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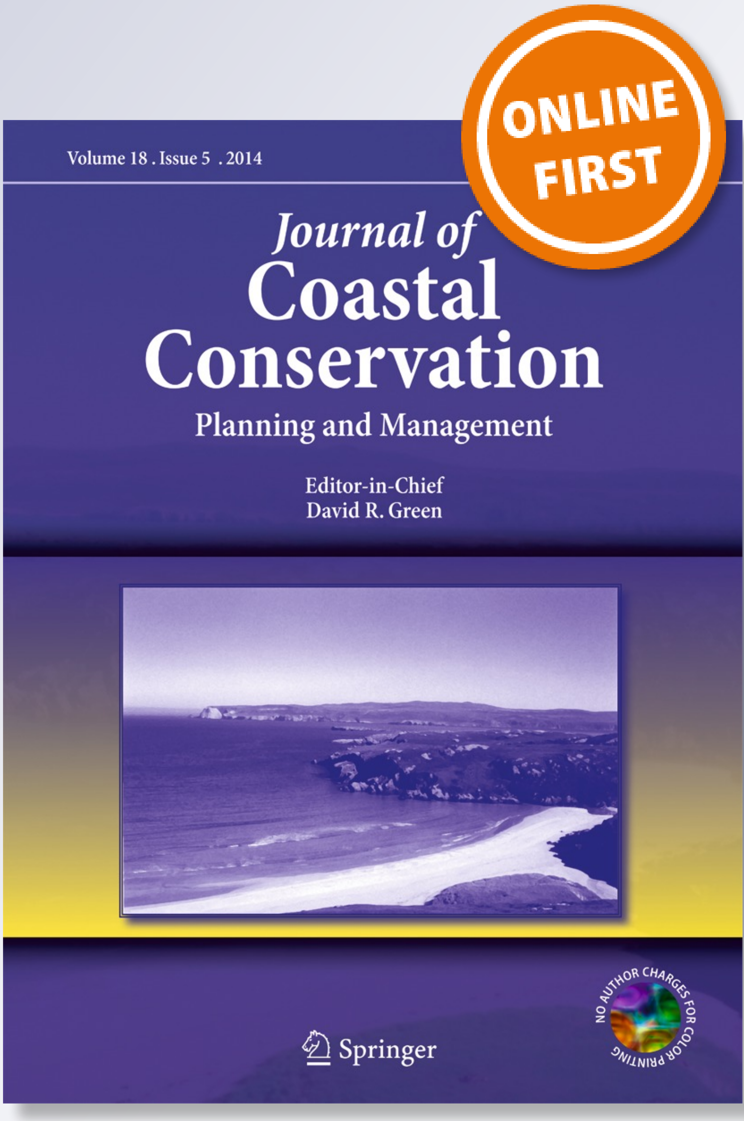

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Abstract With sea level rise, erosion, and human disturbances affecting coastal areas, strategies to protect and stabilize existing shorelines are needed. One popular solution to stabilize while conserving intertidal habitat is the use of "living shoreline" techniques which are designed to mimic natural shoreline communities by using native plants and animals. However, little information is available on the success of living shoreline stabilization. This project evaluated the wave energy attenuation associated with living shorelines that contained *Crassostrea virginica* (eastern oyster) and/or *Spartina alterniflora* (smooth cordgrass) in a wave tank. Four living shoreline techniques were assessed, including a control (sediment only), oysters alone, cordgrass alone, and a combination of oysters plus cordgrass. Time since deployment (newly deployed, one-year after deployment) was also assessed to see how wave energy attenuation changed with natural oyster recruitment and plant growth. Wave energy was calculated for each newly deployed and one-year old shoreline stabilization treatment using capacitance wave gauges and generated waves that were representative of boat wakes in Mosquito Lagoon, a shallow-water estuary in Florida. All one-year old treatments attenuated significantly more energy than newly-deployed treatments. The combination of one-year old *S. alterniflora* plus live *C. virginica* was the most effective as this treatment reduced 67 % of the wave energy created by a single recreational boat wake, compared to bare sediment. Natural resource managers and landowners facing shoreline erosion issues can use this information to create effective stabilization protocols that preserve shorelines while conserving native intertidal habitats.

Keywords Wave tank · Shoreline erosion · Soft stabilization · Spartina alterniflora · Crassostrea virginica · Wave attenuation

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Introduction

Coastal counties only occupy 17 % of the land area in the continental United States, yet these same counties contain 53 % of the nation's population (U.S. Census Bureau 2012). Shorelines are not only attractive areas for human development, they also provide habitat for multitudes of marine, terrestrial and estuarine species that require water-land interfaces for feeding, refuges, and nurseries (Beck et al. 2001; Boesch and Turner 1984; Herke 1971; Kneib 1997; Minello et al. 1994; Rakocinski et al. 1992). With sea level rise, erosion, and human disturbances all affecting coastal areas, resource managers and landowners are concerned about current and future shoreline stability (Klein et al. 2001; Yohe and Neumann 1997).

The Florida Department of Environmental Protection (2012) states that erosion currently affects 59 % of the state's coastline with 47 % being classified as "critically eroded." This means that environmental interests and human development landward of these areas are seriously threatened and require shoreline stabilization or beach nourishment to remain operational (Clark 2008; FDEP 2012). Erosion is a natural force caused by winds, waves and currents (Hayden 1975; Morton et al. 2004). However, the erosion observed on more than half of Florida's coastlines over the past 50 years is believed to be a combination of natural and anthropogenic

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sources (Clark 2008; El-Ashry 1971). Loss of local shoreline sediments can be partly attributed to the construction of waterfront buildings, the creation and maintenance of boating inlets, and recreational or commercial boating activities (Dean 1976; Dolan and Vincent 1972; Houser 2010; Komar 2000; López and Marcomini 2013; Pilkey 1991; Schoellhamer 1996).

Wakes produced from recreational and commercial boating have been shown to cause erosion on shorelines consisting of sand, silt and peat (Schroevers et al. 2011). Cargo ships can create large waves (≥70 cm in height), causing erosion of loose and consolidated materials up to 200 m away from the boating activity (Schroevers et al. 2011). The increased wave height due to boat wakes leads to larger orbital velocities, and shear stress along the seabed up to 2.5 N/m^2, thereby destabilizing benthic sediments (Schroevers et al. 2011). Studies have quantified the energy caused by boat wakes, and suggest they are more detrimental to shoreline stability than tidal flow and natural wind waves in areas with sandy shorelines (0.06 mm to 2 mm grain diameter) (Foda 1995; Foda et al. 1999; Limerinos and Smith 1975; Parnell et al. 2007; Wentworth 1922). This increased shear stress and energy associated with boat wakes ultimately causes sediment loss along shorelines (Bauer et al. 2002; Fredsøe and Deigaard 1992; Komar and Miller 1973; Soomere and Kask 2003). Boat wakes can also be detrimental to aquatic organisms and their habitats in systems with small fetches, which are normally exposed to low natural wave activity (Bourne 2000; Parnell et al. 2007; Soomere and Kask 2003). Keddy (1982, 1983) documented that high wave energy was correlated with low biodiversity and biomass of shoreline plants, including the marsh cordgrass *Spartina alterniflora*. Wave energy also played a role in subtidal seagrass survival, growth and dispersal (Fonseca and Bell 1998). In shallow bodies of water (<3 m deep) with recreational boating traffic, clay and silt shorelines can experience 0.01–0.22 mm of erosion per boat passage (Bauer et al. 2002). The wakes not only caused disturbances to the sediment, they also physically impacted intertidal oysters and shoreline vegetation (Bauer et al. 2002; Walters et al. 2002; Wall et al. 2005).

To combat the loss of local sediments from boating and natural occurrences, many shoreline stabilization techniques have been employed. Techniques including seawalls, breakwaters, beach nourishment, and "living shorelines" are commonly employed to counteract both natural and anthropogenic erosion (Castellan and Wall 2006). Seawalls, groins, jetties, rip-rap, and breakwaters are classified as hard structural stabilization, and involve armoring the shoreline with cement, rock or wooden structures (Hillyer et al. 1997). These types of shoreline hardening can be useful in slowing and stopping upland erosion, but this is often at the cost of losing beach areas and intertidal habitat in front and to the sides of the structures (Bozek and Burdick 2005; Kraus and McDougal 1996; Pilkey and Wright 1988). Soft, non-structural stabilizations, often called living shorelines, use only plants, animals, shell, and organic materials to armor shorelines. These organisms and organic materials absorb wave energy and bind shoreline sediments with below-ground biomass from vegetation (Berman et al. 2007; Broome et al. 1992; Currin et al. 2010). By creating a natural progression of flora and fauna from the subtidal to the supratidal, waves created by wind, storm events, and boats should be attenuated and erosion limited (Charlier et al. 2005; Knutson et al. 1982; Leonard and Croft 2006; Morgan et al. 2009).

Two species commonly used in living shoreline stabilization projects along the western Atlantic coastline of the US are the native eastern oyster *Crassostrea virginica* and the native smooth cordgrass *Spartina alterniflora* (Castellan and Wall 2006; Charlier et al. 2005; Meyer et al. 1997). *Crassostrea virginica* is a filter-feeding bivalve that forms three-dimensional, fringing reefs in the low to middle intertidal zone, as well as patch reefs and expansive subtidal reefs (Dame 2011). This species of oyster is gregarious, with free-swimming larvae that locate adult oyster shell for settlement through chemical cues (Tamburri et al. 1992). The native range for this bivalve stretches from the St. Lawrence River in Canada to the Yucatan Peninsula, covering the entire U.S. eastern seaboard and Gulf of Mexico (Buroker 1983). *Spartina alterniflora* is a perennial marsh grass that typically grows between 0.3 and 3.0 m in height (Adams 1963). This grass uses underground rhizomes to create monospecific stands along sandy, middle and upper-intertidal shorelines (Gleason et al. 1979). Stands of *S. alterniflora* can be found along the western Atlantic seaboard plus the Gulf of Mexico, from Newfoundland to Texas, and can reach densities over 108 stems/m^2 with optimal growing conditions (Gleason et al. 1979).

Oysters and marshgrasses deployed for living shoreline stabilization should significantly slow erosional processes on coastal habitats compared to bare shorelines by attenuating wave energy and binding sediments. To evaluate the ability of *C. virginica* and *S. alterniflora* to attenuate recreational boat wakes, a wave tank and wave gauges were used to calculate and compare changes in wave energies associated with different treatment densities and with time since treatment deployment. Previous studies have used wave tanks to investigate the roles of waves in sediment transport, swell decay, and wave attenuation through plants such as kelp and seagrass (Fonseca and Cahalan 1992; Hasselmann et al. 1973; Messaros and Bruno 2010). In this study, wave attenuation was calculated through newly planted living shoreline materials, and shoreline materials that matured and recruited organisms for 1 year. Key questions pursued were: 1) How much wave energy was attenuated by vertical oyster shells and marshgrass immediately after deployment? 2) How much does wave attenuation increase after 1 year, once live oysters have recruited to the shells and marshgrass has produced new above-ground and below-ground biomass?

Methods

Study area

The motivation for this wave tank study comes from documented shoreline erosion in Mosquito Lagoon, which is a shallow water, microtidal estuary (average depth less than 1.7 m) on the east coast of Central Florida (Grizzle 1990; Walters et al. 2001). Approximately 230 km^2 of northern Mosquito Lagoon is managed by Canaveral National Seashore (CANA). Although the most influential water movement in the majority of this microtidal system is a direct result of wind-driven currents, some locations are more influenced by tidal currents (Dubbleday 1975; Hansell 2012; Smith 1987). Mosquito Lagoon is also a world-renowned fishing area. As such, it experiences intense recreational boating pressures (Johnson and Funicelli 1991; Scheidt and Garreau 2007). Interpretations of aerial photos in the lagoon have shown that a high frequency of recreational boating traffic along popular boating channels was positively correlated with loss of live oyster reefs in CANA (Garvis 2012; Grizzle et al. 2002).

The natural assemblage of flora and fauna on intact shorelines in Mosquito Lagoon includes *C. virginica* in the low to middle intertidal zone, and *S. alterniflora* in the middle to upper intertidal zone. Landward of the upper intertidal zone, we find four different types of mangroves including red (*Rhizophora mangle*), black (*Avicennia germinans*), white (*Laguncularia racemosa*), and buttonwood (*Conocarpus erectus*).

Shoreline stabilization methods used in CANA

To provide stabilized oyster shell as substrate for oyster recruitment in the lower intertidal zone, oyster restoration mats were used. These mats consisted of 0.25 m^2 square of Vexar™ (mesh diameter: 1.5 cm) to which 36 adult oyster shells with a drill hole near the umbo were attached with 50 lb. test cable ties. Shells were attached to the mats in a vertical orientation to mimic natural intertidal oyster reef formation (Stiner and Walters 2008). The oyster shells were collected from local shucking facilities and quarantined on land for a minimum of 6 months to avoid the transfer of diseases and invasive species (Bushek et al. 2004; Cohen and Zabin 2009). Once constructed, the mats were placed on top of the sediment in the low to middle intertidal zone, where natural larval recruitment was expected to occur on deployed shells. In 1.5–5.5 years, over 300 live oysters can be counted on one individual mat, at a mean density of 472 live oysters per m^2 after 5.5 years (Walters et al. 2013). To create stands of *S. alterniflora*, individual shoots with attached rhizomes from shorelines within CANA were collected and potted for four to 6 months prior to deployment. The plants were then transplanted to the middle to upper intertidal zone, and grew into monospecific stands. To test the effectiveness of this type of shoreline stabilization, we brought oyster mats and cordgrass transplants from CANA to a wave tank to quantify wave energy attenuation.

Wave tank experiment

An indoor wave tank at Florida Institute of Technology in Melbourne, FL was used for our manipulative trials to quantify wave attenuation through living shorelines. The tank measured 9.08 m in length, 0.57 m in width, and 0.91 m in depth, and generated waves using a 0.91 m flap paddle located 0.6 m from the back wall of the tank. The tank was constructed of 5 cm clear acrylic, supported with metal beams at 1.22 m intervals along the length of the tank. This particular tank has been used in biological and engineering studies since 1990 (e.g. Lohmann et al. 1990).

Three capacitance wave gauges were used with Ocean Sensor Systems Incorporated (OSSI) V3_1 software to measure free-surface displacements within the tank. Displacement was recorded 2.5 m from the paddle at a water depth of 0.30 m (well-developed wave), 4 m from the paddle at a depth of 0.22 m (before shoreline treatment), and 5.5 m from the paddle at a depth of 0.13 m (after shoreline treatment) (Fig. 1). These displacements were then converted to wave heights using the statistical zero-crossing method (Arhan et al. 1979). All testing within the tank was designed to maintain a scale ratio of 1:1. This allowed the wave tank model to match the model field dimensions (Hughes 1993). Changes in water level associated with tides and attenuation caused by shoaling, were not tested in the wave tank.

The sediment shoreline was constructed in the wave tank at the tank end opposite from the paddle to mimic a non-eroded shoreline in CANA. A slope of 15:1 was selected to represent the natural bathymetry of Mosquito Lagoon after vertical shoreline profiles were completed every 10 cm from the supratidal to subtidal at 10 representative, intertidal, CANA shorelines (15:1 m±SE 2.6 m). To construct the benthic substrate profile, 719 kg (0.389 m^3) of sediment was excavated from shorelines of Mosquito Lagoon, transferred to the wave tank, and graded to the specifications described in Fig. 1. This sediment (mean density: 1.85±SE 0.4 g cm^{-3}) was placed on the bottom of the tank and reached the still water line (SWL) at 0.3 m creating a 15:1 slope (Fig. 1). All sediments were returned to shoreline donor sites post-experiment.

To determine desired wave heights for our tank trials, boat wake surveys were completed in Mosquito Lagoon. Over 3 months, forty-five separate thirty-minute surveys on eroded

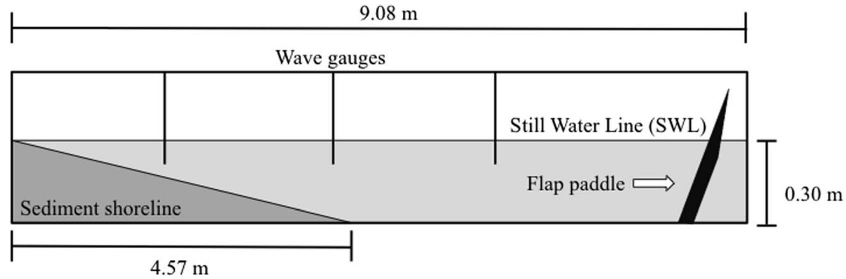

Fig. 1 Wave tank setup with flap paddle, 15:1 sloped sediment shoreline, and capacitance wave gauges

shorelines were used to determine an average wave height, period, and number of waves in a wave train generated by individual recreational boat passes. On average, individual boat wave trains consisted of 10 waves (±0.7 SE) with a mean wave height of 12.7 cm (±2.0 SE) and a period of 1.8 s. To mimic a boat pass in the tank, 10 waves that were 12.7 cm high were used. Because the tank could produce a 12.7 cm wave, the physical settings in the field were closely mimicked throughout this experiment. The water within the tank was allowed to settle between each boat wake wave train to avoid errors associated with residual water movement, and the sediment shoreline was reshaped to the appropriate slope after each trial.

For each trial there were ten boat passes (10 waves per pass) per treatment, and three trials overall (300 waves per treatment). Four treatments and two time variables were tested in the wave tank. The treatments consisted of: 1) control with sediment only (Fig. 2a), 2) cordgrass at depths from 0.17 to 0.13 m below the SWL (Fig. 2b), 3) stabilized oyster shell at depths from 0.26 to 0.22 m below the SWL (Fig. 2c), and 4) combination of oyster shell and cordgrass at these depths (Fig. 2d). The depth for each species in wave tank correlated with intertidal depths found in Mosquito Lagoon.

Time was the other variable considered, with new restoration compared to one-year old, established restoration. New restoration included new oyster mats with clean, stabilized shell and newly-planted cordgrass. This consisted of two new oyster mats with a combined total of 72 disarticulated oyster shells (mean shell height±S.E.: 6.9±1.5 cm; weight of individual oyster mat: 1.95±0.08 kg; water displacement of individual oyster mat: 0.0017±0.0002 m^3) in a 0.5 m^2 area (1 m long x 0.5 m wide), and five individual cordgrass shoots (individual mean height±S.E.: 85.0±8.0 cm; individual mean blotted dry, above-ground biomass: 3.17±0.69 g; individual water displacement: 7.4±0.90 ml) in a 0.25 m^2 area (0.5 m long x 0.5 m wide) (Fig. 2). The one-year old established restoration included oyster shell that had recruited live oysters for 1 year, and cordgrass in densities equivalent to one-year post planting. The two oyster mats combined had a mean (±SE) of 158±6.2 live oysters attached to the original 72 disarticulated shells in a 0.5 m^2 area (mean shell height± S.E.: 15.0±1.6 cm; mean mat weight: 4.00±0.26 kg; mean

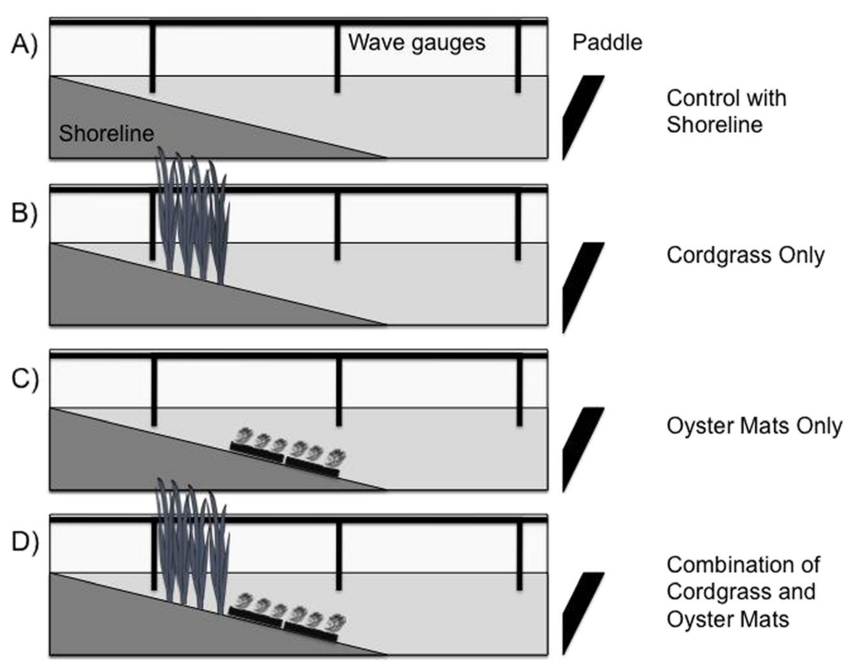

Fig. 2 Wave tank setup with newly-deployed living shoreline treatments displaying relative placement of cordgrass (5 shoots) and oyster mats (two mats, 72 total shells) on the shoreline

mat water displacement: 0.0026±0.0005 m^3). The year-old cordgrass treatment consisted of 37 individual *S. alterniflora* shoots in a 0.25 m^2 area (individual mean height: 89±10 cm; individual mean blotted-dry, above-ground biomass: 28.55±2.16 g; individual water displacement: 28.55±2.17 ml) (Fig. 3).

Spartina alterniflora was collected from three populations within Mosquito Lagoon, and oyster mats with live adult oysters were obtained from one Mosquito Lagoon restoration site. Cordgrass and oyster mats were replaced between trials to retain independence of replicates, but their physical characteristics remained as similar as possible.

Simulated wave height (m) and energy loss (J m^{-2}) over newly-deployed, 1-year old and control living shorelines were evaluated inside the wave tank using

$$E = \frac{1}{8}\rho g H^2, \quad (1)$$

where E is wave energy based on linear wave theory, ρ is fluid density, g is the acceleration of gravity, and wave height is represented by H. Fresh water was used within the wave tank (ρ=1000 k m^{-3}) instead of the brackish water (ρ=1025 k m^{-3}) found in Mosquito Lagoon. This was due to the cost of salt, potential corrosion issues within the tank, and gauge effectiveness. Ultimately, a relative comparison was made that eliminated the role of density (Eq. 2). After calculating wave energy with both fresh and brackish water densities, there was no difference in the final percent change of wave energy through the treatments. The wave height for each treatment was calculated, and a separate number for each wave gauge was produced (well-developed wave, before stabilization, after stabilization). The heights of waves 4–8 of each wave train were used to calculate average height, and then wave energy using Eq. 1. Taller wave heights equated to higher wave energy, and lower wave heights equated to lower wave energy.

An attenuation coefficient, K_t, also was calculated for all treatments using

$$K_t = H_{\mathrm{transmitted}}/H_{\mathrm{input}}, \quad (2)$$

where $H_{\mathrm{transmitted}}$ was the attenuated wave height after progressing through the treatment (m), and H_{input} was the incoming wave height (m). A coefficient of 1 would signify no energy attenuation, while a coefficient of 0 would mean all of the energy was absorbed (Möller 2006). Therefore, a lower coefficient represents a more effective living shoreline treatment (Möller 2006).

Statistical analyses

A block ANOVA was used to determine differences in average wave heights among treatments (R Development Core Team 2011). Three independent blocks were run on separate days, with each block containing one replicate of each shoreline treatment (4 treatments×2 times) in random order. Prior to analysis, all data were tested for normality using a Shapiro-Wilk test, and a Levene's test for equality of variances. Post-hoc Tukey's HSD tests were used for pairwise comparisons among shoreline treatment types when overall ANOVA values were p<0.05.

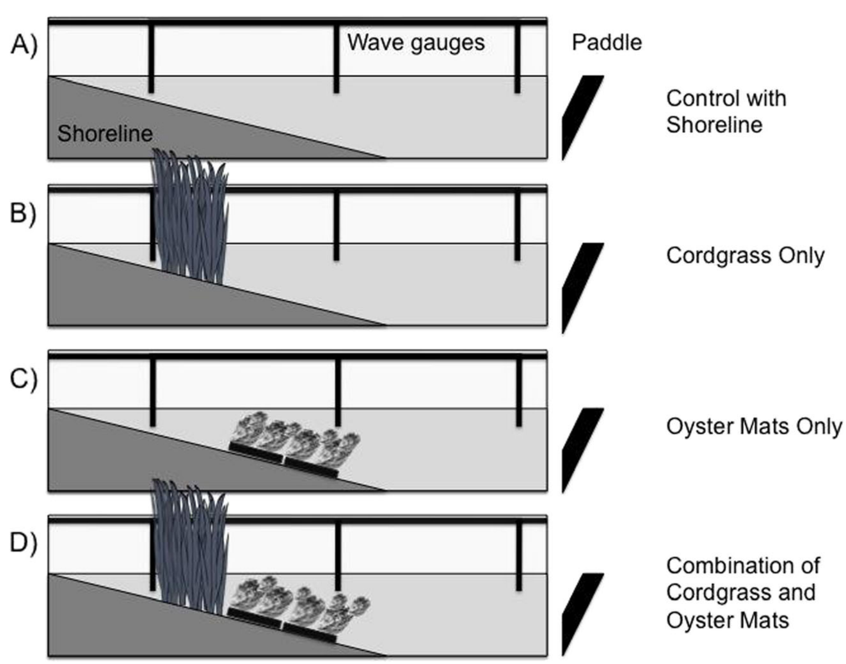

Fig. 3 Wave tank setup with one-year old established living shoreline treatments displaying relative placement of cordgrass (37 shoots) and oyster mats (two mats, 72 shells plus recruited live oysters) on the shoreline

Results

Wave height and energy

Because blocks were not significantly different from one another (ANOVA: $p=0.730$), a two-way ANOVA was used to analyze differences among treatments and time-since-deployment (control, new deployment, 1-year old deployment). Both time-since-deployment and treatments were found to be significantly different (ANOVA: $p<0.001$) with a significant interaction effect (ANOVA: $p<0.001$).

Mean wave height was calculated after each wave train progressed through the living shoreline treatment. The mean change in wave heights for controls was significantly lower than for all other tested treatments (Tukey's post-hoc test; Fig. 4). Excluding the controls, all one-year old established treatments reduced wave height significantly more than newly deployed treatments. Also, the combination of established live oysters and cordgrass showed the highest mean (±S.E.) wave height reduction at 5.52±0.31 cm (Fig. 4). This equates to a 67.3 % decrease in total wave energy (Table 1). Established live oysters alone had the second largest reduction in wave energy (44.7 %), with a mean wave height reduction of 3.36±0.23 cm. Excluding the controls, the lowest wave height reduction (0.51±0.20 cm) was associated with newly deployed cordgrass. This equated to a 6.9 % reduction in wave energy (Table 1).

The total energy contained in one wave created by a boat pass and one entire wave train was calculated using Eq. 1 (Table 1). One wave traveling over the control treatment retained the most energy (mean ± S.E.) at 19.55±3.80 J m^{-2}. The lowest mean wave energy occurred after waves progressing through the one-year old combination treatment (6.32±2.46 J m^{-2}). Boat wakes with 10 waves per train were compared by multiplying the energy by 10. A bare shoreline was impacted by 195.50 J m^{-2} for each boat pass, while a shore with a one-year old combination with cordgrass and live oysters was only impacted by 63.19 J m^{-2} (Table 1).

Attenuation coefficient

We determined an attenuation coefficient for each living shoreline treatment by comparing the initial wave height (12.7 cm) to the final attenuated wave heights (Table 2). Given that a coefficient of 1 indicates no energy dissipated, the control treatment (attenuation coefficient=0.99) reduced almost none of the initial wave height (Möller 2006). Three more treatments, newly deployed cordgrass, stabilized oyster shell, and the combination of newly deployed cordgrass and stabilized oyster shell, also had coefficient values of 0.90 or greater (Table 2). These values contrasted sharply to the established combination treatment (oysters plus cordgrass), which had a coefficient of 0.57, meaning that nearly half of the wave energy from the boat wake was attenuated through this treatment. The next lowest coefficients were from the one-year old live oysters at 0.74, and the one-year old established cordgrass at 0.82 (Table 2).

Discussion

Our findings suggest that living shorelines composed of intertidal oysters and cordgrass attenuate a significant amount of wave energy produced by boat wakes. 1 year post-deployment, our living shoreline stabilization attenuated 67 % of the

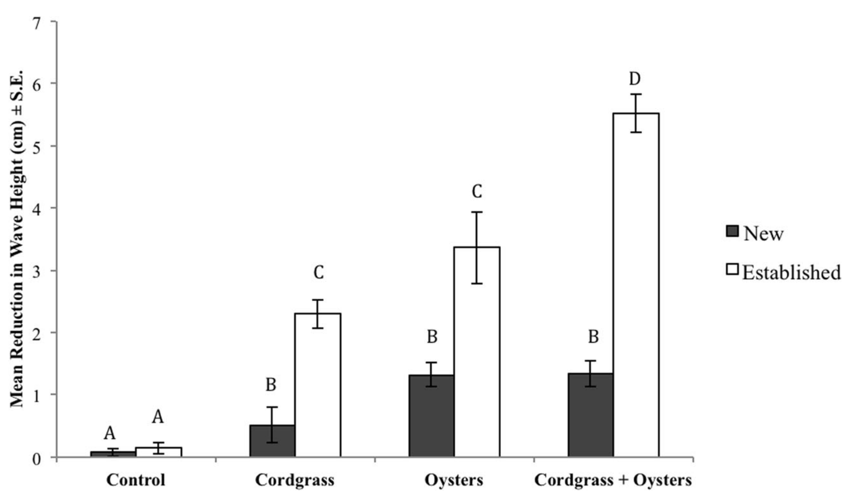

Fig. 4 Mean change in wave height after encountering shoreline stabilization treatment (±SE). Wave heights were compared using a two-way ANOVA. New=new restoration with new, clean, stabilized oyster shells and cordgrass plugs at planting densities used in Mosquito Lagoon. Established=one-year old restoration that included oyster shell that had recruited live oysters for 1 year, and cordgrass densities equivalent to one-year post planting. Treatments with different letters are significantly different at $p<0.05$ level as determined by Tukey's post hoc tests

Table 1 Energy of one wave impacting a shoreline, percent of energy reduction after encountering shoreline stabilization treatment, and energy of one wave train (10 waves) impacting a shoreline

	Energy (J m^{-2}) Impacting Shoreline - One Wave	Wave Energy Reduction After Treatment	Energy (J m^{-2}) Impacting Shoreline - Wave Train
Control	19.55	1.0 %	195.5
Newly planted cordgrass	18.21	6.9 %	182.1
Newly deployed oyster shell	15.9	18.7 %	159.0
Newly deployed cordgrass and oyster shell	15.85	19.0 %	158.5
1-year established cordgrass	13.26	31.4 %	132.6
1-year established oyster	10.69	44.7 %	106.9
1-year established cordgrass and oyster	6.32	67.3 %	63.2

boat wave energy. Also, given that some wave energy was diminished immediately after deployment, this reduction could reasonably be expected to continue to increase well beyond 1 year.

As waves progress over an intertidal oyster reef, energy is dissipated by the three-dimensional structure, thereby protecting the landward shoreline (Coen et al. 1999). Oysters and oyster shell have been used in many forms of wave attenuation and shoreline stabilization, from breakwaters, to oyster-infused reef balls, to oyster restoration mats (Brumbaugh and Coen 2009; Scyphers et al. 2011; Walters et al. 2012). Through wave attenuation, intertidal oyster reefs and breakwaters have been found to reduce erosion and shoreline loss by 40 % (Scyphers et al. 2011). The primary differences between the two tested times (newly-deployed vs. one-year old) with our living shoreline treatments were vertical relief created by the oysters and oyster density. As oyster larvae recruit to oyster shells, they create a shell of calcium carbonate that adds to the overall vertical height of the existing reef (Korringa 1952). This overall reef height was evident when comparing mean (±S.E.) heights of newly-deployed individual oyster shells (6.9±1.5 cm) to shells that were glued together and taller as clusters following the recruitment of live oysters for 1 year in the field (15.0±1.6 cm). Not only did height increase over a year of growth, but significantly more water was displaced by recruited oysters when compared to newly-deployed shells. The established oysters attenuated more than twice as much wave energy as newly deployed shells (Table 1). This was attributed to the average 8.1 cm of extra vertical height that newly recruited oysters added to the reef and a 130 % mean increase in oyster density.

Table 2 Wave attenuation coefficients of treatments tested in wave tank from an input (incident) wave height of 12.7 cm

	Mean Transmitted Wave Height (cm)	Wave Attenuation Coefficient
Control	12.60	0.99
Newly planted cordgrass	12.19	0.96
Newly deployed oyster shell	11.39	0.90
Newly deployed cordgrass and oyster shell	11.37	0.90
1-year established cordgrass	10.40	0.82
1-year established oyster	9.34	0.74
1-year established cordgrass and oyster shell	7.18	0.57

Spartina alterniflora is commonly used in association with oysters in living shoreline stabilization projects. This is due to: 1) its typical position along the intertidal zone of the shoreline profile, 2) its range from Newfoundland to central Florida, and 3) its ability to grow in salinities from almost fresh to ocean water (Bush and Houck 2008). Wave energy is attenuated by this cordgrass as waves pass through the aboveground biomass (Anderson et al. 2011; Dalrymple et al. 1984). As cordgrass density increased, more energy was attenuated. A comparison on the wave attenuation of newly-deployed cordgrass (5 individual shoots in a 0.25 m^2 area) to the year-old established cordgrass (37 individual shoots in a 0.25 m^2 area) shows that the higher density attenuated 4.5 times more energy (Table 1). Blotted-dry wet weights and mean water displacement (±S.E.) of year-old potted individuals of *S. alterniflora* were, likewise, 9 times higher than new plants. An analogous study conducted on *S. alterniflora* in a 30.5 m flume tank documented similar results, with high densities (49 shoots in a 0.25 m^2 area) of *S. alterniflora* attenuating up to 16 % more wave energy than low densities (24 shoots in a 0.25 m^2 area) (Augustin et al. 2009).

Newly deployed shorelines did not show additive effects of oyster shells plus cordgrass. The attenuation effect was minimal (6.9 % energy reduction) when the newly planted cordgrass consisted of only five plants. It appeared that the stabilized oyster shell caused most of the wave attenuation (18.7 % energy reduction) that was seen in the combination treatment (19.0 % energy reduction). When observing the one-year old combination, the attenuation was additive with both the oyster shell and the cordgrass contributing to the total energy reduction. The interaction effect of treatment and age caused the one-year old combination treatment (live oysters plus cordgrass) to significantly reduce wave energy more than any other treatment ($p<0.001$; Table 1).

In order to compare our living shorelines composed of intertidal oysters and cordgrass to other stabilization techniques, an attenuation coefficient (K_t) was assigned to each treatment. The combination of cordgrass and oyster shell in the newly deployed treatment had a coefficient of 0.90, while the 1-year old established combination produced a coefficient of 0.57 (Table 2). By assigning a standardized value to this stabilization, it can be compared to other forms of shoreline protection. Although the attenuation coefficient is not commonly used in restoration literature, it could theoretically be a standard descriptive unit allowing natural resource managers and home owners to choose a stabilization method that best suits their needs. The attenuation coefficient reflects a direct comparison between incident (input) and transmitted wave heights past or over an obstacle (Eq. 2). Therefore, if the transmitted wave height is smaller than the input, due to the presence of a living shoreline attenuating wave energy, it is likely that potential sediment transport (erosion) may be reduced resulting in a more stabilized shoreline.

An important aspect not evaluated in this experiment is the change in wave energy hitting the shoreline at different water levels. Previous studies have shown that energy attenuation in non-microtidal areas is dependent on the depth of the water (Duncan 1964; Ellis et al. 2002; Lugo-Fernandez et al. 1998; Roberts and Suhayda 1983). We chose to replicate in the wave tank a water level typically associated with the summer season when there is more boat activity. In Mosquito Lagoon, there is a "high water season" (mean: 27 cm higher) that annually runs from August to December that could temporarily decrease the ability of a living shoreline to attenuate boat wakes (Hall et al. 2001; Provost 1973; Smith 1986, 1993). However, the temporary subtidal experience for oysters and cordgrass may encourage vertical growth in both species. Future experiments should evaluate how the attenuation coefficient changes over seasons and years with varying water levels, along with how sediment transport is affected by wave energy impacting a shoreline.

A positive aspect of using native flora and fauna as a form of soft stabilization is that it can be significantly less expensive than hard armoring (Grabowski et al. 2012; Swann 2008). Grabowski et al. (2012) demonstrated that a bulkhead or similar rock revetment cost between \$630 and \$752 per linear meter. In comparison, a living shoreline consisting of marshgrass and oyster shell can cost as little as \$150 per linear meter (Davis and Luscher 2008). Having an average life span of 8–10 years, seawalls and other types of hard armoring require maintenance to remain effective over time (Griggs and Fulton-Bennett 1988). A living shoreline stabilization that includes *C. virginica* and *S. alterniflora* can be potentially self-sustaining if there is sufficient recruitment and survival of oyster spat, and if there are no major disturbances to the plants (Meyer et al. 1997; Piazza et al. 2005).

In addition to being substantially less expensive to create than a bulkhead or seawall, living shorelines maintain important ecosystem services such as providing structural habitat, water filtration, nitrogen removal, and enhanced foraging grounds for economically important fisheries (Coen et al. 1999; Harding and Mann 2001; Jones et al. 1994). Hard-armoring also does not provide the habitat complexity that many species need to live and reproduce (Beck et al. 2001; Bilkovic and Roggero 2008; Heck et al. 2003). The three-dimensional structure provided by oyster reefs and marshgrass allows for higher biodiversity (Beck et al. 2001; Bilkovic and Roggero 2008; Heck et al. 2003). Intertidal oyster reefs within Mosquito Lagoon provide habitat for over 140 species of flora and fauna due to this structural complexity, and increase in the surface area of suitable habitat (Barber et al. 2010; Manley et al. 2009). Therefore, living shorelines can both limit shoreline erosion by attenuating wave energy, while at the same time maintain vital shoreline habitat complexity.

When discussing shoreline stabilization, it is also important to recognize sea level change is taking place, and consider how change will impact future erosional rates. The presence of flora and fauna on a shoreline allows sediment and above-ground organic materials, such as fallen leaves, to be trapped on the shore promoting shoreline accretion (Gleason et al. 1979; Redfield 1965, 1972; Yang 1998). Historically, this process allowed salt marsh shorelines to maintain elevation equilibrium, even in the face of sea level rise and land subsidence (Morris et al. 2002; Redfield 1965, 1972). Studies have shown that shoreline areas containing native vegetation, such as mangroves and marshgrasses, have higher accretion rates than bare shorelines under the same erosion pressures (Cahoon and Lynch 1997; Gleason et al. 1979; Kumara et al. 2010; Morris et al. 2002). In some instances, complete land submergence would be inevitable without the addition of organic material from shoreline vegetation (McKee et al. 2007). Some vegetation, such as *S. alterniflora* on intertidal shorelines, has been shown to elevate shorelines by producing underground biomass, and by disrupting the water movement to allow for sediment deposition (Cahoon et al. 2004; Gleason et al. 1979; Morris et al. 2002; Yang 1998). Thus, stabilization in the form of living shorelines could potentially help compensate for sea level rise by accumulating mineral sediment and organic matter, allowing plants and animals to maintain their relative position in the intertidal zone as water levels increase (Orson et al. 1985; Reed 1995).

Conclusions

As coastal development continues, and natural shorelines are impacted by other anthropogenic factors such as recreational boat wakes, shoreline reinforcement is likely to continue to occur. Our findings document that living shorelines composed of intertidal oysters and cordgrass can attenuate a significant amount of wave energy produced by boat wakes. One-year post-deployment, our living shoreline stabilization attenuated 67 % of the boat wave energy and some attenuation began immediately post-deployment. We recommend that shallow-water estuaries, such as Mosquito Lagoon, should focus on living shorelines for stabilization. Natural resource managers and landowners facing shoreline erosion issues can use this information to create effective stabilization protocols that preserve shorelines while conserving native intertidal habitats.

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