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STRATEGIES FOR SUCCESSFUL MANGROVE LIVING SHORELINE STABILIZATIONS
IN SHALLOW WATER SUBTROPICAL ESTUARIES

by

REBECCA FILLYAW
B.S. University of Central Florida, 2017

A thesis submitted in partial fulfillment of the requirements
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Major Professor: Linda Walters

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ABSTRACT

Mangrove living shorelines are an effective alternative to hard-armoring, which combat erosion while also increasing habitat. To improve the success of future mangrove deployments, an experimental *Rhizophora mangle* living shoreline was deployed within Mosquito Lagoon, FL. A factorial design was used to test the impact of mangrove age, breakwater presence, and mangrove placement on mangrove survival and growth. Environmental factors were monitored to isolate the reason for mangrove mortalities. Mangrove age was represented by 3 developmental stages: “seedlings” at 11-months-old, “transitionals” at 23-months-old, and “adults” between 35 and 47-months-old. Mixed mangrove age groups were included to identify if seedling survival could be facilitated by the presence of transitionals and adults; control groups were used to test the impact of restoration materials on recruitment of wrack and mangrove propagules. The majority of mangrove mortalities (62%) occurred 2 months after the onset of high-water season and these dead mangroves showed signs of flooding stress. Breakwaters alleviated stress through the reduction of water velocity and wave height, and increased the odds of survival by 197% and 437% when mangroves were planted in the landward and seaward rows, respectively. Due to their larger stems and greater number of prop roots, older mangroves were better able to survive; compared to seedlings, transitionals increased survival odds by 186% and adults by 1087%. For treatments composed of adults and a breakwater, 88% of the mangroves survived and 64% of these survivors produced flowers or flower buds by 12 months after the restoration. Planting seedlings haphazardly among older mangroves did not attenuate enough wave energy to significantly increase seedling survival, and the complexity of restoration materials did not significantly impact propagule or wrack abundance.

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CHAPTER 1: INTRODUCTION

Historical efforts to stabilize shorelines have focused on the hard-armoring of extensive portions of coastlines with artificial structures such as seawalls, jetties, and breakwaters (e.g., Dugan et al., 2011; Gittman et al., 2016; Peterson et al., 2019). As a result, 14% of shorelines in the United States are hard-armored, with 64% of that total occurring in estuaries and lagoons (Gittman et al., 2016). Unfortunately, these methods have caused loss of natural habitats through direct removal of native plants, increased scouring adjacent to the structures, shading, and competition with exotic species (e.g., Bozek and Burdick, 2005; Beck and Airolidi, 2007; Bulleri and Chapman, 2010; Heerhartz et al., 2015). Overall, seawalls support 23% less biodiversity and 45% fewer organisms when compared to natural shorelines (Gittman et al., 2016). Additionally, seawalls reduce the ability of plant communities to migrate landward as sea levels increase, resulting in further habitat loss over time (Doody, 2004; Pontee, 2013; Phan et al., 2015).

“Living shoreline” is the term for shoreline stabilizations that use natural materials such as native vegetation to reduce erosion while also providing habitat (Currin, 2019). This method of restoration allows for habitat migration over time, wildlife movement between terrestrial and marine habitats, and increased wave attenuation as the vegetation grows larger (Bilkovic et al., 2016). For areas of high wave energy, structural components such as breakwaters are often placed in front of the planted vegetation to aid their survival (e.g., Bilkovic et al., 2016; Moosavi, 2017; Hardy and Wu, 2020). The presence of a breakwater has an impact on wave energy by lowering wave height and reducing incoming velocity (Losada et al., 2005; Spiering et al., 2018). For example, an oyster shell bag breakwater was reported to decrease near-bed velocity by 62%

and to reduce wave height by 42% when the water level was 5 cm above the structure (Spiering et al., 2018).

In tropical and subtropical areas, mangroves are frequently used in living shoreline stabilization efforts, are considered a foundational taxon, and are used by over 1300 animal species for shelter, foraging, and nesting (e.g., Baran and Hambrey, 1999; Ellison et al., 2005; Rusnak, 2016). Increasing areas of mangrove habitat, therefore, has the potential to increase local biodiversity and provide multiple ecosystem services, including fisheries production, carbon sequestration, and ecotourism (e.g., Carlton, 1974; Faunce and Serafy, 2006; Estrada et al., 2014; Gorman and Turra, 2016; Spalding and Parrett, 2019). Many mangrove species include complex, above-ground root structures that slow water movement, capture suspended sediments, and provide microhabitats for invertebrates and fish (Carlton, 1974; Zhang et al., 2019). Moreover, McClenachan et al. (2020) demonstrated that combining the results of multiple small-scale mangrove living shoreline projects reversed system-wide erosion patterns. In their example, 14 mangrove living shoreline deployments ranging from 104 to 327 m in length and 2-7 years in age resulted in a net shoreline gain of $347.62 \text{ m}^2 \text{ yr}^{-1}$ in a Florida estuary.

Rhizophora mangle (red mangrove) is frequently used in living shoreline and mangrove restoration efforts in the Southeastern United States, Caribbean, and Central America. (e.g., Teas, 1997; Winterwerp et al., 2013; Peters et al., 2015; Bayraktarov et al., 2016; Donnelly et al., 2017). Successful planting of *R. mangle* on Florida's east coast has traditionally had a northern limit of approximately 40 km south of Fort George Inlet, where the natural expansion of *R. mangle* populations have been restricted by the frequency of freeze events that drop below -4° C (Kangas and Lugo, 1990; Cavanaugh et al., 2014; Cavanaugh et al., 2019). Compared to the other mangrove species native to Florida [*Avicennia germinans* (black mangrove), *Laguncularia*

racemosa (white mangrove)], *R. mangle* is able to settle and survive amid greater magnitudes of flooding due to larger propagules and interspecific differences in root aeration (McKee, 1993, 1996; Elster, 2000). According to data collected from Tampa Bay, FL, *R. mangle* occupy elevations ranging from +0.06 to +0.49 m, where the base of the mangroves are flooded on average 30% each day (Lewis, 2005). In order to imitate the observed hydrology of naturally-recruited fringe mangrove stands, *R. mangle* used for living shorelines are planted in the middle to high intertidal zone (Primavera and Esteban, 2008; Samson and Rollo, 2008; Donnelly et al., 2017). Frequent inundation alleviates high pore-water salinity and increases phosphorus abundance, but extended periods of submersion can deplete a mangrove's stored oxygen, negatively impacting survival and growth through the accumulation of ethanol (Ball, 1988; Ball, 1998; Krauss et al., 2006; Lara and Cohen, 2006). *Rhizophora mangle* can withstand a greater range of flooded conditions as they get older due to larger stems and the growth of prop roots above the sediment surface, both of which have lenticels and aerenchyma for the intake and storage of oxygen (Tomlinson, 1986; Ball, 1988).

Wave energy contacting the portion of a mangrove submerged in water is a primary source of seedling mortality; individuals can be lost through dislodgement or failure at the stem (e.g. Balke et al., 2011; Boizard and Mitchell, 2010). Wind wave energy is produced based on wind speed and direction, bathymetry, and fetch. It can be enhanced by nearby boating activity, with resulting boat wakes contacting the shorelines (e.g. Gorman and Neilson, 1999; Bilkovic et al., 2017; Walters et al. 2021). Boizard and Mitchell (2010) found that the probability of dislodgement of seedling *R. mangle* by wave energy was inversely related to grain size; in their study, *R. mangle* anchored 3.5 times better in coral rubble than sand. Areas that are solely made up of small grains can vary in soil strength based on cohesiveness, and this impacts plant

anchorage (Schutten et al., 2005). The shear strength of the sediment inhabited by mangroves spans from 2.5 to 46 kPa (Cahoon et al., 2003).

Sediment accretion and erosion, which is influenced by the amount of wave energy at a site, can have an impact on young mangrove survival. Pilato (2019) showed that removal force of *R. mangle* seedlings increased by 0.20 N for every gram increase in root biomass. As the sediment around a mangrove erodes, the buried root biomass decreases, and less wave energy is required to displace the plant (Bywater-Reyes et al., 2015). Previous research, however, also indicates that accretion of sediment can lead to hypoxic conditions that result in mangrove mortality (Craighead and Gilbert, 1962; Terrados et al., 1997). For example, survival of 6-month-old planted *Avicennia marina* (grey mangrove) seedlings was significantly impacted by sediment accretion once burial reached 14 cm above the original sediment level at time of planting (Kamali and Hashim, 2011). Additionally, seedling *Rhizophora apiculata* (tall-stilt mangrove) experienced a 3% increase in mortality rate for every cm of sediment added, and there was 0% survival for the 32 cm of additional sediment treatment at the 321-day mark (Terrados et al., 1997).

Coexisting with other vegetation has proven to have both negative (competition) and positive (facilitation) impacts on mangrove recruitment, survival, and growth (McKee et al., 1988; Farnsworth and Ellison, 1996; Donnelly and Walters, 2014; Teutli-Hernandez et al., 2019). Surrounding vegetation can influence young mangroves negatively by reducing light availability (Farnsworth and Ellison, 1996). *Rhizophora mangle* was once considered a shade-tolerant mangrove species due to their ability to establish as propagules and grow to the seedling stage under shaded conditions (Sousa et al. 2003). However, further research revealed that *R. mangle* required canopy openings resulting in at least 20% light availability to proceed from a

seedling to a juvenile (López-Hoffman et al., 2007). The presence of mature mangroves encourages propagule recruitment through increased surface complexity and decreased wave energy (Donnelly et al., 2017), and seedling mangroves can benefit from establishing near *A. germinans* and *R. mangle* secondary roots since they increase soil redox potential and lower sulfide concentrations (McKee et al., 1988). The presence of vegetation such as *Batis maritima* (saltwort) and *Sarcocornia perennis* (glasswort) was shown to have a positive impact on *R. mangle* establishment by increasing propagule retention time, reducing interstitial salinity, and increasing nutrients such as carbon, nitrogen, and phosphorus (e.g. Donnelly and Walters, 2014; Teutli-Hernandez et al., 2019). Wrack (collections of decaying organic matter) have also been observed to retain propagules of the 3 Florida mangrove species (Pinzón et al., 2003; Ruiz-Delgado et al., 2014; Breithaupt et al., 2019; Smith et al., 2020). Moreover, if wrack abundance is not great enough to smother mangroves, wrack presence can lead to increased growth due to the nutrient additions (Chapman and Roberts, 2004; Breithaupt et al., 2019).

Mangrove survival is crucial for reversing patterns of shoreline erosion and providing natural habitat (Faunce and Serafy, 2006; McClenachan et al., 2020), but many mangrove living shoreline projects have reported low levels of survival even with a breakwater present (Riley and Kent, 1999; Primavera and Esteban, 2008; Hashim et al., 2010; Tamin et al., 2011; Motamedi et al., 2014; Cuong et al., 2015; and Jayarathne et al., 2020). For example, a living shoreline in Malaysia costing \$175,000 per 0.01 km², that utilized a breakwater and planted 1030 *A. marina* and *R. apiculata* seedling mangroves (height: ~20 cm), reported a survivability index of 5% (Motamedi et al., 2014). A separate living shoreline in Malaysia, costing a total of \$85,000, utilized a breakwater and planted *A. marina* saplings (~40 cm) in coir logs; the restoration had 30% survival after 8 months (Hashim et al., 2010). A review paper by Kodikara et al. (2017)

revealed that out of 67 mangrove plantings in Sri Lanka, 97% of which were *Rhizophora spp.*, 37 of the deployments resulted in 100% mortality. The reported reasons for these mortalities included drought, flooding, smothering by wrack, browsing and trampling by vertebrates, and infestation by insects and barnacles (Kodikara et al., 2017).

As demonstrated above, living shorelines can be expensive to deploy, and few studies start with pilot experiments to test different living shoreline designs at each deployment site and monitor them closely enough to identify the reason(s) for failures (Myszewski and Alber, 2016; Morris et al., 2018). In order to fill this gap and explore mangrove success when used in living shoreline stabilization in a shallow, subtropical estuary, I asked: 1) How does initial mangrove age, breakwater presence, and mangrove placement impact mangrove survival and growth? 2) Which structural characteristics of mangroves were most influential for survival? 3) What was the source of observed mangrove mortalities? 4) Is seedling survival enhanced by being planted with older mangroves? 5) How did the living shoreline impact local wrack and mangrove propagule abundance?

CHAPTER 2: METHODS

Study Site

Mosquito Lagoon is located on the east coast of central Florida and makes up the northernmost portion of the Indian River Lagoon (IRL) system. The IRL is classified as one of the most biodiverse estuaries in the continental United States, which supports over 4,000 species of plants and animals (Dybas, 2002). This area experiences an annual high water season each fall, and water movement is primarily wind-driven (Smith, 1987, 1993; Brockmeyer et al., 1996). An experimental living shoreline was deployed in Mosquito Lagoon within the boundaries of Canaveral National Seashore (CANA) (Fig. 1). The 30 m of shoreline between the 2 sections was not stabilized because this stretch had no obvious erosion due to cover of mature *A. germinans*.

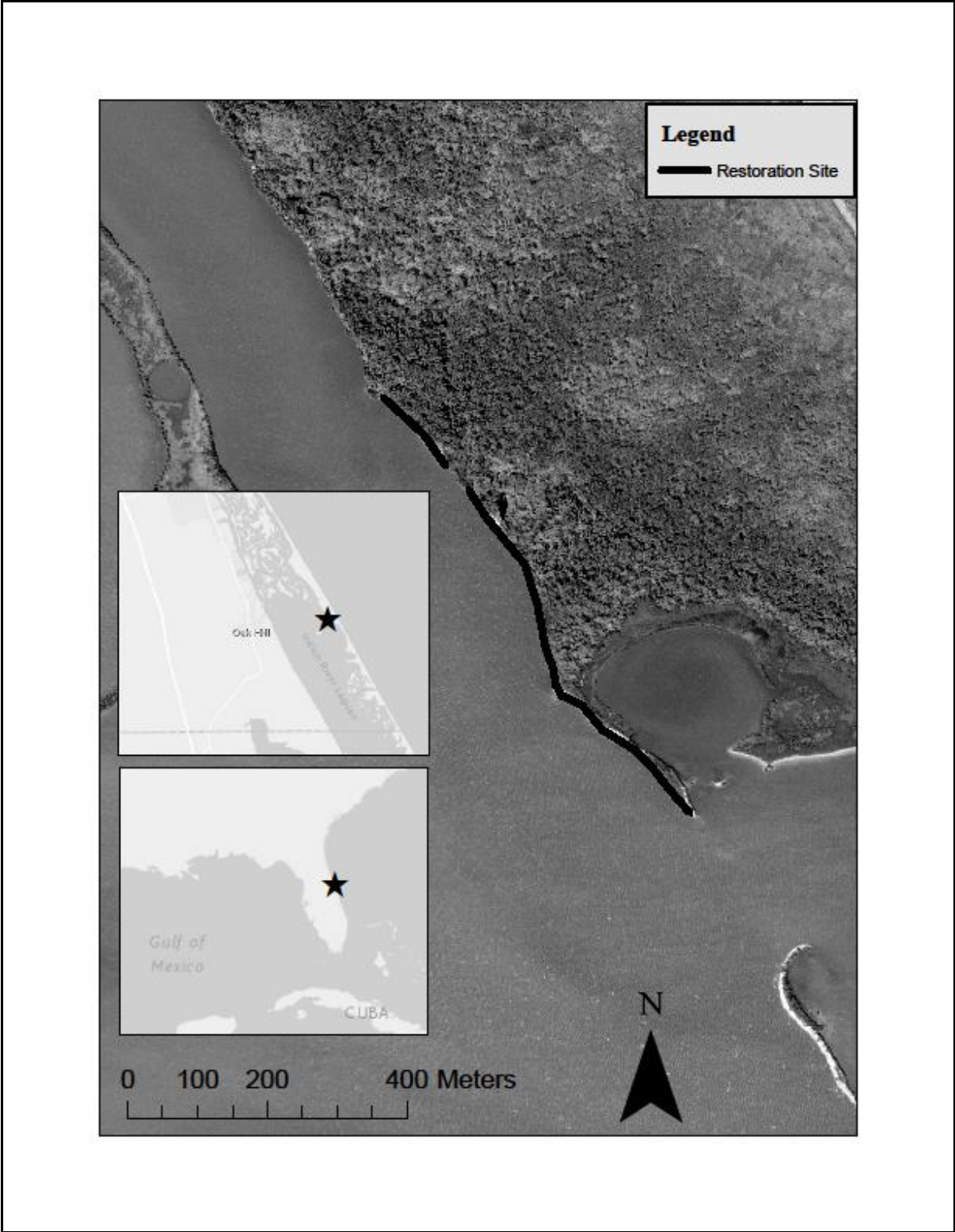


Figure 1: Location of restoration site in Canaveral National Seashore in the Indian River Lagoon system on the east coast of Florida, USA.

The experimental living shoreline was planted along a shell-dominated shoreline once occupied by the Timucuan people (800 to 1400 CE) (National Park Service, 2020). Tribes harvested large amounts of oysters (*Crassostrea virginica*) and clams (*Mercenaria mercenaria*), discarding the empty shells in large piles (middens) along shorelines in Mosquito Lagoon (Donnelly et al., 2017). These shell middens contain culturally significant items, including broken pottery and animal bones (National Park Service, 2020). The US National Park Service is dedicated to protecting these historic sites with as little disturbance as possible, and stabilizing this area using living shoreline techniques directly supports this goal. Due to the shelly substrate, it is difficult for mangroves to naturally recruit to the shoreline; a total of 4 *R. mangle* developed into mature trees along the 650 m of adjacent shoreline (Donnelly et al, 2017).

Experimental Design and Restoration

To test the efficacy of different living shoreline designs, an experimental living shoreline was deployed between 14 and 21 June 2019. In total, 1,050 oyster shell bags were deployed as breakwaters and 640 *R. mangle* planted with the help of 51 volunteers (324 volunteer hours). Oyster shell bags were constructed from DelStar Technologies Naltex nylon mesh filled ~3/4 full with recycled oyster shells (18.9 L) collected from restaurants and quarantined outdoors at Marine Discovery Center in New Smyrna Beach, FL for a minimum of 6 months. Individual bags were 1 m long, 0.4 m wide, weighed approximately 18 kg, and were hand-knotted at each end. The breakwater consisted of 25 units based on Florida Department of Environmental Protection permitting requirements, with each unit no more than 6.6 m in length and a minimum of 6.6 m stretches between each unit to enable wildlife movement. Shell bags were never placed closer than 1 meter to any existing seagrass (*Halodule wrightii*) or other wetland plants. Each

breakwater unit consisted of 2 stacked rows of 21 oyster shell bags (total shells bags/unit = 42), attached together with cable ties (304 x 7 mm).

Rhizophora mangle used for the stabilization were collected as propagules from over 100 trees within the boundaries of CANA and grown at the University of Central Florida greenhouse in Orlando. Propagules were planted in 3.7 L pots with topsoil for approximately 1 year and then transferred to 11.3 L pots with additional topsoil. These pots were kept in shallow, plastic pools filled approximately to 14 cm with freshwater.

Rhizophora mangle used in the experimental living shoreline were separated into 3 developmental stages based on known plant ages and observations of the mangrove stems at the time of deployment. Mangroves were either seedlings at 11 months-old, transitional plants at 23 months-old (hereafter referred to as “transitionals”), and adults that ranged in age from 35 to 47 months-old. These developmental stages were identified by the percentage of woody tissue on the stem. Seedlings had 0%, transitionals had between 25 and 75%, and adults had 100% woody tissue.

A factorial design was used to test all combinations of mangrove developmental stages with the presence or absence of a wave break for a total of 10 treatments along the experimental living shoreline: seedlings only, seedlings with a breakwater, transitionals only, transitionals with a breakwater, adults only, adults with a breakwater, mixture of the developmental stages, mixture of the developmental stages with a breakwater, no mangroves (control), and no mangroves (control) with a breakwater. Each treatment with a mixture of developmental stages had between 5 and 7 seedlings, a minimum of 3 adults, and a minimum of 3 transitionals; however, the exact ratio of seedlings to transitionals to adults and the placement of each developmental stage within the treatment replicate was haphazard (Table 1). This planting scheme was intended to imitate a

restoration strategy that uses a haphazardly deployed mixture of developmental stages with the goal of increasing seedling survival.

Each treatment was replicated 5 times along the shoreline. The placement of each treatment along the shoreline was randomly determined prior to restoration using a random number generator (*random.org*) (Table 1). For treatments with a breakwater, shell bags were placed 1 meter seaward of the planted *R. mangle*. Treatments with *R. mangle* included 16 plants in 2 staggered rows of 8 at an elevation inhabited by the closest naturally recruited adult *R. mangle*. Within each treatment replicate, mangroves were centered with approximately 0.7 m distance to adjacent mangroves. One week after the deployment, all mangroves were checked to ensure the root balls and topsoil from the pots were completely buried by sediment. Three *R. mangle* (0.4% of total) did not meet this standard and were replaced.

Table 1: Location, developmental stage, and breakwater status of each treatment replicate.

Section #	X-Coordinate	Y-Coordinate	<i>R. mangle</i> Treatment	Oyster Shell Bag Treatment
1	-80.789387	28.867135	16 Transitional	Yes
2	-80.789464	28.867207	6 Seedling, 7 Transitional, 3 Adult	Yes
3	-80.789530	28.867289	16 Adult	Yes
4	-80.789603	28.867353	None	No
5	-80.789672	28.867419	16 Seedling	Yes
6	-80.789734	28.867507	16 Seedling	No
7	-80.788250	28.867671	5 Seedling, 6 Transitional, 5 Adult	No
8	-80.789857	28.867708	None	Yes
9	-80.789968	28.867817	6 Seedling, 7 Transitional, 3 Adult	No
10	-80.790020	28.867896	16 Transitional	Yes
11	-80.790115	28.867967	16 Adult	No
12	-80.790191	28.868045	16 Transitional	No
13	-80.790281	28.868080	16 Adult	Yes
14	-80.790362	28.868143	16 Adult	No
15	-80.790448	28.868188	16 Seedling	No
16	-80.790496	28.868212	16 Transitional	No
17	-80.790614	28.868281	None	Yes
18	-80.790695	28.868357	16 Seedling	No
19	-80.790705	28.868382	None	No
20	-80.790802	28.868521	16 Transitional	Yes
21	-80.790867	28.868565	16 Seedling	Yes
22	-80.790934	28.868622	None	No
23	-80.791048	28.868688	16 Seedling	No
24	-80.791136	28.868805	6 Seedling, 3 Transitional, 7 Adult	Yes
25	-80.791150	28.868895	6 Seedling, 5 Transitional, 5 Adult	Yes
26	-80.971151	28.868982	16 Seedling	Yes
27	-80.791260	28.869223	None	Yes
28	-80.791269	28.869330	6 Seedling, 6 Transitional, 4 Adult	Yes
29	-80.791349	28.869392	16 Seedling	Yes
30	-80.791324	28.869494	16 Adult	No
31	-80.791334	28.869611	6 Seedling, 3 Transitional, 7 Adult	No
32	-80.791335	28.869715	6 Seedling, 6 Transitional, 4 Adult	Yes
33	-80.791354	28.869789	16 Adult	Yes
34	-80.791363	28.869871	None	Yes
35	-80.791393	28.869991	None	No
36	-80.791416	28.870048	16 Adult	Yes
37	-80.791421	28.870058	16 Transitional	No
38	-80.791429	28.870330	16 Seedling	No
39	-80.791576	28.870384	16 Adult	No
40	-80.791649	28.870485	16 Transitional	Yes
41	-80.791694	28.870525	7 Seedling, 4 Transitional, 5 Adult	No
42	-80.791774	28.870621	16 Transitional	No
43	-80.791825	28.870682	16 Adult	Yes
44	-80.791891	28.870823	None	No
45	-80.792007	28.870969	16 Transitional	Yes
46	-80.792095	28.871066	None	Yes
47	-80.792754	28.871885	16 Adult	No
48	-80.792962	28.872139	6 Seedling, 4 Transitional, 6 Adult	No
49	-80.793023	28.872177	16 Seedling	Yes
50	-80.793203	28.872368	16 Transitional	No

Mangrove Survival

A numbered Haglöf log tag (length: 43.0 mm, width: 27.0 mm, weight: 3.9 g), which did not cause bending or damage to any of the mangroves in pilot trials, was attached to each *R. mangle* with flagging tape halfway up the stem for identification. Flagging tape was not tightened and did not visibly restrict growth. All tags and tape were removed at the end of monitoring.

Survival was monitored monthly from 28 June 2019 through 28 June 2020, plus on 19 September 2019, 1 week after Hurricane Dorian (Category 2, wind speeds 56 – 96 kmh⁻¹), to isolate any impacts of the storm (Cappucci, 2019). The eye of the storm was approximately 160 km east of the living shoreline (Butler, 2019). Categories included: 1) “alive” if the mangrove remained in place and had foliage, 2) “standing dead” if the mangrove remained in place with no foliage, 3) “dead” if the stem was bent or partially snapped at the base to the point that the entire mangrove was lying flush on the sediment and had no foliage, and 4) “missing” if the mangrove was no longer in the planted location. A category was not created for mangroves that were bent or partially snapped to the point where the entire mangrove was lying flush on the sediment but still had foliage due to lack of occurrence. “Missing” included loss from uprooting or stem breakage. Stem breakage encompassed individuals in which the root and a small stub of the mangrove stem remained in the sediment. Mangroves that were “missing” due to stem breakage, “dead”, or “standing dead” were monitored throughout the year to account for the possibility of regrowth and new leaf production (Anderson and Lee, 1995; Feller, 1995; Imbert et al., 2000; Duke, 2002).

Environmental factors that potentially varied along the shoreline at the start of the trial and could have impacted survival results included slope, distance to other established vegetation,

fetch, and direction of the shoreline. The slope was calculated 1 week after the living shoreline was implemented (Cannon et al., 2020). A level and laser were used to find the change in elevation between the shoreline 1 m seaward of the shell bags and the shoreline 3 m landward of the planted mangroves. The distance to adjacent shoreline vegetation to the left (northward) and right (southward) of the planted mangroves was determined using a transect tape, with the maximum distance being the start of another replicate, not including controls. Shoreline vegetation included naturally-recruited, mature *R. mangle*, *A. germinans*, and *Conocarpus erectus* (buttonwood). ArcMap 10.6 software was used to find the fetch value for each treatment replicate from the S, SW, W, and NW directions using aerial imagery from 2017. Other directions were excluded because they all had a fetch of 0. The experimental shoreline was curved in nature (Fig. 1), so shoreline direction (in degrees) was determined for each treatment replicate by pointing a compass towards the water, perpendicular to the water line. This value accounted for possible variation in mangrove protection from wave energy that could arise from the shoreline orientation.

Factors that could vary along the shoreline throughout the monitoring period included sediment transport and shading. To measure local erosion or accretion for each treatment replicate, a polyvinyl chloride (PVC) pipe (length: 0.6 m, diameter: 12.7 mm) was placed in the center of the intertidal zone where plants were deployed or the comparable area for control treatments (Rick et al., 2006). Each piece of PVC pipe was secured until 50% of the PVC pipe was belowground and secure. The height of each PVC pipe above the sediment was measured in mm with a meter stick at the beginning of the experiment and every 3 months thereafter for 12 months. None of the planted *R. mangle* were placed directly beneath a canopy created by other plants at the start of the trial. To account for any change over the 12-month period, shading from

all plants was recorded as a binary variable (presence or absence) for each deployed *R. mangle* at the end of each month. Shading was visually classified as “present” if the *R. mangle* had another deployed *R. mangle* or naturally-recruited plants growing directly above it.

Mangrove Growth

To track individual mangrove growth, initial measurements of each plant were recorded 1-week post-deployment on 28 June 2019 and every 3 months afterwards for 1 year. To ensure that mangrove growth results were not confounded by survival, only mangroves classified as “alive” at 12 months were included in the analyses of mangrove growth (N=316). Measurements included height, diameter, and number of branches, leaves, anchored prop roots, free-hanging prop roots, flowers, flower buds, and propagules. Without manipulation of the mangrove, height was recorded to the nearest cm from the base of the stem to the highest point with a meter stick. To account for any impact that change in sediment level had on height measurements at the end of the monitoring period, the value of accretion or erosion based on the month 12 erosion stake measurement was added or subtracted from the month 12 mangrove heights. Diameter was measured with calipers to the nearest mm at the thickest portion of the stem. Branches were classified as an extension at least 2 cm long with a minimum of 1 attached leaf. After the initial 1-week measurement, leaf counts were not recorded above 100 leaves due to reduced accuracy. Free-hanging prop roots were defined as secondary roots originating from the stem and at least 2 cm long, but not touching the sediment. Anchored prop roots originated from the stem and contacted the surface of the sediment. Prop roots were not included if they were shriveled and black in color. Prop roots were not tagged for the experiment; therefore, when analyzing change in number of prop roots from month 0 to month 12, free-hanging and anchored prop roots were

combined. This method was used to account for any free-hanging prop roots that grew into anchored prop roots over the 12-month period.

Temporal Environmental Factors

Temporal factors that could collectively impact the deployed mangroves included water level, wind speed, precipitation, and minimum temperature. To determine the mean water level experienced by the plants, 5 PVC pipes (length: 0.6 m, diameter: 12.7 mm), with colored zip-ties attached 2 cm apart, were secured into the sediment at the same elevation as the planted *R. mangle*. The spacing between zip-ties was re-calibrated each month. These PVC pipes were placed along the restoration site at (28.867443, -80.789642), (28.868232, -80.790535), (28.868587, -80.790872), (28.869684, -80.7913247), and (28.870932, -80.791996). This even-spacing of camera locations captured any variation that was present along the entire restoration site. One Bushnell Trophy Cam HD wildlife camera was faced towards each PVC pipe and programmed to capture a 10-second video every 30 minutes from 7:30 a.m. to 7:30 p.m., 5 days each month from July 2019 to June 2020. Water level was quantified by identifying where the water's surface was in relation to the color-coded zip-ties. If waves were present, the mean wave crest value was recorded as the water level. This timing sequence allowed the mean water height recorded for each month and to account for daily semidiurnal tides. To compare water level from the monitoring period to historical water level, data for Haulover Canal from 2008 to 2020 was retrieved online from the United States Geological Survey (USGS). Salinity was measured with an optical refractometer twice each month during monitoring. Minimum temperature, wind speed, and precipitation data was accessed from weatherunderground.com (station: 29.03, -80.93).

Wrack and Mangrove Propagules

Wrack and propagule quantification took place in September 2019, December 2019, March 2020, and June 2020. To determine wrack cover and wrack thickness in a nondestructive manner, 5 quadrats (0.25 m²) were haphazardly placed within each replicate of each treatment at the elevation of the planted *R. mangle*, landward of the breakwater, if present. Wrack cover was identified based on percent cover calculated from the point-intercept method (Breithaupt et al., 2019). Mean wrack thickness (mm) was calculated by taking the mean of 5 haphazard points within the quadrat (Walters et al., 2021).

Within the same 5 quadrats placed for wrack quantification, the abundance and diversity of mangrove propagules were also recorded. To further explore propagule abundance and location in the area, a transect tape was extended from 3 m seaward of the planting zone to the terrestrial ecotone within each treatment replicate. In order to capture the same location each monitoring period, the transect tape was placed to the right of the erosion stake at a 90° angle to the water line. The bottom left corner of a 0.25 m² quadrat was placed at each meter of the transect tape starting at 0. In addition to propagule species and count, the percent cover of substrate (shell, sand, wrack, woody debris, vegetation) within each quadrat was calculated using the point-intercept method (Jonasson, 1983). Species of vegetation were also recorded.

Table 2: List of measurements taken to answer each study question with the reason why that measurement was taken.

Study Question	Measurement	Subject of Measurement	Scientific Basis for Measurement
1	*Survival (yes/no)	Individual	↑ Survival: ↑ living shoreline success
1	*Growth (change in height, diameter, branches, prop roots, and flowers over 12 months)	Individual	↑ Growth: ↑ living shoreline success
1	Breakwater (yes/no)	Treatment	Breakwater presence: ↑ wave energy attenuation
1	Mangrove treatment (adult, transitional, seedling, mixed)	Treatment	↑ Mangrove age: ↑ ability to survive
1	Placement (seaward or landward row)	Individual	Seaward row: ↑ flooding and wave energy
1	Slope	Treatment	↑ Slope: ↑ difference in flooding and wave energy between landward and seaward row
1	Distance to established vegetation	Treatment	↓ Distance: ↑ protection from wave and wind energy
1	Fetch (S, SW, W, and NW)	Treatment	↑ Fetch: ↑ wave energy
1	Direction of shoreline	Treatment	Depending on wind direction, certain shoreline directions provide ↑ wind and wave energy protection
1	Erosion or accretion measurement	Treatment	↑ Erosion: ↓ wave energy required for removal. ↑ Accretion: ↑ smothering
1	Wrack (thickness and cover)	Treatment	↑ Wrack: ↑ smothering. ↑ Wrack: ↑ nutrition
1	Shading (yes/no)	Individual	↑ Shading: ↓ photosynthesis
2	*Survival (yes/no)	Individual	↑ Survival: ↑ living shoreline success
2	Original size metrics (height, diameter, branches, leaves, free-hanging prop roots, and anchored prop roots at month 0)	Individual	↑ Size: ↑ ability to survive

Study Question	Measurement	Subject of Measurement	Scientific Basis for Measurement
3	*# New mangrove mortalities each month	Individual	↑ Mortality: ↓ living shoreline success
3	Water level	Restoration Site	↑ Water level: ↑ flooding stress
3	Wind speed	Restoration Site	↑ Wind speed: ↑ wave and wind energy stress
3	Precipitation	Restoration Site	↑ Precipitation: ↑ flooding stress
3	Minimum temperature	Restoration Site	↓ Temperature: ↑ freeze stress
3	Mangrove mortality type (standing dead, dead, or missing)	Individual	Alive → missing: forceful removal. Alive → standing dead: flooding stress, insect predation, or freeze stress. Dead: stem weakening
4	*Seedling survival	Individual	↑ Survival: ↑ living shoreline success
4	Mangrove treatment (mixed, seedling)	Treatment	Mixed treatments: ↑ seedling protection from wave energy
5	*Wrack (thickness and cover)	Treatment	↑ Wrack: ↑ smothering. ↑ Wrack: ↑ nutrition
5	*Propagule abundance	Treatment	↑ Propagule trapping: ↑ natural mangrove recruitment
5	Mangrove treatment (adult, transitional, seedling, mixed, control)	Treatment	↑ Mangrove age: ↑ complexity
5	Breakwater (yes/no)	Treatment	Breakwater presence: ↑ complexity

Notes: “Subject of Measurement” indicates whether the measurement was taken for each individual mangrove, each treatment replicate, or for the entire restoration site. *indicates the measurement was a dependent variable.

Site Characteristics

Grain size, wave height, and boating pressure at a restoration site can all impact erosion rate of a shoreline; these measurements were therefore reported to provide context for future living shoreline endeavors. To find the mean grain size on this shell midden shoreline, 100 fossil oyster or clam shells were sampled, from the intertidal portion of the shoreline at the elevation where the mangroves were deployed, using the Wolman pebble count method (Wolman, 1954). Shell midden shorelines, such as this one, have 100% cover of recent and historic oyster and clam shell. Calipers were used to measure the B-axis (width) of each shell. This process was repeated 3 times for a total of 300 shells sampled. Mean wave height was established by referencing literature that ran a Simulating WAVes Nearshore model for this site (Kibler et al., 2020). Wind data from 1979-2018 and water level data from 2010-2018 were obtained from the North American Land Data Assimilation System and the United States Geological Survey, respectively, to inform this model (Kibler et al., 2020). To identify boating pressure that would increase the local wave energy, a Bushnell Trophy Cam HD was located at the center of the restoration site, facing the main channel (width: ~ 150 m). Cameras had the capability of capturing boats that passed within 60 meters of the restoration shoreline. The camera took a 10-second video when activated by motion 24 hours a day, for 5 days each month from July 2019 to June 2020.

GIS Analysis

To determine historical erosion rates of the restoration site, aerial imagery from the years 2007 and 2017 was analyzed using ArcMap 10.6 software as described in McClenachan et al. (2020). The vegetation line from the 2 time periods was compared to identify if land mass had

receded or accumulated over the 10-year period. This change along the shoreline was calculated in $m\ y^{-1}$. This analysis served two purposes: 1) to establish if stabilization of the sediment was needed and 2) to ascertain how the elevation at which the 4 adult red mangroves were rooted may have changed since time of recruitment.

Statistical Analyses

A logistic regression was used to determine the impact mangrove treatment, breakwater presence, and mangrove placement had on mangrove survival at 12 months (Table 3, Test # 1). Survival indicated the mangrove was classified as “alive” at the end of the monitoring period. Mangrove placement was a binary covariate used to separate mangroves planted in the row closest to the water (seaward row) from those planted in the row furthest from the water (landward row). A likelihood-ratio test was used to identify the overall impact of developmental stage on survival. To account for any variation along the shoreline that may influence individual mangrove survival, distance to other vegetation (left and right of treatment), fetch (S, SW, W, NW), shoreline direction, wrack thickness, wrack cover, and total erosion or accretion at 12 months were considered as possible predictors of mangrove survival and growth. Shading was removed from consideration since there were only 4 occurrences. Scatter plots of the possible influential covariates were analyzed to create multiple plausible models. Each model was tested for multicollinearity and covariates with a variable inflation factor exceeding 10 were removed (Hair et al., 1995). The best model was identified using weighted Akaike information criterion (AIC), data visualization, and covariate p-values. All statistical analyses were conducted using R, version 3.5.1 (R Core Team, 2018).

A logistic regression was used to identify the impact of temporal environmental factors on overall mangrove survival (Table 3, Test # 2). Each mangrove was marked as a “1” for the month in which they changed from “alive” to “standing dead”, “dead”, or “missing”, and as a “0” if there was no change. This time of death marker was used as the binary response variable. Plots were analyzed comparing mean salinity, wind speed, minimum temperature, precipitation, and water level to time of death to create a plausible model.

A Welch’s t-test was utilized to detect if erosion and accretion patterns along the shoreline were being impacted by breakwater presence (Table 3, Test # 3). The assumption of normality was first documented with a Shapiro-Wilk test and homogeneity of variance with a Levene test.

A Kruskal-Wallis H test and a chi-square test were used to identify if the mangroves and breakwater were impacting wrack thickness and wrack cover, respectively (Table 3, Tests # 4, 5). Wrack thickness and wrack cover measurements were taken on 4 separate occasions throughout the monitoring period. The mean of the 4 measurements was taken separately for wrack thickness and wrack cover, and utilized as the response variables.

Table 3: List of statistical tests.

Test #	Study Question	Test	Dependent Variable	Independent Variables
1	1	Logistic Regression	Mangrove Survival	Mangrove Treatment, Breakwater, Row
2	3	Logistic Regression	Time of Death	Mean Wind Speed, Precipitation, Minimum Temperature, Salinity, Water Level
3	3	Welch's T-Test	Mean Erosion and Accretion at Month 12	Breakwater
4	5	Kruskal-Wallis H Test	Wrack Thickness	Breakwater, Mangrove Treatment
5	5	Chi-square Test	Wrack Cover	Breakwater, Mangrove Treatment

CHAPTER 3: RESULTS

Mangrove Survival

At the end of the 12-month monitoring period, 49.4% of mangroves survived. The best model chosen by AIC, explaining mangrove survival after 12 months, included breakwater, mangrove treatment, and row with an interaction between row and mangrove treatment (Table 4). However, a significant interaction was only present between row and mangrove treatments with mixed developmental stages. Since the placement of developmental stages within mixed treatments was haphazard, by chance, a greater amount of younger mangroves were placed in the seaward row (58.5% of seedlings and transitionals). The significant interaction between row and mangrove treatment was therefore an artifact of experimental design as opposed to mangrove age. Consequently, the second model chosen by AIC was selected as the best model; this model included breakwater, mangrove treatment, and row with an interaction between breakwater and row. (Table 4).

Table 4: AIC results for the statistical models explaining mangrove survival at month 12.

Model	AICc	ΔAICc	Weight
Survival~Mangrove*Row+Breakwater	713.6	0.0	0.8209
Survival~Mangrove+Row*Breakwater	717.9	4.3	0.0933
Survival~Mangrove+Row+Breakwater	718.4	4.8	0.0741
Survival~Mangrove*Row*Breakwater	723.1	9.5	0.0071
Survival~Mangrove*Breakwater+Row	724.0	10.4	0.0046

The overall influence of mangrove treatment was substantial ($p < 0.001$), with an increase in mangrove survival from seedlings to transitionals ($p < 0.001$) and from transitionals to adults ($p < 0.001$) (Fig. 2, Table 5). The total survival for each developmental stage was 28.2% for seedlings, 45.5% for transitionals, and 75.6% for adults. Increased survival was also associated with presence of a breakwater (Fig. 2, Table 5), with total survival being 62.5% for mangroves with a breakwater and 36.2% for those without a breakwater. The increase in survival associated with breakwater presence was greater for mangroves in the seaward row compared to those in the landward row (Fig. 2, Table 5). Mangrove survival increased by 21.7% and 31.3% when a breakwater was present, for the landward row and seaward row, respectively. Survival was significantly increased for mangroves placed in the landward row (Fig. 2, Table 5), with total survival being 61.4% for mangroves in the landward row and 37.3% for those in the seaward row. Observations within mixed treatments were consistent with trends of survival observed for mangroves in a single developmental stage, with increased survival for older mangroves, presence of a breakwater, and mangroves planted in the landward row.

Table 5: Logistic regression output for the best model explaining mangrove survival at month 12.

Covariate	Estimates	Std. Error	Z value	Pr(> z)
Transitional	1.0525	0.2560	4.112	<0.001*
Mixed	0.5863	0.2557	2.293	0.022*
Adult	2.4739	0.2884	8.577	<0.001*
Landward Row	1.5643	0.2779	5.629	<0.001*
Breakwater Present	1.6805	0.2791	6.022	<0.001*
Landward Row: Breakwater	-0.5904	0.3743	-1.577	0.114

Notes: Colon represents an interaction between two covariates.

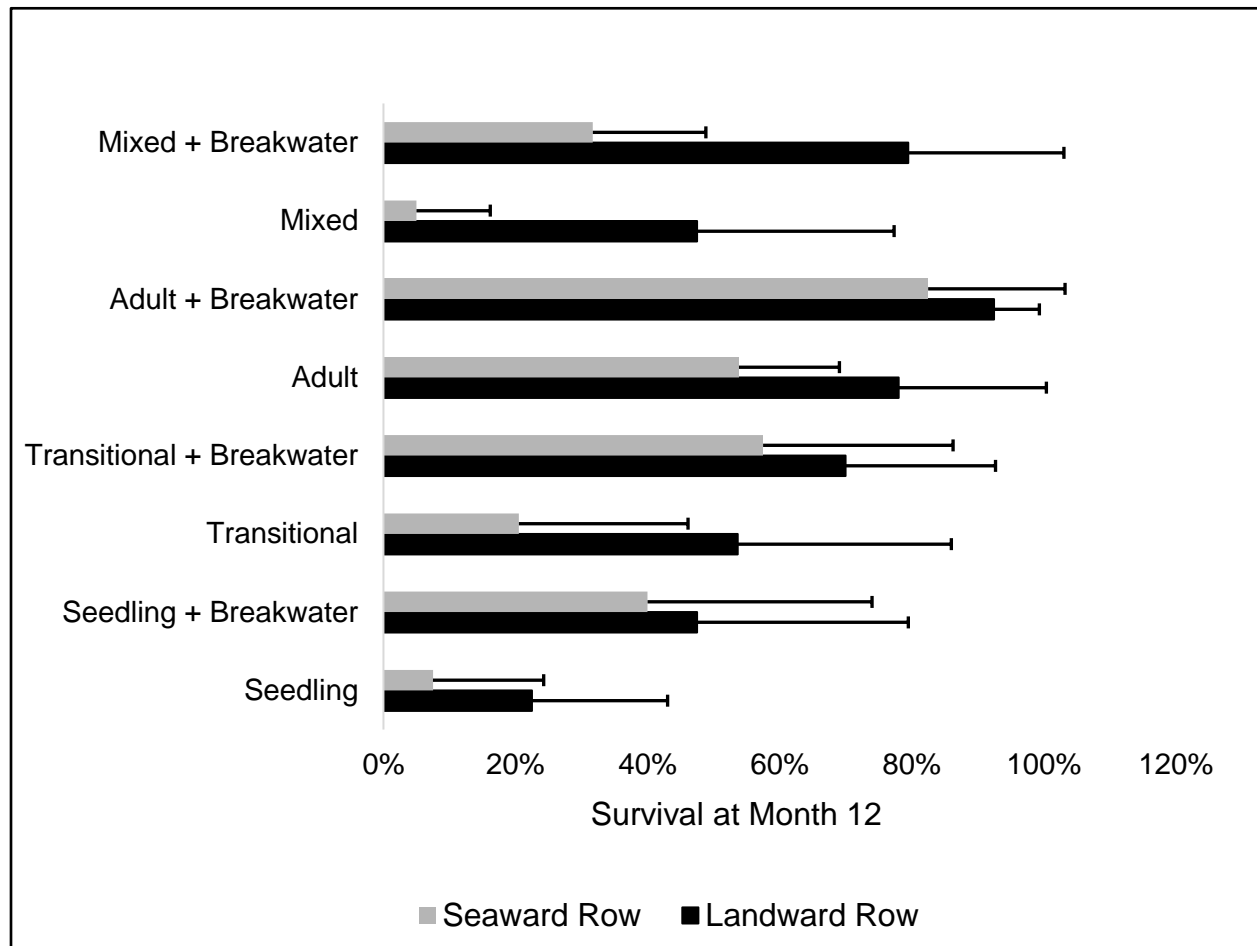


Figure 2: Survival (+ SD) at month 12 based on mangrove, breakwater, and row treatment.

At the end of the 12-month monitoring period, 50.6% of the deployed mangroves did not survive. Of these mangroves, 54.0% were classified as “standing dead”, 42.3% “missing”, and 3.7% “dead”. Greater percentages of missing mangroves were associated with younger developmental stages, absence of a breakwater, and being in the seaward row for mangroves in both mixed and single developmental stage treatments (Figs. 3, 4). Compared to adults, the number of mangroves classified as missing increased by 255.6% for transitionals and 966.7% for seedlings.

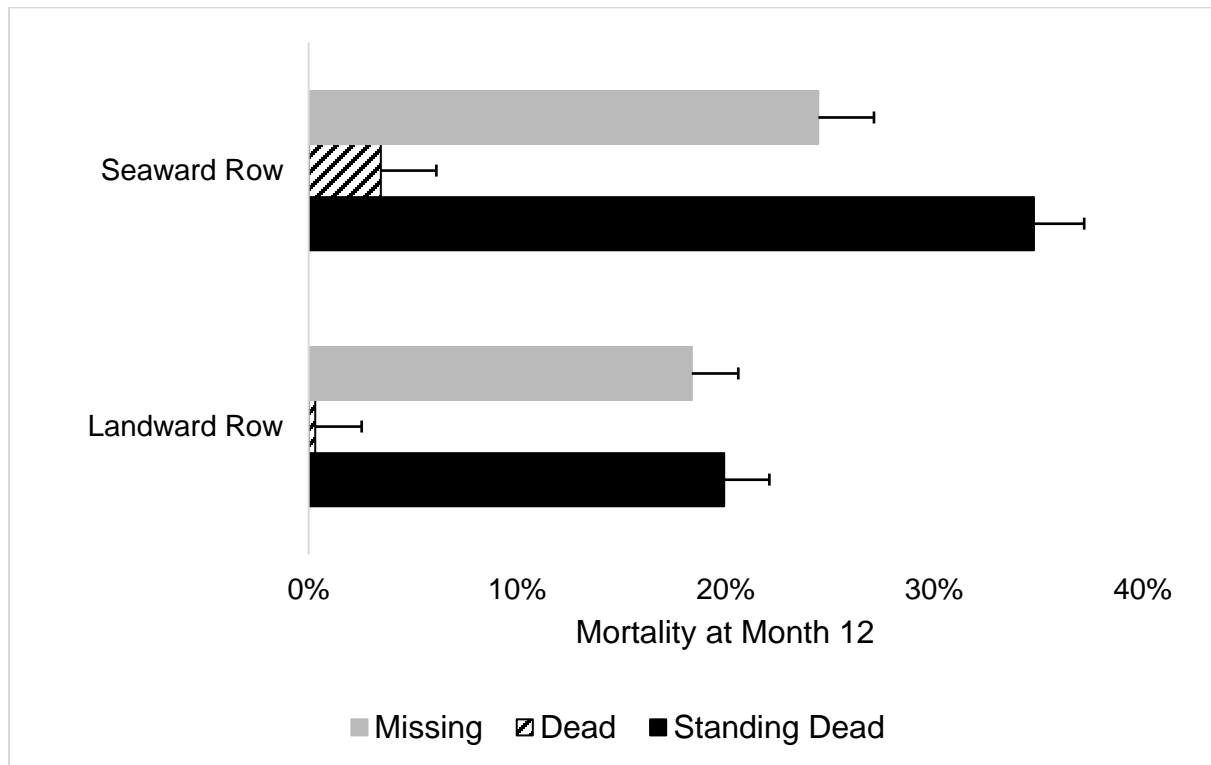


Figure 3: Percentage of standing dead, dead, and missing mangroves (+SE) at month 12 based on mangrove placement within the treatments.

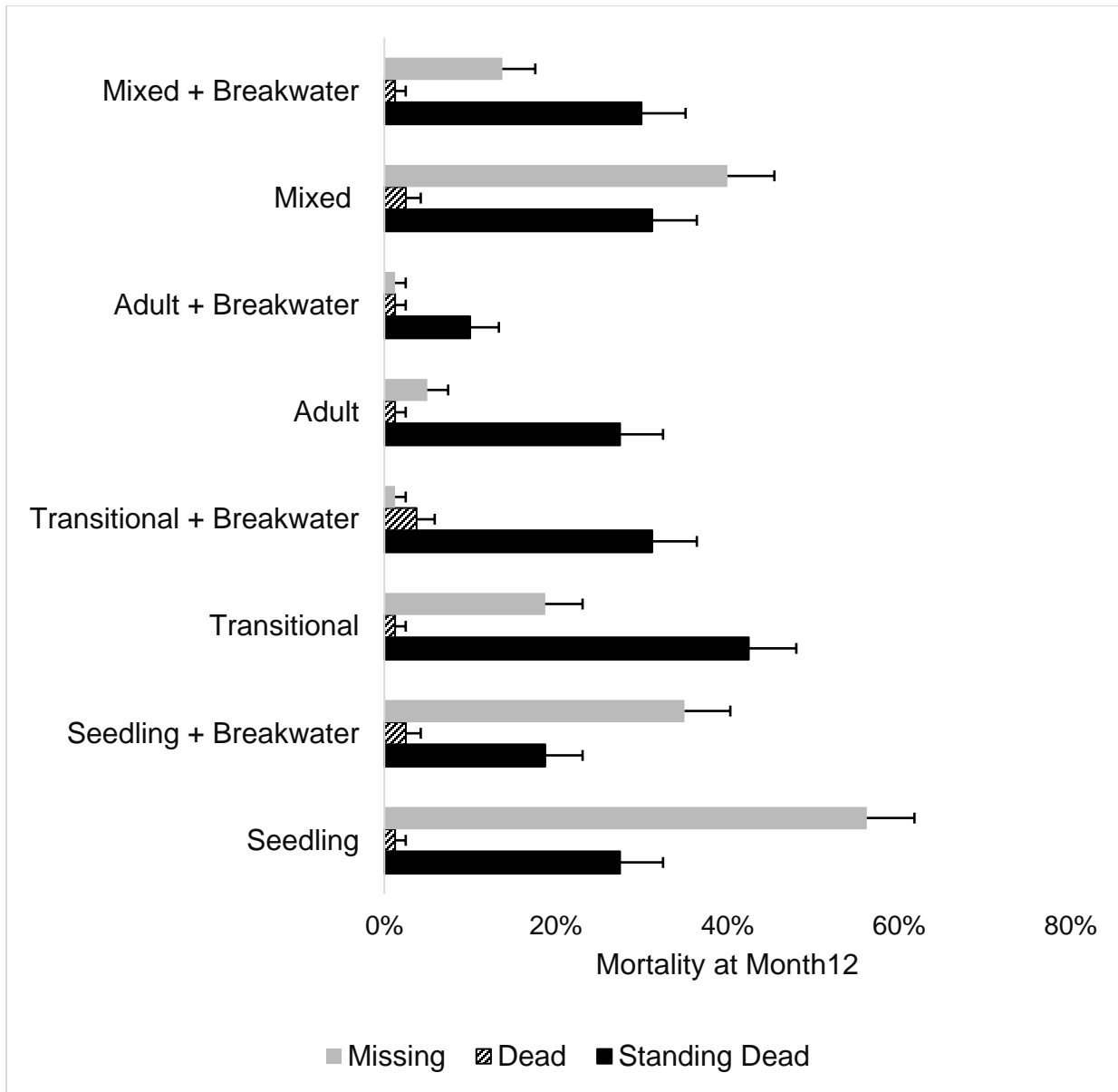


Figure 4: Percentage of standing dead, dead, and missing mangroves (+SE) at month 12 based on mangrove and breakwater treatments.

Mangrove Growth

Seedling vertical growth rate (11.9 cm yr^{-1}) was 22.8% greater than transitional growth rate (9.7 cm yr^{-1}) and 59.2% greater than adult growth rate (7.5 cm yr^{-1}). Presence of a breakwater increased growth for seedlings, adults, and mixed treatments, but decreased growth for transitionals (Fig. 5). Growth rate for adults (5.6 cm yr^{-1}) increased by 67.0% and growth for seedlings (9.9 cm yr^{-1}) increased by 39.9% when a breakwater was present. For the transitional developmental stage, the absence of a breakwater increased vertical growth rate by 82.4% from 6.87 cm yr^{-1} .

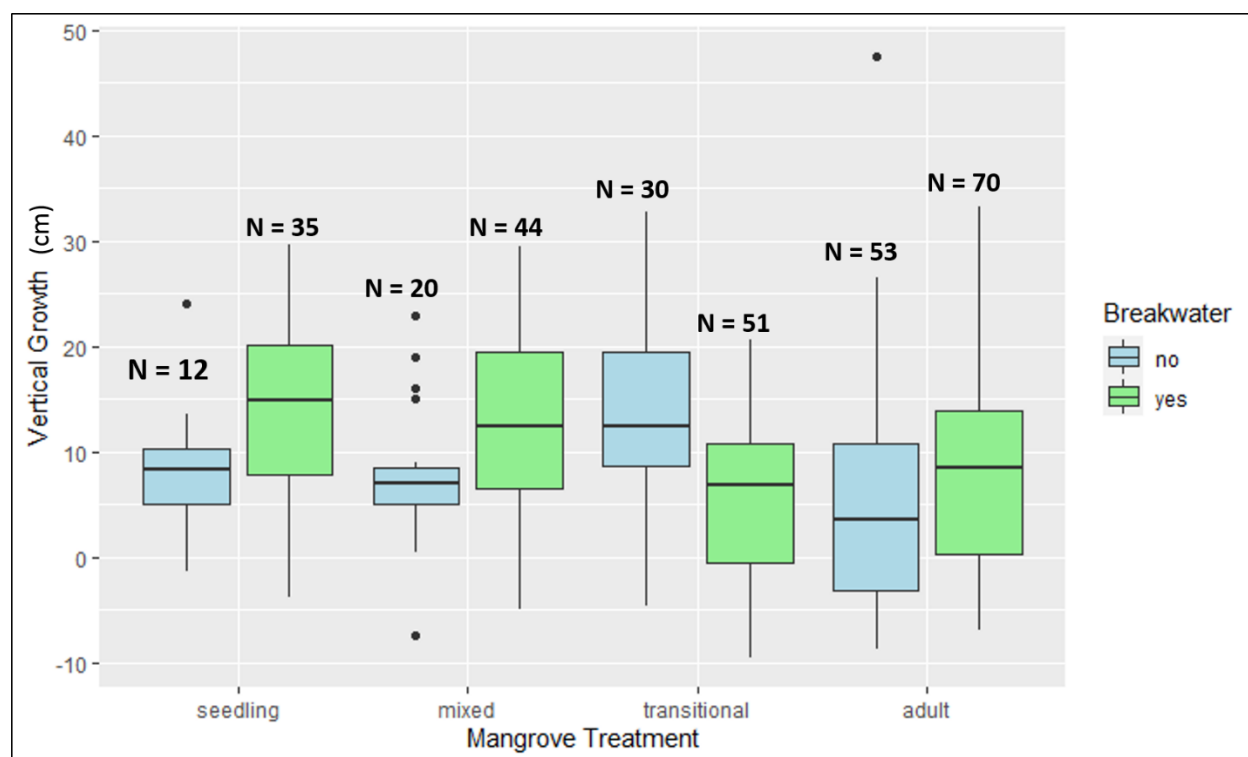


Figure 5: Vertical growth (cm) from month 0 to month 12 based on mangrove and breakwater treatment. "N" represents sample size of each treatment.

Diameter growth over the 12-month period was consistent for all treatments regardless of mangrove age or breakwater presence. Decreased ranges were observed for seedling treatments without a breakwater and mixed treatments without a breakwater due to the decreased sample size (Fig. 6).

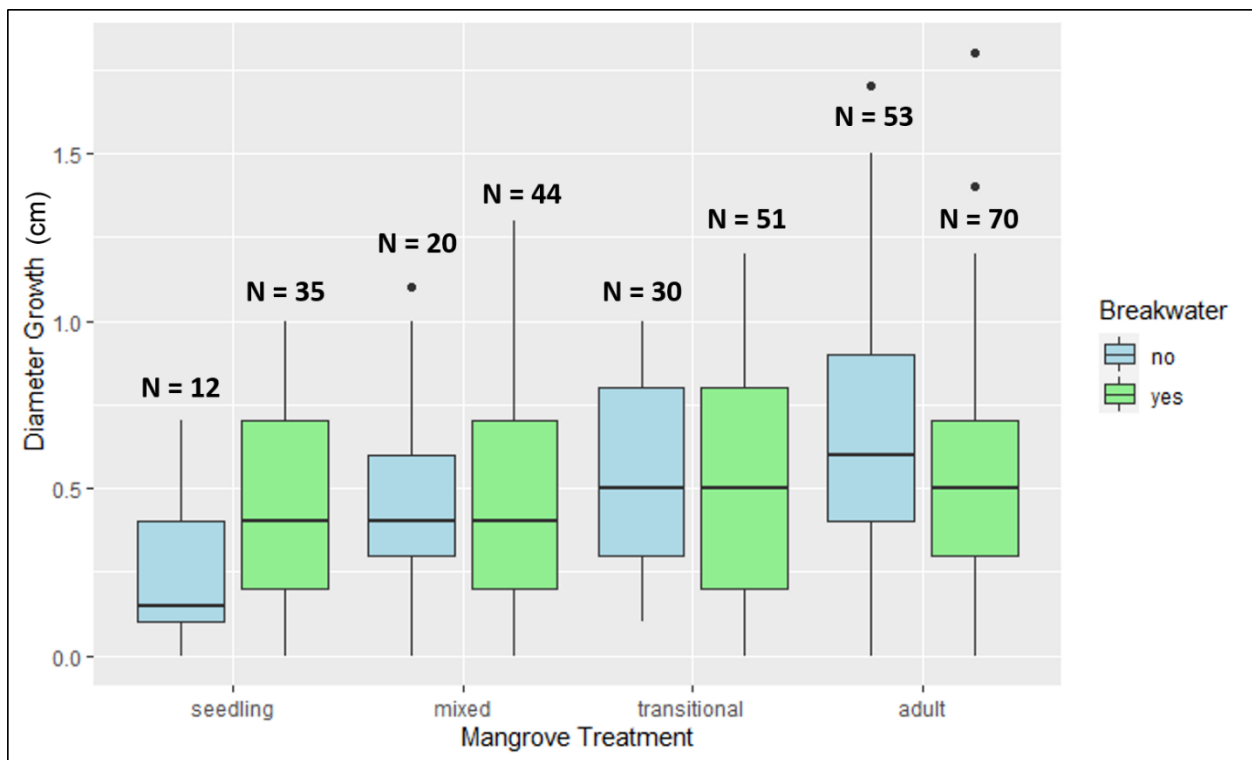


Figure 6: Diameter growth (cm) from month 0 to month 12 based on mangrove and breakwater treatment. “N” represents sample size of each treatment.

Mean change in branch count over the 12-month period was 2.90 branches (44.2%) greater for mangroves with a breakwater compared to those without a breakwater (6.58 branches). Although mean change in branch count was similar among all mangrove treatments, range increased as mangrove age increased (Fig. 7)

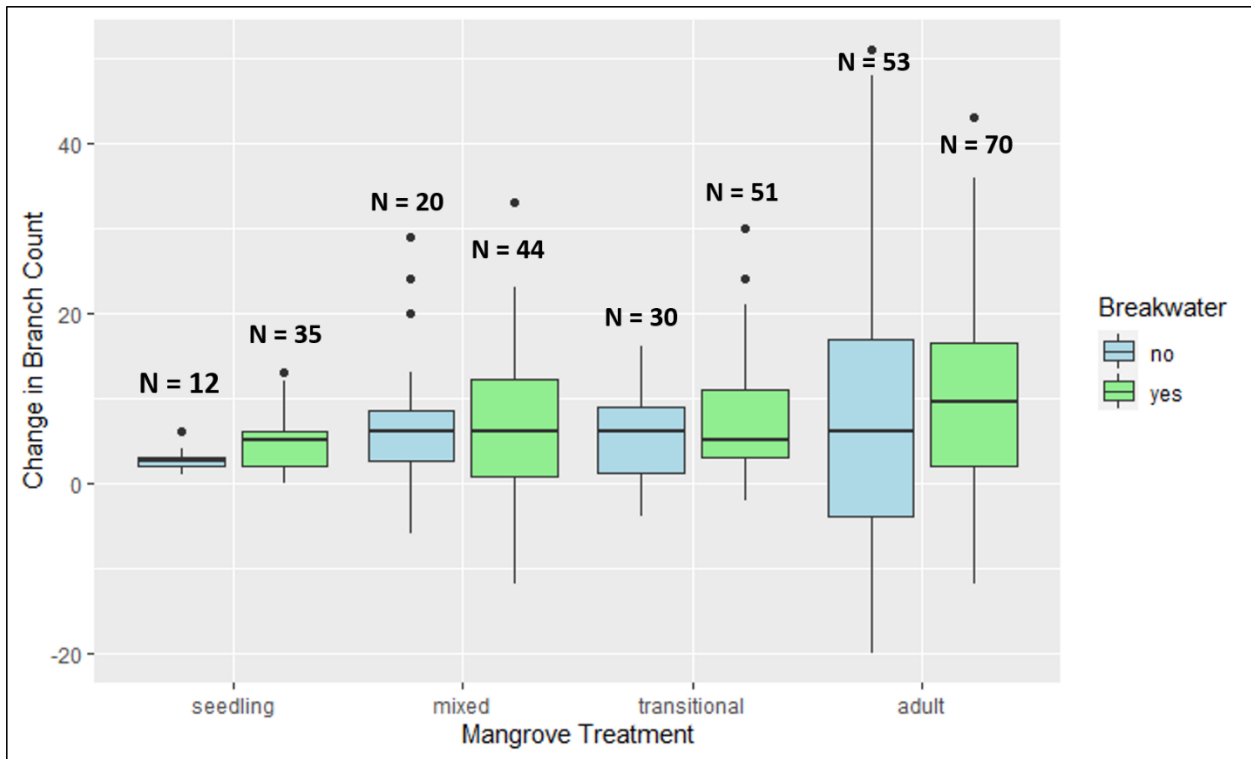


Figure 7: Change in branch count from month 0 to month 12 based on mangrove and breakwater treatment. “N” represents sample size of each treatment.

Of the 316 mangroves that were alive at the end of the experiment, 111 (35.2%) were flowering or had buds in June 2020. None of the mangroves had flowers or buds at month 0, making all counts positive. Increased number of flowers was associated with older developmental stages and breakwater presence (Fig. 8). By month 12, 2 adult mangroves each had a single propagule hanging from their branches. The buds of these 2 propagules were first observed 9 months after the restoration (March 2020).

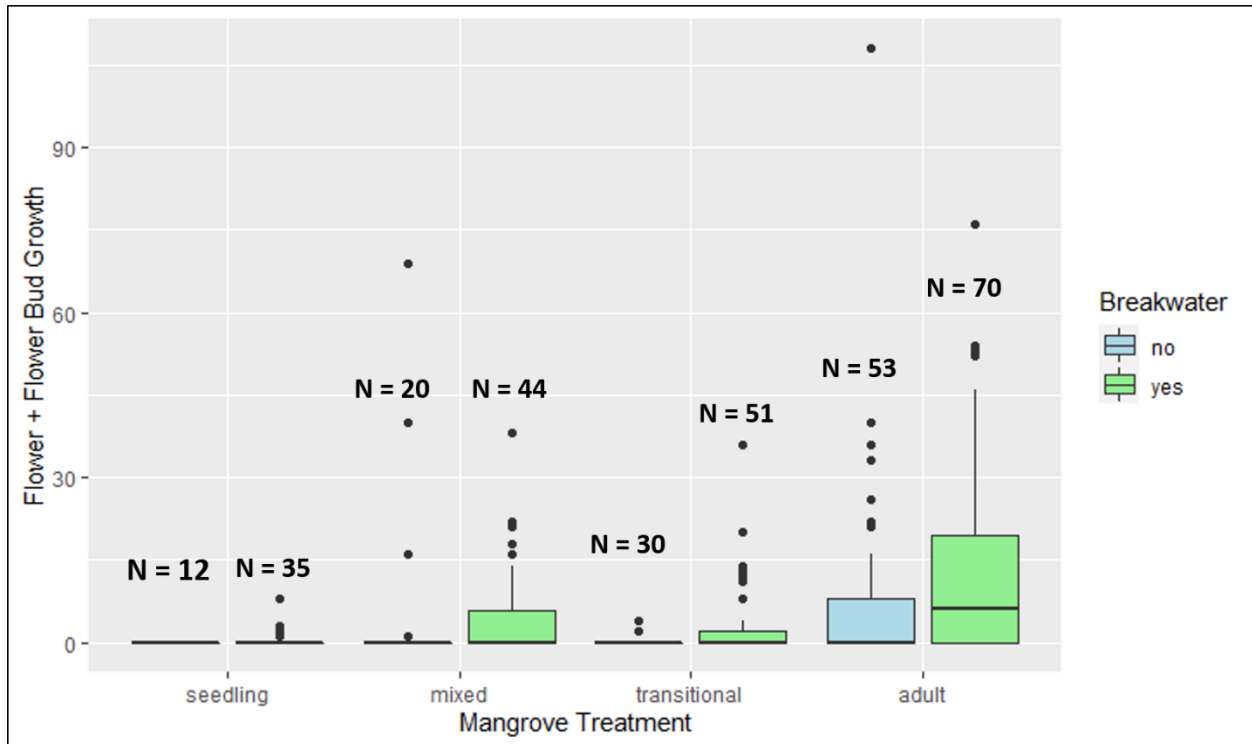


Figure 8: Combined flower and flower bud growth from month 0 to month 12 based on mangrove and breakwater treatment. “N” represents sample size of each treatment.

Neither anchored nor free-hanging prop roots ever developed on the seedling mangroves. Out of the total 253 transitional and adult mangroves that were alive at the end of the monitoring period, 134 (53.0%) produced prop roots after they were deployed. Of these mangroves, 44.8% were free-hanging and 55.2% were anchored. Increased growth was associated with older mangroves (Fig. 9). Compared to the mean prop root growth over the 12-month period for transitionals (0.77 roots), prop root production increased by 185.4% for adult mangroves (2.20 roots). There were 3 instances on 3 different mangroves where a single free-hanging prop root was shriveled and black in color. These mangroves were in the standing dead category.

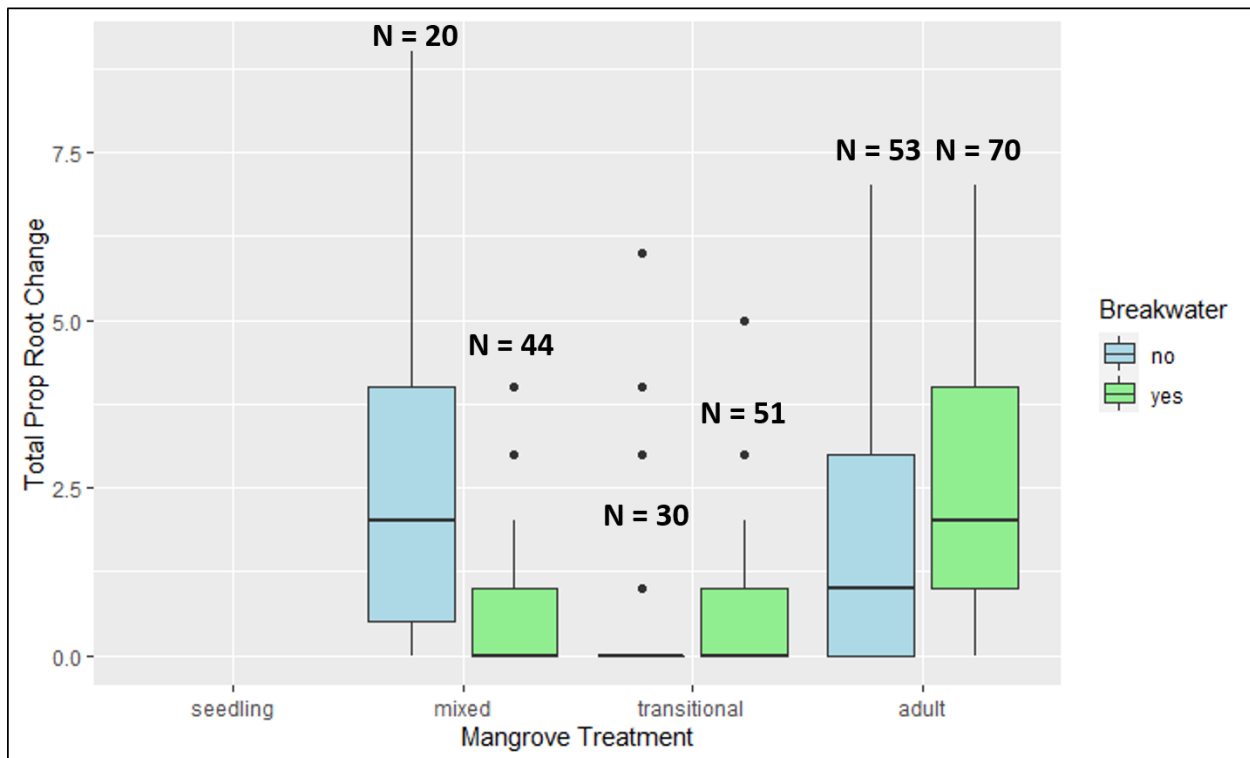


Figure 9: Combined change in anchored and free-hanging prop root count from month 0 to month 12 based on mangrove and breakwater treatment. "N" represents sample size of each treatment.

Differences in branch growth based on row position were consistently observed among mangrove treatments (Fig. 10). For mangroves planted in the landward row, mean branch growth over the 12-month period was 57.1% (2.87 branches) greater than those planted in the seaward row (5.02 branches yr⁻¹). Consistent changes in growth based on row placement were not observed for height, diameter, prop roots, and flowers.

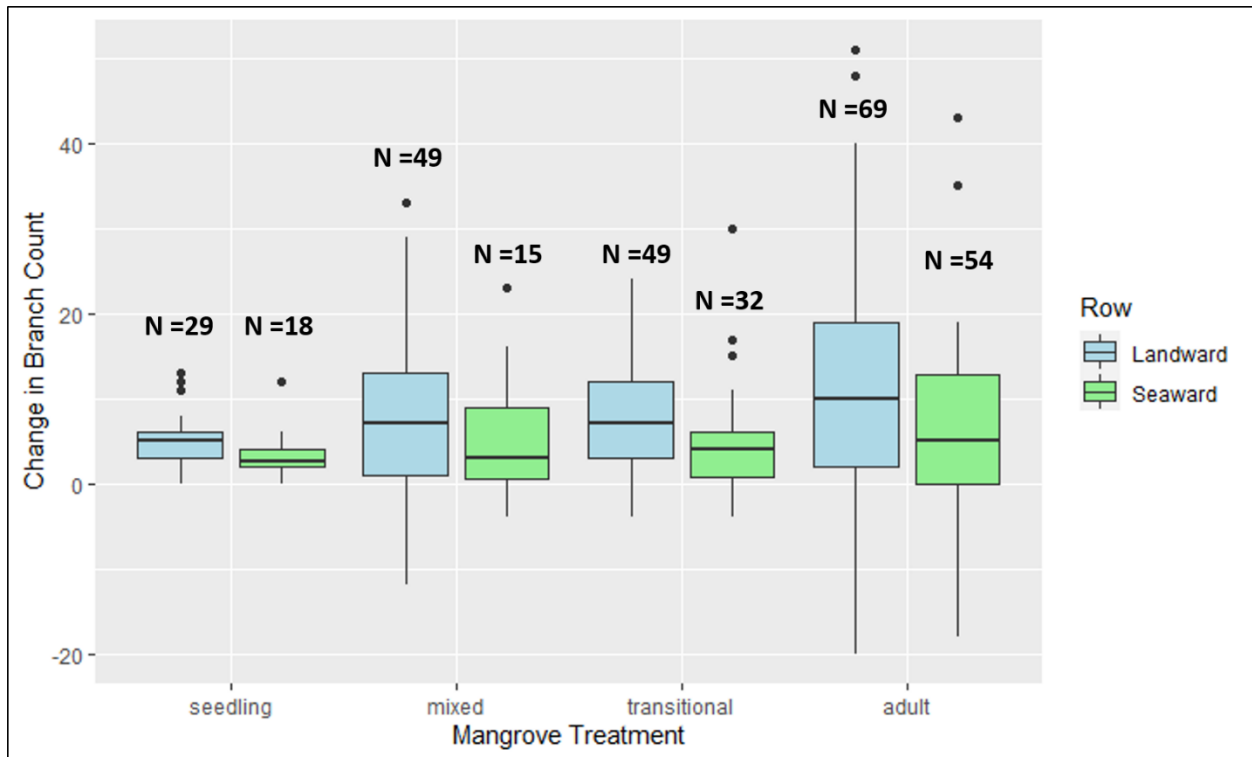


Figure 10: Change in branch growth from month 0 to month 12 based on mangrove and row treatment. “N” represents sample size of each treatment.

Structural Characteristics

Mean starting size dimensions for each developmental stage are displayed in Table 6. All mangroves had leaves at the start of the trial; none had flowers, flower buds, or propagules. According to ANOVA tests, starting size dimensions among each developmental stage were significantly different ($p < 0.05$) from one another with exception of free-hanging prop roots. The category of free-hanging prop roots was therefore removed from consideration as a main driver of increased survival with increased mangrove age.

Table 6: Starting size dimensions for each mangrove developmental stage.

Mean Starting Size Dimensions	Seedling	Transitional	Adult
Height \pm SE (cm)	38.41 \pm 0.41	48.37 \pm 0.51	61.59 \pm 0.65
Diameter \pm SE (cm)	1.24 \pm 0.01	1.60 \pm 0.07	2.20 \pm 0.10
Branch # \pm SE	1.11 \pm 0.04	4.81 \pm 0.12	13.44 \pm 0.51
Leaf # \pm SE	9.20 \pm 0.19	35.40 \pm 0.92	93.35 \pm 3.05
Free-hanging Prop Root # \pm SE	0.00 \pm 0.00	0.00 \pm 0.00	0.18 \pm 0.05
Anchored Prop Root # \pm SE	0.00 \pm 0.00	0.17 \pm 0.05	0.56 \pm 0.13

Height, diameter, and anchored prop roots had the greatest variation between mangroves that survived and those that did not at month 12. Greater starting size measurements for these categories all had a positive impact on survival (Fig. 11). Odds of survival increased by 11.0% for every mm of diameter, 3.5% for every cm of height, and 26.3% for every anchored prop root.

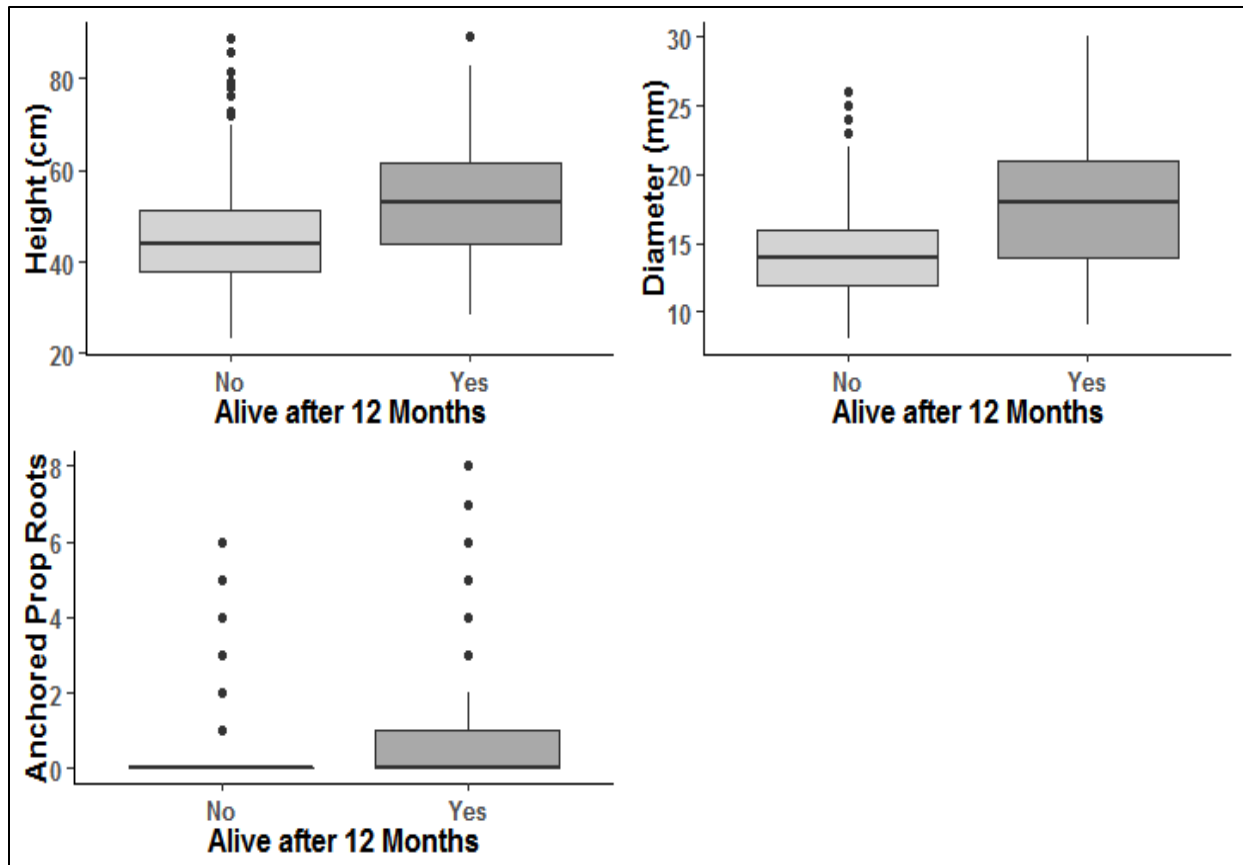


Figure 11: Size metric distribution for mangroves that survived versus mangroves that did not survive over the 12-month monitoring period.

For mangroves without a breakwater, a linear increase in mangrove survival was observed as height and diameter increased (Fig. 12). The same pattern was observed for mangroves with a breakwater until diameter reached 2.0 cm. After this point, height had a smaller overall influence on survival (Fig. 13). The required starting height and diameter for survival was lowered whenever a breakwater was present (Fig. 12, 13).

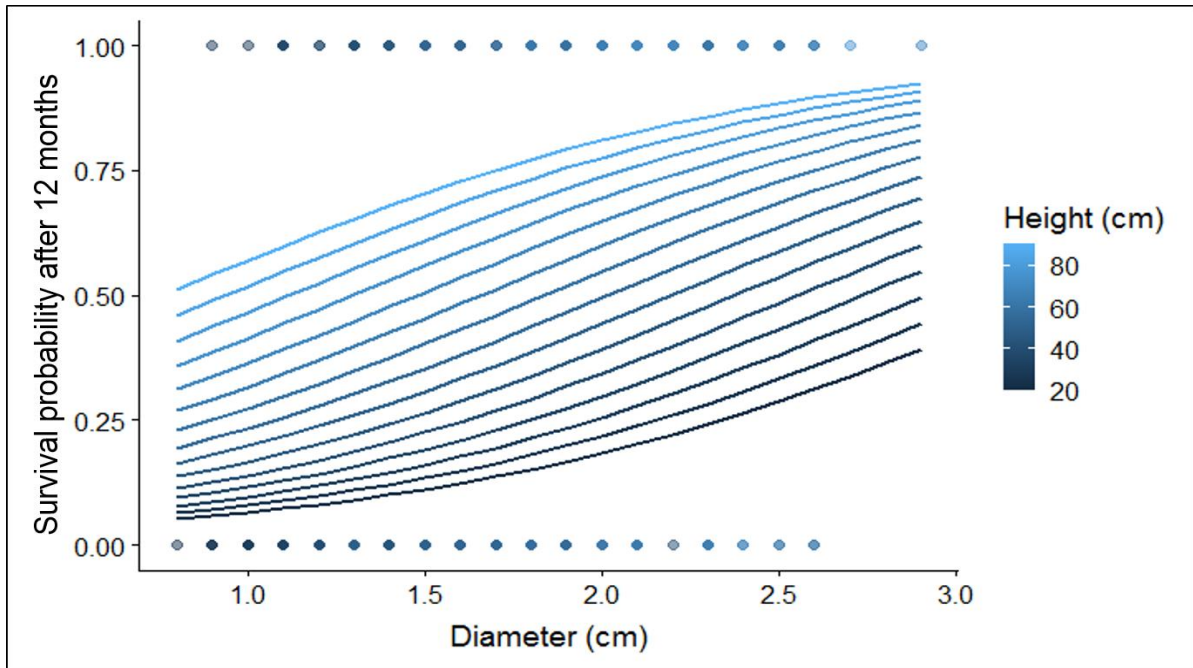


Figure 12: Survival probability after 12 months determined by starting height and diameter for mangroves without a breakwater.

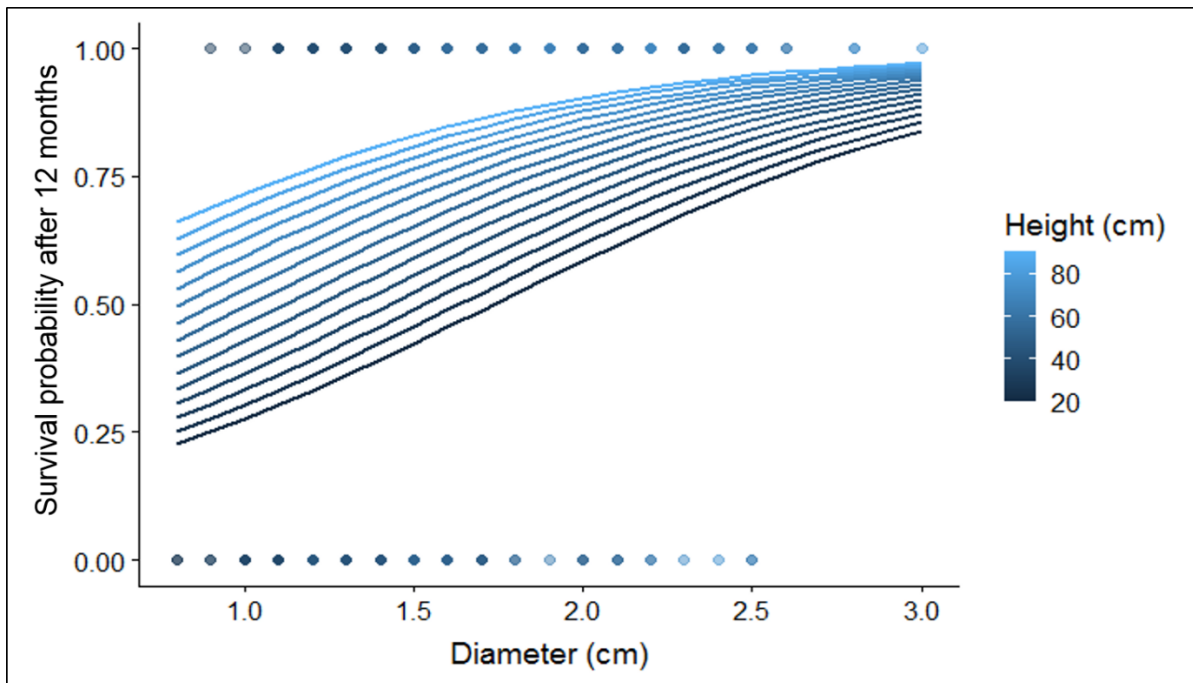


Figure 13: Survival probability after 12 months determined by starting height and diameter for mangroves with a breakwater.

Source of Mangrove Mortalities

Water level ($p < 0.001$) was the best model predicting total mangrove mortality. The majority of the 324 mangrove mortalities occurred 4 months after the restoration, approximately 2 months after the onset of the annual high water season. More specifically, these mortalities occurred in October (62.0%), November (16.0%), September (8.0%), and December (4.6%) when the mean water levels (cm) (\pm SE) above the sediment interface of the mangroves were 20.5 ± 0.2 , 14.2 ± 0.8 , 20.0 ± 0.2 , and 15.4 ± 0.4 , respectively (Fig. 14). Total percent mortality for the remaining months ranged from 0% to 2.2% when mean water level was between 0.0 and 5.6 cm above the sediment (Fig. 14). Post Hurricane Dorian monitoring was conducted on 19 September 2019, at which time only 3 (11.5%) of the 26 mortalities from the month of September occurred. This equates to less than 1.0% of total mortalities.

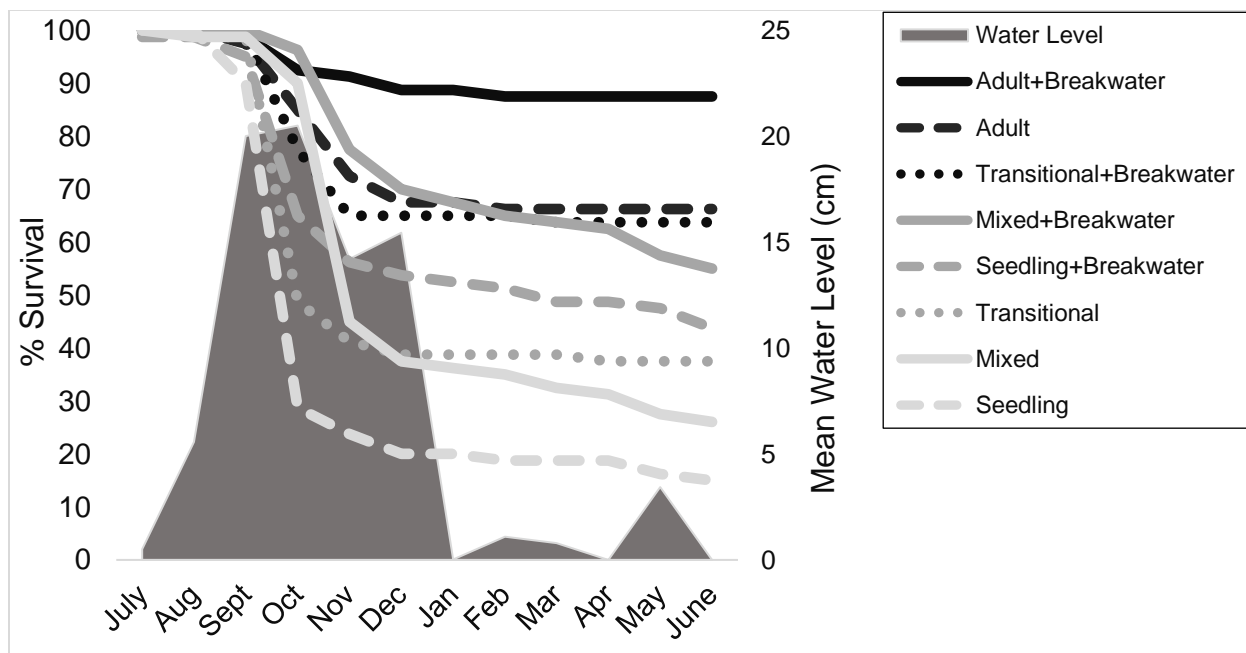


Figure 14: Mean survival (“alive” *R. mangle*) of each treatment per month and mean water level (cm) per month at the base of the mangroves.

Growth measurements taken on 28 September 2019 revealed that at the beginning of the high water season, seedling mangroves were a mean (\pm SE) of 42.7 ± 0.5 cm tall, transitionals 50.9 ± 0.6 cm, and adults 64.6 ± 0.7 cm. During September and October survival monitoring, when water level was ~ 20 cm, 3 seedlings were completely submerged underwater, and 41 mangroves had only the top portion of their highest leaf bundle exposed (21 seedlings, 19 transitionals, 1 adult). Of these 44 mangroves that experienced extreme submersion, 81.8% experienced mortality by October and 88.6% by the end of the 12-month monitoring period. For the 5-day period that water level was monitored each month, video footage revealed that mangroves were completely exposed (water level = 0.0 cm) for part of every month except September and October. For the remainder of months, none of the mangroves experienced complete submersion.

Temperatures were never below freezing (0° C) during the 12-month monitoring period. The lowest temperature reached was 1.6° C for 6 hours in January; during this month, total mortality for the month was 1.5%. Although small changes in precipitation and wind speed were not correlated with overall mangrove mortality, maximum monthly precipitation (8.3 mm) and wind speed (16.3 km h^{-1}) values occurred in October and September, respectively (Tab. 7). These factors most likely contributed to the overall stress of flooding and wave energy. Mean values for water level, minimum temperature, salinity, precipitation, and wind speed are provided in Table 7.

Table 7: Mean values (\pm SE) for water level, minimum temperature, salinity, precipitation, and wind over the monitoring period.

Month	Water Level (cm)	Min. Temp. ($^{\circ}$ C)	Salinity (ppt)	Precipitation (mm)	Wind (km h^{-1})
July 2019	0.5 \pm 0.1	23.2 \pm 0.2	27.0 \pm 0.7	7.1 \pm 2.6	7.7 \pm 0.5
Aug. 2019	5.6 \pm 0.2	23.8 \pm 0.1	24.5 \pm 0.7	5.3 \pm 2.0	10.2 \pm 0.6
Sept. 2019	20.0 \pm 0.2	21.1 \pm 0.4	35.0 \pm 0.4	4.8 \pm 1.2	16.3 \pm 1.5
Oct. 2019	20.5 \pm 0.2	22.2 \pm 0.4	32.5 \pm 0.4	8.4 \pm 3.8	11.8 \pm 1.0
Nov. 2019	14.2 \pm 0.8	13.3 \pm 0.8	32.0 \pm 0.4	1.5 \pm 0.6	10.9 \pm 1.1
Dec. 2019	15.4 \pm 0.4	13.3 \pm 0.8	37.0 \pm 0.0	2.7 \pm 1.3	12.4 \pm 0.4
Jan. 2020	0.02 \pm 0.0	11.6 \pm 1.1	32.5 \pm 0.7	0.2 \pm 0.3	11.1 \pm 0.5
Feb. 2020	1.1 \pm 0.1	12.2 \pm 0.7	31.0 \pm 1.6	2.0 \pm 0.8	12.7 \pm 0.5
Mar. 2020	0.8 \pm 0.1	15.5 \pm 0.4	34.0 \pm 0.7	0.0 \pm 0.0	10.5 \pm 0.4
Apr. 2020	0.0 \pm 0.0	16.6 \pm 0.6	31.0 \pm 0.6	3.0 \pm 1.3	13.1 \pm 0.3
May 2020	3.4 \pm 0.1	18.3 \pm 0.6	34.0 \pm 0.7	2.5 \pm 1.0	12.2 \pm 0.3
June 2020	0.0 \pm 0.0	22.2 \pm 0.2	29.0 \pm 0.9	4.0 \pm 1.3	9.1 \pm 0.3



Figure 15: Water level of 0 cm in January 2020 at the base of the deployed mangroves at low tide.



Figure 16: Water level of ~20 cm in September 2020 at the base of the deployed mangroves at mid-tide.

According to historical water gage data collected from Haulover Canal by USGS, between the years of 2008 and 2020, mean water level within the tidal zone was 23.7 cm. Mean water level for each year ranged from 18.6 cm (2008) to 30.4 (2020). For the experimental monitoring period at the restoration site (July 2019 to June 2020), the mean water level at the Haulover Canal station was 31.5 cm (Fig. 17).

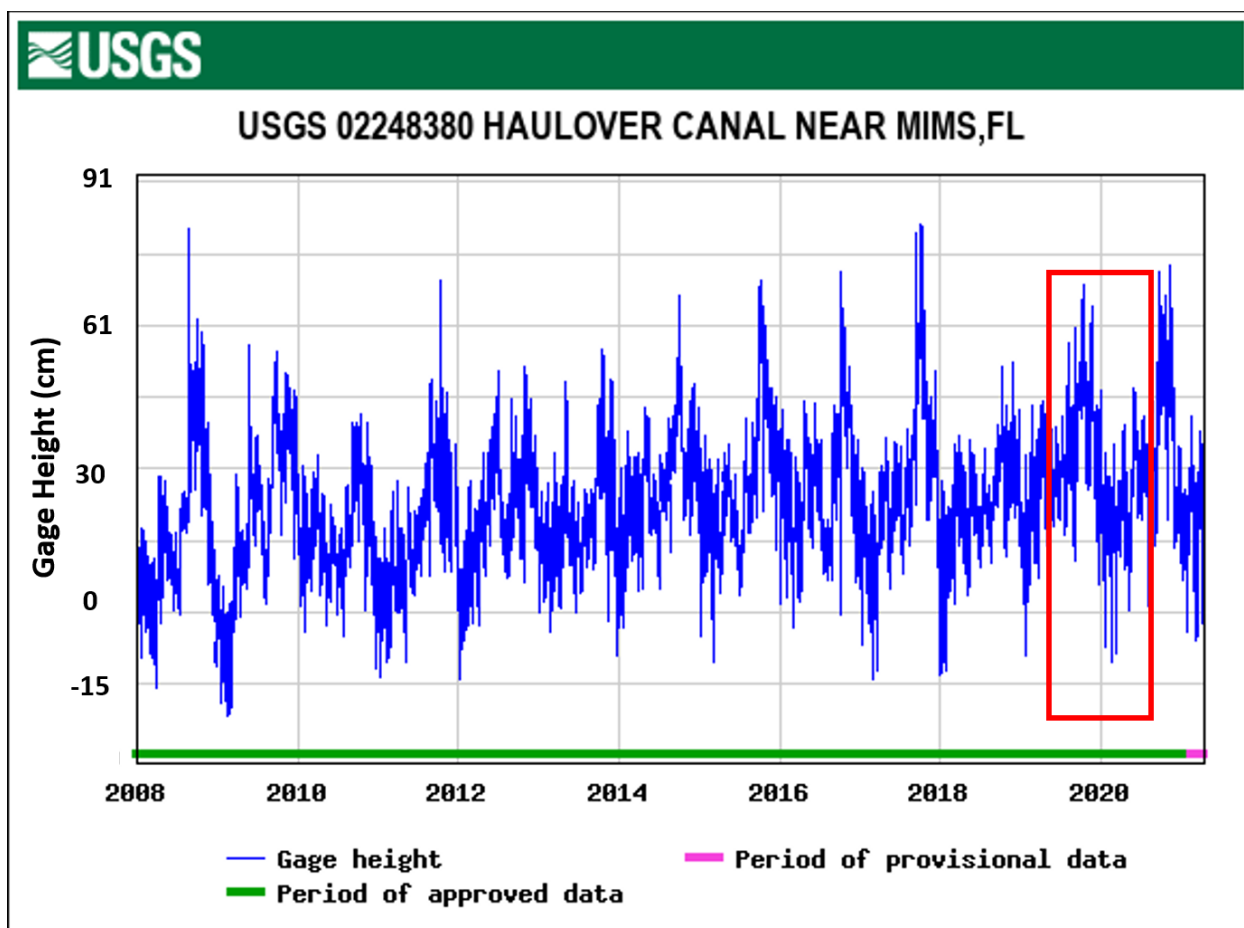


Figure 17: Historical water level data from USGS for Haulover Canal from 2008 to 2020. Red box represents experimental period.

Out of the 137 *R. mangle* that were missing by 1 year post-restoration, 26.3% went straight from “alive” to “missing”, and 73.7% were classified as “standing dead” or “dead” before progressing to “missing.” For mangroves that progressed from standing dead or dead to the missing category, individuals remained in the standing dead or dead state between 1 and 8 months [mean (\pm SD): 4.74 ± 0.21 months]. Twenty-nine of the missing *R. mangle* had the base of the stem still visible, suggesting stem breakage and the retention of roots. Of the 21 displaced mangroves found along the shoreline, 100% did not have a root system attached. Moreover, 2

adults, 1 transitional, and 6 seedling *R. mangle* recovered from being classified as standing dead and were declared alive at the end of the 12 months. Between 1 and 8 months elapsed before new growth was observed [mean (\pm SD): 4.88 ± 0.67 months]. New growth was either in the form of leaves growing from the top of the stem or from a branch with leaves growing from the side of the stem. Fig. 18 illustrates how mortality status (standing dead, dead, or missing), changed from the start of high water season in September 2019 to the end of monitoring in June 2020. The majority of standing dead mortalities occurred during October and November. After this point, the standing dead trend line decreased, indicating progression to the missing stage (Fig. 18). Although the proportion of dead mangroves remained steady over the 12-month period, this was due to the transitory state of progressing from standing dead to missing.

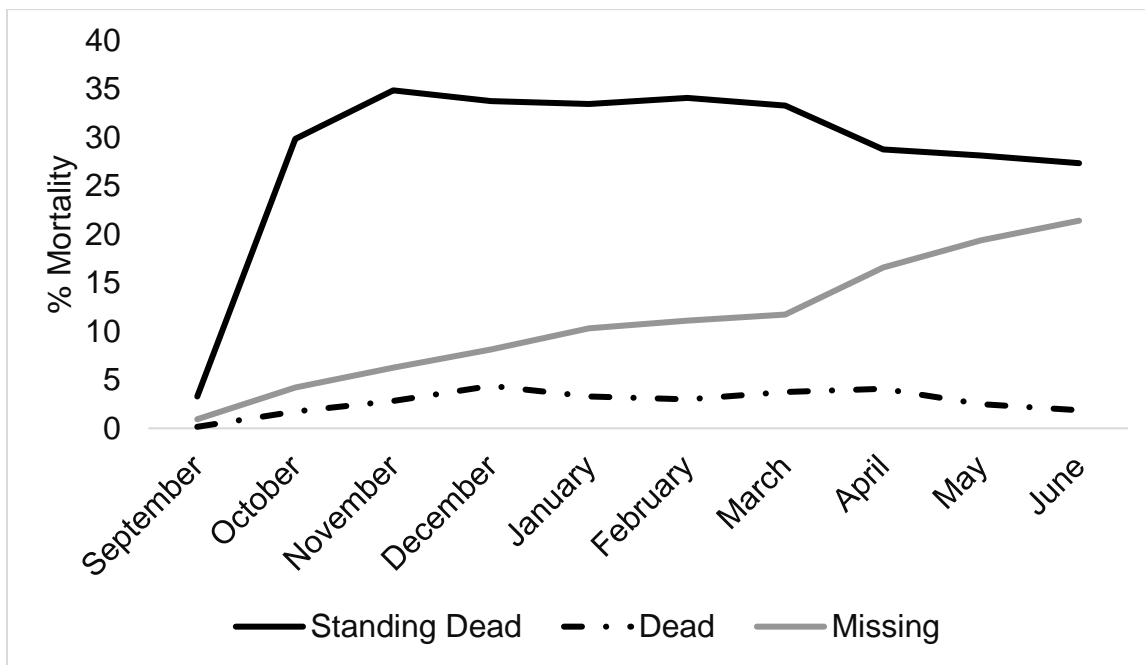


Figure 18: Distribution of mangrove mortality type from the onset of high water season to the end of the monitoring period.

Distribution of erosion and accretion (mm) at month 12 based on breakwater presence can be visualized in Fig. 19. One erosion stake (control with no breakwater) was dislodged from the sediment and removed from the analysis. Results of the Shapiro-Wilks test indicated the change in sediment level data was normally distributed (breakwater: $p=0.74$, no breakwater: $p=0.93$); results of the Levene test showed homogeneity of variance ($p = 0.12$), satisfying the assumptions of the Welch's t-test. Results of the t-test indicate that the mean of the 2 groups was not significantly different ($p=0.18$).

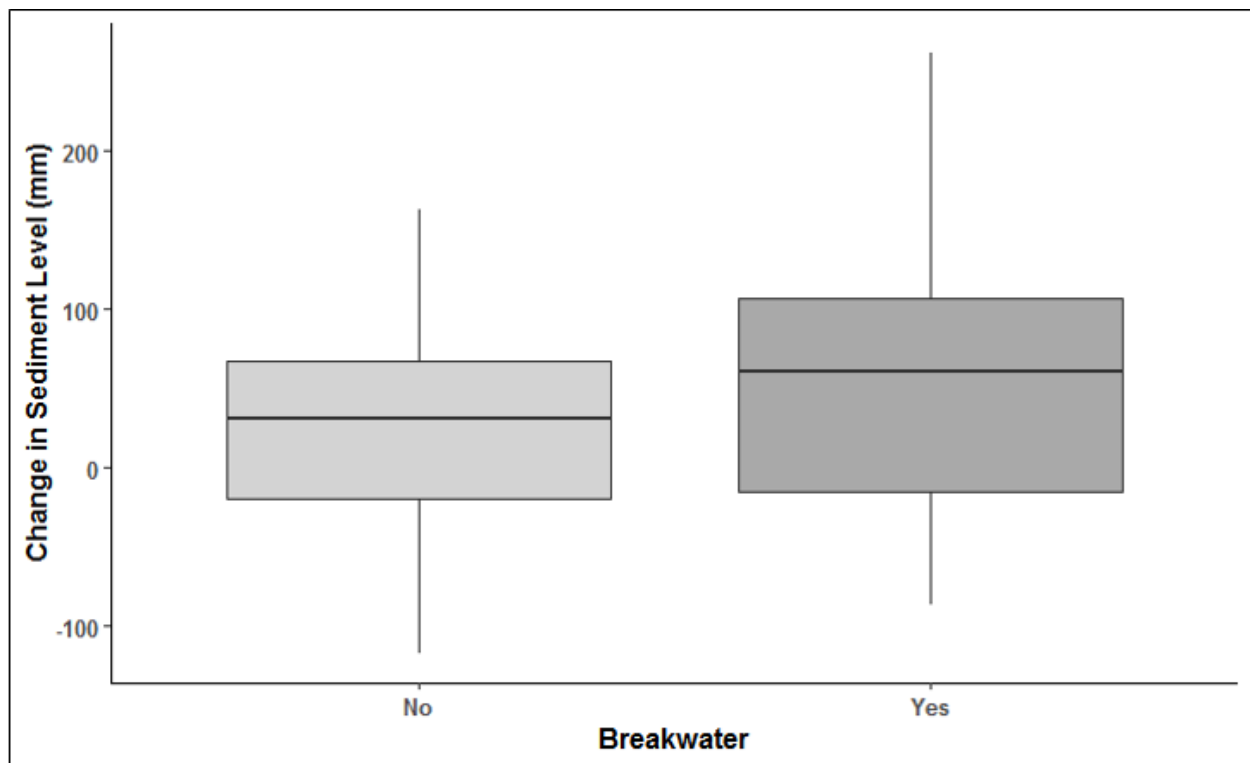


Figure 19: Change in sediment level (erosion and accretion in mm) between treatments with and without a breakwater.

Seedling Survival

For treatments with and without a breakwater, the proportion of seedlings that survived did not vary between seedlings in mixed treatments and seedlings in seeding-only treatments (Fig. 20). Seedling-only treatments had 29.4% survival, and seedlings part of mixed treatments had 25.0% survival. The starting height for all seedling mangroves, regardless of treatments, ranged from 23.0 to 53.0 cm. For seedlings that survived the 12-month period, the mean starting height was 39.9 cm; for seedlings that did not survive, the mean starting height was 37.6 cm.

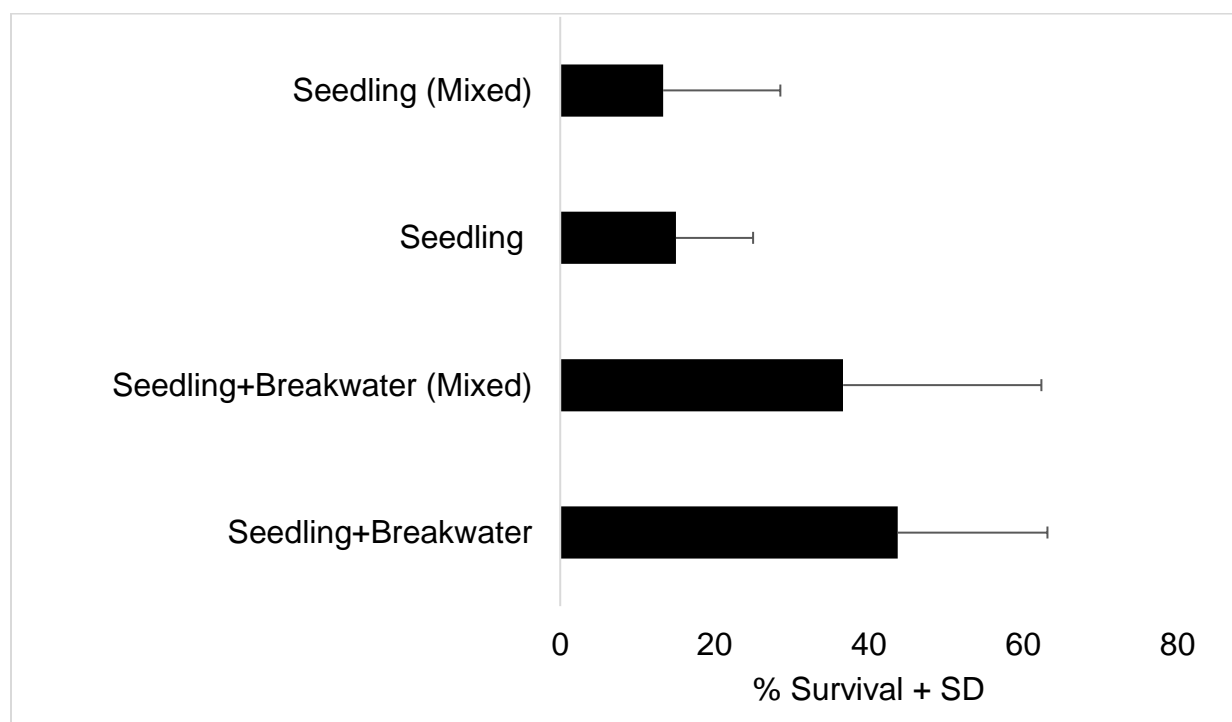


Figure 20: Survival (+SD) of seedlings in mixed vs. single developmental stage groups divided by breakwater presence.

Wrack and Propagule Abundance

There was no significant difference in wrack thickness among mangrove treatments ($p = 0.19$) or between treatments with and without a breakwater ($p = 0.16$) (Fig. 20). Additionally, there was no significant difference in wrack cover among mangrove treatments ($p=0.79$) or between treatments with and without a breakwater ($p=0.41$) (Fig. 21). The majority of wrack observed along the restoration site was *Halodule wrightii* (shoal grass) shoots, but algal and leaf debris from intertidal and terrestrial plants were also present (Brighthaupt et al. 2019). Monitoring conducted at 3 months (5 October 2019) and 6 months (2 January 2020) after deployment of the living shoreline showed very minimal amounts of wrack cover (mean=0.0%). Transects conducted perpendicular to each treatment replicate during the high water season placed the wrack line between 2 and 3 meters landward of where the *R. mangle* were deployed. Outside of the high water season (month 9 and 12), the high water line varied between the breakwater and deployed mangroves, but wrack presence was minimal and did not to form a visible wrack line (Figs. 21, 22).

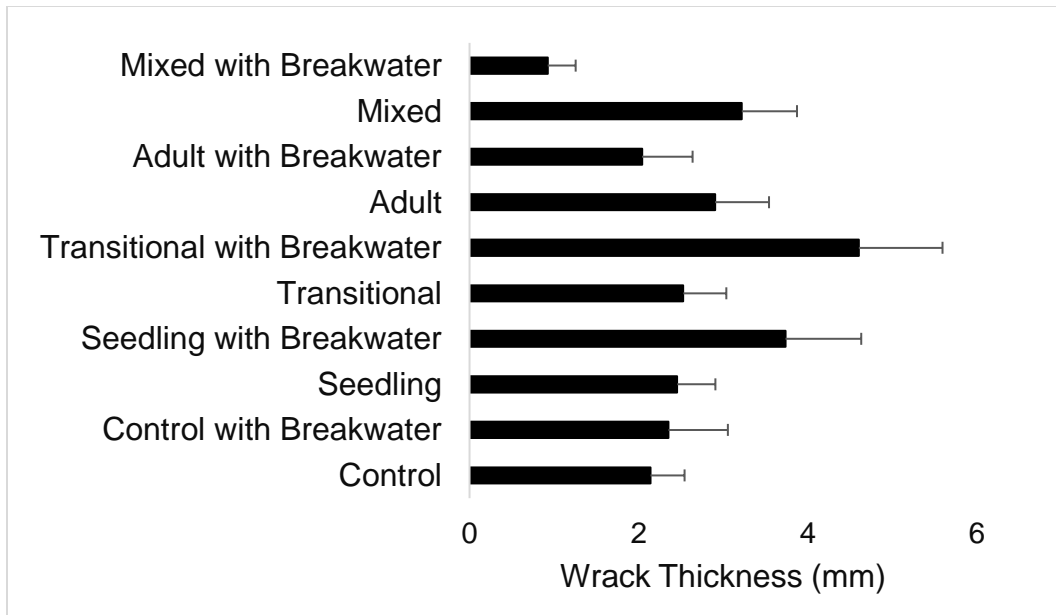


Figure 21: Mean wrack thickness (+SE) at each treatment combined for months 9 and 12.

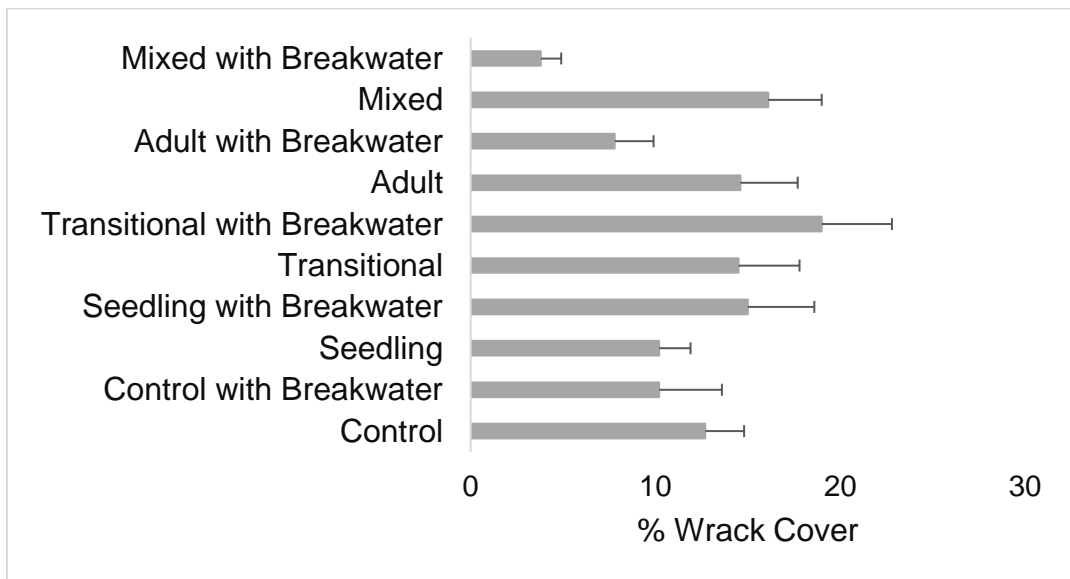


Figure 22: Mean wrack cover (+SE) at each treatment combined for months 9 and 12.

There were no naturally recruited mangrove propagules or seedlings recorded in wrack quadrats throughout the 12-month period. Sixty-two propagules were found in the transects run perpendicular to the shoreline and into the ecotone, averaging to 0.05 ± 0.02 propagules per 0.25m^2 for the year. These propagules were not marked, so it is unknown if any of these propagules were ever recounted or if they recruited to become seedlings. Only 1 *R. mangle* propagule was recorded within a quadrat that also had a deployed mangrove, and 1 *A. germinans* propagule was found caught in the mesh of a breakwater, the latter of which had started to germinate. Four mangrove propagules (2 *R. mangle*, 2 *A. germinans*) were found at the elevation of the deployed mangroves, whereas the remaining propagules were between 1 and 3 meters landward of this area.

A total of 33 naturally-recruited mangrove seedlings were recorded along the transects, placed perpendicular to each treatment replicate over the 12-month monitoring period. Seedlings were defined as recruited propagules. The observed seedlings were recently recruited, visible by their small size and the seed casings still attached to some of the *A. germinans* seedlings. All seedlings were found 1 to 3 meters behind the elevation at which the deployed mangroves were placed. There was only 1 quadrat that had mangrove seedlings in it consistently through months 3 to 12. This quadrat had 16 mangrove seedlings (1 *R. mangle*, 15 *A. germinans*) in month 3 and 3 mangrove seedlings (1 *R. mangle*, 2 *A. germinans*) in months 6, 9, and 12. The substrate of this quadrat included of *A. germinans* pneumatophores, shell, woody debris, and wrack.

Table 8: Mangrove propagule and seedling counts found in the shoreline transects.

Month	<i>R. mangle</i> Propagules	<i>A. germinans</i> Propagules	<i>L. racemosa</i> Propagules	<i>R. mangle</i> Seedlings	<i>A. germinans</i> Seedlings
July 2019	1	0	0	0	0
October 2019	29	17	0	1	15
January 2020	5	9	1	1	6
April 2020	0	0	0	1	5
July 2020	0	0	0	1	3

Site Characteristics

Table 9: Mean and standard error for shell size, slope, wave height, and boating activity at the restoration site.

Measurement	Shell Size (cm)	Slope	Wave Height (cm)	Boating Activity (boats day ⁻¹)
Mean	4.48	0.12	5.37	0.01
SE	0.12	0.00	0.14	0.00

Out of the 12 total boats that were captured by the wildlife cameras in the study, all had motors, but only 2 produced wakes. These 2 boats were far enough away from the shoreline that the waves originating from the boat dissipated offshore. The remaining 10 boats were either actively fishing or trolling with fishing rods visible in the boats.

Of the 300 shells sampled along the restoration site, 67.7% were clam shells and 32.3% were oyster shells. Clam shells ranged from 1.50 to 9.10 cm with a mean of 5.16 cm; oyster shells ranged from 0.70 to 6.90 cm with a mean of 3.07 cm.

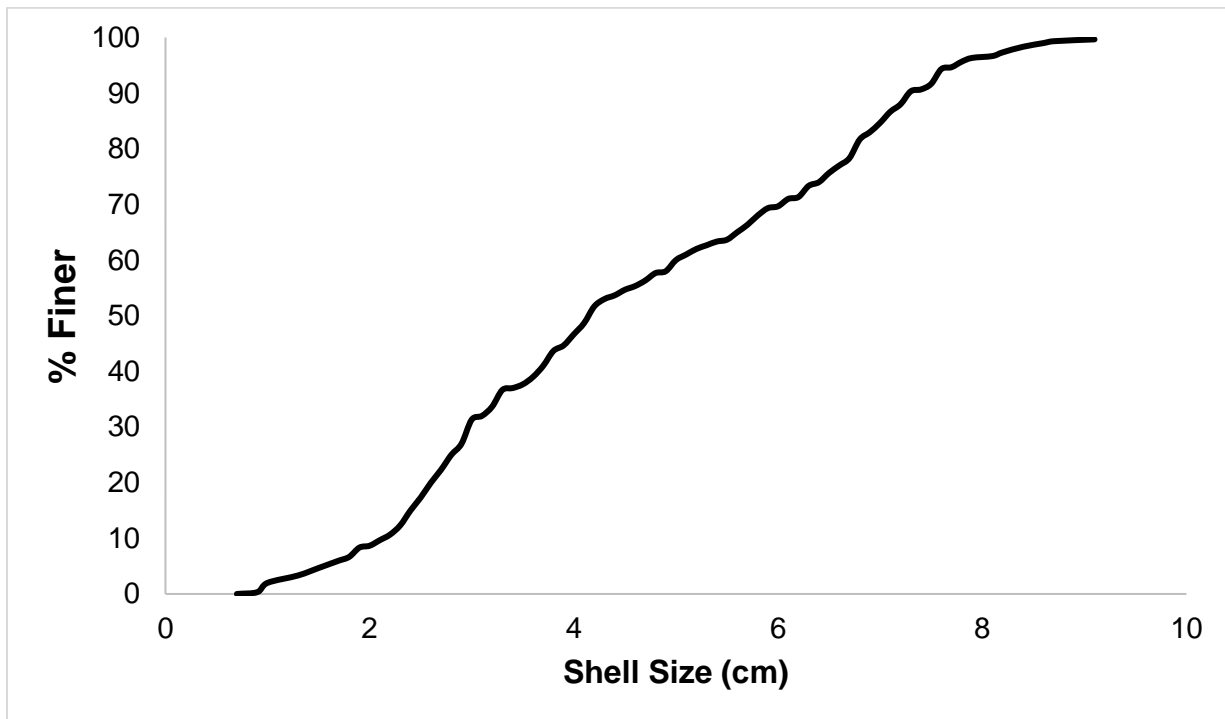


Figure 23: Shell size distribution curve results from the Wolman pebble count conducted along the restoration site.

GIS Analysis

Between 2007 and 2017, the mean loss of shoreline per year was -0.38 m ranging from $+1.18 \text{ m y}^{-1}$ to -1.38 m y^{-1} , using the methodology developed by McClenachan et al. (2020) (Fig. 24). The majority of the shoreline was dominated by patterns of erosion (dark grey and black), and areas denoting land acquisition (white and light grey) corresponded to the location of mature *A. germinans*.

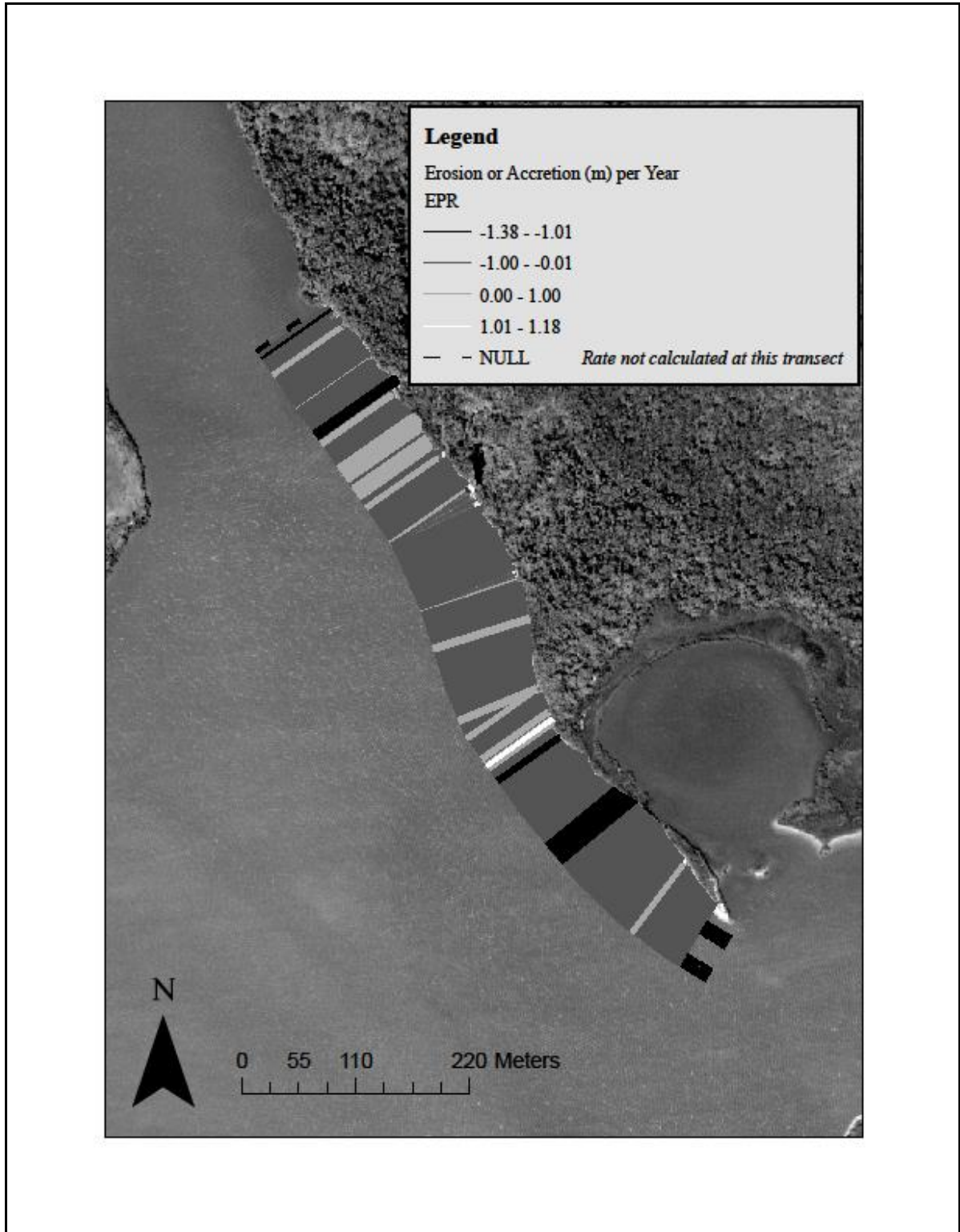


Figure 24: Land loss and gain at restoration site between 2007 and 2017.

CHAPTER 4: DISCUSSION

Deploying 3 to 4 year-old adult mangroves and utilizing a breakwater were important strategies for the retention and thus success of the living shoreline stabilization project in Mosquito Lagoon, a shallow-water estuary in Florida with wind-driven circulation and an annual fall high water season (Provost, 1973; Smith, 1987, 1993; Brockmeyer et al., 1996). Compared to planting seedlings, survival odds increased by 186.4% when transitional mangroves were used and 1,086.9% when adult mangroves were used. For mangroves in the seaward and landward row, respectively, survival odds increased 436.8% and 197.5% when a breakwater was present. The timing of mangrove mortalities throughout the 12-month period indicates that flooding stress was the most important factor influencing success of the living shoreline (Fig. 14). Increased survival in older plants was linked to their larger starting diameter, height, and number of anchored prop roots at the beginning of the experiment (Fig. 11), which increased access and storage of oxygen needed for cellular respiration (Chen et al., 2006). Breakwater presence did not influence sediment patterns around the base of the mangroves within 1 year of the restoration (Fig. 19) but increased survival significantly, indicating that wave energy was a source of stress, compounding the effects of flooding (Balke et al., 2013). Since initial size metrics were crucial for withstanding flooded conditions, deployment after the commencement of high-water season, presents an effective strategy to allow for maximum growth before flooding stress. Due to higher temperatures and greater exposure to sunlight, the majority of *R. mangle* shoot growth occurs during the summer months and the threat of freeze events are minimal (Gill and Tomlinson, 1971). Consequently, the beginning of summer offers optimal conditions for deploying a mangrove living shoreline in the IRL and other areas with similar conditions.

The cause of mangrove mortalities was reinforced by the magnitude of standing dead mangroves, which is consistent with flooding stress as opposed to forceful removal of the plant by wave energy (Bouchon et al., 1994; Hoppe-Speer et al., 2011). Of the 229 mangrove mortalities which occurred by the end of October, 83.4% were standing dead. With the largest recorded wave height being 34 cm, it is plausible that wave energy played a role in the physical removal of mangrove leaves. However, since mangroves as tall as 89 cm progressed to standing dead and number of leaves was not a main influencer of mangrove survival, forceful removal of leaves was most likely an additive effect as opposed to the sole influence. Leaf loss, as a result of flooding stress, was reinforced by Hoppe-Speer et al. (2011), who documented the same response in 3-month old *Rhizophora mucronata* (red mangrove) seedlings that were partially submerged in 30 cm of water, with top foliage remaining exposed, for 14 weeks, 24 hours a day. Mangrove mortalities at the restoration site were most likely not the result of poor deployment practices; planting error usually results in displacement of the roots and mortalities occur within the first month post-deployment (MD, pers. obs.). Mangrove leaf damage has been documented in previous research as a result of insect herbivory, but damage to *Rhizophora* spp. often does not result in complete defoliation of the plant (Feller, 1995; Ellison and Farnsworth, 1996; Feller and Mathis, 1997; Duke, 2002; Burrows, 2006). Evidence of herbivory on the deployed mangroves was present on 4 plants in 4 different treatment replicates; 1 mangrove had a *Phocides pigmalion* (mangrove skipper) and insect bite marks present, whereas 3 mangroves had just insect bite marks. Wildlife cameras that were placed along this restoration site captured 1,419 observations of mammals (56% raccoons, 37% hogs, and 4% deer) (Rifenberg et al., 2021). Of those observations, 15 captured mammals contacted the deployed mangroves, but no dislodgement or consumption occurred. Standing dead trees can also be the result of freezing temperatures

(Osland et al., 2019), but this threshold was never reached over the 12-month monitoring period. If branches with a diameter > 2.5 cm are not broken and if root rot has not occurred, standing dead *R. mangle* have the possibility of recovering (Snedaker, 1995; Smith et al., 2009; Kodikara et al., 2017). However, the exact length of time *R. mangle* can be standing dead before recovering is not known (Snedaker, 1995; Barr et al., 2012; Goldberg and Hein, 2018). Monitoring conducted 4 months, after the destruction of Typhoon Sudal in April 2004, revealed that 64% of the *R. mangle* refoliated after being classified as standing dead (Kauffman and Cole, 2010). Comparatively, 9 of the mangroves had refoliated within 8 months on my experimental living shoreline,

The results of this study highlight the importance of understanding year-long hydrology for choosing the proper planting elevation. Florida restoration recommendations state mangroves should be placed in the middle to upper intertidal region where they are flooded on average 30% per day (Lewis, 2005; Primavera and Esteban, 2008). Provost (1973), however, points out the difficulty that Florida is faced with defining the high-water mark due to its seasonal variation. Data acquired from Haulover Canal showed that over 13 years, mean high water level varied by 12 cm, which can have large impacts on young mangroves survival (Figs. 14, 17). For example, mangroves in the seaward row were placed ~ 0.7 m away from the landward row over a mean slope of 0.12; therefore, the seaward row underwent greater flood and wave energy. Being planted in the seaward row decreased survival probability by 241.1% and increased the need for a breakwater structure. Seasonal changes in water level are not restricted to the coasts of Florida, but are also observed in tropical areas of Asia, Australia, West Africa, and the Americas that experience a monsoon season (Webster et al., 1998). The negative impacts of monsoonal flooding on the survival of both planted and naturally recruited mangroves have been reported in

Malaysia, Sri Lanka, and South Vietnam (Hashim et al., 2010; Nguyen, 2013; Motamedi et al., 2014; Jayarathne et al., 2020).

While mangroves placed at lower elevations are exposed to greater flooding (Ball, 1980; Ellison and Farnsworth, 1997; Pezeshki et al., 1997), mangroves placed at higher elevations risk desiccation, increased salinity stress, slower growth, competition with other plants, and increased predation (Pool et al., 1977; Ball, 1980; Ellison and Farnsworth, 1997; Ball, 1998; Imbert et al., 2000, Lewis, 2005; Hoppe-Spear et al., 2011; Lovelock et al., 2017). At the time of the deployment in mid-June, planted mangroves were placed at the elevation observed by the closest mature, naturally-recruited *R. mangle*, where they were slightly flooded (1-2 cm) in that month during high tide. This seemingly appropriate placement was not suitable for the survival of younger mangroves. The discrepancy could be explained by the results of the GIS analysis that revealed the restoration site underwent erosion from the years 2007 to 2017, suggesting that when the mature *R. mangle* trees first recruited, the elevation may have been higher than it was at the time of the restoration. Additionally, the center of the adult red mangrove prop root mass (~1.5 m wide) was referenced for planting the young mangroves. Based on the extensive flooding the deployed mangroves experienced, placing the seaward row at the elevation where the adult prop roots originated could be an effective future planting strategy for combatting seasonal flooding. Even with historical data readily available, predicting seasonal flooding magnitude and duration can be very challenging (Webster et al., 1998). Fortunately, my study showed that using adult *R. mangle* and a breakwater was a successful strategy for combatting extreme flooded conditions.

Choosing mangroves for a shoreline stabilization project based on development of a woody stem is a simple guideline that removes the need for extensive measurements. This

method was an easy way to identify age and guarantee a plant with larger dimensions. Terms such as “seedling” and “sapling” are commonly used to describe young mangroves, but the exact definition can be based on age or various size measurements depending on the author and type of research being conducted. For example, Tamai and Iampa (1988) classified a sapling as greater than 1 year-old, Ashton and Macintosh (2002) considered saplings to be greater than 1 meter tall but less than 4 cm diameter, and Farnsworth and Ellison (1996) as having 1 to 2 aerial roots and between 1 and 3 growing shoot tips. Whether a restoration manager decides to grow their own mangroves or purchase them from a nursery, tracking age for a large number of mangroves can be difficult. Furthermore, mangroves that are the same age have been observed to grow at different rates and allocate growth to different areas based on environmental factors such as shading and soil nutrients (Farnsworth and Ellison, 1996; Feller et al., 2003). Classifying mangroves based on the progression from a soft, herbaceous stem to a fully woody stem is an easy method for choosing mangroves. Moreover, for potential living shoreline restorations that parallel the conditions of my site, choosing mangroves with a woody stem is a proven method for increasing *R. mangle* survival.

Parallel with previous observations of mangrove growth, younger mangroves allocated the majority of growth to increasing height, whereas older mangroves spent energy on branch, prop root, and flower production (Kathiresan and Bingham, 2001; Nagarajan et al., 2008; Primavera and Esteban, 2008). Vertical growth rate for adults and seedlings increased with breakwater presence, consistent with attenuation of wave energy (Balke et al., 2013); but vertical growth rate for transitionals decreased with breakwater presence (Fig. 5). It is possible that the critical flooding pressure needed to trigger allocation of more resources to vertical growth was different for transitionals with and without a breakwater (Ellison and Farnsworth, 1997; He et al.,

2007; Balke et al., 2013; Hoppe-Spear et al., 2011). The mean starting height required for overall mangrove survival was lowered by 5.1 cm when a breakwater was present, indicating the structures were effectively reducing wave height (Achab et al., 2014; Spiering et al., 2018). During high water season, when water level was between 12 and 34 cm around the deployed mangroves, transitional mangroves were a mean of 50.9 cm tall. Transitional mangroves without a breakwater allocated more growth to increasing height (12.5 cm yr^{-1}) compared to transitionals that had a breakwater (6.9 cm yr^{-1}) (Fig. 5), indicating that transitional mangroves with a breakwater may have represented the threshold where allocation to vertical growth was no longer activated by the seasonal flooding.

Of the 157 living adults after 12 months, 51.6% produced flowers or flower buds, illustrating that flooding or wave energy stress were minimized. Propagule production from a biological standpoint indicates reproductive success and flower growth has been used as an indicator of long-term mangrove restoration success (Nagarajan et al., 2008; Primavera and Esteban, 2008). At my site, mangroves that were adults at the beginning of the experiment showed propagule growth, and compared to seedlings, odds of flower and flower bud production increased by 933.1% for transitional mangroves and 6724.5% for adult mangroves. Only 5 seedlings showed signs of flowering after 12 months for a total of 16 flower buds.

Forty-four mangroves (transitionals and adults) experienced branch loss over the monitoring period but never reached the standing dead category (Fig. 7). A branch was not counted if it did not have any leaves, making branch number a good representation of leaf number as well. Rate of branch loss was highly variable (-1 to -20 branches), and small changes could be attributed to natural leaf loss (Gill and Tomlinson, 1971). Of the mangroves that lost branches, 51.6% had > 50 leaves. Further monitoring would be required to see if these trends of

branch loss reverse over time or eventually lead to the standing dead state. Seedlings only had 1 or 2 branches with between 5 and 18 leaves. Therefore, all seedlings that lost branches experienced mortality by the end of the monitoring period.

Stem diameter growth was not greater for younger mangroves as would be expected (Kathiresan and Bingham, 2001) (Fig 6.). Changes in sediment level could have reduced the accuracy of the diameter measurements since they were taken at the base of the mangroves (the thickest point). As sediment accreted, the point at which diameter was taken changed to a higher, possibly thinner portion of the stem. Four mangroves had a negative reading for change in diameter from month 0 to month 12.

Planting seedlings haphazardly among older mangroves was not an effective way to increase seedling survival in Mosquito Lagoon. Some studies have found positive effects of deploying mixed age plant communities in restoration (Cody, 1993; Ashton et al., 1997; Dulohery et al., 2000; Valenzuela et al., 2016), while others have not seen these facilitative effects (De Steven, 1991; Callaway and Walker, 1997; Coomes and Allen, 2007; Gomez-Aparico, 2009). In the case of this experiment, the reason for seedling failure was most likely two-fold. During flooding pressure in the fall, seedlings at a mean height of 42.7 cm were partially submerged for approximately 2 months with maximum wave heights up to 32 cm. When the site was visited in October for survival monitoring, 192 (87.2%) of the seedlings had their leaves exposed. Secondly, wave energy attenuation was not sufficient to significantly increase survival.

The shoreline materials used for the restoration were not adequate for trapping wrack and mangrove propagules for time periods long enough to be registered by a quarterly monitoring scheme. Seagrass wrack contains nutrients that could potentially increase the growth of living

shoreline restorations (Goforth and Thomas, 1980; Breithaupt et al., 2019), but wrack distribution was evenly dispersed along the shoreline, so there was no discernable difference in growth rate or survival of planted *R. mangle* based on wrack thickness or cover. The dominant component of wrack in Mosquito Lagoon is *H. wrightii*, with highest abundance in the fall from the natural senescence cycle of seagrass (Breithaupt et al., 2019). Although wrack compositions were similar (RF, pers. obs.), abundance was close to 0 in fall of 2019. This inconsistency was most likely caused by the differences in methodology between the 2 studies. Breithaupt et al. (2019) collected the wrack from the high water line, whereas I collected wrack at the elevation of the planted *R. mangle*, which was ~2 m below the high water line during the fall and early winter. Goforth and Thomas (1980) found that wrack accumulation can result in the smothering and failure of planted *R. mangle* under 18 months-old. Seeing as Breithaupt et al. (2019) reported monthly wrack accumulation in Mosquito Lagoon to be between 32 and 44 g m⁻², it is possible that the elevated location of the wrack line increased deployed mangrove survival through the decreased chance of smothering. When the site was visited after Hurricane Dorian (19 September 2019), wrack was observed hanging on some of the planted *R. mangle*. When the site was revisited 2 weeks later for wrack monitoring, the majority of the hanging wrack on the mangroves had been removed by the high water levels and pushed further inland. Although desiccated, single shoots of *H. wrightii* still remained on some of the plants, it did not appear thick enough to block sunlight from the leaves or cause any of the mangroves to bend. Of the mangroves that died throughout the 12-month period, the 8% that occurred in September may have been influenced by wrack.

Propagule abundance at the site was surprisingly low considering that the shorelines of Mosquito Lagoon are dominated by mangroves (Dybas, 2002). A similar finding had been noted

previously by Donnelly et al. (2017); they found lower mangrove propagule and seedling abundance at shell middens compared to shorelines with smaller grain sizes. Donnelly et al. (2017) additionally demonstrated in greenhouse experiments that *R. mangle* propagules were capable of penetrating shelly substrates and surviving rooted in the sediment. Thus, they hypothesized that the low natural seedling recruitment at shell middens was due to the unstable nature of the disarticulated shell. Video footage which captured boating activity indicated that the wave energy experienced by the restoration site was mostly natural. The ~150 m channel adjacent (west) to the restoration site (Fig. 1) is bordered on the other side by an island. The channel progressively gets deeper as it approaches the island across from the restoration site. As a result, the only boats that were observed on plane were far enough away that the waves dissipated before reaching the mangroves. The analysis was limited by the range of the wildlife cameras (60 m) and could explain the minimal number of boats captured on plane. However, the indication of the footage that few boats came to the area and that boats were planing on the other side of the channel did line up with observations made in the field while monitoring (RF, pers. obs.). Because boating activity can increase local wave energy and limit mangrove recruitment (Donnelly et al., 2017), the low boating activity at the site indicates the low natural mangrove recruitment rate was likely the result of the shelly substrate.

Natural mangrove propagule dispersal was most abundant during fall monitoring (5 October 2019), when the area experienced a high-water season. The timing of the high water season most likely limited the number of propagules being placed among the planted *R. mangle* and indicated the complexity of the *R. mangle* and breakwater was not able to trap passing propagules. The majority of propagules discovered were landward of the *R. mangle* ecotone where there was more surface complexity from wrack, dead wood, and upland vegetation. Lack

of complexity at the elevation suitable for mangrove settlement has been previously observed to result in propagules appearing at elevations too high for survival (Smith et al., 2020). A similar process could have been responsible for the seedling abundance along the restoration site that was observed to decrease from October to July. Thus, further research is needed on living shoreline methodology that increases propagule recruitment within the first year of a shell midden restoration. Possible solutions would be to plant a wider range of plant species to cover a greater range of elevations. Plants that have already been shown to trap Florida mangrove propagules species include *A. germinans*, *Spartina alterniflora* (marshgrass), *Sesuvium portulacastrum* (sea purslane), *Distichlis spicata* (salt grass), *B. maritima* and *S. perennis* (Lewis, 2005; McKee et al., 2007; Donnelly and Walters, 2014; Millan-Aguilar et al., 2016). Although my experimental living shoreline method was not effective at trapping propagules, it is possible that propagule retention will increase over time as the planted *R. mangle* grow more complex and produce their own propagules.

The substrate at the restoration site was very shelly (Fig. 23), which is characteristic of shell middens (Alvarez et al., 2011). Compared to finer sediments (i.e. sand, mud), sediment transport at the experimental shoreline should be reduced and anchoring strength of the planted mangrove increased (Schutten et al., 2005; Boizard and Mitchell, 2010; Peterson et al., 2014). The shelly sediment provided an opportunity to isolate the impact of flooding and wave energy on mangrove survival and growth, separate from the threat of removal from erosion, regardless of breakwater presence. Many other restoration sites are made up of sandy and muddy sediments, therefore the results of this experimental living shoreline may not be directly comparable. Detached breakwaters can effectively reduce incoming velocity, lower wave height, and lead to the buildup of sediment landward of the structure. The effectiveness of a breakwater, however,

depends on many factors including breakwater design, sediment supply, land use, distance of the structure from the coast, tide level, sediment type, slope, and wave energy (Akbar et al., 2017; Palinkas et al., 2017; Fitri et al., 2019; Vona et al., 2020). There is evidence that *R. apiculata* seedlings planted in sediment with a mean grain size of 0.016 mm could be forcibly removed by high tides even with the presence of a breakwater (Motamedi et al., 2014). On the opposite spectrum, a study by Kamali and Hashim (2011) showed that breakwater presence can lead to accretion extreme enough to smother planted *A. marina* seedlings. Larger mangroves have the potential to combat both erosion and accretion through larger root and shoot systems, respectively (Terrados et al., 1997; Balke et al., 2011; Tamin et al., 2011; Pilato, 2019), but further research is needed to know how the experimental design would impact *R. mangle* living shoreline success at sites composed of different sediment types.

CHAPTER 5: CONCLUSION

Testing different restoration strategies is an essential step for increasing future living shoreline stabilization success defined by the survival and growth of the planted vegetation. The results of my experiment indicate that a site's annual hydrology should be considered before deploying a mangrove living shoreline. Seasonal flooding, which is often highly unpredictable, can result in large mortality rates of young mangroves through the combined impact of flooding and wave energy stress. Using older mangroves, which have woody stems, in combination with a breakwater structure, is an effective strategy for combatting flooded conditions. Mangrove survival in these treatments in my experimental *R. mangle* living shoreline was 87.5%; of these surviving mangroves, 64.3% showed signs of biological success in the form of flowers and flowers buds within 1 year. Older mangroves increase survival through their larger stems and greater number of prop roots which are better able to withstand wave and flooding pressure.

Breakwater presence increased mangrove survival through the reduction of wave velocity and wave height. If seasonal flooding occurs, mangroves should be placed above the lowest high water level (HWL) to avoid flooding pressure and below the highest HWL to avoid smothering by wrack. Observing the elevation of nearby, naturally-recruited adult mangroves of the same species is a good first step for choosing planting elevation. To analyze how nearby elevation may have changed since the time when the mature mangroves recruited, historical aerial imagery can be evaluated using ArcMap 10.6 software as described in McClenachan et al. (2020). Planting younger mangroves haphazardly among older mangroves is not an effective method for increasing living shoreline success if the site experiences extensive seasonal flooding. Ensuring

the success of planted mangroves is especially important along shell middens since natural mangrove propagule recruitment is severely limited by the shelly substrate.

CHAPTER 6: MANAGEMENT RECOMMENDATIONS

To establish that a shoreline is in need of stabilization, look for: 1) minimal natural mangrove recruitment, and 2) signs of erosion. Signs of erosion include the presence of scarps and a receding vegetation line, visualized through aerial imagery and field visits. Analyze the hydrology of the restoration site to choose the proper restoration materials and planting location. High wave energy can be natural, driven by large fetches and high wind speeds, or it can be the byproduct of boating activity. If the site has high wave energy or a seasonal high water season, utilize a breakwater structure and select mangroves with a woody stem and anchored-prop roots. If feasible, monitor water level of the site for a minimum of 1 year prior to restoration either directly or by accessing a nearby monitoring station that has historical data. Place the mangroves where they will be inundated ~30% of the year. To reach this optimal goal, reference nearby naturally-recruited mangroves of the same species. If *R. mangle*, plant the young mangroves where the adult red mangrove first recruited (landward side of the prop root mass). If GIS analysis of the restoration site indicates a quickly receding vegetation line, young mangroves may need to be placed further landward of the reference adult mangroves. Boundaries on the landward side, where mangroves should not be planted, include the dominant wrack line, above the highest high water line, and areas where transitional or upland vegetation are present. If possible, deploy the mangroves at the commencement of the high water season and the beginning of growing season.

REFERENCES

- Achab, M., Ferreira, O., & Dias, J. A. (2014). Evaluation of sedimentological and morphological changes induced by the rehabilitation of sandy beaches from the Ria Formosa Barrier Island system (South Portugal). *Thalassas*, 30(2), 21-31.
- Akbar, A. A., Sartohadi, J., Djohan, T. S., & Ritohardoyo, S. (2017). The role of breakwaters on the rehabilitation of coastal and mangrove forests in west Kalimantan, Indonesia. *Ocean & Coastal Management*, 138, 50-59. doi:10.1016/j.ocecoaman.2017.01.004
- Álvarez, M., Briz Godino, I., Balbo, A., & Madella, M. (2011). Shell middens as archives of past environments, human dispersal and specialized resource management. *Quaternary International*, 239(1-2), 1-7. doi:10.1016/j.quaint.2010.10.025
- Anderson, C., & Lee, S. Y. (1995). Defoliation of the mangrove *Avicennia marina* in Hong Kong: cause and consequences. *Biotropica*, 218-226.
- Ashton, P. M., Gamage, S., Gunatilleke, I. A., & Gunatilleke, C. V. (1997). Restoration of a Sri lankan Rainforest: Using Caribbean Pine *Pinus Caribaea* as a nurse for Establishing Late-Successional tree species. *The Journal of Applied Ecology*, 34(4), 915. doi:10.2307/2405282
- Ashton, E. C., & Macintosh, D. J. (2002). Preliminary assessment of the plant diversity and community ecology of the sematan mangrove forest, Sarawak, Malaysia. *Forest Ecology and Management*, 166(1-3), 111-129. doi:10.1016/s0378-1127(01)00673-9
- Azzalini, A. (1985). A class of distributions which includes the normal ones. *Scandinavian journal of statistics*, 171-178.
- Balke, T., Bouma, T., Horstman, E., Webb, E., Erfteimeijer, P., & Herman, P. (2011). Windows of opportunity: Thresholds to mangrove seedling establishment on tidal flats. *Marine Ecology Progress Series*, 440, 1-9. doi:10.3354/meps09364
- Balke, T., Webb, E. L., Den Elzen, E., Galli, D., Herman, P. M., & Bouma, T. J. (2013). Seedling establishment in a dynamic sedimentary environment: A conceptual framework using mangroves. *Journal of Applied Ecology*, 50(3), 740-747. doi:10.1111/1365-2664.12067
- Ball, M. C. (1980). Patterns of secondary succession in a mangrove forest of Southern Florida. *Oecologia*, 44(2), 226-235. doi:10.1007/bf00572684
- Ball, M. C. (1988). Ecophysiology of mangroves. *Trees*, 2(3). doi:10.1007/bf00196018

- Ball, M. C. (1998). Mangrove Species Richness in Relation to Salinity and Waterlogging: A Case Study Along the Adelaide River Floodplain, Northern Australia. *Global Ecology and Biogeography Letters*, 7(1), 73. doi:10.2307/2997699
- Baran, E., & Hambrey, J. (1999). Mangrove Conservation and Coastal Management in Southeast Asia: What Impact on Fishery Resources? *Marine Pollution Bulletin*, 37(8-12), 431-440. doi:10.1016/s0025-326x(99)00076-4
- Barr, J. G., Engel, V., Smith, T. J., & Fuentes, J. D. (2012). Hurricane disturbance and recovery of energy balance, CO₂ fluxes and canopy structure in a mangrove forest of the Florida Everglades. *Agricultural and Forest Meteorology*, 153, 54-66. doi:10.1016/j.agrformet.2011.07.022
- Bayraktarov, E., Saunders, M. I., Abdullah, S., Mills, M., Beher, J., Possingham, H. P., Mumby, P.J., & Lovelock, C. E. (2016). The cost and feasibility of marine coastal restoration. *Ecological Applications*, 26(4), 1055-1074. doi:10.1890/15-1077
- Beck, M., & Airoldi, L. (2007). Loss, Status and Trends for Coastal Marine Habitats of Europe. *Oceanography and Marine Biology Oceanography and Marine Biology - An Annual Review*, 345-405. doi:10.1201/9781420050943.ch7
- Bilkovic, D. M., Mitchell, M., Mason, P., & Duhring, K. (2016). The role of living shorelines as estuarine habitat conservation strategies. *Coastal Management*, 44(3), 161-174. doi:10.1080/08920753.2016.1160201
- Bilkovic, D. M., Mitchell, M., Davis, J., Andrews, E., & King, A. (2017). Review of boat wake impacts on shoreline erosion and potential solutions for the Chesapeake Bay (pp. 20-22, Rep.). Edgewater, Maryland: STAC. doi: https://scholarworks.wm.edu/reports/1271
- Boizard, S. D., & Mitchell, S. J. (2010). Resistance of red mangrove (*Rhizophora mangle* L.) seedlings to deflection and extraction. *Trees*, 25(3), 371-381. doi:10.1007/s00468-010-0512-z
- Bouchon, C., Bouchon-Navaro, Y., & Louis, M. (1994). Changes in the coastal fish communities following Hurricane Hugo in Guadelope Island (French West Indies). *Atoll Research Bulletin*.
- Bozek, C. M., & Burdick, D. M. (2005). Impacts of Seawalls on Saltmarsh Plant Communities in the Great Bay Estuary, New Hampshire USA. *Wetlands Ecology and Management*, 13(5), 553-568. doi:10.1007/s11273-004-5543-z
- Breithaupt, J. L., Duga, E., Witt, M., Fillyaw, R., Friedland, N., Donnelly, M. J, Walters, L.J., & Chambers, L. G. (2019). Carbon and nutrient fluxes from seagrass and mangrove wrack are mediated by soil interactions. *Estuarine, Coastal and Shelf Science*, 229, 106409. doi:10.1016/j.ecss.2019.106409

- Brockmeyer, R. E., Rey, J. R., Virnstein, R. W., Gilmore, R. G., & Earnest, L. (1996). Rehabilitation of impounded estuarine wetlands by hydrologic reconnection to the Indian River Lagoon, Florida (USA). *Wetlands Ecology and Management*, 4(2), 93-109. doi:10.1007/bf01876231
- Bulleri, F., & Chapman, M. G. (2010). The introduction of coastal infrastructure as a driver of change in marine environments. *Journal of Applied Ecology*, 47(1), 26-35. doi:10.1111/j.1365-2664.2009.01751.x
- Burrows, D. W. (2006). The role of insect leaf herbivory on the mangroves *Avicennia marina* and *Rhizophora stylosa* (Unpublished master's thesis). James Cook University.
- Butler, A. (2019, September 07). Spared: Hurricane Dorian: Tropical storm. Retrieved March 10, 2021, from <https://daytonatimes.com/2019/09/07/spared/>
- Bywater-Reyes, S., Wilcox, A. C., Stella, J. C., & Lightbody, A. F. (2015). Flow and scour constraints on uprooting of pioneer woody seedlings. *Water Resources Research*, 51(11), 9190-9206. doi:10.1002/2014wr016641
- Cahoon, D. R., Hensel, P., Rybczyk, J., Mckee, K. L., Proffitt, C. E., & Perez, B. C. (2003). Mass tree mortality leads to mangrove peat collapse at Bay Islands, Honduras after Hurricane Mitch. *Journal of Ecology*, 91(6), 1093-1105. doi:10.1046/j.1365-2745.2003.00841.x
- Callaway, R. M., & Walker, L. R. (1997). Competition and facilitation: A synthetic approach to interactions in plant communities. *Ecology*, 78(7), 1958-1965. doi:10.1890/0012-9658(1997)078[1958:cafasa]2.0.co;2
- Cannon, D., Kibler, K., Donnelly, M., McClenachan, G., Walters, L., Roddenberry, A., & Phagan, J. (2020). Hydrodynamic habitat thresholds for mangrove vegetation on the shorelines of a microtidal estuarine lagoon. *Ecological Engineering*, 158, 106070. doi:10.1016/j.ecoleng.2020.106070
- Cappucci, M. (2019, September 06). Stunning portraits of hurricane Dorian's fury. Retrieved January 5, 2021, from <https://www.washingtonpost.com/weather/2019/09/07/stunning-portraits-hurricane-dorians-fury/>
- Carlton, J. M. (1974). Land-building and Stabilization by Mangroves. *Environmental Conservation*, 1(4), 285-294. doi:10.1017/s0376892900004926
- Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., & Feller, I. C. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *Proceedings of the National Academy of Sciences*, 111(2), 723-727. doi:10.1073/pnas.1315800111

- Cavanaugh, K. C., Dangremond, E. M., Doughty, C. L., Williams, A. P., Parker, J. D., Hayes, M. A., Rodriguez, W., & Feller, I. C. (2019). Climate-driven regime shifts in a mangrove–salt marsh ecotone over the past 250 years. *Proceedings of the National Academy of Sciences*, *116*(43), 21602-21608. doi:10.1073/pnas.1902181116
- Chapman, M. G., & Roberts, D. E. (2004). Use of seagrass wrack in restoring disturbed Australian saltmarshes. *Ecological Management and Restoration*, *5*(3), 183-190. doi:10.1111/j.1442-8903.2004.00207.x
- Chen, L. Z., Lin, P., & Wang, W. Q. (2006). Mechanisms of mangroves waterlogging resistance. *Acta Ecologica Sinica*, *26*(2), 586-593.
- Cody, M. L. (1993). Do cholla cacti (*Opuntia* spp., subgenus *Cylindropuntia*) use or need nurse plants in the Mojave Desert? *Journal of Arid Environments*, *24*(2), 139-154. doi:10.1006/jare.1993.1013
- Coomes, D. A., & Allen, R. B. (2007). Effects of size, competition and altitude on tree growth. *Journal of Ecology*, *95*(5), 1084-1097. doi:10.1111/j.1365-2745.2007.01280.x
- Craighead, F. C., & Gilbert, V. C. (1962). The Effects of Hurricane Donna on the vegetation of Southern Florida. *Quarterly Journal of the Florida Academy of Sciences*, *25*(1), 1-28. doi:https://www.jstor.org/stable/24315186
- Currin, C. A. (2019). Living Shorelines for Coastal Resilience. *Coastal Wetlands*, 1023-1053. doi:10.1016/b978-0-444-63893-9.00030-7
- De Steven, D. (1991). Experiments on mechanisms of Tree establishment in Old-Field Succession: Seedling survival and growth. *Ecology*, *72*(3), 1076-1088. doi:10.2307/1940607
- Donnelly, M., & Walters, L. (2014). Trapping of *Rhizophora mangle* Propagules by Coexisting Early Successional Species. *Estuaries and Coasts*, *37*(6), 1562-1571. doi:10.1007/s12237-014-9789-2
- Donnelly, M., Shaffer, M., Connor, S., Sacks, P., & Walters, L. (2017). Using mangroves to stabilize coastal historic sites: Deployment success versus natural recruitment. *Hydrobiologia*, *803*(1), 389-401. doi:10.1007/s10750-017-3155-x
- Doody, J. P. (2004). ‘Coastal squeeze’— an historical perspective. *Journal of Coastal Conservation*, *10*(1), 129-138. doi:10.1007/bf02818949
- Dugan, J., Airoidi, L., Chapman, M., Walker, S., & Schlacher, T. (2011). Estuarine and coastal structures. *Treatise on Estuarine and Coastal Science*, 17-41. doi:10.1016/b978-0-12-374711-2.00802-0

- Duke, N. C. (2002). Sustained high levels of foliar herbivory of the mangrove *Rhizophora stylosa* by a moth larva *Doratifera stenosa* (Limacodidae) in north-eastern Australia. *Wetlands Ecology and Management*, 10(5), 403-419. doi:10.1007/bf03263357
- Dulohery, C., Kolka, R., & McKevlin, M. (2000). Effects of a willow overstory on planted seedlings in a bottomland restoration. *Ecological Engineering*, 15. doi:10.1016/s0925-8574(99)00072-5
- Dybas, C. L. (2002). Florida's Indian River Lagoon: An Estuary in transition. *BioScience*, 52(7), 555. doi:10.1641/0006-3568(2002)052[0555:fsirla]2.0.co;2
- Ellison, A. M., & Farnsworth, E. J. (1996). Spatial and Temporal variability in growth of *Rhizophora Mangle* Saplings on Coral Cays: Links with variation insolation, herbivory, and local sedimentation rate. *The Journal of Ecology*, 84(5), 717. doi:10.2307/2261334
- Ellison, A. M., & Farnsworth, E. J. (1997). Simulated sea level change alters anatomy, physiology, growth, and reproduction of red mangrove (*Rhizophora mangle* L.). *Oecologia*, 112(4), 435-446. doi: 10.1007/s004420050330
- Ellison, A. M., Bank, M. S., Clinton, B. D., Colburn, E. A., Elliott, K., Ford, C. R., Foster, D.R., Kloeppe, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B., & Webster, J. R. (2005). Loss of foundation species: Consequences for the structure and dynamics of forested ecosystems. *Frontiers in Ecology and the Environment*, 3(9), 479-486. doi:10.1890/1540-9295(2005)003[0479:lofscf]2.0.co;2
- Elster, C. (2000). Reasons for reforestation success and failure with three mangrove species in Colombia. *Forest Ecology and Management*, 131(1-3), 201-214. doi:10.1016/s0378-1127(99)00214-5
- Estrada, G. C., Soares, M. L., Fernadez, V., & Almeida, P. M. (2014). The economic evaluation of carbon storage and sequestration as ecosystem services of mangroves: A case study from southeastern Brazil. *International Journal of Biodiversity Science, Ecosystem Services & Management*, 11(1), 29-35. doi:10.1080/21513732.2014.963676
- Farnsworth, E. J., & Ellison, A. M. (1996). Sun-shade adaptability of the red mangrove, *Rhizophora mangle* (*Rhizophoraceae*): Changes through ontogeny at several levels of biological organization. *American Journal of Botany*, 83(9), 1131-1143. doi:10.1002/j.1537-2197.1996.tb13893.x
- Faunce, C., & Serafy, J. (2006). Mangroves as fish habitat: 50 years of field studies. *Marine Ecology Progress Series*, 318, 1-18. doi:10.3354/meps318001

- Feller, I. C. (1995). Effects of nutrient enrichment on growth and herbivory of dwarf red mangrove (*Rhizophora mangle*). *Ecological Monographs*, 65(4), 477-505. doi:10.2307/2963499
- Feller, I. C., & Mathis, W. N. (1997). Primary herbivory by wood-boring insects along an architectural gradient of *Rhizophora mangle*. *Biotropica*, 29(4), 440-451. doi:10.1111/j.1744-7429.1997.tb00038.x
- Feller, I. C., Whigham, D. F., McKee, K. L., & Lovelock, C. E. (2003). Nitrogen limitation of growth and nutrient dynamics in a disturbed mangrove forest, Indian River Lagoon, Florida. *Oecologia*, 134(3), 405-414. doi:10.1007/s00442-002-1117-z
- Fitri, A., Hashim, R., Abolfathi, S., & Abdul Maulud, K. N. (2019). Dynamics of sediment transport and erosion-deposition patterns in the locality of a detached low-crested breakwater on a cohesive coast. *Water*, 11(8), 1721. doi:10.3390/w11081721
- Gill, A. M., & Tomlinson, P. B. (1971). Studies on the growth of red mangrove (*Rhizophora mangle* L.) 3. Phenology of the shoot. *Biotropica*, 3(2), 109. doi:10.2307/2989815
- Gittman, R. K., Scyphers, S. B., Smith, C. S., Neylan, I. P., & Grabowski, J. H. (2016). Ecological Consequences of Shoreline Hardening: A Meta-Analysis. *BioScience*, 66(9), 763-773. doi:10.1093/biosci/biw091
- Goforth Jr, H. W., & Thomas, J. R. (1980). Plantings of Red Mangroves (*Rhizophora Mangle* L.) for Stabilization of Marl Shorelines in the Florida Keys. Naval Ocean Systems Center San Diego, CA.
- Goldberg, N., & Heine, J. (2018). Effects of Hurricane Matthew on *Rhizophora mangle* L. survivorship and reproduction at its northern range limit in northeastern Florida. *Florida Scientist*, 81(1), 55-60.
- Gómez-Aparicio, L. (2009). The role of plant interactions in the restoration of degraded ecosystems: A meta-analysis across life-forms and ecosystems. *Journal of Ecology*, 97(6), 1202-1214. doi:10.1111/j.1365-2745.2009.01573.x
- Gorman, D., & Turra, A. (2016). The role of mangrove revegetation as a means of restoring macrofaunal communities along degraded coasts. *Science of The Total Environment*, 566-567, 223-229. doi:10.1016/j.scitotenv.2016.05.089
- Gorman, R. M., & Neilson, C. G. (1999). Modelling shallow water wave generation and transformation in an intertidal estuary. *Coastal Engineering*, 36(3), 197-217. doi:10.1016/s0378-3839(99)00006-x
- Hair, J.R., Anderson, R.E., Tatham, R.L., & Black, W.C. (1995) *Multivariate Data Analysis with Readings*. Prentice Hall, Englewood, NJ

- Hardy, T., & Wu, W. (2020). Impact of different restoration methods on coastal wetland loss In Louisiana: Bayesian analysis. *Environmental Monitoring and Assessment*, 193(1). doi:10.1007/s10661-020-08746-9
- Harris, R. J., Milbrandt, E. C., Everham, E. M., & Bovard, B. D. (2010). The effects of reduced tidal flushing on mangrove structure and function across a disturbance gradient. *Estuaries and Coasts*, 33(5), 1176-1185. doi:10.1007/s12237-010-9293-2
- Hashim, R., Kamali, B., Tamin, N. M., & Zakaria, R. (2010). An integrated approach to coastal rehabilitation: Mangrove restoration in Sungai Haji Dorani, Malaysia. *Estuarine, Coastal and Shelf Science*, 86(1), 118-124. doi:10.1016/j.ecss.2009.10.021
- He, B., Lai, T., Fan, H., Wang, W., & Zheng, H. (2007). Comparison of flooding-tolerance in four mangrove species in a diurnal tidal zone in the Beibu Gulf. *Estuarine, Coastal and Shelf Science*, 74(1-2), 254-262. doi:10.1016/j.ecss.2007.04.018
- Heerhartz, S. M., Toft, J. D., Cordell, J. R., Dethier, M. N., & Ogston, A. S. (2015). Shoreline Armoring in an Estuary Constrains Wrack-Associated Invertebrate Communities. *Estuaries and Coasts*, 39(1), 171-188. doi:10.1007/s12237-015-9983-x
- Hoppe-Speer, S. C., Adams, J. B., Rajkaran, A., & Bailey, D. (2011). The response of the red mangrove *Rhizophora mucronata* Lam. to salinity and inundation in South Africa. *Aquatic Botany*, 95(2), 71-76. doi:10.1016/j.aquabot.2011.03.006
- Imbert, D., Rousteau, A., & Scherrer, P. (2000). Ecology of mangrove growth and recovery in the Lesser Antilles: State of knowledge and basis for restoration projects. *Restoration Ecology*, 8(3), 230-236. doi:10.1046/j.1526-100x.2000.80034.x
- Jayarathne, K., Rajapaksha, R., Suraweera, P., & Wickramasinghe, W. (2020). Establishing of mangroves in Malala Lagoon, Bundala, Sri Lanka: successes, failures, and future prospects. *Wildlanka*, 8(3), 120-127.
- Jonasson, S. (1983). The point intercept method for non-destructive estimation of biomass. *Phytocoenologia*, 11(3), 385-388. doi:10.1127/phyto/11/1983/385
- Kamali, B., & Hashim, R. (2011). Mangrove restoration without planting. *Ecological Engineering*, 37(2), 387-391. doi:10.1016/j.ecoleng.2010.11.025
- Kangas, P., & Lugo, A. (1990). The distribution of mangroves and saltmarsh in Florida. *Tropical Ecology*, 31, 32-29.
- Kathiresan, K., & Bingham, B. (2001). Biology of mangroves and mangrove ecosystems. *Advances in Marine Biology*, 81-251. doi:10.1016/s0065-2881(01)40003-4

- Kauffman, J. B., & Cole, T. G. (2010). Micronesian mangrove forest structure and tree responses to a severe typhoon. *Wetlands*, 30(6), 1077-1084. doi:10.1007/s13157-010-0114-y
- Kibler, K. M., Donnelly, M., Cannon D., Phagan J., Walters L., McClenachan G., & Roddenberry A. (2020). Developing a Shoreline Restoration Suitability Model for North Indian River and Mosquito Lagoon, Phase II. Indian River Lagoon National Estuaries Program Report IRL2018-017. <https://stars.library.ucf.edu/shorelines/>
- Koch, M. S. (1997). *Rhizophora mangle* L. seedling development into the sapling stage across resource and stress gradients in subtropical Florida. *Biotropica*, 29(4), 427-439. doi:10.1111/j.1744-7429.1997.tb00037.x
- Kodikara, K. A., Mukherjee, N., Jayatissa, L. P., Dahdouh-Guebas, F., & Koedam, N. (2017). Have mangrove restoration projects worked? An in-depth study in Sri Lanka. *Restoration Ecology*, 25(5), 705-716. doi:10.1111/rec.12492
- Krauss, K. W., Doyle, T. W., Twilley, R. R., Rivera-Monroy, V. H., & Sullivan, J. K. (2006). Evaluating the relative contributions of hydroperiod and soil fertility on growth of south Florida mangroves. *Hydrobiologia*, 569(1), 311-324. doi:10.1007/s10750-006-0139-7
- Lara, R. J., & Cohen, M. C. (2006). Sediment porewater salinity, inundation frequency and mangrove vegetation height in Bragança, North Brazil: An ecohydrology-based empirical model. *Wetlands Ecology and Management*, 14(4), 349-358. doi:10.1007/s11273-005-4991-4
- Lewis, R. R. (2005). Ecological engineering for successful management and restoration of mangrove forests. *Ecological Engineering*, 24(4), 403-418. doi:10.1016/j.ecoleng.2004.10.003
- Losada, I. J., Lara, J. L., Christensen, E. D., & Garcia, N. (2005). Modelling of velocity and turbulence fields around and within low-crested rubble-mound breakwaters. *Coastal Engineering*, 52(10-11), 887-913. doi:10.1016/j.coastaleng.2005.09.008
- Lovelock, C. E., Feller, I. C., Reef, R., Hickey, S., & Ball, M. C. (2017). Mangrove dieback during fluctuating sea levels. *Scientific Reports*, 7(1). doi:10.1038/s41598-017-01927-6
- López-Hoffman, L., Ackerly, D. D., Anten, N. P., Denoyer, J. L., & Martinez-Ramos, M. (2007). Gap-dependence in mangrove life-history strategies: A consideration of the entire life cycle and patch dynamics. *Journal of Ecology*, 95(6), 1222-1233. doi:10.1111/j.1365-2745.2007.01298.x
- Nagarajan, B., Pandiarajan, C., Krishnamoorthy, M., & Sophia, P. (2008). Reproductive fitness and success in mangroves: implication on conservation. In *Proceedings of Taal The 12th World Lake Conference* (pp. 29-33).

- McClenachan, G. M., Donnelly, M. J., Shaffer, M. N., Sacks, P. E., & Walters, L. J. (2020). Does size matter? Quantifying the cumulative impact of small-scale living shoreline and oyster reef restoration projects on shoreline erosion. *Restoration Ecology*, 28(6), 1365-1371. doi:10.1111/rec.13235
- McKee, K. L., Mendelssohn, I. A., & Hester, M. W. (1988). Reexamination of pore water sulfide concentrations and redox potentials near the aerial roots of *Rhizophora mangle* and *Avicennia germinans*. *American Journal of Botany*, 75(9), 1352-1359. doi:10.1002/j.1537-2197.1988.tb14196.x
- McKee, K. L. (1993). Soil Physicochemical Patterns and Mangrove Species Distribution-- Reciprocal Effects? *The Journal of Ecology*, 81(3), 477. doi:10.2307/2261526
- McKee, K. L. (1996). Growth and physiological responses of neotropical mangrove seedlings to root zone hypoxia. *Tree Physiology*, 16(11-12), 883-889. doi:10.1093/treephys/16.11-12.883
- McKee, K. L., Rooth, J. E., & Feller, I. C. (2007). Mangrove recruitment after forest disturbance is facilitated by herbaceous species in the Caribbean. *Ecological Applications*, 17(6), 1678-1693. doi:10.1890/06-1614.1
- Millán-Aguilar, O., Manzano-Sarabia, M., Nettel-Hernanz, A., Dodd, R., Hurtado-Oliva, M., & Velázquez-Velázquez, E. (2016). Genetic diversity of the black mangrove *Avicennia germinans* (L.) Stearn in Northwestern Mexico. *Forests*, 7(12), 197. doi:10.3390/f7090197
- Moosavi, S. (2017). Ecological coastal protection: Pathways to living shorelines. *Procedia Engineering*, 196, 930-938. doi:10.1016/j.proeng.2017.08.027
- Morris, R. L., Konlechner, T. M., Ghisalberti, M., & Swearer, S. (2018). From grey to green: Efficacy of eco-engineering solutions for nature-based coastal defense. *Global Change Biology*, 24(5), 1827-1842. doi:10.1111/gcb.14063
- Motamedi, S., Hashim, R., Zakaria, R., Song, K., & Sofawi, B. (2014). Long-Term Assessment of an Innovative Mangrove Rehabilitation Project: Case Study on Carey Island, Malaysia. *The Scientific World Journal*, 2014, 1-12. doi:10.1155/2014/953830
- Myszewski, M., & Alber, M. (2016). Living Shorelines in the Southeast: Research and Data Gaps (pp. 21-23, Rep.). Athens, GA: Georgia Coastal Research Council.
- National Park Service. (2020, July 21). Timucuan ways of life. Retrieved January 6, 2021, from <https://www.nps.gov/timu/learn/timuwaysoflife.htm>
- Nguyen, C. (2013). Processes and factors controlling and affecting the retreat of mangrove Shorelines in South Vietnam (Unpublished master's thesis). Kiel, Christian-Albrechts-Universität, Diss.

- Osland, M. J., Hartmann, A. M., Day, R. H., Ross, M. S., Hall, C. T., Feher, L. C., & Vervaeke, W. C. (2019). Microclimate influences mangrove freeze damage: Implications for range expansion in response to changing macroclimate. *Estuaries and Coasts*, *42*(4), 1084-1096. doi:10.1007/s12237-019-00533-1
- Palinkas, C. M., Sanford, L. P., & Koch, E. W. (2017). Influence of shoreline stabilization structures on the nearshore sedimentary environment in mesohaline Chesapeake Bay. *Estuaries and Coasts*, *41*(4), 952-965. doi:10.1007/s12237-017-0339-6
- Peters, J. R., Yeager, L. A., & Layman, C. A. (2015). Comparison of fish assemblages in restored and natural mangrove habitats along an urban shoreline. *Bulletin of Marine Science*, *91*(2), 125-139. doi:10.5343/bms.2014.1063
- Peterson, C. H., Bishop, M. J., D'Anna, L. M., & Johnson, G. A. (2014). Multi-year persistence of beach habitat degradation from nourishment using coarse shelly sediments. *Science of the Total Environment*, *487*, 481-492. doi:10.1016/j.scitotenv.2014.04.046
- Peterson, N. E., Landry, C. E., Alexander, C. R., Samples, K., & Bledsoe, B. P. (2019). Socioeconomic and environmental predictors of estuarine shoreline hard armoring. *Scientific Reports*, *9*(1). doi:10.1038/s41598-019-52504-y
- Pezeshki, S., Delaune, R., & Meeder, J. (1997). Carbon assimilation and biomass partitioning in *Avicennia germinans* and *Rhizophora mangle* seedlings in response to soil redox conditions. *Environmental and Experimental Botany*, *37*(2-3), 161-171. doi:10.1016/s0098-8472(96)01051-9
- Phan, L. K., Jaap S.m. Van Thiel De Vries, & Stive, M. J. (2015). Coastal Mangrove Squeeze in the Mekong Delta. *Journal of Coastal Research*, *300*, 233-243. doi:10.2112/jcoastres-d-14-00049.1
- Pilato, C. (2019). Hydrodynamic limitations and the effects of living shoreline stabilization on mangrove recruitment along Florida coastlines (Unpublished master's thesis). University of Central Florida.
- Pinzón, Z. S., Ewel, K. C., & Putz, F. E. (2003). Gap formation and forest regeneration in a Micronesian mangrove forest. *Journal of Tropical Ecology*, *19*(2), 143-153. doi:10.1017/s026646740300316x
- Pontee, N. (2013). Defining coastal squeeze: A discussion. *Ocean & Coastal Management*, *84*, 204-207. doi:10.1016/j.ocecoaman.2013.07.010
- Pool, D. J., Snedaker, S. C., & Lugo, A. E. (1977). Structure of mangrove forests in Florida, Puerto Rico, Mexico, and Costa Rica. *Biotropica*, *9*(3), 195. doi:10.2307/2387881

- Primavera, J. H., & Esteban, J. M. (2008). A review of mangrove rehabilitation in the Philippines: Successes, failures and future prospects. *Wetlands Ecology and Management*, 16(5), 345-358. doi:10.1007/s11273-008-9101-y
- Provost, M. W. (1973). Mean high water mark and use of tidelands in Florida. *Florida Scientist*, 36(1), 50-66. <https://www.jstor.org/stable/24319677>
- R Core Team. 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. <https://www.R-project.org/>.
- Rick, T. C., Erlandson, J. M., & Vellanoweth, R. L. (2006). Taphonomy and Site Formation on California's Channel Islands. *Geoarchaeology*, 21(6), 567-589. doi:10.1002/gea.20124
- Rifenberg, J., Litwak, J., & Fillyaw, R. (2021). Vertebrate Impact on a Newly Deployed Shoreline Stabilization Project by Wildlife Camera Analysis. *The Pegasus Review: UCF Undergraduate Research Journal*.
- Riley, R., & Kent, C. (1999). Riley encased methodology: Principles and processes of mangrove habitat creation and restoration. *Mangroves and Salt Marshes*, 2, 207-213.
- Ruiz-Delgado, M. C., Vieira, J. V., Veloso, V. G., Reyes-Martínez, M. J., Sallorenzo, I. A., Borzone, C. A., Sanchez-Moyan, J.E., & García, F. J. (2014). The role of wrack deposits for supralittoral arthropods: An example using Atlantic sandy beaches of Brazil and Spain. *Estuarine, Coastal and Shelf Science*, 136, 61-71. doi:10.1016/j.ecss.2013.11.016
- Rusnak, E. (2016, April 05). The Critical Role Mangroves Play with Regards to Ocean Health. Retrieved January 08, 2021, from <https://sharkresearch.rsmas.miami.edu/the-critical-role-mangroves-play-with-regards-to-ocean-health/>
- Samson, M. S., & Rollon, R. N. (2008). Growth performance of planted mangroves in the Philippines: Revisiting forest management strategies. *AMBIO: A Journal of the Human Environment*, 37(4), 234-240. doi:10.1579/0044-7447(2008)37[234:gpopmi]2.0.co;2
- Schutten, J., Dainty, J., & Davy, A. J. (2005). Root anchorage and its significance for submerged plants in shallow lakes. *Journal of Ecology*, 93(3), 556-571. doi:10.1111/j.1365-2745.2005.00980.x
- Smith, N. P. (1987). An Introduction to the Tides of Florida's Indian River Lagoon. I. Water Levels. *Florida Scientist*, 50(1), 49-61. <https://www.jstor.org/stable/24319868>
- Smith, N. P. (1993). Tidal and Nontidal Flushing of Florida's Indian River Lagoon. *Estuaries*, 16(4), 739. doi:10.2307/1352432

- Smith, T. J., Anderson, G. H., Balentine, K., Tiling, G., Ward, G. A., & Whelan, K. R. (2009). Cumulative impacts of hurricanes on Florida mangrove ecosystems: Sediment deposition, storm surges and vegetation. *Wetlands*, 29(1), 24-34. doi:10.1672/08-40.1
- Smith, R., Blaze, J., & Byers, J. (2020). Negative indirect effects of hurricanes on recruitment of range-expanding mangroves. *Marine Ecology Progress Series*, 644, 65-74. doi:10.3354/meps13351
- Snedaker, S. C. (1995). Mangroves and climate change in the Florida and Caribbean region: Scenarios and hypotheses. *Asia-Pacific Symposium on Mangrove Ecosystems*, 43-49. doi:10.1007/978-94-011-0289-6_6
- Sousa, W. P., Quek, S. P., & Mitchell, B. J. (2003). Regeneration of *Rhizophora mangle* in a Caribbean mangrove forest: Interacting effects of canopy disturbance and a stem-boring beetle. *Oecologia*, 137(3), 436-445. doi:10.1007/s00442-003-1350-0
- Spalding, M., & Parrett, C. L. (2019). Global patterns in mangrove recreation and tourism. *Marine Policy*, 110, 103540. doi:10.1016/j.marpol.2019.103540
- Spiering, D., Kibler, K. M., & Kitsikoudis, V. (2018). Hydrodynamic Change Following Living Shoreline Restoration Based on a Before-After-Control-Impact Experiment. *World Environmental and Water Resources Congress 2018*. doi:10.1061/9780784481424.006
- Tamai, S., & Iampa, P. (1988). Establishment and growth of mangrove seedling in mangrove forests of southern Thailand. *Ecological Research*, 3(3), 227-238. doi:10.1007/bf02348582
- Tamin, N. M., Zakaria, R., Hashim, R., & Yin, Y. (2011). Establishment of *Avicennia marina* mangroves on accreting coastline at Sungai Haji Dorani, Selangor, Malaysia. *Estuarine, Coastal and Shelf Science*, 94(4), 334-342. doi:10.1016/j.ecss.2011.07.009
- Teas, H. J. (1977). Ecology and Restoration of Mangrove Shorelines in Florida. *Environmental Conservation*, 4(1), 51-58. doi:10.1017/s0376892900025042
- Terrados, J., Thampanya, U., Srichai, N., Kheowvongsri, P., Geertz-Hansen, O., Boromthanarath, S., Panapitukkul, N., & Duarte, C. (1997). The Effect of Increased Sediment Accretion on the Survival and Growth of *Rhizophora apiculata* Seedlings. *Estuarine, Coastal and Shelf Science*, 45(5), 697-701. doi:10.1006/ecss.1997.0262
- Teutli-Hernández, C., Herrera-Silveira, J. A., Comín, F. A., & López, M. M. (2019). Nurse species could facilitate the recruitment of mangrove seedlings after hydrological rehabilitation. *Ecological Engineering*, 130, 263-270. doi:10.1016/j.ecoleng.2017.07.030
- Tomlinson, P. (1986). *The Botany of Mangroves*. Cambridge, United Kingdom: Cambridge University Press.

- Valenzuela, P., Arellano, E. C., Burger, J. A., & Becerra, P. (2016). Using facilitation microsites as a restoration tool for conversion of degraded grasslands to *Nothofagus* forests in Southern Patagonia. *Ecological Engineering*, *95*, 580-587. doi:10.1016/j.ecoleng.2016.06.116
- Van Cuong, C., Brown, S., To, H. H., & Hockings, M. (2015). Using *Melaleuca* fences as soft coastal engineering for mangrove restoration in Kien Giang, vietnam. *Ecological Engineering*, *81*, 256-265. doi:10.1016/j.ecoleng.2015.04.031
- Vona, I., Gray, M., & Nardin, W. (2020). The impact of submerged breakwaters on sediment distribution along marsh boundaries. *Water*, *12*(4), 1016. doi:10.3390/w12041016
- Walters, L., Sacks, P., & Campbell, D. (2021). Boating impacts and boat-wake resilient restoration of the eastern oyster *Crassostrea virginica* in Mosquito Lagoon, Florida, USA *Proceedings of Indian River Lagoon Symposium 2020*.
- Webster, P. J., Magaña, V. O., Palmer, T. N., Shukla, J., Tomas, R. A., Yanai, M., & Yasunari, T. (1998). Monsoons: Processes, predictability, and the prospects for prediction. *Journal of Geophysical Research: Oceans*, *103*(C7), 14451-14510. doi:10.1029/97jc02719
- Winterwerp, J. C., Erfteimeijer, P. L., Suryadiputra, N., Eijk, P. V., & Zhang, L. (2013). Defining Eco-Morphodynamic Requirements for Rehabilitating Eroding Mangrove-Mud Coasts. *Wetlands*, *33*(3), 515-526. doi:10.1007/s13157-013-0409-x
- Wolman, M. G. (1954). A method of sampling coarse river-bed material. *Transactions, American Geophysical Union*, *35*(6), 951. doi:10.1029/tr035i006p00951
- Zhang, Y., Ding, Y., Wang, W., Li, Y., & Wang, M. (2019). Distribution of fish among *Avicennia* and *Sonneratia* microhabitats in a tropical mangrove ecosystem in South China. *Ecosphere*, *10*(6). doi:10.1002/ecs2.2759