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NOVA SOUTHEASTERN UNIVERSITY
HALMOS COLLEGE OF NATURAL SCIENCES AND OCEANOGRAPHY

The Relationship between Near Shore Hardbottom Exposure and Benthic Community Composition and Distribution in Palm Beach County, FL

By

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Nova Southeastern University Oceanographic Center
in partial fulfillment of the requirements for
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Abstract

Anthropogenic changes to the landscape, storm events and sea level rise are contributing to the erosion of beaches leading to an increase of the sediment load in near shore marine environments. Palm Beach, Florida is host to unique near shore hardbottom habitats. These areas are distinct from the vast expanses of surrounding sediments and play and important role of habitat and shelter for many different species. In this study, remotely sensed images from 2000-2015 were used to look at the movement of sediment and how it contributes to exposure rates of near shore hardbottom habitats in Palm Beach, Florida and how these factors affect the benthic community.

GIS was used to determine areas of hardbottom with high exposure (exposed in >60% of aerial images), medium exposure (40-60%), and low exposure (<40%). Remotely sensed imagery and manual GIS interpretation were successful in determining hardbottom exposure over time. Large differences in exposed areas were seen in relatively short time periods, and beach nourishments coincided with decreases in exposure.

I strived to determine if one can detect a successional relationship of benthic communities in a dynamic environment with annual mapping. I also examined if areas with higher exposure rates have more complex successive communities than those with lower exposure rates, and what implications this has on near shore benthic communities. *In situ* surveys conducted at 117 sites determined the community structure (corals, octocorals, macroalgae, and hydroids).

This study confirmed that periodic mapping was successful in identifying hardbottom burial and exposure, which fluctuate both spatially and temporally. This periodic mapping along with manual delineation did identify hardbottom burials and exposures that fluctuate between years and relate to benthic community differences. The near shore hardbottom coral reef communities aligned with the observed exposure categories with the greater coral species richness and octocoral morphologies found at sites classified as highly exposed. Statistical analyses showed differences in communities shallower and deeper than three meters’ depth. Increasing the frequency of imagery captures and *in situ* observation would further increase our comprehension of the metrics of hardbottom exposures in reference to community structure.

Keywords: Near shore hardbottom – Palm Beach, FL – Change detection – Benthic habitat mapping – Sediment movement– Spatial analysis – Exposure – Periodic mapping

1. Introduction

1.1 Benthic Community Succession

Ecological succession is the process of change in the species structure and population of a community over time, and is disturbance driven (Connell and Slatyer, 1977). Succession begins after a disturbance takes place that results in bare or semi-bare substrate. The colonizing stage of succession occurs when pioneer species move in to colonize the bare substrate. These pioneer or R-strategy species have high reproductive and growth rates, and typically have a shorter life span. Following this stage, the successional stage of species begins. During the successional stage K-strategy species begin to colonize. These are species with lower reproductive rates, slow growth rates and long life span. They also have a high competitive ability (Littler *et al.* 1983; Weinbauer and Höfle, 1998; CSA International, Inc. 2009). Eventually, without disturbance, a climax community of a diverse variety of slower growing species develop. As time passes following a disturbance species size, population and species richness, would usually increase, thus community structure metrics can give incite to the length of time since the last disturbance or frequency of disturbance a community has experienced (Walker and Alberstadt, 1975).

Many factors can cause succession of communities on benthic substrate. Sediment movement can be a major perturbation on shallow marine benthic communities. Increased sedimentation can significantly impact the health of corals and other sessile organisms. Sedimentation is considered a major cause of coral reef ecosystem degradation worldwide (Nugues and Roberts, 2003). Rogers (1990) noted that both the structure and the function of benthic ecosystems could be negatively affected by physical and biological processes altered by excessive sediment movement.

1.2 Sediment Movement

Sediment movement and sedimentation are natural processes of the erosion and accretion, which operate in dynamic equilibrium (Dean *et al*. 2013). Sedimentation refers to the deposition of sediment grains that were once suspended in the water column. Sediment movement refers to the movement of particles too large to be suspended in the water column, so they move by saltation or by rolling along the bottom. Excessive amounts of sedimentation in one location cause accretion and possibly burial of benthic organisms or entire hardbottom features. Mass sediment movements can also expose previously buried hardbottom below depending on energy regimes.

Increased sediment movement and sedimentation (as well as erosion) can be a direct result of anthropogenic activities. Coastal construction, disruption of natural sediment flows, poor land use patterns, sea level rise, dredging and the removal of mangroves, sea grasses, and marshes have increased sedimentation and erosion on coasts, depleting beaches and producing elevated levels of sedimentation in coastal waters. Elevated coastal erosion has necessitated attempts to restore beaches, shorelines and property through beach nourishment, which involves depositing dredged or hauled in sand onto beaches from other locations. Sediments suspended by dredging are carried by currents, which lead to much larger overall impacted area. The increase in turbidity (sediment suspended in the water column) causes stressful conditions for corals and leads to a reduction in photosynthetic efficiency and potential smothering (Rogers, 1990). Dodge and Vaisnys (1977) point out that even years after the dredging takes place, the deposited sediment can continue to be re-suspended due to the deterioration of the substrate and loss of benthic fauna, leading to potential long-term effects. Consequently, construction projects can have lasting effects increasing stress in organisms which can lead to a decrease in fecundity and increase disease and/or death rates affecting the overall benthic community composition (Erftemeijer *et al*. 2012).

Increased sediment on coral communities can decrease coral abundance, density, productivity and biodiversity (Dodge and Vaisnys, 1977). High levels of sediment reduce fecundity (Gilmour, 1999), survival of coral recruits (Babcock and Smith, 2002), calcification rates (Erftemeijer et al., 2012) and rates of photosynthesis (Fisher *et al*. 2008). Elevated sediment levels can also increase energy needed for sediment removal, alter coral morphology (Jordan et al. 2010), cause smothering (Loya, 1976), change recruit behavior (Babcock and Davies, 1991), affect coral distribution (Hodgson, 1990) and change community structure (Dodge and Vaisnys, 1977; Erftemeijer et al., 2012). Sedimentation on near shore habitats has increased globally with the increase of anthropogenic practices such as dredging, beach nourishment, coastal construction, removal of mangroves, dune grass and sea grass beds. Up to fifty percent of all reefs are considered threatened due to increases in sedimentation resulting directly from anthropogenic activities (Prouty *et al*. 2014). The increase in burials and exposures brought on by mass sediment movement caused by natural and anthropogenic events have widespread impacts throughout the near shore hardbottom ecosystem. The loss of primary productivity, structure, and function affect organisms across all functional groups.

Although near shore hardbottom burial and exposure occurs naturally through seasonal changes and storm events, burial resulting from beach nourishment projects can intensify and prolong the loss of hardbottom habitats (CSA International, Inc. 2009). Beach nourishment is a common practice throughout southeastern Florida due to the loss of sand, stemming from the creation and hardening of inlets and shorelines, coastal erosion brought on by rising sea levels and the changes in the coastal morphology. To restore beaches depleted by erosion or anthropogenic activities, sand is either suctioned or excavated from offshore borrow areas or imported from terrestrial sources (Jordan *et al*. 2010). Sand is then pumped on to beaches in quantities much larger than would naturally occur, temporarily widening the shoreline (Colosio *et al*. 2007). The increased sediment load decreases productivity and function on near shore and beach habitats (Peterson and Bishop, 2005). When beach nourishment projects occur, it increases the cross-shore sediment transport to the offshore, and eventually causes sediment accumulation in the lower part of beach profile. When waves re-shape such profile massive volumes of nourished sand is eroded from the fill which often causes sediment accumulation over nearshore hardbottom (Kosmynim per comm).

1.2.1 Florida's Littoral Processes

Southern Florida beaches are classified as intermediate beaches, where sediment migrates towards the shoreline building up beaches in the summer months during phases of lower wave heights (Benedet *et al*. 2004). During periods of higher wave energy, as those typically seen during the winter months, sediments are transported from the upper beach profile and deposited to the lower part of the beach profile, further offshore (Absalonsen and Dean, 2011). Sediment transport seasonality creates a cycle of cross-shore sediment movement, ensuing periods of erosion and accretion of sediments throughout southeastern Florida (Absalonsen and Dean, 2011). Along with sediment cross-shore movement, longshore drift is also a factor. Longshore drift, which carries sediment along the coast, is generated by waves breaking at an angle to the coastline (Dean *et al*. 2013). Longshore and cross shore currents account for most of the near shore sediment transport, corresponding to annual changes in coastal energy regimes (Stauble, 1993). The dynamics of southeast Florida beaches are unique, with high rates of sediment transport greatly influenced by both geographical characters and by the presence of inlets and other man-made structures. The south Florida coastline is composed of a series of long barrier islands. There are semi-diurnal tides with a mean tidal range of 1 m (Stauble, 1993). South Florida's waters are influenced by the Florida Current which flows north through the corridor between southeast Florida and the Bahamas (Banks *et al*. 2008). The continental shelf is composed of linear reefs and hardbottom ridges which run parallel to shore (Finkl and Andrews, 2008; Banks *et al*. 2007; Walker, 2012).

1.3 Near Shore Hardbottom Characteristics of South Florida

Near shore hardbottom habitats are areas of exposed rock or immobile coarse sediments that facilitate benthic communities. Near shore hardbottom habitats are found in patchy or expansive distributions in southeast Florida. They are unique from the surrounding loose sediment accumulations in shallow marine or intertidal environments at depths less than 6 m (Street *et al*. 2005; CSA, 2009), and are characteristic geologic features prevalent off the shores of south Florida (Van Dolah *et al*. 1987, Walker, 2012). In south Florida, hardbottom habitats generally have low relief, broad flat surfaces, are non-continuous and typically run parallel to shore (Walker *et al*. 2008; Walker *et al*. 2009; CSA, 2009; Walker, 2012). The near shore hardbottom benthic community changes latitudinally southward from Palm Beach County, typically with an increase in the complexity of communities illustrated by the increase of coral species diversity and abundance (Banks *et al*. 2008; CSA, 2009; Klug 2015).

Throughout south Florida, near shore hardbottom is typically classified as an ephemeral habitat, depending on its distance from shore and relief. Ephemeral habitats are disturbance-mediated non-equilibrium systems (FDEP NHB Study; CSA 2009). The shallowest portions of the near shore hardbottom are greatly affected by wave energy, and thus are highly susceptible to sediment movement. Along with the natural progressions of sediment movements with the summer and winter seasons, anthropogenic activities and tropical storm systems often increase the sediment movement leading to mass burial, scouring and exposure events. Frequency of stress from sediment movement is variable, depending on the relief of the hardbottom and its proximity to the beach. Typically, highly stressed environments would have low relief, be geographically close to the beach, and dominated by species adjusted to ephemeral conditions, e.g. turf algae and a few species of macroalgae. Less stressed environments would typically be found farther offshore in areas of higher relief. Habitats experiencing less disturbance from sediment stress are more stable and have longer succession, leading to complex hardbottom communities with perennial macroalgae, higher numbers of coral species and higher diversity of the benthos.

Near shore hardbottom provides an important ecological role in the south Florida marine ecosystem, acting as habitat, settlement sites, nesting and spawning sites, nursery areas, and feeding sites and shelter across many functional groups (CSA, 2009). These areas serve as substrate for many benthic species of algae, sponges, stony corals and octocorals (Moyer *et al*. 2003; CSA, 2009; Walker, 2012; Walker and Gilliam 2013; Klug 2015). Octocorals, hydroids and macroalgae are some of the most abundant organisms on south Florida’s hardbottom habitats (Gilliam *et al*. 2013; Klug 2015). Although less abundant, corals and sponges are important components of near shore hardbottom communities. The living organisms and their skeletons create habitat complexity that attracts many important fish and invertebrate species, increasing the biodiversity of the ecosystem (Van Dolah *et al*. 1987).

CSA (2009) reports that the near shore hardbottom throughout southeast Florida serves as habitat to an estimated 520 invertebrate species, 300 algal species and 250 fish species, with a large population of juveniles. Hardbottom is listed as Essential Fish Habitat, and as a habitat Area of Particular Concern (SAFMC, 1998). The hardbottom also serves as an important habitat for juvenile green turtles, which are currently classified as threatened under the Endangered Species Act (U.S. Fish and Wildlife Service, 1999, 2016; Holloway-Adkins, 2005; CSA, 2009). Hardbottom in Broward and Miami-Dade counties also host some of the oldest known *Orbicella faveolata* colonies (Walker and Klug, 2015) and dense *Acropora cervicornis* patches (Vargas-Ángel *et al.* 2003; Klug, 2015) which are listed as threatened under the Endangered Species Act.

1.4 Sedimentation Impacts on Scleractinian Corals and Octocorals

Stony corals are important primary producers on near shore hardbottom habitats and play a key role in providing structural complexity. The corals structure provides protection and cover for fish and many invertebrates, and serves as a site for fish spawning activities or as juvenile nurseries. They also work to disperse wave energy, protecting the coastline and reducing erosion. An increase in the sediment load can have a number of negative impacts on coral species. Suspended particles reduce light penetration and increase scattering, reducing photosynthetic potential in corals (Dodge and Visanys, 1977; Fisher *et al*. 2008; Erftemeijer *et al*. 2012). Sediment deposited on corals can lead to smothering or burial, reducing productivity and decreasing respiration (Fisher *et al*. 2008). Elevated turbidity (suspended sediments in the water column) under high-energy conditions can abrade coral tissue and erode coral heads, leading to a reduction in reef rugosity (Nugues and Roberts, 2003). Reduced reef rugosity decreases available habitat space affecting reproduction and survival rates. Though corals are able to remove sediment through cilia movement, polyp swelling and mucus production, the physical removal of sediment is energetically taxing and can reduce fecundity, growth rate, and increases coral stress (Erftemeijer *et al*. 2012). However, impacts from increased sedimentation can vary depending on local oceanographic conditions, relief of the benthic habitat, the area and duration of the impact and the presence of other known stressors (Díaz-Ortega and Hernández-Delgado, 2014).

Sedimentation on near shore hardbottom habitats strongly affects survival of coral recruits, thus affecting populations and biodiversity (Babcock and Davies, 1991). Sediment that settles on near shore hardbottom or substrate greatly reduces the success of larval recruitment due to the lack of suitable areas for anchorage (Hodgson, 1990). If coral recruits do manage to settle, they are susceptible to burial by sediment, resulting in post-settlement mortality. Babcock and Davies (1991) found that sedimentation resulted in significant changes in *Acropora millepora* settlement patterns including fewer recruits relative to a control environment and an increase in settlement to undesirable vertical substrates and undersurfaces. Babcock and Smith (2002) stated that fewer coral recruits combined with decreased post-settlement survival rates would significantly affect reef population structure and diversity.

Members of the subclass Octocorallia (octocorals) are responsible for most of the living structural complexity found on the near shore hardbottom habitats in southeast Florida, serving as invaluable habitat for many organisms (Klug 2015). Octocorals also function as a substratum for benthic invertebrates and algae. Therefore, impacts from sedimentation and burial can play a role in reducing biodiversity and the abundance of octocorals (Yoshioka and Yoshioka, 1989). Like stony corals, octocorals recruits can also be negatively affected by the burial of hardbottom due to the loss of suitable settlement sites. However, octocorals are thought to be some of the most sediment tolerant species in Florida (Erftemeijier *et al*. 2012). Rogers (1990) suggested that their morphology was more resistant to the accumulation of sediments, which increased their tolerance to heavier levels of sedimentation when compared to that of stony corals. However, burial of the holdfast ceases holdfast growth while the rest of the octocoral continues to develop, sometimes resulting in the eventual death of the octocorals.

1.5 Sediment Impacts on Macroalgae and Sponges

Macroalgae is found throughout the near shore hardbottom habitats of south Florida, where the hardbottom provides suitable substrate for attachment and growth.

The algae play a number of important roles on hardbottom habitats. They are responsible for a large amount of primary productivity and contribute to complex trophic interactions (Duarte, 2000; CSA, 2009). Some macroalgae species are also important nitrogen fixers on near shore hardbottom communities (Blair and Flynn, 1989). Macroalgae also contribute to benthic structural complexity and serve as shelter and/or food for many organisms including endangered and threatened species of sea turtles (Blair and Flynn, 1989; CSA, 2009). Sediment movement affects benthic algae's ability to survive much in the same way that it affects coral recruits. Burial, increased turbidity and lack of suitable substrate for settlement greatly increases algal mortality and prevents the growth of new algae.

Crustose coralline algae (CCA) is also an important contributor to near shore coral reef ecosystems. These algae deposit calcium carbonate, cementing and reinforcing reef structure e.g. by filling in cracks in the substrate (Fabricius and De'Ath, 2001). Some of these algae act as a substrate on which some corals depend for locating appropriate settlement sites (Harrington *et al*. 2004). CCA also serves as substrata for many other species of benthic invertebrates (Harrington *et al*. 2005). Coverage of CCA is inversely related to sediment levels (Fabricius and De'Ath, 2001). Hardbottom inundated with sediments will prevent the CCA from settling, thus reducing coral recruitment and consequently the biodiversity and composition of the reef community (Fabricius and De'Ath, 2001).

Along with stony and soft corals, sponges also increase the structural complexity of the hardbottom habitat and provide shelter for a number of organisms spanning many trophic levels. Some species such as brittle stars even depend on sponges as necessary habitat. Along with their importance adding to the structural complexity, sponges serve as a food source for some fish and sea turtle species and may play a role in removing nutrients from the water column. Like the other structural counterparts of the near shore hardbottom communities, sediment burial and mass movements of sand also negatively affect sponges; however, infrequent fragmentation events resulting from sand scour could aid in sponge distribution (CSA International, Inc. 2009).

1.6 Sediment Impacts on Motile Species

In south Florida, green (*Chelonia mydas*), loggerhead (*Caretta caretta*) and hawksbill (*Eretmochelys impricata)* turtles use near shore hardbottom habitats as crucial resting sites, foraging grounds, as shelter and as developmental habitats for juveniles (Makowski, 2006, Garrido, 2007). Green turtles, which are currently listed as endangered are highly associated with near shore hardbottom habitats (Baillie, 2004; Makowski, 2006; CSA, 2009). Previous studies suggest that green and hawksbill turtles use the near shore hardbottom as developmental habitat between 2-5 years (CSA International, Inc. 2009). In Palm Beach County, juvenile green turtles move out to the open ocean for their first years of life, and then recruit to near shore hardbottom habitats until they reach sexual maturity (Makowski, 2006). It has also been observed that green sea turtles have specific ranges consisting of feeding grounds and resting places. When sediment movement and accretion reduces the complexity of the hardbottom relief, sea turtles' home ranges, resting places and food sources (macroalgae, sponges, crustaceans) can be negatively affected (Makowski, 2006; Garrido, 2007; CSA, 2009).

Many fish depend on near shore hardbottom habitats for refuge, spawning sites, juvenile nurseries, and feeding grounds. Therefore, they can be negatively impacted by increased sedimentation rates (Street *et al*. 2005). Fish populations on near shore hardbottom are juvenile-dominated, making the habitat important nursery grounds for many species. Studies performed in Palm Beach found that over 80% of fish sampled on near shore hardbottom sites were juveniles (Lindeman and Snyder, 1999; Fisco, 2016). Reduction in the relief complexity of the near shore hardbottom habitat as a result of increased sediment loads decreases the available space for juvenile fish to shelter and settle and has wide spread impacts on fish survival rates and successful spawning. These habitats are unique and act as oases for the fish because they are surrounded by vast areas of sediment.

Reductions in growth rates, reproduction, and photosynthetic ability of benthos because of sedimentation and burial can cause ecosystem-wide impacts on hardbottom communities, affecting not only coral and algal species, but many coral–associated and –dependent organisms. Reductions in overall biodiversity and population sizes on hardbottom habitats affect organisms across all trophic levels (Nugues and Roberts, 2003).

1.7 Change detection through time using visual interpretations and GIS

Change detection is the analysis of remotely sensed imagery on a temporal scale (Costa *et al*. in press). Temporal change detection is accomplished using a number of different methodologies, typically utilizing aerial or satellite imagery to visually or through automated processes, interpret change through time depending. This process depends on the quality of imagery required, spatial extent of the project, budget and the resolution of the desired output (Aronoff, 2005; Costa *et al*. in press).

Aerial photography is an effective way to detect spatial and temporal change in coral reefs and other benthic habitats (Goodman *et al*. 2013). Imagery acquired aerially typically produces excellent spatial resolution and high thematic accuracy with little interference from noise such as cloud cover or sun angle. However, large spatial extents are expensive and difficult to collect (Mumby *et al*. 1998; Goodman *et al*. 2013; Costa *et al*. in press).

Visual interpretation (manual digitization with GIS) is an effective method to detect change on a temporal scale, or map the extent of benthic habitats (Goodman *et al*. 2013). Visual interpretation is useful in areas with smaller spatial extents, and those that need precise delineations. The finer detail mapped using visual interpretation allows for changes occurring overtime to be better represented (Goodman *et al*. 2013). However, visual interpretations are difficult to replicate, rely on the knowledge and skill of the interpreter and depending on the size of the team of interpreters needed, may be less efficient than digital interpretations (Coppin and Bauer, 1996; Costa *et al*. in press).

Digital interpretation approaches are potentially more efficient for mapping large scale (whole reef systems) when compared to that of visual interpretation (Maeder *et al*. 2002; Mishra *et al*. 2006; Costa *et al*. in press). Large projects with varying spatial scales would potentially find digital interpretations more effective (Costa *et al*. in press). However, digital interpretations are prone to misclassification of areas where reflectance reads as conditions that are not present (Coyne *et al*. 2003; Costa *et al*. 2013). Digital interpretations also are susceptible to error when imagery is poor quality or has excess noise (Costa *et al*. in press).

The movement of sediment in Palm Beach, Florida can be seen in remotely sensed imagery over the past 14 years. The imagery has been used by several entities to delineate exposed hardbottom throughout the extent of Palm Beach. These images provide an opportunity to observe how the exposed hardbottom footprint has changed over the years. Presumably, the change in outline of hardbottom exposure from year-to-year reflects changes in sediment burials, based on the assumption that hardbottom not seen in the imagery is likely buried. Thus, the differences in hardbottom footprints provide some indication of relative hardbottom exposure and burial rates through time.

1.8 Study Site

The Florida Reef Tract (FRT) spans approximately 595 km from the Dry Tortugas in the southwest to Martin County in the northeast (Walker 2013). The southern 135 km portion is oriented east west, then it arcs northeast over a 245 km span. The final 215 km extends north through Martin County. The northern part is comprised of three main reefs and extensive near shore hardbottom (Walker, 2012). The northern region has been subdivided based on benthic habitat morphology (Walker, 2012; Walker and Gilliam 2013) and corroborated with benthic (Klug, 2015) and fish results (Fisco, 2016). The south Palm Beach region, where this study takes place, reef habitats are mainly the outer reef and deep ridges (Walker, 2012); however, some near shore hardbottom parallel to shore exists as well.

The near shore hardbottom of Palm Beach County is represented by limestones of the Anstasia Formation, which are dated by the late Pleistocene, and is a part of the Anastasia Formation (Stauble, 1993). Just south of Lake Worth Inlet, in northern Palm Beach County, is the site of the northern terminus of coral reef growth in Holocene (Banks *et al*. 2007; Finkl and Andrews 2008; Walker, 2012; Walker and Gilliam, 2013). Palm Beach is subjected to a large volume of sediment flowing from the sediment rich environments of the north in comparison to the rest of southeast Florida (Banks *et al*. 2007). The Lake Worth Inlet has been deepened significantly, which typically has a great effect on sediment transport processes, leading to high levels of erosion causing concern for coastal management (Dean *et al*. 2013). In 1996, “The United States Army Core of Engineers(USACE) estimated that 98,000 m^3/year of sediment reaches Lake Worth Inlet, in contrast to 4,590 m^3/year reaching Government Cut just north of Biscayne Bay in Miami-Dade County” (Banks *et al*. 2007). Palm Beach County sites near Lake Worth Inlet are important to study, because historically there has been a high abundance of benthic organisms present on the near shore hardbottom that contribute to the productivity in the ecosystem (Blair and Flynn, 1989), and they have been nourished many times due to high erosion rates. It is also some of the northern-most near shore hardbottom with tropical reef communities on the Florida reef Tract (Walker, 2012).

1.9 Purpose of Study

Palm Beach Florida’s near shore hardbottom habitats play an important ecological role in the south Florida marine ecosystem and serve as settlement sites and juvenile habitat for many ecologically beneficial species of fish, turtles, algae, sponges, and corals. This study aimed to achieve the following:

1) Describe the near shore hardbottom benthic communities.

2) Evaluate the current near shore hardbottom designations (near shore, intermediate, and offshore).

3) Elucidate how sediment movement is affecting the benthic communities on these habitats across depth and latitude.

4) Determine if a successional relationship can be detected a dynamic environment with periodic mapping.

5) Serve as baseline data for future studies in the area, including monitoring projects conducted as a part of the better management practices of beach nourishment projects in south Florida.

A better understanding of the near shore benthic community and sediment movement, especially with the influx of external sediment sources from beach nourishment, can help inform us of community succession and help advise near shore habitat management and conservation practices.

1.9.1 Objectives

The main objective in this study was to understand if we could detect a successional relationship of benthic communities in a dynamic environment with annual mapping. We wanted to find how the frequency of sediment burial affects near shore hardbottom benthic communities in Palm Beach County, Florida. Benthic community structure (measured by diversity and size of corals and octocorals; e.g. larger corals are older and therefore are a part of a more complex community) was surveyed in hardbottom areas of differing burial and exposure rates and depth. More established communities were expected in areas with higher rates of exposure (less burial) and in deeper water because the longer exposure allows more time for organisms to settle and grow, when compared to sites with more frequent burial. More established communities were expected in deeper water presumably due to the lessening of wave energy with depth.

2. Methods

This study was conducted along 15.7 miles of Palm Beach County Florida coastline between north Lake Worth Inlet and south Lake Worth (Boynton Beach) Inlet (Figure 1).

Figure 1. The study area, Palm Beach County, Florida. The extent of the area begins just south of Lake Worth Inlet at reference monument 76, and ends north of the Boynton Beach inlet at reference monument 137+400 (Reaches 1-9 in accordance to the BMA management plan (appendix I).

A comprehensive dataset from previous work at the local, state, and federal level including all the Town of Palm Beach and Palm Beach County aerial photographs and hardbottom delineations (2000-2012) was assembled in ArcGIS to support seafloor feature identification. This data was used as a reference to help guide polygon delineations, classification, and exposure.

2.1 Benthic Habitat Mapping

Aerial photographs were collected within the Town of Palm Beach limits (R-76 south to R-137, in accordance with the BMA management plan (appendix I) on July 3, 2014, November 11 and 14, 2014 and May 20, 2015 and provided by the Town of Palm Beach and Palm Beach County.

Imagery were imported into a geodatabase as a mosaic dataset in ArcGIS and visually interpreted where color variations and textural disparities visible at a 1:500 scale indicated exposed hardbottom. Temporary histogram gamma stretches of 2 and 2.5 standard deviations were used for optimal visualization. Polygons were drawn at a minimum mapping unit of 0.75 m^2 (8 ft^2) for each set. All polygons were then checked against known artificial structure areas. Polygons that crossed previously designated artificial habitat were clipped and categorized as artificial over hardbottom, artificial or hardbottom.

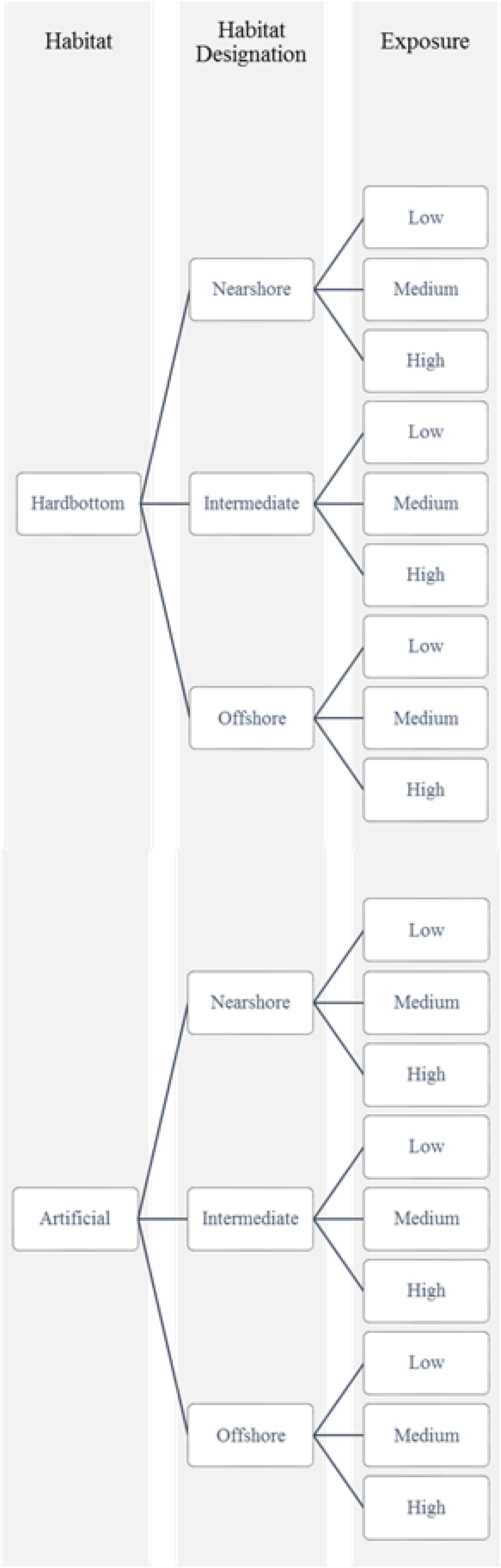
All polygons (Hardbottom, Artificial and Artificial over Hardbottom) were classified by distance from shore/depth as Near shore, Intermediate, and Offshore (Figure 2) using a previously derived polygon layer supplied by the Florida Department of Environmental Protection. The three zones (Figure 3) were based on Town of Palm Beach profile data collected by Sea Diversified Inc. on August 9, 2010 and are referenced to North American Vertical Datum (NAVD) 88, North American Datum (NAD) 83/90. Any near shore hardbottom habitats that occurred within each classification were categorized accordingly.

***Near shore**: categorized as - ~mean high water line to the -13.1 ft (~-4 m). North American Vertical Datum (NAVD) 88 depth contour*

***Intermediate**: between -13.1 ft. and -26.2 ft. (~-8 m) NAVD 88 depth contour*

***Offshore**: between -26.2 ft. and -40.0 ft. (~-12 m) NAVD 88 depth contour (as defined by FDEP in the BMA).*

The exposure categorization was accomplished by evaluation of frequency of exposure in previous mapping efforts. The number of times each area was mapped in the previous imagery was assumed to relate to the exposure frequency; therefore, the polygons of all previous exposed hardbottom mapping efforts in the same area of interest were compiled and unioned together into a file with all previous delineations. Reaches were mapped 15 times between August 2000 and March 2012. The polygons were merged into 3 classes with frequency values of: <6 (i.e. exposed in <6 years of imagery) as Low, 6-10 as Medium, and >10 as high exposure. The bins for the 15 mapped exposure areas equated to the percentages of exposure: <40% = Low, 40%-60% = Medium, and >60% = High exposure (Figure 5).



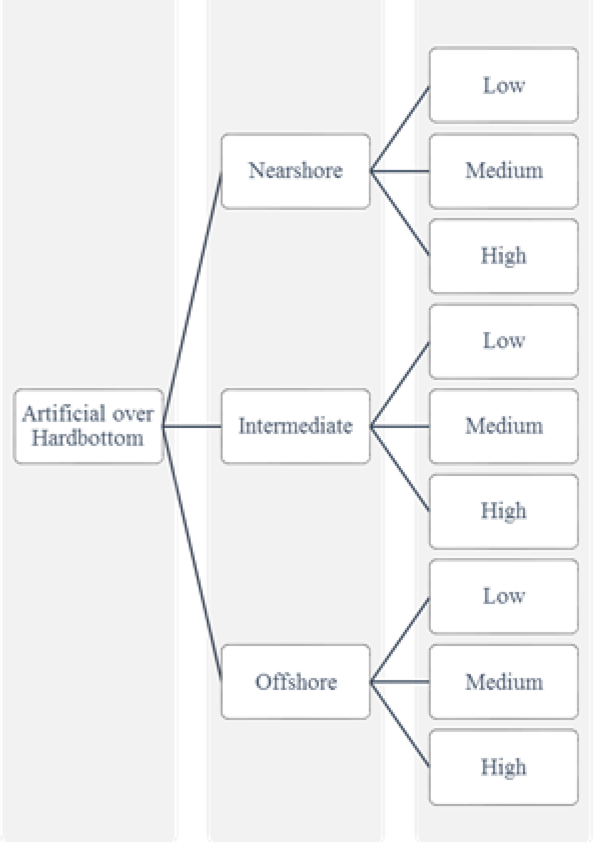


Figure 2. Classification scheme, Hardbottom, Artificial and Artificial over Hardbottom followed by near shore, intermediate and offshore than categorized by low, medium and high exposure**.**

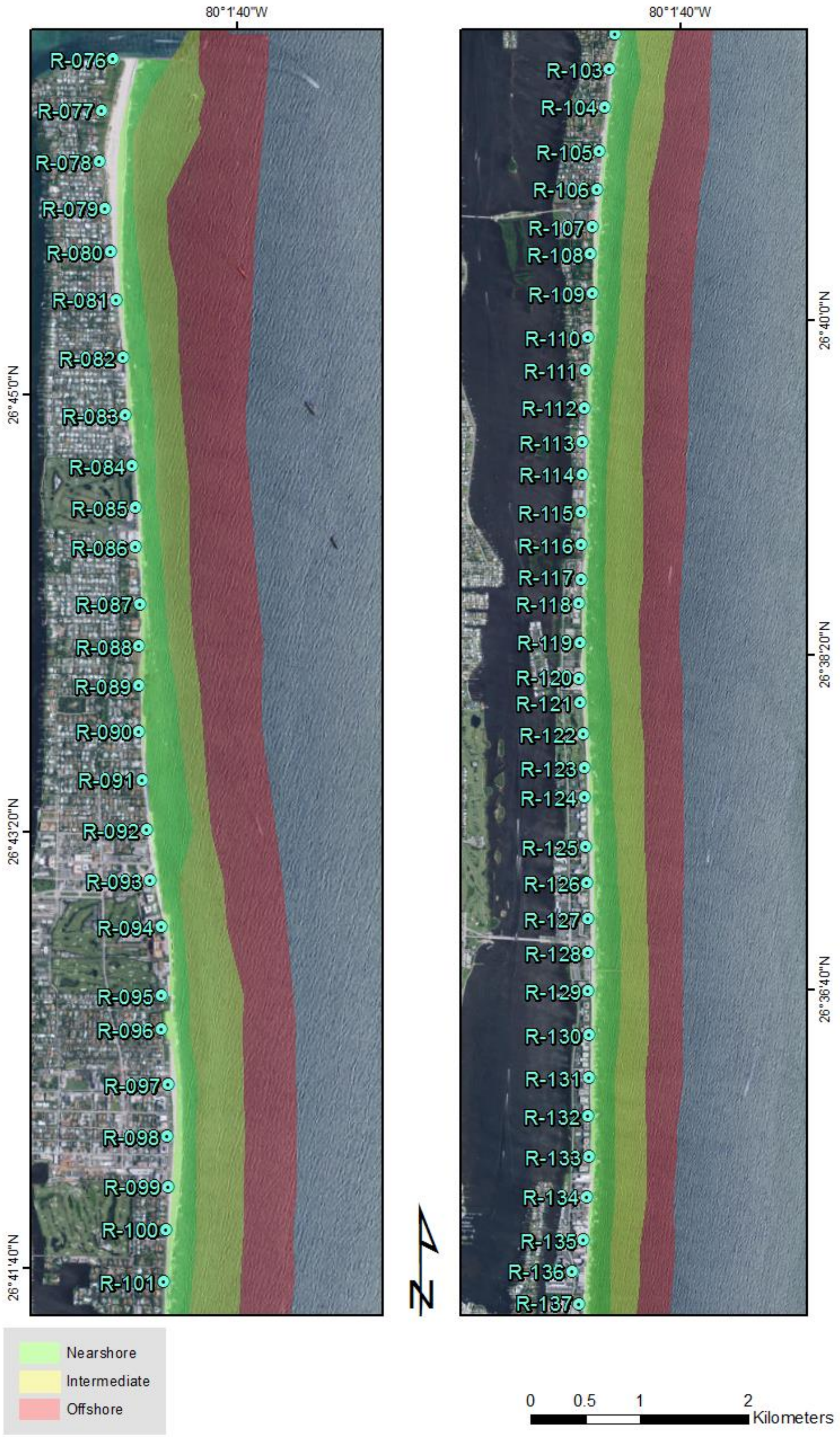


Figure 3. Near shore, Intermediate and Offshore hardbottom designations (based on shapefiles provided by FDEP based on 2010 Town of Palm Beach profile data).

Allocations of sites to be ground truthed were based on the proportional area of the combination of exposure frequency (Figure 4) and a previously assigned hardbottom designation (Figure 3) which equated to nine classes (near shore, intermediate, offshore and low, medium and high exposure rates) in the final layer (Figure 5). Six sites per class (e.g. near shore intermediate exposure) were chosen at random, and then the remaining sites were split proportionally between classes (Figure 6). The site locations were spatially reviewed to ensure that each was assigned to the correct category and that each transect would be logistically feasible (enough area to contain transect, far enough from other sites) to survey. Directional heading limitations were noted on sites that were closer than 20 m from each other or any habitat boundary in order to avoid surveys overlapping or crossing habitats.

2.2 Field surveys

Surveys were conducted to assess the community structure of the near shore hardbottom in relation to their mapped exposures and distance from shore/depth. To establish the study sites, a combination of the last four years (2010-2014) of aerial imagery and exposed hardbottom delineations were used. The hardbottom polygons from July 2010 – March 2012 (July/2010, October/2010, May/2011, October/2011 and March/2012) were unioned into one layer and dissolved to form a single polygon displaying the footprint of all exposed hardbottom since 2010. This was used to clip the total exposed hardbottom file to remove any areas not mapped (exposed) in the past four years.

A total of 117 ground truthing sites (Figure 6) were randomly selected stratified by exposure frequency (Figure 4) and distance/depth (near shore, intermediate and offshore) regions (Figure 3). This equated to nine classes (near shore, intermediate, offshore and low, medium and high exposure rates) in the final layer. Allocations of sites were based on the proportional area of the combination of exposure frequency (Figure 4) and a previously assigned hardbottom designation (Figure 3) with a minimum of six sites per stratum. The site locations were reviewed to ensure that each was assigned to the correct category and that each transect would be logistically feasible to survey. Transect headings were defined for those sites that were closer than 20 m from each other or any habitat boundary in order to avoid surveys overlapping or crossing habitats. If any part of the transect was covered with sand but over hardbottom the area was still included in the analysis. However, if no buried hardbottom was found, sites were excluded.

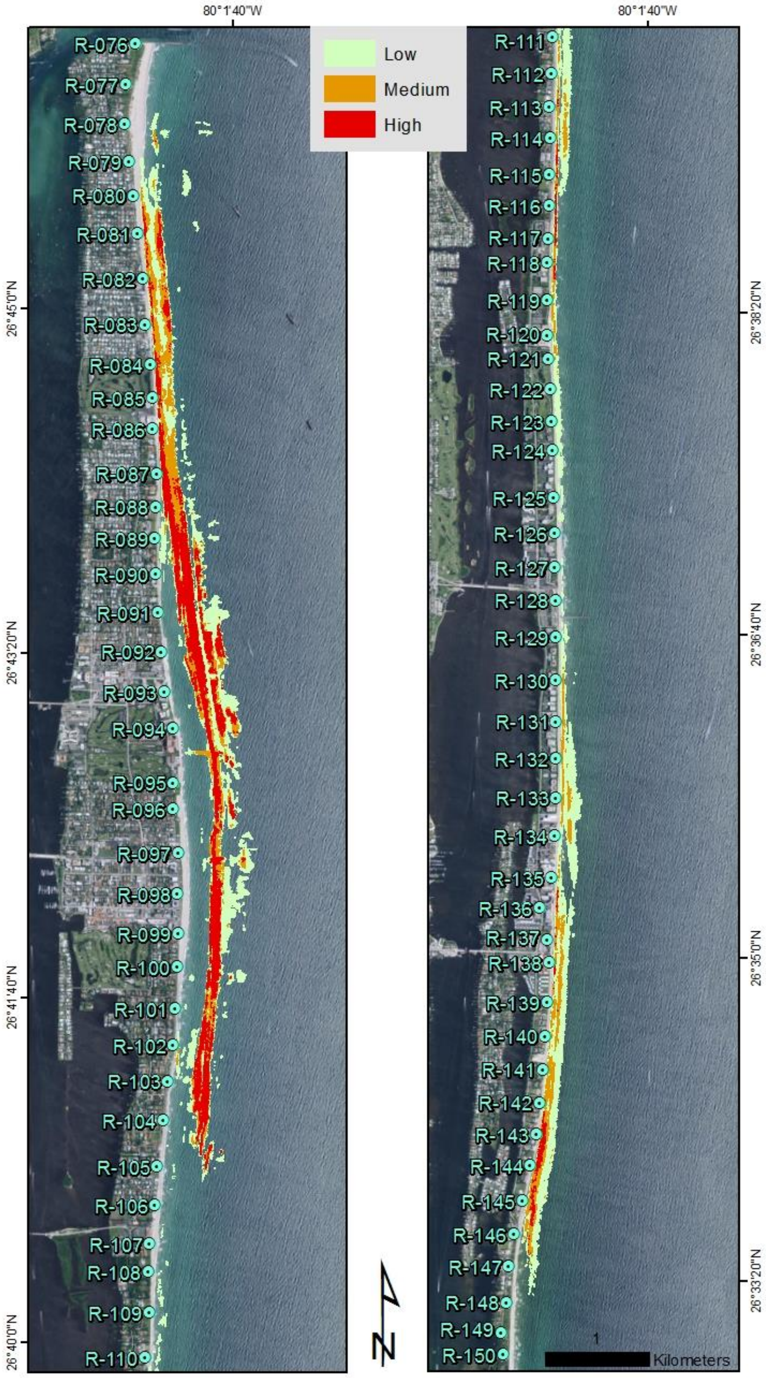


Figure 4. Exposure Frequency 2000-2014

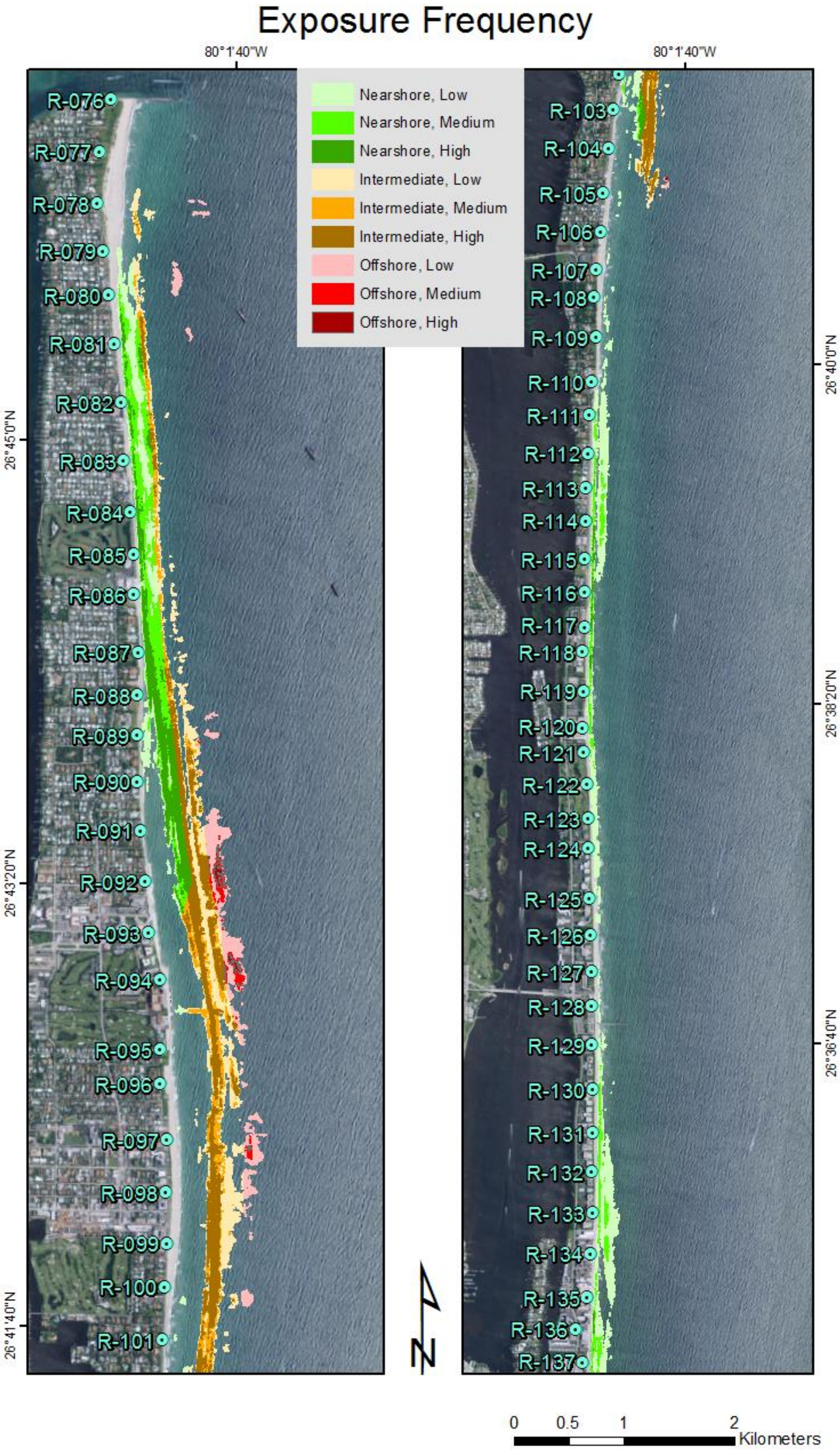


Figure 5. Exposure frequency and hardbottom designation habitat map.

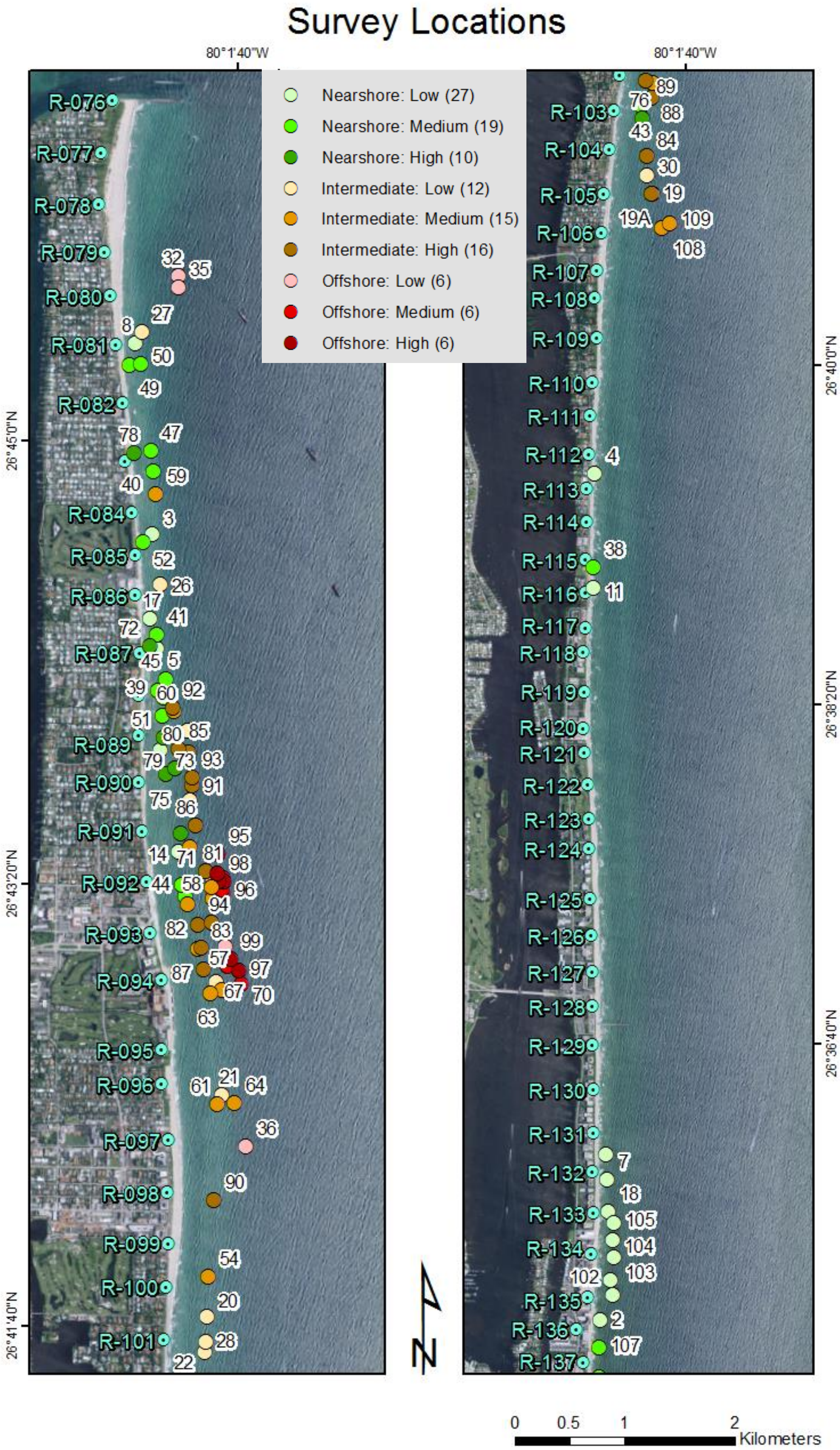


Figure 6. In situ survey locations based on the frequency exposures of hardbottom and spatial designations.

A handheld Global Positioning System (GPS) (Garmin GPSMAP 76CSx) or survey-grade Trimble Differential GPS was used to locate each randomized survey site (Figure 6). A dive flag was deployed to mark the location of each site. Each survey site consisted of a transect used to determine stony coral and octocoral density, and a belt quadrat transect to quantify percent cover of algae, hydroids, and sediment, and maximum sediment depth. The following survey methods were conducted at each site:

- ***General area assessment***

  The general area was assessed and categorized as: exposed hardbottom, partially exposed hardbottom, buried hardbottom, mostly sediment, all sediment, or established benthic communities. The presence of high relief ledges (> 1 m in height) within 5 m of transect were also noted.

- ***One 20 x 0.5 meter transect* (10 m^2)**

  Stony coral colonies were identified to species, and categorized by size class (diameter) (2-5 cm, >5-10 cm, 10-25 cm and >25 cm). Octocorals were identified by morphology (rod, plume, fan, or encrusting) and categorized by size classes (height) (<5 cm, 5-10 cm, 10-25 cm, >25 cm).

- ***Five 0.5 m^2 quadrats***

  The quadrats were set at intervals of 5 m (0, 5, 10, 15, 20 m) along each 20 m transect. Percent cover of algae, hydroids, sponges, sediment, and exposed bare substrate and crustose coralline algae were calculated, and the two most dominate algae species were identified and cover was estimated. Maximum sediment depth was measured within each quadrat location.

2.3 Statistical Analysis Methods

A cluster analysis and corresponding non-metric, multi-dimensional scaling (MDS) plot was constructed using Bray-Curtis similarity indices (PRIMER v6) of the percent benthic cover quadrat data and hard and soft coral density (square-root transformed) to evaluate similarities between sites. The MDS plot shows statistical similarities and differences in multivariate data by plotting them in two dimensions where the relative distance apart is indicative of their similarity. Thus, sites very close together are more similar than those further apart and the sites furthest apart are the least similar. The sites were analyzed by several factors in PRIMER (e.g. hardbottom designation, exposure frequency, depth) to evaluate how well these factors relate to the similarities in the community data. A cluster analysis was performed on each dataset to determine similarities. A MDS plot was configured to illustrate the analyses' results by factors. An Analysis of Similarity (ANOSIM) was performed for different factors to determine significance. The R statistic indicates the strength of the relationship where the closer the value is to 1, the stronger the dissimilarity between groups. Then Similarity Percentages by factor (SIMPER) were calculated to determine which species were driving the similarities identified in the ANOSIM. One way non-parametric analysis of variance (ANOVA) tests were used to find significant univariate differences. A post hoc Wilcoxon Each Pair test was used in JMP (v 10.0) to determine which habitats significantly differed.

3. Results

3.1 Benthic Habitat Mapping

The total mapped area for July 2014 was 212.15 acres (0.86 km^2; 85.85 hectares) (Figure 7), for November 2014 was 212.64 acres (0.86 km^2; 86.05 hectares) (Figure 8), and for May 2015 was 188.81 acres (0.76 km2; 76.41 hectares) (Figure 9). Although July and November 2014 have very similar total acreages, many reaches had very different acreages. For example, Reach 7 has an exposed area of 10.13 acres in July 2014, than four months later in November 2014 there are only 5.47 acres exposed (Table 1).

The acreage was broken down into the nine reaches on Palm Beach Island in accordance with the BMA management plan (appendix I) to compare across all years (Table 1). Reach 1 (just south of Lake Worth Inlet to slightly past reference monument 98) was the smallest area (0.87 km of coastline) and had the lowest amount of exposed hardbottom ($\bar{x}$ = 0.07 acres). Reach 2 was the largest area (3.99 kilometers of coastline) and had the highest acreage of exposed near shore hardbottom ($\bar{x}$ = 62.11 acres) (Table 2). Reach 7 was the second largest area (3.76 km of coastline), but averaged only 4.58 acres of exposed hardbottom (Table 2.) The area of exposed hardbottom fluctuated greatly in each reach. Reaches 2, 3 and 4 had the greatest variance, 202, 105 and 100 respectively. These were also the three reaches with the greatest overall area of exposed hardbottom averaging 62 acres, 52 acres and 41 acres. Although reach 6 only averaged 7 acres, it had a relatively large variance of 45 indicating large fluctuations of exposed hardbottom (Figure 10).

Average exposed acreage for all reaches was 200.84 (Table 2). May 2015 had the second lowest area of exposed hardbottom since 2010. The lowest recorded hardbottom exposed was July 2003 (163.76 acres), while the highest was October 2008 (244.43 acres) (difference 80.67 ac). Exposed hardbottom fluctuated throughout the years from a minimum of 0.33 acres (July 2010 – October 2010) to a difference of 56.21 acres (May – October 2011) (Table 1, Figure 11).

The area of exposed hardbottom fluctuated greatly in each reach. Reaches 2, 3 and 4 had the greatest variance, 201.56, 104.84 and 100.29 respectively. These were also the three reaches with the greatest overall area of exposed hardbottom averaging 62.11 acres, 52.56 acres and 41.06 acres (Table 2). Although reach 6 only averaged 6.52 acres, it had a relatively large variance of 44.86 indicating large fluctuations of exposed hardbottom.

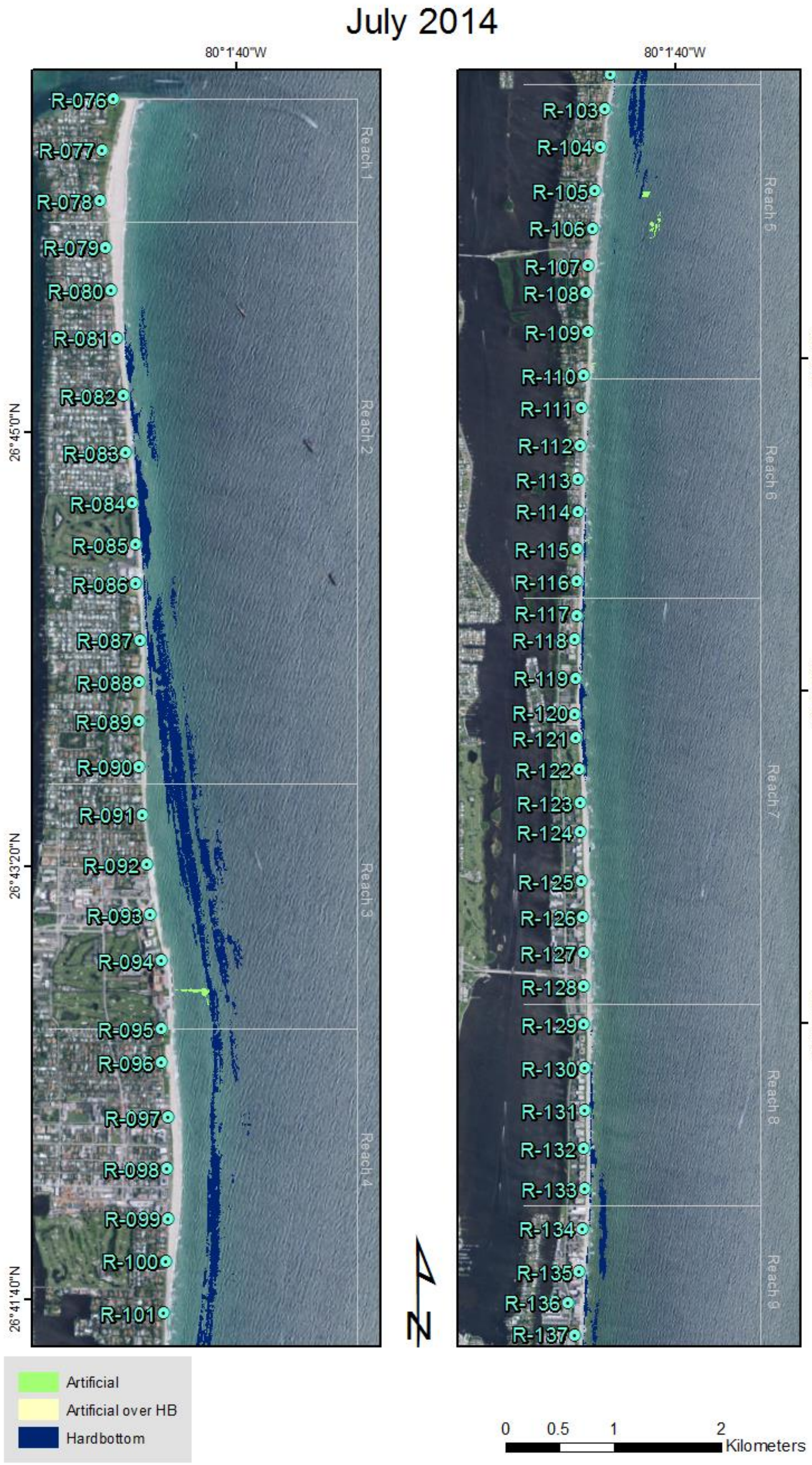


Figure 7. July 2014 hardbottom polygon delineation

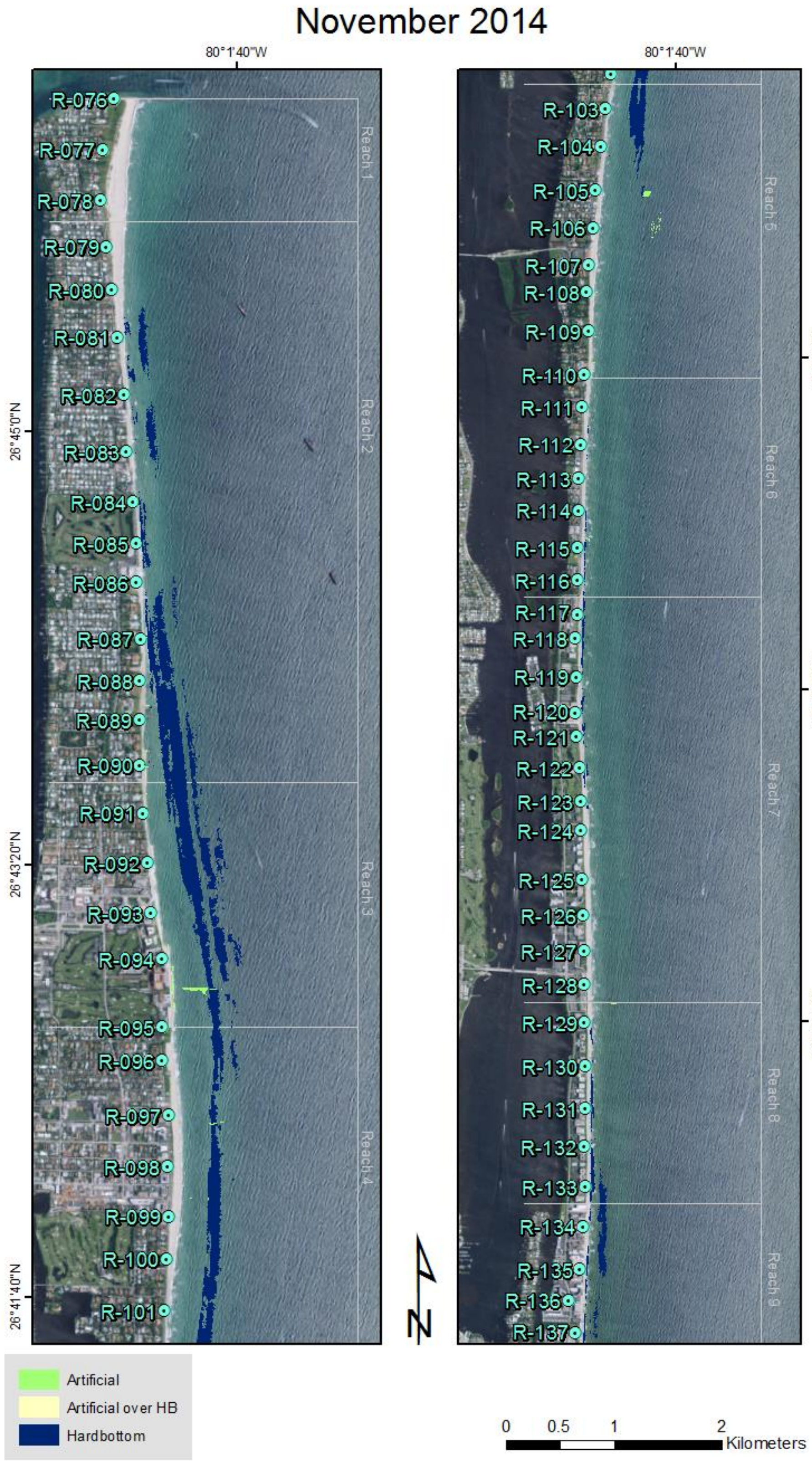


Figure 8. November 2014 hardbottom polygon delineation

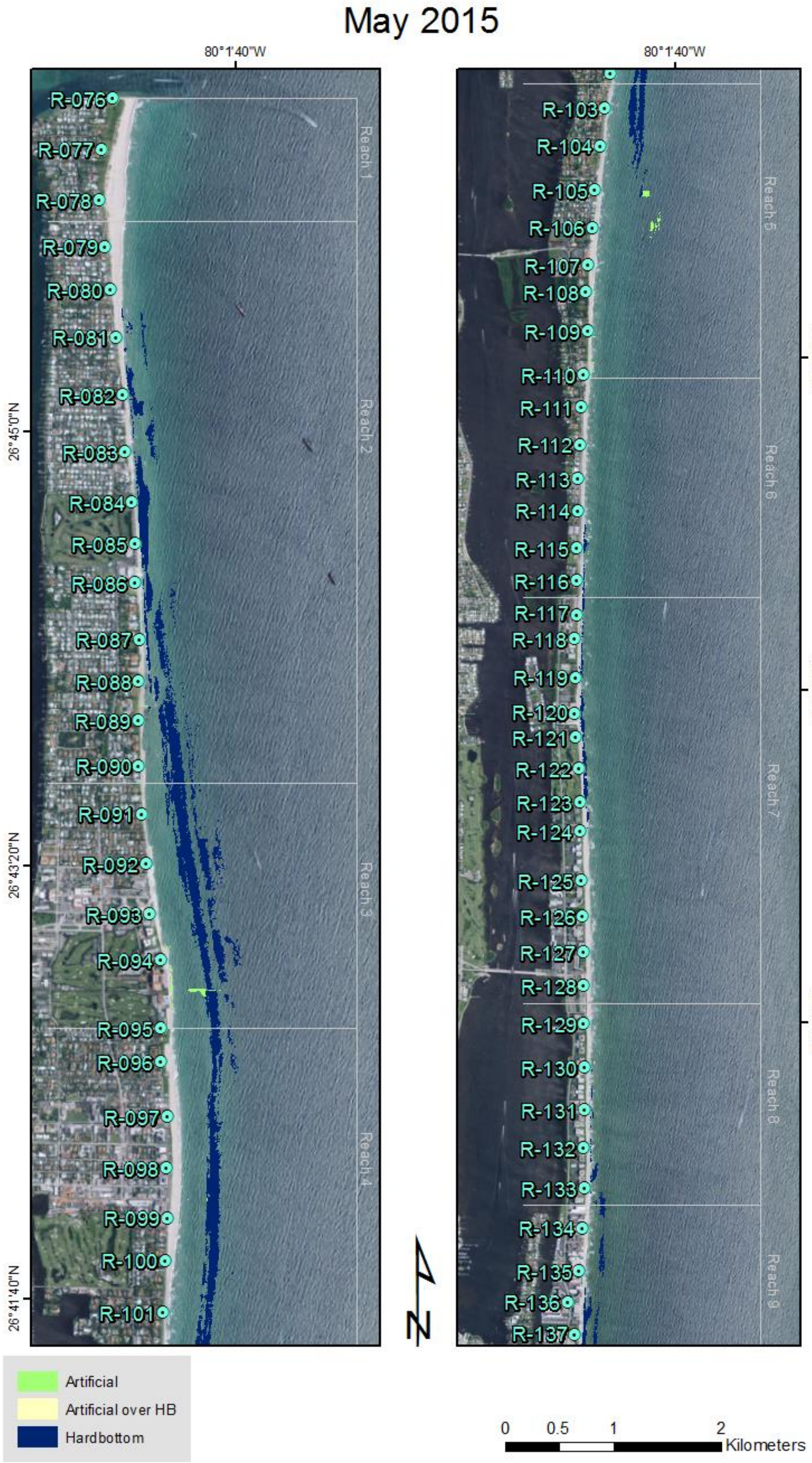


Figure 9. May 2015 hardbottom polygon delineation

Table 1. Acreage of mapped hardbottom by Reach for each

Reach	08/2000	7/8/2001	7/2003	7/16, 7/17, 2004	7/25 & 8/6, 2005	7/26/2006	7/15, 7/20, 2007	7/31/2008	10/22/2008	10/2, 10/-3, 2009	7/10/2010	10/10/2010	5/8/2011	10/10/2011	3/30/2012	7/3/2014	11/11, 11/13, 2014	5/20/2015
Reach 1	0.05	0.44	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.00	0.00	0.26	0.26	0.06	0.00	0.00	0.00
Reach 2	46.69	43.96	36.35	42.86	87.81	67.37	68.47	56.04	72.75	77.23	74.43	68.98	80.42	62.59	59.61	63.46	59.83	49.08
Reach 3	39.39	45.03	44.95	42.28	48.29	47.49	44.43	44.43	81.21	53.92	53.95	63.59	67.93	51.86	55.46	53.10	57.06	51.77
Reach 4	34.53	43.48	48.00	47.27	17.11	39.08	26.73	26.81	50.57	50.70	51.40	50.69	50.68	31.80	41.93	39.76	44.50	43.95
Reach 5	18.35	18.07	19.79	17.46	16.58	19.69	17.32	10.42	17.09	19.57	19.97	21.08	21.57	12.57	19.68	18.41	19.40	18.05
Reach 6	19.01	20.58	4.87	1.77	13.74	17.23	11.67	4.84	5.01	1.10	2.13	1.23	1.73	4.17	2.15	2.26	1.77	2.03
Reach 7	1.72	2.67	6.58	9.18	18.70	0.09	1.28	3.04	5.53	1.72	1.86	1.45	0.34	1.02	2.24	10.13	5.47	9.45
Reach 8	7.44	8.60	2.61	5.34	12.91	14.90	11.37	2.23	9.21	0.45	3.53	2.72	1.86	4.35	2.53	9.57	10.05	3.63
Reach 9	8.52	2.17	0.61	8.62	10.90	18.77	12.85	8.81	3.06	1.03	6.21	3.42	5.19	5.07	2.52	15.46	14.57	10.86
Total Acreage	**175.70**	**185.00**	**163.76**	**174.79**	**226.04**	**224.63**	**194.12**	**190.61**	**244.43**	**205.92**	**213.48**	**213.15**	**229.97**	**173.70**	**186.17**	**212.15**	**212.64**	**188.81**

Table 2. Averages of hardbottom acreages by reach, throughout all years.

Reach	Average Acreage
Reach 1	0.07
Reach 2	62.11
Reach 3	52.56
Reach 4	41.06
Reach 5	18.06
Reach 6	6.52
Reach 7	4.58
Reach 8	6.29
Reach 9	7.70
Total Av. Acreage	**200.84**

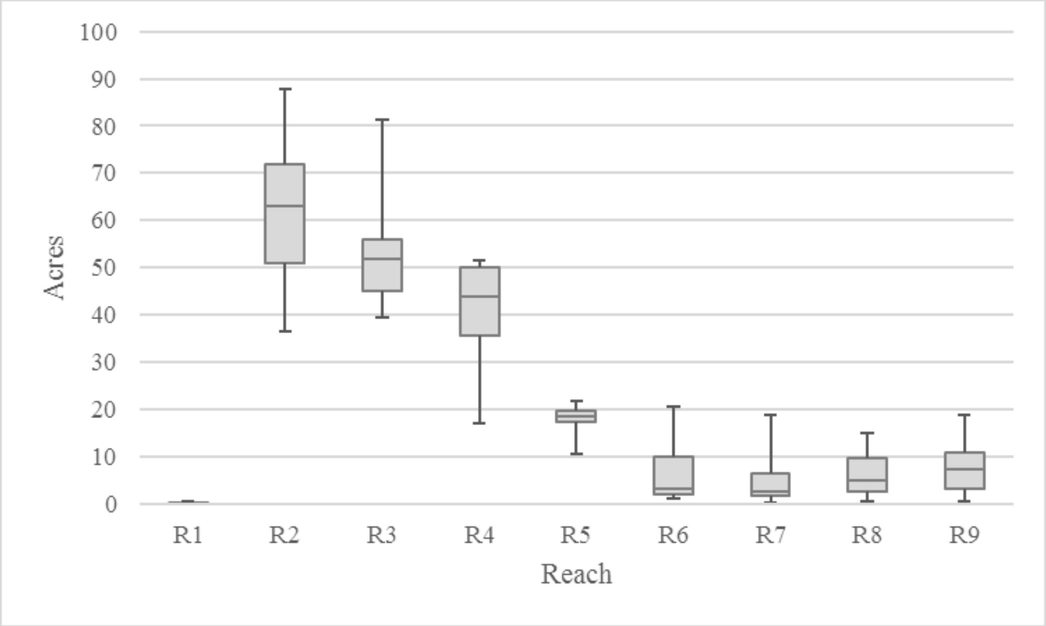


Figure 10. Box and whisker plot of the variability of total hardbottom area by reach, with reaches across the x axis and acres across the y. Whiskers represent the minimum and maximum values observed, while the boxes display the 1st and 3rd quartiles and the middle line represents the median of the data.

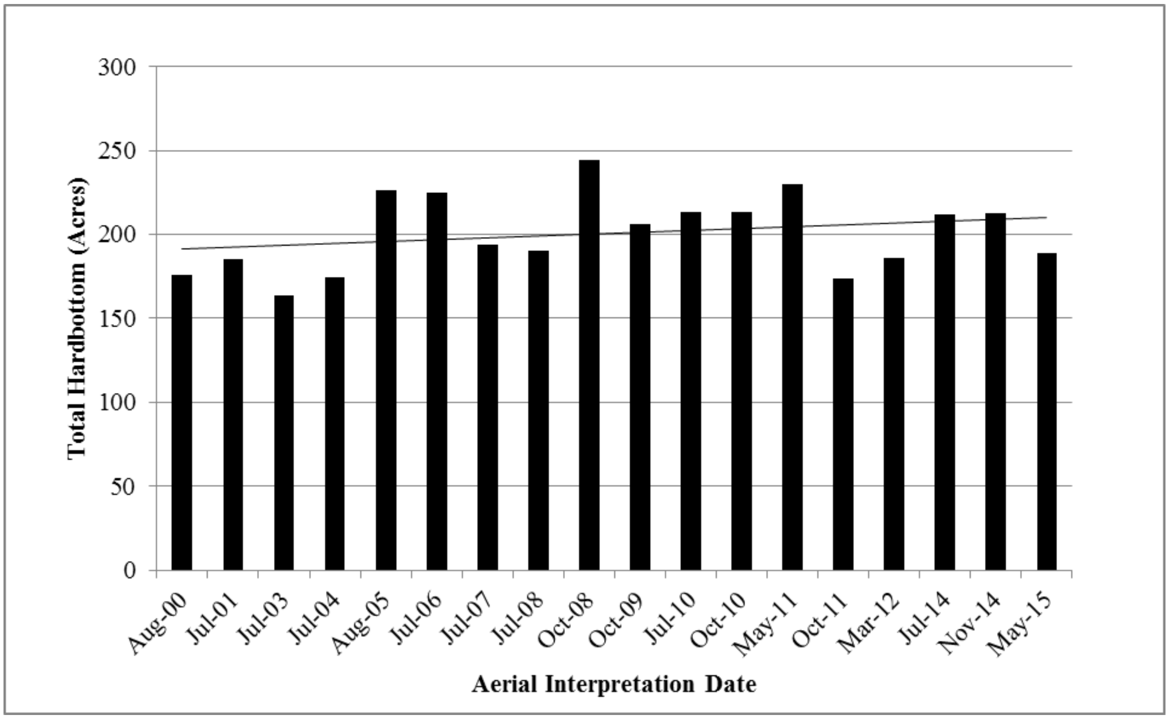


Figure 11. Total mapped hardbottom by year. October 2008 had the highest amount of mapped hardbottom, the line represents the linear trend.

3.1.2 Inter-Annual Hardbottom Exposure

The maps created in July 2014 - May 2015 were examined to evaluate seasonal sediment movement. July and November 2014 total areas of exposed hardbottom were very similar (212.15 acres and 212.65 acres respectively) (Table 3). In May 2015, total exposed hardbottom dropped to a total of 188.62 acres. The major losses from 2014 to 2015 were seen in Reaches 2, 3, 8, and 9 (Table 3; Figure 12). However, in May 2015 Reach 6 had a net gain of 0.26 acres and Reach 7 had a net gain of 3.98 acres since November. Although acreage was gained in the six-month period, total acreages for Reaches 6 and 8 were still not as high as in July 2014. November 2014 showed the highest overall acreage of near shore hardbottom, with higher areas in Reaches 3, 4, 5, and 8.

Table 3. Areas of exposed hardbottom (acres) of reaches 1-9 as mapped for July and November 2014, and May 2015.

BMA Reach Designation	**FDEP Reference Monuments**	**July 3, 2014**	**November 11-13, 2014**	**May 20, 2015**
Reach 1	R-76 to R-78	0.00	0.00	0.00
Reach 2	R-78 to R-90+400	63.46	59.83	49.08
Reach 3	R-90+400 to R-95	53.10	57.06	51.77
Reach 4	R-95 to R-102+300	39.76	44.50	43.95
Reach 5	R-102+300 to R-110+100	18.41	19.40	18.05
Reach 6	R-110+100 to R-116+500	2.26	1.77	2.03
Reach 7	R-116+500 to R-128+530	10.13	5.47	9.45
Reach 8	R-128+530 to T-133+500	9.57	10.05	3.63
Reach 9	T-133+500 to R-137+400	15.46	14.57	10.86

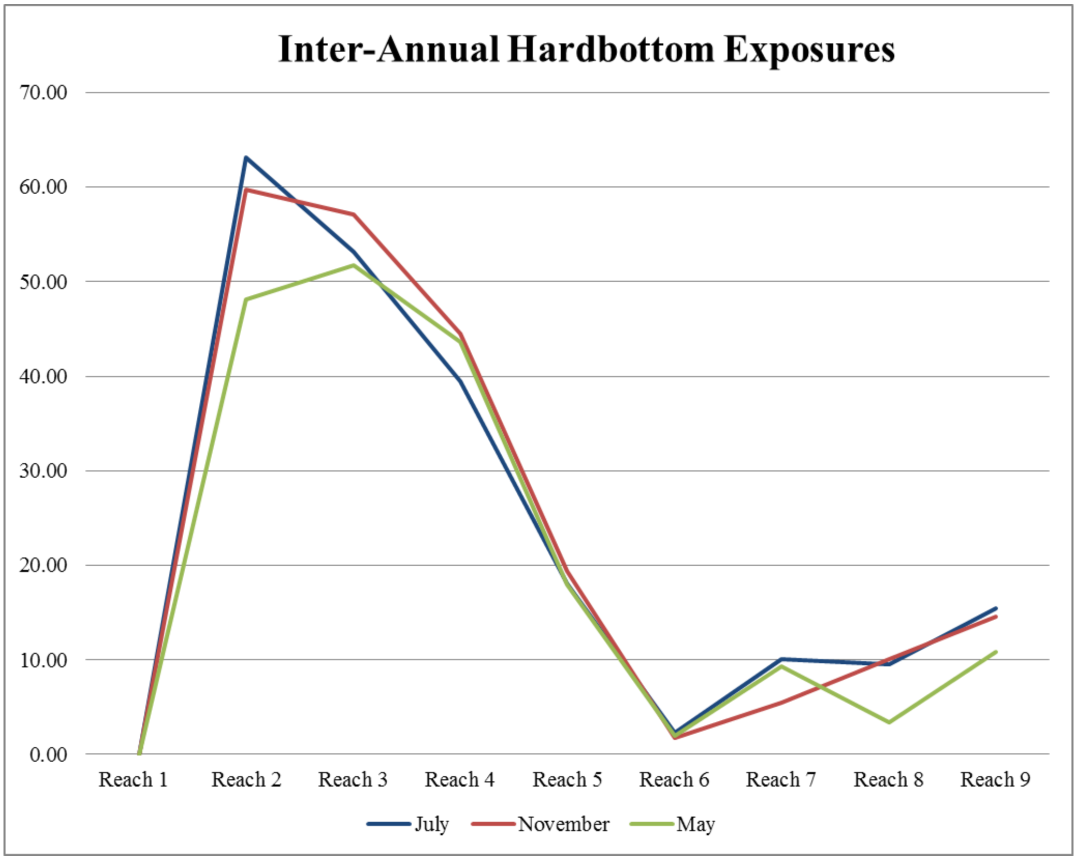


Figure 12. Area of mapped hardbottom since July 2014 by reach. November 2014 shows the highest amount of hardbottom present in 2014-2015.

3.2 Community Structure

A total of 117 sites were surveyed between September 9 and September 18, 2014. Twenty-eight sites were bare sediment and excluded from the hardbottom analyses. The 93 sites with hardbottom were used to assess benthic cover, density and species composition.

3.2.1 Stony Corals

A total of 865 colonies comprised of 7 stony coral (scleractinian) species were identified. *Siderastrea* spp. was the dominant coral throughout the survey sites. *Siderastrea* spp. contributed 92.1% (797 of the 865 colonies) to the total stony coral assemblage. Most of the colonies (81%) were less than 5 cm in diameter. Other species identified in the survey area included (in decreasing abundance) *Solenastrea bournoni* (36), *Stephanocoenia intersepta* (27), *Porites astreoides* (2), *Montastraea cavernosa* (1), *Pseudodiploria clivosa* (1), and *Oculina diffusa* (1) (Table 3). Ten *Millepora alcicornis* hydrozoan colonies were also counted. High exposure sites had significantly higher abundance of *Siderastrea* spp. than low or medium exposure sites. *Stephanocoenia intersepta* were also found in higher abundance in areas of high exposure. *Solenastrea bournoni* had the highest abundance in areas classified as medium exposure.

Mean (±SD) stony coral density within the 93 hardbottom sites was 0.96 ± 1.56 colonies/m^2, but was 0.085 ± 0.215 colonies/m^2 when *Siderastrea* spp. colonies were excluded. Mean (±SE) density of all stony coral colonies $\leq$5 cm diameter within the hardbottom sites was 0.83 ± 1.31 colonies/m^2, but was 0.057 ± 0.17 colonies/m^2 with *Siderastrea* spp. colonies excluded.

3.2.2 Octocorals

A total of 1,371 octocoral colonies of four morphologies (rods, plumes, fans or encrusting) were identified within the 117 sites (Table 4). Rods were the dominant octocoral throughout the survey sites contributing 73.7% (1,010 of the 1,371 colonies) to the total octocoral assemblage. Of all octocorals observed, 42.7% of the colonies (586) were less than 5 cm in diameter. Plumes were the next dominant octocoral contributing to 25.6% (352 colonies) of the assemblage. Fans and encrusting octocorals were rarely encountered (Table 5). High exposure sites had significantly higher plume abundance than low or medium exposure sites and fans were only found at high exposure sites. Encrusting octocorals were found at medium and high exposure sites

Mean (±SD) octocoral density within the 93 hardbottom sites was 1.51 ± 4.2 colonies/m², but was 0.4 ± 1.3 colonies/m² when rod colonies were excluded. Mean (±SD) density of all octocoral colonies ≤ 5 cm (diameter) within the hardbottom sites was 0.64 ± 2.37 colonies/m², but was 0.10 ± 0.27 colonies/m² with rod colonies excluded.

Table 4. Stony coral species abundance based on size identified within the survey sites. Species are listed in decreasing abundance and density within the 93 hardbottom sites**.**

***Coral* spp.**	***Size Class (Diameter)***	***Abundance***	***Mean Density (m^2)***	***SD***
Siderastrea spp	< 5 cm	711	0.867	1.331
Siderastrea spp	5 - <10 cm	82	0.100	0.298
Stephanocoenia intersepta	< 5 cm	26	0.031	0.129
Solenastrea bournoni	< 5 cm	21	0.025	0.094
Solenastrea bournoni	5 - <10 cm	13	0.015	0.073
Siderastrea spp	10 - < 25 cm	4	0.005	0.027
Solenastrea bournoni	10 - < 25 cm	2	0.002	0.016
Pseudodiploria clivosa	5 - <10 cm	1	0.001	0.011
Oculina diffusa	5 - <10 cm	1	0.001	0.011
Porites astreoides	< 5 cm	1	0.001	0.011
Porites astreoides	5 - <10 cm	1	0.001	0.011
Stephanocoenia intersepta	10 - < 25 cm	1	0.001	0.011
Montastraea cavernosa	5 - <10 cm	1	0.001	0.011
Total		**865**		

Table 5. Octocoral abundance and mean density across the 93 hardbottom sites**.**

Octocoral Class	*Size Class*	*Abundance*	*Mean Density (m^2)*	*SD*
Rod	< 5 cm	490	0.598	2.306
Rod	5 - <10 cm	263	0.321	1.350
Rod	10 - < 25 cm	146	0.178	0.564
Plume	10 - < 25 cm	123	0.150	0.556
Plume	5 - <10 cm	114	0.139	0.530
Rod	≥ 25 cm	111	0.304	0.567
Plume	< 5 cm	90	0.110	0.276
Plume	≥ 25 cm	25	0.031	0.129
Fan	< 5 cm	4	0.005	0.044
Encrusting	< 5 cm	2	0.002	0.022
Encrusting	10 - < 25 cm	2	0.002	0.022
Encrusting	5 - <10 cm	1	0.001	0.011
Total		**1,371**		

3.2.3 Benthic Macroalgae

Mean total macroalgae percent cover was assessed in 5, 0.5 m^2 quadrats along the 20m transect. Mean percent cover between all hardbottom sites was 13.5% (±1.75). Macroalgae were found on every hardbottom site and in relatively high occurrence between all sites. Of the 546 quadrats assessed on hardbottom, 66% (361) recorded macroalgae. Twenty-five genera were documented as one of the two dominant genera within a quadrat (Table 6). dictyota was found as one of the most dominant algae the most frequently (37.4%) followed by dasycladus (23.1%), gelidiella (15%), and dasya (8.6%). Mean cover hardbottom sites indicated that dictyota was highest (5.4%)

Table 6. Frequency of occurrence and mean percent cover of the macroalgae species.

Genera	*Overall Frequency of Occurrence*	*% Frequency of Occurrence*	*Mean Percent Cover*
Dictyota	204	37.4%	5.44
Dasycladus	126	23.1%	1.41
Gelidiella	82	15.0%	0.94
Dasya	47	8.6%	0.84
Gelidium	43	7.9%	0.76
Bryothamnion	39	7.1%	0.35
Halimeda	39	7.1%	0.28
Jania	23	4.2%	0.07
Caulerpa	10	1.8%	0.06
Ceramium	8	1.5%	0.05
Wrangelia	8	1.5%	0.04
Laurencia	7	1.3%	0.03
Padina	7	1.3%	0.03
Hypnea	7	1.3%	0.03
Digenea	7	1.3%	0.02
Chondria	4	0.7%	0.02
Chaetomorpha	3	0.5%	0.02
Udotea	2	0.4%	0.02
Dictyopteris	2	0.4%	0.02
Sargassum	2	0.4%	0.01
Herposophinia	2	0.4%	0.01
Gracilaria	2	0.4%	0.004
Amphiroa	1	0.2%	0.002
Avrainvillea	1	0.2%	0.001
Acetabularia	1	0.2%	0.001

followed by dasycladus (1.4%), bryothamnion (0.94%), gelidiella (0.84%), and dasya (0.76%) (Table 6). Low exposure sites were composed of 79% dictyota, whereas medium and high exposure sites were comprised of 30% and 40% dictyota respectively (Figure 13). Low exposure sites had lower number of algal genera. Bryothamnion, chondria, and hypnea were only found in medium and high exposure sites and dictyopteris and avrainvillea were only found in sites classified as highly exposed.

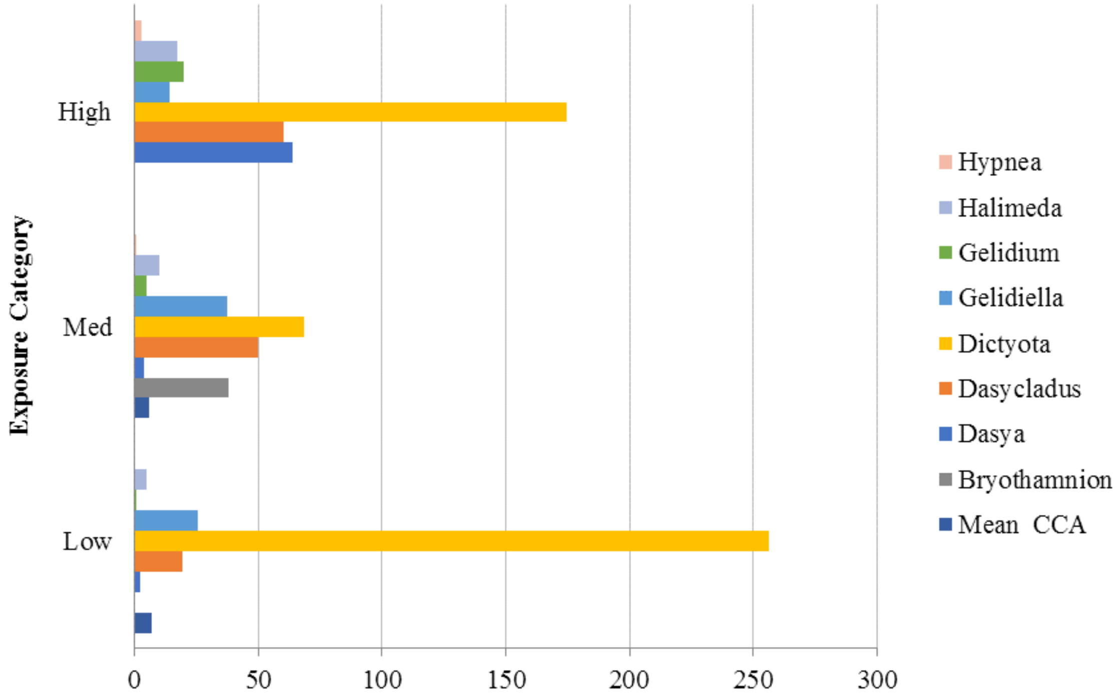


Figure 13. Sum of average percent cover by exposure type. Dictyota was dominant across all exposre categories. The highest total cover occuried in areas of high exposure

3.2.4 Sponges

Sponges were assessed by presence/absence. Of the hardbottom sites assessed, 32 sites documented sponges >10 cm (height), equating to 38.5% of all sites. Broken down by exposure category, 15.6% of high exposure sites were observed to have sponges larger than 10 cm. Sites categorized as medium exposure had the lowest percentage of presence of observed large sponges with only 9.6%. Low exposure sites fell just below that of high exposure with 13.2% of sites with large sponges present. (Figure 14)

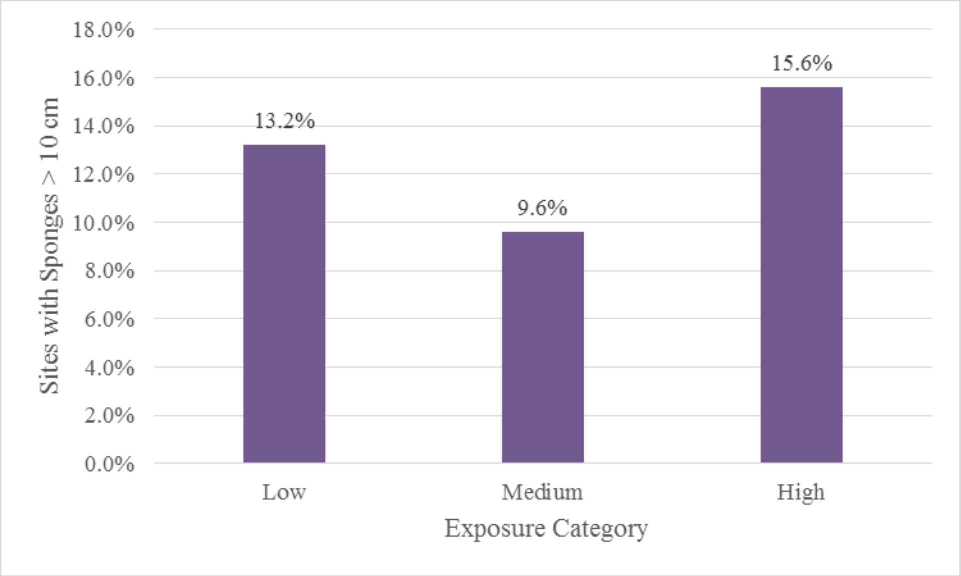


Figure 14. Percentage of sites with sponges >10 cm by exposure

3.4 Community Analysis

3.4.1 Hardbottom Designation

Multivariate statistical analyses uncovered patterns in the benthic community data and their relationship to the present hardbottom designations and the exposure categories. Density data for each size class of every coral species and octocoral morphology were placed into a matrix. Sites categorized by hardbottom designation showed no significant relationship in benthic cover (Figure 15; Table 7). Hard and soft coral size class densities categorized by near shore and offshore Hardbottom Designations were significantly dissimilar from each other, yet the difference was slight (Figure 16; Table 8). This weak dissimilarity was driven by the Offshore having a lower number of *Siderastrea* spp < 5 cm and higher large octocoral densities (Table 9).

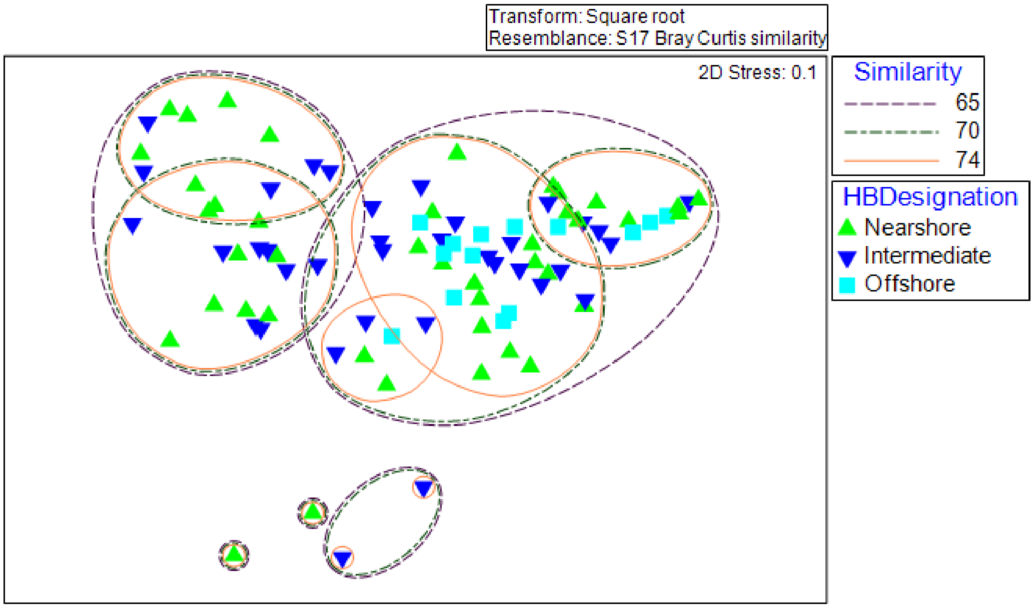


Figure 15. Multidemensional scaling plot of benthic cover (sand, macroalgae and hydriods) data categorized by hardbottom designation.

Table 7. Analysis of similarity results testing hardbottom designation classes by benthic cover site data.

Pairwise Tests					
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Near shore, Intermediate	0.007	25.5	Very large	999	254
Near shore, Offshore	-0.086	97.6	Very large	999	975
Intermediate, Offshore	-0.027	68.6	Very large	999	685

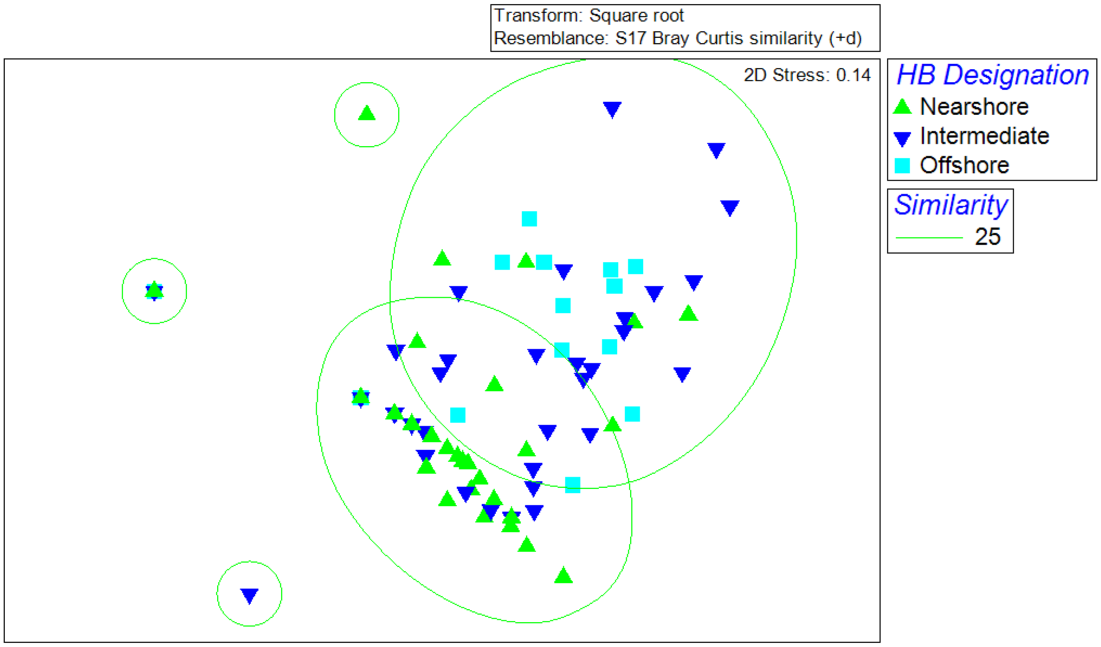


Figure 16. MDS plot of coral and octocoral density data categorized by Hardbottom Designation.

Table 8. Analysis of similarity results testing Hardbottom Designation classes by density and size class data.

Pairwise Tests					
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Near shore, Intermediate	0.064	0.8	Very large	999	7
Near shore, Offshore	**0.197**	**1.4**	**Very large**	**999**	**13**
Intermediate, Offshore	0.047	23	Very large	999	229

Table 9. Similarity Percentages (SIMPER) analysis of coral density (individuals per m²) data between near shore, intermediate and offshore Hardbottom Designations. SID = *Siderastrea.*

Average dissimilarity = 94.73

	Near shore	**Offshore**				
Species	Av. Dens	Av.Dens	Av.Diss	Diss/SD	Contrib%	Cum.%
SID spp < 5 cm	0.01	0.03	2.56	1.02	2.71	100.00
ROD 10 - < 25 cm	0.08	0.28	25.63	1.02	27.06	27.06
ROD 5 - <10 cm	0.34	0.18	21.33	0.84	22.51	74.84
ROD < 5 cm	0.28	0.12	23.94	0.83	25.28	52.33
PLUME 10 - < 25 cm	0.04	0.33	21.27	0.73	22.45	97.29

3.5 Spatial Analysis

The multivariate data were then evaluated as outlined in Costa *et al*. (in press) to determine if sites of more similar data exhibited a spatial relationship. The main clustering in the benthic cover data occurred at 74% similarity (Figure 17) whereas the density data showed distinct clusters at 25% (Figure 18). Factors were created at the respective similarity levels for cover and density and the sites were categorized by in which cluster they resided. These data were then displayed in GIS to visualize where the different clusters spatially occurred and overlain on the hardbottom designations to visualize any relationships. There was no obvious spatial patterning between habitat designation and benthic cover (Figure 19). Clusters of sites with similar data were interspersed and spread throughout much of the map and across all classes. Although density cluster B (green dots in Figure 20) mostly occurred in the deeper areas in Mid-town.

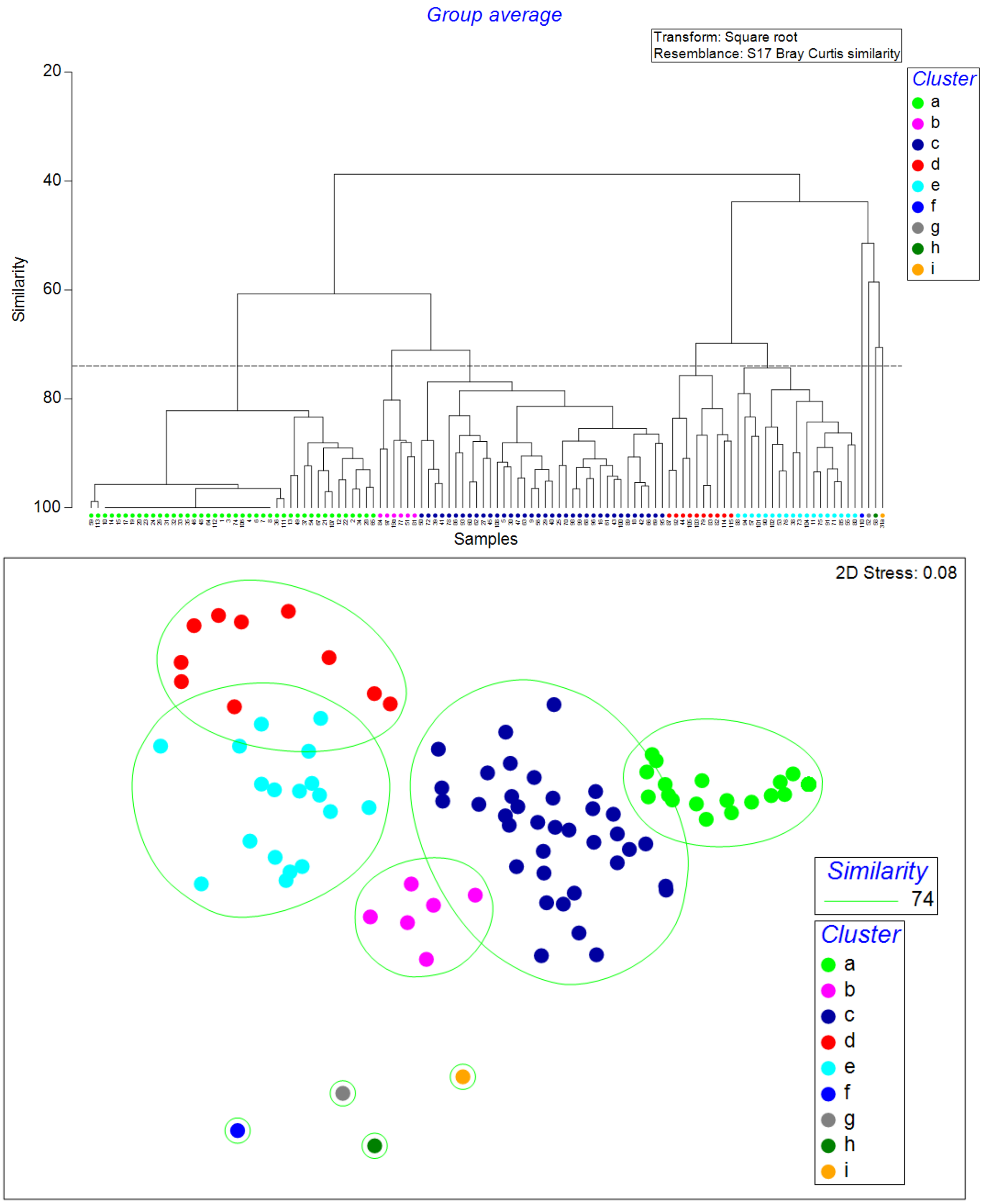


Figure 17. Benthic cover dendrogram from Cluster analysis (top) and corresponding MDS plot (bottom). The dashed line in top and symbology in both represent clusters at 74% similarity.

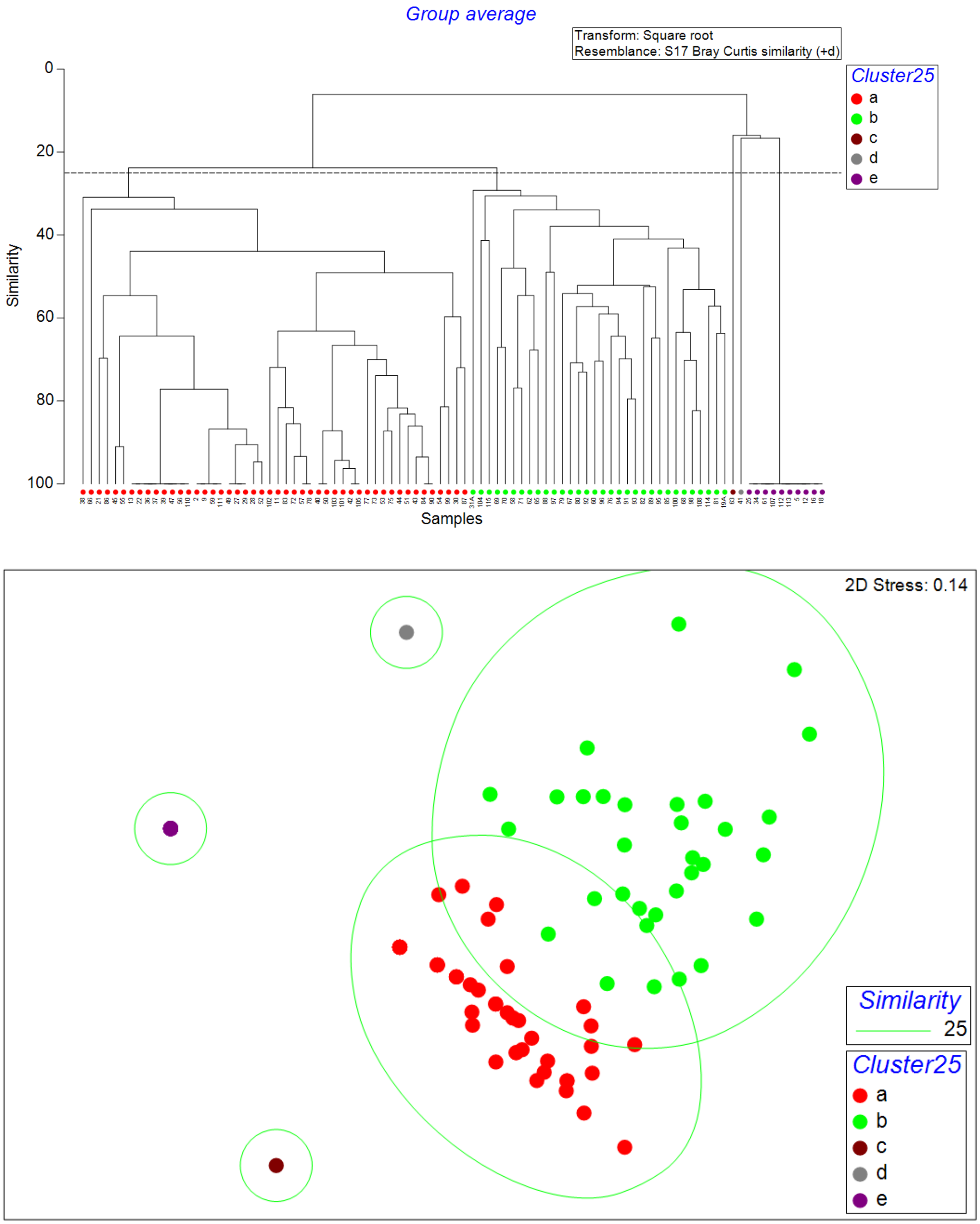


Figure 18. Hard and soft coral density data dendrogram from Cluster analysis (top) and corresponding MDS plot (bottom). The dashed line in top and symbology in both represent clusters at 25% similarity.

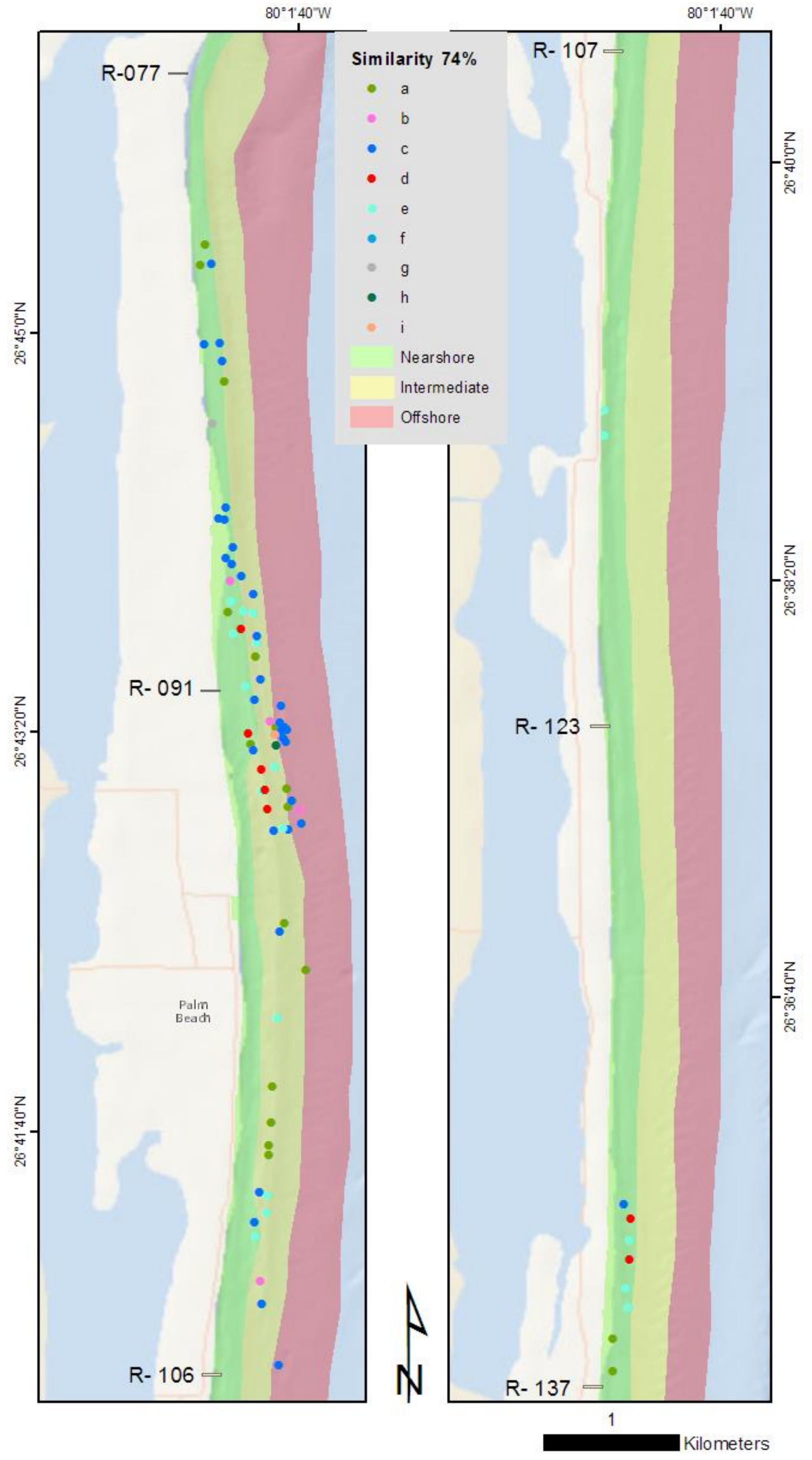


Figure 19. Map of groundtruthing sites categorized by multivariate cluster ananlysis (74% similarity) of benthic cover data overlaying the previously-defined hardbottom designations. Green is high sand cover and red and teal are high algae cover.

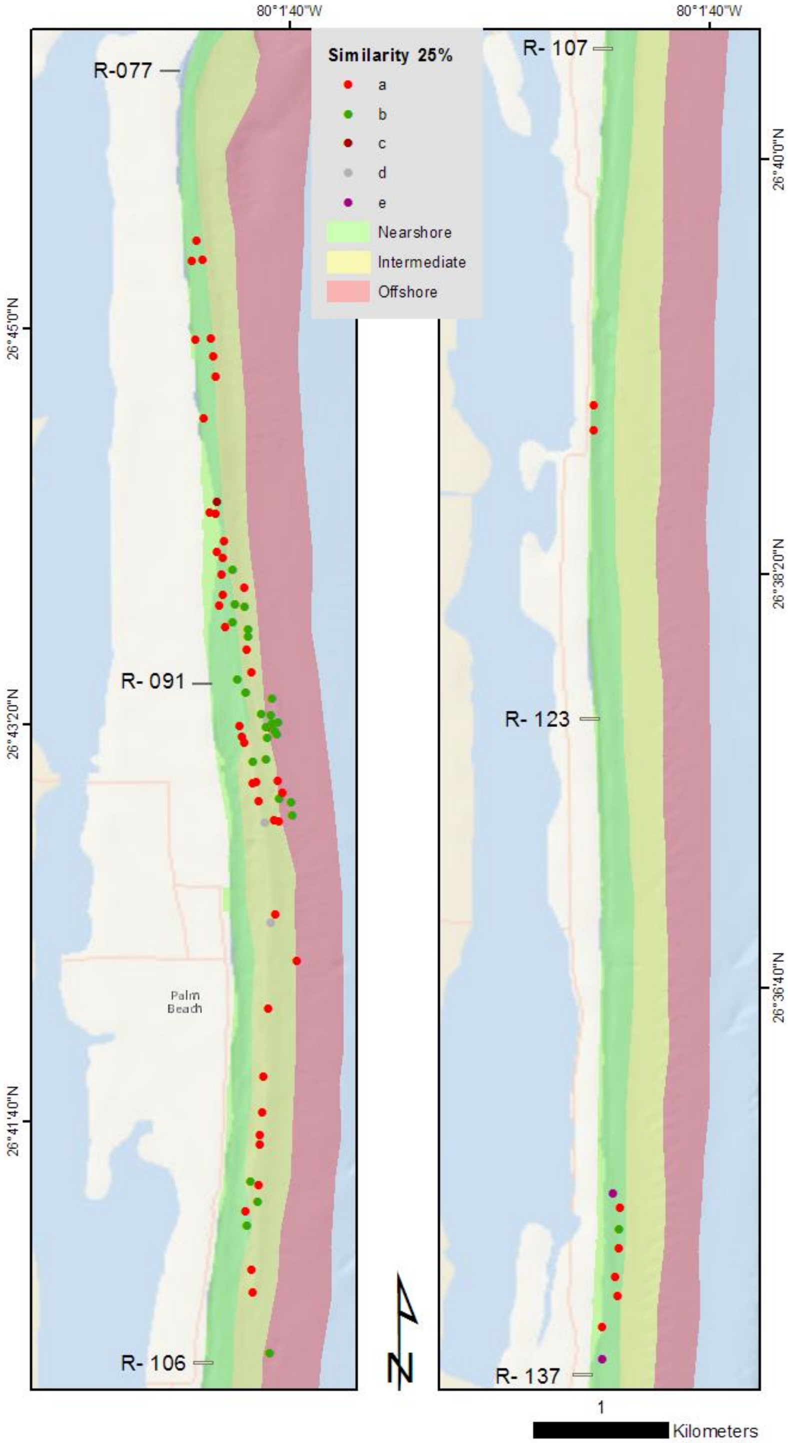


Figure 20. Map of groundtruthing sites categorized by multivariate cluster ananlysis (25% similarity) of hard and soft coral size class density data overlaying the previously-defined hardbottom designations.

3.6 Exposure

The community had significant relationships with the exposure frequency categories. A cluster analysis, MDS plot, and analysis of similarity (ANOSIM) of sites categorized by exposure frequency showed significant relationships in benthic cover between all comparisons (Figure 21; Table 10). The ANOSIM of benthic cover data indicated a medium strength dissimilarity between the high and low exposure classes. Mean percent cover of macroalgae and hydroids increased going from low to high exposures (1.45, 2.16, 3.74 and 0.66, 1.10, 1.52 respectively), while sand cover decreased along the same gradient (7.83, 5.92, 3.70). (Table 11). Sand cover was significantly higher at low exposure sites than at medium, or high exposure.(ANOVA; $p < 0.0088$) (Figure 22). Hydroid cover was higher in high exposure sites versus medium and low (ANOVA; $p < 0.0390$). Macroalgae cover was highest in high exposure sites, followed by low exposure sites and lowest in medium exposure sites (ANOVA; $p < 0.0157$).

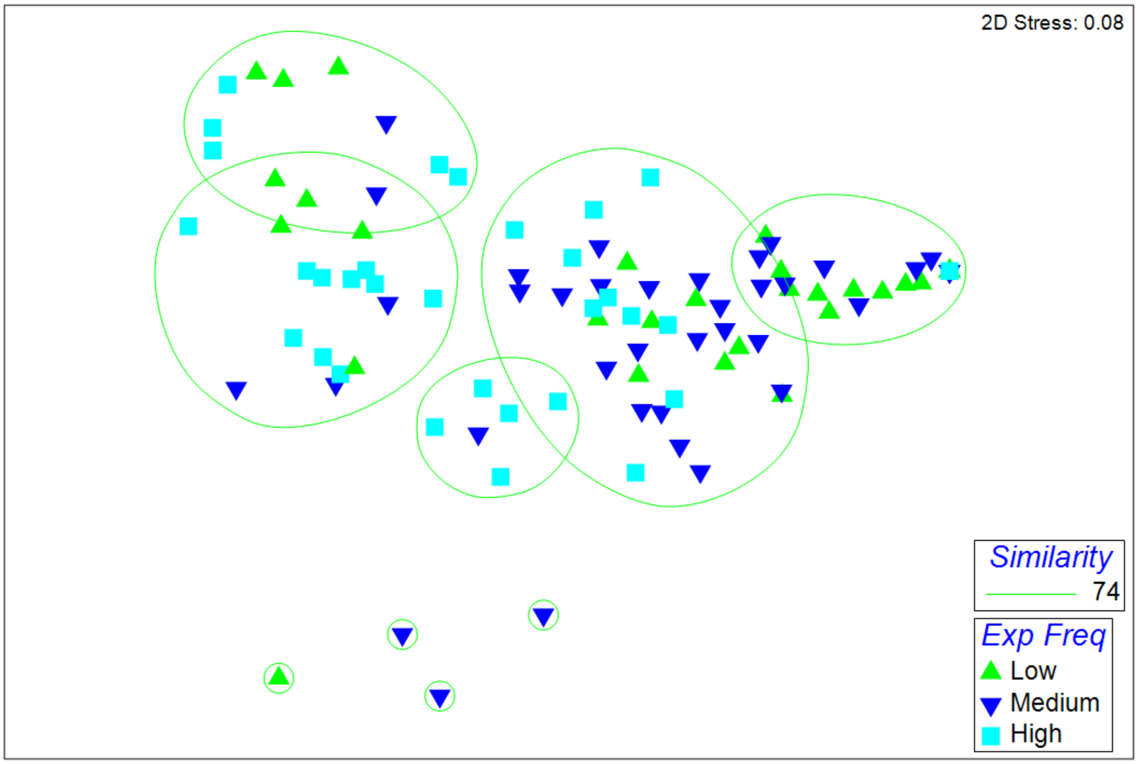


Figure 21. Multidemensional scaling plot of benthic cover data by site categorized by exposure frequency.

Table 10. Analysis of similarity results testing exposure frequency classes by benthic cover site data. Bold indicates medium strength significance.

Pairwise Tests					
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Low, Medium	0.09	0.3	Very large	999	2
Low, High	**0.335**	**0.1**	**Very large**	**999**	**0**
Medium, High	0.164	0.1	Very large	999	0

Table 11. Similarity Percentages (SIMPER) analysis of benthic cover site data between exposure frequency categories.

Groups Low & Medium

Average dissimilarity = 37.97

	Group Low	**Group Medium**				
Species	Av.Cover	Av.Cover	Av.Diss	Diss/SD	Contrib%	Cum.%
Sand	7.83	5.92	19.81	1.22	52.17	52.17
Macroalgae	1.45	2.16	10.85	1.26	28.57	80.74
Hydroid	0.66	1.10	5.25	1.30	13.84	94.57

Groups Low & High

Average dissimilarity = 50.87

	Group Low	**Group High**				
Species	Av.Cover	Av.Cover	Av.Diss	Diss/SD	Contrib%	Cum.%
Sand	7.83	3.70	25.65	1.66	50.43	50.43
Macroalgae	1.45	3.74	15.54	1.69	30.54	80.97
Hydroid	0.66	1.52	6.29	1.62	12.37	93.35

Table 11. Continued.

Groups Medium & High

Average dissimilarity = 41.36

	Group Medium	**Group High**				
Species	Av.Cover	Av.Cover	Av.Diss	Diss/SD	Contrib%	Cum.%
Sand	5.92	3.70	19.28	1.43	46.61	46.61
Macroalgae	2.16	3.74	12.91	1.44	31.22	77.83
Hydroid	1.10	1.52	5.70	1.41	13.77	91.61

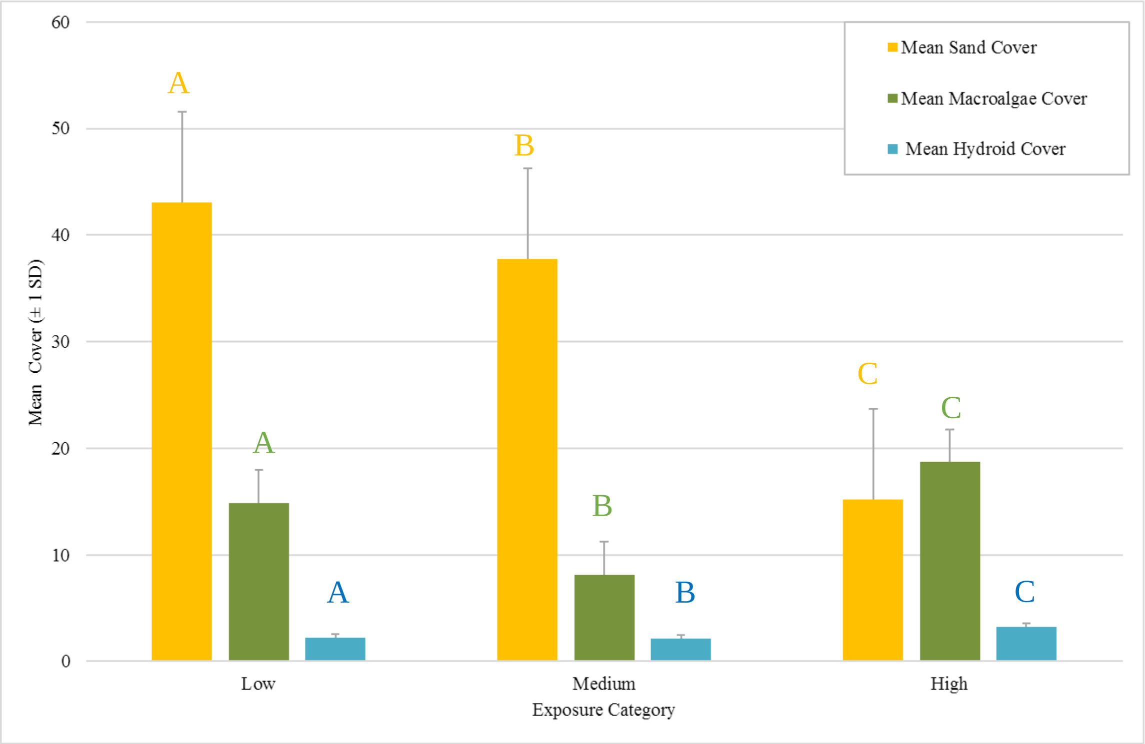


Figure 22. Mean sand cover is significantly lower at high exposure categories and vice versa for macroalgae and hydroids. Error bars indicate ±1 standard error of the mean. Letters indicate significance between exposure frequencies within categories.

Scleractinian and octocoral coral densities had no statistically significant increase with the increase of hardbottom exposure frequency. There were weak significant dissimilarities in density between low and high and medium and high exposures (Figure 23; Table 12). Manyof the differences in similarity in all comparisons came from SID Spp < 5 cm, ROD < 5 cm, SID Spp 5 - <10 cm, and ROD 5 - <10 cm (Table 13). Rods < 5 cm were significantly higher on areas of high exposure (ANOVA; $p < 0.0070$) (Figure 24). Rods 5-10 cm in high exposure were significantly different from low and medium exposure sites, with the lowest cover in areas of medium exposure (ANOVA; $p < 0.056$). SID spp. < 5 cm were increasingly higher with higher exposures, with high exposure sites significantly different from medium and low sites (ANOVA; $p < 0.0100$). SID spp. 5-10 cm densities were significantly higher in high exposure sites than low (ANOVA; $p < 0.0276$).

Hard coral and octocoral densities were significantly higher on high exposure sites than in medium or low sites (ANOVA; $p < 0.0113$ and $p < 0.0008$ respectively) (Figure 25). High exposure sites had significantly higher plume density than low or medium exposure sites (ANOVA; $p < 0.0001$). Rod density was significantly between every exposure site (ANOVA; $p < 0.0001$) with high exposure having the highest densities and medium exposure having the lowest (Figure 26).

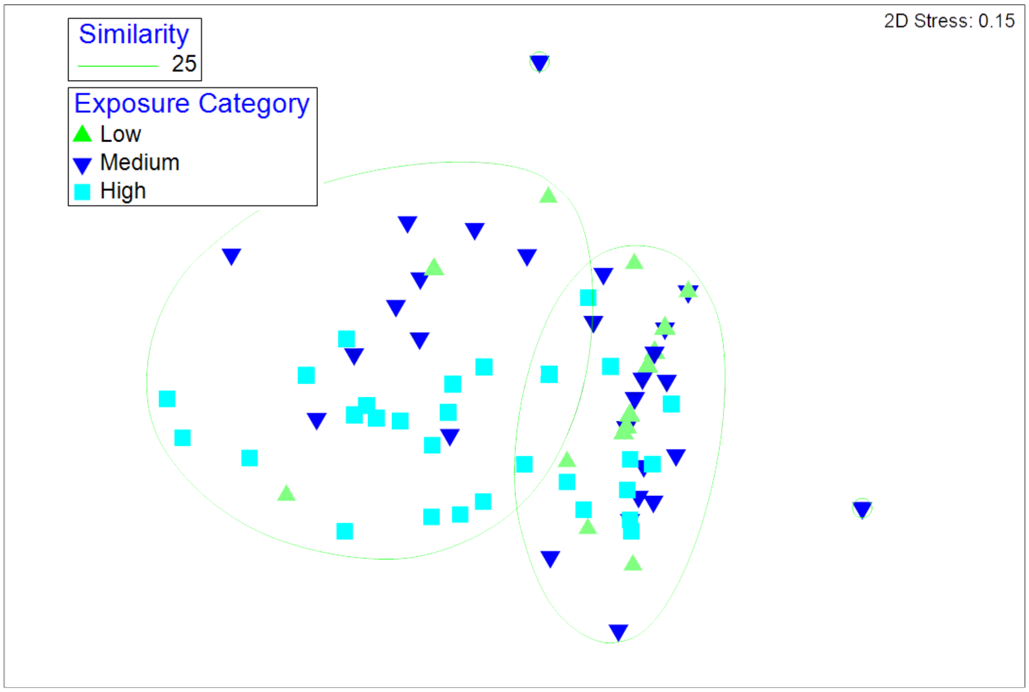


Figure 23. MDS plot of coral and octocoral density data by site categorized by exposure frequency.

Table 12. Analysis of similarity results testing exposure frequency classes by coral and octocoral density. Bold indicates medium strength significance**.**

Pairwise Tests					
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Low, Medium	-0.033	75.4	Very large	999	753
Low, High	**0.253**	**0.1**	**Very large**	**999**	**0**
Medium, High	**0.127**	**0.1**	**Very large**	**999**	**0**

Table 13. Similarity Percentages (SIMPER) analysis of coral density data between exposure frequency categories.

Groups Low & Medium
Average dissimilarity = 61.08

Species	**Group Low** Av.Density	**Group Medium** Av.Density	Av.Diss	Diss/SD	Contrib%	Cum.%
SID Spp < 5 cm	0.71	0.66	20.26	1.10	33.17	33.17
SID Spp 5 - <10 cm	0.05	0.17	6.23	0.54	10.19	53.69
ROD < 5 cm	0.18	0.15	6.31	0.75	10.33	43.50
ROD 5 - <10 cm	0.19	0.12	5.08	0.58	8.32	62.01
ROD 10 - < 25 cm	0.08	0.16	4.42	0.62	7.24	69.25
ROD ≥ 25 cm	0.00	0.12	2.75	0.38	4.50	73.75

Groups Low & High
Average dissimilarity = 69.02

Species	**Group Low** Av.Density	**Group High** Av.Density	Av.Diss	Diss/SD	Contrib%	Cum.%
SID Spp < 5 cm	0.71	0.90	14.38	1.11	20.84	20.84
ROD < 5 cm	0.18	0.66	10.61	0.92	15.37	36.21
ROD 5 - <10 cm	0.19	0.41	7.16	0.95	10.37	46.58
SID Spp 5 - <10 cm	0.05	0.19	6.16	0.69	8.92	55.50
PLUME < 5 cm	0.04	0.33	5.88	0.90	8.52	64.03
PLUME 5 - <10 cm	0.04	0.31	5.32	0.81	7.71	71.73

Groups Medium & High
Average dissimilarity = 68.80

Species	**Group Medium** Av.Density	**Group High** Av.Density	Av.Diss	Diss/SD	Contrib%	Cum.%
SID Spp < 5 cm	0.66	0.90	13.82	1.05	20.09	20.09
ROD < 5 cm	0.15	0.66	9.86	0.93	14.33	34.42
SID Spp 5 - <10 cm	0.17	0.19	6.50	0.80	9.45	43.87
ROD 5 - <10 cm	0.12	0.41	6.44	1.02	9.36	53.24
PLUME < 5 cm	0.05	0.33	5.55	0.91	8.07	61.30
ROD 10 - < 25 cm	0.16	0.27	5.22	0.87	7.59	68.89
PLUME 5 - <10 cm	0.06	0.31	5.01	0.84	7.28	76.17

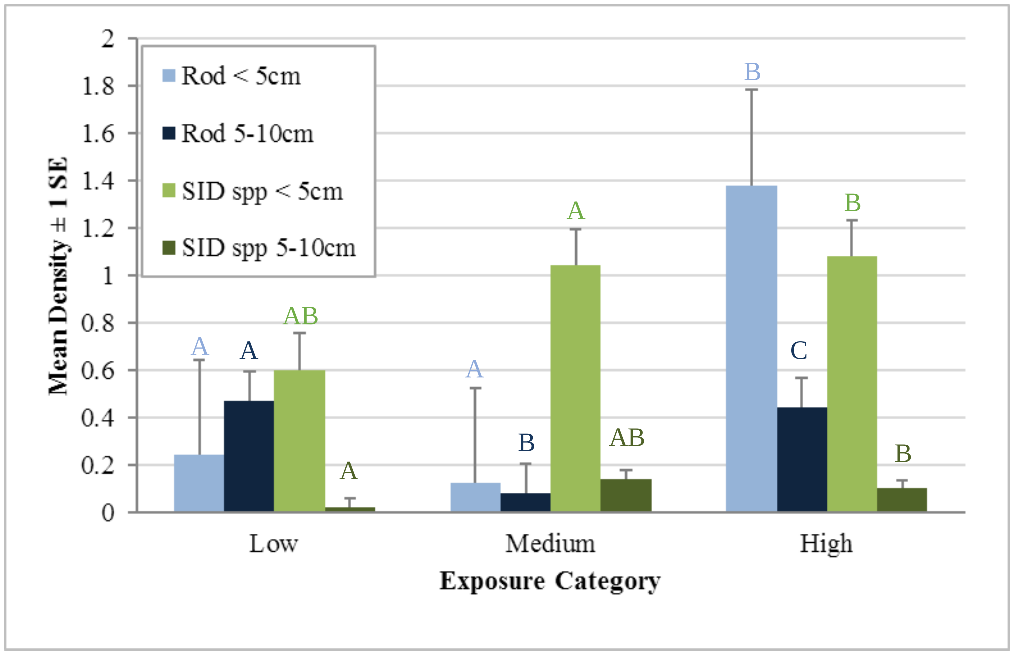


Figure 24. Mean cover of top four species/morphology size classes identified by SIMPER. Error bars indicate ±1 standard error about the mean. Letters indicate significance between exposure frequencies.

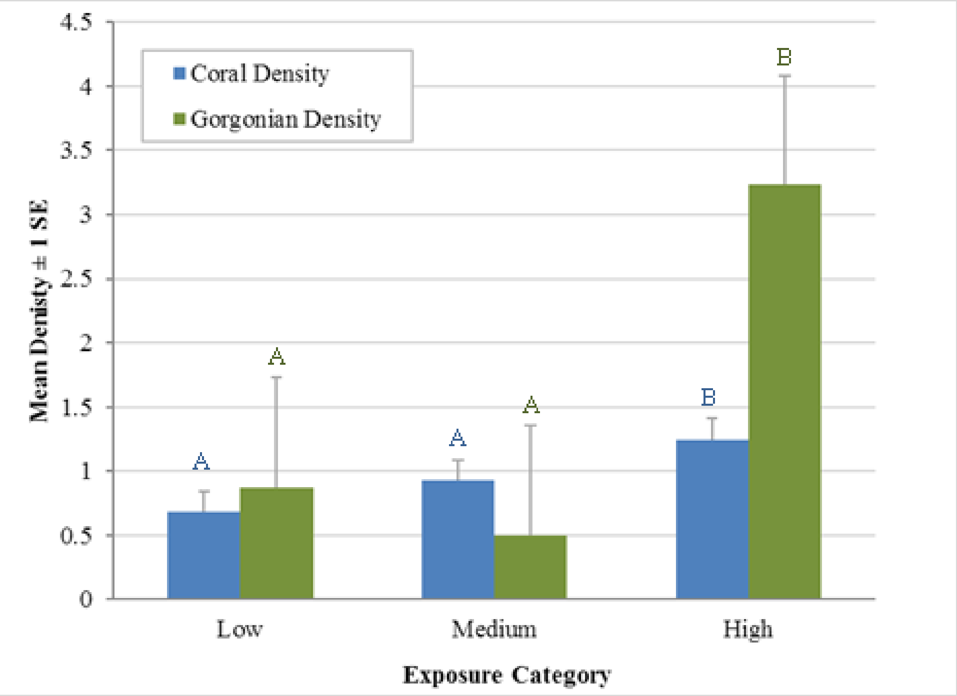


Figure 25. Mean coral and octocoral density by exposure categories. Error bars indicate ±1 standard error of the mean. Letters indicate significance between exposure categories.

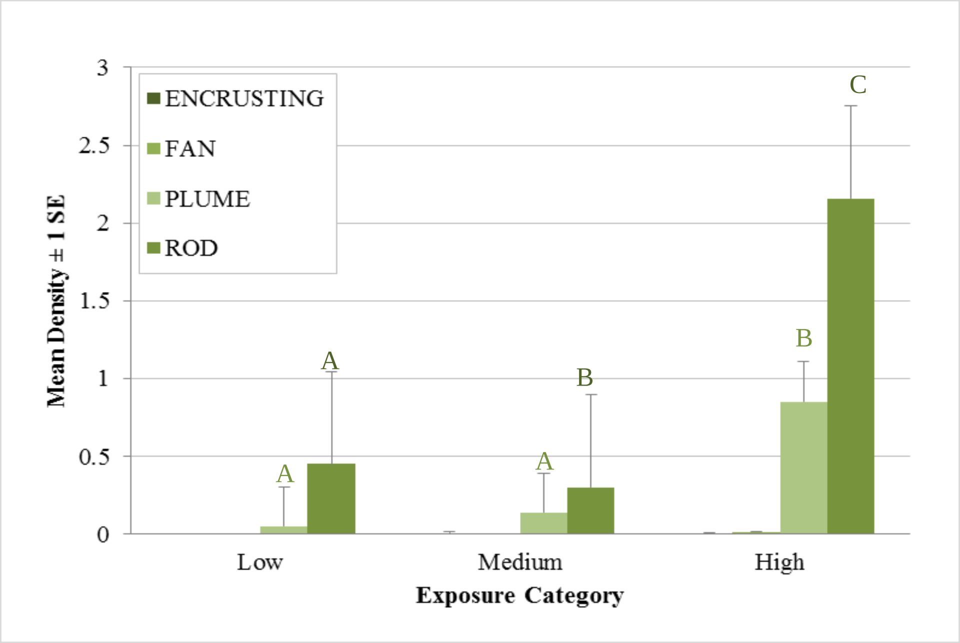


Figure 26. Mean octocoral density by morphology and exposure category. Error bars indicate ±1 standard error of the mean. Letters indicate significance between exposure categories.

At high exposure sites, the dominant stony coral densities were *Siderastrea* spp (1.18), *Stephanocoenia intersepta* (0.07), and *Solenastrea bournoni* (0.02). Only *Siderastrea* spp. were dense enough to perform analyses of variance by exposure category. *Siderastrea* spp. were significantly higher in density comparing sites of low to medium, and medium to high exposure (ANOVA; $p < 0.0001$) (Figure 27). Mean coral richness was also significantly higher at high exposure sites (ANOVA, $p < 0.0134$) (Figure 28). The absence of a species in record does not mean that species is not present in that habitat, but it can be some indication of rarity given the total sampling effort (93 x 10 m^2). *Montastraea cavernosa* and *Oculina diffusa* were only found at high exposure sites and *Porites astreoides* was only found at medium and high exposure sites. Fan octocorals were only found at high exposure sites and encrusting were found at medium and high exposure sites.

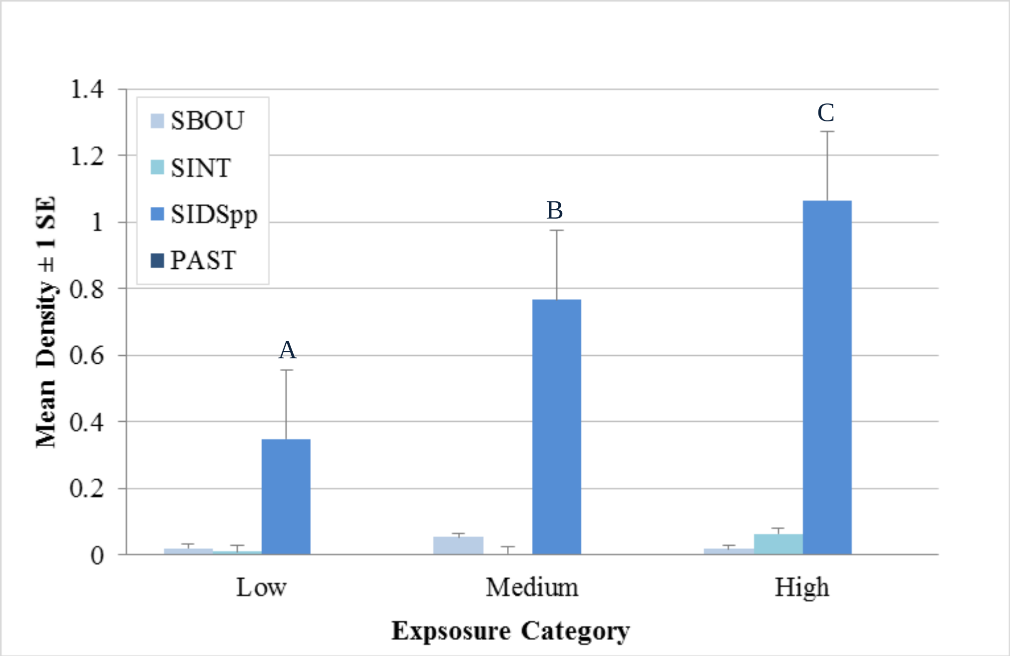


Figure 27. Mean density of the four most prolific corals. Error bars indicate ±1 standard error of the mean. Letters indicate significance between exposure categories.

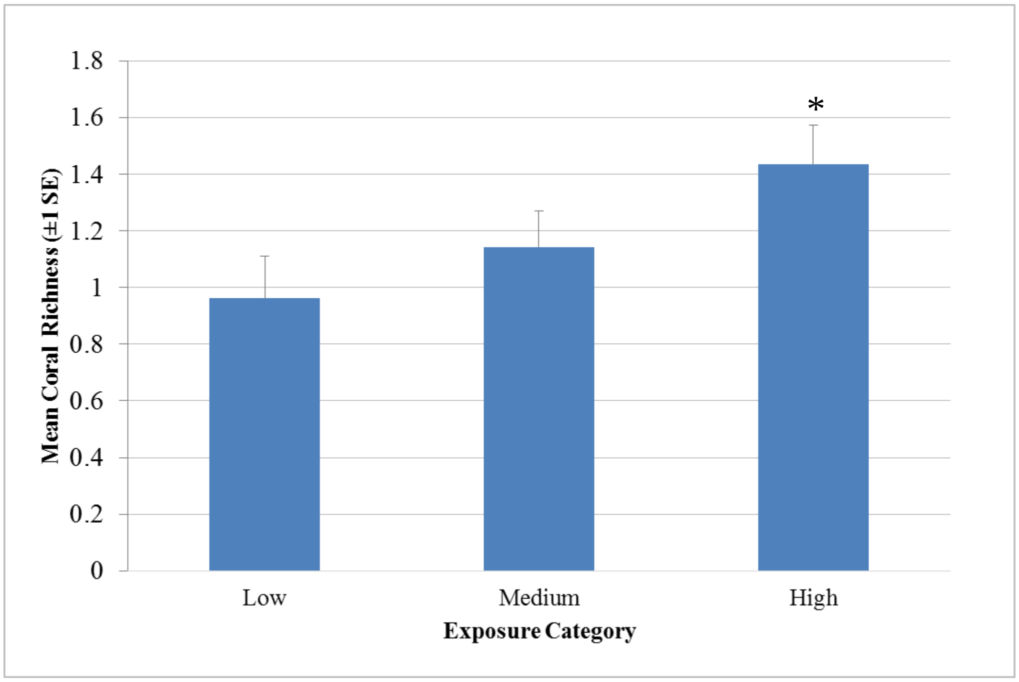


Figure 28. Mean coral richness by exposure category. Error bars indicate ±1 standard error of the mean. Asterisk indicates a significant result.

3.7 New Hardbottom Designations

The hardbottom in Palm Beach County was previously defined into three designations (Near shore, Intermediate, and Offshore) based on different depth zones (0-4 m, 4-8 m, and >8 m) and general distances from shore and did not consider exposure frequency. Many of the community differences were found in areas of high exposure. Multivariate analyses of cover showed that high exposure sites were most dissimilar from the low exposure sites and that medium and low were not very different overall (Table 11). Hard coral and octocoral density and hard coral richness were also significantly higher in high exposed sites while medium and low were not different from each other. Furthermore, Macroalgae and hydroid cover were significantly highest at high exposure sites. Therefore, two exposure classes were created for the new stratification, Low and High. The high exposure class equated to the hardbottom being mapped (exposed) greater than 60% of the time since 2000. Everything else was considered Low exposure.

The community data indicated that depth affected the benthic community composition as well. Octocorals were found in depths ranging from 6.7 to 30.1 feet (2 to 9.2 m) (Figure 29) however, 94% of them were deeper than 9 ft (2.7 m). Large octocorals (>10 cm) occurred between 6.7 and 30.1 ft, but 99% of them (403) occurred deeper than 9 ft (2.7 m). *Siderastrea spp.* < 5 cm were found from 0.25 to 29.6 feet (0.07 to 9 m) in this study, however 80.6% were found shallower than 12 ft (3.7 m) (Figure 30). All octocoral types were found in both depths; however, fans were only found in less than 9.8 feet (3 m) depth at one site.

Model	**AICc**	**BIC**	**SSE**	**MSE**	**RMSE**	**R-Square**
Linear —	509.62174	516.91438	1280.616	14.229066	3.7721435	0.7939804

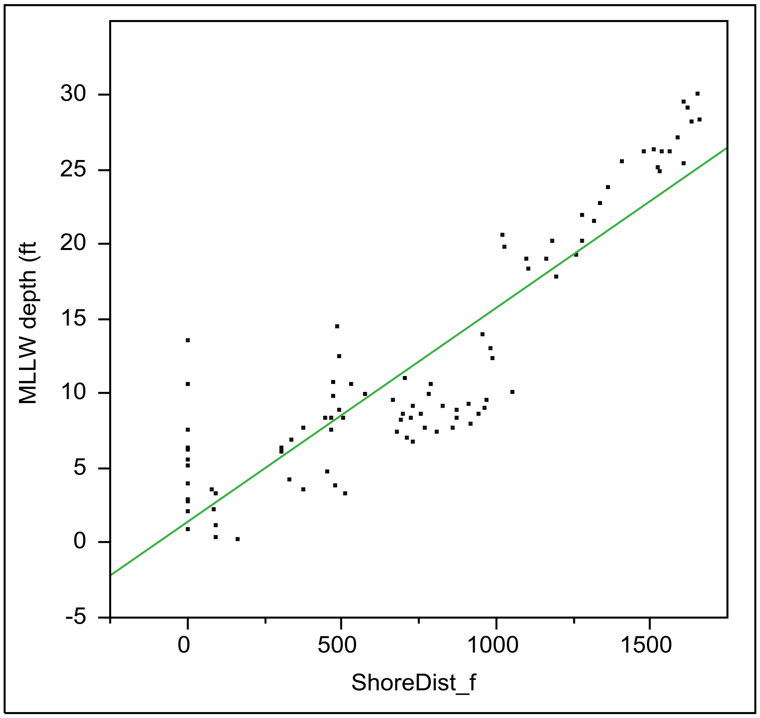


Figure 29. Linear regression of depth versus distance to shore at the BMA groundtruthing sites.

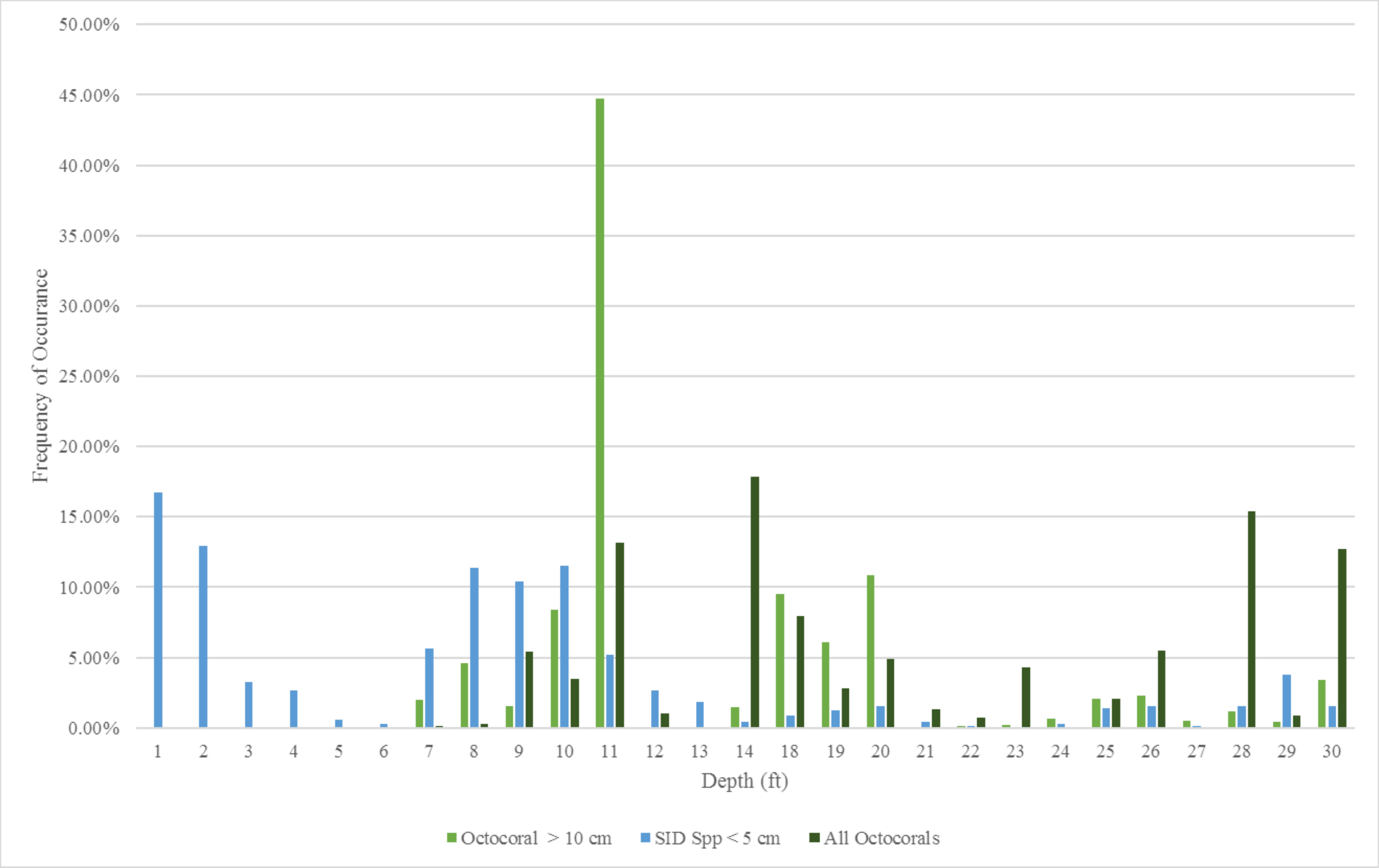


Figure 30. Frequency of all octocorals, octocorals >10 cm and Sid spp < 5 cm by site depth in feet of sites where they were found.

The benthic data analyses showed that 3 m was a point where many community metrics changed. Octocoral density was higher at sites deeper than 3 m, especially octocorals >10 cm in height, whereas small *Siderastrea spp*. density was lower on deeper sites. Additionally, five of the seven coral species were only found deeper than 3 m. Therefore, the data were also stratified by the 3m depth contour.

The new hardbottom designation stratification defined by the community data were Shallow High Exposure, Shallow Low Exposure, Deep High Exposure, and Deep Low Exposure. Using this classification, low exposure sites had the lowest mean density of all species/morphology by size class and high exposure had the highest (Figure 31, 32, Table 14, 15).

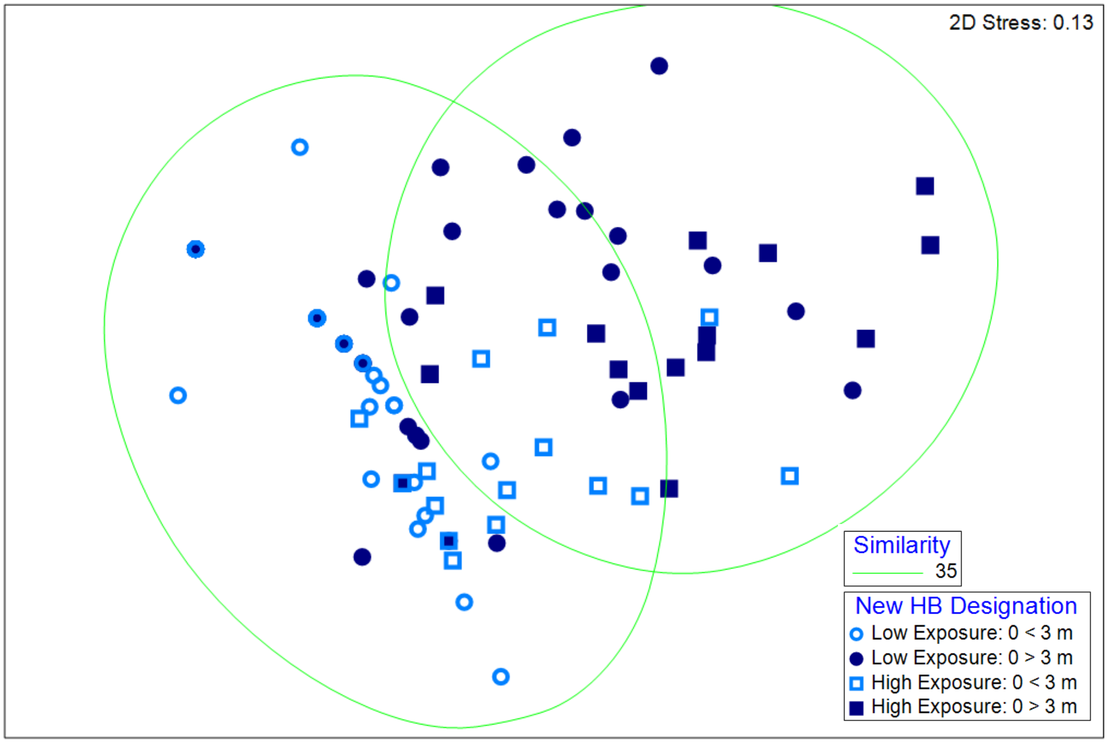


Figure 31. Multidemensional scaling plot of density belt transect data by site categorized by Proposed Hardbottom Designation.

Table 14. Analysis of similarity results testing Proposed hardbottom designation classes by coral and octocoral density and size class data.

Pairwise Tests

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Low Exposure: 0 < 3 m, Low Exposure: 0 > 3 m	**0.21**	**0.1**	**Very large**	**999**	**0**
Low Exposure: 0 < 3 m, High Exposure: 0 < 3 m	**0.386**	**0.1**	**Very large**	**999**	**0**
Low Exposure: 0 < 3 m, High Exposure: 0 > 3 m	**0.68**	**0.1**	**Very large**	**999**	**0**
Low Exposure: 0 > 3 m, High Exposure: 0 < 3 m	0.073	10.7	Very large	999	106
Low Exposure: 0 > 3 m, High Exposure: 0 > 3 m	**0.172**	**0.3**	**Very large**	**999**	**2**
High Exposure: 0 < 3 m, High Exposure: 0 > 3 m	**0.294**	**0.1**	**77558760**	**999**	**0**

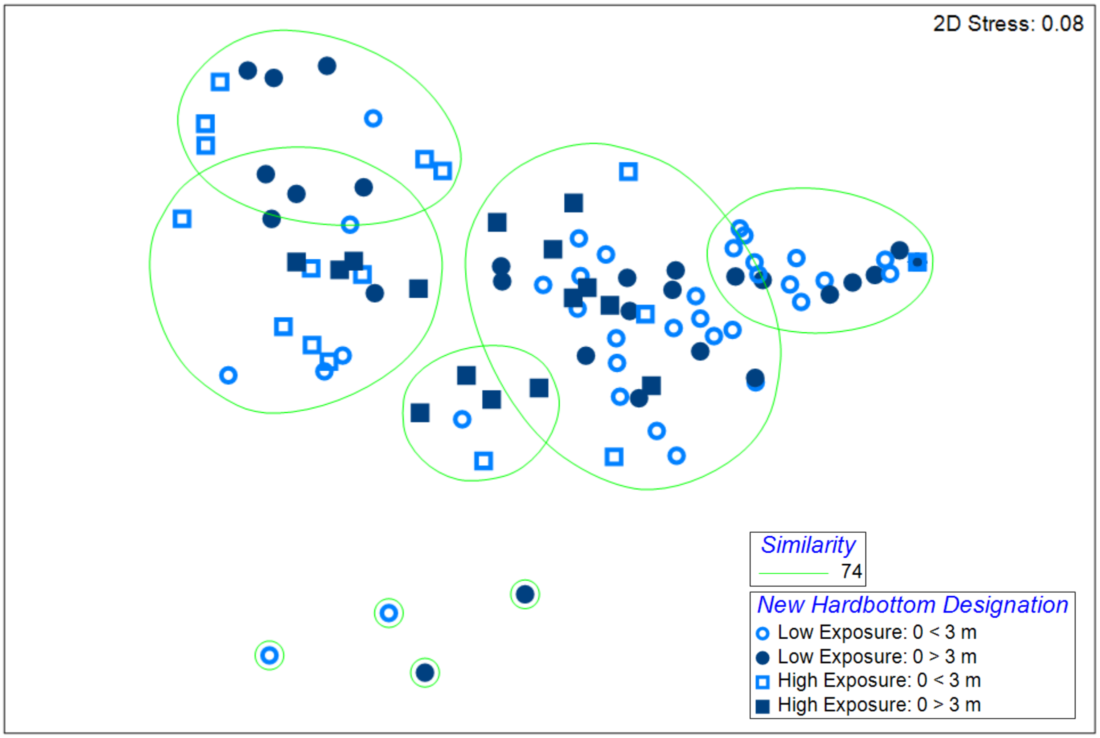


Figure 32. Multidemensional scaling plot of benthic cover data by site categorized by Proposed Hardbottom Designation.

Table 15. Analysis of similarity results testing Proposed Hardbottom Designation classes by benthic cover site data.

Pairwise Tests					
Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
Low Exposure: 0 < 3 m, Low Exposure: 0 > 3 m	**0.053**	**4.1**	**Very large**	**999**	**40**
Low Exposure: 0 < 3 m, High Exposure: 0 < 3 m	**0.378**	**0.1**	**Very large**	**999**	**0**
Low Exposure: 0 < 3 m, High Exposure: 0 > 3 m	**0.28**	**0.1**	**Very large**	**999**	**0**
Low Exposure: 0 > 3 m, High Exposure: 0 < 3 m	**0.15**	**0.6**	**Very large**	**999**	**5**
Low Exposure: 0 > 3 m, High Exposure: 0 > 3 m	0.069	9.7	Very large	999	96
High Exposure: 0 < 3 m, High Exposure: 0 > 3 m	**0.104**	**2.5**	**5.66E+08**	**999**	**24**

Significant relationships were found in the community data between the new hardbottom designation classes. Mean percent cover of sand was higher in the low exposure shallow and deep than in the high exposure shallow and deep (ANOVA $p < 0.0003$) Hydroids were moderately significantly different between low exposure sites (ANOVA 0.0118) and significantly different between low shallow and high deep (ANOVA $p < 0.0015$). Macroalgae cover was highest in high shallow exposure sites, followed by high deep. High shallow and deep were significantly different from low shallow (ANOVA $p < 0.0001$) and high deep was significantly different from low deep (ANOVA; $p < 0.0001$) (Figure 33).

Octocoral cover also had significant relationships between the new designation classes. Rod density was significantly higher in the high deep exposures (ANOVA $p < 0.0001$) Plumes were also significantly higher in the high deep class (ANOVA $p < 0.0001$). Plume density in the low deep to the high shallow were not significant (Figure 34). *Siderastrea spp*. density was highest in the high exposure shallow class (ANOVA p

< 0.0001). *Siderastrea spp*. density in high deep and shallow was significant from that of low deep and shallow (ANOVA p < 0.0001). *Siderastrea spp*. density was not significant between low shallow and deep (Figure 35).

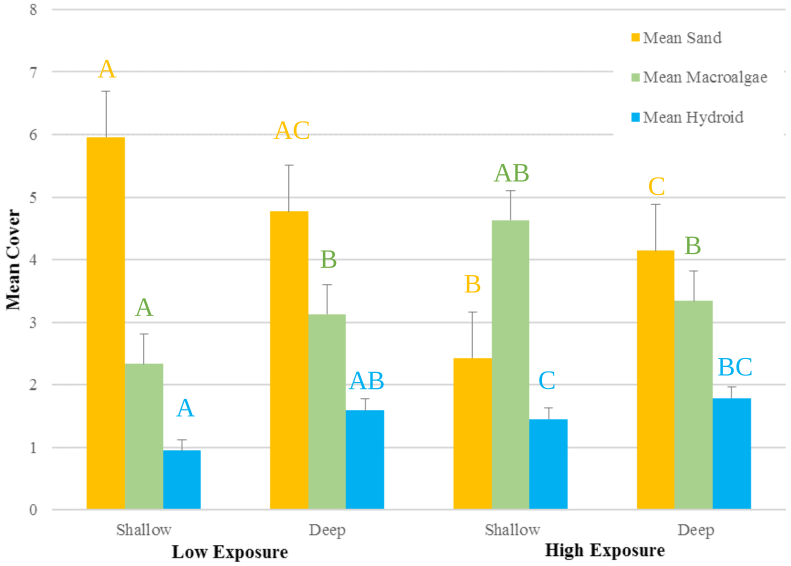


Figure 33. Belt transect data by the new hardbottom designation classes. Low and high shallow (0 < 3 m) and low and high deep (0 > 3 m).

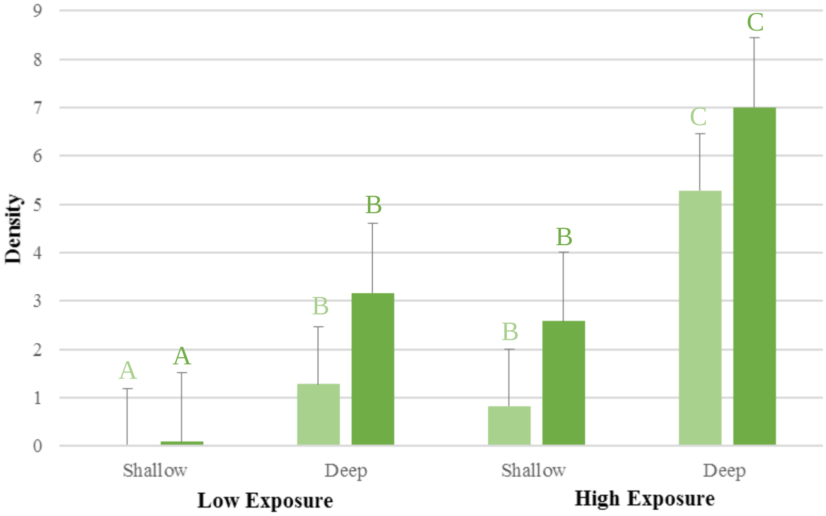


Figure 34. Octocoral cover data by the new hardbottom designation classes. Low and high shallow (0 < 3 m) and low and high deep (0 > 3 m).

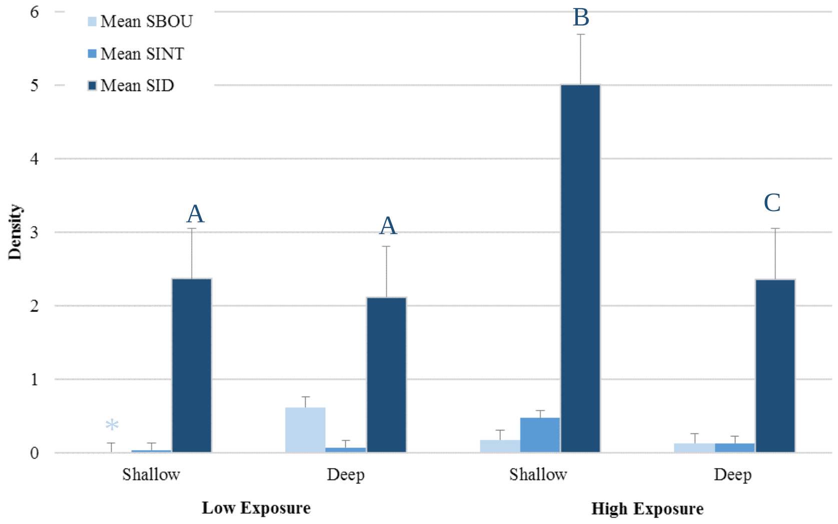


Figure 35. Top three coral cover by the new hardbottom designation classes. Low and high shallow (0 < 3 m) and low and high deep (0 > 3 m).

4. Discussion

Environments that are subjected to frequent disturbance are expected to consist of habitats with smaller organisms, less diversity and more species present that exhibit R- life strategies (Walker and Alberstadt, 1975). We have seen that the hardbottom habitats in Palm Beach are continually changing, indicating they are disturbance driven with constant cycles of disturbance and recovery. Within these habitats we would expect to find more weedy species and other fast growing organisms in areas that experience higher frequencies of burial. Those areas with less burial events, we would expect to begin to see larger organisms, more stony corals and species exhibiting K- life strategies. Following burial events the first to recolonize a benthic community is usually encrusting algae, macroalgae followed by octocorals and colonization of stony corals (Littler *et al*. 1983; Fairfull and Harriott, 1999). It is important to note that the near shore hardbottom environments we looked at are frequently disturbed, and communities would typically never reach climax communities. Also in this study we do not know the time series of the burial and exposure events, but communities are reminiscent of what you would expect to observe.

4.1 Community Structure

Cycles of burial and exposure both prevent and promote new growth, influencing the structure and complexity of all species associated with the near shore hardbottom (Sousa, 2001). Exposed substrate is essential to host benthic communities with sessile organisms because it provides substrate for colonization (Street *et al*. 2005). Due to the higher energy environment, sedimentation can be elevated on near shore hardbottom habitats, especially those with low relief. This cycle of exposure and burial on near shore hardbottom habitats influences the structure and complexity of benthic communities. The disturbance caused by burial can result in mortality of sessile organisms and prevents new organisms from settling. Once the hardbottom becomes exposed, the succession of coral communities begins. The more frequent burial events occur the less opportunity for organisms to settle. When burial events occur on younger communities less sediment is required to bury the recruits, possibly affecting recruit survival (Babcock and Davies, 1991). In my thesis, benthic community structure and complexity measured by diversity and size of corals, octocorals, sponges and macroalgae were tested against exposure rates over time and by depth to determine the effect of sediment burial on the frequently disturbed near shore communities. Areas with higher rates of exposure (less burial) were expected to have more complex successive communities.

Benthic communities mostly aligned with exposure rates with only a few expectations. The highest number of coral species and octocoral morphologies were found at high exposure sites (Figure 25). Mean species richness was also significantly higher at high exposure sites; signifying sites that were more often exposed did in fact have more established coral communities. Additionally, mean sand cover was lower in areas of high exposure as compared to that of areas of low exposure, indicating exposure categories were relatively accurate.

Siderastrea spp. were the most dominant scleractinian throughout the survey sites accounting for 92.1% of colonies found. They were also found in significantly higher densities at medium and high exposure sites (Figure 24). Throughout the northern portion of the Florida Reef Tract (nFRT), *Siderastrea* spp. are abundant in both marginal and disturbed environments (Moyer *et al.* 2003; Lirman and Manzello, 2009; Stein, 2012). *Siderastrea radians* are typically classified as stress-tolerant and an early successional species due to their life strategy; brooding, small size of maturity, and high recruitment rates (Lirman and Manzello, 2009). *Siderastrea siderea* are known for their abundance on the nFRT, as well as tolerance to temperature anomalies (Banks *et al.* 2008; Gilliam *et al.* 2013; Walker and Gilliam, 2013; St. Gelais *et al*. 2016). Most of the *Siderastrea* spp. colonies observed (81%) were less than 5 cm in diameter, indicating relatively new growth (Yaughan, 1915; Bak, 1976; Rogers *et al*. 1984; Yan Moorsel, 1988; Chiappone and Sullivan, 1996). The relatively high numbers of *Siderastrea* spp. observed in areas when compared to other species may be due to the ephemeral nature of the near shore hardbottom their life history, and typical abundance on the nFRT (Banks *et al.* 2008).

Coral richness was also highest in high exposure areas. *Montastraea cavernosa,* a massive reef building species with low recruitment rates (Miller and Barimo, 2001; Lirman and Miller, 2003) and *Oculina diffusa* were only found at high exposure sites, further supporting our expectation that areas having less frequent sediment burial have more complex communities. *Porites astreoides* are known to have an opportunistic life strategy in the FRT because they are brooders with high levels of recruitment and their relatively small size of sexual maturity (Lirman and Miller, 2003). These were only found at medium and high exposure sites.

Octocorals had the highest densities compared to stony corals, macroalgae and sponges observed. This aligns with previous studies that found high octocoral abundance on southeast Florida reefs (Goldberg 1973; Moyer et al. 2003). Gilliam, *et al.* (2013) noted that the Outer Reef in Palm Beach County had the highest density of octocorals in the nFRT and that rod morphotypes were most abundant. In the Palm Beach near shore hardbottom (this study), high exposure sites had significantly higher rod and plume density than low or medium exposure sites. Furthermore, fans were only found in areas of high exposure and encrusting morphologies were found on sites classified as medium and highly exposed. Most octocorals found were less than 25 cm in height, indicating that colonies were not well- established or long-lived (Goldberg, 1973).

Macroalgae was found in every exposure category in between all sites. Of the 546 quadrats assessed on hardbottom, 361 (66%) recorded macroalgae. In sites that were infrequently exposed, you would expect to find algal species that are more resilient to disturbance; usually fast growing with high reproduction rates (FDEP NHB Study; Eriksson and Johansson, 2005; CSA International, Inc. 2009). Dictyota was found most frequently (37.4%) and with the highest mean cover (5.4 %) throughout all the sampled sites which matches previous studies (Foster *et al*. 2006). The success of dictyota is presumably due to its opportunistic r-strategy, with high net photosynthesis and high reproductive rates due to continuous spore release and the ability to disperse via fragmentation (Beach and Walters, 2000; CSA International, Inc. 2009). Low exposure sites had the highest occurrence of dictyota accounting for 79% of benthic macroalgae, which makes sense because dispersion by fragmentation is useful in unstable or changing environments, increasing the likelihood of encountering suitable substrate and decreasing post-settlement mortality (Eriksson and Johansson, 2005).

Furthermore, avrainvillea exhibits k-strategy life histories, which indicates that it is typically slow growing with low productivity rates and that much of its energy is devoted to structural development (CSA International, Inc. 2009). This alga was only found at high exposure sites.

4.1.2 Regional Community Differences

The near shore hardbottom in Palm Beach represents a unique near shore hardbottom community. Although near shore hardbottom exists throughout the northern portion of the nFRT, benthic community structure differs with latitude. Walker (2012) partitioned the nFRT into six regions that were statistically distinct in the number and amount of major benthic habitat types. These regions were supported in Klug (2015), which mapped and evaluated how benthic communities differed along the coast. On the nFRT, near shore hardbottom turf algae cover is higher with increasing latitude, while macroalgae and stony coral cover decreases (Moyer *et al*. 2003; Walker, 2012; Klug, 2015). The number of coral species present on the reef tract decreases from 38 in the

Florida Keys to 9 in Martin County (Banks *et al.* 2008; Walker and Gilliam, 2013). Fish assemblages on the near shore hardbottom differ with latitude and more than 80% of the fish found on these near shore habitats were juveniles (Fisco, 2016).

The southern portion of the nFRT typically sees lower wave energy due to shadowing from the Bahamas banks. It is subtropical in climate, and hosts a higher density and diversity of tropical biota. The northernmost portion of the nFRT transitions to a temperate climate zone (Banks *et al*. 2008; Walker, 2012; Klug, 2015). It is host to successional communities and experiences higher wave energy. It also crosses the Bahamas Fracture Zone, which is the terminus of historical outer reef growth. There the shelf broadens and the Florida Current moves away from the coast, allowing for current meandering that produces strong upwelling in summer months where benthic water temperatures can fluctuate greatly for long periods of time. These temperature fluctuations have been theorized to be the cause of the benthic community differences observed along the nFRT (Banks *et al*. 2008; Walker, 2012; Walker and Gilliam, 2013; Klug, 2015).

The near shore communities in the Biscayne region (southernmost defined coral reef ecosystem region in southeast FL) are defined by its lack of stony coral cover, high density of plume octocorals and most notably by the presence of sea grass, which is dominated by *Thalassia testudinum* and *Syringodium filiforme* (Klug, 2015). This is the only occurrence of seagrass in the nFRT. The Biscayne region is also host to the widest section of near shore hardbottom. Moving northward, the hardbottom in the Broward-Miami region is characterized by the presence of the reef building coral *Acropora cervicornis* and the high density of *Porites astreoides* and rod octocoral densities (Klug, 2015). Broward-Miami had the largest area of near shore hardbottom (49.31km^2) and the highest density of corals.

Our study area crossed the north and south Palm Beach regions derived in Walker (2012). The South Palm Beach region had the second lowest occurrence of near shore hardbottom which was the closest to shore in comparison to hardbottom of all other regions, making it more vulnerable to anthropogenic activity and wave action. This section of Palm Beach hardbottom is isolated due to the vast expanses of unconsolidated sediments. The North Palm Beach region is the shallowest and narrowest occurrence of near shore hardbottom in the nFRT (Walker, 2012; Klug, 2015). It is also noted that the North Palm Beach region’s hardbottom is made up of accretionary ridges of coquina mollusks and tube-building polychaete worms, unlike the hardbottom further south (Banks *et al.* 2008).

4.1.3 New hardbottom designations

Many factors affect benthic communities like wave energy, depth, light levels, temperature, relief and turbidity. Several of these often co-vary with depth. For example, wave energy, light levels, and temperature usually decrease with increasing depth, whereas distance from shore increases with increasing depth (Figure 19). Other local studies have found relationships between reef communities and depth (Walker, Riegl and Dodge, 2008; Walker, Jordan and Spieler, 2009; Walker, 2012; Walker and Gilliam, 2013). Although we do not know the true causative factor(s) controlling the communities, depth can be used as a surrogate to investigate differences.

The previously derived hardbottom designations (near shore, intermediate, and offshore) did not coincide with the community data (Section 3.4.1). Multivariate analyses showed low similarity in benthic community data with previous designations; therefore, the current hardbottom designations are a poor stratification of the benthic communities. The community data had a stronger relationship resulting in a stratification in benthic communities when looking at high exposure categories at the <3m depth contour (Figure 32; Table 14). Therefore, a new Hardbottom Designation classification is proposed that stratifies the hardbottom habitats shallower and deeper than 3 m that have at least 60 % exposure or not. This modification changes the designation to four classes: Shallow High Exposure, Shallow Low Exposure, Deep High Exposure, and Deep Low Exposure. I recommend that this stratification is used in all subsequent study planning, site selections, and data analyses.

4.2 Detecting reef burial/exposure through remote sensing

Near shore hardbottom communities are a large part of the southeast Florida coral benthic ecosystem (Chiappone and Sullivan, 1994). They play an essential role acting as habitat, settlement sites, nesting and spawning sites, nursery areas for juvenile fish, feeding grounds and shelter for many species including the listed green turtle (CSA, 2009). In Palm Beach County, the near shore hardbottom is a comparatively small area with low relief. It is found in shallow depths, close to the beach, and surrounded by vast amounts of motile sediments. The surrounding sediments are continually shifting driven by waves and currents causing these low relief hardbottom habitats to be buried and exposed through natural processes of sediment transport (Street *et al.* 2005; Díaz-Ortega and Hernández-Delgado, 2014). Periodic hardbottom burial can impede the growth and development of hardbottom communities. The impacts from cross-shore and longshore sediment transport are heightened in areas like Palm Beach that have fixed inlets or at fabricated structures, which sit perpendicular to shore and impede natural littoral processes (Dean *et al.* 2013).

My study showed that periodic mapping from aerial photographs and manual delineation can identify hardbottom burials and exposures that fluctuate between years and relate to benthic community differences. Periodic mapping using remotely sensed imagery has been shown useful in identifying significant changes in area of coral reefs at regional scales (Shapiro and Rohmann 2005; Moufaddal 2005; Hedley *et al*. 2016). These techniques prove effective in change detection on near shore coral reef habitats. For example, imagery from 1984 and 2000 were used to assess impacts from shoreline restoration and re-nourishment. Burial from infill resulted in a total loss coral reefs in the inshore reef zone. Change detection coupled with field observations we able to accurately display the impact to the inshore reef zone (Moufaddal, 2005). The use of remote sensing coupled with manual delineation allows the study of extensive expanse of coral reef habitat and assess temporal patterns in a region (Hedley *et al*. 2016). Using remotely sensed imagery, the distribution and total area of near shore hardbottom habitats can be determined along with the change in exposed hardbottom through time, and the impact of disturbance to the hardbottom ecosystems when coupled with *in situ* measurements.

However, it is difficult to know the actual exposure frequency through time given the dynamic nature of the near shore environment and the infrequency of mapping. The mapping frequency affects the relationship between the remotely measured exposure and the benthic community data. Better relationships between exposure and community structure require seasonal mapping because the south Florida coast has intermediate beaches that are geomorphologically affected seasonally. Sediment moves toward the beach during lower energy summer months, and is washed out during the higher energy winter months. (Benedet et al. 2004; Absalonsen and Dean, 2011). Thus, dates and timing of imagery acquisition are crucial to assessing change detection (Choppin, 1996). Although relationships between exposure and benthic communities were evident, the sporadic dates of image acquisition did not allow for the assessment of seasonal hardbottom exposure and thus did not capture this seasonal variability of sand movement. Images collected in July 2014, November 2014 and May 2015 were used to evaluate such fluctuations. There was little change in the overall calculated area of exposed hardbottom between July 2014 and November 2014, but a steep decline from November to May. May 2015 also showed the third lowest area of exposed hardbottom since 2000, with just 188.62 acres, with the lowest recorded hardbottom exposed July 2003 at 171.56 acres. Large differences in exposure in relatively short time frames were observed. For example, an area of hardbottom east of R-136 exposed in July 2014 was completely buried in November (Figure 37). The variability in exposure was missed when hardbottom exposure was mapped annually. Because the benthic communities advance in the succession depending on the time of exposure and seasonal changes are significant, annual mapping does not capture or quantify the hardbottom exposure very well and is likely the reason stronger relationships between the benthic community and exposure weren’t found. Conducting a winter and summer survey would greatly improve the understanding of these relationships.

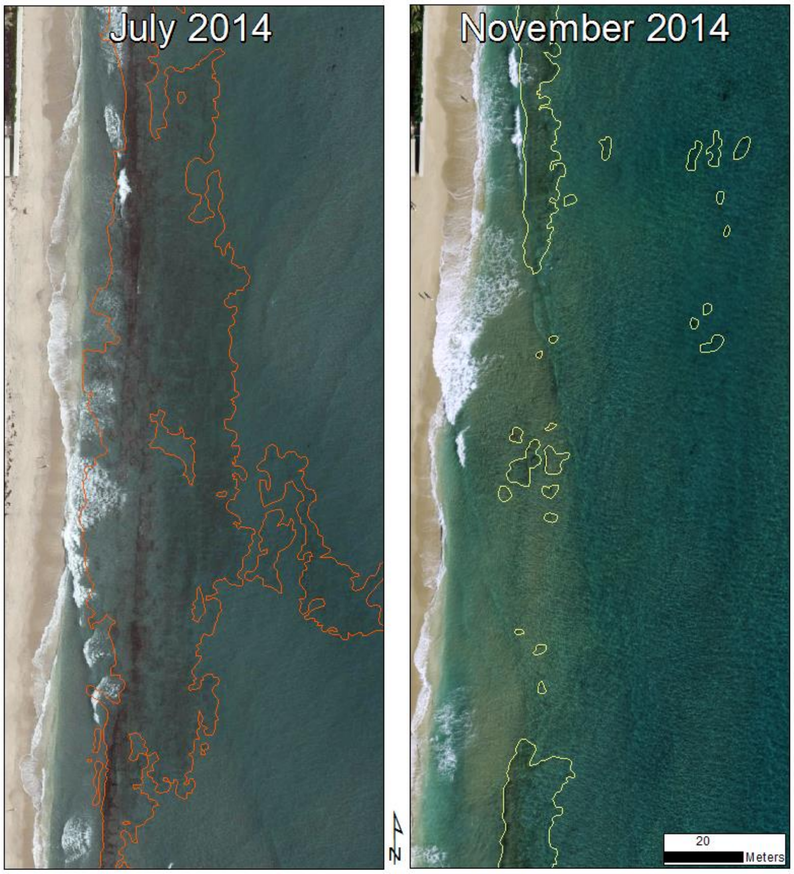


Figure 37. Near shore hardbottom east of R-136 that was exposed in July 2014 and completely buried in November 2014. Here the delineation of the exposed hardbottom in July 2014 is shown over the November 2014 imagery, where no hardbottom is exposed.

When monitoring near shore hardbottom ecosystems, it is often difficult to differentiate natural disturbance and anthropogenic impacts (Chiappone and Sullivan, 1994). Sediment movement may coincide with either thus, the frequency of mapping should also consider anthropogenic activities and major storm events. Several variables including seasonality could account for the decrease between November 2014 and May 2015, but it is important to note the Palm Beach Midtown Dredge and nourishment Project deposited approximately 800,000 cy. in Reaches 3 and 4 encompassing 2.4 miles of shoreline from January through April in 2015 (Palm Beach County Shoreline protection plan Environmental Enhancement & Restoration Division, 2014). Reach 2 decreased 3.63 acres from July 2014 to November 2014 and then another 10.75 acres from November to May. Reach 3 initially increased 3.96 acres from July to November, but then lost 5.29 acres from November to May. Reaches 4, 5 and 6 showed no significant changes throughout the year (Table 19). A dune restoration encompassing 2.1 miles of shoreline also took place during this period three miles south of the Mid-Town Dredge project throughout Reaches 7 and 8. From November 2014 to May 2015 exposed hardbottom in the reaches south of the Mid-Town Dredge project decreased by 6.42 acres in Reach 8, and 3.71 acres in Reach 9. Hardbottom exposure in Reach 7 increased 3.98 acres, although total exposure was still 0.68 acres less then what was recorded in July 2014.

Previous years showed similar patterns with other nourishment projects (Figure 38). October 2010 – October 2011 incurred a loss of 39.45 acres following the partial re-nourishment and dune restoration at Mid-Town Palm Beach, the Phipps Ocean Park restoration (Reach 7), and Reach 8 Dune restoration and partial nourishment project where a total of 189,000 cubic yards of sediment was added. Exposed hardbottom area loss succeeded all beach nourishment projects with the exception of May 2011, which saw an initial gain, followed by a loss in October. The Mid-Town Beach Expansion Project (Reaches 3 & 4) has had one full-scale re-nourishment in 2003, a hurricane restoration project in 2006, and a dune restoration in 2015. The Phipps Ocean Park project (Reach 7 and 8) first took place in 2006 and again in 2011 (Palm Beach County Shoreline protection plan Environmental Enhancement & Restoration Division, 2014).

Even though beach nourishment is correlated with negative impacts to the near shore hardbottom environment (Banks *et al.* 2008; CSA, 2009; Gilliam *et al.* 2013), it is a necessary practice to maintain our beaches, keep the southeastern Florida tourism economy strong and protect real estate (Absalonsen and Dean, 2011). However, it is imperative that managers balance the financial need to maintain a strong economy as well as a healthy marine environment (Smith *et al*. 2007). The severity of dredging impacts on near shore coral communities depends on the intensity, frequency, and duration of sedimentation (Erftemeijer *et al*. 2012).

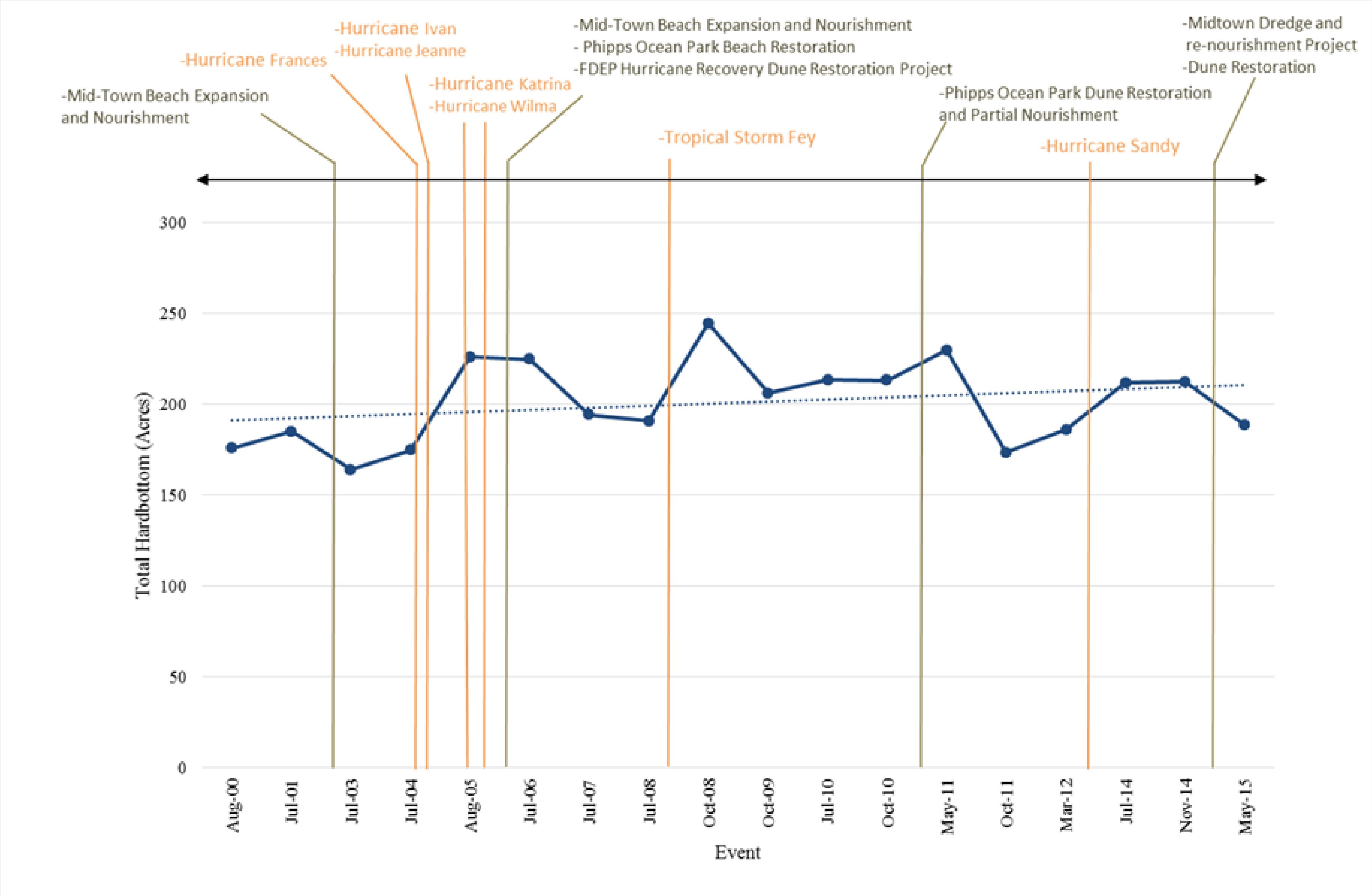


Figure 38. Total hardbottom exposure by year in relation to beach nourishment projects and major storm events, adapted from previous studies by Cheryl Miller and CEG.

Major storms which can greatly affect shorelines and sediment loads on near shore environments in over short time periods (Miller and Kosmynin 2008; CSA, 2009; Absalonsen and Dean, 2011) coincided with massive sediment movement in the imagery. Since 2000, Palm Beach was directly affected by hurricanes Frances (Category 2) and Jeanne (Category 3) in 2004 and Hurricane Wilma (Category 3) in 2005 (Palm Beach County Department of Environmental Resources Management Environmental Enhancement & Restoration Division, 2014). An increase in exposed near shore hardbottom area coincided with the mapped hardbottom after each storm, with the exception of hurricane Wilma that was followed by a drop in 2 acres of exposed hardbottom. In 2008, hardbottom exposure increased dramatically following Tropical Story Fey. Fey impacted Palm Beach August 19 2008, between two imagery capture dates July 2008 and October 2008. The exposed hardbottom increased from 190.61 acres to 244.43 acres from July to October. Between these dates, reaches in the northern most region (2-6) gained the greatest area of exposed hardbottom. In October 2008, Reach 3 documented its highest exposure; 81.21 acres, which is significantly higher than the average, exposed hardbottom in Reach 3 of 52.56 acres (Table 1 & 2). To accurately capture the affects natural and anthropogenic events have on the burial and exposure of hardbottom in Palm Beach, FL, mapping frequency should be modified to include assessments after major storms and planned construction.

5. Conclusions

Palm Beach Florida is host to a unique shallow near shore hardbottom ecosystem. This study confirmed that periodic mapping with manual delineation did identify hardbottom burials and exposures that fluctuate between years and relate to benthic community differences. These techniques prove effective in change detection on hardbottom habitats. Large differences in exposure were seen in relatively short periods, and most of the aerial imagery did not effectively capture the seasonal variability of sediment movement throughout each year. Change in exposed hardbottom can be seen through time, and the impact of disturbance when coupled with *in situ* measurements can be determined however, it is difficult to know the actual exposure frequency through time given the dynamic nature of the near shore environment and the infrequency of mapping. The more frequently the mapping is conducted, the better our understanding of hardbottom exposure will be, allowing for more predictable relationships between exposure and the benthic communities.

The near shore hardbottom coral reef communities of Palm Beach, Florida, did indeed align with the observed exposure categories with the highest number of coral species and octocoral morphologies found at sites classified as highly exposed, however our classifications were not perfect. It was also noted that the current hardbottom designations of near shore, intermediate, and offshore did not represent the striations of communities, and instead a depth limit of <3m and >3m would be more representative of the observed community differences. This study also was successful in creating baseline data of the near shore hardbottom community structure and composition in Palm Beach, Florida.

Anthropogenic activities were found to coincide with the decrease of near shore hardbottom exposure, while major storms seemed to greatly increase the observed exposed hardbottom. Seasonal and targeted imagery collection after known mass sediment movement events will help hone the near shore hardbottom areas affected by sediment burial and exposure.

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Literature Cited

Absalonsen, Luciano, and Robert G. Dean. 2011. "Characteristics of the shoreline change along Florida sandy beaches with an example for Palm Beach County." Journal of Coastal Research 27, no. 6A: 16-26.

Aronoff, Stanley. 2005. "Visual Interpretation of Aerial Imagery." Remote Sensing for GIS Managers. Redlands: ESRI: 260-283. Print.

Babcock, R., and L. Smith. 2002. "Effects of sedimentation on coral settlement and survivorship." In Proceedings of the Ninth International Coral Reef Symposium, Bali, 23-27 October 2000, vol. 1, pp. 245-248.

Babcock, Russ, and Phil Davies. 1991."Effects of sedimentation on settlement of Acropora millepora." Coral Reefs 9, no. 4: 205-208

Baillie JM, Hilton-Taylor C, Stuart SN. 2004. IUCN red list of threatened species. A global species assessment. IUCN, Gland, Switzerland and Cambridge, UK

Banks, K. W., B. M. Riegl, E. A. Shinn, W. E. Piller, and R. E. Dodge. 2007. "Geomorphology of the southeast Florida continental reef tract (Miami-Dade, Broward, and Palm Beach counties, USA)." Coral Reefs 26, no. 3: 617-633.

Banks, Kenneth W., Bernhard M. Riegl, Vincent P. Richards, Brian K. Walker, Kevin P. Helmle, Lance KB Jordan, Janet Phipps, Mahmood S. Shivji, Richard E. Spieler, and Richard E. Dodge. 2008. "The reef tract of continental southeast Florida (Miami-Dade, Broward and Palm Beach counties, USA)." In Coral Reefs of the USA, pp. 175-220. Springer Netherlands

Beach, K.S. and Walters, L.J., 2000. Dictyota bloom in the Florida Keys National Marine Sanctuary: fragments and fouling. In AAUS 20th Symposium pp. 61-63.

Benedet, Lindino, Charles W. Finkl, Thomas Campbell, and Antonio Klein. 2004. "Predicting the effect of beach nourishment and cross-shore sediment variation on beach morphodynamic assessment." Coastal Engineering 51, no. 8: 839-861.

Blair, S. M. and Flynn, B. S. 1989. Biological monitoring of hardbottom reef communities off Dade County Florida: community description. In Land M, Jaap W (eds), Proc Am Ac Underwater Science 9th Annual Science Diving Symposium, Woods Hole, MA, 9–24

BMA (Cell-Wide Monitoring & Mitigation Plans) 2013. “Palm Beach Island Beach Management Agreement” Florida Department of Environmental Protection

Chiappone, M., and K. M. Sullivan. 1994. "Ecological structure and dynamics of near shore hardbottom communities in the Florida Keys." Bulletin of Marine Science 54, no. 3: 747- 756.

Chiappone, M., & Sullivan, K. M. 1996. “Distribution, abundance and species composition of juvenile scleractinian corals in the Florida reef tract.” Bulletin of Marine Science, 58(2), 555-569.

Colosio, F., M. Abbiati, and L. Airoldi. 2007."Effects of beach nourishment on sediments and benthic assemblages." Marine pollution bulletin 54, no. 8: 1197-1206.

Connell, J.H. and Slatyer, R.O. 1977. “Mechanisms of succession in natural communities and their role in community stability and organization.” The American Naturalist, 111(982), pp.1119-1144.

Coppin, P. R., & Bauer, M. E. 1996. Digital change detection in forest ecosystems with remote sensing imagery. Remote sensing reviews, 13(3-4), 207-234.

Coyne, M. S., T. A. Battista, M. Anderson, J. Waddell, W. Smith, P. Jokiel, M. S. Kendall, and M. E.Monaco. 2003. Benthic Habitats of the Main Hawaiian Islands, 1–49. Silver Spring, MD: NOAA Technical Memorandum NOS NCCOS CCMA 152, Biogeography Branch.

Costa, B.M., Kendall MS, Edwards K, Kagesten G, Battista T. 2013. Benthic Habitats of Fish Bay, Coral Bay and the St. Thomas East End Reserve. NOAA Technical Memorandum NOS NCCOS 175. Silver Spring, MD. 80 pp.

Costa B, Walker BK, Dijkstra J. In press. “Spatial patterning in the sea: Mapping and quantifying seascape patterns.” In: Seascape Ecology: Taking landscape ecology into the sea. Pittman SJ ed. Chapter 3.

CSA International, Inc. 2009. Ecological functions of near shore hardbottom habitat in east Florida: A literature synthesis. Prepared for the Department of Environmental Protection Bureau of Beaches and Coastal Systems, Tallahassee, FL. 186 pp. + apps.

Dean, R. G., T. L. Walton, J. D. Rosati, and L. Absalonsen. 2013."Beach erosion: causes and stabilization." In Coastal Hazards, pp. 319-365. Springer Netherlands.

Dean, Robert G., Renjie Chen, and Albert E. Browder. 1997. "Full scale monitoring study of a submerged breakwater, Palm Beach, Florida, USA." Coastal Engineering 29, no. 3: 291- 315.

Díaz-Ortega, G. and Hernández-Delgado, E.A., 2014. Unsustainable land-based source pollution in a climate of change: a roadblock to the conservation and recovery of Elkhorn Coral Acropora palmata (Lamarck 1816). *Natural Resources 2014.*

Dodge, Richard E., and J. Rimas Vaisnys. 1977. "Coral populations and growth patterns: responses to sedimentation and turbidity associated with dredging." Journal of Marine Research 35, no. 4: 715.

Dompe, Philip E. 1993."Natural fluctuations in near shore turbidity and the relative influences of beach renourishment." PhD diss., University of Florida.

Duarte, Carlos M. 2000. "Marine biodiversity and ecosystem services: an elusive link." Journal of experimental marine Biology and Ecology 250, no. 1: 117-131.

Erftemeijer, Paul LA, Bernhard Riegl, Bert W. Hoeksema, and Peter A. Todd. 2012. "Environmental impacts of dredging and other sediment disturbances on corals: a review." Marine Pollution Bulletin 64, no. 9: 1737-1765.

Eriksson, Britas Klemens, and Gustaf Johansson. 2005. "Effects of sedimentation on macroalgae: species-specific responses are related to reproductive traits." Oecologia 143, no. 3: 438- 448.

Fabricius, K., and G. De'Ath. 2001."Environmental factors associated with the spatial distribution of crustose coralline algae on the Great Barrier Reef." Coral Reefs 19, no. 4: 303-309

Fairfull, S.J.L. and Harriott, V.J., 1999. “Succession, space and coral recruitment in a subtropical fouling community.” Marine and Freshwater Research, 50(3), pp.235-242.

Finkl Jr, C. F., J. Walker, and I. Watson. 1988."Shoreline erosion: management case history from southeast Florida." Ocean and Shoreline Management 11, no. 2: 129-143.

Fisco, Dana P. 2016. "REEF FISH SPATIAL DISTRIBUTION AND BENTHIC HABITAT." Thesis. Nova Southeastern. Print.

Fisher, L., K. Banks, D. Gilliam, R.E. Dodge, D. Stout, B. Vargas-Angel, B.K. Walker. 2008. "Real- time coral stress observations before, during, and after beach nourishment dredging offshore SE Florida." Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, Florida, 7-11 July 2008. Vol. 1: 34-37.

Foster K, Foster G, Riegl B (2006) Reef Mapping Final Report. National Coral Reef Institute for Broward County Environmental Protection Department, Biological Resources Division, 117 pp

Garrido, Katharine. 2007."Observations of Immature Sea Turtles at a Near shore Hardbottom Developmental Habitat in Palm Beach, Florida." PhD diss., Florida Atlantic University.

Gilliam, D.S., Brinkhuis, V., Ruzicka, R., and Walton, C.J., 2013. Southeast Florida Coral Reef Evaluation and Monitoring Project 2012 Year 10 Final Report. Florida DEP Report #RM085. Miami Beach, FL. pp. 53.

Gilmour, J. 1999. "Experimental investigation into the effects of suspended sediment on fertilisation, larval survival and settlement in a scleractinian coral." Marine Biology 135, no. 3 (1999): 451-462.

Goldberg, Walter M. 1973 "The ecology of the coral-octocoral communities off the southeast Florida coast: geomorphology, species composition, and zonation." Bulletin of Marine Science 23, no. 3: 465-488.

Goodman, J., Purkis, S., & Phinn, S. (Eds.). (2013). Coral reef remote sensing: a guide for mapping, monitoring and management (pp. 15-405). Springer.

Harrington, Lindsay, Katharina Fabricius, Geoff Eaglesham, and Andrew Negri. 2005. "Synergistic effects of diuron and sedimentation on photosynthesis and survival of crustose coralline algae." Marine Pollution Bulletin 51, no. 1: 415-427

Harrington, Lindsay, Katharina Fabricius, Glenn De'Ath, and Andrew Negri. 2004. "Recognition and selection of settlement substrata determine post-settlement survival in corals." Ecology 85, no. 12: 3428-3437.

Hedley, J.D.; Roelfsema, C.M.; Chollett, I.; Harborne, A.R.; Heron, S.F.; Weeks, S.; Skirving, W.J.; Strong, A.E.; Eakin, C.M.; Christensen, T.R.L.; Ticzon, V.; Bejarano, S.; Mumby, P.J. 2016. Remote Sensing of Coral Reefs for Monitoring and Management: A Review. Remote Sens. *8*, 118.

Halpern, B.S.; Walbridge, S.; Selkoe, K.A.; Kappel, C.V.; Micheli, F.; D'Agrosa, C.; Bruno, J.F.; Casey, K.S.; Ebert, C.; Fox, H.E.; et al. A global map of human impact on marine ecosystems. Science 2008, 319, 948–952.

Hodgson, Gregor. 1990. "Sediment and the settlement of larvae of the reef coral Pocillopora damicornis." Coral Reefs 9, no. 1: 41-43.

Holloway-Adkins, K. G. 2005. "Green turtles using near shore reefs in Brevard County, Florida as developmental habitat: A preliminary investigation." In Kalb, H.; Rohbe, A.; Gayheart, K., and Shanker, K.(Compilers). Proceedings of the 25th Annual Symposium on Sea Turtle Biology and Conservation, p. 92.

Hubbard, Dennis K. 1986. "Sedimentation as a control of reef development: St. Croix, USVI." Coral Reefs 5, no. 3: 117-125.

Jordan, L.K.B., K. W. Banks, L. E. Fisher, B. K. Walker, and D. S. Gilliam. 2010. “Elevated sedimentation on coral reefs adjacent to a beach nourishment project.” Marine Pollution Bulletin 60, no2: 261-271

Klug, Katelyn. 2015. "Cross-Shelf and Latitudinal Benthic Community Investigation in the Near shore Habitats of the Northern Florida Reef Tract." Thesis. Nova Southeastern. Print.

Lindeman, Kenyon C., and David B. Snyder. 1999. "Near shore hardbottom fishes of southeast Florida and effects of habitat burial caused by dredging." Fishery Bulletin 97, no. 3: 508- 5

Lirman, D. and Manzello, D., 2009. Patterns of resistance and resilience of the stress-tolerant coral Siderastrea radians (Pallas) to sub-optimal salinity and sediment burial. Journal of Experimental Marine Biology and Ecology, 369(1), pp.72-77.25.

Lirman, D. and Miller, M.W., 2003. Modeling and monitoring tools to assess recovery status and convergence rates between restored and undisturbed coral reef habitats. Restoration Ecology, 11(4), pp.448-456.

Littler, Mark M., and Diane S. Littler. 1980 "The evolution of thallus form and survival strategies in benthic marine macroalgae: field and laboratory tests of a functional form model." American Naturalist 25-44.

Littler, Mark M., and Keith E. Arnold. 1982 "Primary Productivity of Marine Macroalagal Functional-Form Groups from Southwestern North America." Journal of Phycology 18, no. 3 307-311.

Littler, Mark M., Diane S. Littler, and Phillip R. Taylor. 1983. "Evolutionary Strategies in a Tropical Barrier Reef System: Functional- Form Groups of Marine Macroalgae." Journal of Phycology 19, no. 2: 229-237.

Loya, Y. 1976. "Effects of water turbidity and sedimentation on the community structure of Puerto Rican corals." Bulletin of Marine Science 26, no. 4: 450-466.

Maeder, J., S. Narumainai, D. C. Rundquist, R. L. Perk, J. Schalles, K. Hutchins, and J.Keck. 2002. “Classifying and Mapping General Coral-Reef Structure Using IKONOS Data. “Photogrammetric Engineering and Remote Sensing 68: 1297–1305.

Makowski, Christopher, Jeffrey A. Seminoff, and Michael Salmon. 2006. "Home range and habitat use of juvenile Atlantic green turtles (Chelonia mydas L.) on shallow reef habitats in Palm Beach, Florida, USA." Marine Biology 148, no. 5: 1167-1179.

Miller, M.W. and Barimo, J., 2001. Assessment of juvenile coral populations at two reef restoration sites in the Florida Keys National Marine Sanctuary: Indicators of success?. Bulletin of Marine Science, 69(2), pp.395-405.

Miller, C. and Kosmynin, V., 2008. The effects of hurricane deposited mud on coral communities in Florida. In Proceedings of the 11th International Coral Reef Symposium (Vol. 1, p. 2).

Mishra, D., S. Narumaiani, D. Rundquist, and M. Lawson. 2006. “Benthic Habitat Mapping in Tropical Marine Environments Using Quickbird Multispectral Data.” Photogrammetric Engineering and Remote Sensing 72: 1037–1048.

Moyer, Ryan P., Bernhard Riegl, Kenneth Banks, and Richard E. Dodge. 2003 "Spatial patterns and ecology of benthic communities on a high-latitude South Florida (Broward County, USA) reef system." Coral Reefs 22, no. 4: 447-464.

Moufaddal, W.M., 2005. Use of satellite imagery as environmental impact assessment tool: a case study from the NW Egyptian Red Sea coastal zone. Environmental Monitoring and Assessment, 107(1-3), pp.427-452.

Mumby, P.J., Green, E.P., Clark, C.D. and Edwards, A.J., 1998. Digital analysis of multispectral airborne imagery of coral reefs. Coral Reefs, 17(1), pp.59-69.

Nugues, M. M., and C. M. Roberts. 2003. "Coral mortality and interaction with algae in relation to sedimentation." Coral reefs 22, no. 4: 507-516.

Palandro, D. A., Andréfouët, S., Hu, C., Hallock, P., Müller-Karger, F. E., Dustan, P., & Beaver, C. R. (2008). Quantification of two decades of shallow-water coral reef habitat decline in the Florida Keys National Marine Sanctuary using Landsat data (1984–2002). Remote Sensing of Environment, 112(8), 3388-3399.

Palm Beach County Department of Environmental Resources Management Environmental Enhancement & Restoration Division 2014. “Palm Beach County

Shoreline Protection Plan.” http://www.co.palm beach.fl.us/erm/coastal/shoreline/beach/pdf/shoreline-protection-plan.pdf

Pandolfi, John M., D. Ross Robertson, and David W. Kirtley. 1998. "Roles for worms in reef- building." Coral Reefs 17, no. 2 (1998): 120-120.

PBS&J, 2008. “Best management practices (BMPs) for construction, dredge and fill and other activities adjacent to coral reefs.” Southeast Florida Coral Reef Initiative, Maritime Industry and Coastal Construction Impacts Focus Team. PBS&J, 126pp.

Peterson, Charles H., and Melanie J. Bishop. 2005. "Assessing the environmental impacts of beach nourishment." Bioscience 55, no. 10: 887-896.

Prouty, Nancy G., Curt D. Storlazzi, Amanda L. McCutcheon, and John W. Jenson. 2014."Historic impact of watershed change and sedimentation to reefs along west-central Guam." Coral Reefs: 1-17.

Rogers, Caroline S. 1990. "Responses of coral reefs and reef organisms to sedimentation." Marine Ecology Progress Series. 62, no. 1: 185-202.

Shapiro, A.C. and Rohmann, S.O., 2005. “Summit-to-sea mapping and change detection using satellite imagery: tools for conservation and management of coral reefs.” Revista de biología tropical, 53, pp.185-193.

Sousa, Wayne P. 2001."Natural disturbance and the dynamics of marine benthic communities." Marine community ecology 4: 85-130.

South Atlantic Fishery Management Council (SAFMC) 1998. Final Habitat plan for the South Atlantic region: Essential fish habitat requirements for fishery management plans of the South Atlantic Fishery Management Council. Charleston, SC. 639 pp

St. Gelais, A.T., Chaves-Fonnegra, A., Moulding, A.L., Kosmynin, V.N. and Gilliam, D.S., 2016. Siderastrea siderea spawning and oocyte resorption at high latitude. Invertebrate Reproduction & Development, 60(3), pp.212-222.

Stauble, Donald K. 1993. "An overview of southeast Florida inlet morphodynamics." Journal of Coastal Research, 1-27.

Street, Michael W., Anne S. Deaton, William S. Chappell, and Peter D. Mooreside. 2005. “North Carolina coastal habitat protection plan.” NC Department of Environment and Natural Resources, Division of Marine Fisheries. Ch 7. Pp. 449-472

Jennifer Stein. 2012. “Juvenile Scleractinian Coral Density, Composition, and Influence on the Adult Coral Population in Southeast Florida” Master's thesis. Nova

Southeastern University. Retrieved from NSUWorks, Oceanographic Center. (173)

U.S. Fish and Wildlife Service (USFWS) 1999. South Florida multi-species recovery plan, Atlanta Georgia. Pp. 2,712

Van Dolah, R.F., Wendt, P.H. and Nicholson, N., 1987. Effects of a research trawl on a hardbottom assemblage of sponges and corals. Fish. Res., 5: 39-54.

Vargas-Ángel, B., Thomas, J.D. and Hoke, S.M., 2003. High-latitude Acropora cervicornis thickets off Fort Lauderdale, Florida, USA. *Coral Reefs*, *22*(4), pp.465-473.

Vermeij,M.J.A., 2005. Substrate composition and adult distribution determine recruitment patterns in a Caribbean brooding coral. Marine Ecology Prog. Ser. 295,123–133

Walker, Brian K. 2012. "Spatial analyses of benthic habitats to define coral reef ecosystem regions and potential biogeographic boundaries along a latitudinal gradient." PloS one 7, no. 1

Walker, Brian K., Bernhard Riegl, and Richard E. Dodge. 2008. "Mapping coral reef habitats in southeast Florida using a combined technique approach." Journal of Coastal Research: 1138-1150.

Walker, Brian K., and David S. Gilliam. 2013 "Determining the Extent and Characterizing Coral Reef Habitats of the Northern Latitudes of the Florida Reef Tract (Martin County)." PloS one 8, no. 11

Walker, Brian K., and Klug, K. 2014. “Southeast Florida shallow-water habitat mapping & coral reef community characterization.” Florida DEP Coral Reef Conservation Program report. Miami Beach, FL. Pp. 83.

Walker, Brian K., and Klug, K. 2015. “Southeast Florida Large Coral Assessment 2015” (pp. 151). Miami Beach, FL: Florida DEP Coral Reef Conservation Program report.

Walker, K.R. and Alberstadt, L.P., 1975. “Ecological succession as an aspect of structure in fossil communities.” Paleobiology, pp.238-257.

Walters, L. and Beach, K., 2000. Algal bloom in the Florida Keys. Underwater Naturalist, 25(3), pp.27-29.

Weinbauer, M.G.; Höfle, M.G. 1998. "Distribution and Life Strategies of Two Bacterial Populations in a Eutrophic Lake". Appl. Environ. Microbiol. 64 (10): 3776–3783

Yoshioka, Paul M., and Beverly Buchanan Yoshioka. 1989"Effects of wave energy, topographic relief and sediment transport on the distribution of shallow-water octocorals of Puerto Rico." Coral Reefs 8, no. 3: 145-152.

Appendix

Appendix I.

The classification definitions were taken directly from Section 2a of Appendix B of the BMA (Cell-Wide Monitoring & Mitigation Plans). The BMA definition for categories are as follows:

> _Distance/Depth_
>
> ***Near shore Hardbottom**: The near shore hardbottom is typically exposed as a 200-400 meter-wide strip from the shoreline, ranging from the supralittoral zone to the depth of -4 meters, and is divided into 3 zones: a) slightly above tidal line (supralittoral zone); b) intertidal area between high spring tide and low spring tide marks (littoral zone); c) from the low spring tide mark to the depths of -4 meters (upper sublittoral zone). The longshore and cross-shore currents, waves and suspended sediments influence this area. Typical communities are adapted to stresses associated with the pounding surf, scour from mobilized sand and naturally elevated turbidity levels. Low relief hardbottom in this area is generally ephemeral and benthic communities' exhibit rapid re-colonization by new growth.*
>
> ***Intermediate Hardbottom**: Hardbottom existing from the depth of -4 meters to the depth of closure (approximately -8 meters). There is generally less stress to the community from sand scour. The hardbottom is typically more persistent, with a more diverse and stable benthic community, depending on the relief.*
>
> ***Offshore Hardbottom**: Hardbottom in water depths deeper than -8 meters, beyond the depth of closure to -12 meters. Benthic communities are more stable here with more developed and older communities. Often larger sized species and fish are present here.*
>
> _Persistence_

***Persistent Hardbottom**: Persistent hardbottom habitats are consistently exposed and generally visible in aerial photography and/or verified by in situ field survey data. This habitat contains stable biological features such as older age classes of benthic species (e.g. corals, algae and sponges) as well as benthic communities in sub-climax/climax status. Burial can occur within persistent habitats, but the time of exposure is sufficient to allow for occupancy by benthic and demersal organisms and associated production functions. Due to the more stable environmental conditions of persistent hardbottom, most macroalgae in these habitats are perennial species and in some cases may live up to 20 years. Larger sponges, scleractinian corals, and octocorals may also be present. Some fish species reside for an entire life cycle. Transient larval and juvenile stages of many species occur year-round with peaks corresponding to species-specific seasons of larval recruitment.*

***Ephemeral Hardbottom**: Ephemeral habitats are disturbance-mediated non-equilibrium systems (FDEP NHB Study; CSA 2009); burial and exposure of these habitats occur with a frequency that promotes new growth, inhibits colonization and growth of the benthic invertebrate community, and along with scouring effect of sediment transport by wave-generated currents, reduces macroalgal cover and herbivore abundance. Benthic community structure is driven by dynamic physical conditions associated with wave activity and sediment scour. Epibiota may persist temporarily under the sand or through the sand. Algal species that persist in this habitat typically are forms with high reproduction rates due to continuous spore release events, and are very resilient to environmental disturbances (FDEP NHB Study; Eriksson and Johansson, 2005;). Communities typically present in ephemeral hardbottom habitats include fast-growing macroalgae (e.g. Chaetomorpha spp. and Ceramium spp.), filamentous turf, Padina, Gracilaria, opportunistic green and brown sheet form algae (e.g. Ulva, Dictyota), and other early succession species with a short life cycle. The annual algal biomass production in these species is highly variable since they allocate much of their resources for speedy reproduction typical for r-strategic species.*

The diversity of algal species is an indicator of the duration of exposure; low diversity is characteristic for ephemeral communities. Benthic forms typical for persistent communities can be present in the ephemeral communities, but normally only as recruits and juvenile forms.