Evaluating the Ability of Constructed Intertidal Eastern Oyster (*Crassostrea virginica*) Reefs to Address Shoreline Erosion in South Carolina

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Abstract. The application of nature-based solutions to address shoreline erosion and the loss of salt marsh in coastal South Carolina has centered around the creation of intertidal oyster (*Crassostrea virginica*) reefs that act as natural breakwaters. The installation of such living shoreline materials often results in a rapid accumulation of fine sediments, followed by wild oyster recruitment to suitable materials, and then more gradually the growth of salt marshes (primarily *Spartina alterniflora*). Leveraging more than two decades of oyster reef restoration and living shorelines research at the South Carolina Department of Natural Resources, this study quantitatively assessed performance rates for both percent oyster cover and marsh protection in relation to reef age. Determining such rates will serve to inform the expectations of prospective adopters of living shorelines as to the timeframes of some of the biological processes, as measures of performance success, that will occur following material installation. Performance success was investigated in terms of recruitment of oysters to installed materials and the creation of new marsh habitat or protection of existing marsh from erosion. Reef age was an important determinant of reef "success", with significant relationships between reef age and both performance success metrics. Percent oyster cover reached 40% by two years post-installation and 50% by four years post-installation, indicative of high rates of oyster recruitment. The relative marsh protection rate of living shorelines compared to unprotected reference plots was 0.4 m yr^{-1} Reef performance differed based on bank substrate firmness, bank width, shoreline morphology, and location relative to the Intracoastal Waterway (ICW). Firmer bank substrate was associated with greater percent oyster cover. Broader bank width was associated with greater marsh protection. Higher percent oyster cover measurements were observed on straight, natural shorelines and reefs located along the ICW. Reefs located on the ICW were also associated with greater marsh protection than reefs at non-ICW sites. Further, this study demonstrates that bagged oyster shell reefs are capable of providing shoreline protection services for more than a decade and can endure multiple intense storm events. The results of this study were also used to facilitate the implementation of new living shoreline regulations in coastal South Carolina in the hope of broadening adoption of this approach to addressing shoreline erosion and salt marsh habitat loss.

INTRODUCTION

Coastal salt marshes have declined in area over the last 200 years in the conterminous United States and are currently being lost at a higher rate than any other wetland category (Dahl 2011). In South Carolina, USA, a recent shoreline assessment classified over 57% of coastal shorelines as erosional, with an average long-term marsh erosion rate of 0.34 m yr^{-1} (Jackson 2017). The loss of salt marsh habitats is of particular concern as they represent some of the most biologically productive and ecologically valuable habitats in the coastal region (e.g., Whittaker and Likens 1973; Odum

1979; Day et al. 1989; Levin et al. 2001; Peterson et al. 2007; Barbier et al. 2011). These marsh habitats are at risk of degradation and loss through synergistic relationships between sea level rise, increased storm frequency, coastal development, and shoreline hardening (Peterson et al. 2008a, b; Titus et al. 2009; Mattheus et al. 2010; Nicholls and Cazenave 2010; Rahmstorf 2010; Grinsted et al. 2013). Furthermore, Beck et al. (2011) highlighted the imperiled nature of oyster reefs on a global scale.

Fringing intertidal oyster (*Crassostrea virginica* Gmelin 1791) reefs, which occur in close spatial proximity to salt marshes, are a dominant feature of the estuarine landscape in South Carolina (Bahr and Lanier 1981; Burrell 1986). These fringing wetland systems support valuable ecological services by improving water quality (Titus 1988; Dame 1999; Dame et al. 2001; Porter et al. 2004), providing shelter and nursery habitat for aquatic species (Minello et al. 1994; Peterson and Turner 1994; Shervette and Gelwick 2008), often at higher diversities and abundances than unvegetated habitats (e.g., Coen et al. 1999; Posey et al. 1999; Kingsley-Smith et al. 2012), and by protecting and preserving adjacent salt marshes (primarily *Spartina alterniflora*) through both the dissipation of wave and tidal energies and the facilitation of sedimentation (Redfield 1972; Meyer et al. 1997; Leonard and Reed 2002; Möller and Spencer 2002; Piazza et al. 2005; Barbier et al. 2008; Shepard et al. 2011).

BACKGROUND AND RELATED WORK

Significant portions of the South Carolina coastal landscape have been adversely affected by rapid human population growth and associated coastal development in recent years, with projections for continued increases in both population and development (CRDA 2018; US Census Bureau 2018; Hauer 2019). Commensurate with such growing coastal human populations, the numbers of registered boats in Charleston County increased steadily from 28,752 in 2010 to 38,951 in 2021 (SCDNR, unpubl. data), likely exacerbating the stresses on fringing oyster reef and salt marsh habitats from the erosional boat wakes. Much of this coastal development has also been accompanied by the hardening of shorelines with bulkheads, riprap revetments, and seawalls. Bulkheads can have deleterious impacts on marsh habitats, including the isolation of the marsh from the upland, which disrupts the land-water continuum, the restriction of the sediment supply to the marsh (Redfield 1972; Chauhan 2009), the limitation of the ability for coastal marshes to migrate landward (Titus 1988; Peterson et al. 2008a), and the reflection of wave energy that can increase erosion and sediment scour (National Research Council 2007). As a result, bulkheads can lead to the transformation of a gently sloped shoreline into a steep transition from the upland to the subtidal zone by eliminating the intertidal area (see Currin et al. 2010). Further, these structures can be associated with reduced abundances of coastal marsh plant species, fishes, crustaceans, and benthic infauna (Bozek and Burdick 2005; Seitz et al. 2006; Bilkovic and Roggero 2008).

The application of living shorelines for shoreline stabilization in South Carolina has been relatively limited compared to traditional shoreline hardening approaches, which is in part due to the lack of a clear regulatory pathway. Living shorelines are valued for the additional ecosystem benefits that they provide, such as improvements to water quality (Onorevole et al. 2018) and habitat provision (Sutton-Grier et al. 2015). The suite of ecological benefits supported by living shorelines cannot be attained through the use of bulkheads or revetments (Bozek and Burdick 2005; Seitz et al. 2006; Bilkovic and Roggero 2008).

One obstacle that the increased adoption of living shorelines faces is that many private property owners assume that bulkheads provide superior protection from erosion and storm damage (Fear and Currin 2012; Scyphers et al. 2014); however, studies supporting this claim are lacking, particularly with regard to storm effects (Shepard et al. 2011). Studies have shown that bulkheads experience considerable damage during extreme storm events (Currin et al. 2008) and that marshes offer better shoreline protection (Gittman et al. 2014) and experience less damage (Smith et al. 2017) from significant storm events than bulkheads.

While engineering performance and cost effectiveness are key criteria for property owners in their selection of a shoreline protection approach (Scyphers et al. 2014), private property owners must have confidence that living shorelines methods will reduce shoreline erosion if broader acceptance and adoption of these biogenic methods over more traditional techniques is likely to occur. Further, a regulatory pathway comparable in scope to traditional engineered methods must also be established. In order to facilitate more widespread adoption, the National Oceanic and Atmospheric Administration (NOAA) developed guidance materials for using living shorelines (NOAA 2015). Since environmental conditions and state regulations vary tremendously across the United States, however, a number of states, such as North Carolina (North Carolina 2006), Virginia (Hardaway et al. 2017), and New Jersey (Miller et al. 2016), through the efforts of both state agencies and non-profit organizations, have developed guidance documents specific to local environmental conditions.

To develop a similar guidance framework for South Carolina and to gain an understanding of the effectiveness of intertidal oyster reefs to address shoreline erosion within its coastal zone, the authors of the present study assessed a suite of existing living shorelines constructed by a long-term program focused on oyster reef habitat restoration and enhancement in South Carolina. The majority of the oyster restoration efforts by the South Carolina Department of Nat-

Figure 1. Time series of photographs from the SCDNR South Carolina Oyster Recycling and Enhancement (SCORE) Program site on Hunting Island, South Carolina (32.34467°N, 80.46765°W), illustrating the ability of intertidal bagged oyster shell reefs to facilitate salt marsh (*Spartina alterniflora*) habitat expansion. (A) April 21, 2009: Pre-installation condition of shoreline; note the lack of suitable substrate for oyster recruitment. (B) June 4, 2009: Creation of an intertidal bagged oyster shell reef through the addition of 400 shell bags (0.016 acres) by volunteers. (C) January 12, 2010: Approximately seven months post-installation. Sediment has begun to accumulate on the marsh side of the reef. Mean (±SE) oyster recruitment at this time was measured as 4,693.3 ± 749.1 oysters m^{-2}. (D) June 13, 2013: Approximately four years post-installation. The salt marsh has grown seaward, as much as 10 m, to the back of the reef, without any directed planting of *S. alterniflora*.

ural Resources (SCDNR), dating back to 2001, have focused on increasing environmental awareness and stewardship centered around the ecological importance of oyster reefs, as well as evaluating their provision of habitat to a diversity of other organisms, as supported by published studies from across the region (Hadley et al. 2010; Kingsley-Smith et al. 2012; see also Coen et al. 1999, 2007; Posey et al. 1999; Grabowski and Peterson 2003; Peterson et al. 2003; Tolley and Volety 2005). More recently, however, the ability of intertidal oyster reefs to serve as natural breakwaters to adjacent fringing marshes (*sensu* Bahr and Lanier 1981; Burrell 1986) has become particularly relevant in the context of efforts to address shoreline erosion through natural material-based approaches (Gittman et al. 2016; Bilkovic et al. 2017). As illustrated in Figure 1, the placement of suitable oyster settlement substrate on intertidal shorelines in South Carolina can facilitate the rapid development of reef habitat, followed some years later by the expansion of salt marsh (*S. alterniflora*) habitat.

PROJECT GOALS

The impetus for the present study was to inform the development of new living shorelines regulations by South Carolina's coastal zone management agency (the South Carolina Department of Health and Environmental Control, SCDHEC) to facilitate their broader adoption as a shoreline erosion control measure (SCDNR 2019). The primary goal of this study was to assess how successful previously-established living shorelines, constructed over 15-plus years of oyster restoration efforts by the SCDNR in coastal South Carolina, addressed the problem of marsh edge shoreline erosion. Success was investigated in terms of recruitment of oysters to installed materials and the creation of new marsh habitat or protection of existing marsh from erosion. A secondary goal was to identify physical factors at these living shoreline sites that were significantly associated with the relative success of oyster recruitment (measured as percent oyster cover) and marsh protection.

MATERIALS AND METHODS

STUDY DESIGN

A suite of existing fringing intertidal bagged oyster (*C. virginica*) shell reefs, representing a chronosequence of oyster restoration efforts dating back to 2001, were assessed on a single occasion in the late fall or winter (October through February) of either 2016 or 2017. These reefs were constructed primarily by the SCDNR's South Carolina Oyster Recycling and Enhancement (SCORE) Program for oyster restoration purposes and a subset of available reefs were selected for this study to encompass a wide range of reef ages (0.4 to 15.5 years) and physical site characteristics. Reefs of multiple ages were present at most sites, which allowed for the assessment of 39 existing reefs at 18 sites (Figure 2; Supplemental Table 1). Reefs evaluated in this study were typically constructed as a single layer of oyster shell bags, arranged in four contiguous rows parallel to the shoreline. Each shell bag typically measured 0.50 m (length) x 0.23 m (width) x 0.15 m (height) and contained two-thirds of a U.S. bushel (~5 gallons; 19 L) of generally harvestable sized (7.82 cm shell height) oyster (*C. virginica*) shell sourced from SCDNR's oyster shell recycling program. The shell was quarantined on land for at least six months prior to use. The mesh bags were comprised of UV-stabilized polypropylene manufactured by Delstar Technologies, Inc. At a subset of sites with soft intertidal substrate, the shell bags were placed on top of wooden pallets to provide additional elevation while distributing weight over a greater surface area to reduce the sinking of the shell bags into the sediment. Both of these benefits served to encourage oyster recruitment and growth by reducing the burial of oyster spat. The edges of all bagged oyster shell reefs were staked down with rebar. The bagged oyster shell reefs evaluated in this study ranged in length along the shoreline from 4.3 m to 120.2 m (mean = 19 m, S.D. = 19 m, n = 39).

One reference plot (i.e., an adjacent shoreline with no installed reef) was established at each site at the time of monitoring. These contemporary reference plots were assumed to approximate conditions at the reef plots had reefs never been installed at those locations. Reference plots were positioned similarly to the adjacent installed reef with respect to location (e.g., tidal elevation) on the intertidal shoreline and identified at the nearest location on the shoreline that was visually representative of the site, and which also lacked existing oyster reefs. The mean distance between reference plots and their nearest paired study reef was 24.1 m, and the distance ranged from 0.3 m to 167.3 m.

DEPENDENT VARIABLES

Responses of reef performance metrics indicative of success in addressing shoreline erosion were as follows: (1) an increase in *percent oyster cover*, which creates a living reef to buffer wave energy and creates a relatively sheltered environment behind the reef; and (2) a shorter distance between the reef and marsh edge relative to the corresponding positions along the reference shoreline (*marsh protection*). To measure the dependent variable percent oyster cover, a 0.5-m x 0.5-m quadrat with an attached ruler (included as a known size reference) was placed at three pre-selected positions on each reef. A photograph was captured directly overhead of each quadrat sample area using a waterproof Olympus® Tough TG-4 digital camera. These photographs were captured as JPEGs and processed using ImageJ (Schneider et al. 2012). The horizontal coverage of oysters that had recruited to the deployed material since reef construction and that were present above the upper surface of the mesh bag within each quadrat was digitally outlined to obtain an estimate of percent oyster cover. Of the recruited oysters, live oysters were not differentiated from dead oysters based on these digital images; nonetheless, this param provided a representative measure of the extent of oyster reef habitat establishment for individual reefs. As part of QA/QC procedures for the digital images used to determine percent oyster cover, one-third (33%) of quadrat photographs were independently digitized by two observers, a primary and secondary observer, and values for percent oyster cover compared. In cases where the differences between observers was ≤ 5%, the value of the primary observer was used. For any case where the difference in coverage was > 5%, the individual quadrat image was re-digitized by both observers until the difference was ≤ 5%, at which point the value of the primary observer was used.

To assess any difference in marsh edge position potentially attributable to the protective presence of a constructed reef (*marsh protection*), the distance from each reef to the marsh edge was subtracted from the distance from the reference plot to the marsh edge and compared at the reef level. This metric can be thought of as an inferred measure of marsh growth seaward attributable to the installation of the living shoreline, with the general trend for the value of this metric to increase over time (i.e., with reef age), but a positive result may also reflect a reduction in the loss of marsh from erosion due to its physical protection by the reef. The horizontal distance from the reef or reference plot to the marsh edge was measured with global navigation satellite system (GNSS) data at three pre-selected locations at each plot. Two reefs that were modified by marsh plantings and two reefs with tree branches overhanging the marsh (interfering with GNSS measurements) were excluded from marsh protection analyses. Horizontal and vertical positions were collected using a Trimble R8 survey-grade GNSS receiver. Position data were corrected in real time using solutions provided by the South Carolina Real Time Network (SCRTN). Horizontal data referenced the North American Datum of 1983 and the Universal Transverse Mercator projected coordinate system. Vertical data were collected as orthometric heights (Geoid 12B) referenced to the North American Vertical Datum of 1988. The accuracy of this GNSS receiver was assessed by comparing data collected at National Geodetic Survey (NGS) benchmarks to the reported established survey data for the benchmarks. In terms of data accuracy, the root mean square error (RMSE) of the GNSS was < 0.05 m for both horizontal and vertical measurements.

INDEPENDENT VARIABLES

A suite of independent variables was compiled to explore relationships between site characteristics and reef performance. *Reef age* at the time of assessment, expressed in decimal years elapsed since reef installation, was included as an independent variable in all analyses. *Bank width* was determined at the site level at the reference plot at the time of sampling (near low tide), using a measuring tape extended from the marsh edge to the waterline (generally +/- 1 hour of the predicted low tide). Reference *bank slope* was calculated from three sets of paired GNSS positions collected in front of (waterbody side) and behind (marsh side) the reference plot, with the goal of approximating the bank slope that was present at that site at the time of the installation of each reef. *Sink depth* was measured at the site level at the reference plot near low tide using a standard cinderblock (20 cm × 20 cm × 40 cm, weighing approximately 17 kg) dropped narrow end down from a 1 m height. A m stick was then used to measure how deeply the block penetrated beneath the sediment surface on the upslope edge of the depression. *March salinity* (psu) was calculated as mean salinity in March from the data sources nearest each site, compiled from publicly available long-term data (1995 to 2014) collected by SCDHEC and available online (https://www.waterqualitydata.us/portal/). As a first step, salinity data were summarized to identify general trends in annual salinity, and March was identified as the month that typically had the lowest mean salinity. Given

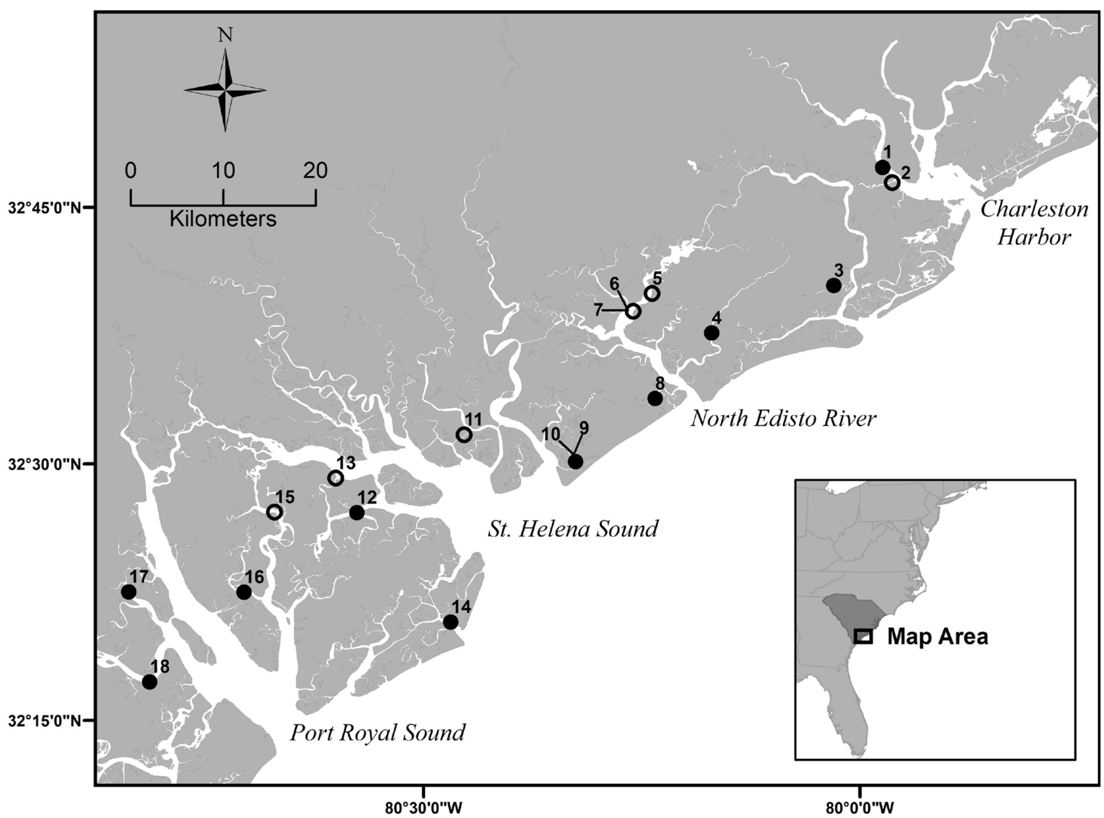

Figure 2. Locations of 39 intertidal bagged shell oyster reefs at 18 sites evaluated as part of this study. Open circles indicate sites located adjacent to the Intracoastal Waterway. Data related to each site and reef are presented in Supplemental Table 1. *Map inset*: Mapped area of study sites within the context of the eastern U.S. seaboard (shaded area represents the state of South Carolina).

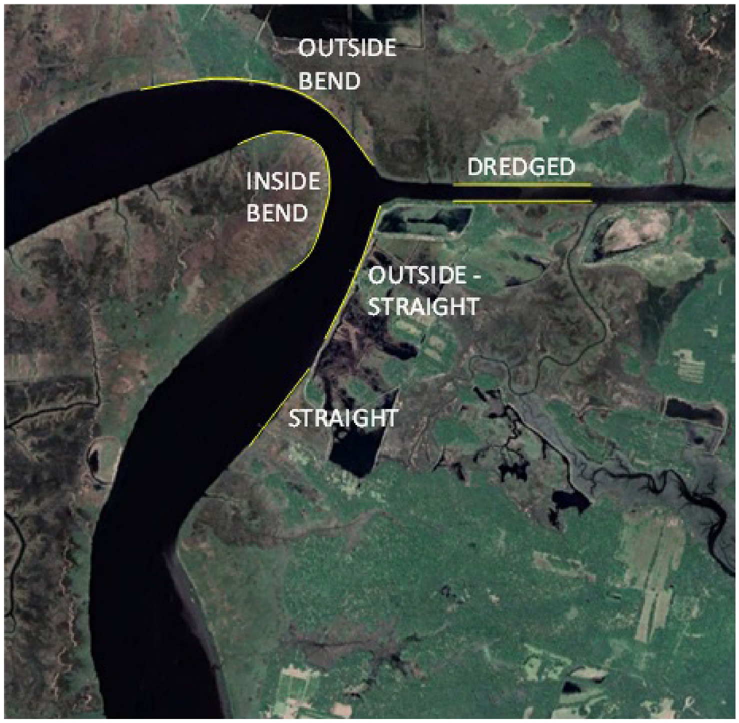

Figure 3. Illustration of categories of shoreline morphology used to investigate the effect of this site characteristic on reef performance metrics. Imagery was obtained from Google Earth, dated October 10, 2016, and includes sections of the Intracoastal Waterway and South Edisto River, South Carolina.

that low salinity conditions can be stressful to oysters and oyster growth rates decrease at salinities 15 psu and lower (La Peyre et al. 2016), mean March salinity was selected for inclusion in analyses. *Waterbody width* (m) perpendicular to the shoreline was determined for each site using aerial imagery. *Shoreline morphology* at each site was characterized using aerial imagery and the following categories: dredged channel, inside bend, outside bend, outside-straight, and straight (Figure 3). Whether each site was located on the Intracoastal Waterway, a potential boat wake exposure factor, was also determined and assigned to a category of yes or no.

DATA ANALYSES

Simple linear regression and logarithmic regression were used to explore relationships between reef age and performance metrics and to identify specific rates of change (slopes of fitted linear regression lines). Stepwise regression analysis was performed to examine the relationships between site characteristics and reef performance metrics. Reef age was included in each model. Due to the large number of independent variables, forward stepwise regression was used to select the most appropriate models. Variables were included if they contributed to the overall model and were prioritized based on their contribution. The residuals of all statistical models presented were tested using the Shapiro-Wilk test ($p > 0.05$) to confirm that distributions were not significantly different from normal. All statistical analyses were conducted using JMP version 15.2 (© 2019 SAS Institute, Inc.). Analysis of covariance (ANCOVA), with Tukey's *post hoc* tests, was used to examine the relationship between shoreline morphology (i.e., dredged channel, inside bend, outside bend, outside-straight, and straight), reef ICW location (yes or no), and performance metrics, after controlling for reef age as the covariate. Interaction terms between the categorical variables (i.e., shoreline morphology and ICW location) and reef age were used to test the assumption of homogeneity of regression slopes, but these were not significant and were removed from subsequent analyses.

RESULTS

SITE CHARACTERISTICS

Each site characteristic is expressed here as mean ± SD and minimum to maximum value (with more details in Supplemental Table 1): *bank width* in m (11.8 ± 7.0, 3.4–34.8); *bank slope* as rise/run (0.10 ± 0.06, 0.01–0.42); *sink depth* in cm (10.0 ± 10.1, 0.3–39.5); *March salinity* in psu (24.0 ± 4.5, 15.6–31.9); and *waterbody width* in m (426 ± 336, 61–1,157). The following numbers of reefs in each shoreline morphology category were evaluated: dredged channel (*n* = 4), inside bend (*n* = 6), outside bend (*n* = 12), outside-straight (*n* = 6), and straight (*n* = 11) (Figure 3). Fourteen reefs were located along the ICW and 25 reefs were not.

RELATIONSHIPS BETWEEN REEF AGE AND REEF PERFORMANCE METRICS

Both of the simple linear regressions and both of the logarithmic regressions performed between reef age and reef performance metrics (*percent oyster cover* and *marsh protection*) were statistically significant (Figure 4). For *percent oyster cover*, logarithmic regression (R^2 = 0.25) explained a greater proportion of variation than linear regression (R^2 = 0.13), indicating that oyster cover increased at the highest rates in the first few years after installation of the reef and then stabilized to a slower rate in subsequent years (Figure 4A). Based on the logarithmic regression equation for reef age versus percent oyster cover, *percent oyster cover* reached

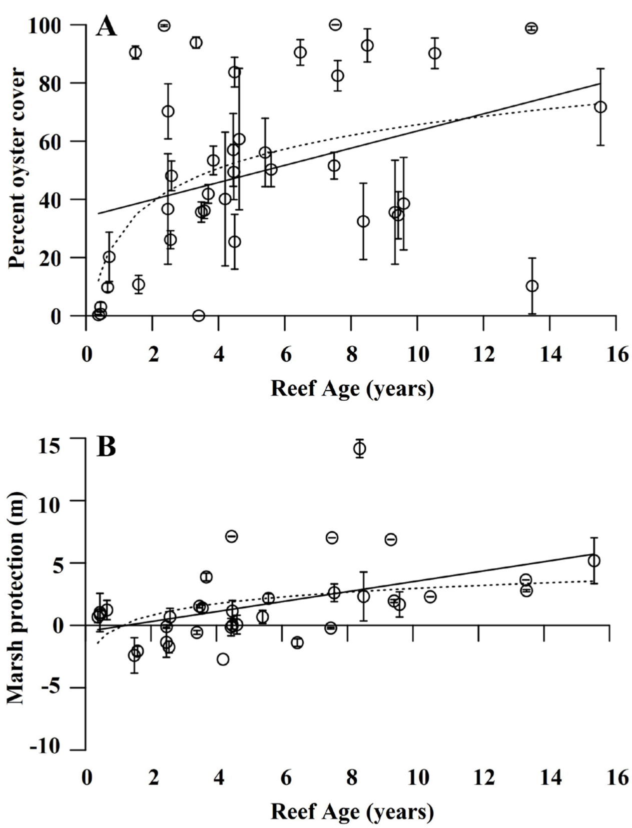

Figure 4. Regression relationships between reef age and reef performance metrics. Error bars indicate standard error calculated from the three replicate measurements at each reef. (A) Percent oyster cover: linear (solid line) equation, $y = 34.013 + 2.948x$, $R^2 = 0.13$, $p = 0.024$; logarithmic (dashed line) equation, $y = 27.968 + 16.366\ln(x)$, $R^2 = 0.25$, $p = 0.001$. (B) Marsh protection: linear (solid line) equation, $y = -0.499 + 0.400x$, $R^2 = 0.22$, $p = 0.004$; logarithmic (dashed line) equation, $y = -0.114 + 1.333\ln(x)$, $R^2 = 0.15$, $p = 0.020$.

40% by two years post-installation and 50% by four years post-installation. For *marsh protection*, linear regression (R^2 = 0.22) explained a greater proportion of variation than logarithmic regression (R^2 = 0.15). Lateral marsh position differences for the studied reefs indicate that the behind-reef position of marsh differed from the adjacent unprotected (reference plot) shoreline with a relative protection rate of 0.40 m yr^{-1}(Figure 4B). This protection rate may represent erosion prevention, marsh expansion, or a combination of the two.

INFLUENCES OF PHYSICAL SITE CHARACTERISTICS ON REEF PERFORMANCE METRICS

Percent oyster cover was highly variable, even at 10–15 years following installation (Figure 4A), prompting an investigation of potential factors explaining the observed variation. After adjusting for reef age, *sink depth* was the most influential independent variable affecting percent oyster cover. Percent oyster cover was significantly inversely associated with substrate sink depth, indicating that firmer sites exhibited greater percent oyster cover (Table 1). For marsh protection, the most influential independent variable after adjusting for age was *bank width*. Reefs located at sites with wider banks were more likely to exhibit greater marsh growth or protection against marsh loss (Table 1). The marsh protection rate behind reefs, determined from the slope of the ANCOVA regression analysis, was slightly lower than for the univariate model, suggesting an average rate of 0.34 m yr^{-1}, with faster rates occurring at sites with wider banks and slower rates at sites with narrower banks. Analyzing older reefs (9-plus years post-installation) separately, the average marsh protection rate decreases to 0.17 m yr^{-1}, indicating a relative stabilization as reefs mature (Figure 4B). Bank slope, waterbody width, and March salinity were not selected by the stepwise regression model (and are therefore not included in either Table 1 or Supplemental Table 1).

SHORELINE MORPHOLOGY AND REEF PERFORMANCE METRICS

There were performance differences among reefs with regard to both shoreline morphology and whether the site was located on the ICW. *Percent oyster cover* was greatest on straight, natural channels, lowest on outside bends, and intermediate at dredged channels, inside bends, and along outside-straight shorelines (Table 2). Reefs located along the ICW were associated with greater marsh protection (more marsh growth or reduced marsh erosion) relative to adjacent reference areas (Table 2). Differences in marsh edge distance from reef and reference areas averaged 3.6 m along the ICW as compared to 0.85 m at non-ICW sites, and the associated relative marsh protection rates were 0.84 m yr^{-1} and 0.36 ms per year, respectively, although the marsh protection

Table 1. Results of stepwise regressions used to identify relationships between site characteristics and reef performance metrics. Reef age was included in each model because it varied widely and is a known factor influencing reef performance. Numbers in parentheses, excluding degrees of freedom, represent *p* values and partial R^2 values, respectively.

	Percent Oyster Cover	Marsh Protection (m)
Adj. R^2	0.22	0.46
F (df_1,df_2)	6.31 (2,36)	15.61 (2,32)
***p* value**	0.005	< 0.0001
Intercept	45.51 (< 0.0001)	-3.02 (0.003)
Reef Age (years)	2.88 (0.019, 0.14)	0.34 (0.003, 0.24)
Sink Depth (cm)	-1.12 (0.017, 0.15)	
Bank Width (m)		0.24 (0.0002, 0.35)

Table 2. ANCOVA results of the effects of shoreline morphology and location along the ICW on reef performance metrics. Location characteristics labeled A correspond to significantly better reef performance success metric values relative to those labeled B, whereas variables with the same letters are not significantly different from one another. n.s. indicates not significant.

		Percent Oyster Cover	Marsh Protection (m)
Shoreline Morphology Test	**Adj. R^2**	0.3	
	F (df_1, df_2)	4.21 (5,33)	
	***p* value**	0.005	n.s.
	Shoreline Morphology (*p* value)	0.017	
	Dredged Channel	AB	
	Inside Bend	AB	
	Outside Bend	B	
	Outside-Straight	AB	
	Straight (natural)	A	
	Reef Age (*p* value)	0.007	
ICW Test	**Adj. R^2**		0.34
	F ($df1$, df_2)		9.57 (2,32)
	***p* value**	n.s.	0.0006
	ICW (*p* value)	n.s.	0.009
	Yes		A
	No		B
	Reef Age (*p* value)		0.0005

relationship for ICW sites was marginally significant (p = 0.054).

DISCUSSION

Anecdotal observations of the recruitment of oysters (*C. virginica*) on intertidal bagged oyster shell followed by the natural expansion of smooth cordgrass (*S. alterniflora)* have now been quantified and related to reef age as well as physical site characteristics. In the oldest living shorelines studied here, quantifiable shoreline protection and ecosystem benefits were present over a decade following initial reef material installation. Many of these living shorelines also persisted through extreme wind and wave events over this same timeline, which complements shorter timescale event-based studies in confirming their potential role in increasing coastal resilience (Polk et al. 2021). These results are valuable in informing expectations for restoration practitioners implementing similar strategies as well as for property owners interested in adopting living shoreline approaches to address marsh edge erosion adjacent to both private and public lands using natural material-based solutions.

The present study revealed significant relationships between reef age and the performance metrics of both percent oyster cover and marsh protection. One benefit of the suite of sites evaluated in this study was that at 11 of the 18 sites, reefs of different ages were created using the same technique (i.e., standardized installation of bagged oyster shell) and were located along the same stretch of shoreline. This allowed for a more direct evaluation of the progression of oyster reef development and marsh protection over time.

The mean living shoreline marsh protection rate reported here (i.e., 0.34 to 0.40 m yr^{-1}) is comparable to rates reported in nearby North Carolina and Florida, where rates were identified using different experimental designs and in differing geographies, and all are on the order of tens of centims per year of either reduced erosion or marsh accretion compared to adjacent shorelines (Polk et al. 2018; McClenachan et al. 2020). While a subset of living shorelines exhibited high oyster cover (> 80%) within a few years following initial installation, oyster coverage on bagged shell living shoreline reefs averaged 40% at two years and 50% at four years but was highly variable in both space and time. This is consistent with the variability and temporal trajectories reported in other studies examining oyster recruitment to shell substrate in both South Carolina (Hadley et al. 2010) and Florida (Safak et al. 2020).

Important site selection characteristics to be considered prior to the installation of living shorelines, or when evaluating alternative approaches for addressing shoreline erosion issues, should include an assessment of substrate firmness, bank width, shoreline morphology, and exposure to boat wakes. Percent oyster cover was inversely associated with sink depth, and greater percent oyster cover was observed at sites located along natural straight channels than on outside bends of tidal creeks and rivers. Marsh protection was positively associated with bank width, and reefs located on the ICW exhibited more than twice the capacity for marsh protection than non-ICW sites, suggesting that the role of oyster reefs in mitigating marsh erosion may be particularly impactful along shorelines characterized by high boat traffic.

While not empirically measured in this study, changes in reef surface elevation over time are also an important consideration regarding the sustainability of living shoreline approaches as compared to fixed manmade structures such as seawalls and bulkheads. Repeated measures for recently installed living shoreline materials, encompassing a broader suite of approaches than presented here, including both oyster- and natural fiber-based approaches (i.e., coir logs), are currently being analyzed (Tweel et al. in prep.), and will complement the work presented here. A key param assessed by Tweel et al. (in prep.) is reef surface elevation. As sea level rises, oyster reef heights may naturally increase such that reefs can persist as natural, growing breakwaters that adjust to tidal elevation, thus offering great potential for shoreline protection in the face of rising sea levels (Rodriguez et al. 2014). This is particularly relevant in coastal South Carolina due to its low elevation, vulnerability to sea level rise, and ongoing coastal flooding concerns (Daniels 1992). Further, global climate change is predicted to affect the intensity, frequency, timing, and distribution of hurricanes and tropical storms (Michener et al. 1997). Shorelines and their associated habitats in the southeastern U.S. have evolved under a particular regime of such events, which in the future could be subject to change. When evaluating the performance of living shorelines, it is important to keep in mind the intensity, frequency, and timing of storm events that these installations have experienced and how the temporal intensity of evaluations has been able to capture the impacts of those storms. For the purposes of the present study, such impacts on both reference and reef plots will have been integrated over the lifespan of these individual plots. When comparing the performance of living shorelines during different timeframes, however, it is important to consider how those timeframes differed in terms of their relative storminess when interpreting the performance metrics of these installations.

In summary, the present study has demonstrated the feasibility of using quantified biological responses, specifically oyster recruitment and change in marsh edge position, to evaluate the performance of oyster reef-based living shorelines in coastal South Carolina. While oyster recruitment happens fairly quickly, often within a matter of months following reef installation, salt marsh expansion into the area between the installed nature-based breakwater and the existing marsh edge occurs more slowly and gradually. The more informed and quantified expectations of living shoreline performance in coastal South Carolina that have been derived from this study have recently been utilized by the state's coastal zone management agency (i.e., SCDHEC) to inform new regulations that were adopted on May 28, 2021. These new regulations will support the adoption of living shorelines as a viable alternative to manmade, engineered structures (e.g., seawalls and bulkheads) to address shoreline erosion and salt marsh habitat loss in coastal South Carolina.

CONCLUSIONS AND RECOMMENDATIONS

Both a long history of community-based oyster restoration at the South Carolina Department of Natural Resources as well as the specific findings presented in this study support the premise that environmental conditions in coastal South Carolina are highly conducive to substrate supplementation-based approaches to intertidal oyster reef creation. As a result, oyster reefs can be effectively created to support a variety of ecosystem services, including but not limited to shoreline protection that facilitates marsh expansion. The marsh protection benefits from these services were quantifiable at the time of monitoring, in some cases 10 or more years following initial reef installation. The southeastern U.S. is experiencing a changing climate that brings with it rising sea levels and increased coastal storm frequency and severity such that resilient, long-term strategies for shoreline protection, both for human and ecosystem dimensions, are critical. The present study provides quantification of the rates of the biophysical processes that result following shoreline stabilization through the addition of bagged oyster shell as substrate by leveraging a long-term restoration program. Further, easily measured site characteristics, such as substrate firmness, bank width, shoreline morphology, and exposure to boat wakes, can serve as valuable predictors of performance success, allowing for more informed site selection to be developed to improve the performance and cost effectiveness of installed living shorelines.

The research team producing this work, however, realizes that additional strategies beyond bagged oyster shell-based approaches are also required, particularly for environmental conditions (e.g., lower salinity regimes, high wave energy environments) where this more traditional approach is unsuitable. As a result, for the past several years the research team has also explored and evaluated both oyster-based (e.g., repurposed crab traps, manufactured wire reefs) and natural fiber-based (e.g., coir logs) living shorelines approaches. Broad spatial-scale evaluations of these materials are required over multi-year timeframes to understand their performance across space and time, and to inform practitioners and citizens wishing to pursue nature-based solutions to address shoreline erosion issues, impacts to upland habitats and infrastructure, and the loss of biologically productive habitats.

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Supplemental Table 1. Reef- and site-specific information for bagged oyster shell reefs evaluated in this study. For sites with multiple reefs of different ages, information is presented for percent oyster cover and marsh protection at the individual reef level and sequentially from youngest to oldest. Other physical characteristics (e.g., bank width, sink depth) are presented at the site level. (Bank slope, waterbody width, and March salinity are not included as they were not selected as significant explanatory variables during stepwise regression analyses.)

Site No. (Figure 1)	Site Name	# Reefs	Reef Ages (Years)	Oyster Cover (%)	Marsh protection (m)	Bank Width (m)	Sink Depth (cm)	Shoreline morphology	ICW?
1	Ashley River	3	0.4,2.5,4.5	3.1, 70.2, 25.4	0.86, -0.10, -0.01	9.3	15.8	Outside-straight	No
2	Wappoo Cut	3	0.4, 2.4, 3.3	0.6, 99.7, 93.9	1.03, planted, N/A	11.3	6.5	Straight	Yes
3	Abbapoola Creek	2	0.7, 4.2	20.2, 40.1	N/A, -2.72	6.0	6.5	Outside-straight	No
4	Boy Scout Camp	1	10.5	90.2	2.29	3.4	14.8	Inside bend	No
5	Wadmalaw River	2	0.4, 3.8	0.2, 53.4	0.68, planted	20.4	6.4	Outside bend	Yes
6	Bears Bluff 1	1	9.3	35.6	6.87	18.8	10.0	Inside bend	Yes
7	Bears Bluff 2	2	3.5, 8.4	35.6, 32.4	1.53, 14.17	34.8	39.5	Inside bend	Yes
8	Ocella Creek	1	4.4	57.0	-0.14	6.0	20.2	Inside bend	No
9	Big Bay Edisto	5	0.7, 3.6, 5.6, 9.4, 13.5	9.7, 36.2, 50.3, 64.5, 10.2	1.23, 1.38, 2.16, 19.5, 2.78	12.3	14.5.	Outside bend	No
10	Big Bay Creek	2	2.6, 7.5	26.1, 51.6	-1.74, -0.22	6.0	7.5	Outside bend	No
11	Coosaw Cut	4	1.6, 3.7, 4.6, 9.6	10.7, 41.9, 60.7, 38.5	-2.06, 3.88, 0.05, 1.67	7.3	2.5	Dredged	Yes
12	Dataw Island	2	7.5, 15.5	100, 71.7	7.02, 5.18	15.4	0.3	Straight	No
13	Lucy Point Creek	1	4.5	49.4	7.13	11.0	1.6	Outside bend	Yes
14	Hunting Island	5	1.5, 2.5, 4.5, 6.5, 8.5	90.5, 36.7, 83.7 90.5, 92.9	-2.41, -1.36, 1.14, -1.38, 2.32	8.0	2.1	Straight	No
15	Pigeon Point	1	3.4	0.0	-0.57	15.0	38.7	Inside bend	Yes
16	Battery Creek	1	7.6	82.5	2.61	6.0	6.0	Outside-straight	No
17	Port Royal Maritime Center	1	2.6	48.1	0.67	6.6	2.6	Straight	No
18	Waddell Mariculture Center	2	5.4, 13.4	56.1, 98.8	0.68, 3.64	18.3	7.3	Outside bend	No