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The effects of large-scale breakwaters on shoreline vegetation

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ABSTRACT

In response to shoreline erosion and coastal wetland loss, living shorelines have been implemented as a natural alternative to shoreline hardening. In high wave energy systems, "hybrid" living shoreline designs incorporating large-scale breakwaters have been increasingly chosen to restore and conserve wetlands. However, evaluations of the effectiveness of breakwaters at preserving natural shorelines and promote the growth of shoreline plantings are limited. To evaluate the effectiveness of large-scale breakwaters at protecting or restoring marshes in high wave energy environments, we conducted an experimental planting and shoreline monitoring program landward of eight-year-old breakwaters and adjacent no breakwater sites along Bon Secour Bay, Alabama. Results showed that breakwaters help natural marsh to maintain its cover at a high level (70%), but have little impact on shoreline planting. Furthermore, breakwater presence reduced the pressure for upland migration, allowing natural marsh patches to expand seaward. Without breakwater protection, fringing marsh retreated upland significantly. Cumulatively, this study suggests that large-scale breakwaters could have an impact on preserving fringing marsh vegetation in high wave energy environments though their effectiveness into the future will require adaptive management in response to local sea-level rise.

1. Introduction

Though coastal wetlands are one of the most ecologically and economically valuable ecosystems on the planet (Costanza et al., 2014), natural and anthropogenic pressures have reduced their extent and increasingly threaten their persistence into the future. Currents and wind-driven waves, during storms in particular, can have a staggering erosive effects on shorelines (Morton and Barras, 2011; Hauser et al., 2015). While these are dynamic processes that can also deposit sediments (McKee and Cherry, 2009; Tweel and Turner, 2012; McKee et al., 2020), their erosive effects are worsening as sea-levels rise and the frequency and intensity of storms increase (Fitzgerald et al., 2008; Wallace and Anderson, 2013). The natural loss of wetlands is worsened by the urbanization of coastal regions. The dredging of wetlands for transportation has led to an alarming loss of wetlands via altered hydrology and edge erosion (Johnson and Gosselink, 1982; Browne, 2017) and wakes from commercial and recreational watercraft have significantly eroded shorelines (Houser, 2010; Bilkovic et al., 2019; Safty and Marsooli, 2020). Furthermore, the construction of urban infrastructure along the coast has limited the ability of wetlands to migrate upland and persist into the future (Torio and Chmura, 2013; Borchert et al., 2018). In fact, due to these interacting natural and anthropogenic effects, about 65% of wetlands world-wide have already been lost (Davidson, 2014) including over 257,000 acres of wetlands along the northern Gulf of Mexico in the five years between 2004 and 2009 (Dahl, 2011).

Shoreline hardening, while preventing land loss for a time, directly and indirectly leads to the destruction of both intertidal habitat and its associated ecosystem services (Lee et al., 2006; McCauley et al., 2013; Aguilera et al., 2020). For instance, shoreline hardening has been shown to reduce the nursery habitat for a variety of economically important species (Long et al., 2011; Dugan et al., 2018) and decrease the resilience of both the ecosystem and coastal communities to storms (Smith et al., 2017; Tomiczek et al., 2020). Additionally, hardened structures often lead to increased erosion at the installation site and in adjacent areas leading to the further losses of these services (Roberts, 2007; Nordstrom et al., 2009). While the loss of ecosystem services is certainly expensive, the installation and maintenance of seawalls, bulkheads, and other shoreline hardening is also costly. The Center for Climate Integrity

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estimated that by 2100, the United States will spend $518 billion to build almost 97,000 km of seawalls to minimize shoreline erosion and flooding (Wiles and LeRoy, 2019). Even after this initial financial investment, these structures must be maintained throughout their life and eventually replaced every 15–50 years (North Carolina Division of Coastal Management, 2011; Herder, 2014).

Because of the detrimental environmental effects of shoreline hardening and their high costs, many areas have turned to the implementation of natural or nature-based features along shorelines, termed "living shorelines" due to the use of living elements such as marsh vegetation. These projects have been shown to not only stabilize shorelines (Herbert et al., 2018; Polk and Eulie, 2018; Smee, 2019), but also to re-introduce crucial ecosystem services. Gittman et al. (2016) showed that living shorelines were able to provide nursery habitat that had not been there prior to project implementation. Similarly, Davis et al. (2006) found that this effect could be immediate for some species. Furthermore, living shorelines provide protection from storms (Gittman et al., 2014), sequester significant amounts of carbon (Davis et al., 2015), and reduce the amount of nutrient pollution entering adjacent waterways (Sparks et al., 2015; Onorevole et al., 2018). While this restoration of services can often be achieved with only living elements like plants and oyster shell, a "hybrid" design using rock or concrete structures to break waves is usually implemented in areas with high wave energy.

In recent years, numerous large-scale living shoreline projects that incorporate near-shore breakwaters have been implemented to both protect shorelines and enhance ecosystem services. This has been especially true in the northern Gulf of Mexico, where projects like these have been installed in response to the Deepwater Horizon oil spill in 2010. In Hancock County, Mississippi, USA, almost six miles (i.e., approximately 9.66 km) of breakwaters have been installed in combination with marsh restoration to reduce shoreline erosion. Likewise, at the Biloxi Marsh in southeast Louisiana, USA, between nine and eleven miles of breakwaters will be constructed to restore the adjacent marsh. These two projects alone cost over $110,000,000 USD to fully implement. Furthermore, there are many more planned living shoreline projects across the northern Gulf of Mexico that incorporate breakwaters (gulfspillrestoration.noaa.gov). Despite the enormous investment in both cost and labor, the effects of large-scale breakwater projects like these on shoreline vegetation and on shoreline plantings that often accompany large-scale living shorelines projects have not been well studied. Breakwaters have many design elements (e.g., crest height, width, construction materials, etc.) that have been studied for their wave dampening abilities ((Chasten et al., 1993); (U.S. Army Corps of Engineers, 2002); Morris et al., 2019). For example, it is widely accepted that the crest height of breakwaters should be at or above mean water level to best reduce the wave energy reaching the shoreline (Allen and Webb, 2011; Webb and Allen, 2017). However, variations in those elements as well as installation decisions such as the distance from the shoreline, the width of gaps between breakwaters, whether to plant along the shoreline, and the density of that planting could have confounding effects on the shoreline (Birben et al., 2007). For instance, large-scale breakwaters made of geotextile materials, rather than rock, were shown to increase sedimentation along multiple beaches in Indonesia, though they were not wholly successful across all test sites due to factors including the placement of the breakwaters from the shore, the materials used to fill the geotextile, and the failure of the geotextile itself in some cases (Simanjuntak, 2018). While many studies show that breakwaters increase sedimentation (Garcia, 2019; Oakley et al., 2020), it is still unclear if they stabilize the shoreline enough to enhance shoreline vegetation. In Southeast Asia, low-crested, rubble breakwaters have been shown to promote the colonization and establishment of mangroves (Kamali and Hashim, 2011; Tamin et al., 2011; Akbar et al., 2017). However, studies showing the effectiveness of breakwaters to enhance shoreline vegetation and planting are limited.

To address this gap, we performed a paired shoreline planting and monitoring project along a breakwater protected and adjacent no breakwater shoreline in Bon Secour Bay, Alabama. We hypothesized that breakwater presence would have the following effects on the shoreline when compared to the non-protected shoreline: 1) Increased percent cover of *Spartina alterniflora*, 2) increased average patch size and area of *S. alterniflora* per unit shoreline length, 3) lower number of *S. alterniflora* patches per unit shoreline length, 4) less upland migration of *S. alterniflora*, and 5) decreased wave heights.

2. Methods

2.1. Management application team

The design and scope of work for this project was developed in collaboration with a management application team (MAT) comprised of individuals from academia, environmental consulting firms, non-profits, and local, state, and federal agencies. The purpose of the interaction with the MAT was to directly address research needs for natural resource managers, which this study was derived from. The project team met with the MAT semiannually throughout the study period to discuss results to date, new potential research questions, and plans to integrate research and outreach products into the applied management of shoreline projects. The MAT was instrumental in the development of the research questions and the experimental design.

2.2. Study site

The Swift Tract (centered at 30.319675, −87.787743) is an eroding shoreline along Bon Secour Bay, which is located in the SE corner of Mobile Bay, AL (Fig. 1). This stretch of shoreline is part of the Weeks Bay National Estuarine Research Reserve and has fringing marsh consisting primarily of *Spartina alterniflora* and *Phragmites australis*. This area is a microtidal system that experiences diurnal tides with an average peak range of 0.41 m. Mean low and high water at this location is −0.098 m and 0.323 m (NAVD88), respectively.

Because erosion at the site is driven by high-wave energy, The Nature Conservancy and Weeks Bay National Estuarine Research Reserve

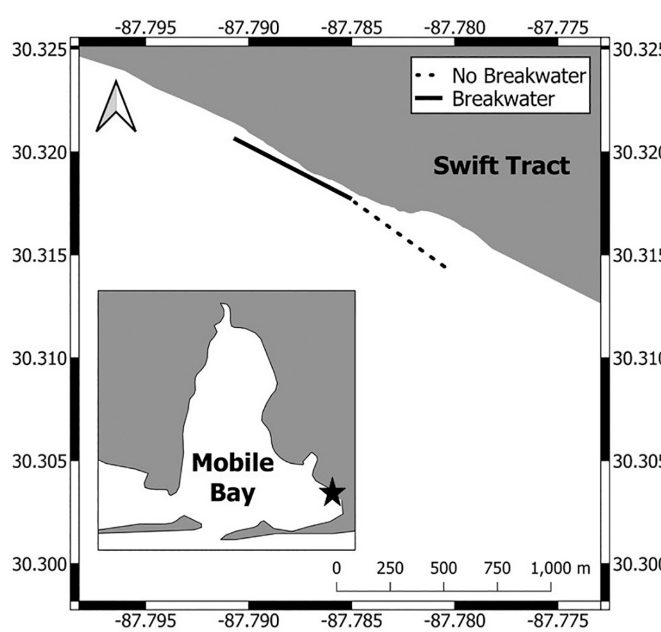

Fig. 1. Map of the swift tract study site. The location of the study site is demarcated in the inset map by a black star. The position of the breakwater and adjacent no breakwater shorelines are indicated by the solid black line and the dotted line, respectively. Latitude and longitude for the site are noted on the perimeter of the map.

installed five breakwaters in spring 2012 to mitigate wave-energy and prevent further erosion. The northernmost breakwater is 80 m long x 3.5 m wide, while the 4 others are 120 m long x 3.5 m wide. Breakwaters were spaced 15 m apart. The breakwaters are constructed of individual wire cages (1 m long x 1 m wide x 0.76 m tall) fastened together to reach the target length. The cages were then filled with rock ranging from about 7.6 to 24 cm and weighing 22.7 kgs or less. Following a settling period, the breakwaters had an average crest height of 0.25 m (NAVD88) which is approximately the same height above the seabed elevation of −0.05 m (NAVD88). Because the shoreline is varying, the breakwaters sit between 30 and 40 m offshore, depending on the given location (Fig. 2).

2.3. Experimental design

In summer 2016, 24 experimental fixed-plots (4 m^2) were established along both the breakwater protected shoreline (0.6 km) and the adjacent reference shoreline (1.2 km) resulting in 48 plots in total and two breakwater treatments, Breakwater (BW) and No Breakwater (NBW). Across these 48 plots, there were also three vegetation treatments (i.e., natural, planted, and open; Fig. 2). To ensure these plots experienced similar tidal influence and were at the same elevation of 0.25 m (NAVD88), elevation markers were placed along the shoreline as a reference for selecting plot areas. Because plant cover was patchy and irregular along the study shoreline, plots were established and vegetation treatments were assigned according to what was achievable at the site.

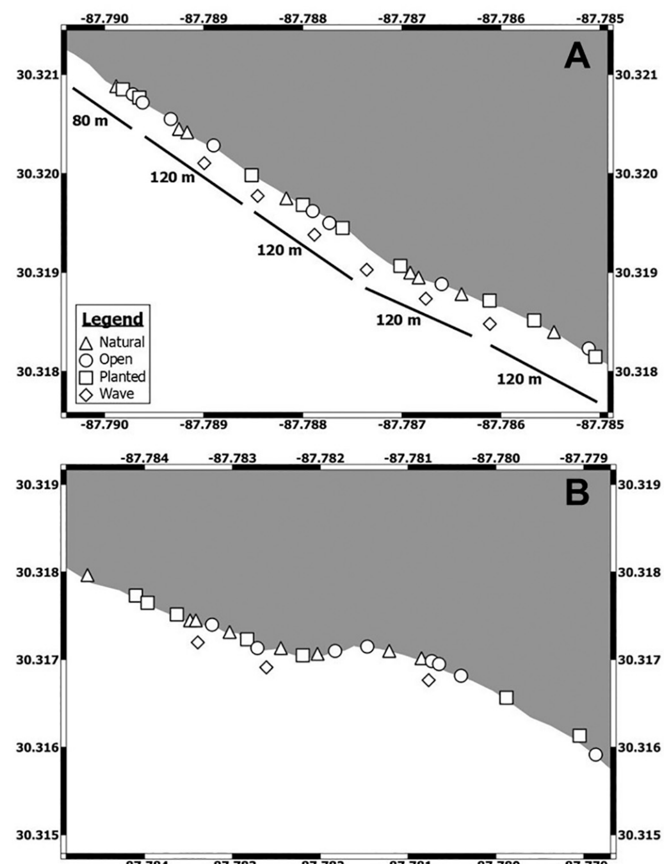

Fig. 2. Breakwater position and experimental design. Breakwater dimensions and their position relative to the shoreline are shown. Text indicates the length of each breakwater segment in meters. The location of the vegetation plots are marked by shapes along the A) breakwater shoreline and B) adjacent no breakwater shoreline. Triangles, circles, and squares represent natural, open, and planted plots, respectively. Latitude and longitude are noted along the perimeters of each map.

While plots with the natural treatment were established in dense, pre-existing stands of *S. alterniflora,* both the planted and open treatments were established at locations along the shoreline that had no pre-existing vegetation. To achieve the planted treatment, nursery-grown *S. alterniflora* was planted in a 4 m^2 plot at 50% cover in a checkerboard pattern. Nursery plants were grown via vegetative growth from native cuttings in 2.74 L containers. All plantings occurred at low tide on June 20th, 2016. Open treatments were areas left without vegetation of any kind.

To investigate hypothesis 1 (i.e., increased *S. alterniflora* percent cover behind breakwaters), percent cover of *S. alterniflora* at each of the 48 plots was visually estimated by the same person (S. Martin) each season between September 2016 and July 2020, resulting in eleven sampling periods. The visual estimation of percent cover followed the procedure established in Brower et al., 1990 and procedures used in a nearby shoreline monitoring project (Martin et al., 2021).

Additionally, beginning in the summer of 2017, the outside perimeter of natural *S. alterniflora* patches along the BW and NBW shorelines were recorded using Real Time Kinematic Global Positioning Systems (RTK GPS) to address hypotheses 2, 3, and 4. The shoreline was divided into 140 m segments to create replicates for analysis and to account for differences in the length of the breakwater protected shoreline and the adjacent no breakwater shoreline. This distance was chosen to encompass one breakwater and half of the 15 m gap between each breakwater. This data was then exported for analyses as GIS polygons to measure how much the patches migrated upland and to determine if this was affected by breakwater presence. The GIS polygons were also used to measure an array of polygon dynamics including the number of polygons, the average area per polygon, and the total area of polygons. This process was repeated again in the summer of 2018 and 2020, resulting in three sampling periods.

Lastly, to determine the breakwater complex's ability to attenuate waves (hypothesis 5), nine DIY wave gauges (Temple et al., 2020) were deployed in February 2019 during a high water period. Three gauges were deployed behind breakwaters, three gauges were deployed in the breakwater gaps (BWG), and three gauges were deployed along the reference no breakwater shoreline (Fig. 2). At each location, the gauges were deployed to a water depth of one meter at approximately the same distance from the shore. Depths were determined by lowering a meter stick until solid substrate was encountered as the wave gauges were attached to anchors that would sink into unconsolidated sediment. They were all deployed within a 15 min timeframe, ensuring a similar water depth across the site. The wave gauges recorded continuous pressure data at a frequency of 10 Hz (10 times per second) for a 3 day period.

2.4. Statistical analyses

2.4.1. Fixed-plots

Because the percent cover data did not meet the assumption of normality for a *T*-Test, the effects of breakwater presence on *S. alterniflora* percent cover were analyzed using the non-parametric, two-sample Wilcoxon test for each of the eleven sample periods in R version 3.5.1. Time was not considered a factor in these analyses because when breakwaters had a significant effect on plant percent cover was not a central question to this study.

2.4.2. Polygon migration

QGIS version 3.10 was used to measure the distance that polygons changed from summer 2017 to summer 2020 on both the landward and seaward edges of the polygons. Summer 2018 was left out of this analysis to capture the total change over the study period rather than the fluctuations across years. To determine the effect of breakwater presence on upland migration of *S. alterniflora,* these distances were analyzed using a two-sample *t*-test, or a two-sample Wilcoxon test when the data did not meet the assumptions of a t-test. These analyses were done R (R Core Team 2020).

2.4.3. Polygon dynamics

QGIS version 3.10 (QGIS Development Team 2020) was also used to measure three types of polygon dynamics across breakwater treatments: 1) the number of *S. alterniflora* polygons, 2) the average area of a *S. alterniflora* polygon, and 3) the total area of *S. alterniflora* polygons. Once these measurements were determined in QGIS, they were entered into a .csv file and analyzed in R to determine the effects of breakwater presence using a two-sample t-test or a two-sample Wilcoxon test when the data at a sampling period did not meet the assumptions of a t-test. These analyses were done by breaking the respective shorelines into 140 m segments to capture one breakwater and half of the gap between breakwaters and to standardize the units for analysis.

2.4.4. Wave attenuation

After retrieval, the raw pressure data was processed using Matlab software to determine the significant wave height at the site. Significant wave height is the average of the top third of wave heights in the entire record.

3. Results

3.1. Fixed plot experiment

4 m^2 plots of standing *S. alterniflora* (i.e., natural treatment) began with 77.63 ± 11.22 (mean ± SE) percent cover when behind breakwaters and 65.13 ± 6.48% cover without breakwaters. A significant effect of breakwater presence on natural percent cover was seen starting with the seventh sampling period (i.e., six years post breakwater construction) and was maintained for the duration of the study (Fig. 3a). Furthermore, the difference in average cover between the breakwater and no breakwater treatments continued to increase at each sampling with percent cover in breakwater protected plots falling slightly to 65 ± 17.37% by the end of the study while those not protected by breakwaters fell sharply to 13.38 ± 12.51% cover in the same period (Table 1). Additionally, the slight decrease in percent cover from the first to last sampling in breakwater protected plots was not significant ($z = 0.5789$, $p = 0.563$).

Plots planted with nursery-grown *S. alterniflora* (i.e., planted treatment) had fallen to 31.63 ± 3.03% cover in those plots behind breakwaters and 4.5 ± 2.17% cover without breakwaters by the time of the first sampling (i.e., 0.25 years after planting; Fig. 3b; Table 2). This was the only sampling in which breakwaters had a statistically significant effect ($z = 3.391$, $p = 0.001$) on percent cover in planted plots until the final sampling which was also significant ($z = 2.304$, $p = 0.021$). While the percent cover in breakwater plots fluctuated through time, the final cover did not significantly differ from that of the first sampling ($z = 0$, $p = 1.0$) as was also the case with the natural plots.

Open plots experienced little to no change in percent cover where the highest cover experienced in those plots with breakwaters and without breakwater protection reached 3.2 ± 3.02 and 3.75 ± 3.83%, respectively (Fig. 3c; Table 3). There was no significant effect of breakwaters at any sampling period across the open plots.

3.2. Spartina alterniflora patch migration

Over three years, the upland migration of the landward edge of *S. alterniflora* patches was significantly affected by the presence of breakwaters ($z = -2.985$, $p = 0.003$). Without breakwaters, patches saw their landward edge migrate upland 3.322 ± 0.871 m while those behind breakwaters migrated upland only 1.685 ± 0.348 m (Fig. 4a; Table 4), resulting in an annual upland migration rate of 1.107 ± 0.290 m and 0.561 ± 0.117 m, respectively.

Likewise, the seaward edges of *S. alterniflora* patches were also significantly affected by breakwater presence over the three years ($t = 6.607$, $p \leq 0.001$). The seaward edge of patches in areas without breakwaters again migrated upland 2.091 ± 0.429 m in total over the study period at a rate of 0.697 ± 0.143 m annually. Interestingly, the seaward edge of patches with breakwater protection did not migrate upland at all. Instead, patches expanded seaward by 0.447 ± 0.325 m (Fig. 4b; Table 4) at a rate of 0.149 ± 0.108 m annually.

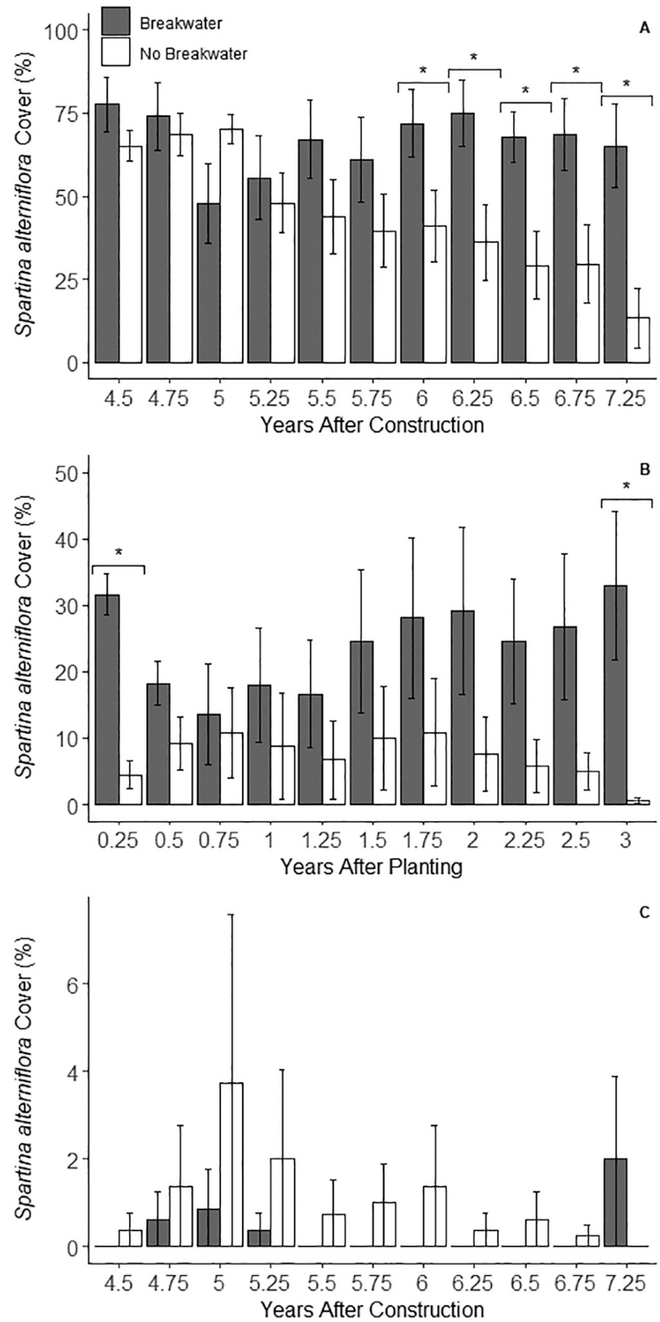

Fig. 3. The effect of breakwater presence on *S. alterniflora* cover. The percent cover of plant *S. alterniflora* over 11 sample dates spanning across (A) natural, (B) planted, and (C) open plots. Sample dates range from 4.5 years to 7.25 years after breakwater construction for natural and open plots and 0.25 years to 3 years after planting for the planted plots. Grey bars represent plots with breakwater protection and white bars represent those plots without. Asterisks denote a significant effect of breakwater presence on *S. alterniflora* percent cover.

Table 1
Outputs of statistical tests for significant effects of breakwater presence on natural *S. alterniflora* percent cover. The N, means, standard errors, and outputs from statistical tests are displayed. The *p*-value is reported along with either a t-score or z-score depending on which test, either the Welch Two Sample *t*-test or the Asymptomatic Wilcoxon-Mann-Whitney Test, was most appropriate for the data collected at each sample period. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwater presence on *S. alterniflora* percent cover.

Years Since Construction	Breakwater Presence	N	Mean Cover	± SE	z-score	t-score	p-value
4.5	Breakwater	8	77.625	8.117		1.365	0.199
	No Breakwater	8	65.125	4.688			
4.75	Breakwater	8	74	29.146		0.466	0.645
	No Breakwater	8	68.5	6.312			
5	Breakwater	8	47.75	12.076		−1.787	0.108
	No Breakwater	8	70.25	4.494			
5.25	Breakwater	8	55.5	12.591		0.497	0.624
	No Breakwater	8	48	8.923			
5.5	Breakwater	8	67.125	11.858		1.466	0.165
	No Breakwater	8	43.875	11.087			
5.75	Breakwater	8	61	12.663		1.2962	0.216
	No Breakwater	8	39.625	11.159			
6	Breakwater	8	71.875	12.157		**2.135**	**0.050**
	No Breakwater	8	41.125	10.687			
6.25	Breakwater	8	75	9.832	**2.261**		**0.024**
	No Breakwater	8	36.125	11.314			
6.5	Breakwater	8	67.875	7.52		**3.128**	**0.008**
	No Breakwater	8	29.125	10.209			
6.75	Breakwater	8	68.5	10.7	**2.052**		**0.040**
	No Breakwater	8	29.625	11.881			
7.25	Breakwater	8	65	12.573	**2.342**		**0.020**
	No Breakwater	8	13.375	9.052			

Table 2
Outputs of statistical tests for significant effects of breakwater presence on planted *S. alterniflora* percent cover. The N, means, standard errors, and outputs from statistical tests are displayed. The p-value is reported along with either a t-score or z-score depending on which test, either the Welch Two Sample t-test or the Asymptomatic Wilcoxon-Mann-Whitney Test, was most appropriate for the data collected at each sample period. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwater presence on *S. alterniflora* percent cover.

Years after planting	Breakwater Presence	N	Mean Cover	± SE	z-score	t-score	p-value
0.25	Breakwater	8	31.625	3.033		**7.442**	**<0.001**
	No Breakwater	8	4.500	2.171			
0.50	Breakwater	8	18.250	3.349		1.764	0.100
	No Breakwater	8	9.250	4.005			
0.75	Breakwater	8	13.625	7.586	0.913		0.361
	No Breakwater	8	10.875	6.802			
1.00	Breakwater	8	18.000	8.548	1.949		0.051
	No Breakwater	8	8.750	7.958			
1.25	Breakwater	8	16.625	8.050	1.591		0.112
	No Breakwater	8	6.750	5.920			
1.50	Breakwater	8	24.500	10.799	1.294		0.196
	No Breakwater	8	10.000	7.845			
1.75	Breakwater	8	28.125	12.097	1.066		0.287
	No Breakwater	8	1.875	8.153			
2.00	Breakwater	8	29.13	12.545	1.153		0.249
	No Breakwater	8	7.625	5.569			
2.25	Breakwater	8	24.625	9.421	1.756		0.079
	No Breakwater	8	5.750	4.024			
2.50	Breakwater	8	26.750	11.018	1.514		0.130
	No Breakwater	8	5.000	2.729			
3.00	Breakwater	8	33.000	11.241	**2.304**		**0.021**
	No Breakwater	8	0.625	0.332			

3.3. *Spartina alterniflora* patch dynamics

The number of *S. alterniflora* patches was significantly higher in the summer 2017 and 2018 surveys along the unprotected shorelines ($p = 0.046$ and $p = 0.037$, respectively), but by summer 2020 no significant difference was observed between protected and unprotected shorelines (Fig. 5a; Table 5).

The average area of *S. alterniflora* patches increased both with and without breakwater protection (Table 5). However, average patch area only significantly differed in the summer of 2017 where unprotected patches had higher average area than those behind breakwaters ($t = -3.904$, $p = 0.011$). However, no significant differences were recorded in the subsequent summers. Still, the patches behind breakwaters did tend to have higher average area in the summers of 2018 and 2020 (Fig. 5b; Table 6).

Like the average area of the *S. alterniflora* patches, the total area of patches (i.e., the area of all patches in each breakwater treatment) also increased from one summer sampling to the next (Fig. 5c). However, total area of patches again only differed significantly in the first summer when the unprotected patches had higher total cover than those with breakwater protection ($t = -6.085$, $p = 0.002$; Table 7).

3.4. Wave attenuation

Over the three days that the gauges were deployed at the experimental site, breakwater presence was shown to have a significant effect

Table 3
Mean percent cover of open plots. The Ns, means, and standard errors are displayed.

Years Since Construction	Breakwater Presence	N	Mean Cover	± SE
4.5	Breakwater	8	0.000	0.000
	No Breakwater	8	0.375	0.384
4.75	Breakwater	8	0.625	0.640
	No Breakwater	8	1.375	1.407
5	Breakwater	8	0.875	0.896
	No Breakwater	8	3.750	3.838
5.25	Breakwater	8	0.375	0.384
	No Breakwater	8	2.000	2.047
5.5	Breakwater	8	0.000	0.000
	No Breakwater	8	0.750	0.768
5.75	Breakwater	8	0.000	0.000
	No Breakwater	8	1.000	0.886
6	Breakwater	8	0.000	0.000
	No Breakwater	8	1.375	1.407
6.25	Breakwater	8	0.000	0.000
	No Breakwater	8	0.375	0.384
6.5	Breakwater	8	0.000	0.000
	No Breakwater	8	0.625	0.640
6.75	Breakwater	8	0.000	0.000
	No Breakwater	8	0.250	0.256
7.25	Breakwater	8	2.000	1.905
	No Breakwater	8	0.000	0.000

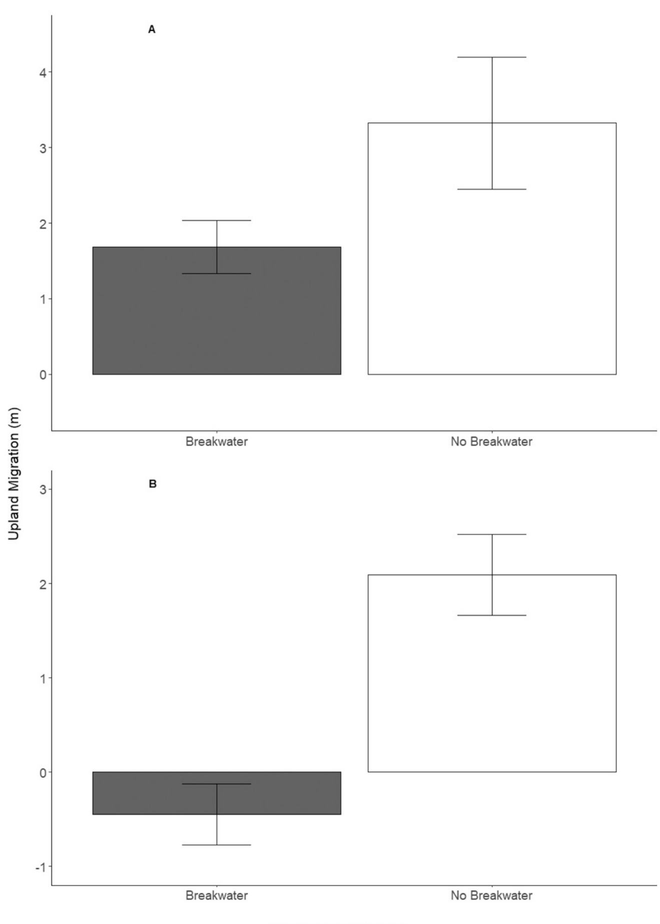

Fig. 4. The Effect of Breakwater Presence on Upland Migration of the (A) Landward and (B) Seaward Edges of *S. alterniflora* Patches.

on the average significant wave height ($p < 0.001$). The recorded wave height was lower in the areas protected by breakwaters. When considering the average significant wave height, breakwater presence lowered the wave height from 8.49 cm to 4.73 cm. The average significant wave height was also lower behind breakwaters than it was in the breakwater gap areas (Fig. 6).

4. Discussion

The results of this study suggest that near-shore breakwaters of this design do, in fact, stabilize natural shoreline vegetation as supported by the fixed plot, patch migration, and patch dynamics data. Furthermore, hypotheses 1, 4, and 5 were supported by this study.

The first hypothesis, that *S. alterniflora* cover would be higher along the breakwater protected shoreline, was confirmed by the percent cover monitoring. Percent cover in plots with natural vegetation was not only higher in those plots protected by breakwaters, but cover was also maintained at a high level (i.e., 65% on average) throughout the course of the study. Additionally, cover was maintained in planted plots from the first sampling to the last when protected by breakwaters, though they did not expand their area of coverage. Those plots without breakwater protection, on the other hand, saw cover levels fall sharply over the course of the four-year study, regardless of planting treatment. As with all field data, visual cover estimates are subject to human error. The strong trends observed between sites with and without breakwater protection suggest that this error did not affect the results. Furthermore, several measures were taken to account for and minimize this error following widely accepted methodology (Brower et al., 1990; Martin et al., 2021; Temple et al., 2021). Still, cover estimates could be improved in future studies by using technology.

In addition, the results from this study suggest that *S. alterniflora* patches are not actually shrinking. Rather, the migration and patch dynamics data reveals that the difference in percent cover could be attributed to upland migration. Plots without breakwater protection migrated upland more than one meter on both the landward and seaward edges, while those behind breakwaters maintained their platform position and even expanded seaward towards the breakwaters over the three-year duration of these measurements. This confirmed the hypothesis that there would be less upland migration behind breakwaters. Similar results have also been shown in high energy shorelines along the northern Gulf of Mexico (Safak et al., 2020) and in South Asia (Chowdhury et al., 2019). From this finding, it can be inferred that natural plots not behind breakwaters were not losing cover because there was less vegetation along the reference shoreline, but instead the vegetation had moved upland and therefore outside of the fixed plot boundaries. This inference is further supported by the patch dynamics data.

Had the difference in percent cover between breakwater and no breakwater plots been due to a loss of vegetation, a significant difference between patch areas would have been expected. However, there was no difference between the number of patches, average patch area, or total patch in breakwater and no breakwater plots by the end of the study. This result does not support the hypotheses that total patch area and average patch size would be increased by breakwater presence or that there would be less patches behind breakwaters. While this result did not meet expectations, the lack of a difference between patches with and without breakwater protection is meaningful. When combined with the migration data, it shows that the patches of *S. alterniflora* are not shrinking, but instead are moving upland, in part as a response to the pressures associated with increased wave energy.

Throughout the study, breakwater protection appeared to have a positive effect on the maintenance of shoreline plantings (Fig. 3b; Table 2). Results from patch migration and dynamics monitoring combined with the percent cover results for planted plots show that the presence of breakwaters may allow shoreline plantings to persist in a high wave energy environment. Though the cover behind breakwaters was reduced from 50% to 32% in planted plots by the time of the first sampling, percent cover of these plots remained unchanged throughout the rest of the study. This constant cover in planted plots, combined with the high cover maintained in the natural plots, suggests that shoreline

Table 4
Significant effects of breakwater presence on upland migration of *S. alterniflora* patches. The N, means, standard errors, and outputs from statistical tests are displayed. The p-value is reported along with either a t-score or z-score depending on which test, either the Welch Two Sample t-test or the Asymptomatic Wilcoxon-Mann-Whitney Test, was most appropriate for the data collected. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwater presence on *S. alterniflora* patches.

Leading edge	Breakwater Presence	N	Mean Upland Migration	±SE	z-score	t-score	p-value
Landward	Breakwater	23	1.685	0.348	−2.985		0.003
	No Breakwater	26	3.332	0.871			
Seaward	Breakwater	23	−0.447	0.325		6.67	<0.001
	No Breakwater	26	2.091	0.429			

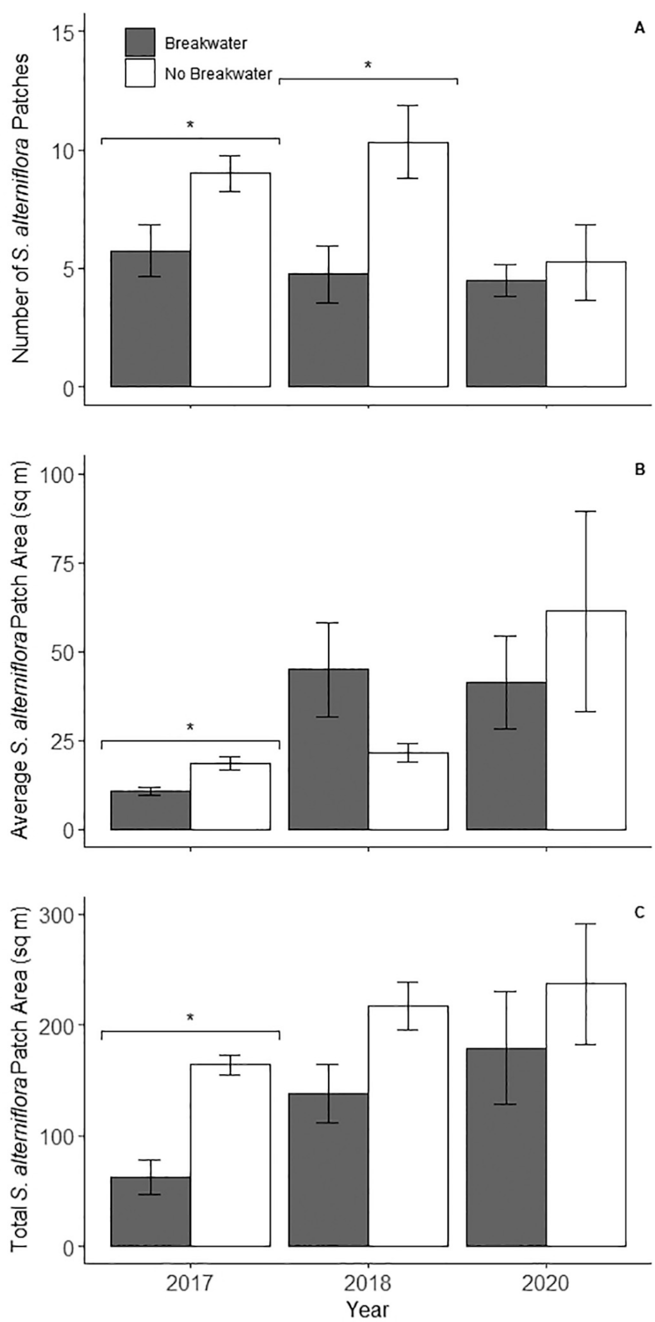

Fig. 5. The effect of breakwater presence on *S. alterniflora* (A) number of patches, (b) average patch area, and (c) total patch area. Dark bars represent patches not behind breakwaters and white bars represent those behind breakwaters. For statistical analysis, each shoreline was broken into standardized 140 m long units. Asterisks denote a significant effect of breakwaters (α = 0.05).

Table 5
The effect of breakwater presence on the average *S. alterniflora* patch area (sq m). The N, means, standard errors, and outputs from statistical tests are displayed. The p-value is reported along with either a t-score or z-score depending on which test, either the Welch Two Sample t-test or the Asymptomatic Wilcoxon-Mann-Whitney Test, was most appropriate for the data collected. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwater presence on the average *S. alterniflora* patch size.

Year	Breakwater Presence	N	Mean Area	± SE	z-score	t-score	p-value
2017	Breakwater	4	10.698	1.173		**−3.904**	**0.011**
	No Breakwater	4	18.55	1.802			
2018	Breakwater	4	44.967	13.249	1.768		0.077
	No Breakwater	3	21.504	2.436			
2020	Breakwater	4	41.188	13.04	0.289		0.773
	No Breakwater	4	61.243	28.098			

Table 6
The effect of breakwater presence on total *S. alterniflora* patch area (sq m). The N, means, standard errors, and outputs from statistical tests are displayed. The p-value is reported along with a t-score from the Welch Two Sample t-test. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwaters.

Year	Breakwater presence	N	Mean area	± SE	t-score	p-value
2017	Breakwater	4	62.935	15.497	**−6.085**	**0.002**
	No Breakwater	4	164.093	8.704		
2018	Breakwater	4	138.012	26.227	−2.478	0.056
	No Breakwater	3	217.303	21.956		
2020	Breakwater	4	179.417	51.066	−0.828	0.44
	No Breakwater	4	237.38	54.699		

Table 7
The effect of breakwater presence on the number of *S. alterniflora* patches. The N, means, standard errors, and outputs from statistical tests are displayed. The p-value is reported along with a t-score from the Welch Two Sample t-test. All values were rounded to the nearest third decimal place. Bold text denotes a significant effect of breakwaters.

Year	Breakwater Presence	N	Mean Number of Patches	± SE	t-score	p-value
2017	Breakwater	4	5.75	1.102	**−2.6**	**0.046**
	No Breakwater	4	9	0.756		
2018	Breakwater	4	4.75	1.185	**−3.055**	**0.037**
	No Breakwater	3	10.333	1.553		
2020	Breakwater	4	4.5	0.69	−0.461	0.668
	No Breakwater	4	5.25	1.596		

planting should be done in high densities (i.e., in clusters; O'Brien and Zedler, 2006; Silliman et al., 2015; Duggan-Edwards et al., 2020), if it is done at all. In sum, to protect shorelines in high wave energy climates, near-shore breakwaters are most effective at conserving existing marsh rather than creating new marsh.

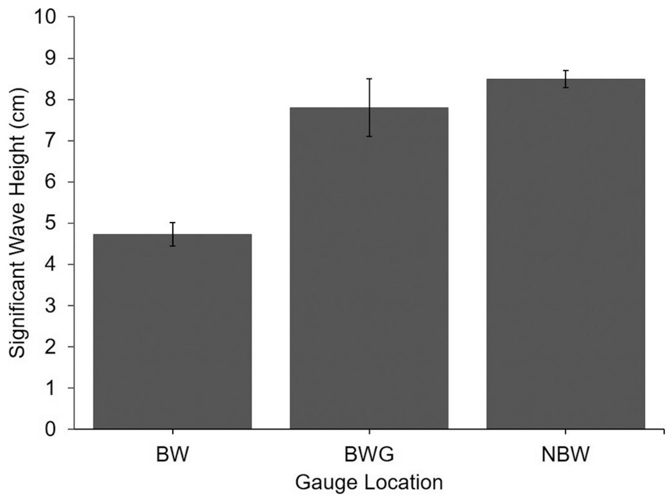

Fig. 6. The effect of breakwater presence on significant wave height. Grey bars represent the average wave height (cm) recorded by wave gauges behind breakwaters (BW), in the breakwater gaps (BWG), and in the adjacent no breakwater area (NBW).

The effects of the breakwaters on the shoreline plant communities could be partially explained by the reduction wave heights and associated energy impacting the shoreline observed in this study. The results from the wave attenuation portion of this study show that the breakwaters are effective at reducing the wave energy impacting the shoreline, thus supporting our fifth hypothesis. This one time measurement was done during a low water period in which the breakwaters were at their most effective. Therefore, observed differences in wave attenuation were likely higher than average at the site. However, the slight difference in wave energy impacting the shoreline could be driving some of the observed vegetation differences. A growing body of literature demonstrates that the increasing exposure to higher magnitude wave events limits the establishment and persistence of coastal plant communities (Roland and Douglass, 2005) and can have varying effects on plant morphological features (Balke et al., 2011; McLoughlin et al., 2015; Sharma et al., 2016; Silinski et al., 2018; Temple et al., 2021). In this study, breakwater presence undoubtedly served to mitigate wave energy impacts on this shoreline, allowing the vegetation to expand both landward and seaward. However, where plants cannot adapt in place, marsh migration is the primary avenue for marsh resilience to environmental change (Brinson et al., 1995; Enwright et al., 2016; Raabe and Stumpf, 2016), as was seen in the plots without breakwater protection. While upland marsh migration can help marshes to persist into the future, marshes are often limited by the presence of upland barriers (e.g., coastal squeeze; Kirwan and Megonigal, 2013; Gittman et al., 2015; Field et al., 2016; Constantin et al., 2019) emphasizing the need to reduce the pressure to migrate. The results of this study have shown that near-shore breakwaters can reduce this pressure on shoreline vegetation.

This study demonstrates the usefulness of near-shore breakwaters at protecting existing shoreline vegetation from wave impacts. However, recent studies have shown that local marsh shoreline erosion leads to sediment deposition elsewhere in the system and subsequently increases marsh platform elevation (Ganju, 2019; Vona et al., 2020). Therefore, the benefits of breakwaters on the shoreline plant community may be short term and ultimately lead to reductions in sediment deposition on the inland marsh platform. This loss of sediment inland could reduce the resilience of the entire marsh system, though future research is needed to validate this relationship.

Regardless of the effect of breakwaters on sediment transport, it is unclear if the effectiveness of near-shore breakwaters will persist as sea levels rise. As sea-level rise increases, breakwaters can become submerged and no longer perform as intended (Allen and Webb, 2011; Webb and Allen, 2017). As a result, shoreline vegetation would experience increased flooding and wave energy. The Weeks Bay NERR, where this study was performed, will see an increase in sea levels between 30.5 and 122 cm by 2060 (Collini et al., 2018). Under even the lowest scenario, 30.5 cm of additional sea level on top of the current MLW (i.e., 9.8 cm) would submerge the breakwaters in this experiment the vast majority of the time. At such a time, it is important that these breakwaters are adaptively managed to add to the crest height and thereby maintaining their effectiveness as a buffer against wave energy for the shoreline vegetation. Such adaptive management of projects has been widely implemented to ensure project success in the face of a changing environment (Wigand et al., 2017; Ellison et al., 2020; Perry et al., 2020).

Additionally, this study emphasizes the need for long-term monitoring of large-scale breakwater implementation projects. Had this project taken place in the initial two to three years following breakwater implementation at the Swift Tract, there would have been no observable effect of breakwater presence on shoreline vegetation as the significant effects on the percent cover in natural plots emerged six years after breakwater construction. Similarly, had this study been continued for an additional three to five years, it is possible that the significant effect of breakwater presence on the planted plots that emerged three years after planting could prove to be a reliable result. Long-term monitoring of projects of this type would allow researchers and land managers to more fully understand the effect that large-scale breakwaters have on not only the shoreline vegetative community, but also on the marsh system as a whole.

5. Conclusions

While this study shows that large-scale breakwaters in high wave energy environments do not promote the growth of shoreline plantings, they do effectively reduce wave impacts on the shoreline and enhance natural fringing vegetation. By reducing wave energy, breakwaters mitigate the pressure on vegetation for upland migration and allow for seaward expansion. This effect is likely to become more important as sea levels rise. However, without adaptive management, breakwaters will not be able to maintain their effectiveness as they become submerged over time. Adaptive management that considers predicted future environmental changes will ensure the success of large-scale shoreline restoration and protection projects like these into the future.

Declaration of Competing Interest

None.

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References

Aguilera, M., Tapia, J., Gallardo, C., Nunez, P., Varas-Belemmi, K., 2020. Loss of coastal ecosystem spatial connectivity and services by urbanization: Natural-to-urban integration for bay management. J. Environ. Manag. 276, 111297. https://doi.org/10.1016/j.jenvman.2020.111297.

Akbar, A., Sartohadi, J., Djohan, T., Ritohardoyo, S., 2017. The role of breakwaters on the rehabilitation of coastal and mangrove forests in West Kalimantan. Indonesia

Ocean & Coast Manag 138, 50–59. https://doi.org/10.1016/j.ocecoaman.2017.01.004.
Allen, R., Webb, B., 2011. Determination of wave transmission coefficients for oyster shell bag breakwaters. Coast Eng Practice 684-697. https://doi.org/10.1061/41190(422)57.
Balke, T., Bouma, T.J., Horstman, E.M., Webb, E.L., Erftemeijer, P.L., Herman, P.M., 2011. Windows of opportunity: thresholds to mangrove seedling establishment on tidal flats. Mar. Ecol. Prog. Ser. 440, 1–9. https://doi.org/10.3354/meps09364.
Bilkovic, D., Mitchell, M., Davis, J., Herman, J., Andrews, E., Mason, P., Tahvildari, N., Davis, J., Dixon, R., 2019. Defining boat wake impacts on shoreline stability toward management and policy solutions. Ocean Coast. Manag. 182, 104945. https://doi.org/10.1016/j.ocecoaman.2019.104945.
Birben, A., Ozolcer, I., Karasu, S., Komurcu, M., 2007. Investigation of the effects of offshore breakwater parameters on sediment accumulation. Ocean Eng. 34 (2), 284–302. https://doi.org/10.1016/j.oceaneng.2005.12.006.
Borchert, S., Oslands, M., Enwright, N., Griffith, K., 2018. Coastal wetland adaptation to sea level rise: Quantifying potential for landward migration and coastal squeeze. J. Appl. Ecol. 55 (6), 2876–2887. https://doi.org/10.1111/1365-2664.13169.
Brinson, M., Christian, R., Blum, L., 1995. Multiple states in the sea-level induced transition from terrestrial forest to estuary. Estuaries. 18 (4), 648–659. https://doi.org/10.2307/1352383.
Brower, J.E., Zar, H., von Ende, C.N., 1990. Field and Laboratory Methods for General Ecology. McGraw-Hill, Boston, MA.
Browne, J., 2017. Long-term erosional trends along channelized salt marsh edges. Estuar. Coasts 40 (6), 1566–1575. https://doi.org/10.1007/s12237-017-0245-y.
Chasten, M., Rosati, J., McCormick, J., Randall, R., 1993. Engineering design guidance for detached breakwaters as shoreline stabilization structures. U.S. Army Corps of Engineers, Washington DC.
Chowdhury, M., Walles, B., Sharifuzzaman, S., Hossain, M., Ysebaert, T., Smaal, A., 2019. Oyster breakwater reefs promote adjacent mudflat stability and salt marsh growth in a monsoon dominated subtropical coast. Sci. Report. 9 (1), 8549. https://doi.org/10.1038/s41598-019-44925-6.
Collini, R., Sweet, W., Weaver, C., Roche, C., Fulford, C., Garfield, N., Hanisko, M., Hintzen, K., Luscher, A., Marcy, D., Scott, G., Spiegler, S., Stiller, H., Sudol, T., 2018. Sea Level Rise Two Pager Resource. masgc.org/northern-gulf-of-mexico-sentinel-site-co/two-pager (accessed 4 December 2020).
Constantin, A., Broussard, P., Cherry, J., 2019. Environmental gradients and overlapping ranges of dominant coastal wetland plants in weeks Bay. AL Southeast Naturalist 18 (2), 224–239. https://doi.org/10.1656/058.018.0202.
Costanza, R., de Groot, R., Sutton, P., van der Ploeg, S., Anderson, S., Kubiszewski, I., Farber, S., Turner, R., 2014. Changes in the global value of ecosystem services. Glob. Environ. Chang. 26, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002.
Dahl, T., 2011. Status and Trends of Wetlands in the Conterminous United States 2004 to 2009. US Department of the Interior, US Fish and Wildlife Service.
Davidson, N., 2014. How much wetland has the world lost? Long-term and recent trends in global wetland area. Mar. Freshw. Res. 65 (10), 934–941. https://doi.org/10.1071/MF14173.
Davis, J., Takacs, R., Schnabel, R., 2006. Evaluating ecological impacts of living shorelines and shoreline habitat elements: an example from the upper western Chesapeake Bay. In: Management, Policy, Science, and Engineering of Nonstructural Erosion Control in the Chesapeake Bay, 55.
Davis, J., Currin, C., O'Brien, C., Raffenburg, C., Davis, A., 2015. Living shorelines: coastal resilience with a blue carbon benefit. PLoS One 10 (11), e0142595. https://doi.org/10.1371/journal.pone.0142595.
Dugan, J., Emery, K., Alber, M., Alexander, C., Byers, J., Gehman, A., McLenaghan, N., Sojka, S., 2018. Generalizing ecological effects of shoreline armoring across soft sediment environments. Estuar. Coasts 41 (1), 180–196. https://doi.org/10.1007/s12237-017-0254-x.
Duggan-Edwards, M., Pages, J., Jenkins, S., Bouma, T., Skov, M., 2020. External conditions drive optimal planting configurations for salt marsh restoration. J. Appl. Ecol. 57 (3), 619–629. https://doi.org/10.1111/1365-2664.13550.
Ellison, A., Felson, A., Friess, D., 2020. Mangrove Rehabilitation and Restoration as Experimental Adaptive Management. Front. Mar. Sci. 7, 327. https://doi.org/10.3389/fmars.2020.00327.
Enwright, N., Griffith, K., Osland, M., 2016. Barriers to and opportunities for landward migration of coastal wetlands with sea-level rise. Front. Ecol. Environ. 14 (6), 307–316. https://doi.org/10.1002/fee.1282.
Field, C., Gjerdrum, C., Elphick, C., 2016. Forest resistance to sea-level rise prevents landward migration of tidal marsh. Biol. Conserv. 201, 363–369. https://doi.org/10.1016/j.biocon.2016.07.035.
Fitzgerald, D., Fenster, M., Argow, B., Buynevich, I., 2008. Coastal impacts due to sea-level rise. Annu. Rev. Earth Planet. Sci. 36, 601–647. https://doi.org/10.1146/annurev.earth.35.031306.140139.
Ganju, N., 2019. Marshes are the new beaches: Integrating sediment transport into restoration planning. Estuar. Coasts 42 (4), 917–926. https://doi.org/10.1007/s12237-019-00531-3.
Garcia, P., 2019. A pilot project for beach restoration using a submerged breakwater-Ponta da Praia, Santos. Brasil J Integrated Coast Zone Manag 19 (1), 43–57. https://ojs.aprh.pt/index.php/rgci/article/view/240.
Gittman, R., Popowich, A., Bruno, J., Peterson, C., 2014. Marshes with and without sills protect estuarine shorelines from erosion better than bulkheads during a Category 1 hurricane. Ocean Coast. Manag. 102, 94–102. https://doi.org/10.1016/j.ocecoaman.2014.09.016.
Gittman, R., Fodrie, F., Popowich, A., Keller, D., Bruno, J., Currin, C., Peterson, C., Piehler, M., 2015. Engineering away our natural defenses: an analysis of shoreline hardening in the US. Front. Ecol. Environ. 13 (6), 301–307. https://doi.org/10.1890/150065.
Gittman, R., Peterson, C., Currin, C., Fodrie, F., Piehler, M., Bruno, J., 2016. Living shorelines can enhance the nursery role of threatened estuarine habitats. Ecol. Appl. 26 (1), 249–263. https://doi.org/10.1890/14-0716. https://www.gulfspillrestoration.noaa.gov.
Hauser, S., Meixler, M., Laba, M., 2015. Quantification of impacts and ecosystem services loss in New Jersey coastal wetlands due to Hurricane Sandy storm surge. Wetl. 35 (6), 1137–1148.
Herbert, D., Astrom, E., Bersoza, A., Batzer, A., McGovern, P., Angelini, C., Wasman, S., Dix, N., Sheremet, A., 2018. Mitigating erosional effects induced by boat wakes with living shorelines. Sustain. 10 (2), 436. https://doi.org/10.3390/su10020436.
Herder, T., 2014. Living Shorelines: A Guide for Alabama Property Owners [Mobile Bay National Estuary Program].
Houser, C., 2010. Relative importance of vessel-generated and wind waves to salt marsh erosion in a restricted fetch environment. J. Coast. Res. 26.2 (262), 230–240. https://doi.org/10.2112/08-1084.1.
Johnson, W., Gosselink, J., 1982. Wetland loss directly associated with canal dredging in the Louisiana coastal zone. In: Proceedings of the Conference on Coastal Erosion and Wetland Modification in Louisiana: Causes, Consequences, and Options. Center for Wetland Resources. Louisiana State University, Baton Rouge, La, p. 1982.
Kamali, B., Hashim, B., 2011. Mangrove restoration without planting. Ecol. Eng. 37 (2), 387–391. https://doi.org/10.1016/j.ecoleng.2010.11.025.
Kirwan, M., Megonigal, J., 2013. Tidal wetland stability in the face of human impacts and sea-level rise. Nature. 504, 53–60. https://doi.org/10.1038/nature12856.
Lee, S., Dunn, R., Young, R., Connolly, R., Dale, P., Dehayr, R., Lemckert, C., Mckinnon, S., Powell, B., Teasdale, P., Welsh, D., 2006. Impact of urbanization on coastal wetland structure and function. Austral Ecol 31 (2), 149–163. https://doi.org/10.1111/j.1442-9993.2006.01581.x.
Long, W., Grow, J., Majoris, J., Hines, A., 2011. Effects of anthropogenic shoreline hardening and invasion by Phragmites australis on habitat quality for juvenile blue crabs (Callinectes sapidus). J. Exp. Marine Biol. And Ecol 409 (1–2), 215–222. https://doi.org/10.1016/j.jembe.2011.08.024.
Martin, S., Sparks, E.L., Constantin, A.J., Cebrian, J., Cherry, J.A., 2021. Restoring Fringing Tidal Marshes for Ecological Function and Ecosystem Resilience to Moderate Sea-level rise in the Northern Gulf of Mexico. Environ. Manag. 67 (2), 384–397. https://doi.org/10.1007/s00267-020-01410-5.
McCauley, L., Jenkins, D., Quintana-Ascencio, P., 2013. Isolated wetland loss and degradation over two decades in an increasingly urbanized landscape. Wetl. 33 (1), 117–127. https://doi.org/10.1007/s13157-012-0357-x.
McKee, K., Cherry, J., 2009. Hurricane Katrina sediment slowed elevation loss in subsiding brackish marshes of the Mississippi River delta. Wetl. 29 (1), 2–15. https://doi.org/10.1672/08-32.1.
McKee, K., Mendelssohn, I., Hester, M., 2020. Hurricane sedimentation in a subtropical salt marsh-mangrove community is unaffected by vegetation type. Estuar., Coast. And Shelf Sci. 239, 106733. https://doi.org/10.1016/j.ecss.2020.106733.
McLoughlin, S., Wiberg, P., Safak, I., McGlathery, K., 2015. Rates and forcing of marsh edge erosion in a Shallow Coastal Bay. Estuar. Coasts 38, 620–638. https://doi.org/10.1007/s12237-014-9841-2.
Morris, R., Bilkovic, D., Boswell, M., Bushek, D., Cebrian, J., Goff, J., Kibler, K., La Peyre, M., McClenachan, G., Moody, J., Sacks, P., Shinn, J., Sparks, E., Temple, N., Walters, L., Webb, B., Swearer, S., 2019. The application of oyster reefs in shoreline protection: are we over-engineering for an ecosystem engineer? J. Appl. Ecol. 56 (7), 1703–1711. https://doi.org/10.1111/1365-2664.13390.
Morton, R., Barras, J., 2011. Hurricane impacts on coastal wetlands: a half-century record of storm-generated features from southern Louisiana. J. Coast. Res. 27.6A, 27–43. https://doi.org/10.2112/JCOASTRES-D-10-00185.1.
Nordstrom, K., Jackson, N., Rafferty, P., Raineault, N., Grafals-Soto, R., 2009. Effects of bulkheads on estuarine shores: an example from Fire Island National Seashore. USA J Coast Res 1, 188–192. https://www.jstor.org/stable/25737563.
North Carolina Division of Coastal Management, 2011. Weighing your options. http://files.nc.gov/ncdeq/Coastal%20Management/coastal-reserve/research/publications/Weighing-your-Options-Final-5x7-11-18-15.pdf.
Oakley, B., Murphy, C., Lee, K., Hollis, R., Caccioppoli, B., King, J., 2020. Sediment deposition following construction of a breakwater harbor: point judith harbor of refuge, Rhode Island, USA. J Marine Sci Eng 8 (11), 863. https://doi.org/10.3390/jmse8110863.
O'Brien, E., Zedler, J., 2006. Accelerating the restoration of vegetation in a southern California salt marsh. Wetl Ecol Mang 14 (3), 269–286. https://doi.org/10.1007/s11273-005-1480-8.
Onorevole, K., Thompson, S., Piehler, M., 2018. Living shorelines enhance nitrogen removal capacity over time. Ecol. Eng. 120, 238–248. https://doi.org/10.1016/j.ecoleng.2018.05.017.
Perry, D., Chaffee, C., Wigand, C., Thornber, C., 2020. Implementing adaptive management into a climate change adaptation strategy for a drowning New England salt marsh. J. Environ. Manag. 270, 110928. https://doi.org/10.1016/j.jenvman.2020.110928.
Polk, M., Eulie, D., 2018. Effectiveness of living shorelines as an erosion control method in North Carolina. Estuar. Coasts 41 (8), 2212–2222. https://doi.org/10.1007/s12237-018-0439-y.
Raabe, E., Stumpf, R., 2016. Expansion of tidal marsh in response to sea-level rise: Gulf Coast of Florida, USA. Estuar. Coasts 39 (1), 145–157. https://doi.org/10.1007/s12237-015-9974-y.
Roberts, S., 2007. The national academies report on mitigating shore erosion along sheltered coasts. In: Management, Policy, Science, and Engineering of Nonstructural

Erosion Control in the Chesapeake Bay, 2006, pp. 3–6. https://doi.org/10.1061/40926(239)124.
Roland, R., Douglass, Scott, 2005. Estimating wave tolerance of *Spartina alterniflora* in coastal Alabama. Coast. Res. 3, 453–463. https://doi.org/10.2112/03-0079.1.
Safak, I., Norby, P., Dix, N., Grizzle, R., Southwell, M., Veenstra, J., Acevedo, A., Cooper-Kolb, T., Massey, L., Sheremet, A., Angelini, C., 2020. Coupling breakwalls with oyster restoration structures enhances living shoreline performance along energetic shorelines. Ecol. Eng. 158, 106071. https://doi.org/10.1016/j.ecoleng.2020.106071.
Safty, H., Marsooli, R., 2020. Ship Wakes and their potential Impacts on Salt Marshes in Jamaica Bay. New York J Marine Sci Eng 8 (5), 325. https://doi.org/10.3390/jmse8050325.
Sharma, S., Goff, J., Moody, R., McDonald, A., Byron, D., Heck, K., Powers, S., Ferraro, C., Cebrian, J., 2016. Effects of shoreline dynamics on saltmarsh vegetation. PLoS One 11 (7), e0159814. https://doi.org/10.1371/journal.pone.0159814.
Silinski, A., Schoutens, K., Puijalon, S., Schoelynck, J., Luyckx, D., Troch, P., Meire, P., Temmerman, S., 2018. Coping with waves: Plasticity in tidal marsh plants as self-adapting coastal ecosystem engineers. Limnol. Oceanogr. 63 (2), 799–815. https://doi.org/10.1002/lno.10671.
Silliman, B., Schrack, E., He, Q., Cope, R., Santoni, A., Van der Heide, T., Jacobi, R., Jacobi, M., Van de Koppel, J., 2015. Facilitation shifts paradigms and can amplify coastal restoration efforts. Proc National Acad Sci 112 (46), 14295–14300. https://doi.org/10.1073/pnas.1515297112.
Simanjuntak, P., 2018. Performance evaluation on low-crest breakwater at north coast of Java Island. J. Civil Eng. Forum 4 (2).
Smee, D., 2019. Coastal ecology: living shorelines reduce coastal erosion. Curr. Biol. 29 (11), R411–R413. https://doi.org/10.1016/j.cub.2019.04.044.
Smith, C., Gittman, R., Neylan, I., Scyphers, S., Morton, J., Fodrie, F., Grabowski, J., Peterson, C., 2017. Hurricane damage along natural and hardened estuarine shorelines: using homeowner experiences to promote nature-based coastal protection. Mar. Policy 81, 350–358. https://doi.org/10.1016/j.marpol.2017.04.013.
Sparks, E., Cebrian, J., Tobias, C., May, C., 2015. Groundwater nitrogen processing in Northern Gulf of Mexico restored marshes. J. Environ. Manag. 150, 206–215. https://doi.org/10.1016/j.jenvman.2014.11.019.
Tamin, N., Zakaria, R., Hashim, R., Yin, Y., 2011. Establishment of *Avicennia marina* mangroves on accreting coastline at Sungai Haji Dorani, Selangor. Malaysia Estuar Coast Shelf Sci 94 (4), 334–342. https://doi.org/10.1016/j.ecss.2011.07.009.
Temple, N., Webb, B., Sparks, E., Linhoss, A., 2020. Low-cost pressure gauges for measuring water waves. J. Coast. Res. 36 (3), 661–667. https://doi.org/10.2112/JCOASTRES-D-19-00118.1.
Temple, N., Sparks, E., Cebrian, J., Martin, S., Firth, D., 2021. Nitrogen removal in constructed marshes at sites protected from and exposed to waves. Wetl. Ecol. Manag. https://doi.org/10.1007/s11273-021-09800-0.
Tomiczek, T., O'Donnell, K., Furman, K., Webbmartin, B., 2020. Rapid damage assessments of shorelines and structures in the florida keys after Hurricane Irma. Nat Hazards Rev 21 (1), 05019006. https://orcid.org/0000-0002-2356-7283.
Torio, D., Chmura, G., 2013. Assessing coastal squeeze of tidal wetlands. J. Coast. Res. 29 (5), 1049–1061. https://doi.org/10.2112/JCOASTRES-D-12-00162.1.
Tweel, A., Turner, R., 2012. Landscape-scale analysis of wetland sediment deposition from four tropical cyclone events. PLoS One 7 (11), e50528. https://doi.org/10.1371/journal.pone.0050528.
U.S. Army Corps of Engineers, 2002. Coastal engienering manual (vols. 1110-2-1100). U. S. Army Corps of Engineers, Washington, DC.
Vona, I., Gray, M., Nardin, W., 2020. The Impact of Submerged Breakwaters on Sediment distribution along Marsh Boundaries. Water. 12 (4), 1016. https://doi.org/10.3390/w12041016.
Wallace, D., Anderson, J., 2013. Unprecedented erosion of the upper Texas coast: Response to accelerated sea-level rise and hurricane impacts. Bulletin 125 (5-6), 728–740. https://doi.org/10.1130/B30725.1.
Webb, B., Allen, R., 2017. Wave transmission through artificial reef breakwaters. Coastal Structures and Solutions to Coastal Disasters 2015: Resilient Coast. Communities 432–441. https://doi.org/10.1061/9780784480304.046.
Wigand, C., Ardito, T., Chaffee, C., Ferguson, W., Paton, S., Raposa, K., Vandemoer, C., Watson, E., 2017. A climate change adaptation strategy for management of coastal marsh systems. Estuar. Coasts 40 (3), 682–693. https://doi.org/10.1007/s12237-015-0003-y.
Wiles, R., LeRoy, S., 2019. High Tide Tax: The Price to Protect Coastal Communities from Rising Seas. http://repo.floodalliance.net/jspui/handle/44111/3067.