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Stephanie Garvis
University of Central Florida

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QUANTIFYING THE IMPACTS OF OYSTER REEF RESTORATION ON OYSTER COVERAGE, WAVE DISSIPATION AND SEAGRASS RECRUITMENT IN MOSQUITO LAGOON, FLORIDA

by

STEPHANIE K. GARVIS
B.S. University of Central Florida, 2009

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Major Professor: Linda J. Walters

ABSTRACT

The goal of this project was to determine the effects of oyster reef restoration on oyster coverage, wave height dissipation and seagrass recruitment. First, to assess the current versus historical coverage of natural, dead and restored oyster reefs within Mosquito Lagoon, aerial photographs from 2009, provided by Saint Johns River Water Management District, were digitized using ArcGIS software. Live reefs, restored reefs and dead reefs were screen digitized using a reef 'signature' in order to estimate the area of each type of reef. The 2009 maps were used as a guide to digitizing the historical aerial photographs (1943, 1951, 1967, 1971, 1984, 1995, 2006). Dead reefs increased both in number and aerial extent during the study period (1943 – 2009), with 2009 having over 10 acres of dead reef coverage. Dead reefs were more likely to be found along major boating channels. Several dead reefs exhibited migration into the mangrove islands located landward of the 1943 footprint, with some dead reefs completely washing up into the shoreline. Restoration of dead reefs added 0.5 acres of live oysters to the Mosquito Lagoon area as of January 2009.

Second, I examined how different oyster reef types (natural reef, restored reef, dead reef) reduced wave height. To determine wave height attenuation on each reef type, experiments were conducted in a 9 meter long wave tank using sensors that measured changes in wave height. For each reef type, replicate reefs were created in the wave tank. Shoreline sediment without oysters was used as a control. Using the wave generator, wave heights similar to Mosquito Lagoon boat wakes were created. Restored reefs reduced the incoming wave height by 25% compared to sediment without oysters.

Lastly, I examined the potential link between oyster reef restoration and recruitment of seagrass fragments. Monthly surveys were performed to quantify the number of seagrass fragments encountering the three oyster reef types: dead reefs, natural reefs and restored reefs. The quantity of seagrass fragments was found to be similar on the three different reef types, but did show a significant trend of seasonality, which corresponds with the growing season of *Halodule wrightii*. Next, I tested retention of experimentally manipulated seagrass fragments on five natural and five restored reefs. Restored reefs retained seagrass fragments for significantly longer than natural reefs. I also measured seagrass fragment entanglement on each reef type inside the wave tank. I found that seagrass fragments were significantly more likely to become entangled and retained on restored reefs compared to dead and natural reefs. Overall, these metrics are important for determining the success of long-term oyster restoration project in Mosquito Lagoon, Florida. This project has found that oyster restoration is increasing the area of oyster habitat as well as providing important ecosystem services.

To Kyle Dennis and Buster Brown,

Love you guys!

ACKNOWLEDGMENTS

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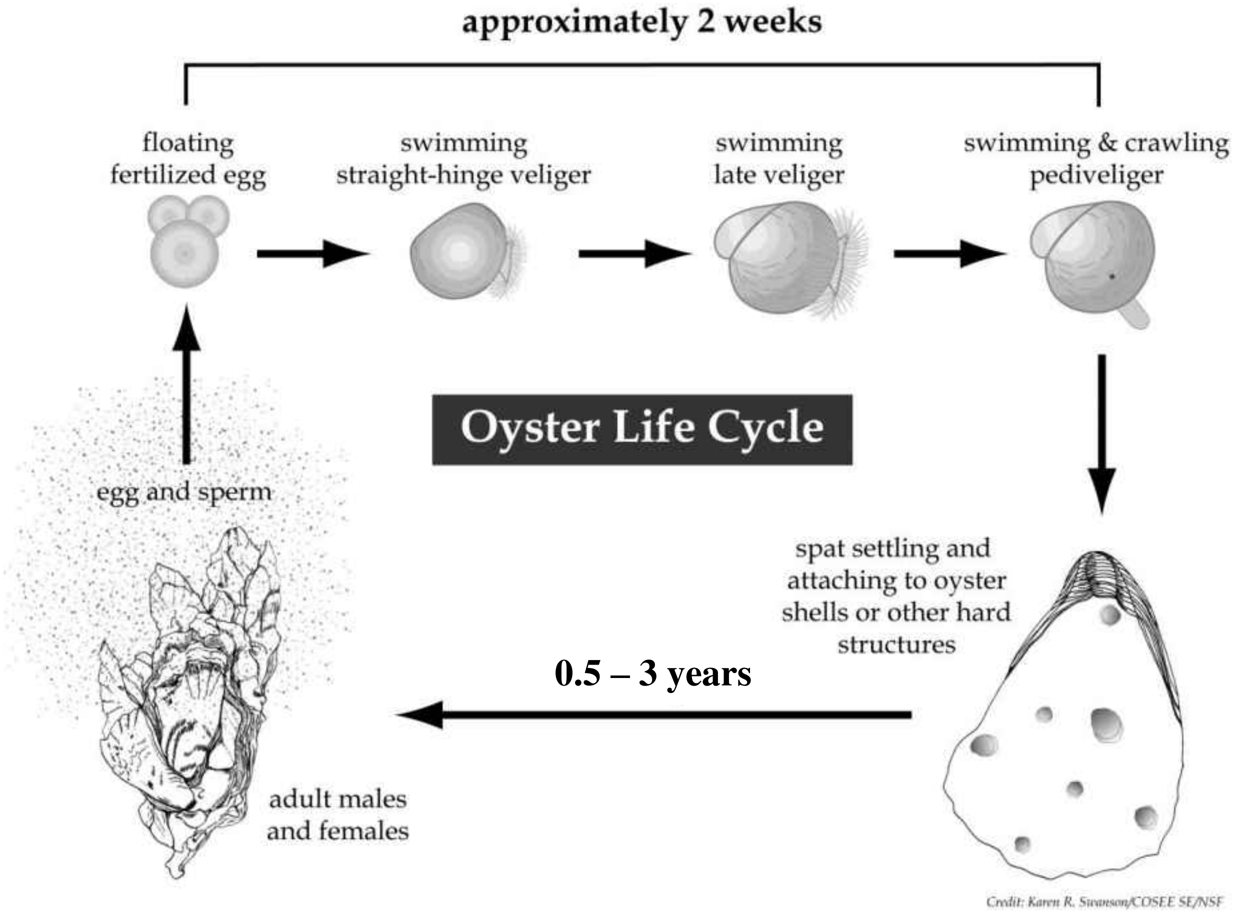
CHAPTER 1: GENERAL INTRODUCTION

Biology of *Crassostrea virginica*

Kingdom	Phylum/Division:	Class:	Order:	Family:	Genus:	Species:
Animalia	Mollusca	Bivalvia	Ostreoida	Ostreidae	*Crassostrea*	*virginica*

In the western Atlantic Ocean, *Crassostrea virginica* has been reported from the Gulf of Mexico up to the Gulf of St. Lawrence in Canada, and can also be found in Argentina and Brazil (Andrews 1991). The southern-most point of the range of expansive *C. virginica* reefs on the Atlantic coast of the United States is found within the Indian River Lagoon, along the east coast of Central Florida (Grizzle & Castagna 1995). *Crassostrea virginica* has an extensive geographical distribution, in part, because it possesses broad salinity and temperature tolerances (Gunter & Geyer 1955). Mature oysters typically thrive at salinities from 10 to 30 ppt (Gunter & Geyer 1955). *Crassostrea virginica* can also tolerate a wider range of salinities, from 2 to 40 ppt, for short periods of time (Gunter & Geyer 1955). The temperature tolerance of oysters is between 10º and 43º C, but their larvae can only develop at temperatures around 20º C (Burrel 1986).



Credit: Karen R. Swanson/COSEE SE/NSF

Figure 1 The life cycle of *Crassostrea virginica*.

Crassostrea virginica reproduces via external fertilization (Burrell 1986, Figure 1). Spawning occurs as a result of environmental cues in the water and the occurrence of conspecific gametes nearby (Kennedy et al. 1996). After spawning, planktotrophic larvae begin to grow and can reach the trochophore phase 12 hours after fertilization (Kennedy et al. 1996). These trochophore larvae will then progress into the veliger stage after approximately 48 hours (Burrell 1986). Once in the veliger form, the larvae spend 2 to 3 weeks feeding while moving with the water currents (Bahr & Lanier 1981). Then, the larva proceeds to the pediveliger stage, which has two eyespots and a foot, enabling the larvae to find suitable substrate (Burrell 1986). When in this stage, larvae can respond to multiple stimuli, including physical factors and chemical cues (Tamburri et al. 1992). In order to settle on a suitable substrate, larvae react to specific chemicals released from adult oysters (Burrell 1986). The larvae respond by moving down the water column, towards the substrate (Tamburri et al. 1992). When an oyster larva finds a suitable substrate, most often a conspecific shell, it then cements itself and remains there permanently (Burrell 1986).

Ecosystem Services of Oysters

Crassostrea virginica has been classified as an ecosystem engineer as well as a keystone species because of the important benefits that oysters and oyster reefs provide (Coen et al. 2007). Due to their three-dimensional structure, oyster reefs are able to maintain high levels of biodiversity (Coen & Luckenbach 2000). It has been shown that oyster reefs can support more fauna than other types of bottom cover (Nelson et al. 2004). More than three hundred different species of fauna are associated with oyster reefs (Wells 1961). Many economically and ecologically important species can be found using the oyster reefs, including shrimp, stone crabs, blue crabs, and spotted sea trout (Walters et al. 2001, Coen & Grizzle 2007, Barber et al. 2010, Stunz et al. 2010). Certain fish species also use oyster reefs as nesting areas (Coen & Luckenbach 2000). Several species of birds, including pelicans, bald eagles and wood storks, utilize oyster reefs for foraging (Walters et al. 2001).

Oysters also influence the surrounding phytoplankton community, which in turn has an effect on water clarity (Riisgård 1988). Oysters affect water clarity by filtering large amounts of water, up to 6.8 liters per hour per oyster (Riisgård 1988, Newell & Langdon 1996). After filtering out particles, oysters then reject what they cannot use as pseudofeces (Newell & Langdon 1996). Pseudofeces consists of mucus-bound aggregates that can settle almost 40 times faster than particles that are not mucus-bound (Newell et al. 2002).

Previous research has demonstrated that oysters can have an ecosystem-wide scale effect on water clarity (Newell 1988, Mann 2000, Newell & Koch 2004). Many studies have shown that oyster reefs can change the energy flow rates of waves, resulting in an indirect influence on resuspended sediments in the water column (Dame et al.1992, Dame 1996). Oysters not only support species richness, but they act as soft-armor for shorelines (Piazza et al. 2005). In doing so, reefs absorb wave energy which helps to preserve emergent vegetation and stabilize sediments along shorelines (Coen et al. 2007).

Oyster Declines

It has been estimated that 85% of shellfish reefs have been lost worldwide (Beck et al. 2011). Oyster coverage has decreased for several reasons: habitat destruction, disease, overharvesting and reduced water quality (Wenner et al. 1996, Coen et al. 1999, Kirby 2004, Johnson et al. 2009). Overharvesting has caused oyster stock to drop. In the United States, oyster reefs produced over 160 million pounds of meat per year between 1880 and 1910 (Coen & Luckenbach 2000). Due to severe overharvesting, the supply of oysters in the United States decreased almost 60% from around 80 million pounds of meats in the early 1950s to less than 33 million pounds in 2006 (Beck et al. 2011).

Oyster Restoration Efforts

The severe loss of *C. virginica* has encouraged different types of restoration efforts throughout its native range. Restoration focuses on providing substrate for larval settlement and, if needed, increasing larval abundances by introducing additional spat, either wild or cultivated. Spat prefer conspecific substrate, but other substrates that have been used for restoration include limestone marl, cement, and clam shells (Ertel & McCall 2005). The simplest form of restoration is dumping the new substrate material directly on the seafloor. Another technique is the use of mesh bags filled with oyster shells, which is more similar to the three-dimensional shape of a pristine reef than simply dumping shells on the benthos (Ertel & McCall 2005). These techniques have been somewhat successful for areas suffering from overharvesting or a disease outbreak. However, in Mosquito Lagoon, boat wakes are a primary cause for oyster reef degradation. A restoration technique was developed to help address this particular problem using oyster restoration mats. This new type of oyster restoration involves aquaculture grade mesh mats that measure 0.25 meter square in size (Figure 2). The mats have 36 oyster shells firmly attached to them using zip-ties so that the shells are perpendicular to the benthos, to maximize larval recruitment (Wall et al. 2005). Then the mats are deployed in a quilt-like fashion on top of a raked-down dead reef, and are held in place with circular weights, which helps the mats resist the water motion created by boat wakes (Figure 2).

Figure 2 Image of oyster mat (left) and several deployed oyster mats in Mosquito Lagoon (right).

Biology of *Halodule wrightii*

Kingdom:	Phylum:	Class:	Order:	Family:	Genus:	Species:
Plantae	Tracheophyta	Angiosperm	Najadales	Cymodoceaceae	*Halodule*	*wrightii*

In the Indian River Lagoon system, the diversity of seagrasses is very high as a result of its location within a transition zone between subtropical and temperate zones (Steward et al. 2006). The Indian River Lagoon has the most species of seagrass of any estuary in the US, which includes *Halophila johnsonii, Halodule wrightii, Halophila engelmannii*, *Syringodium filiforme, Halophila decipiens, Thalassia testudinum,* and *Ruppia maritima* (Steward et al. 2006).

Halodule wrightii is the predominant seagrass species in Mosquito Lagoon (Hall et al. 2001). It is also found in a broad range of coastal habitats including the western Atlantic coast as far as North Carolina, Gulf of Mexico, the Caribbean, South America, northwestern Africa, the Indian Ocean and the Pacific coast of Mexico (Eiseman & McMillian 1980). *Halodule wrightii* occupies a central role as a pioneer species by colonizing disturbed areas of sediment, which facilitates the establishment of other species of seagrass (den Hartog 1971). *Halodule wrightii* is a pioneer species due to the fact that it can endure larger variations in water depth, salinity, and clarity than other seagrass species (Sargent et al. 1995, Dunton 1996). On the Atlantic coast of Florida, *H. wrightii* occurs most frequently at a depth range of 0.35 – 0.65 meters (Bortone 2000). This species will grow on substrate that ranges from sand to mud, typically containing less than 6% organic composition (Hemminga & Duarte 2000). *Halodule wrightii* needs approximately twenty percent direct light penetration to thrive (Steward et al. 2006).

Halodule wrightii grows fastest in spring and summer months, and then becomes dormant in late fall and winter (Phillips 1960). Most seagrass bed maintenance and new shoot production occurs via rhizome elongation (Phillips 1960, Hemminga & Duarte 2000, Marbá et al. 2006). Reproduction via seeds in *H. wrightii* is considered to be infrequent or rare (Phillips 1960, Eleuterius 1971). This species typically colonizes new areas by vegetative fragments (Hall et al. 2006a). A study conducted in Mosquito Lagoon found that these fragments can remain viable during spring for up to 4 weeks, which is important for dispersal over long distances (Hall et al. 2006a). These fragments have been shown to be successful in settling to the sediment level and subsequently rooting themselves within the duration of viability (Hall et al. 2006a). Hall et al. (2006a) found that *H. wrightii* was more effective at vegetative fragment recruitment in spring compared to fall. The potential dispersal distance of fragments also depends on factors such as tidal currents and wind (Hall et al. 2006a).

Ecosystem Services of Seagrasses

Seagrass beds produce large amounts of organic material and provide substrate for diatoms, epiphytic algae, as well as some fauna (McRoy & Helfferich 1977). Seagrasses provide an important energetic relationship between terrestrial and marine systems because approximately 25% of their net production is exported to surrounding ecosystems (Duarte & Cebrián 1996, Duarte 2002). The beds also act as sediment traps, causing suspended inorganic and organic particles to settle into the sediment (McRoy & Helfferich 1977). Friction from seagrass blades diminishes wave velocity, thereby reducing the impact of wave motion on the sediments (Rybicki et al. 1997). This decrease in wave energy helps to improve the recruitment of marine organisms (Eckman 1983, Peterson 1986). Seagrass beds are habitat for a rich fauna assemblage, including echinoderms, gastropods, bivalves, pelecypods, annelids, and copepods (McRoy & Helfferich 1977). They also directly provide food to herbivorous grazers, such as certain fish species, sea urchins, marine turtles, waterfowl and manatees, and they also supply a source of prey for other species (McRoy & Helfferich 1977, Hemminga & Duarte 2000). Additional species feed on the epiphytic algae on the leaves on the seagrass (McRoy & Helfferich 1977). Seagrasses play an important role in primary productivity and nutrient cycling, and have a profound effect on the chemical dynamics of the surrounding coastal system (Hemminga & Duarte 2000). Due to the fact that seagrasses serve as nurseries, habitat, and food sources for economically and ecologically significant fauna and flora, seagrasses are a very important constituent of estuarine ecosystems (Zieman 1982).

Seagrass Declines

Globally, the total known area of seagrass coverage has decreased 29% since 1879 (Hughes et al. 2009). From 1943 to present, the quantity of seagrass in the Indian River Lagoon system has decreased on average 18%; some areas have experienced losses of up to 100% of historical seagrass cover (Virnstein et al. 2007). As a result of seagrass declines, estuaries are considered one of the most degraded ecosystems (Beck et al. 2001). Unfortunately, most of these ecosystem declines are from direct and indirect human impacts (Erftemeijer & Lewis 2006). The indirect causes that contribute to seagrass declines are natural disasters, global climate change, and other widespread effects, such as nutrient runoff (Erftemeijer & Lewis 2006).

In addition to these stresses, human activities are adding to the pressure on seagrass beds by way of hydro-morphological changes in the seagrass environment. These include dredging, benthic trawling, fishing activity, sediment disturbance, ship anchoring and scouring, beach replenishment; building sea walls, groins, or breakwaters; industrial expansion, land reclamation, harbor construction, cooling water discharge, waste dumping, sewage and storm water discharge (Foden & Brazier 2007). In Florida, boat propellers have scarred many seagrass beds, and this number is increasing rapidly (Grablow 2008). Scars recolonize by *H. wrightii* very slowly and sometimes not at all (Grablow 2008). Currents and tides can move sediments around, causing the scars to deepen, which makes recolonization more difficult (Kirsch et al. 2005). These trenches are typically void of new seagrass growth without some form of assistance or restoration (Kirsch et al. 2005).

Another major factor in seagrass declines is changes in water quality (Koch 2001). Water quality refers to the chemical makeup of water, which is being altered as a result of human activity. One example of this is the accumulation of heavy metals in eelgrass (Austen & McEvoy 1997). Tributyltin, an antifouling agent, was banned, but then boat manufacturers started to employ herbicides instead (Chesworth et al. 2004). As a result, herbicides are often present in the waters and sediments of coastal areas and they can cause seagrass die-offs (Thomas et al. 2001). The chemicals inhibit seagrass growth as well as photosynthetic activity (Chesworth et al. 2004).

Nutrient runoff and subsequent enrichment in coastal and estuarine areas also have a negative impact on surrounding seagrass beds. In certain temperate areas, *Zostera Marina* habitat has decreased as a result of excessive nitrogen loading due to runoff (Hauxwell et al. 2003). Not only does the enrichment cause direct seagrass death, but it also enhances the growth of opportunistic algal species, such as *Chaetomorpha, Ectocarpus* and *Enteromorpha,* which can lead to the deterioration or elimination of the bed via smothering (Dennison et al. 1993). This decrease is not simply the consequence of augmented nitrogen amounts from nitrate and ammonium, but also the result of shading by algal coverage (van Katwijk et al. 1997).

Light penetration is one of the most significant limiting factors for seagrass growth (Kemp et al. 1983, Dennison et al. 1993, Koch 2001). Seagrass meadows are often restricted to shallow waters, where desiccation due to low tide (Koch 2001) or wave exposure is the limiting factor for survival (Koch et al. 2006). Studies show that even small decreases in light availability can result is major declines of both growth and distribution (Ralph et al. 2006), with diminishing survival rates at seston concentrations higher than roughly 15 mg L^{-1} (Batiuk et al. 1992). Temporary changes in water transparency occur due to increases in TSS (total suspended solids), burial due to bioturbation and deposition, and the removal of substrate as well as associated plants from the bed itself (Erftemeijer & Lewis 2006).

Conservation and Restoration Strategies for Seagrass

In response to the global crisis facing seagrass meadows, many areas have implemented detailed conservation plans to offset or prevent human impacts of the seagrass ecosystem. Legislation has been passed in many places to designate certain areas as marine protected areas (MPAs). These MPAs bar human activities to give protection to marine resources and help support sustainable fisheries (Green & Short 2003). While protection is rarely designated exclusively for seagrass beds, they are listed as a key species within the protection area, if present (Green & Short 2003). Seagrasses are not taxonomically diverse (den Hartog 1971), so protecting the small number of species that exist is important because of the ecosystem services provided exclusively by seagrass (Green & Short 2003). By losing just one species of seagrass, a cascade effect could occur that would severely impact the surrounding coastal ecosystem (Hemminga & Duarte 2000).

The loss of seagrass species can affect the chemical makeup of nutrient cycles and food webs within coastal systems (Bach et al. 1986). These processes are important, and when lost, can lead to economic problems such as the collapse of fisheries, the decline of other living resources, and a reduction in water quality (De Jonge & De Jong 1992). One problem that faces the conservation of seagrass is due to the fact that seagrasses may occur on community owned lands and may cross national or international boundaries, it is difficult to successfully implement a conservation plan that all involved parties will obey (Green & Short 2003). Another problem is that limited research is available to managers on current seagrass declines and the possible reasons for these declines (Green & Short 2003). Scientists need to focus on providing information that addresses the conservation needs of seagrass ecosystems (Green & Short 2003).

Most methods of seagrass restoration involve the transplantation of seagrass plugs from existing beds to the new site (Hemminga & Duarte 2000). An attempt in Western Australia involved harvesting large sods (0.25 m^2), which was made possible by utilizing a machine transplanting system (Paling et al. 1998). The disadvantage of techniques like this is that holes are formed in the bed of the donor site, and as a result, the donor site becomes highly susceptible to erosion even when the holes are filled (Hemminga & Duarte 2000). Another technique that has proven to be successful is using garden staples to hold down individual fragments of seagrass that contain a growing tip (Fonseca et al. 1988). This technique involves digging up fragments of seagrass and attaching the section of rhizome to a garden with a twist-tie, then planting in desired location (Fonseca et al. 1988). Using garden staples has been shown to be cheaper and less time-consuming than the sod method (Fonseca et al. 1988). Also, by using fragments, the impact to the original donor bed is minimized (Hall et al. 2006a, 2006b). In areas that have moderate to high wave motion, using a twist-tie to hold the fragment in place has been shown to increase transplanting success (Fonseca et al. 1984). The simplest way to restore a bed is by broadcasting seeds across a potential new site (Hemminga & Duarte 2000). While this technique is very easy to administer, it is rarely used, due to low seed availability and survival (Hemminga & Duarte 2000). *Halodule wrightii* has been shown to produce seeds only under extremely rare circumstances (Phillips 1960, McMillan & Moseley 1967, Eleuterius 1971). However, transplantation attempts using *H. wrightii* fragments have higher success rates, in part because it is the pioneer species (Fonseca et al. 1998).

CHAPTER 2: CURRENT AND HISTORICAL ANALYSIS OF *CRASSOSTREA VIRGINICA* DISTRIBUTION IN MOSQUITO LAGOON, FLORIDA

Introduction

Crassostrea virginica has been classified as an ecosystem engineer as well as a keystone species because of the important benefits that oysters and oyster reefs provide (Coen et al. 2007). Due to their three-dimensional structure, oyster reefs are able to maintain high levels of biodiversity (Coen & Luckenbach 2000). Many economically and ecologically important species can be found using the oyster reefs, including shrimp, stone crabs, blue crabs, and spotted sea trout (Coen & Grizzle 2007, Stunz et al. 2010). Previous research has demonstrated that oysters can have an ecosystem-wide scale effect on water clarity (Newell 1988, Newell & Koch 2004). Oysters not only support species richness and increase water clarity, but they also act as soft-armor for shoreline stabilization (Piazza et al. 2005).

It has been estimated that 85% of shellfish reefs have been lost worldwide (Beck et al. 2011). Oyster coverage has decreased for several reasons: habitat destruction, disease, overharvesting and reduced water quality (Wenner et al. 1996, Coen et al. 1999, Kirby 2004, Johnson et al. 2009). The severe loss of *C. virginica* has encouraged different types of restoration efforts throughout its native range. Restoration focuses on providing substrate for larval settlement and, if necessary, increasing larval abundances by introducing additional spat, either wild or cultivated. The simplest form of restoration is dumping the new substrate material (shell, cement, marl, etc.) directly on the seafloor. Another technique is the use of mesh bags filled with oyster shells, which is often more similar to the three-dimensional shape of a natural reef than dumping shells on the benthos (Ertel & McCall 2005). These techniques have been somewhat successful for areas suffering from overharvesting or a disease outbreak. However, in Mosquito Lagoon, boat wakes are a primary cause for oyster reef degradation. Water motion from boat wakes causes live clusters and oyster shells to wash up on top of one another, which can push the oyster clusters above the mean high water level (Grizzle et al. 2002, Wall et al. 2005). These clusters die due to lack of inundation, causing the formation of a dead reef (Grizzle et al. 2002, Wall et al. 2005, Stiner &Walters 2008).

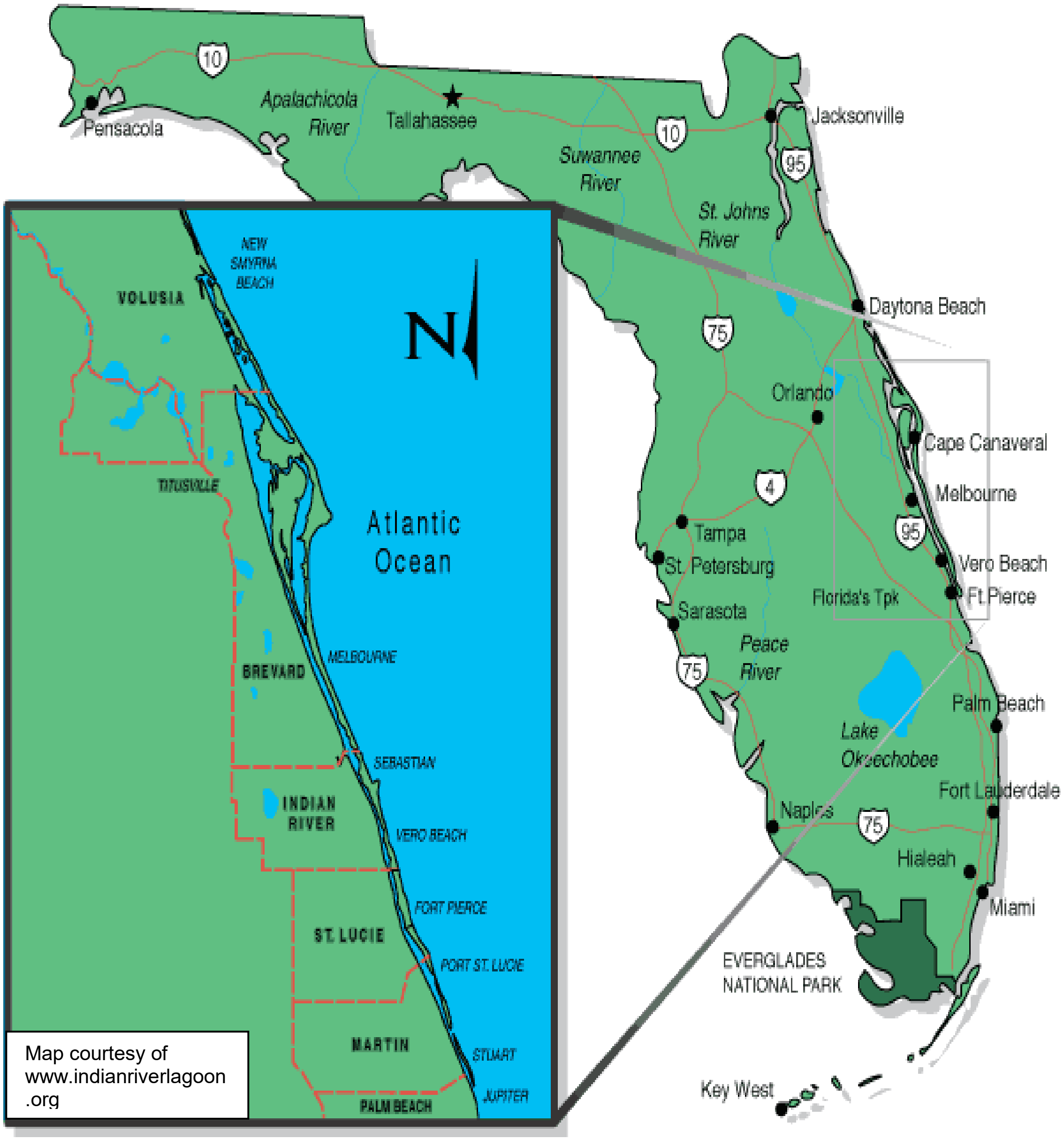
In Mosquito Lagoon, FL, a restoration technique was developed for shallow, intertidal areas with intense boating pressures (>50 boats/hour) involving the use of oyster mats has been developed to help stabilize the reefs and prevent the dislodgement of oysters. Oyster mat restoration involves aquaculture grade mesh mats that measure 0.25 m^2 in size. The mats have 36 oyster shells firmly attached to them using zip-ties so that the shells are perpendicular to the benthos (Wall et al. 2005). The mats are deployed in a quilt-like fashion on top of a leveled dead reef, and held in place with cement weights. Restored oyster reefs have been shown to have equal densities of live oysters (316/m^2) when compared to natural oyster reefs after 3.5 years.

In 2002, Grizzle and colleagues conducted a study of the historical changes in intertidal oyster reefs in select areas of Mosquito Lagoon using year 2000 imagery (Grizzle et al. 2002). I aim to expand that study to include the entirety of Mosquito Lagoon and incorporate recent restoration efforts that have occurred starting in the summer of 2007. The goals of this project are to: 1) assess the change in natural oyster habitat coverage over time (1943 – 2009), 2) document the formation of dead reefs over time, and 3) quantify the amount of new oyster habitat created by restoration.

Methods and Materials

Study Site

The project study site was Mosquito Lagoon, which is the northern-most estuary in the Indian River Lagoon system. The Indian River Lagoon system (IRL) is a group of estuaries that comprises approximately 250 km of Florida Atlantic coast (St. Johns River Water Management District 2006). The IRL is one of the most biologically diverse estuaries in the world, in part because is lies in a transitional area between temperate and subtropical climate zones (Dybas 2002). The majority of the lagoon is a complex system of shallow, open water areas with almost 100 mangrove dominated islands (Walters et al. 2001). Mosquito Lagoon is a microtidal system, with a mean water depth of 1.7 m and a tidal range of approximately 10 cm in our study area (Smith 1993, Hall et al. 2001, Steward et al. 2006). Water motion in Mosquito Lagoon is largely wind-driven, with wind speeds ranging from 0-50 kmh that create wave heights of 10 to 30 cm (Smith 1993, Hall et al. 2001). The GIS analysis will encompass all of Mosquito Lagoon, south of Ponce de Leon Inlet in New Smyrna Beach to the southern end of the lagoon, including the Canaveral National Seashore (CANA) (Figure 3).



Map courtesy of www.indianriverlagoon.org

Figure 3 Map of Mosquito Lagoon, Florida.

Preliminary Field Checking

We conducted pre-photointerpretation fieldwork in the summer of 2010 in order to assist with accurate photointerpretation. This preliminary field checking was directed at advanced signature identification prior to beginning digitization. The fieldwork was completed in order to identify and classify the signatures appearing on the 2009 aerial photographs. Prior to visiting the field, check sites were selected representing the assumed oyster signatures existing within the project area.

Photointerpretation Process

To assess the current area of natural, restored and dead oyster reefs within Mosquito Lagoon, imagery from the most recent available year, 2009, were screen digitized using ArcGIS software. The aerial photographs used in this study were obtained from Saint Johns River Water Management District (SJRWMD) and Volusia County GIS Department (Table 1). The aerial photographs had high enough contrast to accurately detect oyster reefs. Also, the aerial photography was captured during ideal conditions for oyster habitat study (i.e during low water season). Natural reefs, restored reefs and dead reefs were digitized in order to estimate the area of each type of reef. Each reef type has a distinct ‘signature’ (i.e. set of identifying characteristics) that was used to identify individual oyster reefs on the aerial photographs. Fringing oyster clusters that were covered by mangroves in the photographs were not included in the study. Also, scattered live clusters were not included. Only dense clusters of live oysters (e.g. patch and fringing reefs) were included in the natural reef class. Natural reefs with dead margins were classified as two reefs with different reef types. All oyster habitats were mapped to the level of detection (no minimum mapping unit). Once the 2009 map and ground truthing were complete, historical aerial photographs were digitized using the 2009 reef locations as references. The process began with 2006 and worked backwards to 1943. Issues with aerial photography such as glare or dark areas were resolved by interpolating reef positions between the previous and following years, a method also used during the Grizzle et al. (2002) study.

Table 1 Source Information for aerial imagery used in the present study.

Year	Type	Source	Resolution
1943	Black & White prints	SJRWMD	1:16,000
1951	Black & White prints	Volusia County	1:16,000
1967	Black & White prints	Volusia County	1:24,000
1971	Black & White prints	Volusia County	1:24,000
1984	National High Altitude Photography	SJRWMD	1:24,000
1995	Color Infrared Prints	SJRWMD	1:6,000
2006	Digital	SJRWMD	1:24,000
2009	Digital	SJRWMD	1:24,000

Ground Truthing

A set of quality control points was created by a GIS Analyst at SJRWMD to avoid bias in the selection of quality control points. The objective was to generate sample points for each class (oysters, non-oysters) using a spatially balanced method. A 1 x 1 kilometer grid was generated for the study area (mapped oyster coverage) resulting in 60 grid cells across the lagoon. Random points were generated within the grid cells, across the final oyster layer. Another set of random points was generated across a non-oyster layer consisting of areas that were likely to be oyster habitat in terms of bathymetry. A minimum distance of 200 m was enforced between points. Ground-truthing was conducted by both UCF and SJRWMD staff. A simple kappa coefficient was calculated to assess the agreement of the field data with the mapping classification.

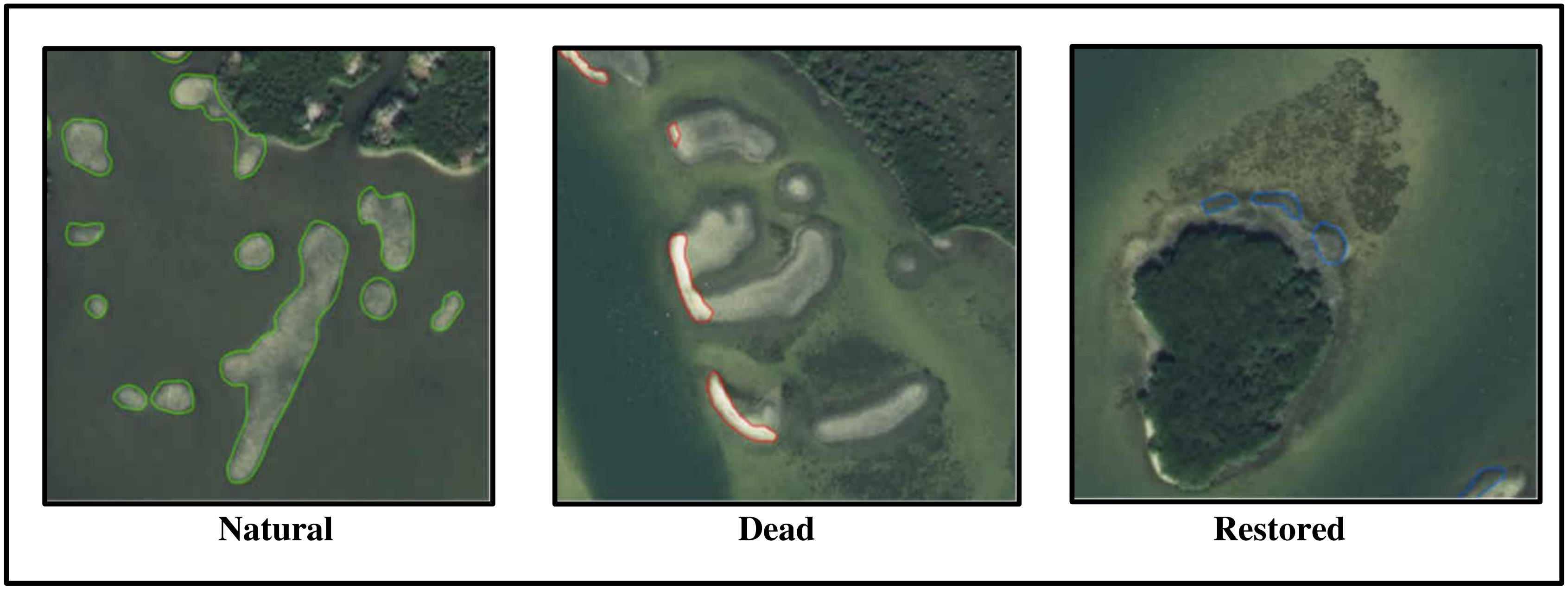
Results

During preliminary field checking in the summer of 2010, three unique reef signatures were developed: natural reef, dead reef and restored reef (Figure 4). Natural oyster reefs were identified based on the following criteria: globular or irregular in shape, with dark margins around the edge of the reef, and a lighter color in the center of the reef. Most natural reefs were located near mangrove islands at shallow bathymetry due to the fact that oysters are intertidal in Mosquito Lagoon. Dead reefs were identified based on the following criteria: bright white reflection due to bleached, disarticulated shells that are continuously exposed, even at high tide. Dead reefs were found either adjacent to existing natural oyster reefs (i.e. dead margin) or standing alone, typically located on main boating channels (Grizzle et al. 2002) (Figure 4). Restored reefs were identified based on the following criteria: darker than natural oyster reefs, due to black mesh mats that support the restored reef, and typically more geometric in shape. Locations of all restored reefs were known before digitizing.

After reef signatures were developed, aerial photography of Mosquito Lagoon in 2009 was digitized using the reef signature to identify different reef types using ArcGIS software (Figure 4). 2542 natural reefs with identified within Mosquito Lagoon (Table 2). The natural reefs ranged in size from < 10 m^2 to over 5000 m^2 and compromised a total of 46.34 hectares of bottom habitat cover in the lagoon (Table 2). Of the 2542 natural reefs identified in this study, 524 reefs were located within CANA park boundaries, which constitute 21% of the total oyster reefs in Mosquito Lagoon (Table 3). In 2009, 247 dead reefs or dead margins were identified in the study area. These dead margins/reefs ranged in size from < 5 m^2 to over 1300 m^2. Dead areas comprised anywhere from 10% (i.e. dead margin) to 100% (i.e. dead reef) of the original natural reef areal coverage. As of the winter of 2009, 19 dead margins/reefs were restored to live oyster habitat. These 19 restored reefs totaled almost half an acre of bottom habitat in the lagoon. The restored reefs ranged in size from 10 m^2 to 317 m^2. All of these reefs were located within Canaveral National Seashore boundaries.

The ground truthing was completed in the spring of 2011. Within the oyster class quality control points, there was 98% accuracy. Within the non-oyster class points, there was 100% accuracy. This resulted in a simple kappa coefficient of 0.9684, which indicates almost perfect agreement of field checked data and mapped data (Viera & Garrett 2005).

Over the 66-year study period, there was a steady decrease in natural reef coverage, in the entire Mosquito Lagoon as well as within the CANA boundaries (Tables 2, 3). Losses in natural reef habitat in 1943 and 1951 consisted mainly of reefs shrinking, potentially due to harvesting pressure or the impacts of dredging in the area. In 1943, four dead margins/reefs were located on the Atlantic Intracoastal Waterway on the western side of Mosquito Lagoon, and one dead reef was located on the main channel on the eastern side of Mosquito Lagoon (Figure 5). By 1951, ten dead margins/reefs had appeared within Mosquito Lagoon, mainly along Shipyard Channel (Figure 6). By 1967, the number of dead reefs had nearly doubled (18) from what was present in 1951 (Figure 7). This trend continued, and in 1971 and 1984, dead margins/reefs were visible along Government Cut and other main boating channels (Figures 8, 9). Between 1984 and 1995, Mosquito Lagoon experienced the largest increase in dead margins/reefs, from 93 dead reefs in 1984 to 230 in 1995 (Table 2) (Figures 9, 10). In 2006, dead reefs covered 3.61 hectares in bottom habitat (Table 2) (Figure 11). In 2009, the areal coverage of dead reefs increased while the total number decreased (4.3 hectares, 247 reefs), which is in part due to the converging of several smaller dead reefs into large dead reefs (Figure 12). Due to restoration focused within CANA boundaries, there was a decrease in dead reefs (109 to 103) within the park boundaries (Table 3). Also, there was a small increase in natural reef acreage (~1 acre) behind and near restored reefs within CANA (Table 3). Despite this gain within the park, the oyster habitat within all of Mosquito Lagoon decreased by almost 15 hectares, which constitutes 24% of the original coverage in 1943.



Natural

Dead

Restored

Figure 4 The three reef signatures used to identify reef types during photointerpretation. Natural reefs are outlined in green (on left), dead margins/reefs are outlined in red (center) and restored reefs are outlined in blue (on right)

Table 2 Mosquito Lagoon oyster coverage 1943 - 2009 as determined by screen digitizing with GIS using aerial photography of the study area.

Year	# Natural Reefs	Hectares of Natural Reefs	# Dead Margins/Reefs	Hectares of Dead Reefs	% Loss of Natural Coverage
1943	2722	61.07	5	0.32	0%
1951	2716	57.46	10	0.75	6%
1967	2715	54.91	18	0.99	10%
1971	2705	54.17	32	1.06	11%
1984	2685	51.82	93	2.33	15%
1995	2609	48.28	230	3.12	21%
2006	2530	46.04	269	3.61	24%
2009	2542	46.34	247	4.30	24%

Table 3 Oyster coverage within CANA boundaries 1943 - 2009 as determined by screen digitizing with GIS using aerial photography of the study area.

Year	# Natural Reefs	Hectares of Natural Reefs	# Dead Margins/Reefs	Hectares of Dead Reefs	% Loss of Natural Coverage
1943	576	24.07	4	0.26	0%
1951	573	20.72	6	0.52	14%
1967	577	19.27	7	0.55	20%
1971	573	18.62	14	0.56	23%
1984	570	16.91	43	1.38	30%
1995	541	15.26	84	1.54	37%
2006	515	14.04	109	1.65	42%
2009	524	14.46	103	1.45	40%

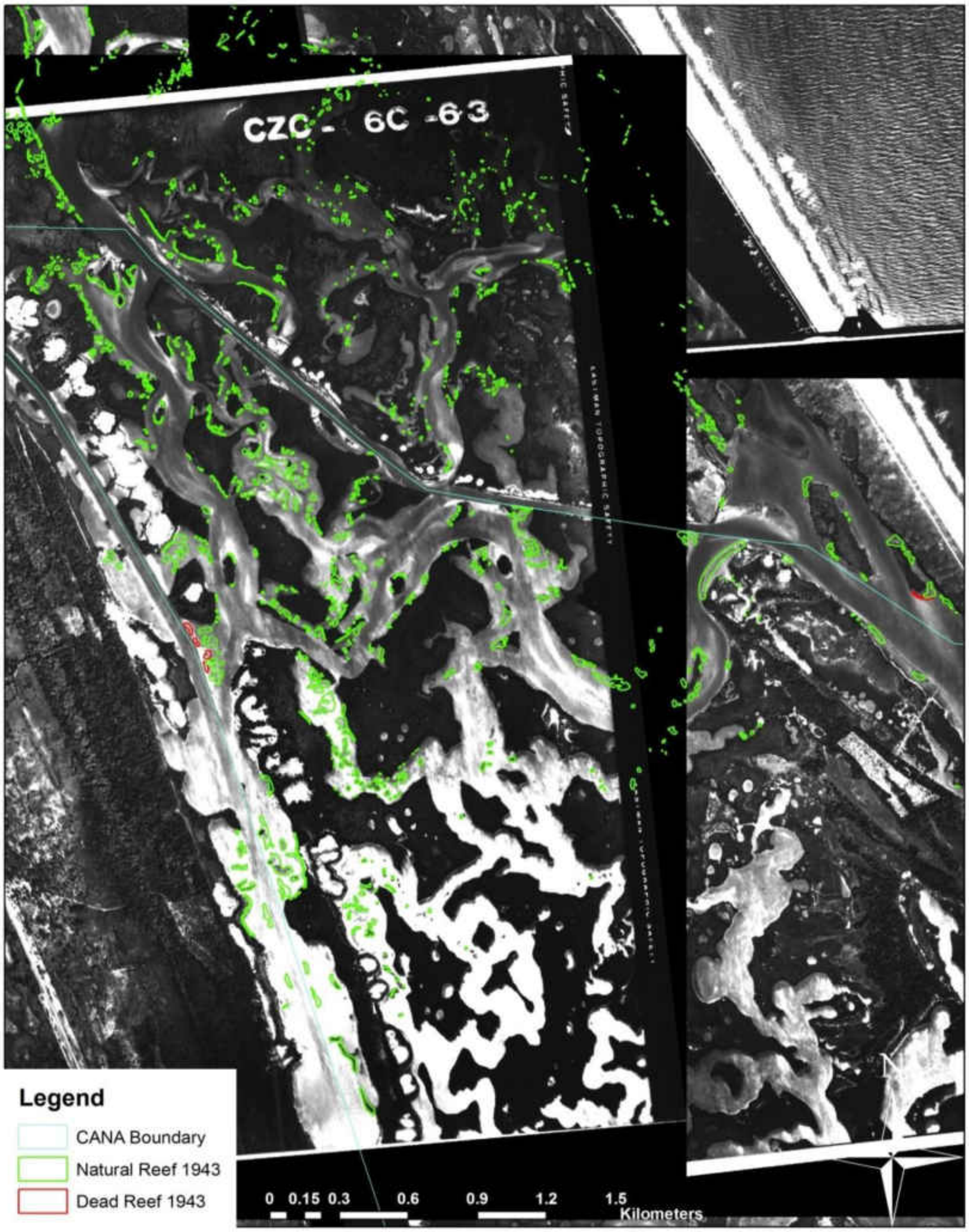


Figure 5 Map of a portion of Mosquito Lagoon oyster reefs in 1943. Red represents dead margins/reefs, green represents natural reefs and blue represents restored reefs. Main boating channels are marked as a reference.

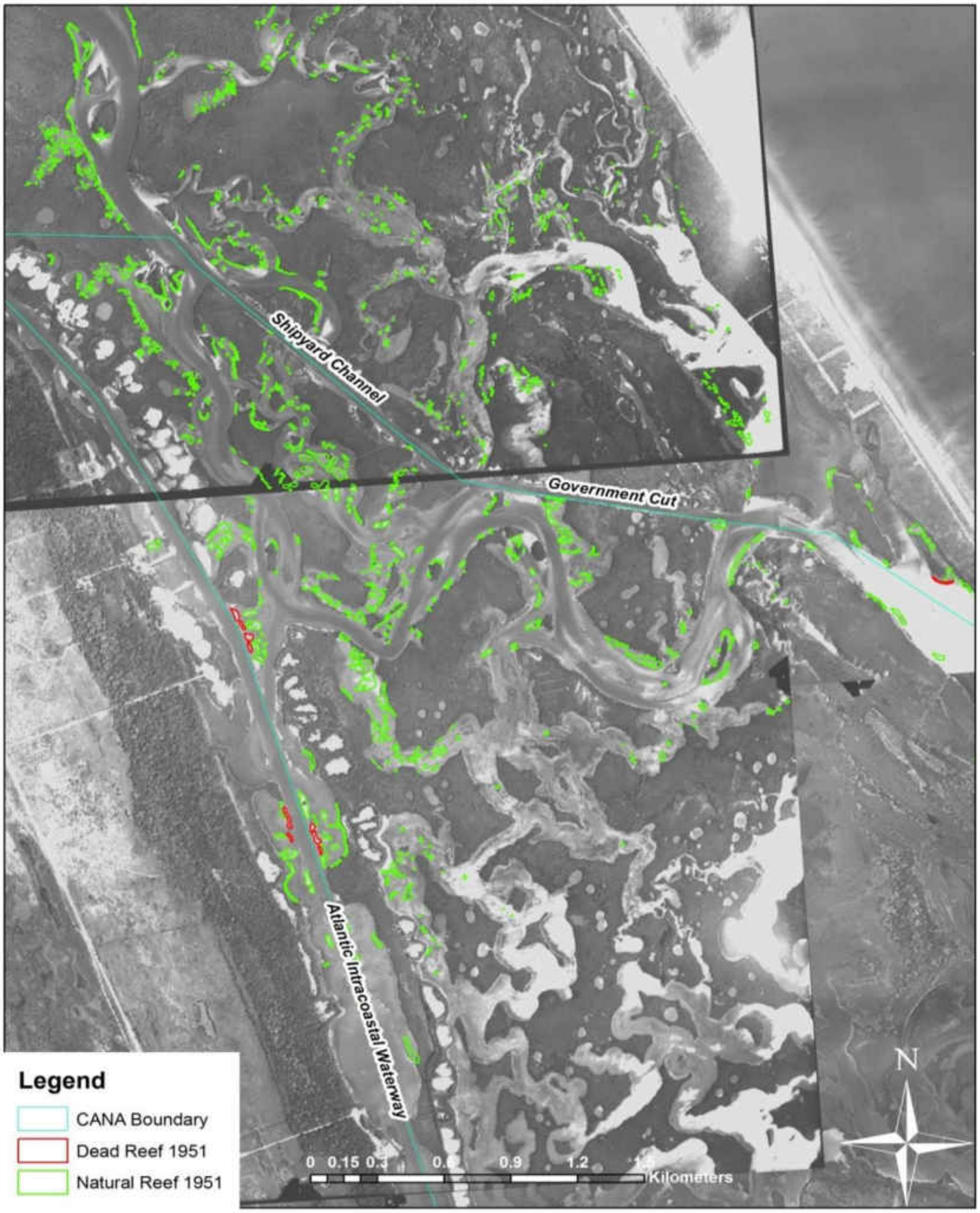


Figure 6 Map of a portion of Mosquito Lagoon oyster reefs in 1951. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

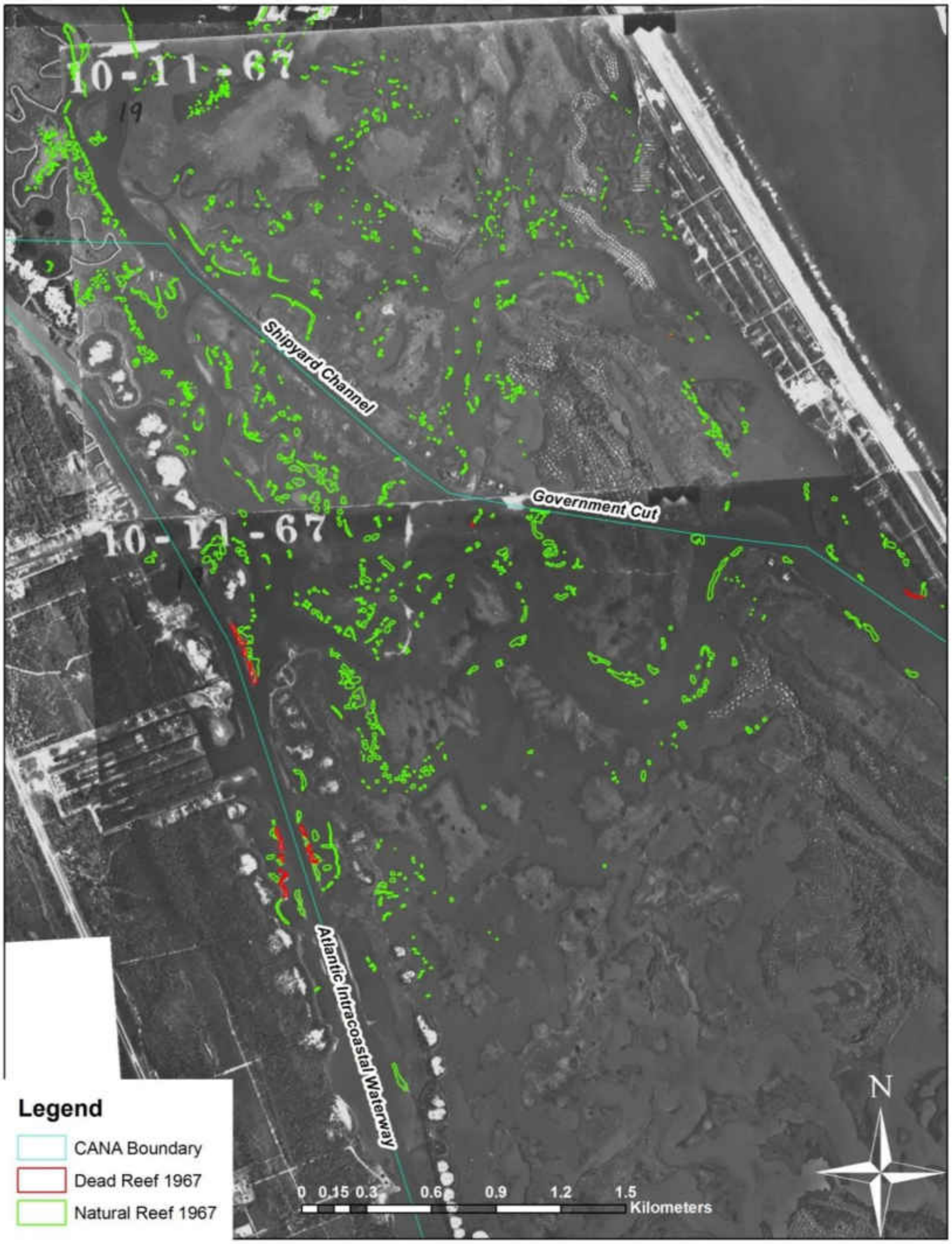


Figure 7 Map of a portion of Mosquito Lagoon oyster reefs in 1967. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

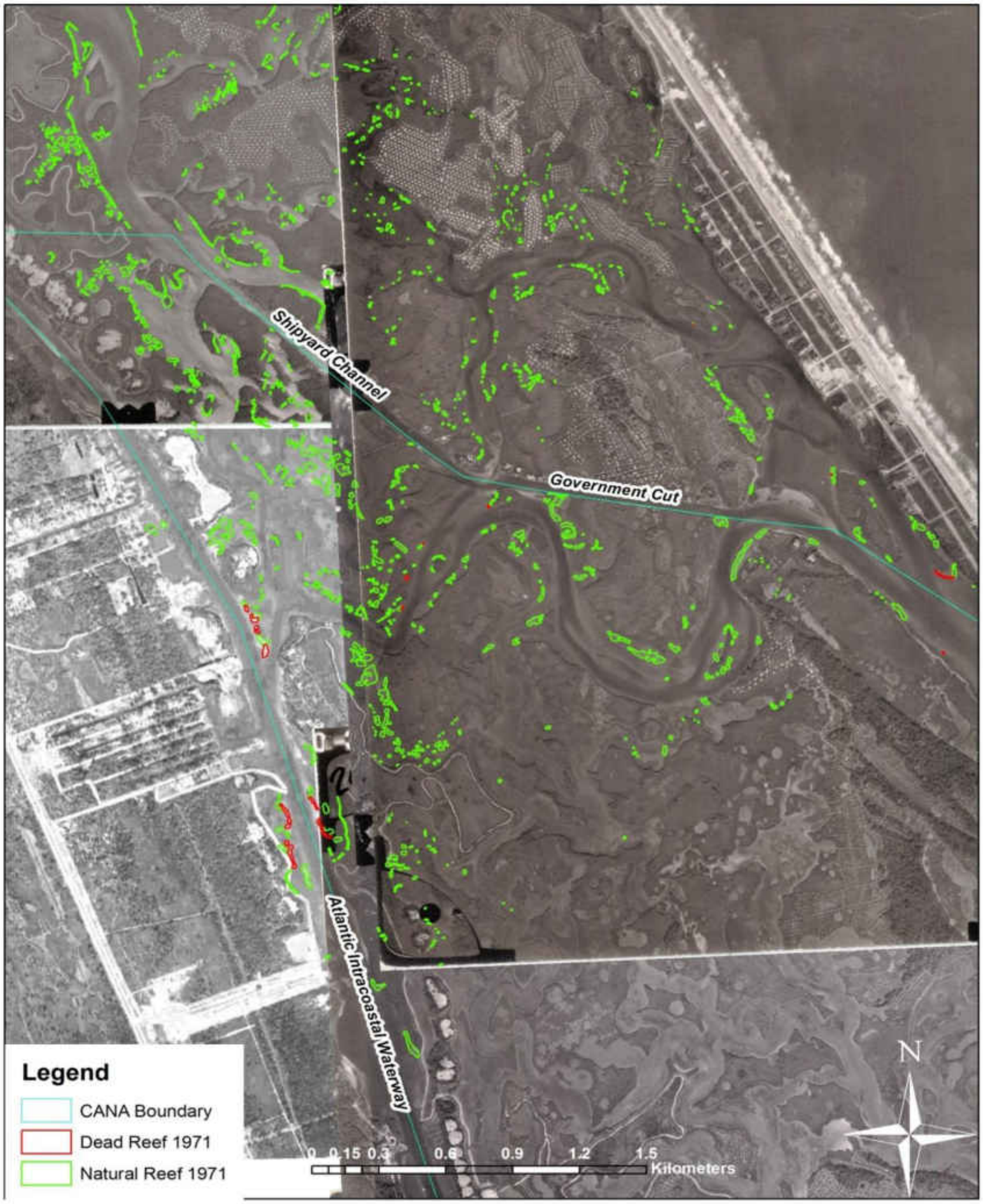


Figure 8 Map of a portion of Mosquito Lagoon oyster reefs in 1971. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

Figure 9 Map of a portion of Mosquito Lagoon oyster reefs in 1984. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

Figure 10 Map of a portion of Mosquito Lagoon oyster reefs in 1995. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

Figure 11 Map of a portion of Mosquito Lagoon oyster reefs in 2006. Red represents dead margins/reefs, and green represents natural reefs. Main boating channels are marked as a reference.

Figure 12 Map of a portion of Mosquito Lagoon oyster reefs in 2009. Red represents dead margins/reefs, green represents natural reefs and blue represents restored reefs. Main boating channels are marked as a reference.

Discussion

Many studies have shown large losses in oyster reef coverage compared to historical levels (Beck et al. 2011). Beck et al. (2011) created four categories of reef condition based on historical oyster abundance indicators or changes in reef coverage as measured using GIS: good = less than 50% lost, fair = 50%- 89 % lost, poor = 90%- 99% lost, and functionally extinct = more than 99% lost. Of the regions evaluated, they found the majority of oyster reefs to be in poor condition (Beck et al. 2011). Many bays and estuaries have been deemed functionally extinct (i.e. < 1 % remaining), especially in Europe, Australia and North America (Beck et al. 2011). In Florida, the overall condition of oyster reefs is fair. For example, in the Big Bend of Florida, researchers found a decrease of 123 hectares of oyster habitat between 1982 and 2010, which represents a loss of 66% of the original oyster habitat (Seavey et al. 2011). Seavey et al. (2011) found these losses to be ecologically significant because the oyster reefs had previously existed for 2,800 – 4,000 years (Grinnell 1972, Wright et al. 2005). Thus an important change has occurred over the last 30 to 40 years to cause such large losses in reef habitat (Seavey et al. 2011). The researchers attributed these losses mainly to changes in freshwater inputs which affected oyster spat recruitment and survival (Seavey et al. 2011). Our study found that reef condition in Mosquito Lagoon is good, with losses of 24% compared to 1943 levels. Most of the losses have been found along major boating channels, suggesting that human impacts are playing a key role in these losses (Grizzle et al. 2002).

The large amount of boating activity within Mosquito Lagoon is due to it being a very popular tourist and fishing destination (Scheidt & Garreau 2007). A 2005 survey found over 100,000 registered boats within counties that are adjacent to Mosquito Lagoon (IRLNEP 2007).

Approximately 46,000 boaters frequented Mosquito Lagoon between 2006 and 2007, and 76% of these were fishing boats (Scheidt & Garreau 2007). Advancements in boating technology has allowed boaters to travel in shallower waters, which can increase the negative impact of boat wakes (Scheidt & Garreau 2007). Boating traffic has been linked with dead reef formation as well as shoreline erosion (Gabet 1998, Grizzle et al. 2002, Wall et al. 2005). High levels of wave motion caused by boat wakes have been shown to dislodge oyster shells and potentially damage spat in the process (Walters et al. 2002).

When assessing reef conditions, it is important to understand the underlying mechanism responsible for causing the rapid loss of reef habitat to better guide future conservation planning. Our study, as well as Grizzle et al. (2002), provides correlative evidence that boat wakes are playing a major role in oyster reef declines. Understanding trends in oyster death could help minimize the impact of boat wakes by implementing no wake zones in key areas. Additionally, using the historical coverage data, natural resource managers can establish a baseline for oyster reef coverage in the area and demonstrate how oyster restoration is affecting the current oyster coverage. By mapping the historical distribution of oysters, managers can use that information to set conservation and restoration goals. The loss of other marine habitats such as mangroves (30%–50%), coral reef (~20%), and seagrass (~ 30%) have greatly influenced policy and conservation actions (Wilkinson 2002, Hughes et al. 2009, Spalding et al. 2010). With a global loss of 85% (Beck et al. 2011), shellfish habitat protection and restoration needs to be the focus of future conservation policy for coastal areas.

CHAPTER 3: QUANTIFYING BOAT WAVE HEIGHT ATTENUATION ASSOCIATED WITH INTERTIDAL *CRASSOSTREA VIRGINICA* REEF RESTORATION

Introduction

Coastal erosion is a top priority around the globe due to climate change and rising sea levels (Piazza et al. 2005). Natural resource management plans typically include the protection of shoreline integrity as well as the prevention of shoreline erosion (Mimura & Nunn 1998, Klein et al. 2001). A common technique for protecting shorelines from erosion involves hardening the shoreline, using materials such as limestone rock, rip-rap, and concrete (Hillyer et al. 1997). Recently, however, there has been a push to transition into living shoreline techniques. Living shoreline plans involve the planting or restoration of native flora and fauna that have numerous ecological benefits, including serving as habitat for many different species (Piazza et al. 2005). Intertidal reefs of the eastern oyster *Crassostrea virginica* that are normally found seaward of salt marshes have become a popular choice in living shorelines in temperate areas (Piazza et al. 2005, Coen et al. 2007).

Crassostrea virginica is classified as an ecosystem engineer because its reefs provide many benefits to coastal ecosystems, including habitat for many species, water filtration, and shoreline stabilization (Coen et al. 2007, Grabowski & Peterson 2008). Oyster reefs contribute to shoreline stabilization by providing rugged, three-dimensional structures that disrupt incoming waves (Coen et al. 2007). In doing this, reefs absorb wave energy which helps to preserve emergent vegetation and stabilize sediments along shorelines (Dame 1996, Coen et al. 2007). As a result, oyster reefs are ideal for shoreline protection and stabilization (Dame & Patten 1981, Meyer et al. 1997, Piazza et al. 2005).

In Mosquito Lagoon, FL, it has been shown that boat wakes play a role in oyster reef declines in the area (Grizzle et al. 2002, Wall et al. 2005). Mosquito Lagoon is a shallow, microtidal system, with a mean water depth of 1.7 m and a tidal range of approximately 10 cm in our study area (Smith 1993, Hall et al. 2001, Steward et al. 2006). Water motion from boat wakes in this area causes disarticulated oyster shells and live oyster clusters to be pushed up on top of one another, causing the formation of dead reefs (Grizzle et al. 2002, Wall et al. 2005, Stiner & Walters 2008). As a result, an oyster restoration technique was created using oyster restoration mats to help prevent deployed shells from being dislodged by boat wakes. This oyster restoration technique involves aquaculture grade mesh mats that measure 0.25 m^2 in size. Each mat has 36 oyster shells firmly attached to it using zip-ties so that the shells are perpendicular to the benthos, to maximize larval recruitment (Wall et al. 2005). Mats are then deployed in a quilt-like fashion on top of a raked-down dead reef, and are held in place with circular weights. These reefs have been shown to have equal densities of oyster clusters compared to natural reefs within 3.5 years and no new dead areas reformed after 5 years (Walters, unpublished data).

We examined the potential for oyster restoration mats to be used for living shoreline protection in Mosquito Lagoon by determining the changes in wave height attenuation of restored versus natural oyster reefs. Additionally, we compared the restored oyster reefs to dead reefs and areas without oysters (control) to determine how much additional wave height was attenuated by live oysters.

Methods and Materials

Study site

The oyster restoration sites are in Mosquito Lagoon, which is the northernmost section of the Indian River Lagoon (IRL) system, along the east coast of central Florida. The IRL is of a group of estuaries that comprises approximately 250 kilometers of the Atlantic coast of Florida (St. Johns River Water Management District, 2006). It is one of the most diverse estuaries in the U.S., in part because is lies in a transitional area between temperate and subtropical climate zones (Provancha et al. 1992, Dybas 2002). The majority of the lagoon is a complex system of shallow, open water areas with mangrove (*Rhizophora mangle*, *Avicennia germinans*) dominated islands (Walters et al. 2001). Water motion in Mosquito Lagoon is largely wind-driven, with wind speeds ranging from 0-50 kph that create wave heights of 10 to 30 cm (Smith 1993, Hall et al. 2001). The water level in the lagoon changes with the annual rise and fall of oceanic water levels by approximately 0.3 m, and as a result, the highest water levels occur in October-November and the lowest water levels occur in April-May (Smith 1993, Hall et al. 2001). In addition to being a biodiversity hotspot, Mosquito Lagoon is considered the redfish capitol of the world, and has become a very popular destination for boaters (Scheidt & Garreau 2007).

Wave attenuation

To examine differences in wave height attenuation, mimic oyster reefs (restored, natural, dead) (n=3) were constructed inside of the wave tank facility at the Florida Institute of Technology's Surf Mechanics Laboratory in Melbourne, FL (Figure 13). The tank measured 9.08 m in length, 0.57 m in width and 0.91 m in depth. Both the bottom and side walls are constructed of clear acrylic supported with metal beams at 1.22 m intervals the length of the tank. The tank has been used in biological and engineering studies since 1990 (Lohmann et al. 1990). To create realistic bathymetry inside the wave tank, sediment was obtained from Mosquito Lagoon and used to form a gradual, sloping (15:1 slope) shoreline within the tank based on shoreline surveys of Mosquito Lagoon. The sediment had a mean density of 1.85 g/cm^3 (± 0.4 SE). The constructed shoreline extended 4.6 meters from the wall of the tank opposite of the wave generator. A water depth of 0.3 m was maintained throughout the trials by filling or draining the tank. To closely replicate waves in nature, a flapper paddle located opposite of the constructed shoreline generated waves through an attached rotary motor. A wave height of 12.7 cm was created by the rotary motor.

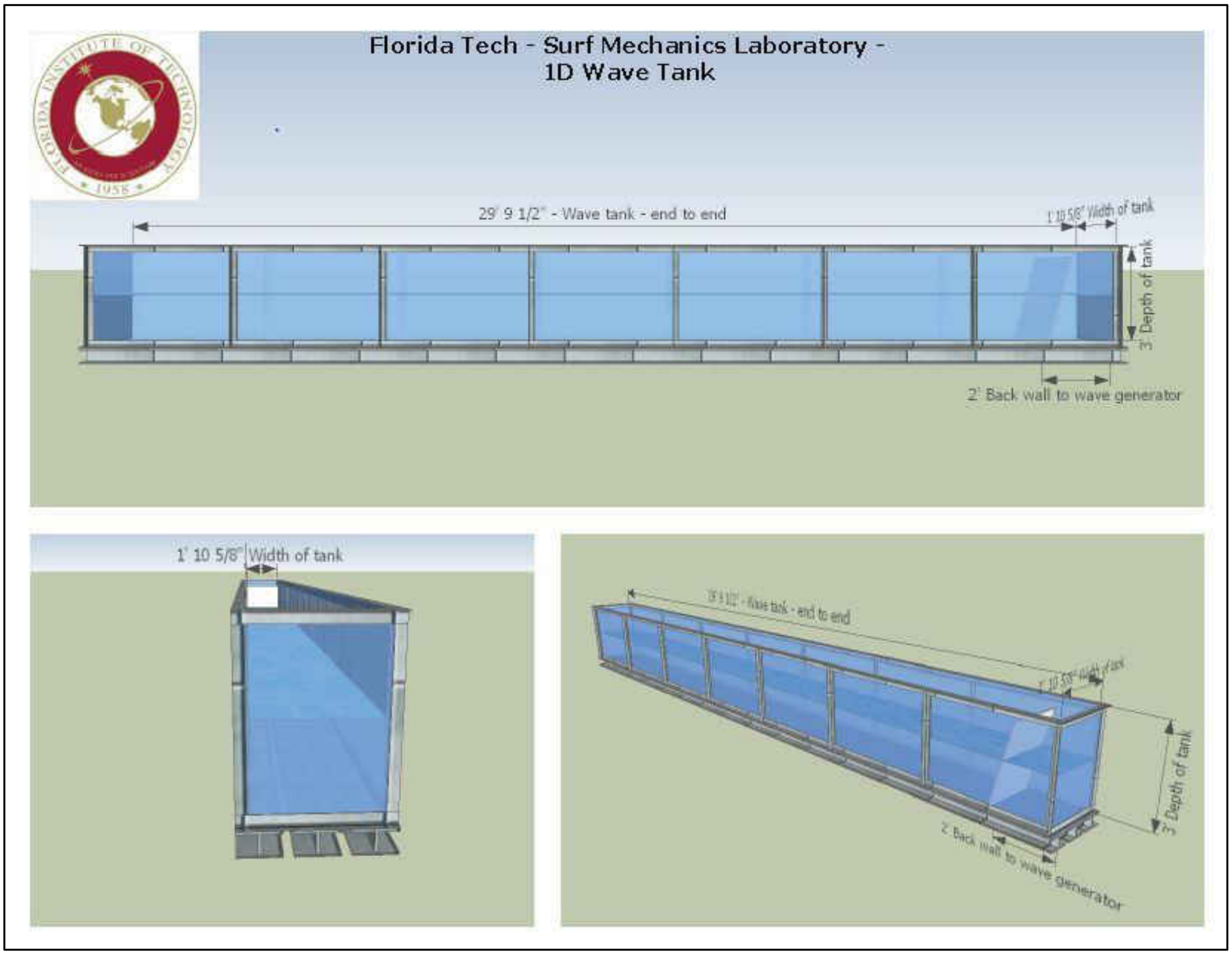


Figure 13 Diagram of wave tank used in study. The wave tank is located in the Surf Mechanics Laboratory at the Florida Institute of Technology.

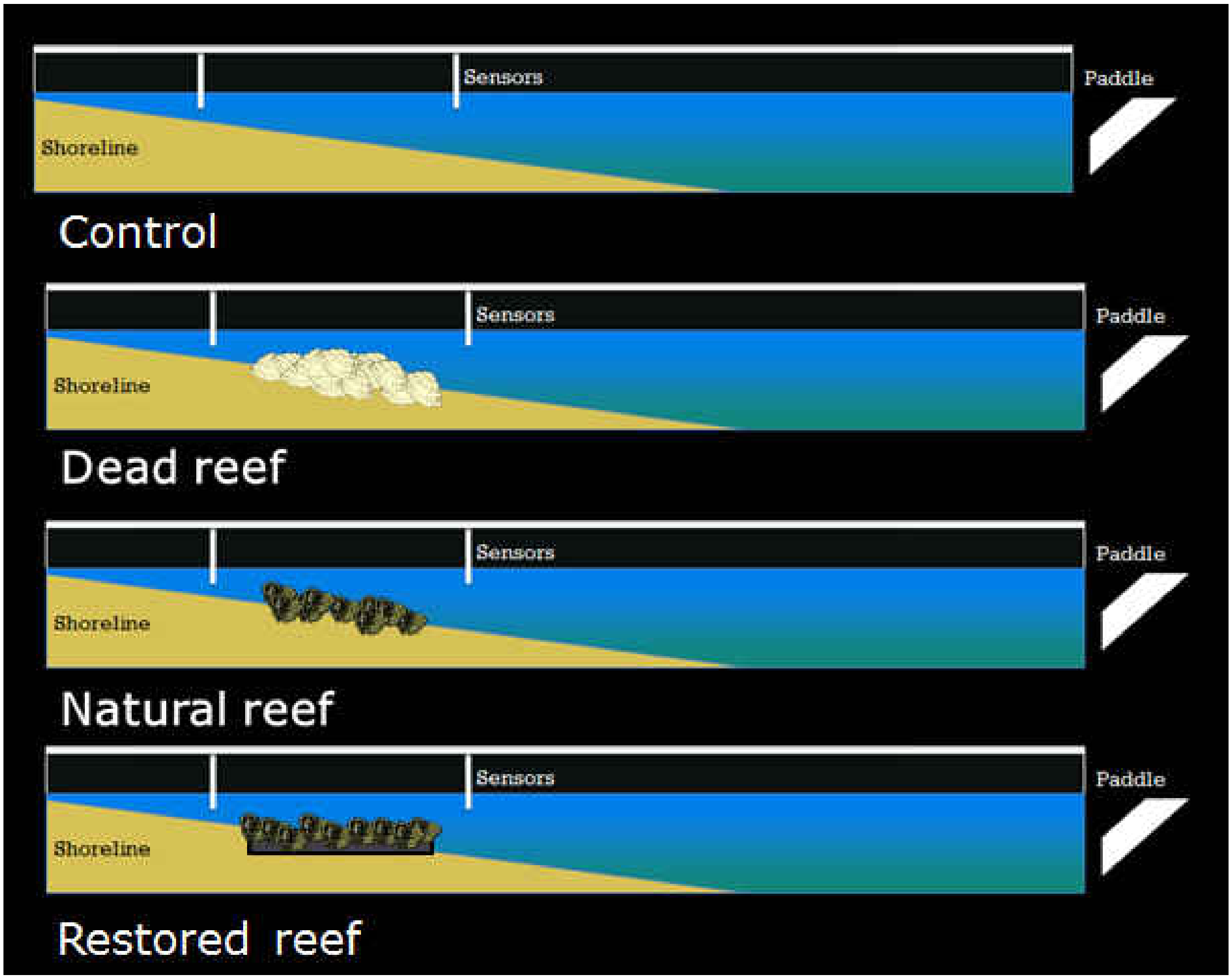


Figure 14 Illustration of control, dead reef, natural reef, and restored reef treatments for the wave tank experiment.

Shoreline sediment without oysters was used as a control (Figure 14). Natural reefs consisted of live adult oysters in clusters (158 oysters/0.5 m^2) pushed approximately 5 cm into the sediment that simulated natural reefs. The average oyster shell height above the sediment was 11.9 cm (± 1.9 SE). For restored reefs, two oyster restoration mats with an equal live adult oyster density (158 oysters/0.5 m^2) was placed on top the shoreline sediment; these oysters extended 11.8 cm (± 1.7 SE) above the sediment surface. For the dead reefs, there was a layer of densely pack oyster shells on top of the shoreline, extending to a height of 12 cm above the sediment. For each reef replicate, 10 sets of wave trains (one train = 10 waves) were generated. The resulting wave heights were measured with pressure sensors (OSSI Wave Staff: OSSI-010-002E). The pressure sensors used Ocean Sensor Systems Incorporated software to measure the wave heights within the tank. Sensors were located 2.5 m from the paddle (before treatment), and 5.5 m from the paddle (after treatment). The experiment was run in a block design, with each treatment being run once in each block. One block was run in August, another in September, and the last block was run in October 2011.

Statistical Analyses

R statistical software was used for statistical analyses (R Development Core Team 2011). All data were tested using the Shapiro-Wilk test for normality and an *F*max test for homogeneity of variance. The mean of each wave train was calculated to get one wave height per train. The resultant wave was then substrated from the incoming wave to obtain a change in wave height. Change in wave height data was analyzed using a block ANOVA. Tukey's HSD tests were used for pairwise comparisons of reef treatment types as appropriate.

Results

Changes in wave heights were found to be significantly different among reef types (ANOVA: $p < 0.001$, Table 4). Blocks were not significantly different from one another (ANOVA: $p = 0.9141$, Table 4). The control and dead reef treatments were similar to each other as determined by a Tukey's HSD test (Figure 15). The mean wave height of the control treatment was 12.4 cm (±0.08 SE), and the dead reef treatment was 12.3 cm (±0.09 SE). Within the control treatment, wave heights ranged from 11.3 cm to 13.1 cm (Figure 15). Within the dead reef treatment, wave heights ranged from 11.2 cm to 12.9 cm (Figure 15). The dead reef treatment reduced the mean wave height by < 1% when compared to the control treatment. Natural and restored reef treatments were similar to each other (Figure 15). The mean wave height of the restored reef treatment was 9.3 cm (±0.07 SE), and natural reef treatment was 9.5 cm (±0.08 SE) (Figure 15). Within the restored reef treatment, wave heights ranged from 8.2 cm to 10.0 cm (Figure 15). Within the natural reef treatment, wave heights ranged from 8.4 cm to 10.2 cm (Figure 15). The natural and restored reef treatments reduced the mean wave height by 23% and 25% respectively, relative to the control treatment and by 22% and 24% relative to the dead treatment.

Table 4 The results of a block ANOVA of change in wave height as explained by reef treatment type (Control, dead, restored and natural).

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Treatment	3	248.9529	82.9843	413.9308	<0.001*
Block	2	0.0361	0.0180	0.0899	0.9141
Residuals	114	0.8546	0.2005		

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

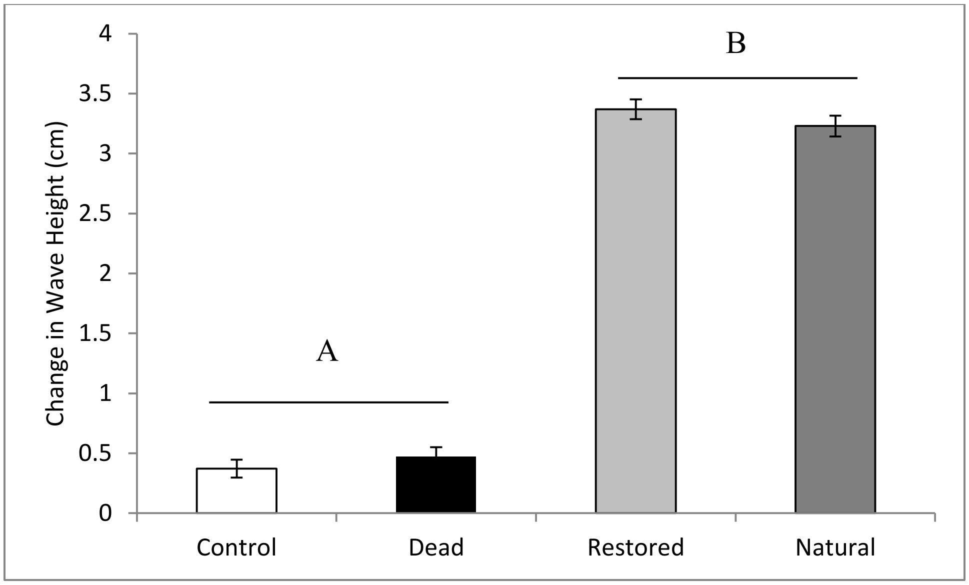


Figure 15 Mean change in wave height after encountering reef treatment (±SE). Wave heights were compared using a block ANOVA, and blocks were not significantly different from each other. Treatments with different letters are significantly different at $p < 0.05$ level as determined by Tukey's post hoc tests.

Discussion

The three-dimensional structure of oyster reefs has been shown to reduce water velocities, and increase the rate of sedimentation, both of which maintain the shoreline integrity behind the reefs (Coen et al. 2007). Scyphers et al. (2011) found that fringing oyster reefs parallel to shorelines can serve to decrease the erosion on coastal marshes caused by wind and boat wakes by up to 40%. Our study found that restored reefs possess the same wave height attenuation capacity as natural oyster reefs. Restored reefs attenuated wave heights by 25% on average while natural reefs attenuated wave heights by 23%. Piazza et al. (2005) also demonstrated that creating fringing restored reefs decreased shoreline erosion by an average of 33%, particularly in low energy environments. By attenuating wave motion, both natural and restored oyster reefs can help protect other coastal habitats such as seagrass beds and salt marshes from erosion (Henderson & O'Neil 2003).

Mosquito Lagoon is a very popular boating destination, with over 100,000 registered boats within neighboring counties (IRLNEP 2007, Scheidt & Garreau 2007). Boating traffic has been linked with loss of live oyster reefs as well as shoreline erosion (Gabet 1998, Grizzle et al. 2002, Wall et al. 2005). High levels of wave motion caused by boat wakes have been shown to dislodge oyster shells and potentially damage spat in the process (Walters et al. 2002). By stabilizing oyster clusters, restored reefs are able to protect spat and adult oysters in major boating channels.

As opposed to hard armoring of the shoreline, living shoreline restoration that includes *Crassostrea virginica* has the potential to be self-sustaining if the recruitment and survival of oyster spat is sufficient (Meyer et al. 1997, Piazza et al. 2005). The recruitment on restored oyster reefs in Mosquito Lagoon has been shown to be high, with adult oyster densities equivalent to natural reefs after 3.5 years (Walters, unpublished data). With equivalent oyster densities, our restored reefs provide structure for over 140 species of flora and fauna in Mosquito Lagoon (Barber et al. 2010). Manley et al. (2010) found that the increase in structural complexity created by the accumulation of adult oysters over time supported the settlement of additional fauna to the reef due to the increase in habitat surface area. Interstitial spaces created by the oyster shells increase settlement of other larvae such as mussels and barnacles by reducing water velocities (Manley et al. 2010). Additionally, restored oyster reefs can increase seagrass recruitment by encouraging entanglement of seagrass fragments (Chapter 4). This is in contrast to hard-armoring of the shoreline, which can reflect wave energies back into the water column, subjecting nearby shorelines to even higher less of erosion and loss of intertidal habitat (Bilkovic & Roggero 2008). Additionally, hard-armoring techniques, such as seawalls and bulkheads, are typically void of the structural complexity seen in oyster reefs, which serves as habitat for many species (Beck et al. 2001, Heck et al. 2003, Bilkovic & Roggero 2008).

Our study found that restored and natural oyster reefs significantly attenuated waves more than dead reefs or bare sediment. This suggests that oyster restoration can help prevent shoreline erosion in areas with increased wave motion due to human impacts, such as boating. Additionally, this study found that dead reefs do not attenuate waves more than a shoreline without oysters, thus these dead reefs may be attributing to shoreline erosion. By restoring dead reefs, we can increase local oyster populations while protecting shorelines.

CHAPTER 4: HOW *CRASSOSTREA VIRGINICA* RESTORATION AFFECTS VEGETATIVE RECRUITMENT OF *HALODULE WRIGHTII*

Introduction

Within the last 150 years, seagrass extent has decreased by over 30% worldwide (Hughes et al. 2009). Most of these declines are from direct and indirect human impacts (Erftemeijer & Lewis 2006, Herrera-Silveira et al. 2010). In response to this, detailed conservation plans to offset or prevent further human impacts on the seagrass ecosystem have been developed. Legislation has been passed in many places to designate certain areas as marine protected areas (MPAs) (Green & Short 2003). While protection was not often designated exclusively to seagrass beds, they were always listed are a key species within the protection areas (Green & Short 2003).

Many federal and state agencies in the US have long-term monitoring projects to continuously assess seagrass health. For example, long-term monitoring in Tampa Bay has found increases in seagrass coverage since the 1950s due to improvements in managing nutrient runoff and subsequent algal blooms (Johansson & Lewis 1992; Lewis et al. 1999). However, some areas of Tampa Bay have not recovered, possibly due to increases in man-made wave energy, such as boat wakes (Lewis 2002). Thus, resource managers turn to manual restoration to meet their seagrass targets (Tampa Port Authority 2011).

Successful seagrass transplantation is dependent on using optimal transplantation techniques for a given area, and previous research has shown significant differences between seagrass transplant survival using different restoration techniques (Calumpong & Fonseca 2001).

The simplest way to restore a seagrass bed is to broadcast seeds across a potential restoration site (Hemminga & Duarte 2000). While this technique is very easy to administer, it is rarely used, due to low seed availability and survival after broadcasting (Hemminga & Duarte 2000). For example, *Halodule wrightii* has been shown to rarely produce seeds (Phillips 1960, McMillan & Moseley 1967, Eleuterius 1971, McGovern & Blankenhorn 2007). Hence, most manual restoration methods involve the transplantation of seagrass plugs from existing beds to the new site, which can negatively impact donor beds (Hemminga & Duarte, 2000). Another technique involves collecting fragments of seagrass that have a growing tip and attaching the section of rhizome to a garden staple with a twist-tie, then planting in desired location (Fonseca et al. 1988). By using loose fragments, the donor beds are not impacted but the genetics of the fragment pool is unknown (Hall et al. 2006a). Few transplant efforts have had success with establishing significant seagrass growth (Fonseca et al. 1998, Grablow 2008). It is very common to have low survival among transplants, or complete failure of the transplants (Fonseca et al. 1998). A review of seagrass restoration studies concluded that only 22% of transplantation efforts resulted in a 50% or higher survival rate (Campbell 2002). In addition to low success rates, seagrass restoration methods are much more expensive than prevention, which includes adding signage and enforcement (Engeman et al. 2008).

In the Indian River Lagoon (IRL), along the east coast of Florida, the overall loss of seagrasses has been 18% since the 1940s (Virnstein et al. 2007). In some areas of the lagoon, 100% loss of seagrasses has occurred (Virnstein et al. 2007). Recently, localized increases in the dominant seagrass species *Halodule wrightii* have been correlated with a new *Crassostrea virginica* restoration technique being implemented in the area. *Halodule wrightii* beds began to establish adjacent to restored oyster reefs within 6 months after initial oyster restoration efforts (Walters, unpublished data). After 4 years, the establishment of new *H. wrightii* beds was recorded adjacent to 22.5% of the restored *C. virginica* reefs. Several studies have demonstrated enhanced growth of *H. wrightii* and other seagrass species in the presence of filter-feeding bivalves such as *C. virginica* (Reusch et al. 1994, Peterson & Heck 2001a, b, Booth & Heck 2009). Researchers attributed this to fertilizing the surrounding sediments with pseudofeces which enhance seagrass growth (Kenworthy & Fonseca 1992, Peralta et al. 2003). By filtering suspended nutrients from the water column and excreting them as psuedofeces that settle into the sediment, bivalves enriched the sediment porewater (Reusch et al. 1994, Peterson & Heck 2001a, Newell et al. 2002). This process is one of the most vital sources of nutrients for seagrasses (Fourqurean et al. 1992a, b). By creating a rich source of available nutrients, bivalves can effectively release seagrasses from nutrient limitation (Reusch et al. 1994, Peterson & Heck 2001b, Newell et al. 2002). *Crassostrea virginica* can also create a more habitable environment for seagrasses by filtering large amounts of water, subsequently increasing water clarity (Haven & Morales-Alamo 1966, Riisgård 1988). Previous research has demonstrated that oysters can have an ecosystem-wide scale effect on water clarity (Newell 1988, Mann 2000, Newell & Koch 2004). Even small decreases in light availability resulted in major declines of both seagrass growth and distribution (Ralph et al. 2006), with diminishing survival rates at seston concentrations higher than 15 mg L^{-1} (Batiuk et al. 1992). Additionally, habitat complexity is increased by the presence of oysters, and the increases in biodiversity include epiphyte grazers which decrease epiphytic loads on seagrass blades (Peterson & Heck 2001b).

The purpose of this study is to determine whether restoration of *Crassostrea virginica* is positively affecting *Halodule wrightii* recruitment in Mosquito Lagoon, Florida. The main objectives of this study are to: (1) measure the quantity of *H. wrightii* fragments on natural, restored, and dead reefs by month, (2) assess the duration of fragment retention on each reef type in summer and winter seasons, and (3) quantify the likelihood of initial fragment entanglement on each reef type.

Materials and Methods

Study site and local oyster reef restoration methodology

Fieldwork occurred in Mosquito Lagoon, which is the northernmost part of the IRL system, along the east coast of central Florida. The IRL is composed of a group of estuaries that extend along approximately 250 km of the central Florida Atlantic coast (SJRWMD 2006). It is one of the most diverse estuaries in the U.S., in part because is lies in a transition zone that includes both temperate and subtropical climates (Provancha et al. 1992, Dybas 2002). Mosquito Lagoon is a shallow, microtidal system, with a mean water depth of 1.7 m and a tidal range of 10 cm in our study area (Smith 1993, Hall et al. 2001, Steward et al. 2006). The water level in the lagoon changes with the annual rise and fall of oceanic water levels by approximately 0.3 m, and as a result, the highest water levels typically occur in October-November and the lowest water levels occur in April-May (Smith 1993, Hall et al. 2001).Water motion in Mosquito Lagoon is largely wind-driven, with normal wind speeds ranging from 0-50 km-h^{-1} that create wave heights of 10 to 30 cm (Smith 1993, Hall et al. 2001).

Mosquito Lagoon is considered the red fish (*Sciaenops ocellatus*) capitol of the world, and has become a very popular tourist destination (Scheidt & Garreau 2007). A 2005 survey found over 100,000 registered boats within counties that are adjacent to Mosquito Lagoon (IRLNEP 2007). In Mosquito Lagoon, it has been shown that boat wakes can play a role in oyster reef declines in the area (Grizzle et al. 2002, Wall et al. 2005). Water motion from boat wakes can cause live clusters and oyster shells to wash up on top of one another, which slowly push the oyster clusters above the mean high water level (Grizzle et al. 2002, Wall et al. 2005). These clusters die due to lack of inundation, eventually resulting in a dead reef, which is composed of disarticulated shells that lie flush on the benthos (Grizzle et al. 2002, Wall et al. 2005, Stiner & Walters 2008). Fifty eight of these dead reefs have been restored using oyster restoration mats (Walters, unpublished data). This type of oyster restoration involves aquaculture grade mesh mats that measure 0.25 m^2. The mats have 36 oyster shells firmly attached to them using zip-ties so that the shells are perpendicular to the benthos (Wall et al. 2005). The mats are then deployed in a quilt-like fashion on top of dead reef that has been leveled to intertidal water depth, and are held in place with cement weights. Spat have been shown to settle within days of deployment (Walters, unpublished data). Restored oyster reefs have been shown to have similar densities of live oysters (316/m^2) as natural oyster reefs after 3.5 years (Walters, unpublished data).

Seagrass fragment quantities

Monthly seagrass surveys were conducted from August 2009 to July 2010. Seven of each of the three oyster reef types (restored, natural, dead) were selected based on absence of seagrass within 10 m of the reefs. Ten 0.25 m^2 quadrats were haphazardly placed on the exposed portion of the intertidal reefs during outgoing tides and all of the seagrass present within each quadrat was collected in separate plastic bags. Later the same day, each fragment was measured for rhizome length (cm), maximum shoot height (cm), number of short shoots, and mass of fragment (all combined within each quadrat)(g).

Fragment retention

To compare the 4-wk retention of *H. wrightii* fragments on restored versus natural oyster reefs, 20 fragments for each of three attachment treatments (twist tie, woven, and control) were tested on five natural and five restored reefs (Figure 16). Dead reefs were not included in this experiment because seagrass fragments would desiccate and not survive on this reef type over the study period. Each fragment had three short shoots; fragment length ranged from 8.8 to 16.9 cm, with a mean of 11.7 cm (±1.2 SE). The twist-tie treatment involved using a twist-tie (metal wire encased in paper) to attach the fragment to either the mesh base of an oyster mat (restored reef) or a live oyster cluster (natural reef). The woven treatment involved weaving the rhizome of the fragment through the mesh or among oysters within a live oyster cluster. The control treatment involved placing fragments on top of the oyster reef to mimic natural movement of fragments in water. For restored reefs, one replicate of each treatment was haphazardly attached to 20 randomly selected mats. The 20 mats were at the same water depth as the natural reefs. For natural reefs, each treatment was present within a 0.25m^2 quadrat (20 quadrats per reef). Quadrats were placed at random along the front edge of each reef at the same height in the intertidal as the restored reefs. One fragment of each treatment was attached haphazardly within each quadrat. The initial location of each fragment was marked using flagging tape. The flagging tape was placed so it did not interfere with fragment attachment or water flow. All treatments were deployed within the same tidal cycle on all reefs. Once deployed, fragments were monitored after 24 hrs and then weekly thereafter for a 4 wk period to record retention.

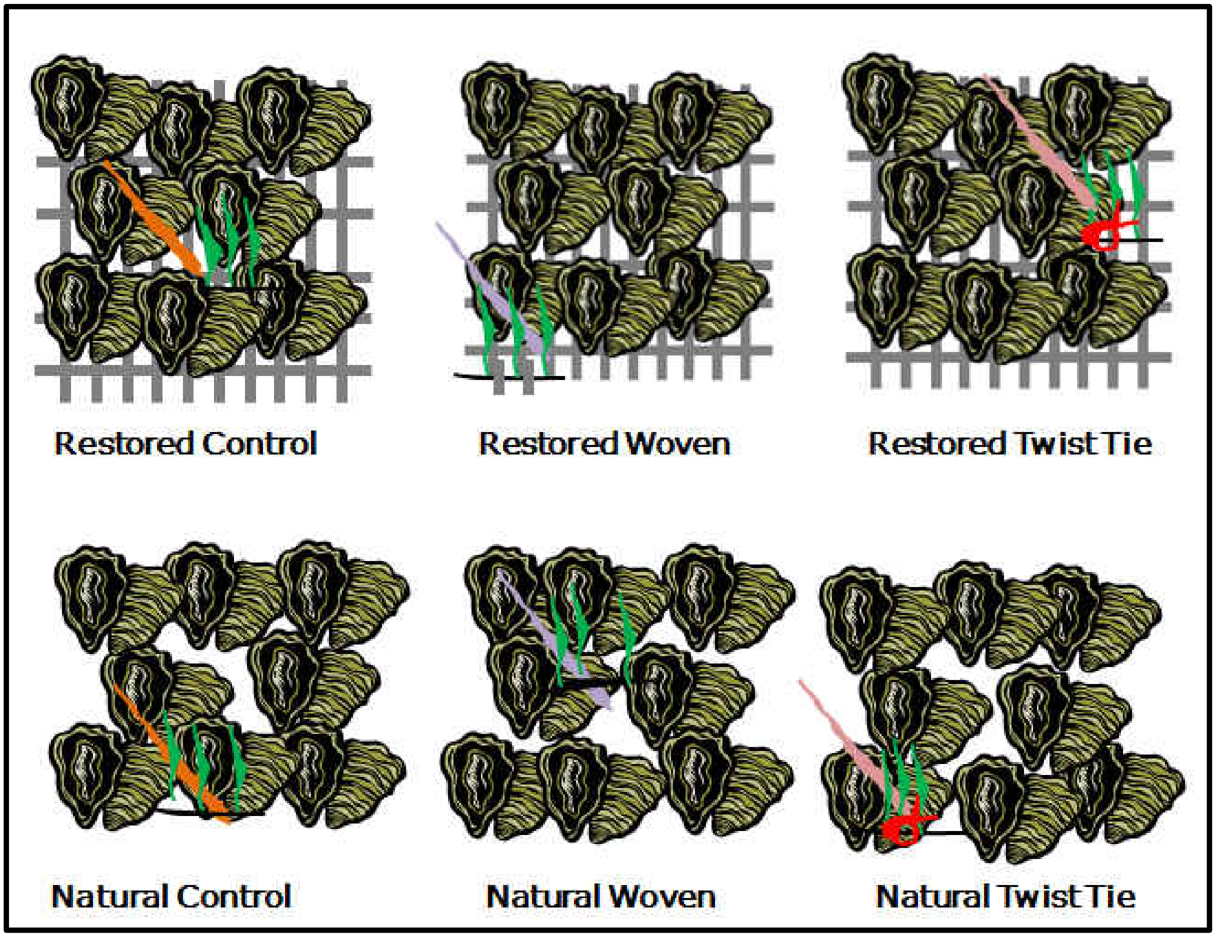


Figure 16 Illustration of the different treatments for the attachment experiment. Different colored flagging tape was used to indicate different treatments.

Fragment entanglement

To examine differences in fragment entanglement in a controlled setting, simulated restored, natural and dead oyster reefs were constructed inside a wave tank at the Florida Institute of Technology's Surf Mechanics Laboratory. The wave tank is 9.08 m in length and 57.15 cm in width. To create realistic bathymetry inside the wave tank, sediment was obtained from Mosquito Lagoon and used to form a gradual, sloping shoreline (0.3 m deep and 4.6 m long) within the tank. A water depth of 0.3 m was maintained throughout the experiment. To closely replicate waves in nature, a flapper paddle at the opposite end of the tank generated waves through an attached rotary motor. A wave height of 12.7 cm was created by the rotary motor. Shoreline sediment without oysters was used as a control. Natural reefs consisted of live oyster clusters (158 clusters/0.5 m^2) pushed into the sediment to simulate natural reef conditions. The average oyster shell height above the sediment was 11.9 cm (± 1.9). For restored reefs, two oyster restoration mats with an equal cluster density (158 clusters/0.5 m^2) was placed on the shoreline sediment, with an average height of 11.8 cm (± 1.7) above the sediment. For the dead reefs, there was a layer of densely packed oyster shells on top of the shoreline sediment, extending 98 cm and a height of 12 cm above the sediment.

For each reef type (restored, natural and dead), 20 seagrass fragments with three short shoots each were placed in the water 1 m in front of the shoreline on the mimic reef. After 2 minutes of constant wave motion (~ 100 waves), fragment locations were recorded. The entanglement rate was measured by the proportion of entangled fragments compared to the free-floating fragments in the water column. There were ten iterations for each mimic reef type.

Statistical analyses

R statistical software package (R Core Development Team 2011) was used for all statistical analyses. All data were tested using the Shapiro-Wilk test for normality and an *F*max test for homogeneity of variance. The monthly seagrass survey data was analyzed with two-way ANOVAs to compare the amount of fragment biomass on different reef types and to examine the seasonal effects on fragment availability. In the manipulative field experiment, retention of fragments was compared using a Kaplan-Meier survival analysis with the Wilcoxon statistic to compare between reef types and treatments. Additionally, final retention of fragments within all treatments was compared simultaneously with a two-way ANOVA. Tukey's HSD test was used for pairwise comparisons of reef type as appropriate. Due to non-normality of the data, wave tank fragment entanglement trials were analyzed using Kruskal-Wallis tests to compare the number of fragments of seagrass becoming entangled on different reef types. Pairwise comparisons of reef type were made using the non-parametric Wilcoxon method.

Results

During the monthly seagrass surveys, *H. wrightii* fragments found on oyster reefs varied in shoot height from 2.5 to 29.0 cm (Table 5). *H. wrightii* rhizome lengths ranged from 0.2 to 15.1 cm (Table 5). The highest number of fragments in any one 0.25 m^2 quadrat was 4, found during the month of July on a restored reef (Figure 17). Analysis of the number of *H. wrightii* fragments showed number of seagrass fragments was similar among reef types, and was significantly different among seasons (ANOVA: $p = 0.508$ and $p < 0.001$, respectively, Table 6).

Throughout the monthly seagrass surveys, we found that *Halodule wrightii* fragment biomass as measured by blotted wet weight similar among reef types (ANOVA: $p = 0.343$, Table 7). However, fragment quantities displayed a strong seasonal pattern ($p < 0.001$, Table 7) that correlates with the *H. wrightii* growing season, which peaks in summer months (Phillips 1960). No fragments were found on the oyster reefs during November - January.

Table 5 Mean ± SE by quadrat of fragment morphological characteristics among reef types by month. Data is a mean of seventy 0.25 m^2 quadrats per reef type per month.

Month	Reef Type	No. Frags		Max Rhizome Length (cm)		Max Shoot Height (cm)		No. of Short Shoots		Mass (g)		No. Growing Tips	
January	**Natural**	0.00	±0.00										
	Restored	0.00	±0.00										
	Dead	0.00	±0.00										
February	**Natural**	0.43	±0.07	2.10	±0.42	5.07	±0.86	0.83	±0.18	0.18	±0.04	0.09	±0.03
	Restored	0.50	±0.08	1.41	±0.29	5.66	±0.89	0.90	±0.17	0.19	±0.04	0.13	±0.04
	Dead	0.44	±0.08	1.44	±0.26	5.22	±0.90	0.96	±0.18	0.19	±0.04	0.10	±0.04
March	**Natural**	0.57	±0.09	1.91	±0.35	5.66	±0.82	1.18	±0.20	0.23	±0.04	0.18	±0.05
	Restored	0.64	±0.09	1.86	±0.26	7.24	±0.95	1.17	±0.17	0.23	±0.03	0.21	±0.05
	Dead	0.61	±0.09	2.06	±0.31	6.75	±0.91	1.13	±0.17	0.22	±0.03	0.14	±0.04
April	**Natural**	0.69	±0.10	2.07	±0.30	7.20	±0.91	1.17	±0.16	0.22	±0.03	0.11	±0.04
	Restored	0.66	±0.08	1.85	±0.25	7.66	±0.91	1.13	±0.15	0.21	±0.03	0.11	±0.04
	Dead	0.67	±0.09	1.70	±0.24	7.27	±0.90	1.24	±0.18	0.22	±0.03	0.10	±0.04
May	**Natural**	0.80	±0.11	2.20	±0.33	7.68	±0.91	1.31	±0.19	0.24	±0.03	0.20	±0.06
	Restored	0.81	±0.10	2.07	±0.30	8.18	±0.87	1.34	±0.18	0.25	±0.03	0.14	±0.04
	Dead	0.77	±0.09	2.20	±0.30	7.92	±0.83	1.37	±0.18	0.24	±0.03	0.16	±0.05
June	**Natural**	0.76	±0.09	1.82	±0.23	8.36	±0.92	1.30	±0.17	0.23	±0.03	0.13	±0.04
	Restored	0.77	±0.09	2.26	±0.29	8.33	±0.88	1.43	±0.18	0.27	±0.04	0.19	±0.05
	Dead	0.71	±0.09	1.95	±0.26	8.51	±0.95	1.27	±0.17	0.24	±0.03	0.07	±0.03
July	**Natural**	1.03	±0.10	2.57	±0.27	10.43	±0.88	1.67	±0.17	0.38	±0.04	0.13	±0.05
	Restored	1.09	±0.12	2.70	±0.34	9.58	±0.92	1.86	±0.24	0.39	±0.05	0.27	±0.06
	Dead	1.03	±0.11	2.24	±0.26	9.73	±0.94	1.60	±0.19	0.36	±0.04	0.10	±0.04
August	**Natural**	1.10	±0.11	2.38	±0.25	9.15	±0.84	1.63	±0.18	0.38	±0.04	0.20	±0.06
	Restored	1.07	±0.12	2.47	±0.27	9.74	±0.94	1.80	±0.21	0.43	±0.05	0.22	±0.06
	Dead	1.03	±0.11	2.40	±0.37	7.99	±0.93	1.34	±0.18	0.29	±0.04	0.17	±0.05
September	**Natural**	0.76	±0.09	2.61	±0.35	8.11	±0.90	1.29	±0.16	0.28	±0.04	0.16	±0.05
	Restored	0.86	±0.11	2.16	±0.28	8.01	±0.92	1.30	±0.17	0.26	±0.03	0.14	±0.04

Month	Reef Type	No. Frags		Max Rhizome Length (cm)		Max Shoot Height (cm)		No. of Short Shoots		Mass (g)		No. Growing Tips	
	Dead	0.77	±0.08	2.35	±0.31	7.91	±0.84	1.20	±0.14	0.24	±0.03	0.14	±0.04
October	**Natural**	0.47	±0.08	1.70	±0.35	4.97	±0.79	0.77	±0.14	0.13	±0.02	0.09	±0.03
	Restored	0.51	±0.08	1.44	±0.33	5.02	±0.72	0.74	±0.13	0.13	±0.02	0.07	±0.03
	Dead	0.41	±0.07	1.33	±0.26	3.92	±0.70	0.64	±0.12	0.12	±0.02	0.09	±0.03
November	**Natural**	0.00	±0.00										
	Restored	0.00	±0.00										
	Dead	0.00	±0.00										
December	**Natural**	0.00	±0.00										
	Restored	0.00	±0.00										
	Dead	0.00	±0.00										

Table 6 Two-way ANOVA comparing the mean biomass of *Halodule wrightii* fragments as explained by reef type (natural, restored, dead) and month by quadrat. Data is a mean of 70 0.25 m^2 quadrats per reef type per month. Surveys occurred between August 2009 and July 2010.

Source	**DF**	**Sum Sq**	**Mean Sq**	**F value**	**Pr(>F)**
Month	11	339.15	30.832	65.369	<0.001*
ReefType	2	0.64	0.319	0.676	0.508
Month*ReefType	22	1.06	0.048	0.102	1.000
Residuals	2484	1171.60	0.472		

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

Table 7 Two-way ANOVA comparing the mean number of *Halodule wrightii* fragments as explained by reef type (natural, restored, dead) and month by quadrat. Data is a mean of 70 0.25 m^2 quadrats per reef type per month. Surveys occurred between August 2009 and July 2010.

Source	**DF**	**Sum Sq**	**Mean Sq**	**F value**	**Pr(>F)**
Month	11	39.89	3.6264	55.4646	<0.001*
ReefType	2	0.140	0.0700	1.0699	0.3432
Month*ReefType	22	0.713	0.0324	0.4957	0.9758
Residuals	2484	162.408	0.0654		

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

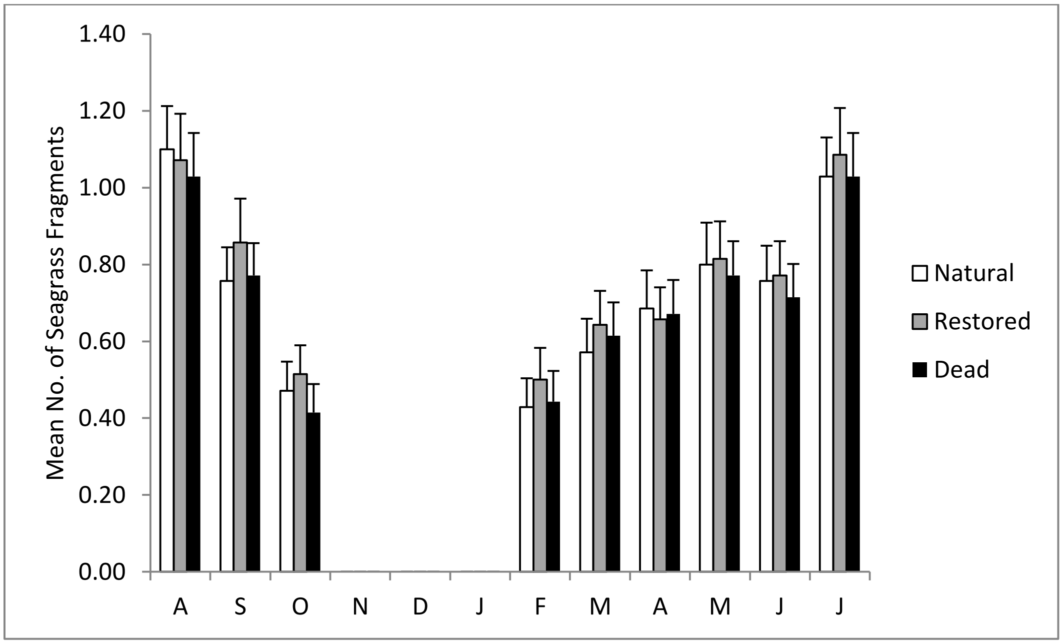


Figure 17 Mean number of seagrass fragments (±SE) per 0.25 m^2 quadrat per reef for each reef type. Surveys occurred between August 2009 and July 2010.

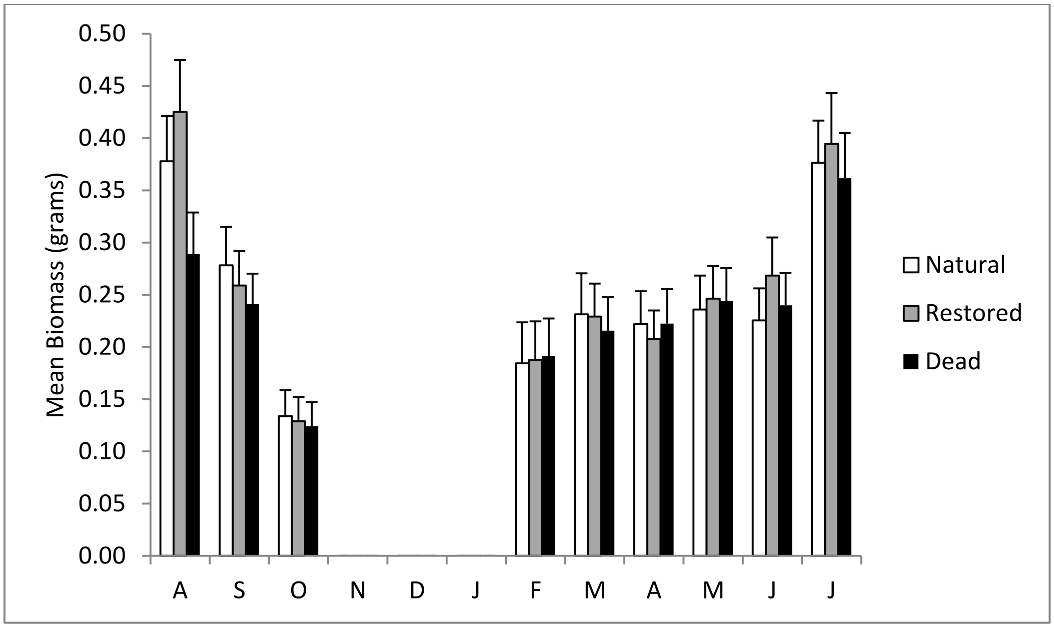


Figure 18 Mean seagrass fragment blotted-dry wet weight (±SE) combined within one 0.25 m^2 quadrat per reef for each reef type. Surveys occurred between August 2009 and July 2010.

In the fragment attachment field experiments, both reef type and attachment treatment were significantly different in both trials (ANOVA: $p < 0.001$; Figures 19, 20). Retention of entangled seagrass fragments at the end of 1 month was significantly higher on restored oyster reefs compared to natural oyster reefs (Survival Analysis: $p < 0.001$; Figures 19, 20). The interaction between reef type and treatment was also significant (Tables 8, 9). The twist-tie treatment was the most successful treatment across both natural and restored reefs, with 51% retained after 3 weeks on restored reefs (Figure 19). The final retention of the twist-tie treatment was significantly higher for both restored and natural reefs in both trials (Figure 21, 22). The woven treatment was significantly more successful on the restored reefs, with 20% retention after 2 wks compared to only 4% retention on the natural reefs (Figure 19). The fragments in the control treatment did not remain on the reefs for more than 24 hr and was not significantly different between restored and natural reefs (Figures 19, 20).

Table 8 Two-way ANOVA of final fragment retention experiment data (February 27 – March 27 2010). Data was a sum of the total time spent attached the reef for each fragment within each reef type.

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Reef Type	1	43.20	43.20	26.319	<0.001*
Treatment	1	715.56	715.56	435.930	<0.001*
Reef*Treat	1	26.52	26.52	16.158	<0.001*
Residuals	596	978.31	1.64		

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

Table 9 Two-way ANOVA of final fragment retention experiment data (June 26 - July 24 2010). Data was a sum of the total time spent attached the reef for each fragment within each reef type.

Source	DF	Sum Sq	Mean Sq	F value	Pr(>F)
Reef Type	1	41.08	41.08	26.507	<0.001*
Treatment	1	635.04	635.04	409.746	<0.001*
Reef*Treat	1	27.04	27.04	17.447	<0.001*
Residuals	596	923.70	1.55		

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

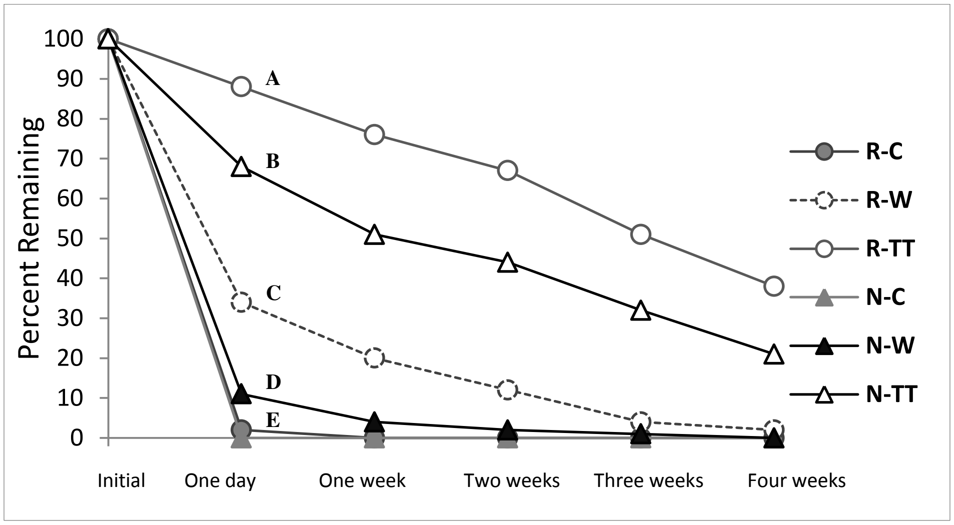


Figure 19 Results from the fragment retention experiment (February 27 – March 27 2010). R=restored, N=natural, C=control treatment, W=woven treatment and TT= twist-tie treatment. Retention of fragments was compared using a Survivor Analysis. Treatments with different letters are significantly different at $p < 0.05$ as determined by a Tukey’s post hoc test.

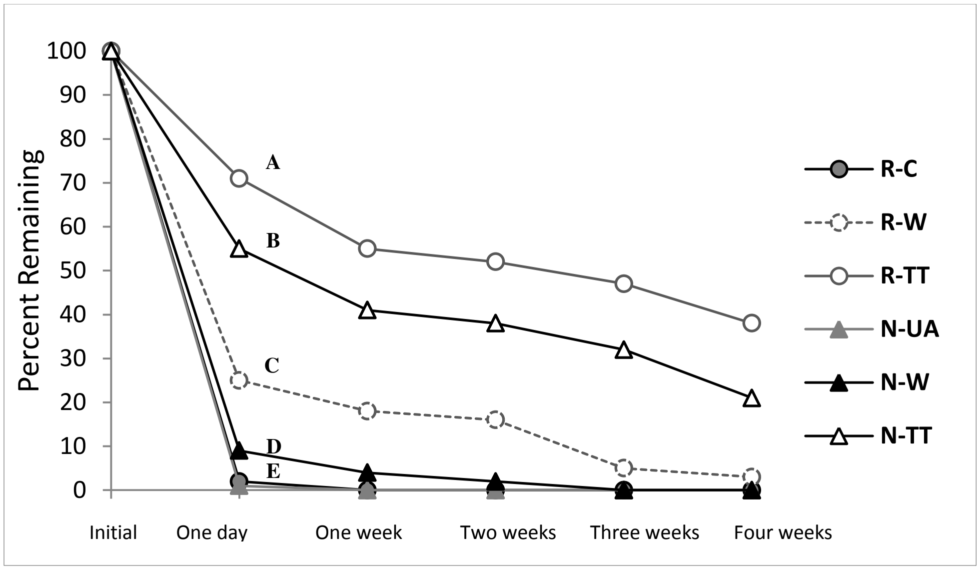


Figure 20 Results from the fragment retention experiment (June 26 - July 24 2010). R=restored, N=natural, C=control treatment, W=woven treatment and TT= twist-tie treatment. Retention of fragments was compared using a Survivor Analysis. Treatments with different letters are significantly different at $p < 0.05$ as determined by a Tukey's post hoc test.

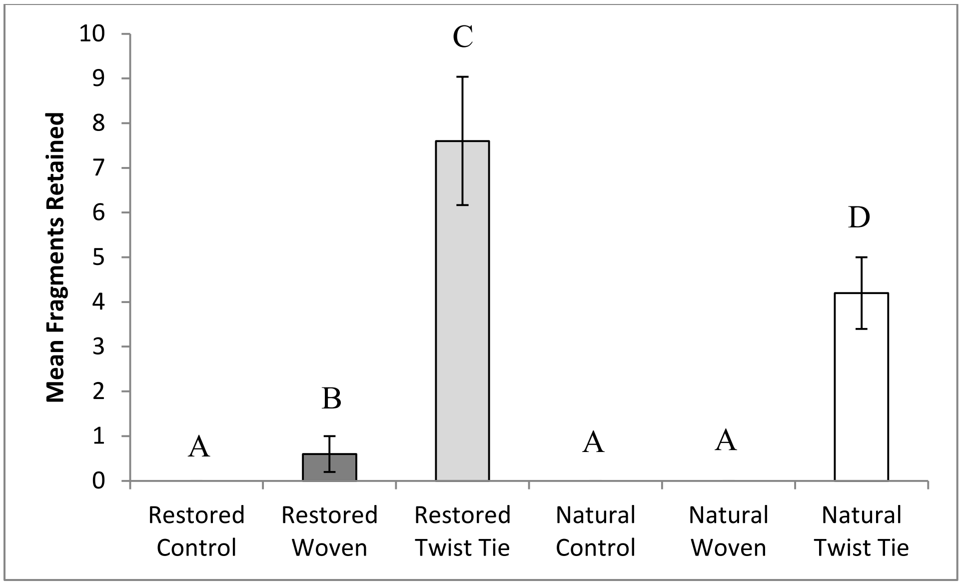


Figure 21 Results from the fragment retention experiment (February 27 – March 27 2010). Final retention (±SE) was compared using a two-way ANOVA. Treatments with different letters are significantly different at $p < 0.05$ as determined by a Tukey's post hoc test.

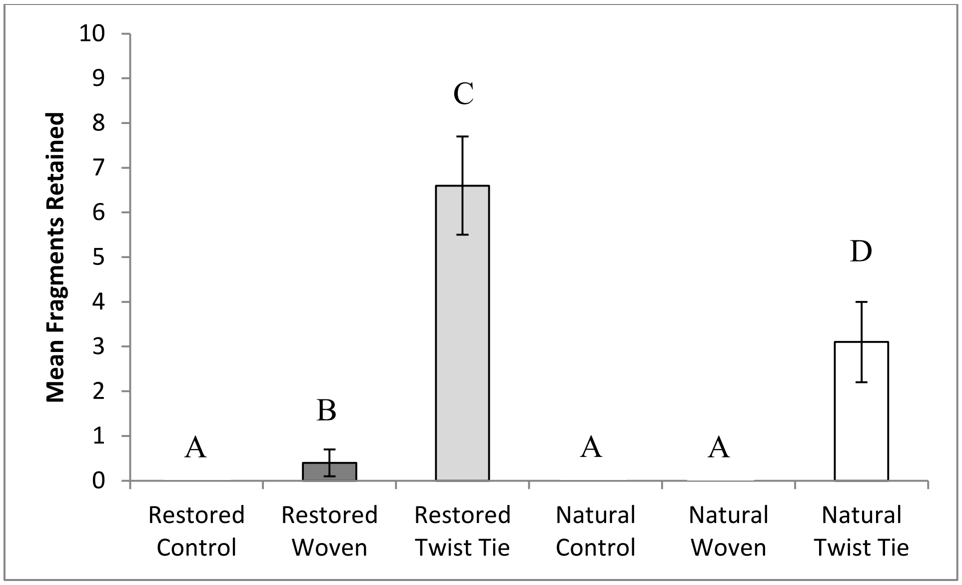


Figure 22 Results from the fragment retention experiment (June 26 - July 24 2010). Final retention (±SE) was compared using a two-way ANOVA. Treatments with different letters are significantly different at $p < 0.05$ as determined by a Tukey's post hoc test.

In the wave tank experiment, seagrass fragment entanglement was significantly different among treatments (Kruskal-Wallis: Chi-square = 187.9, DF = 2, $p < 0.001$, Figure 23). Restored reefs had an average of 11.7 (±1.7 SE) entangled fragments out of 20 released fragments, an entanglement rate of 58.5% (Figure 23). Natural reefs had an average of 8.2 (±1.6 SE) entangled fragments, an entanglement rate of 41.0% (Figure 23). Significantly fewer fragments became entangled in the dead reef and no oyster treatments (4.1±1.7 SE and 0.4±0.6 SE respectively, Figure 23). Many of the seagrass fragments were washed above the water line in the dead reef and no oyster treatments, which could lead to fragment desiccation and subsequent death during low tides.

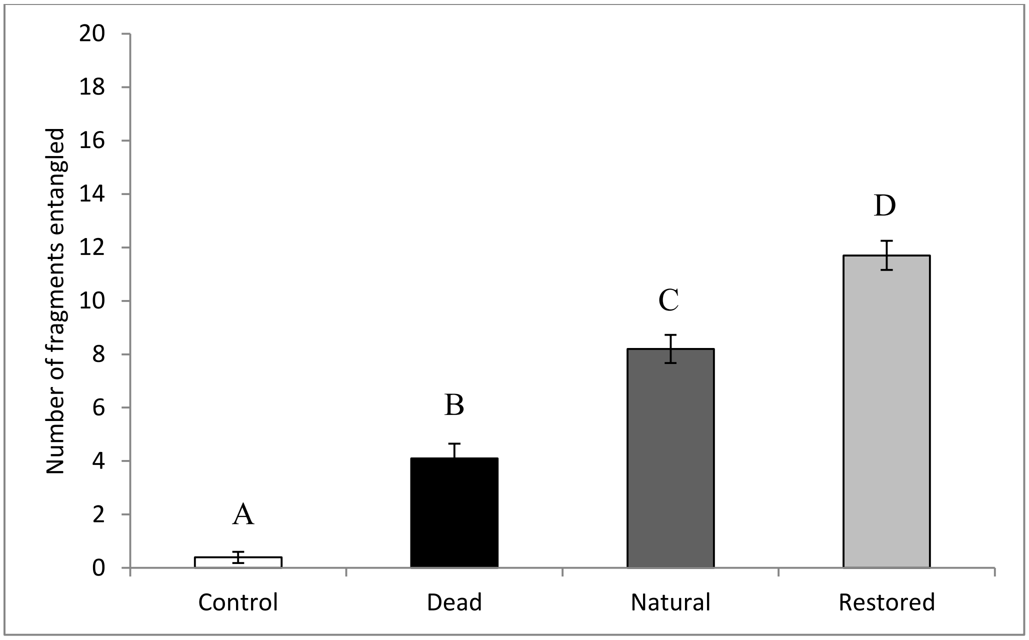


Figure 23 Fragment entanglement as explained by reef type in wave tank. Restored and natural treatments had same number of live oyster clusters. Entangled fragments were compared between treatments using a Kruskal-Wallis test. Treatments with different letters are significantly different at $p < 0.05$ using the non-parametric Wilcoxon method.

Discussion

The results from these experiments suggest that oyster restoration positively effects vegetative fragment recruitment of *H. wrightii* in Mosquito Lagoon. Vegetative fragmentation is an important dispersal strategy for seagrasses that lack seed banks, such as *H. wrightii* (McConchie & Knox 1989, Campbell 2003, Hall et al. 2006a). Researchers detected a rapid recovery of *H. wrightii* in the IRL after an existing seagrass bed died off completely (Morris & Virnstein 2004). The recovery initially occurred in small patches, which suggested vegetative recruitment by *H. wrightii* fragments as the recovery mechanism. Another IRL study found that vegetative fragments of *H. wrightii* can remain viable during spring for up to 4 wks, which is important for dispersal over long distances (max: 9 km/day) (Hall et al. 2006a). Fragments of *H. wrightii* have been shown to be successful in settling to the sediment level and subsequently rooting themselves within the duration of viability in mesocosms (Hall et al. 2006a). The potential dispersal distance of fragments also depends on factors such as tidal currents and wind (Hall et al. 2006a). In our study, the number and biomass of seagrass fragments was similar on the three reef types, and seasonal differences in availability of fragments was significant ($p<0.001$, Tables 6, 7). *H. wrightii* has its peak growing season during the spring and summer months, and then becomes dormant during the late fall and winter (Morris et al. 2001). Hall et al. (2006a) found that *H. wrightii* was more effective at vegetative fragment recruitment in spring compared to fall.

Fragments of *Halodule wrightii* were significantly more likely to be retained longer by the restored reefs compared to the natural and dead reefs ($p<0.001$, Figures 19, 20). Restored reefs may provide needed additional structure for fragment entanglement due to the mesh mats that are part of the assembly of these restored reefs. The control treatment did not remain attached for more than 24 hr, suggesting that fragments need some sort of entanglement to maintain their position.

Enhanced fragment retention may also be the result of increased sediment stabilization due to the construction of the restored reefs, which consists of oyster restoration mats held in place with cement weights. Many studies have shown that oyster reefs can change the energy flow rates of waves, resulting in increased sediment stabilization near the oyster reef (Dame et al. 1992, Dame 1996, Newell & Koch 2004, Chapter 3). In Tampa Bay, researchers have found that the presence of longshore sandbars have historically sheltered seagrass beds from destructive wave impacts, such as large boat wakes (Lewis et al. 1985, Lewis 2002). Due to erosion, longshore bars have been lost in numerous areas of the bay and this is believed to have added to the extensive seagrass losses in Tampa Bay over the last 50 years (Lewis 2002). Currently, the Tampa Bay Estuary Program (TBEP) in conjunction with the Tampa Port Authority (TPA) is investigating whether the construction of artificial longshore bars can be used as a potential management option to increase seagrass coverage and to protect current seagrass areas from high wave energy (Lewis 2002, Cross et al. 2011). Experimental longshore bars have improved the stability of the sediment and facilitated an increase in seagrass coverage compared to pre-construction coverage (Tampa Port Authority 2011).

In many species of seagrasses, as well as seaweeds, entanglement is essential to the successful attachment and growth of vegetative fragments (Kilar & McLachlan 1986, Harwell & Orth 2001, Herren et al. 2006). We found that *Halodule wrightii* fragments are significantly more likely to become entangled in restored reefs (Figure 23). This is potentially the result of the added three-dimensional structural complexity of the restored oyster reefs. Studies have found that structural complexity leads to increased accumulation and retention of vegetative fragments compared to less complex structures, thus creating a significant increase in the capture of potential recruits (Davis et al. 2009). Balestri et al. (2011) found that fragments of the seagrass *Posidonia oceanica* that became entangled in submerged artificial reefs had 80-96% survival and could theoretically re-establish in that area. Researchers have also found that *P. oceanica* fragments had higher success rooting in the valleys between cement rubble mounds compared to the fragments that were located outside the valleys (Di Carlo et al. 2005, Di Carlo et al. 2007).

This is thought to be the result of increased protection from wave action as well as enhanced sediment deposition near the mounds (Di Carlo et al. 2005, Di Carlo et al. 2007).

This study provides evidence of enhanced seagrass recruitment near restored oyster reefs in Mosquito Lagoon. These findings complement many other studies that have also found a facultative mutualism between oysters and seagrasses (Newell & Koch 2004). Newell and Koch (2004) suggested that coordinating seagrass and oyster restoration efforts could improve the recovery of seagrasses in the restoration area. *H. wrightii* began to re-establish within 6 months after initial oyster restoration at some sites in Mosquito Lagoon. To date, 22.5% of restored oyster reefs have newly established seagrass beds within 10 m of the reef (Walters, unpublished data). Our restoration technique provided additional substrate for fragment entanglement and retention. Fragments were retained at a higher rate and for a longer duration by restored reefs compared to natural reefs, thus encouraging rooting. Restoration mats also increased sediment stability and dissipated more wave energy compared to dead reefs or areas without oysters (Chapter 3). Given that oysters also increase water clarity and nutrient availability in shallow systems, the presence of oyster reefs in an estuary can potentially boost seagrass establishment and growth (Booth & Heck 2009). Seagrass restoration is very labor intensive and expensive, so enhancing vegetative recruitment via oyster restoration is an ideal and cost-effective way to help support seagrass populations while also providing habitat for oysters. Understanding the interaction between seagrass and oyster reef restoration will enable coastal resource managers to develop better conservation plans for both of these species.

CHAPTER 5: GENERAL DISCUSSION

With ecosystems being degraded at an alarming rate, many natural resource managers turn to habitat restoration to help reestablish natural habitats. Habitat restoration takes place in all ecosystems, including marine systems such as salt marshes, mangroves, and oyster reefs. Some restoration projects are unsuccessful due to lack of initial scientific research before the restoration project as well as a lack of continued monitoring after the restoration. With multiple restoration techniques available and many projects occurring across the globe, we need comparable metrics to judge the success or failure of our restoration projects (Coen & Luckenbach 2000). This project focuses on three important metrics of restoration: the creation of accurate historical baseline data, the retention of ecosystem services by the restoration project, and how the restoration project affects other important species within the ecosystem.

In any restoration project, it is important to establish a baseline as a goal for the project. Some habitat loss is greatly exaggerated or underestimated due to lack of historical abundance data. With the introduction of GIS, the creation of accurate historical abundance data has become feasible for both small scale and large scale restoration projects across many different habitat types. Specifically, historical data for oyster abundance has shown that current oyster abundance is 15% of its historical coverage (Beck et al. 2011). However, this number is highly variable for local oyster populations. In Mosquito Lagoon, we have shown that oyster losses are 23% of the historical coverage in 1943. Most of these losses are located along main boating channels, which are now the focus of a long term restoration project. The loss of natural reef coverage is higher with CANA park boundaries (40%). Using the information obtained during this study, CANA park managers can set more realistic goals and focus restoration effects on key areas within the lagoon.

Another important metric for restoration projects is the retention of important ecosystem services associated with the natural habitat that is being restored. Many restoration projects focus on ecosystem engineers because the loss of the ecosystem services they provided is negatively impacting both the environment and the people that utilize that environment. Often times, restoration projects involve recreating lost habitat through a wide variety of techniques. Oyster restoration projects often involve artificial structures to provide substrate for oyster spat settlement, such as reef balls, oyster grates and oyster mats. Beyond simply providing habitat for oysters, these structures need to provide the ecosystem services associated with natural oyster reefs, such as serving as habitat for local biota, improving water clarity via filtration and protecting shorelines by attenuating wave motion (Coen et al. 2007). Our study found that restored oyster reefs are able to attenuate waves similar to natural oyster reefs. Thus, by restoring oyster reefs, we are providing additional oyster habitat and maintain shoreline integrity behind the reef.

Lastly, the impact that the restoration has on other important species within the local environment is an important criteria for a successful project. In Mosquito Lagoon, we have found that 22.5% of restored oyster reefs have newly established seagrass beds adjacent to the restored reef (Walters, unpublished data). In this study, we have found that our oyster mat restoration technique provided additional substrate for enhanced fragment entanglement and retention. Restoration mats also increased sediment stability and dissipated more wave motion compared to dead reefs or bare sediment. Thus, restored oyster reefs could be a potential management option for both increasing oyster abundance and encouraging seagrass growth in the area.

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