investment costs could be minimized by using a community-based or grassroots approach to restoration (see below), which may in-turn promote greater community involvement, ecological awareness, and sustainability (Leigh 2005).

Although oyster recruitment was largely absent during the 2 years of monitoring, both breakwater configurations provided substrate for mussel recruitment, which may also enhance the ecosystem functions and services provided. For instance, the filter-feeding of mussels may reduce seston in the water column and promote benthic–pelagic coupling (Galimany et al. 2013). Mussels are also important prey sources for decapod crustacean and fish predators, including blue crabs and Sheepshead (Hsueh et al. 1992). Dense aggregations of mussels have also been shown to benefit or facilitate the presence of other structured habitats, such as seagrasses and saltmarsh (Bertness 1984; Peterson and Heck 2001). The bagged oyster breakwaters supported the greatest abundance of mussels, but quantitatively measuring potential differences between treatments was difficult because of differences in structural complexity, substrate type, and surface available for recruitment. Regarding oyster recruitment, our study site was located at the northern limit and lower extreme for larval supply of oysters in Mobile Bay (Kim et al. 2010), but the presence of sparsely distributed oysters on adjacent structures suggests that episodic recruitment events do occur. Further experimentation is needed to determine the relative effects of oyster recruitment and post-settlement mortality under various contexts of substrate type and complexity.

Nekton sampling revealed that more than 35 species of fishes, shrimp, and crabs inhabited or utilized the complex structure provided by the breakwaters, and many of these species are considered recreationally or commercially important. For instance, catches of juvenile Sciaenids (Family: Sciaenidae), which includes Atlantic Croaker (*Micropogonias undulatus*), Red Drum (*Sciaenops ocellatus*), Sand Seatrout (*Cynoscion arenarius*), and Spotted Seatrout (*Cynoscion nebulosus*), were fivefold or greater near breakwaters than mudflats. The species richness of smaller fishes and crustaceans captured by seines was also higher near breakwaters than mudflat controls, but the abundance and richness of larger fishes were similar between treatments. Although it is widely recognized that complex habitats typically harbor greater densities of fishes and mobile fauna than less structurally complex areas (Rozas and Odum 1987), the relative influences of food availability versus refuge in promoting this pattern is less clear (Bostrom and Mattila 1999; Scheinin et al. 2012). Our results could indicate that enhanced densities of common prey species may not necessarily promote higher abundances of larger mobile consumers. However, the lack of treatment differences could also represent an artifact of our study design, considering that many species of larger fishes travel and forage over much greater distances than those separating the treatments (50 m). While larger scale oyster breakwaters have been shown to broadly enhance demersal fishes and crustaceans (Scyphers et al. 2011), decades of empirical studies have shown that species densities and interactions in oyster reefs are often highly context-dependent (e.g., Coen et al. 1999; Geraldi et al. 2009; Grabowski et al. 2005; La Peyre et al. 2014; Scyphers and Powers 2013). Disentangling the impacts of highly fragmented and heterogeneous habitats on ecological communities may be challenging, but it is increasingly relevant and necessary for understanding and managing urbanized coastal ecosystems.

An important area of inquiry left unresolved by our study was the efficacy of the submerged breakwaters for reducing erosive wave energies and protecting upland properties. The historically vegetated shoreline at our study site has been retreating for at least the past half-century, and the wave climate is directly affected by adjacent armored shores and boat-driven wakes from a nearby shipping channel, multiple marinas, and yacht clubs (Jones et al. 2009). Quantitatively evaluating how the breakwaters affected wave energy and marsh retreat was an initial objective of our study, but several unexpected events prevented us from effectively doing so. These include prescribed burns and sediment excavation targeting eradication of the common reed (*Phragmites australis*) that were conducted on adjacent uplands during the course of our study. These activities prevented us from being able to effectively and confidently measure changes in shoreline location or bathymetric profile. However, the two breakwater designs exhibited essential durability during storm-driven waves and storm surges following tropical cyclones Gustav and Ike. From pre- and post-event surveys, the only visible impact to our study site was a loss of PVC markers outlining the extent of each breakwater. Previous field studies have demonstrated the potential value of oyster reefs for mitigating vegetation retreat in some scenarios (La Peyre et al. 2014; Meyer et al. 1997; Piazza et al. 2005; Scyphers et al. 2011), but longer and more comprehensive empirical studies are needed to better define the biological and geophysical contexts when oyster breakwaters may be most effective.

In Mobile Bay, the average length of private, residential shorelines is approximately 30 m (Scyphers et al. 2014), and to install breakwaters at this scale, our study suggests that property owners could be forced to choose between a higher financial investment and reduced labor requirements for the Reef Ball configuration or face the opposite scenario for bagged oyster shell. For example, the staging and deployment of all four breakwaters required approximately 600 person hours, of which shell bagging and deployment demanded five times more labor than the Reef Ball breakwaters. However, the cost of Reef Balls and freight was three times more expensive than oyster shells and bagging materials. The community-based approach for implementing both configurations was successful and did not rely upon any specialized skills or equipment other than small aluminum or pontoon boats (<5 m). For homeowner efforts, the recruitment of volunteers could minimize or eliminate labor costs, as well as provide opportunity for "citizen science" monitoring of outcomes.

The two breakwater configurations tested in this study proved feasible and provided habitat for filter-feeding mussels and diverse assemblages of fishes. However, larger fishes did not appear to benefit from breakwaters as much as smaller and young fishes, and large scale land transformations and altered hydrology prevented an accurate assessment of how the breakwaters may have affected erosion rates and other geophysical processes. As noted above, several previous studies have demonstrated that oyster reefs constructed along impacted shorelines can promote fisheries enhancement, and to some extent mitigate erosion (La Peyre et al. 2014; Piazza et al. 2005; Scyphers et al. 2011), but the presence and magnitude of these benefits will undoubtedly exhibit spatial and temporal variability (Koch et al. 2009). To further advance our knowledge of the ecological and biophysical effects of breakwaters, and ultimately optimize their design and deployment, research efforts should involve collaboration between coastal ecologists, geologists, engineers and social scientists. Most importantly these efforts should evaluate structures and ecosystem functions at spatial scales relevant to waterfront property owners and other key decision makers.

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