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Assessing The Effectiveness Of Living Shoreline Restoration And Quantifying Wave Attenuation In Mosquito Lagoon, Florida

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ASSESSING THE EFFECTIVENESS OF LIVING SHORELINE RESTORATION AND QUANTIFYING WAVE ATTENUATION IN MOSQUITO LAGOON, FLORIDA

by

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ABSTRACT

Coastal counties make up only 17% of the land area in the continental United States, yet 53% of the nation's population resides in these locations. With sea level rise, erosion, and human disturbances all effecting coastal areas, researchers are working to find strategies to protect and stabilize current and future shorelines. In order to maintain shoreline stability while maintaining intertidal habitat, multipurpose living shorelines have been developed to mimic natural shoreline assemblages while preventing erosion. This project determined the effectiveness of a living shoreline stabilization containing *Crassostrea virginica* (eastern oyster) and *Spartina alterniflora* (smooth cordgrass) in the field and through controlled wave tank experiments. First, fringing oyster reefs constructed of stabilized oyster shell and smooth cordgrass plugs were placed along three eroding shoreline areas (shell middens) within Canaveral National Seashore (CANA), New Smyrna Beach, FL. For each shell midden site, four treatments (bare shoreline control, oyster shell only, *S. alterniflora* only, and oyster shell + *S. alterniflora*) were tested in replicate 3.5 x 3.5 meter areas in the lower and middle intertidal zones. Each treatment was replicated five times at each site; erosion stakes within each replicate allowed measurement of changes in sedimentation. After one year in the field, the living shoreline treatments that contained oyster shells (oyster shell only and oyster shell + *S. alterniflora*) vertically accreted on average 4.9 cm of sediment at two of the sites, and an average of 2.9 cm of sediment at the third, while the controls lost an average of 0.5 cm of sediment. *S. alterniflora* did not significantly contribute to the accretion at any site due to seagrass wrack covering and killing plants within one month of deployment.

Next, the reduction in wave energy caused by these living shoreline stabilization techniques relative to bare sediment (control) was quantified. The energy reduction immediately after deployment, and the change in energy reduction when *S. alterniflora* had been allowed to grow for one year, and the stabilized shell was able to recruit oysters for one year was tested. Laboratory experiments were conducted in a nine-meter long wave tank using capacitance wave gauges to ultimately measure changes in wave height before and after treatments. Wave energy was calculated for each newly deployed and one-year old shoreline stabilization treatment. Boat wake characteristics from CANA shorelines were measured in the field and used as inputs to drive the physical modeling. Likewise, in the wave tank, the topography adjacent to the shell midden sites was measured and replicated. Oyster shell plus *S. alterniflora* attenuated significantly more wave energy than either the shells or plants alone. Also, one-year old treatments attenuated significantly more energy than the newly deployed treatments. The combination of one-year old *S. alterniflora* plus live oysters reduced 67% of the wave energy.

With the information gathered from both the field and wave experiments, CANA chose to utilize living shorelines to stabilize three shell middens within the park. Oyster shell, marsh grass and two types of mangroves (*Rhizophora mangle, Avicennia germinans*) were deployed on the intertidal zones of the eroding middens. Significant accretion occurred at all middens. Two sites (Castle Windy and Garver Island) vertically accreted an average 2.3 cm of sediment after nine months, and six months respectively, and the other site (Hong Kong) received on average 1.6 cm of sediment after six months. All control areas (no stabilization) experienced sediment loss, with erosion up to 5.01 cm at Hong Kong. Plant survival was low (< 20%) at Castle Windy and Garver Island, while Hong Kong had moderate survival (48-65%). Of the surviving marsh grass and mangroves on the three sites, almost all (> 85%) had documented growth in the form of increased height or the production on new shoots. Landowners facing shoreline erosion issues, including park managers at CANA, can use this information in the future to create effective shoreline stabilization protocols. Even though the techniques will vary from location to location, the overall goal of wave attenuation while maintaining shoreline habitat remains. As the research associated with the effectiveness of living shorelines increases, we hope to see more landowners and land managers utilize this form of soft stabilization to armor shorelines.

To Dr. Justine, Dr. Peter, Diana and Laura Manis,

I love you, and couldn't have done this without your support.

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CHAPTER 1: GENERAL INTRODUCTION

Coastal counties make up only 17% of the land area in the continental United States, yet 53% of the nation's population resides in these locations (U.S. Census Bureau 2012). Shorelines not only are attractive areas for human development, but also provide habitat for multitudes of marine, terrestrial and estuarine species that require water-land interfaces for feeding, refuge, and nurseries (Herke 1971; Boesch & Turner 1984; Rakocinski et al. 1992; Minello et al. 1994; Kneib 1997; Beck et al. 2001). With sea level rise, erosion, and human disturbances effecting coastal areas, researchers and landowners are concerned about current and future shoreline stability (Yohe & Neumann 1997; Klein et al. 2001). In Florida, the Department of Environmental Protection (2012) states that erosion currently affects 780 km, or 59% of the state's marine coastline; of this total, 47% is classified as "critically eroded" (FDEP 2012). This means that environmental interests and human development landward of these areas are "seriously threatened" (Clark 2008; FDEP 2012). Some erosion is natural, which is caused by wind, waves and currents (Hayden 1975; Morton et al. 2004). However, the unnaturally high erosion rates identified on more than half of Florida's coastlines over the past 50 years are believed to be anthropogenic in origin (T El-Ashry 1971; Clark 2008). The loss of local shoreline sediments can in part be attributed to the construction of waterfront buildings, the creation and maintenance of boating inlets, and recreational or commercial boating activities (Dolan & Vincent 1972; Dean 1976; Pilkey 1991; Schoellhamer 1996; Komar 2000; Houser 2010; López & Marcomini 2012).

To combat shoreline erosion and to protect landward structures from both natural and anthropogenic occurrences, many stabilization practices have been developed. Techniques including seawalls, groins, jetties, breakwaters, beach nourishment, and "living shorelines" are commonly used to counteract local sediment loss and to abate upland erosion (Castellan & Wall 2006). Seawalls, groins, jetties and breakwaters are classified as "hard, structural stabilization," and involve armoring the shoreline with cement, rock or wood structures (Hillyer et al. 1997). These types of shoreline hardening can be useful in slowing and stopping upland erosion, but often at the cost of losing beach areas and intertidal habitat in front and to the sides of the structures (Pilkey & Wright 1988; Kraus & McDougal 1996; Bozek & Burdick 2005). Also, these types of structures can be costly to install, anywhere from $100 to $3,000 per linear foot, and even more costly to maintain and repair (Charlier et al. 2005).

On the other end of the spectrum, "soft, non-structural stabilization" uses only living plants, animals and organic materials (i.e. bivalve shells) to attenuate wave energy and secure shoreline sediments (Currin et al. 2010). By creating a natural progression of animals and plants from the subtidal to the supratidal, waves created from wind, storm events, and boats should be attenuated and erosion limited (Knutson et al. 1982; Charlier et al. 2005; Leonard & Croft 2006; Morgan et al. 2009). While this type of restoration might not be suitable for high wave energy environments (coasts directly impacted by ocean waves), it is appropriate for many estuarine locations and can be much less expensive than hard, structural stabilizations over time (Petersen et al. 2002; Charlier et al. 2005; Grabowski et al. 2012).

Soft stabilization not only costs less, it also has the possibility to adapt to sea level change (Reed 1990). The presence of flora and fauna on a shoreline allows sediment and aboveground organic materials, such as fallen leaves, to be trapped on the shore promoting shoreline accretion (Redfield 1965, 1972; Gleason et al. 1979; Yang 1998). This historically has allowed salt marsh shorelines to maintain elevation equilibrium, even in the face of sea level rise and land subsidence (Redfield 1965,1972; Morris et al. 2002). Studies have shown that shoreline areas containing native vegetation, such as mangroves and marsh grasses, have higher accretion rates than bare shorelines under the same erosion pressures (Gleason et al. 1979; Cahoon and Lynch 1997; Morris et al. 2002; Kumara et al. 2010). In some instances, complete land submergence would be inevitable without the addition of organic material from shoreline vegetation (McKee et al. 2007). Thus, using living shorelines and other soft stabilization techniques over hard armoring could potentially ameliorate short-term sea level rise (Reed 1990) (Figure 1).

Beach nourishment is a unique type of soft stabilization commonly used in highly populated sandy ocean coasts, consisting of an addition of dredged benthic sediments to the eroded shoreline (Miller et al. 2002). While this method is effective and environmentally sustainable when used periodically, it has a specified design life; the erosion processes will continue to act on the restored beach, requiring perpetual restoration that can amount to enormous costs every few years (Hamm et al. 2002; Miller et al. 2002).

Hybrid stabilization, a combination of erosion control techniques, is commonly used in estuaries and intertidal habitats that are experiencing moderate to high wave energy. In these types of environments, soft stabilization may not be sufficient to ameliorate the erosion caused by high amounts of wind and boat wakes. When wave energy in these habitats is too much for the shoreline to withstand, the use of hard structures can be used in conjunction with natural flora and fauna, thereby creating a hybrid stabilization (Castellan & Wall 2006). Many areas in the Gulf Coast of Alabama and Mississippi have utilized hybrid stabilizations by using a combination of natural materials (i.e. bivalve shell) and artificial materials (i.e. limestone or concrete) to create offshore wave breaks (Swann 2008; Pace 2010). These structures dissipate wave energy, which allows plants to grow on the shoreline landward of the structures (MBNEP 2002; Scyphers et al. 2011). While this type of stabilization will lower wave energy impacting the shoreline, it can change local shoreline currents and sediment transport (NRC 2007).

Figure 1 Image of hard stabilization versus soft stabilization.

Study Species

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* (Meyer et al. 1997; Charlier et al. 2005; NRC 2007) (Figure 2). *Crassostrea virginica* is a filter-feeding bivalve that is able to form three-dimensional fringing reefs in the low to middle intertidal zone, as well as intertidal patch reefs, and expansive subtidal reefs (Dame 2011). Along the western Atlantic, this oyster can be found as far north as St. Lawrence, Canada, and as far south as the Indian River Lagoon, located on the east coast of central Florida (Andrews 1991). The large physical range can partly be attributed to the broad temperature and salinity tolerances of *C. virginica* (Gunter & Geyer 1955). *Crassostrea virginica* is gregarious, with free-swimming larvae that locate adult oyster shell through chemical cues (Burrell 1986; Tamburri et al. 1992).

Figure 2 Species used in living shoreline treatments, *Crassostrea virginica* (left) and *Spartina alterniflora* (right).

Spartina alterniflora is a perennial marsh grass that typically grows from 0.3 to 3.0 m in height (Adams 1963). With salinity tolerance ranging from 0 – 40 ppt, *S. alterniflora* can be found anywhere there is standing water, including lakes, retention ponds, and estuarine coasts (Gleason et al. 1979). The grass commonly uses underground rhizomes to create monospecific stands along sandy, middle and upper-intertidal shorelines (Mooring et al. 1971; Gleason et al. 1979). Because the main mode of reproduction is asexual within stands, one *S. alterniflora* individual can produce hundreds of new shoots in three to five months, quickly covering a vacant shoreline with natural protection and habitat for intertidal species (Mooring et al. 1971; Dennis et al. 2011). Seeds are used to naturally establish *S. alterniflora* in new habitats, and are rarely found as a form of reproduction within stands (Mooring et al. 1971). Stands of *S. alterniflora* can be found along the western Atlantic seaboard from Newfoundland to Florida, and along the northern Gulf of Mexico (Gleason et al. 1979). Healthy populations can reach densities up to 108 stems/m^2 in optimal growing conditions (Gleason et al. 1979). This plant has been included in living shoreline blueprints in many areas, including the Chesapeake and Delaware Bays (Berman et al. 2007; Davis & Luscher 2008). *Spartina alterniflora* is commonly used because it has long, sediment securing roots, and can live submerged by up to 45 centimeters of water for short periods of time (Maricle & Lee 2007). The upright plant structures cause turbulence and dissipation in the flow of water, reducing the erosion caused by wind waves and boat wakes (Neph 1999). When used in living shorelines, *S. alterniflora* is capable of attenuating wave energy while binding sediment with its roots and rhizomes (Gleason et al. 1979; Knutson et al. 1982; Neph 1999).

Study Site

The field portion of this study took place in Mosquito Lagoon, which is a shallow water estuary (average depth: 1.7 m), located on the east coast of Central Florida (Grizzle 1990; Walters et al. 2001) (Figure 3). It encompasses the northernmost section of the Indian River Lagoon (IRL) system. The IRL is one of the most biologically diverse estuaries in the world, which is mostly due to its location spanning both temperate and sub-tropical climates (Provancha et al. 1992; Dybas 2002; Steward et al. 2006). Two hundred thirty square kilometers of northern Mosquito Lagoon are managed by Canaveral National Seashore (CANA), and the rest is divided among the state, NASA, and the Merritt Island Wildlife Refuge (FDEP 2012). The most influential water movement in the majority of this system is a direct result of wind-driven currents; however, some locations are more influenced by tidal currents (Smith 1993; Dubbleday 1975; Hansell 2012). The tidal range in Mosquito Lagoon is variable; around our study site it is microtidal with a range of 10 cm (Smith 1987, 1993; Hall et al. 2001; Steward et al. 2006; Hansell 2012).

Mosquito Lagoon is also a world-renowned fishing area, which introduces intense recreational boating pressure (Johnson & Funicelli 1991; Scheidt & Garreau 2007). Interpretations of aerial photos in the lagoon have shown that a high frequency of recreational boating traffic (location of channels) could be correlated to observed erosion within CANA (Grizzle et al. 2002; Garvis 2012). The combination of microtidal shallow waters with intense boating activity has caused severe erosion along the shorelines of multiple Native American archaeological sites within the national park (Walters 2010; Walters et al. 2012).

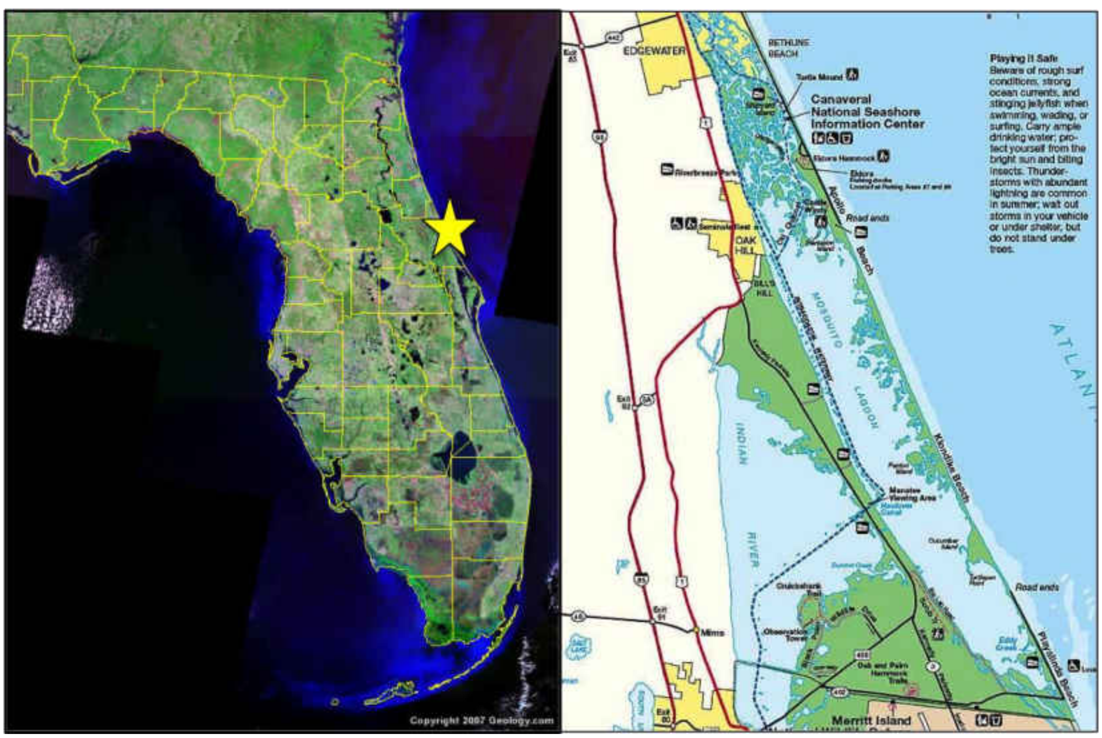

Figure 3 Map of Mosquito Lagoon, Florida.

The natural assemblage of flora and fauna on non-eroding shorelines in Mosquito Lagoon includes *C. virginica* in the low to middle intertidal zone, and *S. alterniflora* in the middle to upper intertidal zone. In the upper intertidal zone continuing landward, four different types of mangroves including red (*Rhizophora mangle*), black (*Avicennia germinans*), white (*Laguncularia racemosa*) and buttonwood (*Conocarpus erectus*) are observed. The mangrove trees in Mosquito Lagoon are somewhat smaller than their Caribbean counterparts in warmer areas due to occasional freezing events in the winter months (Lugo & Zucca 1977).

Mangroves are commonly used in shoreline stabilization projects in tropical and subtropical areas (Odum et al. 1982). These trees are important to shoreline stabilization due to the root structures that either reach down from the trunk of the tree (*Rhizophora mangle*), or grow outward underneath the sediment (*Avicennia germinans*). Both red and black mangrove species are native and have been used in living shorelines in Mosquito Lagoon, but they will not be included in the first two sections of this study. Although these species play a large role in stabilizing shorelines, their effect is only significant when established for long periods of time, up to 10 years for some species (Kathiresan & Bingham 2001). When assessing vacant shores eroding due to wind and boat wakes, planting only mangroves is not a successful way to immediately protect these intertidal shorelines (Kamali & Hashim 2011).

To evaluate the effectiveness of living shorelines containing oysters and *S. alterniflora* to attenuate recreational boat wakes, a field study and wave tank experiments were completed. A large-scale restoration project was also completed using the information gathered from the first two experiments. The goal of each study was unique but complimentary. One study focused on the effect of living shorelines in the field, while the other focused on calculating wave attenuation through living shorelines through the use of a wave tank. The last looked at how the living shoreline preformed on a large scale. Key questions pursued were: 1) Considering living shoreline stabilization with *C. virginica* and *S. alterniflora*, what combination limits shoreline erosion best? 2) How much wave energy is attenuated by oyster shells and marsh grass immediately after deployment? 3) How much does wave attenuation improve after one year, once live oysters have recruited to the shells and marsh grass has produced new aboveground and belowground biomass? 4) Using the results gathered from pervious testing, how effective will living shorelines be at limiting erosion on a large scale in the field?

CHAPTER 2: TECHNIQUES FOR MITIGATING COASTAL EROSION ON SHELL MIDDENS USING LIVING SHORELINES

Introduction

Shoreline erosion has been observed within CANA boundaries in northern Mosquito Lagoon (Walters et al. 2012). While erosion is an ongoing issue within this national park, there is much concern for sediment loss at three archaeological sites. These sites, known as shell middens, contain the only remaining artifacts from the Timucuan Native Americans that lived in this area 800-1200 years ago (Bushnell 1915; Ehrmann 1940). The middens are composed of oyster, clam, and coquina shells, along with broken pieces of pottery and animal bones (Butler 1915). Erosion from natural and anthropogenic sources has caused archaeological artifacts to wash out into the surrounding waters, permanently losing the only records of these Native Americans (Walters et al. 2012).

The eastern oyster *C. virginica*, and smooth cordgrass *S. alterniflora* are native to intertidal shorelines in Mosquito Lagoon. Oysters and marsh grass act as soft armoring for shorelines by increasing wave absorption (Coen & Luckenbach 2000; Piazza et al. 2005). They also maintain high biodiversity for many economically important species with their three-dimensional structures (Newell 1988; Coen et al. 1999, 2007; Newell & Koch 2004; Coen & Grizzle 2007; Stunz et al. 2010). Intertidal oyster reefs within Mosquito Lagoon have been documented to provide habitat for 140 species (Barber et al. 2010). Oysters not only support species directly through structure, but also through water filtration and sediment fertilization (Peterson & Heck 2001a, b; Booth & Heck 2009). *Spartina alterniflora* not only is useful at attenuating wave energy, but also is important for absorbing terrestrial nutrient run-off such as nitrogen and phosphorous before it enters the aquatic system (Broome et al. 1975; Johnston et al. 1984; Anderson et al. 2011).

In previous field experiments testing the effectiveness of living shorelines, the combination of stabilized oyster shell and *Spartina alterniflora* was shown to mitigate shoreline erosion better then either one individually (Walters 2010). The purpose of this project was to determine whether stabilized oyster shell and *Spartina alterniflora* separately or in combination will best mitigate shoreline erosion on three Timucuan shell middens found in the northern region of Mosquito Lagoon.

Methods and Materials

Study Site

National Park Service archaeologists and Canaveral National Seashore Resource Management Specialists prioritized and determined the three sites needing stabilization (Figure 4). The three middens were Castle Windy (28°53'38.88"N; 80°48'29.35"W) Garver Island (28°51'50.81"N; 80°47'15.86"W) and Hong Kong (28°52'11.63"N; 80°49'14.35"W). The middens were 4.07 ± 0.33 m from the mean high tide line (mean distance ± SE), and had visible erosion scarps where fossil shells were spilling into the water. All of the middens were separated by a minimum of 1500 meters, and had a mean eroding shoreline length (± SE) of 217.3 ± 45.7 meters. The average shoreline slope ratio of 15:1 (± 0.6) was found by sampling 10 haphazard perpendicular transects along each midden shoreline. The sediment found on all three midden shores consisted of a mixture of oyster and clam shell fragments, mixed within Astatula fine sand that had a density ranging from 1.84 to 2.04 g cm^{-3} with a mean density of 1.85 ± SE 0.4 g cm^{-3} (Baldwin 1980).

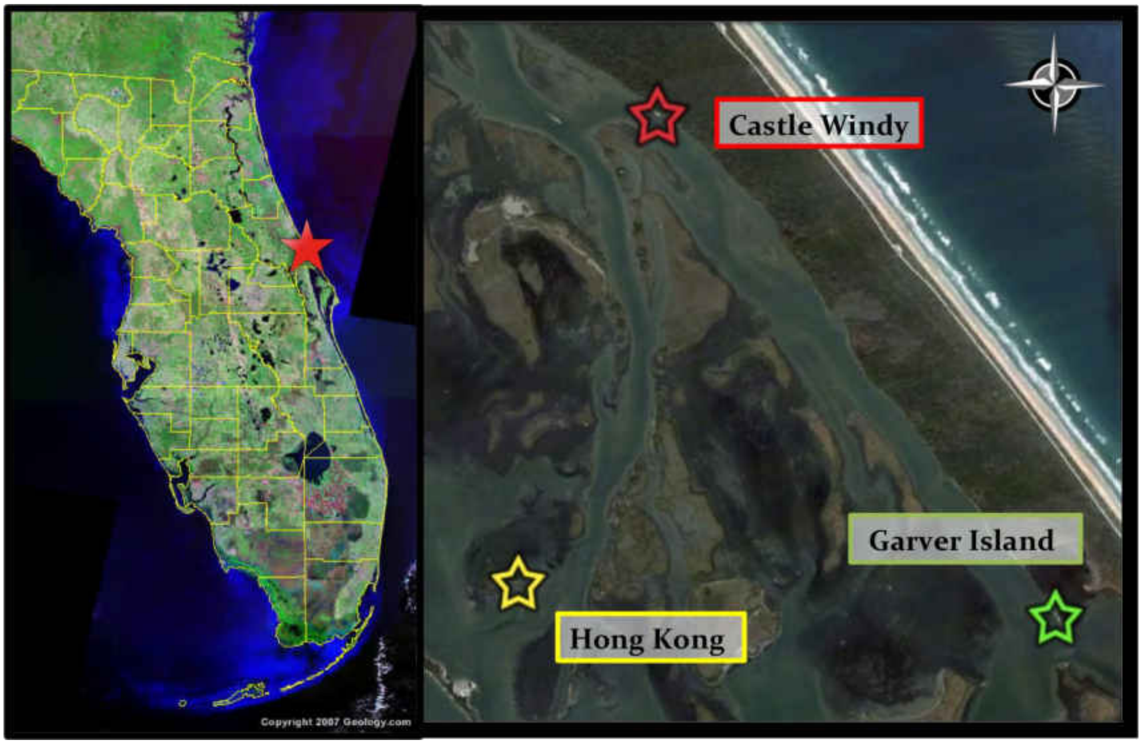

Figure 4 Map of shell middens locations: Castle Windy, Garver Island and Hong Kong.

Shoreline Stabilization Methods Used in CANA

To provide stabilized substrate for oyster recruitment in the lower intertidal zone, oyster restoration mats were used. These mats consisted of 0.25 m^2 of Vexar™ mesh with 1.5 cm openings. Thirty-six adult oyster shells are attached to the mesh with 50 lb. test plastic cable ties, so the shells were retained in a vertical orientation (Wall et al. 2005). The oyster shells used for the mats were collected and quarantined on land for at least six months to avoid the transfer of diseases and invasive species (Bushek et al. 2004; Cohen & Zabin 2009). Once constructed, the mats were placed on top of the bare sediment in the low to middle intertidal zone in a quilt like fashion, where natural recruitment to deployed shells occurred. In two to six years, over 300 live oysters can be counted on one individual mat, or a mean density of 472 live oysters per square meter after 5.5 years (Walters et al. 2013) (Figure 5).

Figure 5 Newly created oyster restoration mat (left) and oyster reef restored using oyster restoration mats after 5.5 years (right).

To create beds of *S. alterniflora*, individual shoots from surrounding shorelines within CANA were collected and potted for up to six months prior to deployment (Figure 6). Once a large root bundle formed, the plants were transplanted to the middle to upper intertidal zone. The newly planted *S. alterniflora* then grows into large monospecific stands along the eroding shoreline (Gleason et al. 1979).

Figure 6 Volunteers potting *S. alterniflora* to be used for stabiliztion (left) and a stabilizated shoreline containing planted *S. alterniflora* after 9 months (right).

Field Experiment

Changes in sediment height were tested on the three eroding shell middens within replicated experimental plots (12.25 m^2) to assess the best type of living shoreline protection for each midden. Each experimental plot was 3.5 meters wide and 3.5 meters long, and all were separated by at least 5 meters of shoreline. Two of the middens had 20 total plots (Castle Windy and Garver), and one midden had 16 total plots (Hong Kong) due to a shorter shoreline length. Five replicates (four at Hong Kong) of four different treatments were created to compare erosion rates. Treatment 1 (combination treatment) consisted of 35 oyster mats in the lower 2.5 m of the plot and 15 plugs of *Spartina alterniflora* in the upper 1 m of the plot. The 35 oyster mats were deployed in the lower intertidal zone, and then weighted down on each corner with donut-shaped concrete sprinkler weights. Mats were attached to the donut weights with 50 lb. test cable ties. Fifteen *Spartina alterniflora* plugs were planted in a haphazard array within the middle intertidal zone. Individual holes were created for the plugs with a gas-powered 43 cc Earthquake auger. Treatment 2 (*Spartina* only) consisted of 15 plugs of *Spartina alterniflora* in one plot. This treatment was deployed by measuring out a 3.5 m long by 1 m wide area in the middle intertidal zone, and by adding the plants the same as treatment 1. Treatment 3 (oyster mats only) consisted of 35 oyster mats in one plot. This treatment was deployed by measuring out a 3.5 m long by 2.5 m wide area in the lower intertidal zone. The mats were weighted down and attached as in Treatment 1. Treatment 4 was a control, with no restoration treatments in the plots (Figure 7).

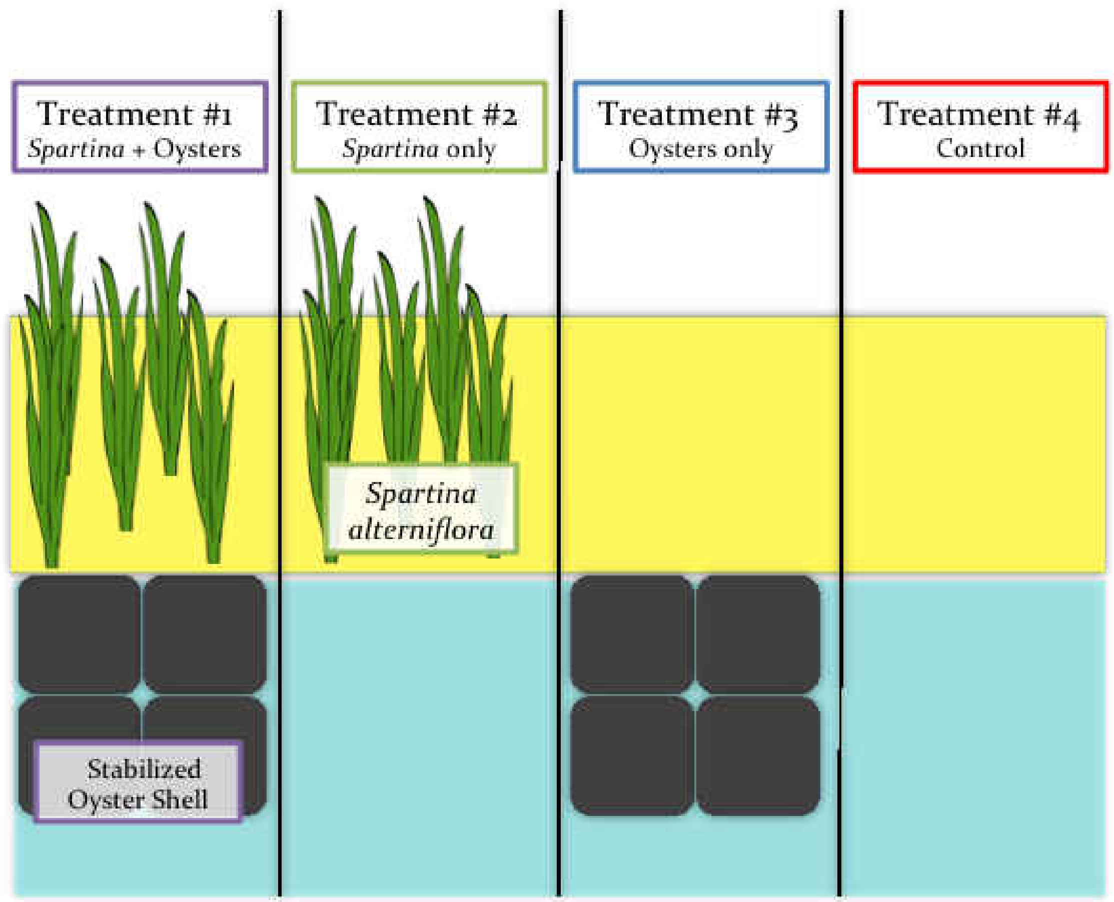

Figure 7 Experimental plots containing different treatments found on each shell midden. When deployed, each plot was separated by at least 5 meters.

The experimental treatment plots were set up to mimic natural living shorelines in the area, with the oyster mats in the lower intertidal zone and *Spartina alterniflora* in the middle intertidal zone. The intertidal zone was designated as the area directly below the mean high tide of 0.782 meters (determined from the mean high tide for Mosquito Lagoon tidal datum).

Erosion stakes were used to measure the change in vertical sediment height at each experimental plot. The stakes were created by drilling five holes at five-centimeter intervals in the topmost 30 cm of a 0.75 m PVC. This allowed for underwater measurements. All stakes were inserted into the sediment with a sledgehammer and aligned to a drilled hole 15 centimeters from the top (middle hole) (Hudson 1957). Once the mats were deployed and the marsh grass was planted, three of these PVC erosion stakes were deployed into the center of each test plot in a transect. The stakes were placed so that there was one meter between each stake, with the most landward stake being placed at the mean high tide line (Figures 8, 9).

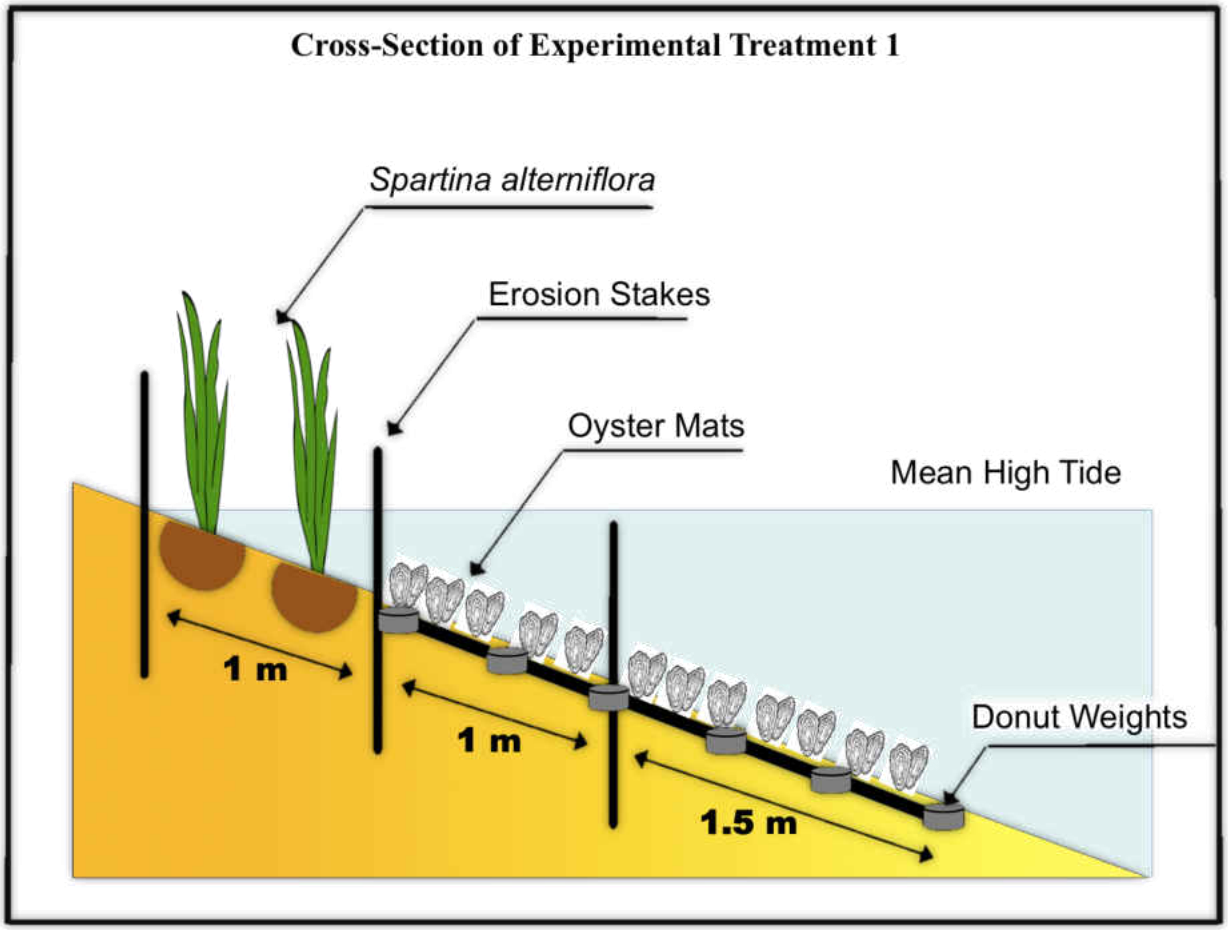

Figure 8 Cross-section of Treatment 1.

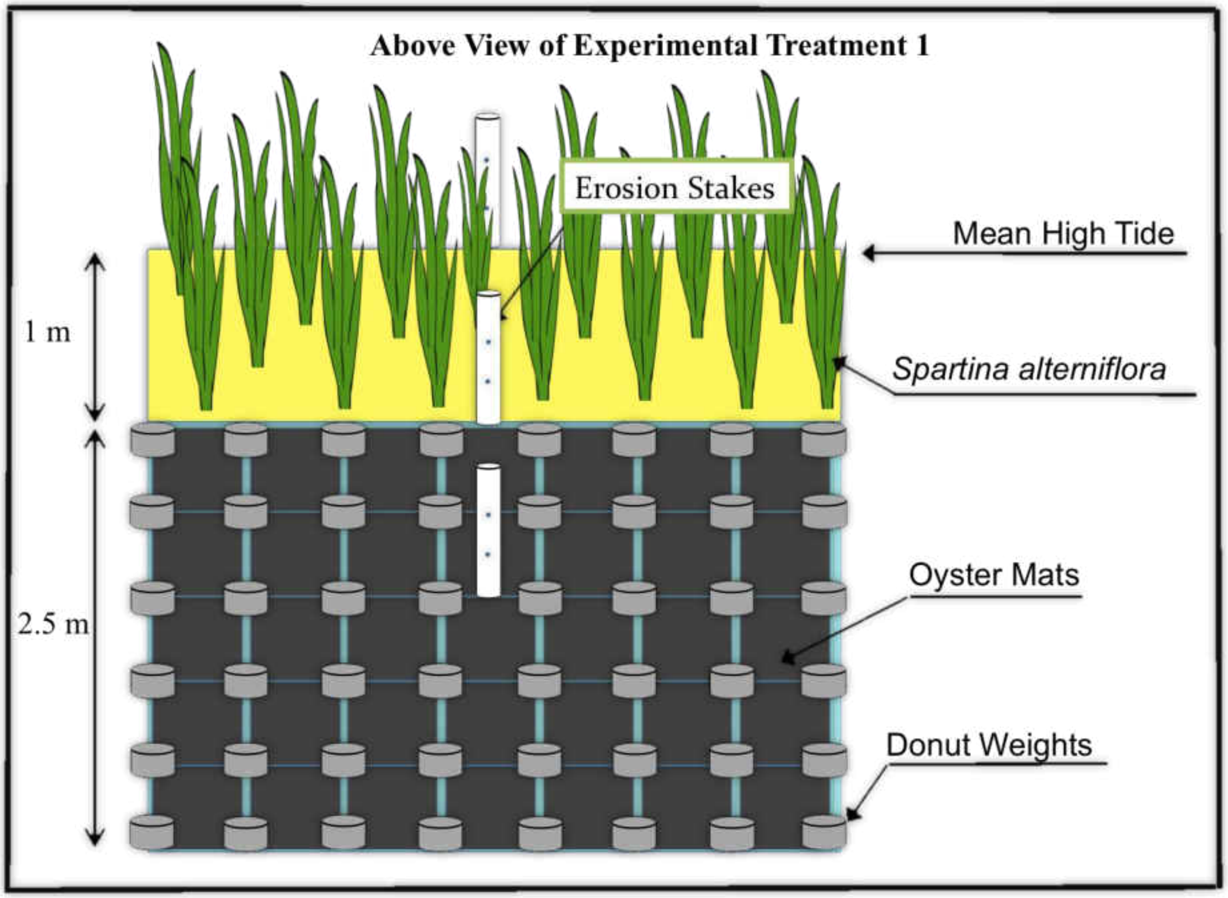

Figure 9 Aerial view of Treatment 1.

All erosion stakes were monitored three days after deployment, one week after deployment, one month after deployment, and then monthly for one year. Erosion or accretion was measured by recording the sediment height on the stakes relative to the drilled holes. A block ANOVA was used to test for significant differences among treatments after one year, with each midden acting as a block. In addition to monitoring erosion, biotic factors including oyster recruitment, plant survival and plant propagation were documented at six months and at one year. All mats were checked for live oysters, and all surviving plants and new shoots were counted. Also, the highest leaf blade on each plant was measured to document growth.

It is important to note that *S. alterniflora* survival was extremely low at all three shell middens, and the causes for this are discussed in the results and discussion section. When examining the figures and tables in the results section, note that the *S. alterniflora* only treatments are in most instances very similar to the control treatments. Plant survival did not change after one month of being in the field.

Results

Overall Erosion and Accretion

There were highly significant differences in sediment heights among treatments after one year in the field ($p < 0.001$; Table1). The combination treatment and oyster mats only treatment accreted significantly more sediment than the *Spartina alterniflora* only treatment and the control (Table 2). The block effect was marginally non-significant ($p = 0.101$; Table 1). To better understand the treatment effects at each site independently, a one-way ANOVA was run for each site, along with appropriate Tukey's post-hoc tests. This was done to allow resource managers to understand the best stabilization method needed at each midden, and to examine the differences in sedimentation at each shell midden. Post-hoc Tukey's tests showed that the control and *S. alterniflora* only treatments were not statistically different ($p = 0.575$) on any midden, and the combination treatment and oyster shell only treatments were not significantly different from each other either ($p = 0.997$). The two groups were however significantly different from each other at all three shell middens ($p < 0.001$).

Table 1 The results of a block ANOVA of change of erosion as explained by shoreline stabilization type after 1 year in the field. Note that some erosion stakes were lost after one year.

Source	DF	Sum Sq	Mean Sq	F value	Pr (>F)
Treatment	3	211.404	70.468	23.283	<0.001*
Block	2	14.643	7.321	2.419	0.101
Residuals	42	127.116	3.027		

***A significant *P* value at p <0.05 level.**

Table 2 Mean change in vertical sediment height (cm) after one year in the field at different treatments.

	Combination	Oyster Shell Only	*Spartina* Only	Control
Castle Windy	4.33	4.77	-0.22	-0.22
Garver Island	4.97	4.9	0.3	-0.93
Hong Kong	2.21	2.42	0.13	-0.72

Castle Windy Erosion and Accretion

Treatments at Castle Windy were found to be significantly different from each other ($p = 0.008$) when compared with a one-way ANOVA (Table 3). The oyster shell only treatment had the highest mean (±SE) accretion amount at 4.77 ± 0.35 cm of sediment (Figure 10). The combination treatment (oyster shell and *S. alterniflora*) had the second highest mean accretion amount at 4.33 ± 0.80 cm. There was no *S. alterniflora* survival at Castle Windy due to smothering from seagrass wrack within one month of deployment. Hence, it is not surprising the *S. alterniflora* only treatments and the control treatments did not differ, with both losing 0.22 ± 1.48 cm of sediment (Figure 10). Three live oysters were found recruited to the stabilized oyster shell 12 months after deployment at this site. All shell was, however, densely covered with striped barnacles (*Balanus amphitrite*) and sea lettuce (*Ulva lactuca*).

Table 3 The results of a one-way ANOVA of change of erosion as explained by shoreline stabilization type after 1 year in the field at Castle Windy.

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Treatment	3	81.532	27.177	6.389	0.008*
Residuals	12	51.047	4.254		

*** A significant *P* value at p < 0.05 level.**

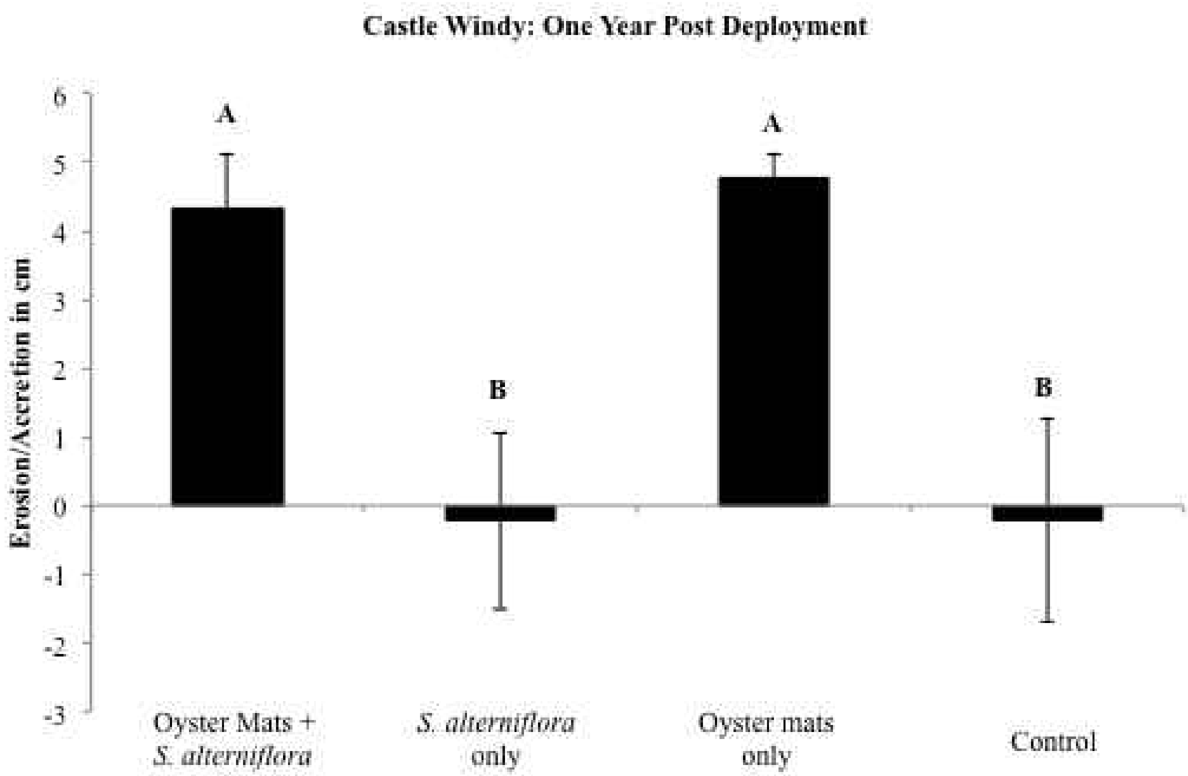

Figure 10 Mean change in sediment height after the treatments were in the field for one year at Castle Windy (±SE). Sediment heights were compared using a one-way ANOVA, and were significantly different from each other ($p = 0.008$). Treatments with different letters are significantly different at $p < 0.05$ level as determined by Tukey's post hoc tests.

Garver Island Erosion and Accretion

Sediment heights associated with different treatments at Garver Island were also significantly different from each other ($p > 0.0001$, one-way ANOVA, Table 4). The combination treatment (oyster shell and *S. alterniflora*) had the highest mean (±SE) accretion amount at 4.97 ± 0.73 cm. The oyster shell only treatment had the second highest mean accretion amount at 4.90 ± 0.90 cm of sediment (Figure 11). There was one surviving *S. alterniflora* plant out of the original 150 deployed (0.6% survival) at Garver Island. The one surviving plant produced 11 new shoots within 12 months. The remaining 149 plants died due to smothering from seagrass wrack within the first month of deployment. Since almost all of the plants died, it is not surprising that the *S. alterniflora* only treatment and control treatment were not significantly different when comparing with a Tukey's test (Figure 11). One live oyster was found recruited to the stabilized oyster shell at this site. All other shells were densely covered with striped barnacles (*Balanus amphitrite*) and sea lettuce (*Ulva lactuca*).

Table 4 The results of a one-way ANOVA of change of erosion as explained by shoreline stabilization type after 1 year in the field at Garver Island.

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Treatment	3	141.626	47.209	14.237	<0.001***
Residuals	16	53.055	3.316		

*** A significant *P* value at p < 0.05 level.**

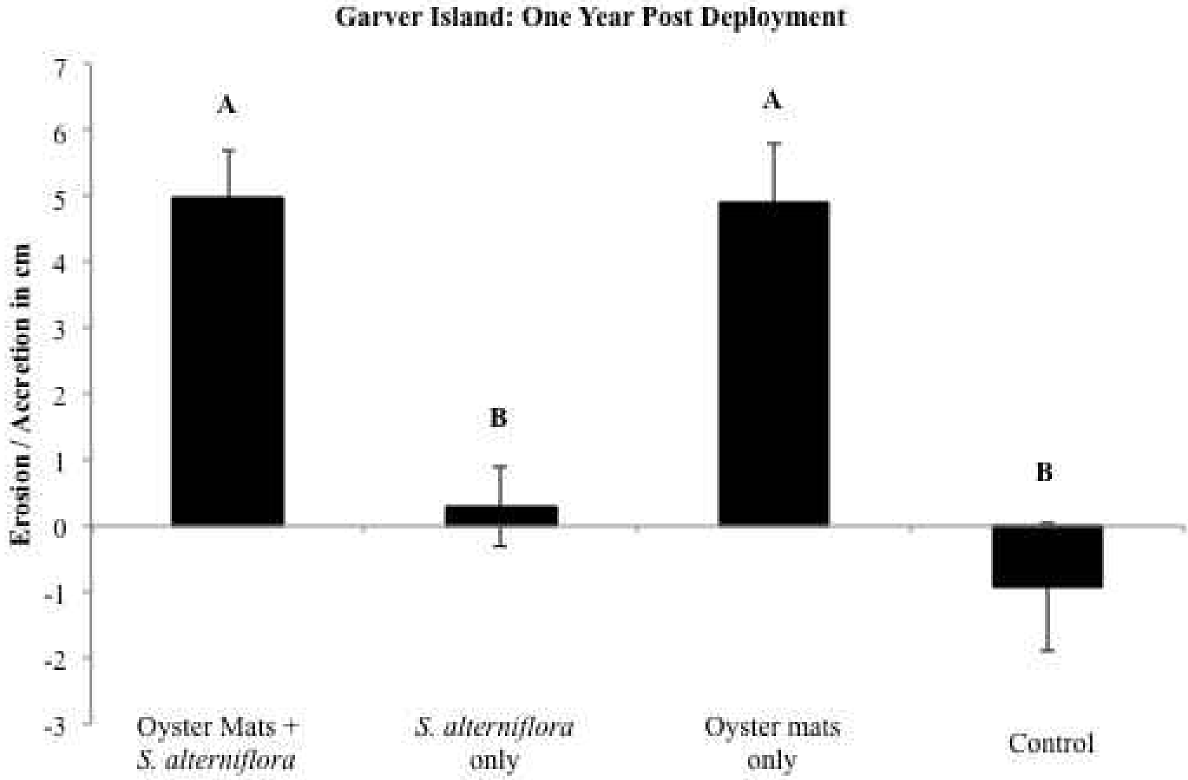

Figure 11 Mean change in sediment height after treatments were in the field for one year at Garver Island (±SE). Sediment heights were compared using a one-way ANOVA, and were significantly different from each other. Treatments with different letters are significantly different at $p < 0.05$ level as determined by Tukey's post hoc tests.

Hong Kong Erosion and Accretion

Sediment heights differed significantly among treatments at the Hong Kong midden ($p = 0.001$, one-way ANOVA, Table 5). The oyster shell only treatment had the highest mean (±SE) accretion amount at 2.42 ± 0.60 cm of sediment (Figure 12). The combination treatment (oyster shell and *S. alterniflora*) had the second highest mean accretion amount at 2.20 ± 0.39 cm. Hong Kong had the highest *S. alterniflora* survival at 22.5%, or 27 individuals out of 120 (Table 6). From these 27 surviving individuals, 160 new shoots were produced within 12 months (5.9 ± 2.7 shoots per plant). No live oysters were found recruited to this site, but the mats were densely covered with striped barnacles (*Balanus amphitrite*) and pleated sea squirts (*Styela plicata*).

Table 5 The results of a one-way ANOVA in change of erosion as explained by shoreline stabilization type after 1 year in the field at Hong Kong.

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Treatment	3	29.179	9.726	11.820	0.001**
Residuals	12	9.874	0.823		

*** A significant *P* value at p < 0.05 level.**

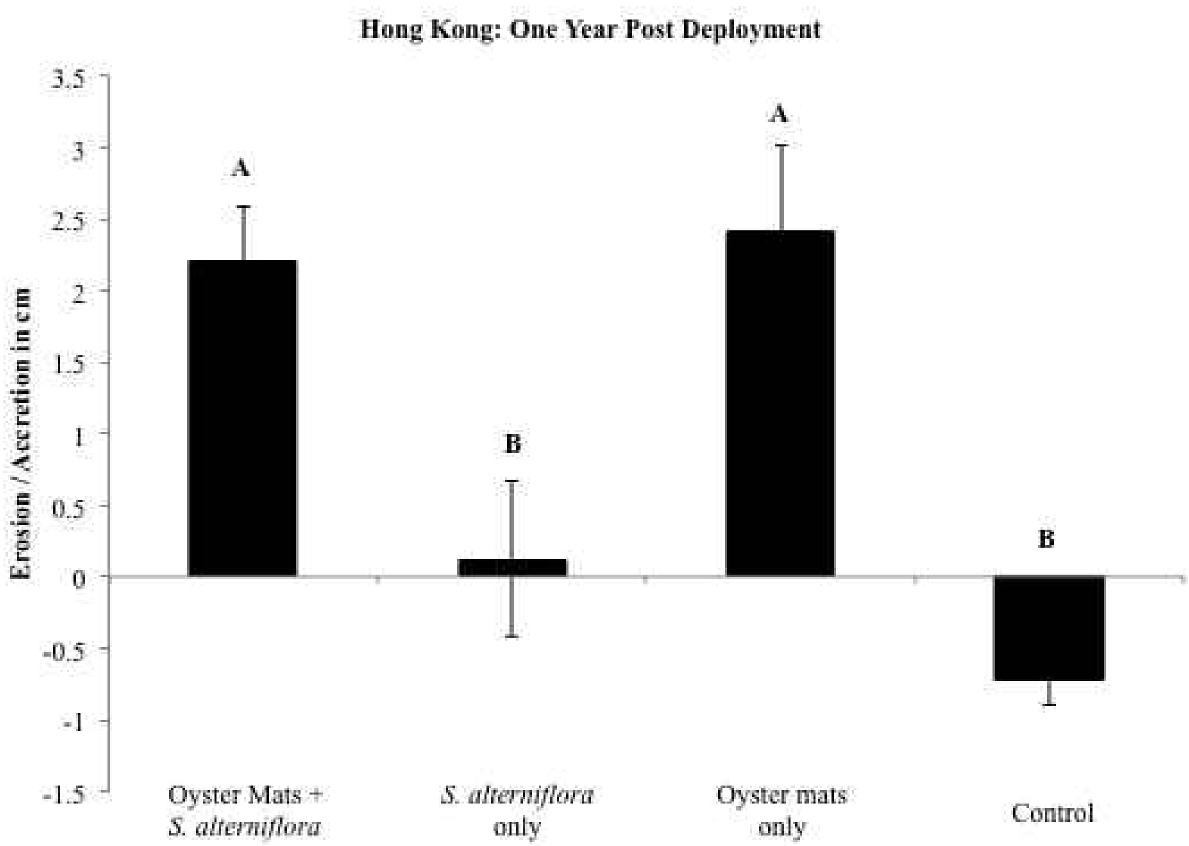

Figure 12 Mean change in sediment height after treatments were in the field for one year at Hong Kong Island (±SE). Sediment heights were compared using a one-way ANOVA, and were significantly different from each other. Treatments with different letters are significantly different at p < 0.05 level as determined by Tukey's post hoc tests.

Table 6 *Spartina alterniflora* survival and new growth at three shell middens after one year in the field.

	Total Planted	Survival	Total New Shoots
Castle Windy	150	0%	0
Garver Island	150	0.60%	11
Hong Kong	120	22.50%	160

Discussion

Living shorelines containing oyster shell (combination treatment and oyster shell only treatment) significantly changed the erosion rates found on shell middens (Figures 10, 11, 12). All three sites displayed similar sediment accretion at treatments with oyster shells plus *S. alterniflora*, and oyster shells only. Without stabilization (control treatment), the shell midden shorelines were eroding at all sites. With this information, CANA began to stabilize these three shell middens (Castle Windy, Garver Island, and Hong Kong) with living shorelines in the summer of 2012.

Our planting techniques for *Spartina alterniflora* had proven to be effective in previous stabilization experiments, so our methods do not appear to be the cause of the poor survival in this experiment (Woodhouse et al. 1974; Craft et al. 2003). Hong Kong midden was the only site with *S. alterniflora* survival (22.5%). Although the difference between *S. alterniflora* only treatments and controls was not significant there, the sites where *S. alterniflora* survived show a mean positive change in sediment height (Figure 12). The failure of *S. alterniflora* was due to seagrass smothering, which may have been due to periodic turnover of seagrass populations in the lagoon. Studies completed in the Indian River Lagoon system have identified decline and recovery periods of seagrass (Morris & Virnstein 2004). After seagrass beds accumulate thick layers of organic detritus (10-15 cm), populations appear to crash, and recover every two to three years (Morris & Virnstein 2004). Since Mosquito Lagoon is a poorly flushed system with slow water movement, it is possible that such an event occurred the summer of our stabilization (Fletcher & Fletcher 1995). This would help explain the large amounts of seagrass wrack found floating in the lagoon, and on the shorelines.

We believe that the results of this restoration would have been even more successful (i.e. greater accretion) if *S. alterniflora* had higher survival rates at all shell middens. Based on previous studies and living shoreline stabilization efforts, when *S. alterniflora* survives on eroding shorelines, it can significantly reduce wave energy, as well as limit erosion (Woodhouse et al. 1974; Berman et al. 2007). With this information, we believe it is still beneficial to include *S. alterniflora* in shoreline stabilizations on these shell middens due to their potential positive impacts on sediment accretion.

Oyster recruitment was minimal at all three shell midden sites. Areas 1.8 km north of these middens within CANA experience extensive oyster recruitment, with more than 300 live oysters per 0.25 m^2 after 2 - 6 years. Low recruitment could be correlated to the meeting of temperate and sub-tropical climate zones that cut through Mosquito Lagoon. More likely, the low oyster recruitment is linked to the seagrass die-offs during the experimental period. Large amounts of dead seagrass trigger hypoxic conditions and cyanobacteria blooms, which have been documented to limit oyster survival and recruitment in Florida estuaries (Phlips et al. 1999). The stabilized shell still recruited other invertebrates such as barnacles and tunicates, which have larval dispersal periods that differ from *C. virginica*, and compete for space once settled on the stabilized shell (Wells 1961; Burrell 1986; Gosling 2008; Wall 2004). Even though we saw very limited oyster recruitment, the oyster restoration mats proved to be effective at stabilizing the shoreline, all while producing a three-dimensional habitat for native flora and fauna.

With numerous boaters (~46,000) accessing the Mosquito Lagoon annually, boat wakes will occur and impact shorelines (Scheidt & Garreau 2007). With 56 kilometers per hour speed limit within CANA, boat wakes from fast moving vehicles dislodge shoreline sediment, oysters, and bank vegetation (Gabet 1998; Grizzle et al. 2002; Walters et al. 2002; Wall et al. 2005). With this study, we have provided evidence that soft armoring not only slows erosion, but also results in accretion.

CHAPTER 3: WAVE ATTENUATION EXPERIMENTS OVER LIVING SHORELINES: A STUDY TO ASSESS RECREATIONAL BOATING PRESSURE

Introduction

Boating traffic has been linked to the loss of intertidal habitat as well as shoreline erosion in estuaries and boat channels (Gabet 1998, Grizzle et al. 2002). 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). Studies have quantified the energy caused by the 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 – 2 mm grain diameter) (Wentworth 1922; Limerinos and Smith 1975; Foda 1995; Foda et al. 1999). This increased shear stress associated with boat wakes, which is more energetic than naturally occurring waves, ultimately causes sediment loss along shorelines (Komar and Miller 1973; Fredsøe and Deigaard 1992; Bauer et al. 2002; Soomere and Kask 2003).

Waves created by boats cause shoreline erosion not only with increased wave heights and energy, but also with wave properties that differ from naturally occurring wind waves (Parnell et al. 2007). Boat wakes are especially detrimental to aquatic organisms and their habitats in systems with small fetches, which are normally exposed to low natural wave activity (Bauer 2002). Keddy (1982, 1983) documented that high wave energy is correlated to low biodiversity and biomass of shoreline plants, including *S. alterniflora* and freshwater ferns. Wave energy has also been shown to play a role in seagrass survival, growth and dispersal (Fonseca and Bell 1998). Increased wave energy can even affect the physical landscape, changing the natural shoreline bathymetry and beach processes (Bourne 2000; Soomere and Kask 2003; Parnell et al. 2007).

In shallow bodies of water (< 3 m deep) where primarily recreational boats are used, clay and silt shorelines can experience 0.01–0.22 mm of erosion per boat passage (Bauer et al. 2002). The wakes cause disturbances to the sediment, and also physically impact intertidal oyster reefs (Walters et al. 2002; Wall et al. 2005). Boats passing oyster reefs in Mosquito Lagoon have been found to dislodge individual oyster shells and oyster clusters from the sediment. These dislodged oysters then get pushed above the high tide line, removing the three-dimensional wave attenuating structure from the benthos (Walters et al. 2002; Wall et al. 2005).

With recreational boating likely causing some of the shoreline erosion observed at Mosquito Lagoon shell middens, we wanted to see how well living shorelines that include oysters and *S. alterniflora* attenuated wave energy created by boat wakes. We asked two questions: 1) How much wave energy is attenuated by oyster shells and marsh grass immediately after deployment? 2) How much does wave attenuation improve after one year, once live oysters have recruited to the shells and marsh grass has produced new aboveground and belowground biomass?

Methods and Materials

The indoor wave tank at the Florida Institute of Technology's Surf Mechanics Laboratory 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 deep, and generated waves using a 0.91 m flap paddle located 0.6 m from the back wall of the tank (Figure 13). 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 (Lohmann et al. 1990).

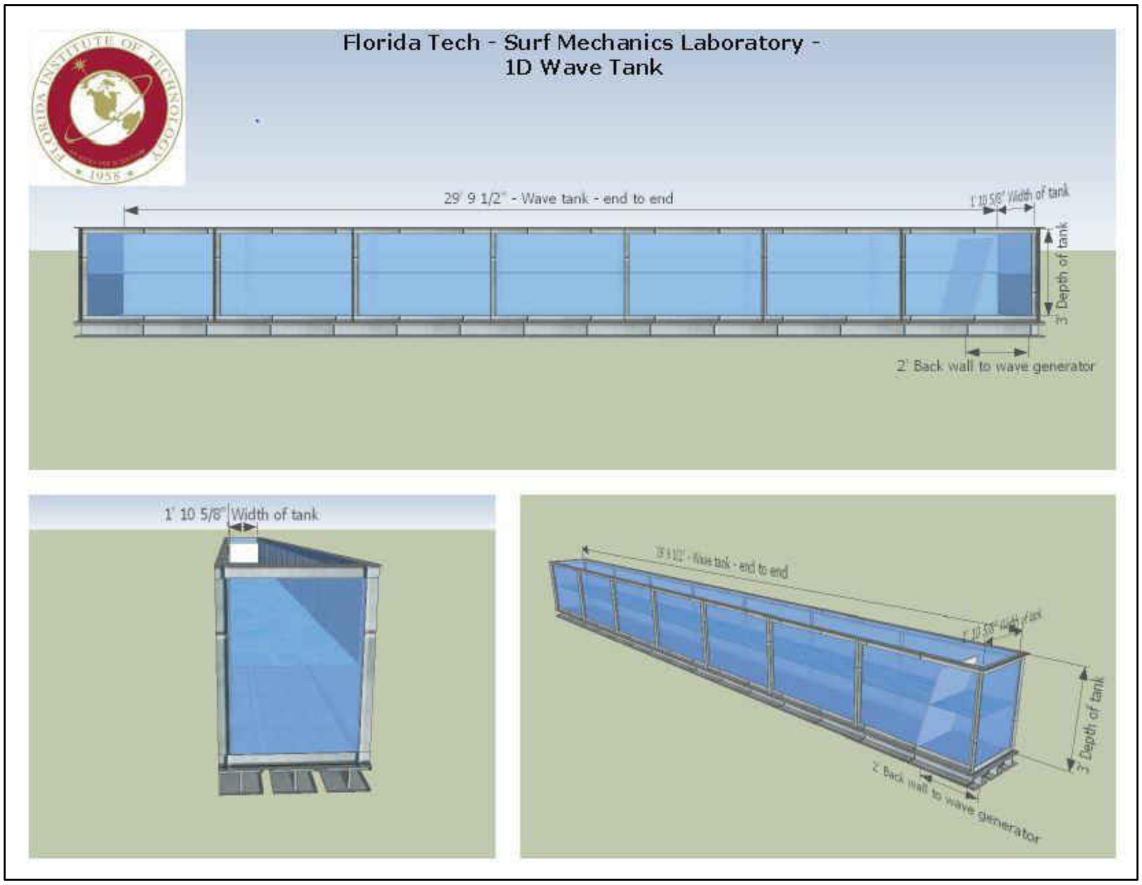

Figure 13 Diagram of wave tank used in study. The wave tank is located in the Surf Mechanics Laboratory at the Florida Institute of Technology.

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). These displacements were 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 field prototype model closely in size (Hughes 1993).

A sediment shoreline was created in the wave tank at the end of the tank opposite from the paddle to mimic shell midden shorelines in CANA. A slope of 15:1 was chosen to represent the natural bathymetry in Mosquito Lagoon. This slope was documented by surveying the intertidal zones of 10 shell midden shorelines (15:1 m ± SE 0.6 m). 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 (Figure 14). For this, 719 kg (0.389 m^3) of sediment was excavated from shorelines of Mosquito Lagoon and transferred to the wave tank and graded to the above specifications. All sediments were returned to Lagoon shoreline donor sites post-experiment.

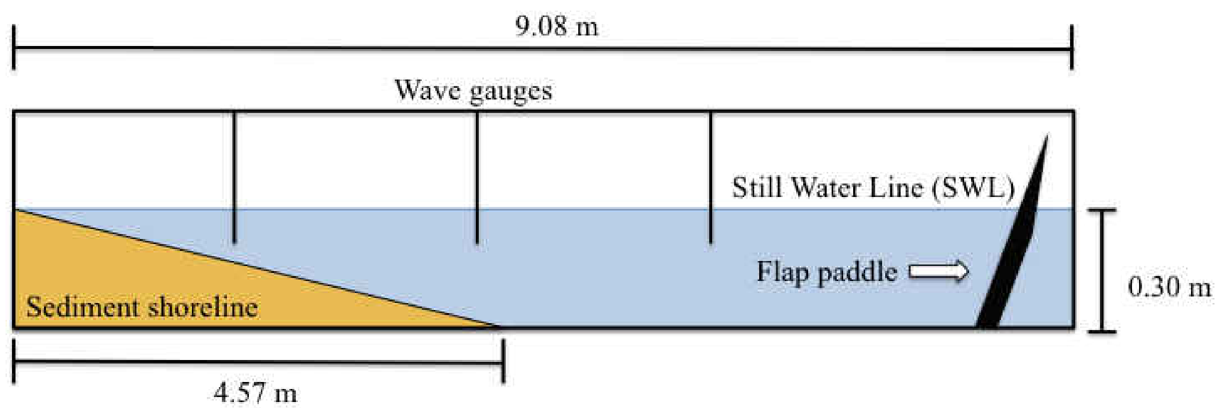

Figure 14 Diagram of wave tank with 15:1 sloped sediment shoreline used in study. The flap paddle is opposite of the shoreline.

To determine desired wave heights for our wave tank trials, boat wake surveys were completed in Mosquito Lagoon. Over three months, forty-five separate thirty-minute surveys on four different shell midden shorelines led to determining the average wave height, period and number of waves in a wave train created from 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 (±1.98 SE) and a period of 1.8 seconds. This is significantly larger than the natural wind waves found in Mosquito Lagoon (t-test; $p < 0.001$) (Figure 15). To mimic a boat pass in the tank, 10 waves that were 12.7 cm tall were used. Because the tank could produce a 12.7 cm wave, a 1:1 ratio was maintained throughout this experiment, therefore closely mimicking the physical settings in the field. The water within the tank was allowed to settle between each boat wake wave train to avoid errors associated with wave reflection.

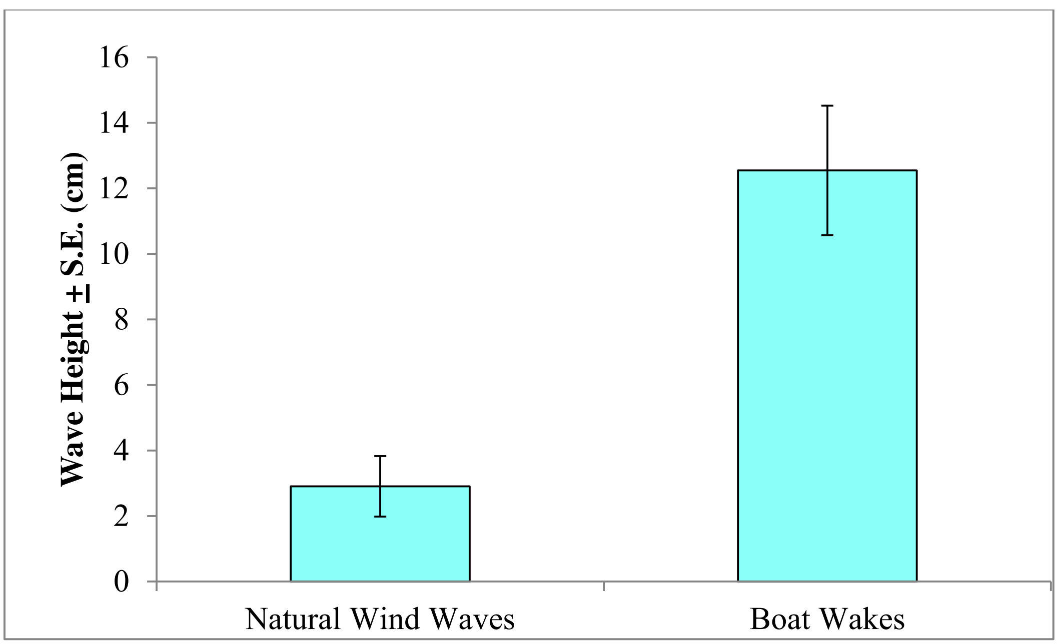

Figure 15 Wave height comparison of natural wind waves versus boat wakes after shell midden surveys. Means are significantly different (t-test; $p < 0.001$).

For each trial there were ten simulated boat passes (10 waves per pass) per treatment, and there were three trials completed 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, 2) stabilized oyster shell at depths from 0.26 m to 0.22 m relative to the SWL, 3) *S. alterniflora* at depths from 0.17 m to 0.13 m from the SWL, and 4) a combination of oyster shell and *S. alterniflora* at these depths (Figures 16, 17). Treatment locations along the sloped shoreline correlated to intertidal depths found in Mosquito Lagoon.

Two time variables for each treatment also were completed to evaluate new restoration and one-year old establish restoration. New restoration included new oyster shell and newly planted *S. alterniflora*, two new oyster mats with 72 disarticulated oyster shells in a 0.5 m^2 area (1 m long x 0.5 m wide), and five individual *S. alterniflora* shoots in a 0.25 m^2 area (0.5 m long x 0.5 m wide). The one-year old established restoration evaluated oyster shell that had recruited live oysters for one year, and *S. alterniflora* in densities equivalent to one-year post planting. The two oyster mats had a mean (± SE) of 158 ± 6.2 live oysters attached to the original 72 disarticulated shells in a 0.5 m^2 area. The marsh grass consisted of 37 individual *S. alterniflora* shoots in a 0.25 m^2 area. *S. alterniflora* was collected from three populations within Mosquito Lagoon, and oyster mats with live adult oysters were obtained from Mosquito Lagoon restoration sites (Figure 17). *Spartina alterniflora* and oyster mats were replaced between trials to retain independence of replicates, but their physical characteristics remained as similar as possible.

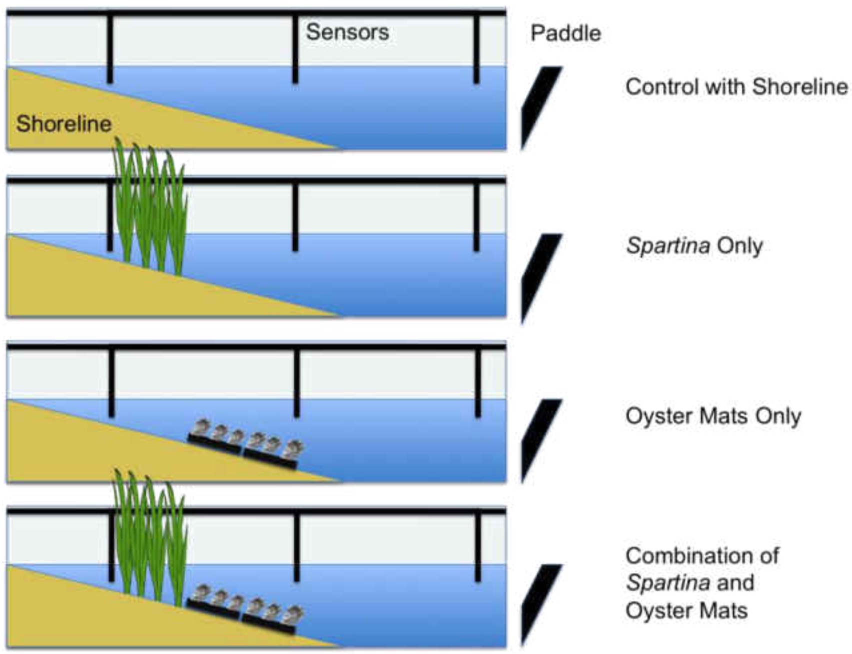

Figure 16 Newly deployed control, *Spartina* only, oyster mat only, and combination (*Spartina* + oyster mat) treatments for the wave tank experiment.

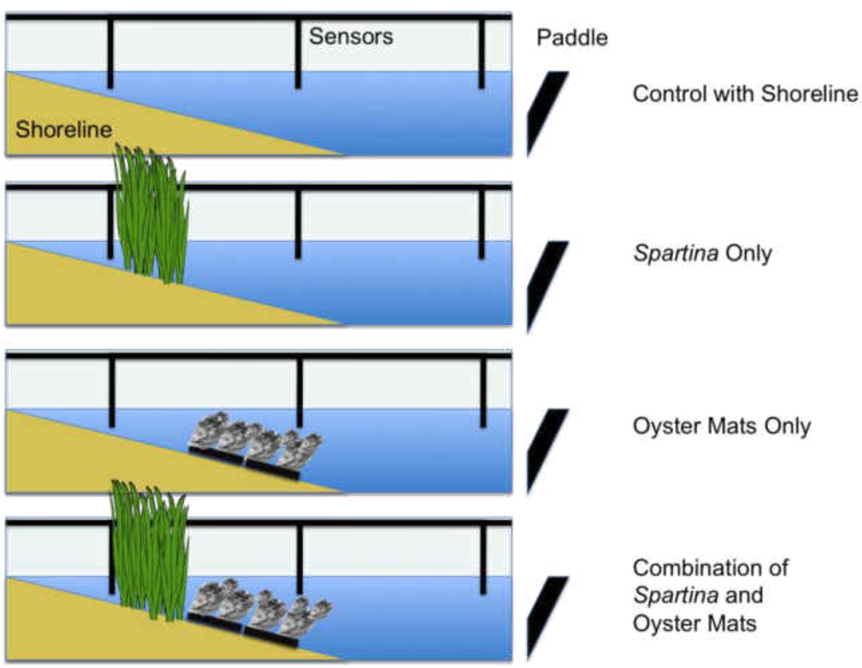

Figure 17 One-year old established control, *Spartina* only, oyster mat only, and combination (*Spartina* + oyster mat) treatments for the wave tank experiment.

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

$$E = \frac{1}{8} \quad gH^2 , \qquad \text{(Eq. 1)}$$

where E is wave energy, is fluid density, g is the acceleration of gravity, and wave height is represented by H. Fresh water was used within the wave tank (= 1000 k m^{-3}) instead of brackish (= 1025 k m^{-3}), which is found at Mosquito Lagoon. This was due to the cost of salt, and potential corrosion issues within the tank and gauge effectiveness. Ultimately, a relative comparison will be made that eliminates the role of density. After calculating wave energy with both a fresh and brackish water density, there was no difference in the final percent change of wave energy through the treatments. The average wave height for each treatment was calculated three separate times, once for each wave gauge (well developed wave, before stabilization, after stabilization). The heights of waves 4-8 of each wave train were used to calculate an average height of one wave, and then wave energy using Eq. 1.

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

$$k_t = H_{transmitted}/H_{input} , \qquad \text{(Eq. 2)}$$

where $H_{transmitted}$ is the attenuated wave height after progressing through the treatment (m), and H_{input} is 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 in this case, a lower coefficient represents a more efficient 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 x 2 time variables) in random order. Prior to analysis, 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 $p < 0.05$.

Results

Wave Height and Energy

Blocks were not significantly different from one another (ANOVA: $p = 0.730$), so a two-way ANOVA was used to analyze differences among treatments and time since deployment (new deployment, 1-year old deployment). Both time since deployment and treatments were found to be significantly different (ANOVA: $p < 0.0001$) with a significant interaction effect (Table 7).

Table 7 Two-way ANOVA comparing the mean wave height as explained treatment type (control, *Spartina* only, oyster mat only, combination) and age (newly deployed and one-year old).

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Treatment	3	35.751	11.917	31.344	<0.001*
Age	1	24.63	24.63	64.783	<0.001*
Treatment*Age	3	12.849	4.283	11.265	<0.001*
Residuals	16	6.083	0.38		

*** A significant *P* value at p < 0.05 level.**

Mean wave height was calculated after each wave train progressed through the living shoreline treatment. Wave heights for controls were significantly higher than for all other tested treatments (Tukey's post-hoc test; Figure 18). 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 *S. alterniflora* showed the highest mean (±S.E.) wave height reduction at 5.52 ± 0.31 cm. This equates to a 67.3% decrease in total wave energy. Established live oyster alone had the second largest reduction in wave energy (44.7%), with a mean wave height reduction of 3.36 ± 0.23 cm. The lowest wave height reduction (0.51 ± 0.20 cm) was associated with newly deployed *S. alterniflora*. This equated to a 6.9% reduction in wave energy (Figure 19).

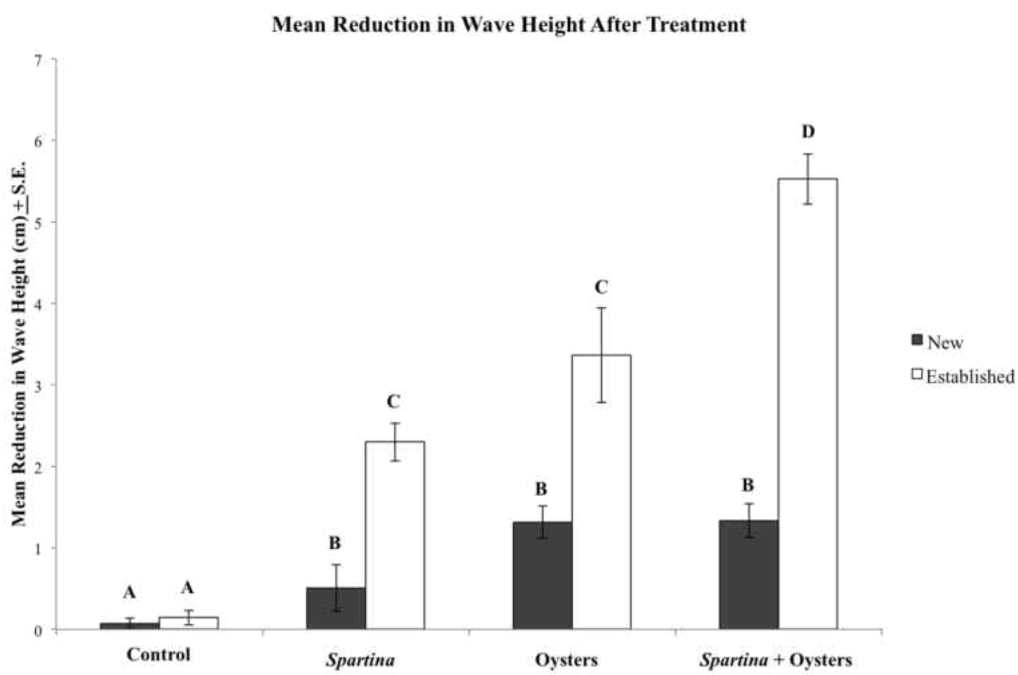

Figure 18 Mean change in wave height of one wave after encountering shoreline stabilization treatment (±SE). Wave heights were compared using a two-way ANOVA. Treatments with different letters are significantly different at $p < 0.05$ level as determined by Tukey's post hoc tests.

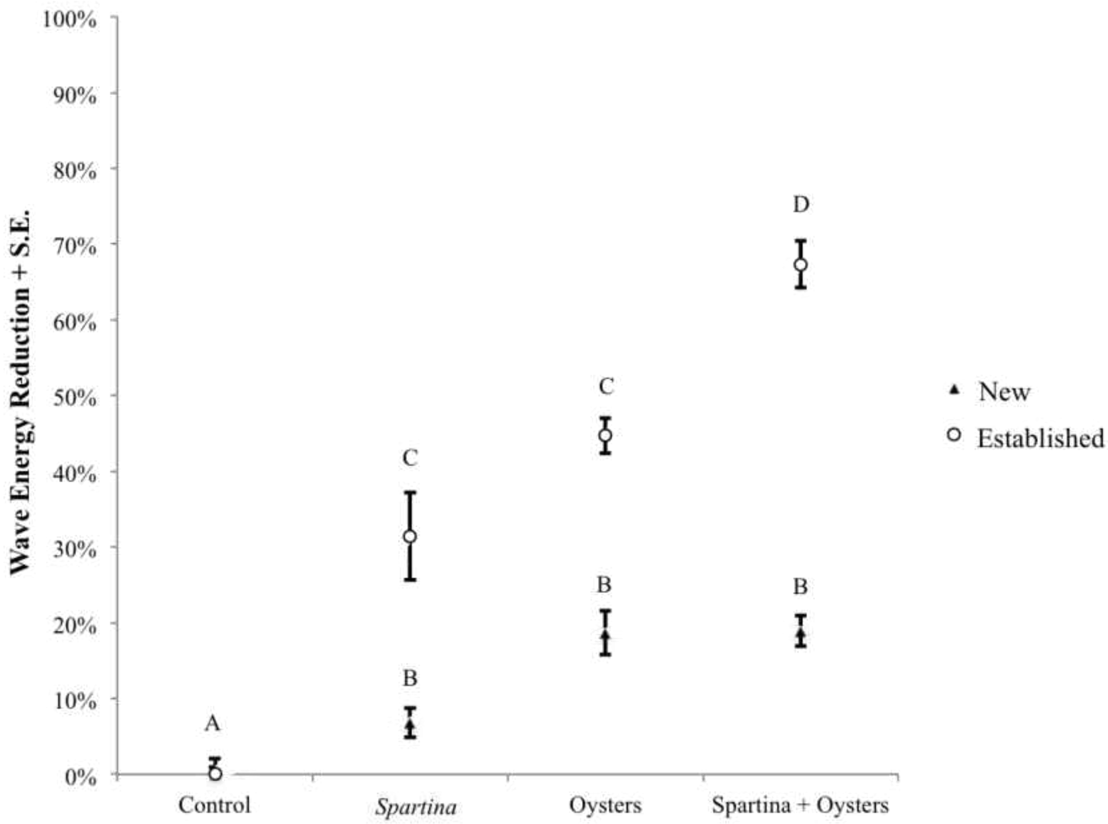

Figure 19 Mean change in wave energy after encountering shoreline stabilization treatment (±SE). Treatments with different letters are significantly different at $p < 0.05$ level as determined by Tukey's post hoc tests.

The total energy contained in one wave created by a boat pass and one entire wave train was calculated using Eq. 1 (Table 8). One wave traveling over the control treatment retained the most energy (mean ± S.E.) at 19.55 ± 3.80 J m^{-2}. The smallest mean wave height occurred after progressing through the one-year old combination treatment (6.32 ± 2.46 J m^{-2}). Boat wake wave trains were compared by multiplying the energy by 10, since the mean number of waves per boat wake is 10. A bare shoreline was impacted by 195.5 J m^{-2} for each boat pass, while a shore with a one-year old combination with *S. alterniflora* and live oysters was only impacted by 63.19 J m^{-2}, 32.7% of the original energy.

Table 8 Mean change in wave energy after encountering shoreline stabilization treatment.

	Wave Energy (J m^{-2}) After Treatment
Control	19.55
Newly planted *S. alterniflora*	18.21
Newly deployed stabilized oyster shells	15.9
Newly deployed *S. alterniflora* and stabilized oyster shell	15.85
1 year established *S. alterniflora*	13.26
1 year live oysters	10.69
1 year established *S. alterniflora* and live oysters	6.32

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 height (Table 9). A coefficient of 1 would mean no energy was absorbed, and a coefficient of 0 would mean all the energy was absorbed (Möller 2006). The control had an attenuation coefficient of 0.99. Newly deployed *S. alterniflora*, stabilized oyster shell, and the combination of *S. alterniflora* and stabilized oyster shell had coefficient values of 0.90 or greater. These values contrasted sharply to the established combination treatment (oysters and *S. alterniflora*), 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 coefficient was from the one-year old live oysters at 0.74.

Table 9 Wave attenuation coefficient after encountering the living shoreline treatments.

	Wave Attenuation Coefficient
Control	0.99
Newly planted *S. alterniflora*	0.96
Newly deployed oyster shell	0.90
Newly deployed *S. alterniflora* and oyster shell	0.90
One-year *S. alterniflora*	0.82
One-year live oysters	0.74
One-year *S. alterniflora* and live oysters	0.57

Discussion

Living shorelines composed of one-year old intertidal oysters plus *S. alterniflora* reduced the amount of wave energy that impacted shorelines by 67% (Figure 19). The increased vertical height and density caused by the recruited oysters, plus the high stem density of the *S. alterniflora* caused this attenuation. The combination of oysters and marsh grass attenuated more wave energy than shorelines containing only oysters or marsh grass. Also, as the living shoreline recruited live oysters and grew new plant biomass, the energy reduction increased.

Oysters have been used in many forms of wave attenuation and shoreline stabilization, from breakwaters, oyster infused reef balls, and oyster mats (Brumbaugh & Coen 2009; Scyphers et al. 2011; Walters et al. 2012). The primary differences among time since deployment (newly deployed vs. one-year old) with our living shoreline treatments were vertical relief caused by the oysters, oyster density, and stem density of the marsh grass vegetation. As oyster larvae recruit to oyster shells, they create a shell of calcium carbonate that adds to the vertical height of the existing reef (Korringa 1952). This was evident when comparing mean (± S.E.) heights of newly deployed oyster shells (6.9 ± 1.48 cm) to shells that had recruited live oysters for one year in the field (15.0 ± 1.63 cm). As waves progress over the intertidal reef, energy is dissipated and attenuated by the three dimensional structure, protecting the landward shoreline (Coen et al. 1999). The established oyster shells attenuated over two times more of the wave energy than the newly deployed shells (Figure 19). This is mostly attributed to the average 8.1 cm of extra vertical height that newly recruited oysters added to the reef. Through wave attenuation, intertidal oyster reefs and breakwaters have been found to reduce erosion, and shoreline loss by 40% (Scyphers et al. 2011).

Spartina alterniflora is commonly used in association with oysters in living shoreline stabilization projects. This is due to where *S. alterniflora* grows in the intertidal zone, the range of *S. alterniflora* along the east and Gulf coast, and its ability to grow in any salinity range from fresh to ocean water (Bush & Houck 2008). This plant is able to attenuate wave energy passing through the emerged and submerged vegetation because the waves lose energy as they move through the stems (Dalrymple et al. 1984; Anderson et al. 2011). As the density of the *S. alterniflora* increased, we found that more energy was attenuated. Comparing the wave attenuation of newly deployed *S. alterniflora* alone (five individual *S. alterniflora* shoots 0.85 ± 0.10 m tall in a 0.25 m^2 area) to the year-old established *S. alterniflora* alone (37 individual *S. alterniflora* shoots 0.89 ± 0.08 m tall in a 0.25 m^2 area) shows that the higher density attenuated over four and a half times more energy (Table 8).

Newly deployed shorelines did not show additive effects of oyster reefs plus *S. alterniflora*. Because the newly planted *S. alterniflora* consisted of only 5 plant stalks, the attenuation effects were small (6.9% energy reduction). 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 the oyster shell attributing 60% of the total and the *S. alterniflora* attributing 40% of the total. The interaction effect of treatment and age caused the one-year old combination treatment (live oysters and *S. alterniflora*) to significantly reduced wave energy more than any other treatment ($p < 0.001$; Table 8).

Using native flora and fauna as a form of soft stabilization can be significantly less expensive than hard armoring (Swann 2008; Grabowski et al. 2012). Grabowski and colleagues (2012) demonstrated that a bulkhead or similar rock revetment cost between $630 and $752 per linear meter. In comparison, a living shoreline consisting of marsh grass and oyster shell can cost as little as $150 per linear meter (Davis & Luscher 2008). Having an average life span of 8-10 years, seawalls and other types of hard armoring need structural maintenance to remain effective over time (Griggs & Fulton-Bennett 1988). A living shoreline stabilization that includes *C. virginica* and *S. alterniflora* can potentially be 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). Not only can living shorelines be substantially less expensive to create than a bulkhead or seawall, ecosystem services are associated with them, including the maintenance of structural habitat, water filtration, nitrogen removal and enhanced foraging grounds for economically important fisheries (Jones et al. 1994; Coen et al. 1999; Harding and Mann 2001).

Our findings suggest that living shorelines composed of intertidal oysters and *S. alterniflora* attenuate a significant amount of wave energy created by the use of recreational boats. Although living shorelines were more effective after one year in the field, they are capable of diminishing some wave energy immediately after deployment. Our results provide additional evidence that soft shoreline stabilization, in the form of living shorelines, can effectively attenuate wave energy. With continued research focusing on the mechanics, effectiveness, and maintenance of living shorelines, landowners and managers should consider this form of soft stabilization when assessing shoreline stabilization alternatives.

CHAPTER 4: LARGE-SCALE LIVING SHORLINE STABILIZATION ON THREE SHELL MIDDENS IN MOSQUITO LAGOON, FLORIDA

Introduction

Over the past sixty years, human impacts have resulted in the destruction and alteration of many essential habitats throughout Mosquito Lagoon, including intertidal oyster reefs (Walters et al. 2002). Areas with vegetation and shallow subtidal/intertidal estuarine ecosystems have been lost, turned into bare and exposed shorelines that are prone to erosion (Walters et al. 2012). This is what has occurred on the shorelines of three archeological sites, known as shell middens, located in northern Mosquito Lagoon: Castle Windy, Garver Island, and Hong Kong. These three middens contain a portion of the only remaining artifacts from the Timucuan Native Americans that lived in this area 800-1200 years ago (Bushnell 1915; Ehrmann 1940). The middens are composed of oyster, clam, and coquina shells, along with broken pieces of pottery and animal bones (Butler 1915).

The shorelines, along with archeological artifacts, are rapidly being lost to shoreline erosion associated with storms, high water events and boat wakes at these three middens. This erosion will only increase with climate change, as storm frequency increases and as the sea level rises. Conserving natural shoreline habitat is necessary for coastal ecosystems to adjust to future climate conditions and preserve essential ecological functions (Erwin 2009). Resource managers and some scientists now agree that the only viable solution for long-term shoreline protection is by utilizing living shoreline techniques.

Results from previous field and lab experiments helped to identify the most effective form of living shoreline stabilization for each shell midden. In the field experiment (Chapter 2), stabilized oyster shell alone accreted the most sediment. The potential additive effects of *S. alterniflora* in the combination treatment were difficult to determine due to *S. alterniflora* die-offs caused by seagrass smothering. In the laboratory experiments, the combination of stabilized oyster shell plus *S. alterniflora* attenuated the most wave energy. Combining our wave tank and field results, a combination of stabilized oyster shell and *S. alterniflora* was used to absorb wave energy and bind shoreline sediments at all three middens, totaling 340 meters of eroding shoreline. To further stabilize areas of the upper-intertidal zone, we planted red mangroves (*Rhizophora mangle*) shoreward of the marsh grass. At Castle Windy, black mangroves (*Avicennia germinans*) were planted along with the red mangroves. The key question for this project was: Using the results gathered from the first two experiments (Chapters 2 and 3), how effective will living shorelines be at limiting erosion on a large-scale in the field?

Materials and Methods

Castle Windy

The first shoreline to be stabilized in Mosquito Lagoon was adjacent (5.4 ± 0.3 m) to the shell midden called Castle Windy. One hundred twelve meters of the shoreline was stabilized with oyster shell, *S. alterniflora*, *R. mangle* and *A. germinans* (Figure 20). To create the living shoreline, 155 red and 121 black mangroves were planted in the upper-intertidal zone (1.5 m wide), 369 *Spartina alterniflora* in the mid-intertidal zone (1 m wide), and placed 1364 oyster restoration mats in the low intertidal zone (2.5 m wide) (Figure 21). This equated to one *S. alterniflora* every 0.3 m, one *R. mangle* every 0.7 m, and one *A. germinans* every 0.9 m. Stabilized oyster shell began at the lower-intertidal zone, and extended seaward 2.5 m (Figure 21). The stabilized oyster was deployed as restoration mats, which consisted of 0.25 m^2 of Vexar^{TM} 1.5 cm mesh with 36 adult oyster shells attached with 50 lb. test cable ties, so the shells were retained in a vertical orientation (Wall et al. 2005). The oyster shells used for the mats were collected and quarantined on land for at least six months to avoid the transfer of diseases and invasive species (Bushek et al. 2004; Cohen & Zabin 2009). Once constructed, the mats were placed on top of the bare sediment in the lower intertidal zone in a quilt-like fashion, where natural recruitment to deployed shells occurred. In two to six years, over 300 live oysters can be counted on one individual mat, or a mean density of 472 live oysters per square meter after 5.5 years (Walters et al. 2013) (Figure 5). This stabilization was completed on April 22, 2012 with the help of over 300 volunteers.

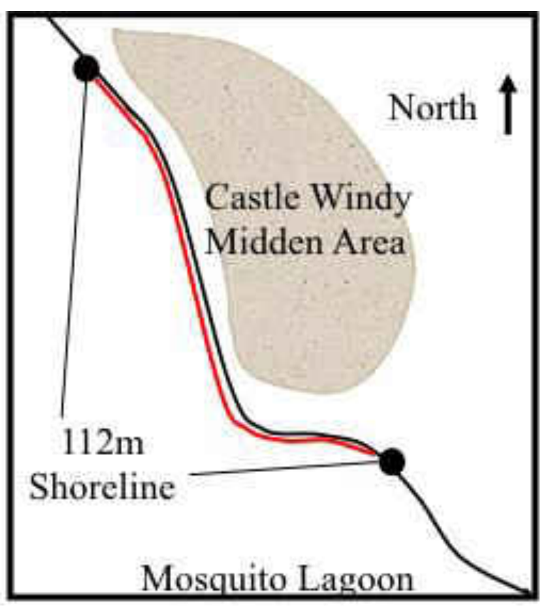

Figure 20 Castle Windy midden area in tan and shoreline stabilization area in red.

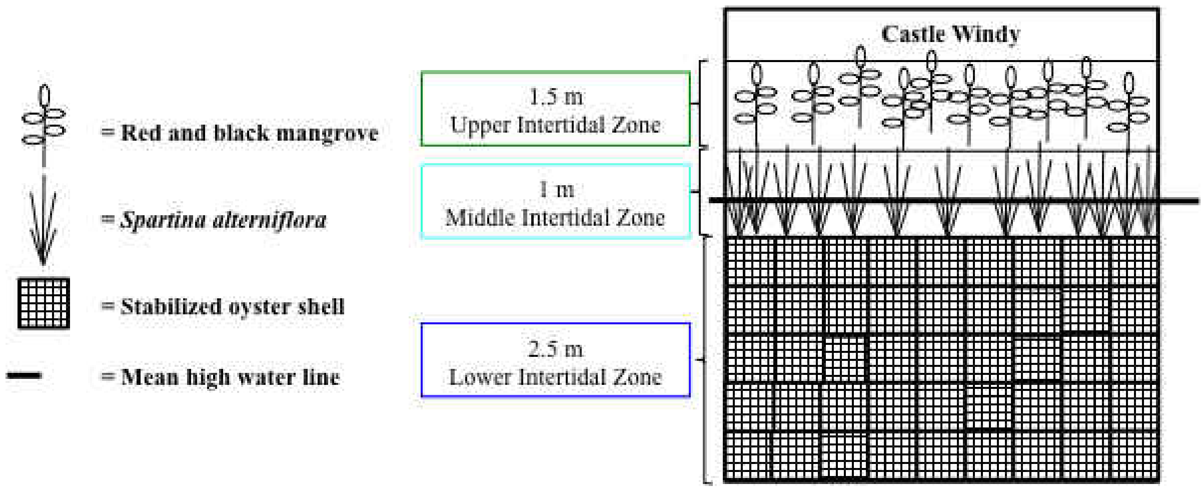

Figure 21 Castle Windy shoreline stabilization plan view.

To determine the effectiveness of the shoreline stabilization, erosion stakes were used to measure changes in sediment height. The stakes were created by drilling five holes at five-centimeter intervals in the topmost 30 cm of a 0.75 m PVC. This allowed for underwater measurements. All stakes were inserted into the sediment with a sledgehammer and aligned to a drilled hole 15 centimeters from the top (middle hole) (Hudson 1957). Fifteen erosion stakes were placed throughout the restoration area, and fifteen were placed in non-restored areas experiencing erosion (control areas) on the shell midden shoreline. Control areas were located outside of the stabilized sections at each midden. At least five meters separated each erosion stake. All shell middens stabilized had erosion stakes inserted immediately after the living shoreline deployment.

Garver Island

There was 150 m of shoreline to be stabilized with oyster shell, *S. alterniflora* and *R. mangle* adjacent (3.4 ± 0.7 m) to the Garver Island midden (Figure 22). To create the living shoreline, we planted 390 red mangroves in the upper-intertidal zone (1.5 m wide), 370 *Spartina alterniflora* in the mid-intertidal zone (1 m wide), and placed 1720 oyster restoration mats in the low intertidal zone (2.5 m wide) (Figure 23). This equated to one *S. alterniflora* plant every 0.4 m, and one *R. mangle* plant every 0.4 m. Stabilized oyster shell was deployed at the lower-intertidal zone, and extended seaward 2.5 m (Figure 23). Stabilization was split into two sections to avoid covering a large area of seagrass (*Halodule wrightii*) found growing close to the shoreline. No black mangroves were planted at Garver Island or Hong Kong due to the long germination and growing periods needed to form large root masses for transplanting. This stabilization was completed on July 12, 2012.

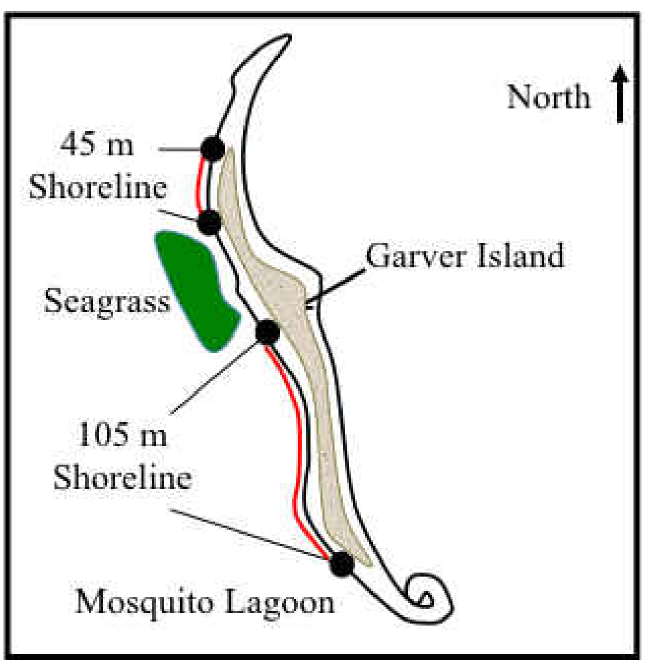

Figure 22 Garver Island midden area, seagrass area, and two shoreline stabilization areas in red.

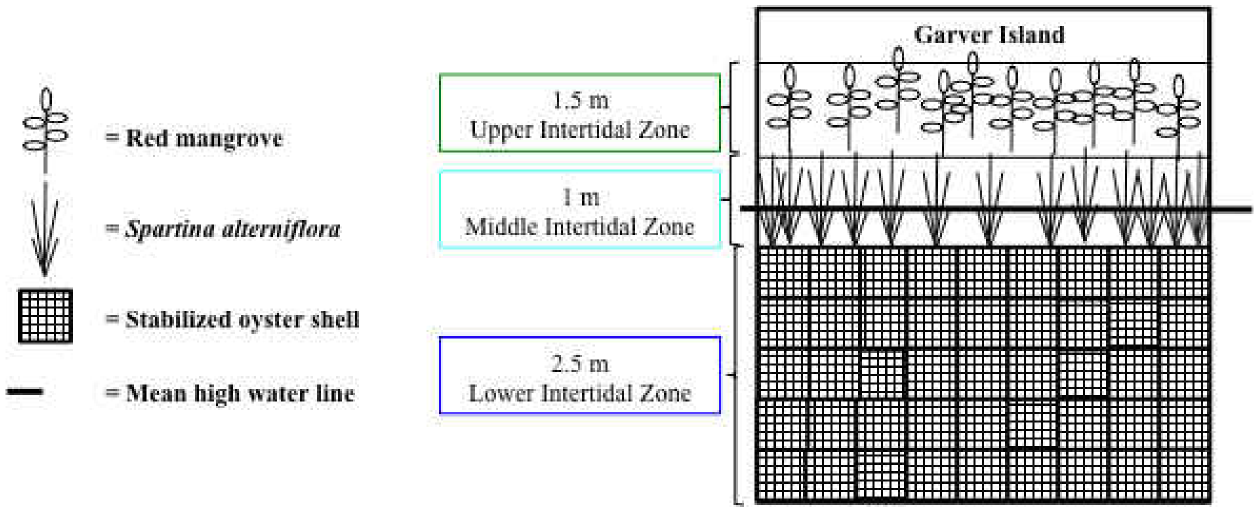

Figure 23 Garver Island and Hong Kong shoreline stabilization plan view.

Hong Kong

The shoreline to be stabilized adjacent (3.2 ± 0.3 m) to the Hong Kong midden was the last area completed. Eighty meters of the shoreline was stabilized with oyster shell, *S. alterniflora* and *R. mangle* (Figure 24). To create the living shoreline, we planted 240 red mangroves in the upper-intertidal zone (1.5 m wide), 240 *Spartina alterniflora* in the mid-intertidal zone (1 m wide), and placed 1150 oyster restoration mats in the low intertidal zone (2.5 m wide) (Figure 23). This equated to one *S. alterniflora* plant every 0.3 m, and one *R. mangle* plant every 0.3 m. Stabilized oyster shell began at the lower-intertidal zone, and extended seaward 2.5 m the same as they do at Garver Island (Figure 23). Stabilization was split into two sections because two small, separate middens are located at Hong Kong. Both stabilization sections were completed on July 15, 2012.

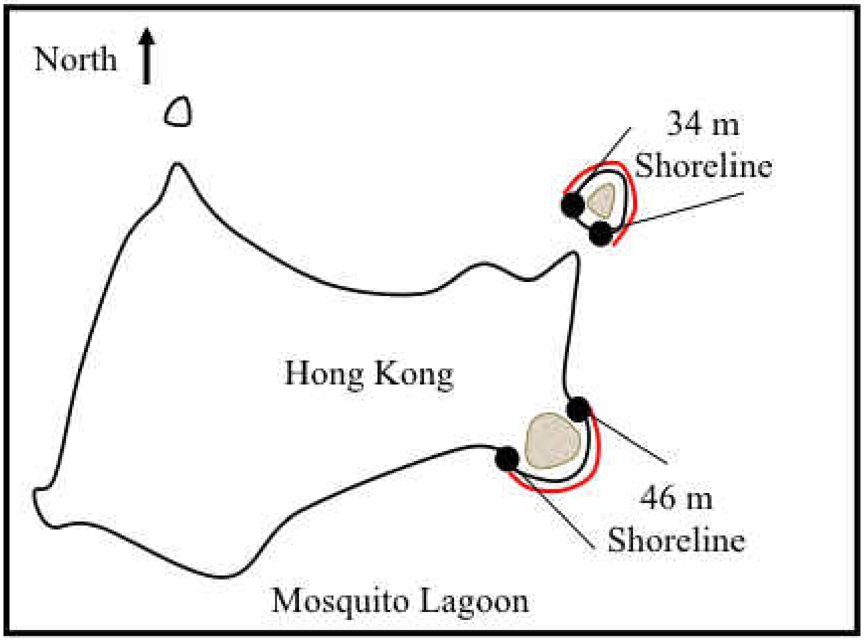

Figure 24 Hong Kong midden area in tan, and shoreline stabilization area in red.

Overall

Plants used in shoreline stabilization had been potted for at least three months for *Spartina alterniflora*, or for at least nine months for the mangroves *Rhizophora mangle* and *Avicennia germinans*. Monitoring of marsh grass, mangroves, oyster mats and erosion stakes commenced immediately post-deployment. Plant survival and new growth was monitored monthly by counting surviving plants, documenting new shoot growth, and by measuring the highest point on each plant. Change in sediment height was monitored at all erosion stakes monthly. Oyster recruitment was documented at six months by randomly choosing 5% of the deployed mats (different for each site) and visually inspecting stabilized shell for recruited live oysters.

Results

Castle Windy

As of January 26, 2013 (nine months after deployment) 26 of the initial 369 *S. alterniflora* plants survived (7.1%) at Castle Windy, with a mean of 12.3 ± 3.1 new shoots for all living *S. alterniflora* plugs. The mean height of surviving *S. alterniflora* at Castle Windy was 57.9 ± 5.6 cm, and 92.3% of the survuving plugs had increased in height since deployment (Table 10). Thirty-four red mangroves survived out of the initial 155 (21.9%) planted, and the mean height for these red mangroves was 50.1 ± 5.0 cm. 84.6% of the surviving mangroves had increased in height since deployment (Table 10). Two black mangroves survived out of the initial 121 planted (1.7%), and the mean height for both black mangroves was 34.0 ± 0.5 cm. Both of the black mangroves had increased in height after nine months (Table 10). Of all the stabilized oyster shell monitored, only three live oysters had recruited.

Table 10 Initial numbers of plants deployed, survival and new growth at Castle Windy.

	Initially Deployed	**Survival**	**New Growth (Increased Height)**
Spartina alterniflora	369	7.1%	92.3%
Rhizophora mangle	155	21.9%	84.6%
Avicennia germinans	121	1.7%	100%

Changes in sediment height were measured nine months after deployment. Within areas that were stabilized, a mean accretion (± SE) of 2.28 ± 0.82 cm occurred. At nearby control areas (no stabilization) erosion stakes showed a mean erosion of -0.22 ± 1.16 cm (Figure 25). The mean of stabilized versus control areas was significantly different when analyzed with a t-test ($p < 0.001$), with restored areas having higher accretion than control areas.

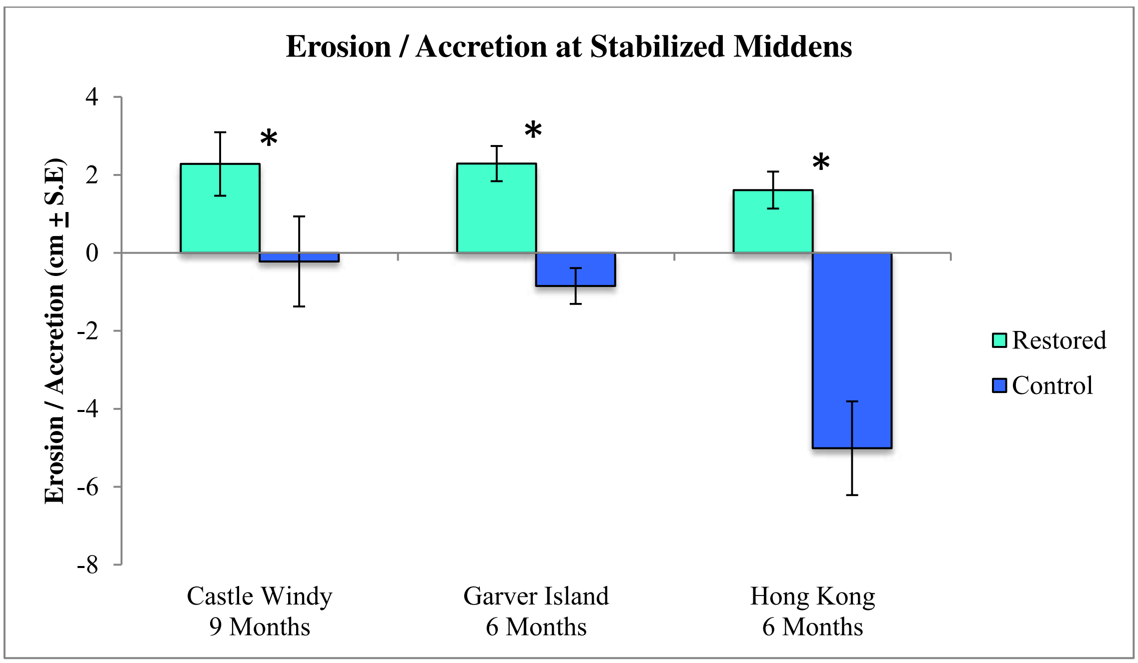

Figure 25 Mean change in sediment height at Castle Windy, Garver Island and Hong Kong as of January 2013 (±SE). Middens with stars were significantly different at $p < 0.05$ level as determined by a t-test.

Garver Island

Six months after deployment, 18 of the initial 370 *S. alterniflora* plants survived since July 2012 (4.9%) at Garver Island, with a mean of 8.1 ± 1.5 new shoots for all living *S. alterniflora* plugs. The mean height of *S. alterniflora* at Garver Island was 64.8 ± 6.3 cm, with

93.7% of the remaining plants increasing in height since deployment (Table 11). Thirty-four red mangroves survived out of the initial 390 (8.7%), and the mean height for red mangroves was 48.3 ± 3.3 cm. 58.1% of the surviving mangroves had increased in height since deployment in July 2012 (Table 11). Of all the stabilized oyster shell monitored, only one live oyster had recruited.

Table 11 Initial numbers of plants deployed, survival and new growth at Garver Island.

	Initially Deployed	Survival	New Growth
Spartina alterniflora	370	4.9%	93.7%
Rhizophora mangle	390	8.7%	58.1%

Changes in sediment height were measured 6 months after deployment. Within areas that were stabilized, a mean accretion (± SE) of 2.29 ± 0.45 cm occurred. At nearby control areas (no stabilization) erosion stakes showed a mean erosion of -0.85 ± 0.46 cm (Figure 25). The mean of stabilized versus control areas was significantly different when analyzed with a t-test ($p < 0.001$), with stabilized areas accreting more sediment than control areas (Figure 25).

Hong Kong

As of January 26, 2013 (six months after deployment), 156 of the initial 240 *S. alterniflora* plants deployed survived (65.1%) at Hong Kong, with a mean of 10.0 ± 4.4 new shoots for all living *S. alterniflora* plugs. The mean height of all surviving *S. alterniflora* at Hong Kong was 70.2 ± 4.9 cm, and 97.6% of the surviving plants increased in height since deployment (Table 12). One hundred sixteen red mangroves survived out of the initial 240 planted (48.5%), and the mean height for red mangroves was 53.8 ± 2.9 cm. 85.3% of the surviving red mangroves had increased in height since deployment (Table 12). Of all the stabilized oyster shell monitored, no live oysters had recruited.

Table 12 Initial numbers of plants deployed, survival and new growth at Hong Kong.

	Initially Deployed	**Survival**	**New Growth**
Spartina alterniflora	240	65.1%	97.6%
Rhizophora mangle	240	48.5%	85.3%

Changes in sediment height were measured 6 months after deployment. Within areas that were stabilized, a mean accretion (± SE) of 1.61 ± 0.48 cm occurred. At nearby control areas (no stabilization) erosion stakes showed a mean erosion of -5.01 ± 1.20 cm (Figure 25). The means of stabilized areas versus control areas were significantly different when analyzed with a t-test ($p < 0.001$), with stabilized areas accreting more sediment than control areas.

Discussion

In order to protect Native American archaeological sites within CANA, living shoreline stabilizations containing stabilized oyster shell, *S. alterniflora*, *R. mangle*, and *A. germinans* were used. Three separate middens, Castle Windy, Garver Island and Hong Kong were stabilized with native flora and fauna to attenuate wave energy and to accrete sediment. Overall, the stabilization has been effective. While each midden experienced different results, overall, areas with the stabilization accreted sediment while non-stabilized areas were experiencing erosion.

Plant survival was variable at each shell midden, with Hong Kong having the highest survival percentages (Table 12). Seagrass wrack played a role in initial plant survival, with the dead seagrass washing over the newly deployed plants and smothering them. Studies completed in the Indian River Lagoon system have identified decline and recovery periods of seagrass (Morris & Virnstein 2004). After seagrass beds accumulate thick layers of organic detritus (10-15 cm) and populations appear to crash, and recover every two to three years (Morris & Virnstein 2004). Since Mosquito Lagoon is a poorly flushed system with slow water movement, it is possible that such an event occurred the summer of our stabilization (Fletcher & Fletcher 1995). This would help explain the large amounts of seagrass wrack found floating in the lagoon, and on the shorelines. The death of seagrass would also help to explain the low oyster recruitment to these middens. Large amounts of dead seagrass trigger hypoxic conditions and, cyanobacteria blooms, which have been documented to limit oyster survival and recruitment in Florida estuaries (Phlips et al. 1999). On top of the seagrass die-offs, a brown algal bloom (*Aureoumbra lagunensis*) impacted Mosquito Lagoon in the summer of 2012 (FWC 2013). It is possible that the bloom affected *S. alterniflora* and mangrove survival, along with oyster recruitment (FWC 2013). While the stabilized oyster shell recruited other invertebrates such as barnacles and tunicates, only four live oysters were found on our restoration substrates. All middens will have more *S. alterniflora* and *R. mangle* planted in stabilized areas to replace lost vegetation during the spring and summer of 2013.

As water quality increases over time with the passing of the brown algal bloom, plant survival and oyster recruitment should increase. Also, older, taller mangrove plants are being used to allow survival through seagrass wrack. Through the entire process of field experiment, lab experiments and shoreline restoration, we have documented that living shoreline stabilizations cause sediment accretion. By creating intertidal habitat where there was once only bare sediment shoreline, we have stopped shoreline erosion and prevented the national park from having to use a hard stabilization technique. We hope that CANA continues to utilize this form of stabilization, and that other parks and landowners facing similar issues will follow our lead in using soft stabilization.

CHAPTER 5: GENERAL DISCUSSION

Living shorelines are theoretically a long-term, sustainable solution to coastal erosion issues in estuaries and bays (Piazza et al. 2005; O'Riordan et al. 2006). Pilkey et al. (2012) found that many “living shorelines” are, however, often improperly classified and frequently are hard stabilizations with some type of vegetation planted landward (Figure 26). To be correctly labeled as a living shoreline, a stabilization technique must fulfill three main requirements: 1) control erosion by mimicking the natural coastal processes of the area, 2) maintain ecosystem services, and 3) allow for migration of intertidal habitat in response to sea level rise (Pilkey et al. 2012). Hard shoreline stabilizations are continually being installed, up to 30 miles per year in some states (Pilkey et al. 2012). Research on the effectiveness of true living shorelines, such as this study, is necessary to inform land managers of the ability to protect coastlines without destroying intertidal habitat.

Figure 26 A "living shoreline" that is actually a hard stabilization (concrete rip-rap) with *Spartina alterniflora* planted landward in Edgewood, Maryland.

Here, our living shoreline is compared to Pilkey's three criteria (Pilkey et al. 2012). The living shoreline stabilization treatments tested in Mosquito Lagoon and in the wave tank mimicked local natural shorelines and did not appear to change any coastal processes. Combining two native intertidal species (*C. virginica* and *S. alterniflora*) to soft-armor shell midden shorelines, the natural assemblage found on non-eroding shorelines elsewhere in the lagoon was restored. The introduction of foreign material with hard stabilizations such as chemically treated wood, highly acidic concrete and metal can have detrimental consequences to invertebrates, macroalgae and benthic fauna in the area (Eisler 1989). Also, as waves hit a hard structure such as a seawall or bulkhead, the energy is reflected back into the water column with little energy loss causing increased erosion rates on adjacent shorelines (Pilkey and Wright 1988; Pilkey 1991; Bozek & Burdick 2005; Bilkovic & Roggero 2008). The change in water motion and the increase in wave energy seaward of a hard stabilization changes sedimentation and accretion for areas up to 150 m down current of the seawall (Griggs & Tait 1988).

This study and additional studies have shown that wave attenuation is increased over live oysters and oyster shell, which in turn causes sedimentation and accretion landward of the reefs and along shorelines (Coen et al. 2007) (Figure 19). *Spartina alterniflora* also absorbs wave energy by increasing surface friction over a salt marsh habitat (Möller et al. 1999). Our wave tank study determined that a newly deployed living shoreline containing *C. virginica* and *S. alterniflora* attenuated 19% of the energy from boat wakes (Figure 19). After recruiting live oysters and producing additional plant biomass for one-year, this form of living shoreline absorbed 67% of the energy from boat wakes. By minimizing the impact of anthropogenic waves (boat wakes), this form of living shoreline maintained the local natural flora and fauna assemblage, all while accreting up to 4.9 cm of sediment a year.

Having met the first criterion to be considered a living shoreline, the next requirement is that this stabilization maintains the local ecosystem services historically provided by the location. Placing a seawall, bulkhead, or rip-rap on a shoreline would instantly remove some portion of the original intertidal habitat through installation alone (Peterson et al. 2000). These forms of hard armoring would also sever the land-water interface between the intertidal and supratidal habitats (Peterson et al. 2000). Using *C. virginica* to attenuate wave energy, the shells provide necessary structural complexity for crabs, shrimp and juvenile fishes, all while providing preferred settling areas for oyster larvae, and many other sessile invertebrates (Beck et al. 2001, Heck et al. 2003; Bilkovic & Roggero 2008). Intertidal oyster reefs within Mosquito Lagoon have been documented to provide habitat for 140 species (Barber et al. 2010). Oysters not only support species directly through structure, but also through water filtration and sediment fertilization, giving them an estimated ecosystem service valued at $40,000 per acre per year (Peterson & Heck 2001a, b; Booth & Heck 2009; Grabowski et al. 2012). Several species of seagrass depend on oyster reefs for nutrients and water clarity, allowing them to grow in low nutrient, deeper areas (Haven & Morales-Alamo 1966; Riisgård 1988; Booth & Heck 2009). *Spartina alterniflora* dominated marshes are the backbone to many fisheries, either through detritus production for food, or as foraging grounds for predatory fish (Boesch & Turner 1984). This fringing marsh grass is important for absorbing terrestrial nutrient run-off such as nitrogen and phosphorous before it enters the aquatic system (Broome et al. 1975; Johnston et al. 1984).

This marsh grass is valued at $20,058 per hectare per year in ecosystem services, which includes its value of stabilizing shorelines (Qin et al. 1997).

The last step to being classified a living shoreline is the ability of intertidal habitat to migrate landward as the sea level rises (Pilkey et al. 2012). If there is no intertidal zone because it has been covered with concrete, or eroded away due to the presence of a seawall or bulkhead, then it is not a living shoreline. Once the sea level rises to a point where waves overtake a hard structure, then it is no longer able to prevent shoreline erosion. The living shoreline stabilization tested in Mosquito Lagoon maintains flora and fauna on the shoreline, allowing sediment and aboveground organic material to be trapped (Redfield 1965, 1972; Gleason et al. 1979; Yang 1998). Multiple studies have shown that shoreline areas containing native vegetation, such as mangroves and marsh grasses, have higher accretion rates than bare shorelines under the same erosion pressures (Gleason et al. 1979; Cahoon and Lynch 1997; Morris et al. 2002; Kumara et al. 2010). Some vegetation, such as *S. alterniflora* in the intertidal zone, has been shown to elevate shorelines by producing underground biomass, and by disrupting the water movement enough to allow for sediment deposition (Gleason et al. 1979; Yang 1998; Morris et al. 2002; Cahoon et al. 2004). Thus, the living shoreline treatments we tested could potentially track sea level rise by accumulating sediment and organic matter, and also allow the marsh grass and oysters to migrate shoreward as water levels increase (Orson et al. 1985; Reed 1995).

Our findings suggest that living shorelines composed of *C. virginica* and *S. alterniflora* attenuate a significant amount of wave energy produced from boat wakes. The results from this study provided strong evidence that living shorelines can effectively stabilize shell middens at CANA. Due to the success in sediment accretion and wave attenuation documented in our experiments, park managers chose living shoreline stabilizations with *C. virginica* and *S. alterniflora* to armor all eroding areas of three shell middens. As invertebrates recruit to the oyster shell, and marsh grass grows over time, the stabilization should become even more effective. CANA land managers will continue to utilize living shorelines to stabilize eroding shorelines within the park.

Both *C. virginica* and *S. alterniflora* have large geographic ranges, therefore this type of living shoreline has broad management applications in any estuary or bay with naturally occurring intertidal oysters and marsh grass. Depending on the wave energies found in the system, the techniques for deploying oyster shell and marsh grass could vary. Loose oyster shell, or bags filled with shell could attenuate different amounts of wave energy depending on the shoreline slope. Larger areas of marsh grass could also be planted based on the bathymetry of the eroding system. Even though the techniques will vary from location to location, the overall goal of wave attenuation while maintaining habitat remains. As the research associated with the effectiveness of living shorelines increases, we hope to see more landowners and land managers utilize this form of soft stabilization to protect shorelines.

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