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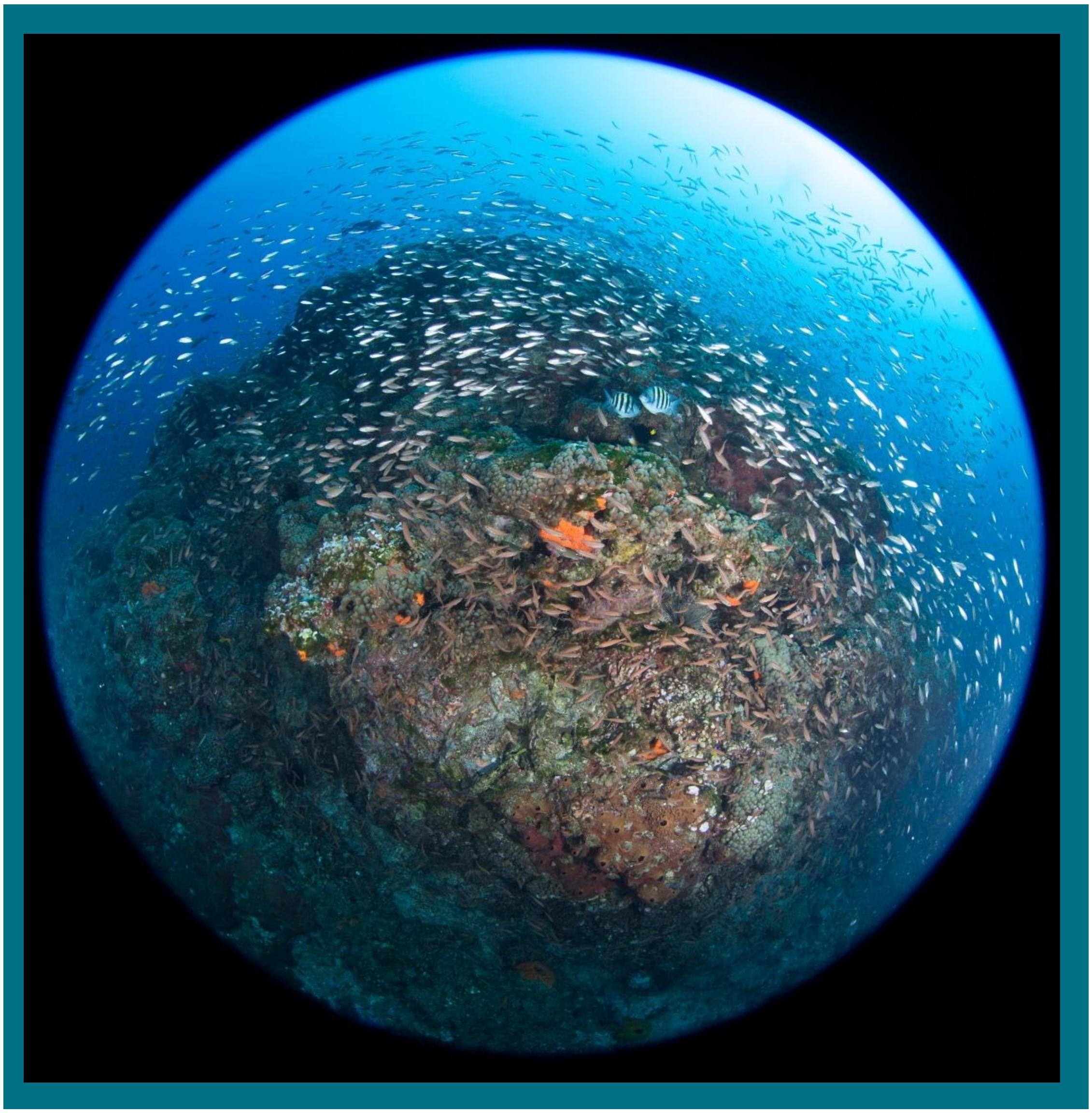
NATIONAL MARINE SANCTUARIES CONSERVATION SCIENCE SERIES

Stetson Bank Long-Term Monitoring: 1993-2015



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NATIONAL MARINE SANCTUARIES CONSERVATION SERIES ONMS-20-06

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Dr. Neil Jacobs, Acting Administrator

National Ocean Service
Nicole LeBoeuf, Acting Assistant Administrator

Office of National Marine Sanctuaries
John Armor, Director

Report Authors:
Marissa F. Nuttall[1,2], Paul Somerfield[3],
Travis K. Sterne[1,2], James MacMillian[1,2], John A. Embesi[1,2], Emma L. Hickerson[2], Michelle A. Johnston[2], G.P. Schmahl[2], and James Sinclair[4].

[1]CPC, Galveston, TX, USA
[2]Flower Garden Banks National Marine Sanctuary, Galveston, TX, USA
[3]Plymouth Marine Laboratory, Plymouth, UK
[4]Bureau of Safety and Environmental Enforcement, Office of Environmental Compliance,
New Orleans, LA, USA

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Cover Photo:
A claystone feature that comprises the pinnacle habitat at Stetson Bank. Various sponges, invertebrates, and fish can be seen using the habitat. Image credit: G.P. Schmahl/NOAA

NATIONAL MARINE SANCTUARIES

About the National Marine Sanctuaries Conservation Series

The Office of National Marine Sanctuaries, part of the National Oceanic and Atmospheric Administration, serves as the trustee for a system of underwater parks encompassing more than 620,000 square miles of ocean and Great Lakes waters. The 14 national marine sanctuaries and two marine national monuments within the National Marine Sanctuary System represent areas of America's ocean and Great Lakes environment that are of special national significance. Within these waters, giant humpback whales breed and calve their young, coral colonies flourish, and shipwrecks tell stories of our maritime history. Habitats include beautiful coral reefs, lush kelp forests, whale migration corridors, spectacular deep-sea canyons, and underwater archaeological sites. These special places also provide homes to thousands of unique or endangered species and are important to America's cultural heritage. Sites range in size from less than one square mile to more than 582,000 square miles. They serve as natural classrooms and cherished recreational spots, and are home to valuable commercial industries.

Because of considerable differences in settings, resources, and threats, each national marine sanctuary has a tailored management plan. Conservation, education, research, monitoring, and enforcement programs vary accordingly. The integration of these programs is fundamental to marine protected area management. The National Marine Sanctuaries Conservation Series reflects and supports this integration by providing a forum for publication and discussion of the complex issues currently facing the National Marine Sanctuary System. Topics of published reports vary substantially and may include descriptions of educational programs, discussions on resource management issues, and results of scientific research and monitoring projects. The series facilitates integration of natural sciences, socioeconomic and cultural sciences, education, and policy development to accomplish the diverse needs of NOAA's resource protection mandate. All publications are available on the Office of National Marine Sanctuaries website (http://www.sanctuaries.noaa.gov).

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Contact

Marissa F. Nuttall
Research Operations Specialist
CPC, contracted to
NOAA Flower Garden Banks National Marine Sanctuary
4700 Avenue U, Bldg. 216
Galveston, TX 77551
409.356.0391
Marissa.Nuttall@noaa.gov

James Sinclair
Marine Ecologist
Marine Trash & Debris Program Coordinator
Bureau of Safety and Environmental Enforcement
Office of Environmental Compliance
1201 Elmwood Pk. Blvd., GE 466
New Orleans, LA 70123
504-736-2789
Jim.Sinclair@bsee.gov

Table of Contents

List of Figures

List of Tables

Abstract

This report reviews historical studies conducted at Stetson Bank in the Gulf of Mexico and presents reanalyzed annual monitoring data from 1993 through 2015. The earliest known documentation of Stetson Bank was in 1930. Between then and 2015, over 40 studies examining the geological and biological components of the site were conducted. Stetson Bank is an uplifted, high relief, claystone feature associated with an underlying salt dome. It supports a well-developed community of tropical marine sponges and corals. The location of the bank provides marginal environmental conditions for coral reef development due to varying temperature and light availability. The fish community is similar to other Caribbean reefs, but has reduced diversity due to the site's isolation, small size, and dynamic environment. The water column at Stetson Bank includes annual anomalies, which stress the biological communities. In the past, when anomalies were experienced in one or two parameters, biotic communities at Stetson Bank remained fairly stable, but when multiple anomalies occurred in the same year, significant changes were observed in the benthic biota.

The benthic community at Stetson Bank has undergone several significant shifts between 1993 and 2015, changing from a habitat primarily composed of hydrocoral and sponges to one dominated by macroalgae and sponges. These changes occurred in years when multiple environmental stressors affected the region. In addition to these rapid changes in benthic community, long-term declines in sponge cover and growth of macroalgae were also observed. The decline in the cover of sponges correlated with a decline in sighting frequency of spongivorous fish. Overall, the fish community was temporally variable, with sporadic recruitment events potentially contributing to this variability. Fish biomass at Stetson Bank is high, with piscivore biomass typically greater than herbivores, thus exhibiting an inverted trophic biomass pyramid.

Keywords

benthic community, fish community, Flower Garden Banks National Marine Sanctuary, long-term monitoring, mesophotic coral ecosystem, Stetson Bank, water quality

Chapter 1

INTRODUCTION AND LITERATURE REVIEW

Stetson Bank, known to fisherman in the 1960s as "10 ½ Fathom Lump" (Pulley 1963) and "Barracuda Rock" (Hofferbert 1963), was named after the Woods Hole Oceanographic Institute geological oceanographer Henry C. Stetson (US BGN ACUF). It is an uplifted mid-Tertiary (Miocene epoch) claystone feature associated with an underlying salt dome, located approximately 70 nautical miles southeast of Galveston, Texas. The surface expression of the bank is comprised of two distinct hard bottom features: a central high relief outcrop and a ring of surrounding low relief outcrops (a rim-syncline). The central feature of the bank, referred to in the report as the bank crest, is approximately 800 m by 250 m and extends 36 m upwards from the seafloor. The center of the bank's crest is relatively flat, with an average depth of 23 m, and has scattered vertically oriented outcroppings of bedrock with ridges, terraces, and pinnacles, perforated from bioerosion, rising to within 17 m of the surface. The outer edges of the bank crest slope steeply (20-70 degrees) down to the seafloor at 53 m. The low relief outcrops surrounding the central feature emerge from the surrounding seafloor at 58 m and have a maximum relief of 3 m, forming a non-concentric ring 450-900 m away from the central feature (Figure 1.1). Both of these habitats harbor distinctive biological communities differentiated by depth, temperature, and turbidity. Multiple natural bank and reef features exist along the continental shelf in the northern Gulf of Mexico, with Stetson Bank considered a mid-shelf bank. The closest known features to Stetson Bank are Claypile Bank, located 11 nautical miles northeast, and 32 Fathom Bank, located 14 nautical miles southwest. Sonnier Bank, located 97 nautical miles to the east of Stetson Bank, occurs along the same latitude and shelf location and has similar geology and biology to Stetson Bank.

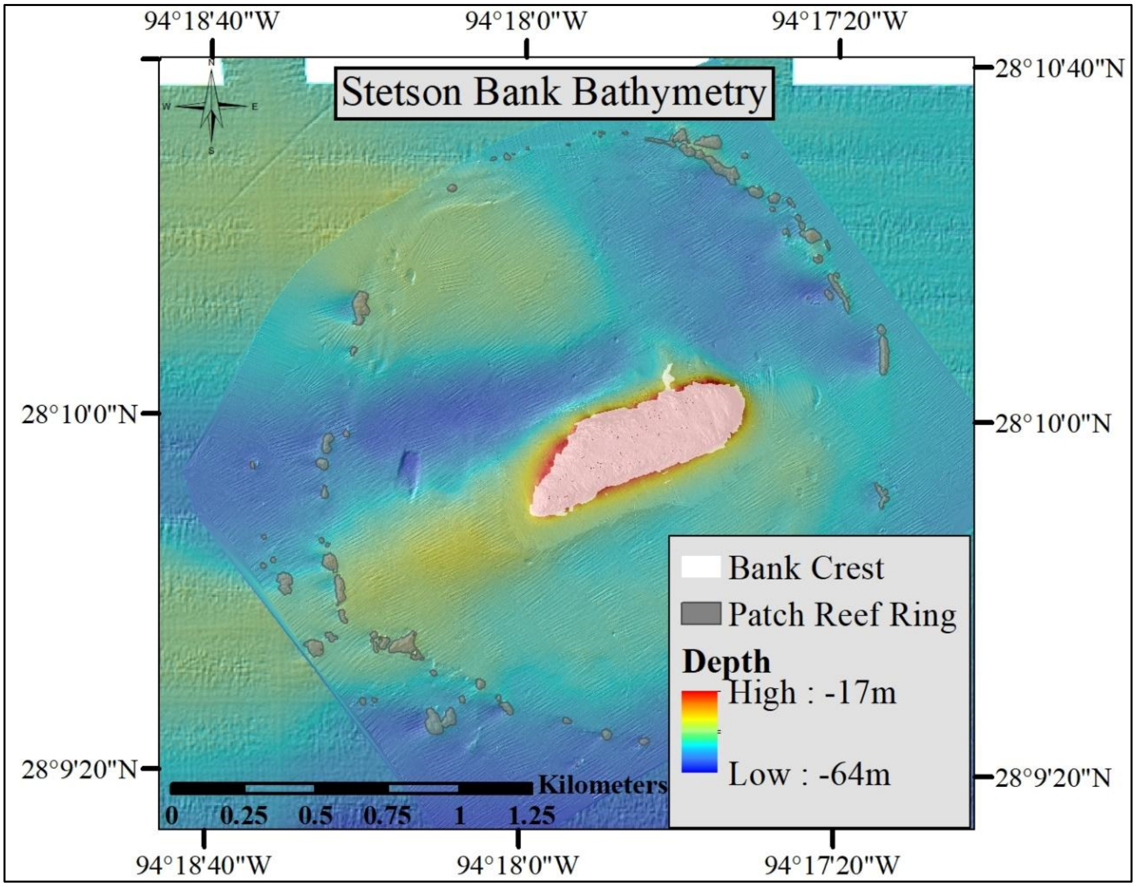


Figure 1.1. Bathymetric map of Stetson Bank. Color denotes depth with bank crest and patch reef ring highlighted. Image: NOAA

Stetson Bank was first documented during a study of the development of the delta of the Mississippi River in 1930 (Trowbridge 1930). The bank was later studied by Shepard (1937) who conducted a hydrographic survey of portions of the Gulf of Mexico with the U.S. Coast & Geodetic Survey (USCGS), now the National Ocean Service, using lead line surveys to document 26 banks along the continental shelf break. The subsurface feature of Stetson Bank appeared on the first nautical charts in 1939 (Figure 1.2). Many of these banks, including Stetson Bank, were hypothesized to be salt dome expressions (Shepard 1937, Lankford & Curray 1957, Nettleton 1957).

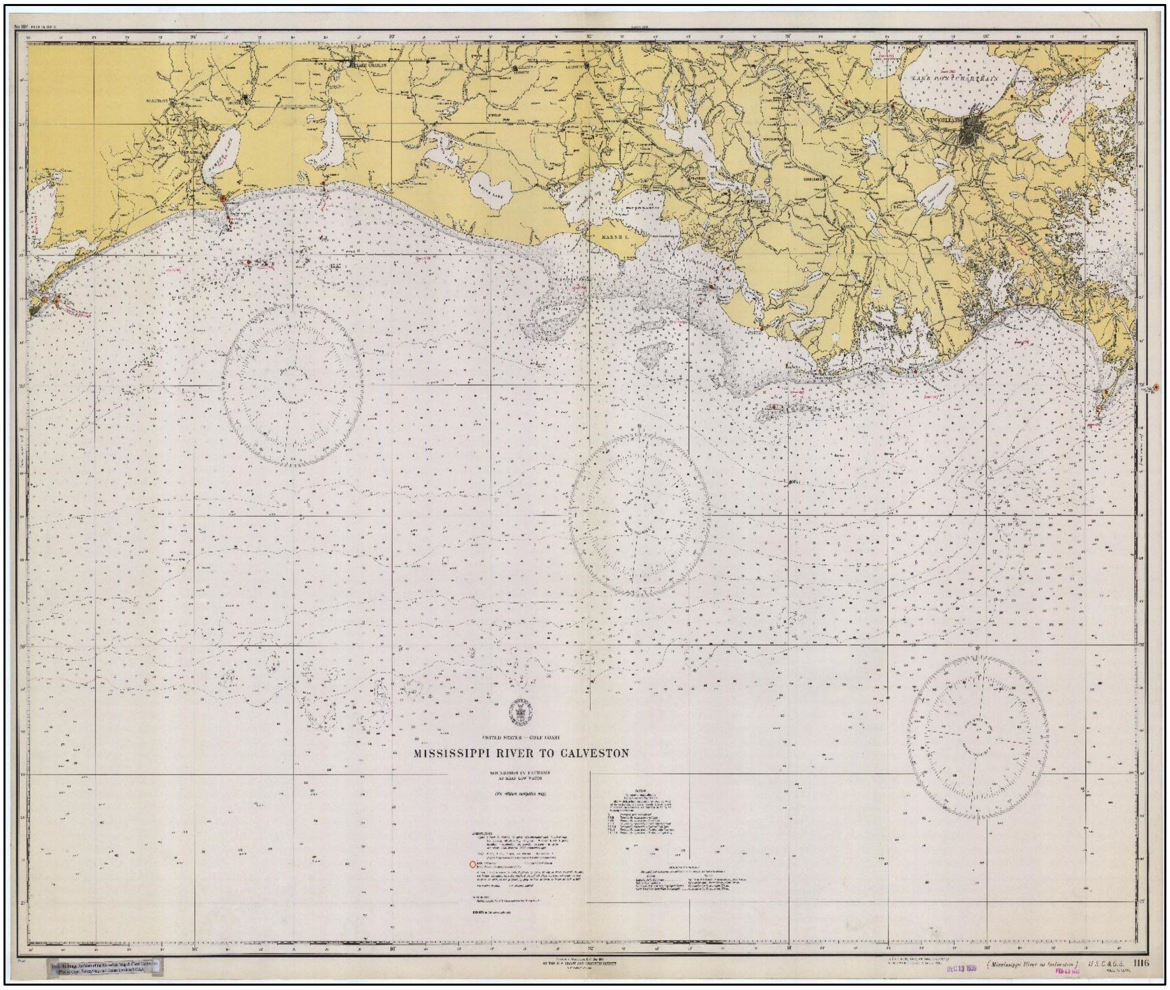


Figure 1.2. U.S. Coast & Geodetic Survey 1937 chart of the Mississippi River to Galveston. This is the first nautical chart where the subsurface features of Stetson Bank first appear. Image: USCGS

Geologic surveys of the bank were conducted by researchers from Scripps Institution of Oceanography (Lankford & Curray 1957) and suggested that the bank was composed of sandstone and claystone from the mid-Miocene. The first fathograms (a graphic representation of the sea floor made using a depth finder) of the bank were conducted in 1957 using line-of-sight triangulation (Neumann 1958), documenting the bank's dimensions, topography, and sediments. This study also noted the presence of the boring clam *Jouannetia quillingi*, one of the species responsible for the perforated appearance of the substrate at Stetson Bank. Neumann (1958) also suggested that the sediments that form the bank were of more recent origins than the mid-Miocene, suggesting mid-Tertiary, and documented four groups of mollusks that represent historical changes in environmental conditions linked to sea level rise, including tropical warm-water species, soft sediment species, nearshore species, and brackish water species. (Samples of the species *Rangia cuneata* were carbon dated to 13,000 years old.) In 1960, extensive reef-building coral communities were documented by scuba divers atop East and West Flower Garden Banks (FGBs), roughly 50 km southeast of Stetson Bank (Pulley 1963). During the same study, the first recorded scuba divers visited Stetson Bank and documented massive rock formations with few reef-building corals (Figure 1.3). The researchers theorized that the differences observed at Stetson Bank were likely due to winter temperatures falling below the generally accepted threshold for coral reef development. A study of foraminiferans, conducted by Loep in 1965, documented West Indian genera including *Amphistegina, Archaias*, and *Peneroplis* at Stetson Bank. Some of the first species records of hermatypic corals at Stetson Bank appear in 1971 (Edwards 1971), where sparse colonies of an unknown *Siderastrea* sp., *Orbicella* sp. (reported as *O. annularis*), *Madracis asperula*, and *Pseudodiploria strigosa*, were documented. All of these species, with the exception of *Orbicella annularis*, are still observed in isolated colonies on the bank.

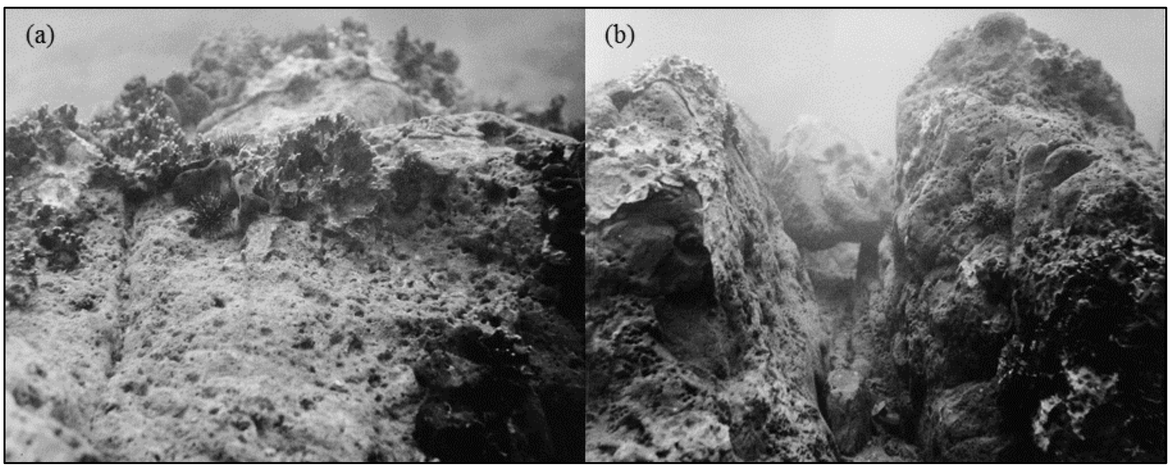


Figure 1.3. Images taken by scuba diver on the Pulley (1963) Expedition. Image A shows a close up of sponges and fire coral and Image B shows high relief pinnacle features. Photos: Dr. T.E. Pulley/Houston Museum of Natural Science

In 1974, due to increased interest in the region for offshore oil and gas exploration, a baseline survey of the reef was conducted by Texas A&M University College of Geosciences, which documented the geological (Bryant et al. 1974) and biological (Bright et al. 1974) components of the bank. Bryant et al. (1974) provided topographic cross sections of the bank and documented the shallowest portion of the bank at 17 m, identified chronically high turbidity water extending downward from 49 m, and noted the potential influence of tropical weather systems, and boring clams (specifically, *Leiosolenus bisulcatus* and *J. quillingi*) in shaping the topography of the bank. Bright et al. (1974) identified the dominant epifauna as an unknown *Neofibularia* sp. (potentially *N. nolitangere*) and fire coral *Millepora alcicornis*, documented the presence of small (<0.3 m) *Stephanocoenia intersepta* colonies around the edges of the bank to a depth of 49 m, noted a lack of leafy algae species, and documented an abundant population of the long-spined sea urchin (*Diadema antillarum*). Fifty-five species of fish were also recorded on the bank crest, predominantly wrasse, small grouper, and butterflyfish. Consequently, in 1974 Bureau of Ocean Energy Management (BOEM, formerly Bureau of Land Management [BLM] and Minerals Management Service [MMS]) developed no-activity zones around significant topographic features, including Stetson Bank, to protect these features by prohibiting seafloor disturbance from oil and gas exploration activities (MMS 2007). In 1976, a follow up study documented a similar community to that observed in 1974, and suggested that there was no evidence of nearby drilling activities impacting the biota (Bright & Rezak 1978). Bright and Rezak (1976) compiled geologic and biologic information from Stetson Bank with several other features along the Texas continental shelf and presented graphics of the communities, identifying major biological zones (Figure 1.4). Concerns about the impact of nearby oil and gas activities on reefs and banks in the northwestern Gulf of Mexico led to additional sediment surveys at several locations, including Stetson Bank by Rezak et al. (1982), and recommended the continuation of restrictions on oil and gas activity.

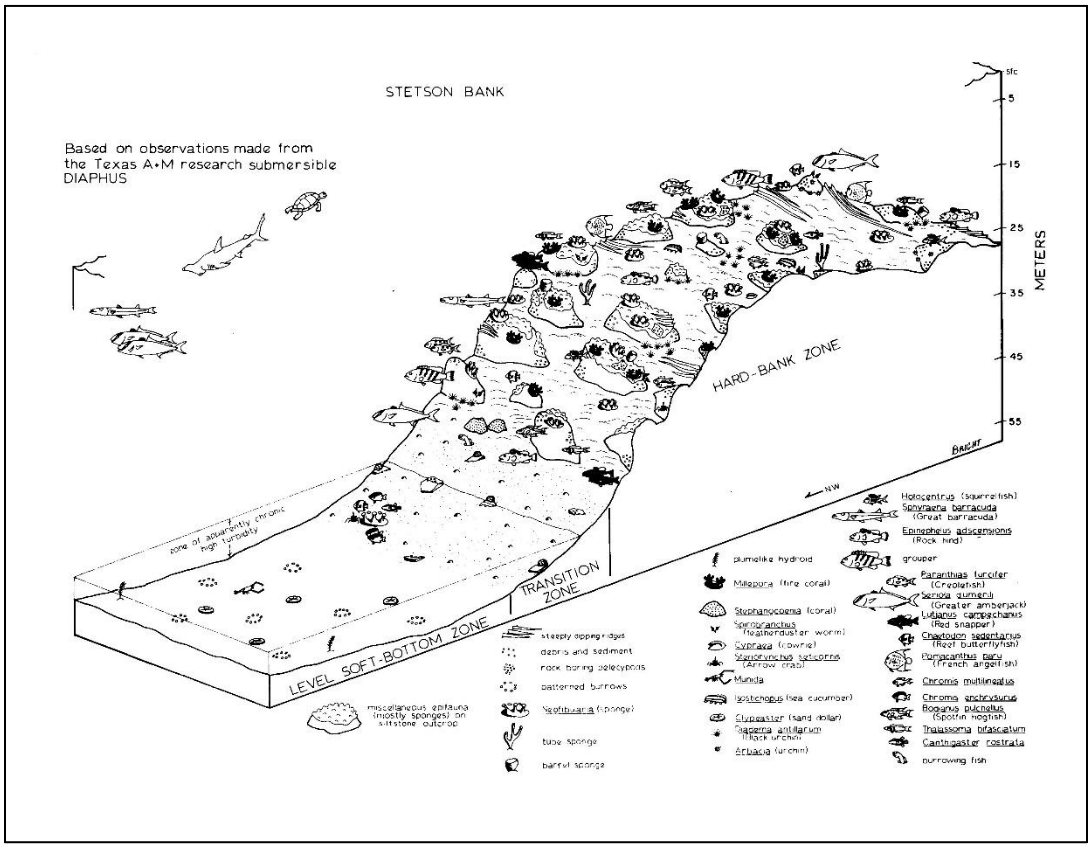


Figure 1.4. Major biological zones at Stetson Bank. Image: Bright & Rezak 1976

Rezak et al. (1985) presented a compilation of information on the reefs and banks along the Texas-Louisiana continental shelf and noted Stetson Bank's similarities to other mid-shelf banks including Sonnier Bank, located approximately 97 nautical miles to the east-northeast. These mid-shelf banks possessed a clear water "minor reef-building" community referred to as the *Millepora*-Sponge Zone, from 18 to 40 m, and a Nepheloid Zone (high-turbidity zone) from 50 to 62 m harboring species tolerant of turbid water. Mid-shelf banks, as characterized by Bryant (1974), are low relief features occurring 45 to 75 nautical miles offshore, rising steeply from the seafloor to depths of 46 to 73 m and cresting around 18 to 63 m. Stetson and Sonnier Banks were also noted for a diverse fish population, with an abundance of reef fish including angelfish, butterflyfish, wrasse, and chromis, in addition to commercially and recreationally valuable fish including snapper. Studies of the reef fish assemblies at multiple banks, including Stetson, by Dennis and Bright (1988) noted the similarities between habitat and communities to other mid-shelf banks, including Sonnier and Claypile Bank. This study documented 43 species of fish and noted the prevalence of chromis, grunt (particularly tomtate, *Haemulon aurolineatum*), goby, and damselfish in these habitats despite stressful hydrographic conditions (thermal variability, turbidity, and storm impacts).

In 1993, a long-term monitoring program was initiated at Stetson Bank by Gulf Reef Environmental Action Team (GREAT), a non-profit organization composed of volunteer divers and citizen scientists. These first monitoring cruises developed benthic maps of a portion of the crest of Stetson Bank, near the permanent mooring buoys on the northwestern portion of the bank crest (Figure 1.5). They also established benthic monitoring with repetitive photostations atop high relief pinnacle features, began semi-quantitative reef fish censuses, collected random photographs of the reef, and installed thermographs on the bank crest. In addition, during the 1993 field work, GREAT installed the first U-bolts for mooring buoys at Stetson Bank to reduce the impacts of anchoring and facilitate access for recreational divers. Preliminary results from this study were reported in Boland et al. (1995), and represented the first major study focused on the bank since the 1970s. Initial results documented 110 fish species and extreme short-period temperature fluctuations of 3.4^{o} C over a 10-day period. This study also documented the presence of argonaut octopus egg cases (G. Bunch, personal communication, February 15, 2001), which have not been observed at the bank since. Data from this study were reprocessed and presented in this report.

Several projects stemmed from the initial monitoring studies conducted by GREAT, including analyses of benthic cover. These analyses documented sponges as the greatest contributor to overall cover at repetitive photostations and measured sea urchin diversity and densities, with *D. antillarum* documented at a density of ~1 per m^2 in 1995-1996 (Matson & Boland 1996). Random transect assessments from data collected in 1994 documented exposed substrata and reef rock as the predominant benthic component and sponges as the predominant biota in the vicinity of mooring buoys 1-3 (Downey 1994). A molluscan community survey documented 54 species, including queen conch (*Lobatus gigas*), at Stetson Bank (Hyde 1995).

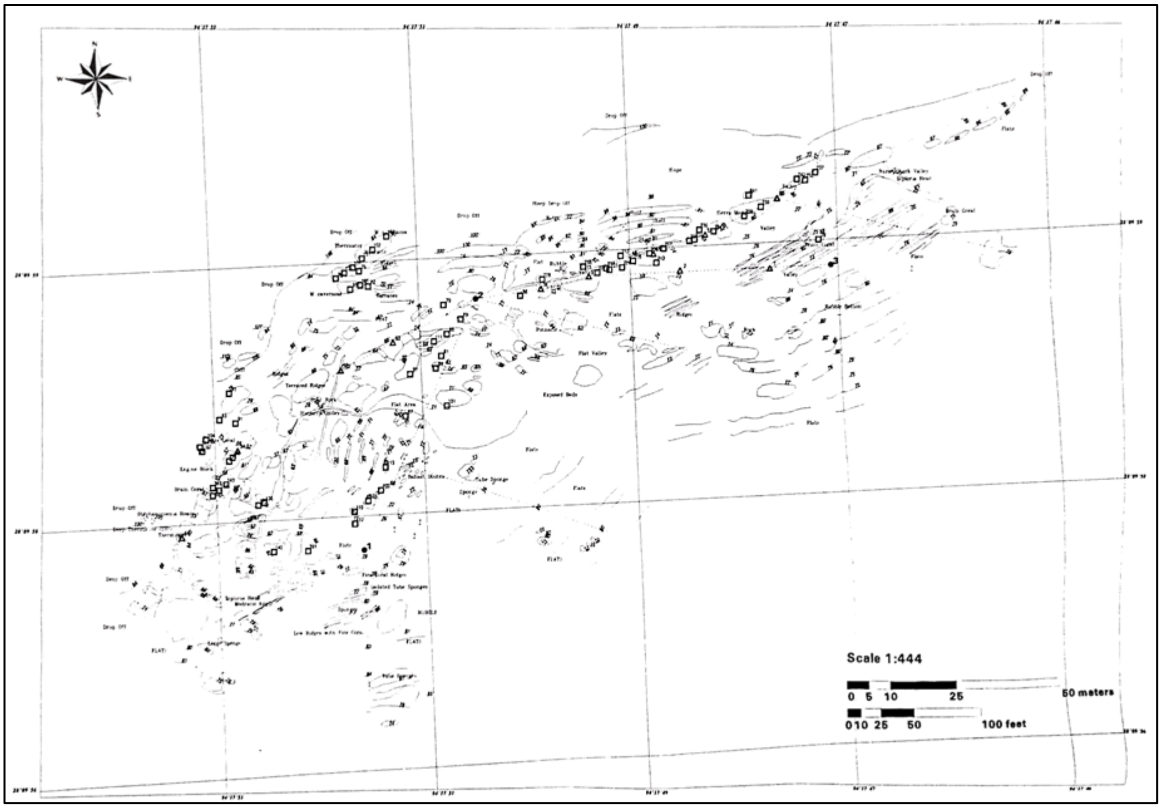


Figure 1.5. Topographic map of Stetson Bank, May 1997. Collected (1993-1996) and provided by GREAT and Flower Garden Banks National Marine Sanctuary, with the principal investigator G.S. Boland. Image: K.J.P. Deslarzes and M.V. Morin/MMS

In 1992, East and West FGB were designated as Flower Garden Banks National Marine Sanctuary (FGBNMS). With growing concern over anchoring impacts and resource depletion by fishing and collecting, the local dive community (notably the Houston Underwater Photographic Society), researchers, and legislators worked to recommend Stetson Bank for inclusion into FGBNMS. Following the development of a bill led by U.S. Representative Solomon Ortiz (H.R. 3886 1994) and various support letters, Stetson Bank was added to the sanctuary system through congressional designation in 1996 (Figures 1.6 and 1.7). The official boundary and regulations were published in the Federal Register in

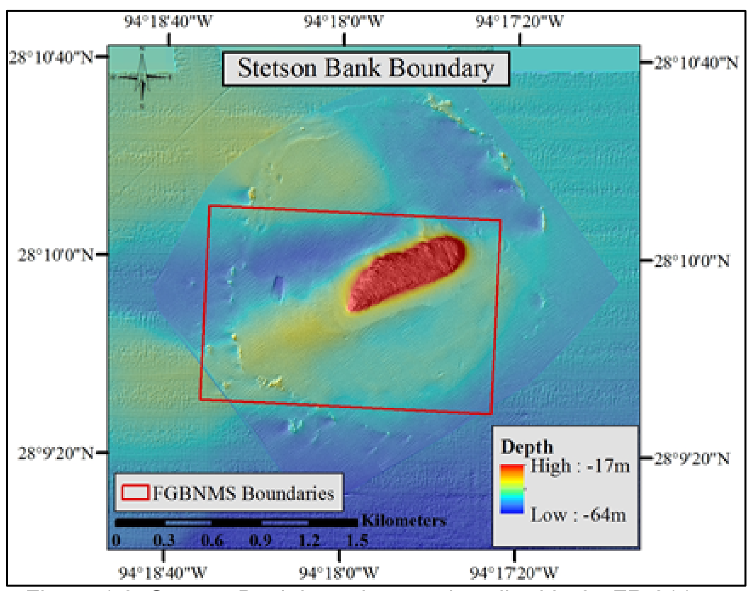


Figure 1.6. Stetson Bank boundary as described in 65 FR 81175. (Image: NOAA)

2000 (National Marine Sanctuaries Preservation Act [P.L. 104-283] 1996, 15 C.F.R. § 922, 65 FR 81175). This new designation afforded Stetson Bank additional protections through the prohibition of bottom-impacting activities (e.g., anchoring, drilling, and explosive use) and of fishing other than conventional hook and line gear. Due to the proximity of FGBNMS to shipping fairways heavily trafficked by international vessels, domestic regulations were deemed insufficient to address potential anchoring issues, leading to the development of no anchoring areas at FGBNMS under the International Maritime Organization (NAV46/3/3). Following the addition of Stetson Bank to FGBNMS, the Center for Coastal Studies at Texas A&M University, Corpus Christi, led by Dr. Quenton R. Dokken, continued monitoring efforts at the bank through 2001, at which time sanctuary staff and volunteers absorbed the monitoring project. These efforts maintained the annual collection of images from benthic repetitive photostations and as well as reef fish censuses and bank crest temperature data, when time and funds permitted (Appendix A: Table A.1).

Figure 1.7. The National Marine Sanctuary System, 2019. Image: NOAA

Following sanctuary designation, studies to further characterize the bank continued. Reports describing the elasmobranch community by Childs (1998, 2001) characterized Stetson Bank as potential forage, mating, and nursery habitat for 11 documented species of sharks and rays. The fish (Pattengill 1995, Pattengill et al. 1997, Pattengill 1998) populations within the sanctuary were found to be similar to tropical reef systems and were characterized by many rare and few abundant species, and maintained a similar trophic structure temporally, but possessed overall reduced species richness in comparison to other locations in the tropical western Atlantic (specifically due to a lack of diversity among parrotfish, grunt, and snapper) (Figure 8). The following year, Pattengill-Semmens (1999) documented a unique golden color morph of the smooth trunkfish, *Lactophrys triqueter*, at both Stetson Bank and the FGBs. In 2000, Bernhardt presented the first temporal monitoring results from Stetson Bank monitoring between 1993 and 1999 (also available in an unpublished report by Bernhardt & Boland 2014). While the study found significant variation between photostations, no overall temporal changes were documented in the benthic community between 1993 and 1999, with the community comprised of ~30% each of *M. alcicornis*, sponges, and exposed substrate. A method for interactive color segmentation to obtain benthic cover was compared to point count methods, presented in the Bernhardt (2000) study and published in Bernhardt and Griffing (2001). In 2000, Hyde continued his work on the mollusks at Stetson Bank (Hyde 1995), presenting a four-year study of the mollusk community that documented 195 species of mollusk within three distinct depth zones to 36.6 m. Additionally, in 2002, Burnside and McNamara conducted another census of conch at Stetson Bank, documenting both queen conch and milk conch (*L. costatus*) in densities of 1 and 0.6 per 100m^2, respectively (Figure 1.9; Burnside & McNamara 2002). Further evaluation of motile invertebrate community was conducted by Mike Tringali and John Hunt (of Florida Fish and Wildlife Research Institute) in 2006, where Caribbean spiny lobster (*Panulirus argus*) were sampled at Stetson Bank. *P. argus* sampled averaged 206 mm (±1.5 SE), larger than *P. argus* reported for other marine protected areas in the Caribbean (Bertelsen et al. 2004) with a lack of smaller individuals.

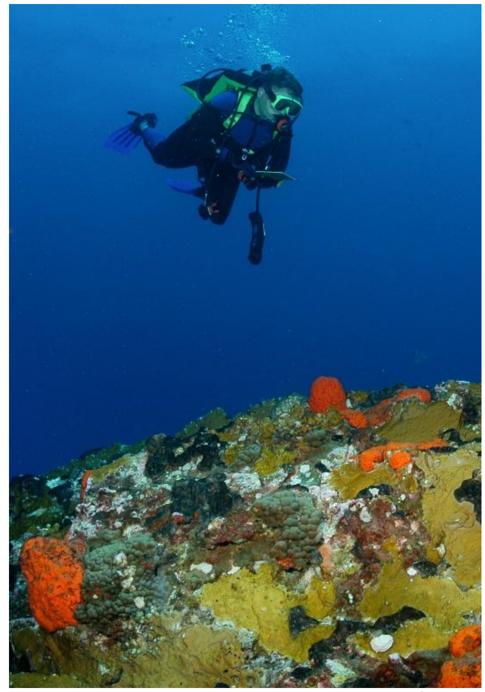


Figure 1.8. A diver conducts a roving fish survey at Stetson Bank in 2004. Photo: G.P. Schmahl/NOAA

Figure 1.9. Queen conch, *L. gigas*, at Stetson Bank. Photo: Mark McNamara

In 2002, Stetson Bank was included in The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States report (Schmahl 2002). The report examined anthropogenic and environmental stressors to the reefs of FGBNMS, including the potential buffering effects of depth on water temperature, the low coral disease incidence at East and West FGBs, and that Stetson Bank appears, through algal nitrogen isotope signatures, to be more consistently impacted by coastal runoff than East and

West FGBs. The impacts of fishing at FGBNMS were identified in this report as lacking information; however, marine debris associated with this activity has been documented at all the banks within the sanctuary. The northern Gulf of Mexico represents one of the most active regions of oil and gas exploration in the US; therefore, the potential impact of oil and gas exploration activities were discussed, with the note that no detectable detrimental impacts had been identified to-date. In 2005, another The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States report (Hickerson & Schmahl 2005) reported on the continuation of anthropogenic and environmental stressors to the reefs of FGBNMS, adding details on the physical scouring and toppling of pinnacle features due to storm impacts from hurricane activity (hurricanes Katrina and Rita). While headboats and fishing charters are known to frequent the vicinity of FGBNMS, the impacts of fishing and diving were again identified as activities lacking information. However, through direct observation and anecdotal catch reports (such as the Reel Report by Joe Kent in the *Galveston Daily News*), it is assumed that Stetson Bank's closer proximity to land results in greater use than at East and West FGBs. The area surrounding Stetson Bank is a popular shrimp trawling ground and impacts from bycatch and marine debris from fishing were documented in this report. A summary on the presence, type, and impacts of marine debris was also presented, including pipelines, fishing line, and nets (figures 11.0 and 1.11).

Figure 1.10. Trawl net and floats entangled in outcroppings at Stetson Bank. (Photo: UNCW-UVP/NOAA)

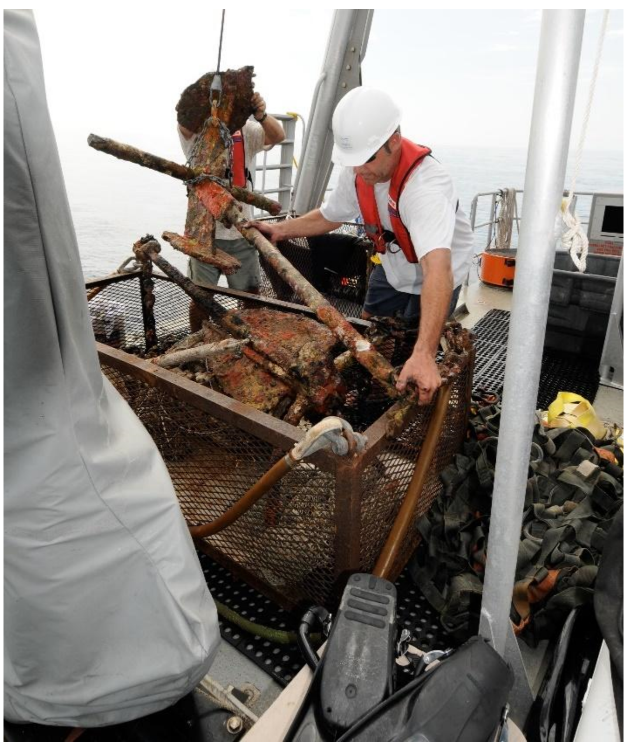


Figure 1.11. Anchors and other marine debris were removed from the crest of Stetson Bank. Photo: G.P. Schmahl/NOAA

With the support of historical studies in the region, the Gulf of Mexico Fishery Management Council (GMFMC) recognized the value of Stetson Bank as a Habitat Area of Particular Concern (HAPC) in 2006 (Figure 1.12) and established regulations for fishing vessels that prohibit anchoring and the use of bottom impacting fishing gear (bottom longlines, trawls, pots, and traps) in an effort to protect these habitats from fishing impacts (50 C.F.R. § 622, GMFMC 2005). In 2008, NOAA's National Ocean Service Office of Response & Restoration Marine Debris Program funded a project to assess, map, and remove derelict fishing gear from Stetson Bank, removing four large anchors, an engine block, and approximately 30 lbs. of miscellaneous fishing debris (DeBose 2008; Figure 1.11).

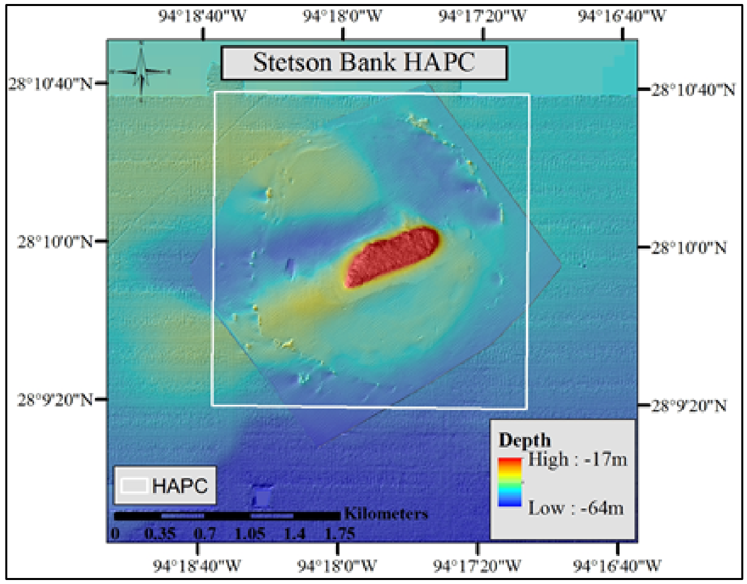


Figure 1.12. Stetson Bank Coral HAPC boundary (50 C.F.R. § 622). Image: NOAA

In 2008, Schmahl et al. expanded on the biotic zone characterizations for reefs and banks in the northwestern Gulf of Mexico. The report described the benthic community at Stetson Bank as a coral community zone: not considered to be a coral reef, but containing low densities of hermatypic coral species, in addition to being characterized by *Millepora* spp. (fire coral), sponges, and tropical macroalgae. Zingula (2008) presented a summary of geology and paleontology of the bank, highlighting foraminiferal fossil records, indicative of the early Miocene age, and cross-bedding in siltstone samples collected from the bank that indicates turbidite sediment deposits. In addition, the report documented a wide variability in microfossil content of the substrate, indicating that while some mudstone was formed in place, some was brought to the area from shallower habitat as part of the turbidites. A 2008 report on FGBNMS, including Stetson Bank, was presented in The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States (Hickerson et al. 2008), expanding on the Hickerson & Schmahl (2005) report. Similar anthropogenic and environmental stressors to the reefs of FGBNMS were evaluated, adding observations of a coral bleaching event and a possible coral disease outbreak in 2005. Recent hurricane paths were examined and a coastal runoff plume was observed via satellite color imagery extending over Stetson Bank in 2005. This report also noted the lack of processed monitoring data to review for Stetson Bank and identified data processing as a priority for improving management capability. This need was addressed in a report by DeBose et al. (2013) that provided a historical analysis of monitoring data from Stetson Bank from 1993 to 2008. The report documented initial community stability in the 1990s and a major shift in community structure following 1999, when macroalgal cover on the reef started to increase, followed by another major shift in community structure in 2005, when the *Millepora*-sponge community was largely replaced by an algal community (Figure 1.13). This report also linked environmental parameters, including coastal runoff and elevated temperatures, to these changes in benthic community (DeBose et al. 2013).

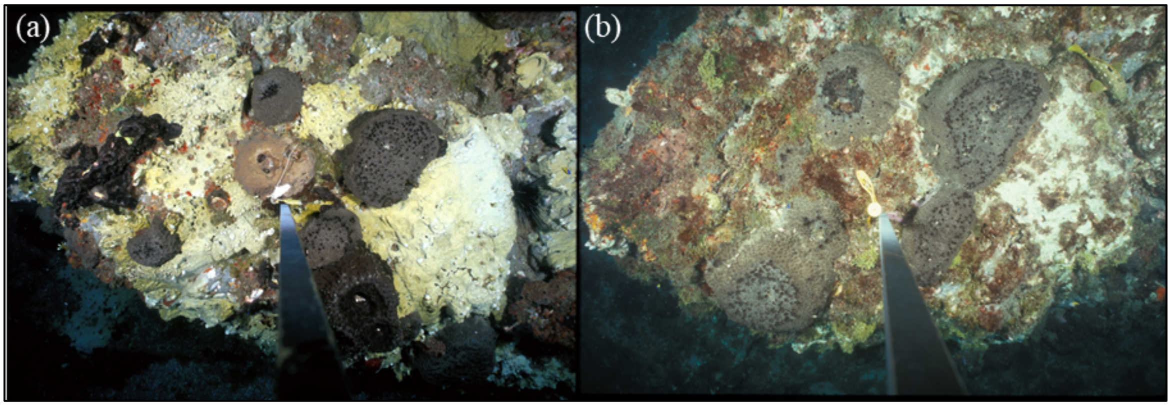


Figure 1.13. Repetitive photostation images in A 2000 and B 2007 show change from *Millepora*-sponge community to an algal-sponge community. Photos: NOAA

In 2010, Stetson Bank underwent sampling for hydrocarbons in response to the *Deepwater Horizon* oil spill Natural Resource Damage Assessment (NRDA). The April 20, 2010, explosion on the *Deepwater Horizon* Macondo drilling platform initiated the largest oil spill in U.S history and occurred approximately 600 km east of Stetson Bank. This assessment detected low concentrations of polynuclear aromatic hydrocarbons in the water column at Stetson Bank (DIVER 2018). However, the source of the detected hydrocarbons is unknown and the low concentrations and lack of apparent physical damage to the biota on the bank suggest the hydrocarbons had no known significant lethal impact on the biota.

Between 2001 and 2014, available funding for monitoring at Stetson Bank only allowed for data collection of repetitive photostation analysis, water temperature, salinity and nutrient monitoring, and sporadic fish censuses. However, in 2015, the Bureau of Safety and Environmental Enforcement (BSEE) and FGBNMS developed an interagency agreement for the continuation and expansion of annual long-term monitoring at Stetson Bank (E14PG00052). The study continued methods initiated in 1993 and enhanced annual monitoring efforts. Continued methods included bank crest repetitive photostations, bank crest random transects, water nutrient analysis, bank crest water parameters, and quantitative bank crest fish surveys. Additional methods incorporated ocean carbonate analysis, expanded bank crest water parameter data collection, expanded water column profile parameters, and monitoring of the habitats inaccessible by standard scuba (Appendix A: Table A.1). The results presented in this report continue to document community changes at Stetson Bank with data incorporated from 1993 through 2015.

Over the years, studies at Stetson Bank have documented undescribed and exotic species. Wicksten and McClure (2003) discovered and described a new species of snapping shrimp, *Alpheus hortensis*, living amongst the rock rubble and holes on the bank. In 2007, Weaver &

Rocha documented and described a new species of wrasse, *Halichoeres burekae*, commonly known as Mardi Gras wrasse due to the terminal male's bright purple, blue-green, and yellow coloration (Figure 1.14). The Mardi Gras wrasse was described from a specimen collected at Stetson Bank, but has been documented at all banks within FGBNMS, with sightings indicative of sporadic recruitment events, and on reefs near Veracruz, Mexico. Several exotic species have also been documented at Stetson Bank, including (1) the nudibranch *Thecacera pacifica*, native to the Pacific, photographically documented in 2006; (2) the orange cup coral *Tubastraea coccinea*, native to the Pacific, first documented on platforms in the Gulf of Mexico in 1991 (Fenner 2001) and on Stetson Bank in 2012 (Precht et al. 2014), and still present on the reef in 2015; and (3) the lionfish, *Pterois volitans*, native to the Pacific and invasive in its exotic range, first documented in 2011 and seen in increasing numbers through 2013, with a minor decline in 2014, and considered established in FGBNMS (Johnston et al. 2016b). Lionfish collected at Stetson Bank were used by Johnson et al. (2016) to present a genetic analysis of the population and by Peake et al. (2018) to discuss feeding ecology.

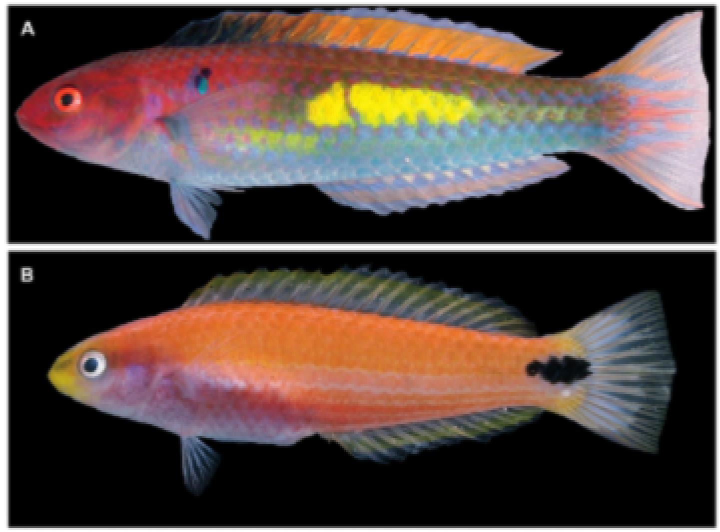


Figure 1.14. Mardi Gras wrasse (*H. burekae*) (A) holotype and (B) paratype. Image: Weaver and Rocha 2007

While the early studies at Stetson Bank focused on the central bank feature, a ring of hard-bottom outcrops were discovered surrounding the central feature following the collection of high resolution multibeam bathymetry (Gardner et al. 1998). The area was re-surveyed in 2004 with higher resolution technology aboard the NOAA Ship *Thomas Jefferson*. The surrounding outcrops are a part of an extensive ecological network that enhance and support adjacent habitats by providing potential refugia, feeding grounds, and spawning areas (Hickerson et al. 2008). The ring of outcrops surrounding Stetson Bank varies in relief from rubble to 3 m outcroppings. Remotely operated vehicles (ROVs) were first used to explore these habitats by FGBNMS researchers in 2001. These surveys documented extensive mesophotic habitat comprised of communities of sponges, black corals, and gorgonians, which exist in a persistent nepheloid layer (Figure 1.15; Rezak & Bright 1981, Rezak et al. 1985). The upper limit of the mesophotic zone is defined by decreased light availability and a distinct change in community, occurring at approximately 30-40 m depth, whereas the lower limit is defined as the lower boundary of the photic zone, occurring at approximately 80-150 m depth (Lesser et al. 2009, Kahng et al. 2010, Baker et al. 2016). A total of 16 ROV surveys were conducted on the outcrops from 2004 to 2013. These surveys expanded the knowledge of these habitats and the biota that use them. Eighty-five species of fish and benthic species from 12 phyla, including 12 species of cnidarian, have been documented. In addition, these surveys have revealed marine debris, predominately longline and trawl nets, which were widely distributed among the outcrop features. As part of the BSEE and FGBNMS partnership for Stetson Bank monitoring in 2015, quantitative surveys of these habitats were included in the monitoring plan. The current FGBNMS boundary does not encompass the entire ring of outcrops. The discovery and exploration of these features highlighted their value and a boundary modification proposal to include them was presented by FGBNMS in 2016 (Office of National Marine Sanctuaries 2016).

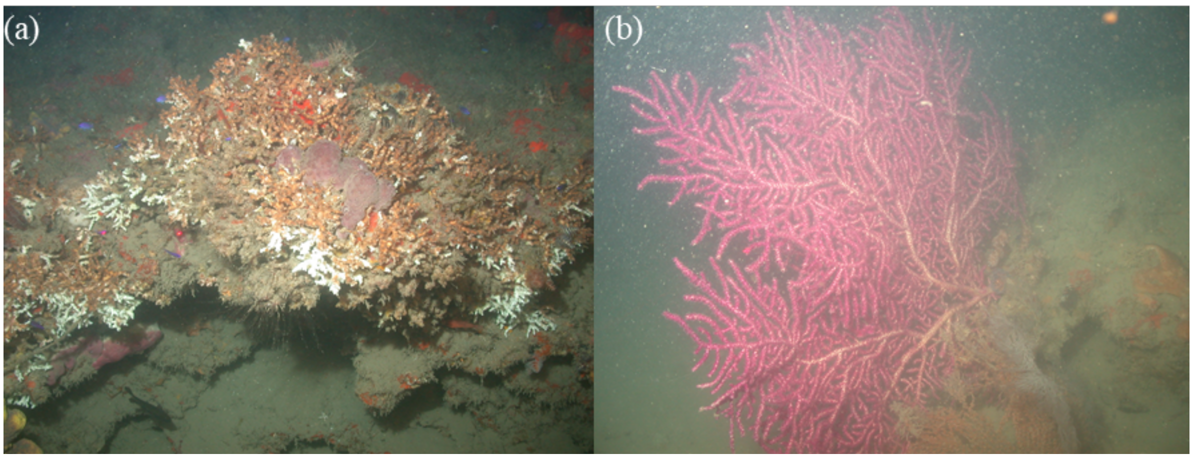


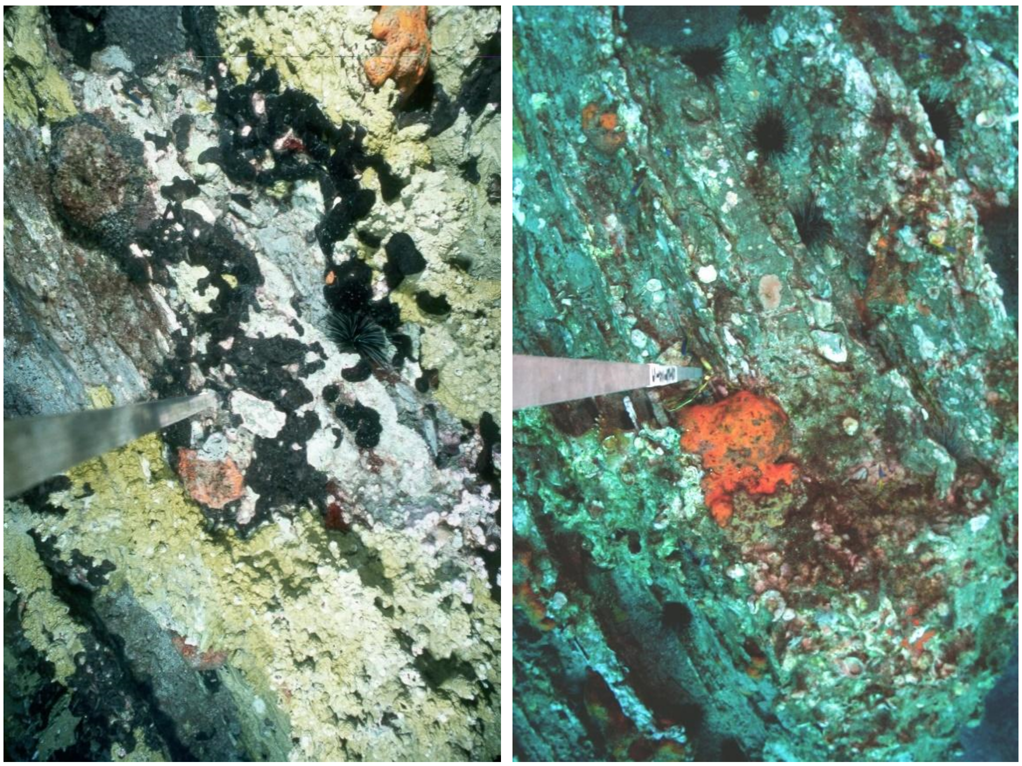
Figure 1.15. Images of the mesophotic reef surrounding Stetson Bank. A: Branching stony corals and sponges and B: octocorals. Photo: UNCW-UVP/NOAA

The most current monitoring data (Nuttall et al. 2019) shows that sponges, primarily *Neofibularia nolitangere*, *Ircinia strobilina*, and *Agelas clathrodes*, still comprise a large portion of the benthic biota on the bank crest, but have been in decline in recent years. Eleven species of hermatypic corals have been documented. Dominant species on the bank crest include *Pseudodiploria strigosa*, *Stephanocoenia intersepta*, *Siderastrea radians*, *Madracis auretenra*, *Madracis decactis*, *Madracis brueggemanni*, and *Agaricia fragilis* (Figure 1.16). The hydrozoan *Millepora alcicornis* (fire coral) was historically the predominant benthic organism in this habitat, but has declined since 2005 due to bleaching. This shallow portion of the reef is considered a high latitude coral community, existing at the northern limit of coral community ranges with “marginal” environmental conditions for coral reef development due to varying temperature and light availability. While considered a coral community rather than a coral reef, due to the presence but not dominance of reef-building scleractinian corals, the shallow portion of this bank is thought to provide habitat for potential future reef development (Schmahl et al. 2008). The sloping edges of the central reef feature from 34 to 52 m support a transitional community where crustose coralline algae abundance increases. The outcrops surrounding the main feature support a mesophotic community, comprised of sponges, black corals, and octocorals. In addition to the sessile benthic community, a diverse fish community is found at Stetson Bank. In comparison to other Caribbean reefs, Stetson Bank possesses low species diversity, primarily due to the absence of certain species such as hamlets, and the low diversity among parrotfish, grunt, and snapper; however, reef fish biomass and density at Stetson Bank, as well as at East and West FGBs, is greater than many other Caribbean reefs (Johnston et al. 2015a). As an offshore bank, both reef-associated and pelagic species are found at Stetson Bank. On the crest of the central reef feature, reef-associated wrasse, blenny, and chromis dominate abundance whereas biomass is predominantly barracuda, seabass, snapper, and jack. On the sloping edges of the central reef feature, reef-associated chromis and damselfish dominate abundance, while jack and snapper dominate biomass. On mesophotic outcrops, snapper, grunt, and jack dominate both abundance and biomass.

Figure 1.16. A diver swims over a pinnacle on the crest of Stetson Bank. Photo: G.P. Schmahl/NOAA

These unique communities attract divers, researchers, and anglers to Stetson Bank (Deslarzes 1998, Hickerson & Schmahl 2005, Hickerson et al. 2008). However, each of these activities has some impact on the environment. Although Stetson Bank is relatively removed from immediate coastal influences, this area can be affected by runoff from river discharges (Dodge & Lang 1983, Hickerson & Schmahl 2005, DeBose et al. 2013), hurricanes and associated high-energy waves (Hickerson & Schmahl 2005, Hickerson et al. 2008, Doyle 2009, Lugo-Fernandez & Gravois 2010), and anthropogenic activities, including trawling, marine debris, and oil and gas exploration and production (Hickerson & Schmahl 2005, Hickerson et al. 2008, Schmahl et al. 2008). While the coral reef communities at sites like Stetson Bank have evolved, matured, and adapted alongside intermittent natural and anthropogenic disturbances (Hughes & Connell 1999), stressors to marine environments (climate change, ocean acidification, oil and gas exploration, development for natural resources, etc.) are projected to increase, making long-term monitoring datasets essential to understanding community stability and ecosystem resilience. Additionally, these long-term datasets are vital in documenting the arrival and establishment of non-native species as well as the subsequent impact to the native population. Continuation and expansion of these extensive datasets will provide valuable insight for both research and management purposes.

Chapter 2
SESSILE BENTHIC COMMUNITY



Repetitive photostation #55 in 1993 (left) and in 2015 (right) shows major changes that have occurred in the benthic community in the 22-year time frame. Photo: NOAA

Introduction

Over the past few decades, benthic communities throughout the Caribbean have undergone significant changes in community composition (Pandolfi et al. 2003, Bellwood et al. 2004). Jackson et al. (2014) used available coral cover data from the Caribbean and surmised that the greatest overall Caribbean-wide changes in coral and macroalgal cover occurred in 1984 (in conjunction with *Diadema antillarum* die off [Lessios et al. 1984, Hughes et al. 1985, Gardner et al. 2003]) and in 1998, suggesting that the bleaching events in 2005 and 2010 had more localized effects on coral and macroalgal cover. Gardner et al. (2003) found that significantly greater rates in coral decline were observed in the Caribbean in the 1980s, in comparison to the 1990s, but found significant regional differences in coral cover decline, attributing these differences to the complex interactions between local stressors. The effects of these Caribbean-wide events were recorded at the nearby coral reefs of the East and West FGBs, located approximately 35 nautical miles to the southeast of Stetson Bank. A notable algal increase was documented for two years at East and West FGBs following the 1984 *D. antillarum* die-off (Gittings & Bright 1986) before stabilizing until 1998 when another increase was documented (Dokken et al. 2003) and followed by a steadily increasing trend (Johnston et al. 2016a). Coral bleaching events were observed in 2005, where, in November, ~10% bleaching was reported at East FGB with low post bleaching mortality rates (Precht et al. 2008, Zimmer et al. 2010), and 2010, where, in August, ~7% bleaching was reported at West FGB (Johnston et al. 2013). It should be noted that the surveys conducted by Johnston et al. (2013) were not conducted during the peak of the bleaching event and therefore underrepresent coral bleaching values. Changes in the benthic community can be highly localized (Bruno et al. 2009) and require multi-decadal datasets to characterize due to the presence of long-lived benthic species. While these extensive datasets are uncommon, the Stetson Bank long-term monitoring program provides over 20 years of continuous benthic data.

In 1993, an annual long-term monitoring program was initiated at Stetson Bank by GREAT to characterize and monitor the benthic community. Initially, the monitoring consisted of marked repetitive photostations, located on high relief areas comprised of coral and sponge species on the bank crest. These stations were photographed annually to monitor changes in the composition of benthic assemblages at each site over time. This study supported the addition of Stetson Bank into FGBNMS in 1996 (National Marine Sanctuaries Preservation Act [P.L. 104-283] 1996). From 1993 through 2014, GREAT, Texas A&M University - Corpus Christi (TAMU-CC), and FGBNMS researchers continued benthic monitoring on the bank crest (from 17-33.5 m), using original repetitive photostations as well as installing new photostations. Of the initial 36 repetitive photostations installed in 1993, 18 were still in use in 2015. These photostations have been critical in documenting and characterizing major shifts in the community structure at Stetson Bank by enabling reoccurring analyses of the same locations, thereby controlling for small-scale environmental heterogeneity (Côté et al. 2005). However, as they were intentionally located on high relief features on the bank crest, they are not representative of the bank community as a whole. This issue was investigated once in 1994 with the collection of random images in the vicinity of the mooring buoys. In order to capture spatial and temporal variations representative of the entire bank, annual random benthic transects were added to the survey protocol in 2013. Random transects were distributed between the two habitat types observed on the bank crest at Stetson Bank: high and low relief. In 2015, BSEE and FGBNMS entered into an interagency agreement (E14PG00052) to fund the continuation and expansion of monitoring efforts. In addition to continuing repetitive photostations and random transects the agreement expanded benthic monitoring to include mesophotic reef characterization and monitoring (Appendix A: Table A.1).

To date, long-term monitoring of the benthic community at Stetson Bank has archived 23 consecutive years of repetitive photostation imagery on the bank crest, one non-consecutive and three consecutive years of random transect imagery characterizing the benthic community in high and low relief habitats on the bank crest, and one year of repetitive photostation imagery of biologically interesting sites and random transect imagery of mesophotic communities.

Methods

Random Transects

In 1994, individual random photographs were collected around the bank crest at Stetson Bank. These images were collected in the vicinity of mooring buoys 1, 2, and 3, and each is a downward-facing image of the seafloor obtained using a Nikonos V film camera with 15-mm lens, mounted on a 1.06 m T-frame with strobes, set 0.76 m apart. A total of 180 images were captured and analyzed using random point counts in Downey and Boland (1994). These random images were not included in this report due to the lack of information regarding the sample design, but may provide an interesting independent study at a later date.

Bank crest stratified random benthic transects were conducted from 2013 to 2015. Transect sites were selected within high and low relief habitat, defined using 1 m^2 resolution bathymetric data. Depth range was calculated with a 5 m x 5 m rectangular window, and reclassified to define low relief habitat (<1 m range) and high relief habitat (>1.1 m range). A 33.5 m contour was used to restrict the extent of the range layer, limiting surveys to within depths that would allow divers sufficient time to conduct surveys within no-decompression limits. A total of 30 random surveys were conducted annually, distributed proportionally by area between habitat types: 10 low relief sites (two transects per site) and five high relief sites (two transects per site) (Figure 2.1). Site selection was conducted using ESRI's ArcGIS 10.3. Each transect was designed to capture at least 8 m^2 of benthic habitat, matching methods used in monitoring at East and West FGBs (Johnston et al. 2015a). A still camera, mounted on a 0.65 m T-frame with a bubble level and strobes, was used to capture non-overlapping images of the reef. Each image captured approximately 0.8 x 0.6 m (0.48 m^2), requiring 17 images to obtain the desired coverage (8.16 m^2). Spooled fiberglass 15 m measuring tapes with 17 pre-marked intervals (every 0.8 m) were used as guides, providing a 0.2 m buffer between each image to prevent overlap. A Canon Power Shot® G11 digital camera in an Ikelite® housing with a 28 mm equivalent wet mount lens adaptor, with two Inon® Z240 strobes set 1 m apart on the T-frame, was used.

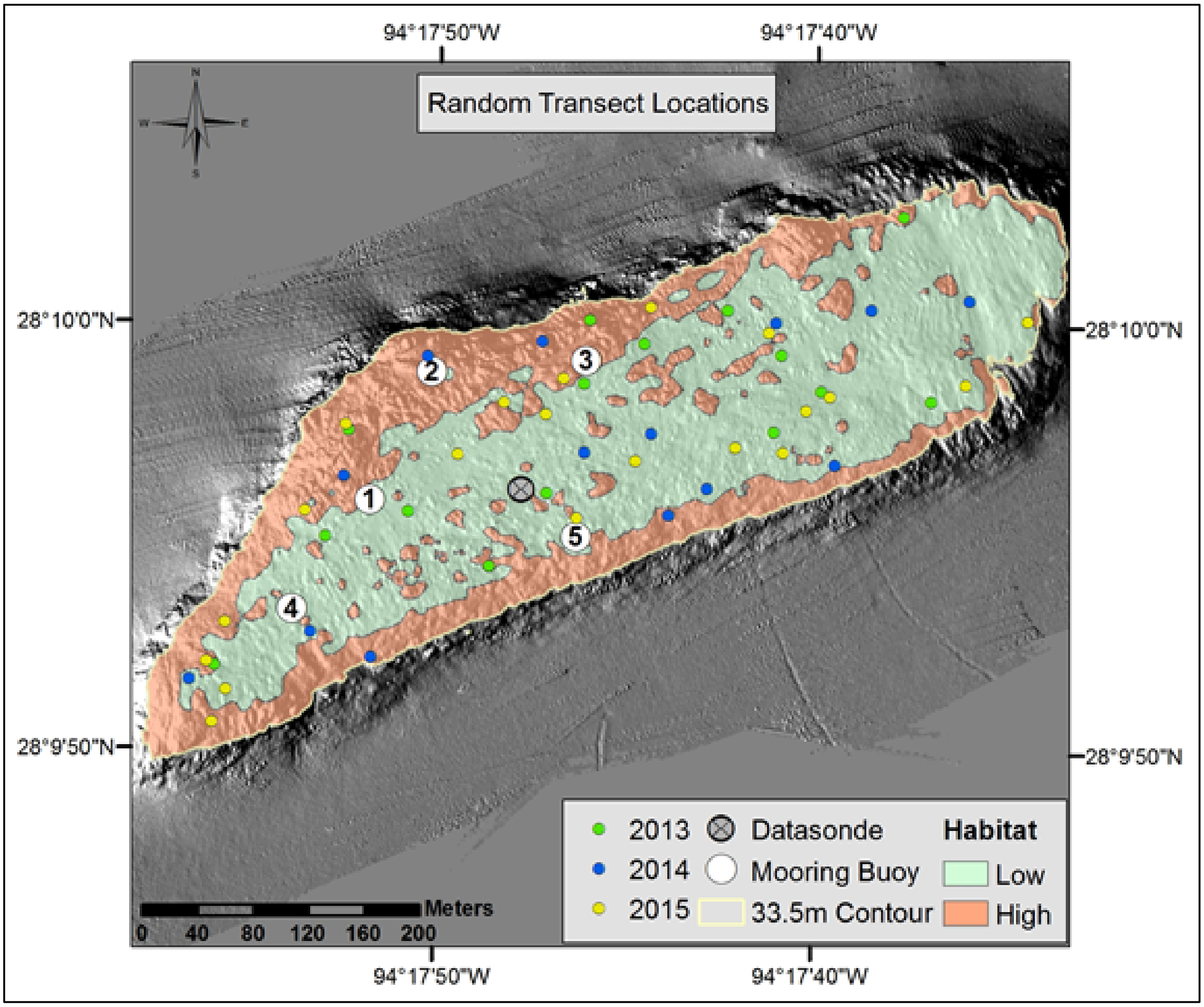
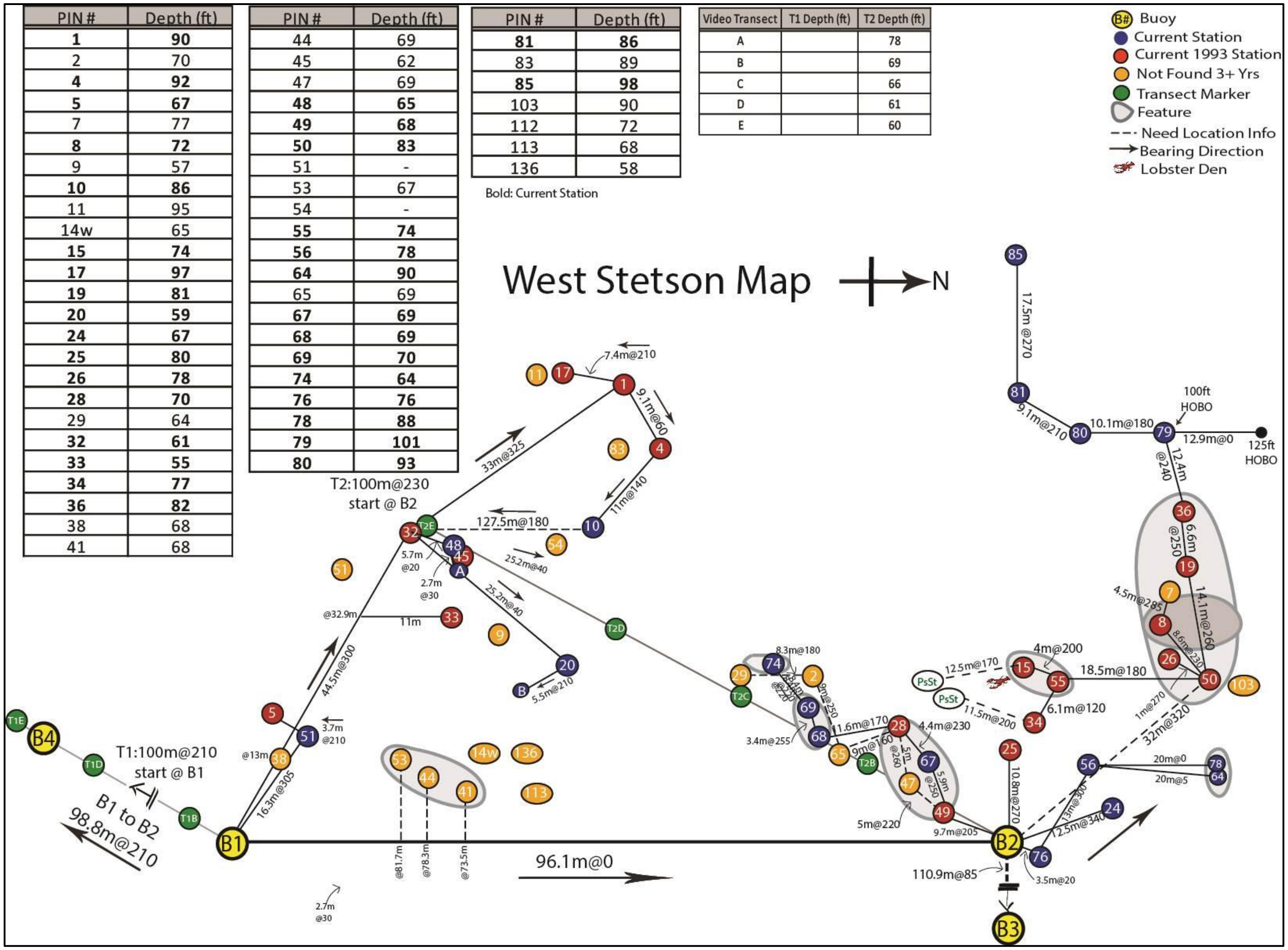


Figure 2.1. Location of bank crest random transect surveys from 2013 through 2015. Green dots represent 2013, blue represent 2014, and yellow represent 2015. High and low relief habitat is denoted with orange for high relief and green for low relief. White circles represent mooring buoys and are labeled with their respective number. The gray circle marks the location of 25 m water monitoring datasonde. Image: NOAA

Repetitive Photostations

Bank crest repetitive photostations were selected within an area of dense high relief features harboring diverse benthic and fish communities and marked by scuba divers using nails or eyebolts and numbered tags. Permanent mooring buoys 1, 2, and 3 were used for this study to enable easy access to the site by scuba divers. Stations were located by scuba divers using detailed maps and photographed annually (Figure 2.2, Figure 2.3). Repetitive photostations were installed on biologically interesting locations on high relief habitat, which included sites with scleractinian corals and sponges. In 1993 a total of 36 permanent photostations were installed. Over time several of these stations have been lost due to the dynamic environment at Stetson Bank (algal overgrowth, storm impacts, and fragile substrate). To maintain a sufficient number of repetitive photostation sites, new stations have been established, as needed, following the same selection criteria. In 2015 a total of 59 stations were photographed, 18 of which were original stations from 1993. While 18 original stations persist, several of these stations have been lost and rediscovered or reinstalled over time and therefore do not necessarily provide a continuous data record. A total of 14 stations have nearly continuous data record from 1994 to 2015, missing no more than one consecutive year.



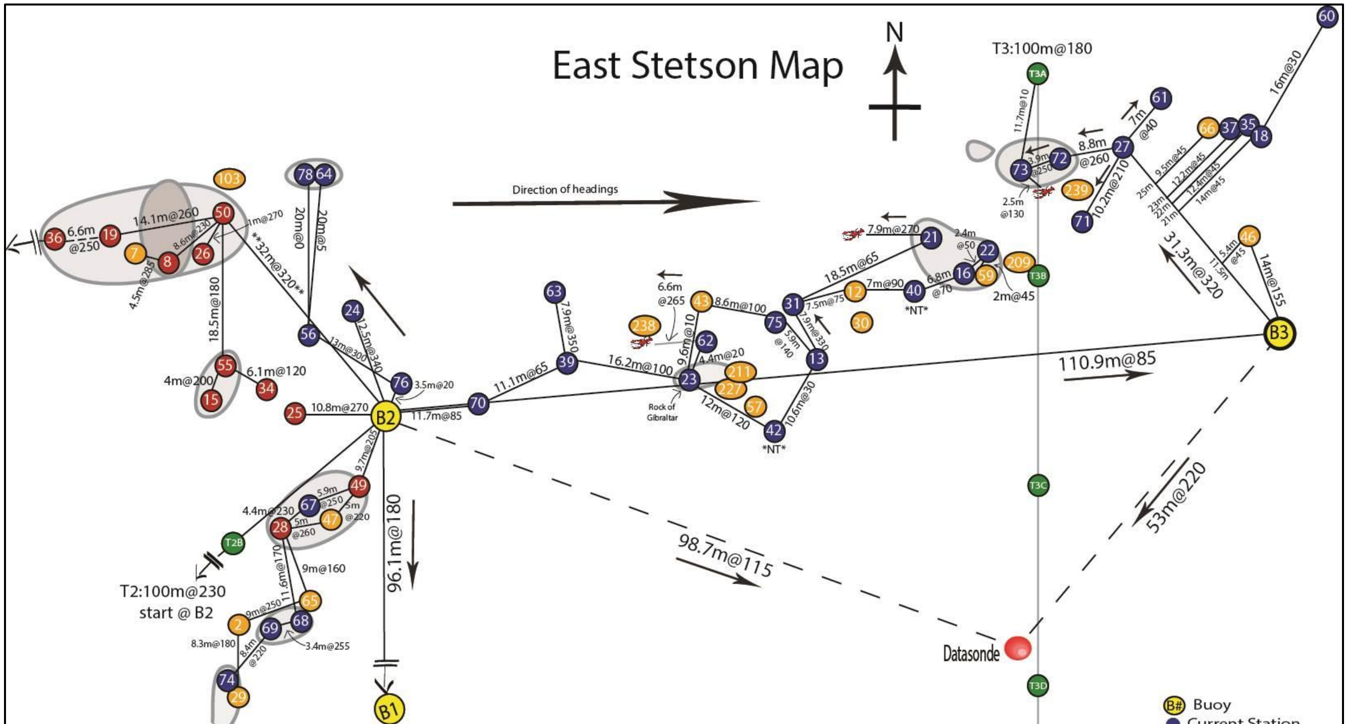
PIN #	Depth (ft)
1	**90**
2	70
4	**92**
5	**67**
7	77
8	**72**
9	57
10	**86**
11	95
14w	65
15	**74**
17	**97**
19	**81**
20	**59**
24	**67**
25	**80**
26	**78**
28	**70**
29	64
32	**61**
33	**55**
34	**77**
36	**82**
38	68
41	68

PIN #	Depth (ft)
44	69
45	62
47	69
48	**65**
49	**68**
50	**83**
51	-
53	67
54	-
55	**74**
56	**78**
64	**90**
65	69
67	**69**
68	**69**
69	**70**
74	**64**
76	**76**
78	**88**
79	**101**
80	**93**

PIN #	Depth (ft)
81	**86**
83	89
85	**98**
103	90
112	72
113	68
136	58

Video Transect	T1 Depth (ft)	T2 Depth (ft)
A		78
B		69
C		66
D		61
E		60

Figure 2.2. West Stetson map of repetitive photostations. Blue and red circles denote current station, with red representing original stations from 1993 that are still in use. Orange circles denote stations that have been lost. Image: NOAA



PIN #	Depth (ft)	PIN #	Depth (ft)	PIN #	Depth (ft)	PIN #	Depth (ft)
2	70	26	78	46	70	66	77
7	77	27	77	47	69	67	69
8	72	28	70	49	68	68	69
12	69	29	64	50	83	69	70
13	64	30	68	55	74	70	73
15	74	31	67	56	78	71	-
16	70	34	77	57	62	72	70
18	77	35	81	59	72	73	70
19	81	36	82	60	80	74	64
21	76	37	79	61	74	75	71
22	72	39	72	62	73	76	76
23	62	40	73	63	74	78	88
24	67	42	65	64	90	103	90
25	80	43	79	65	69	209	-

PIN #	Depth (ft)
211	-
227	-
238	-
239	-

Bold: Current Station

Video Transect	T2 Depth (ft)	T3 Depth (ft)
A	78	95
B	69	70
C	66	77
D	61	77
E	60	77

Figure 2.3. East Stetson map of repetitive photostations. Blue and red circles denote current station, with red representing original stations from 1993 that are still in use. Orange circles denote stations that have been lost. Image: NOAA

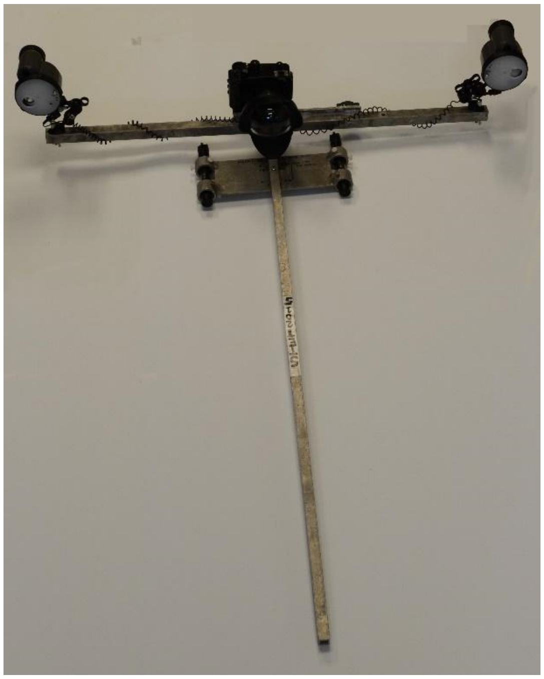


Figure 2.4. 2015 T-frame configuration. A G11 Fisheye FIX® housing mounted to the frame, set at 1.5 m from the substrate, with two Inon® Z240 strobes, set 1 m apart. Photo: NOAA

Camera and strobes were mounted on a "T-frame" with a bubble level and compass (Figure 2.4). Stations were photographed annually in the same orientation (north) to capture 1.6 m^2 coverage of the site. From 1993 to 2007, a Nikonos V film camera with 15-mm lens affixed to a T-frame with a 1 m pole length was used, and effectively captured approximately a 1.6 m^2 field of view. In 2008 and 2009, a Nikon Coolpix P5000 digital camera, housed in an Ikelite underwater housing, with a wide-angle adapter, was used on a T-frame with 1.25 m pole length providing 1.86 m^2 of coverage. From 2010 to 2015, a Canon G11 camera inside a FIX fisheye housing fitted with a 165-degree dome port was used on a T-frame with a 1.5 m pole length that provided 7.68 m^2 coverage. This setup included a set of lasers, fixed at 30 cm apart, to provide a spatial scale in the photo. Film slides dating from 1993 to 2008 were commercially developed and digitized by scanning at 1,200 pixels per inch (Nikon LS1000). Digital still images from 2008 to 2015 were downloaded from the camera and imported into Adobe Photoshop for image distortion correction and cropping to obtain 1.6 m^2 of benthic coverage.

Mesophotic Random Transects

Stratified random transects were conducted in colonized mesophotic reef habitat in 2015. ESRI's ArcGIS 10.3 was used to select transects sites within potential hard bottom habitat, defined using 2 m^2 resolution bathymetric data. Range was calculated with a 4 m x 4 m rectangular window, and reclassified to define potential hard bottom (>1 m range). Area shallower than 33.5 m was excluded from the layer. In alignment with the Coastal and Marine Ecological Classification Standard, the substrate was identified as rock substrate and the biotic community was identified as colonized mesophotic reef (CMECS 2012). The habitats were further classified using FGBNMS habitat classification scheme into two: coralline algae reef and deep reef (Schmahl et al. 2008). Areas on the flanks of the main reef feature were identified as coralline algae reef due to the high abundance of crustose coralline algae. Outcrops surrounding the main reef feature were identified as deep reef owing to the lack of crustose coralline algae. Thirty transects were randomly distributed within the polygon. Each point, representing the start location of a transect, was generated with a minimum of 30 m between sites (Figure 2.5). Surveys were conducted using an ROV with a downward-facing still camera. A SubAtlantic Mohawk 18 ROV was used for these surveys and was equipped with a Kongsberg Maritime OE14-408 10 MP digital still camera, OE11-442 strobe, and two Sidus SS501 50mW green spot lasers set at 10 cm in the still camera frame for scale. Transects started at each of the random drop sites and continued for 10 minutes along hard bottom habitat. The ROV traveled 1 m above the bottom, at a speed of 0.5 knots, taking downward facing still images every 30 seconds during the transect. Transect images were processed to remove silted, shadowed, out of focus, or soft bottom images. From the remaining images, 11 images were randomly selected for processing (if each transect did not have at least nine useable images, it was removed from the analysis). The size of each image was calculated in ImageJ.

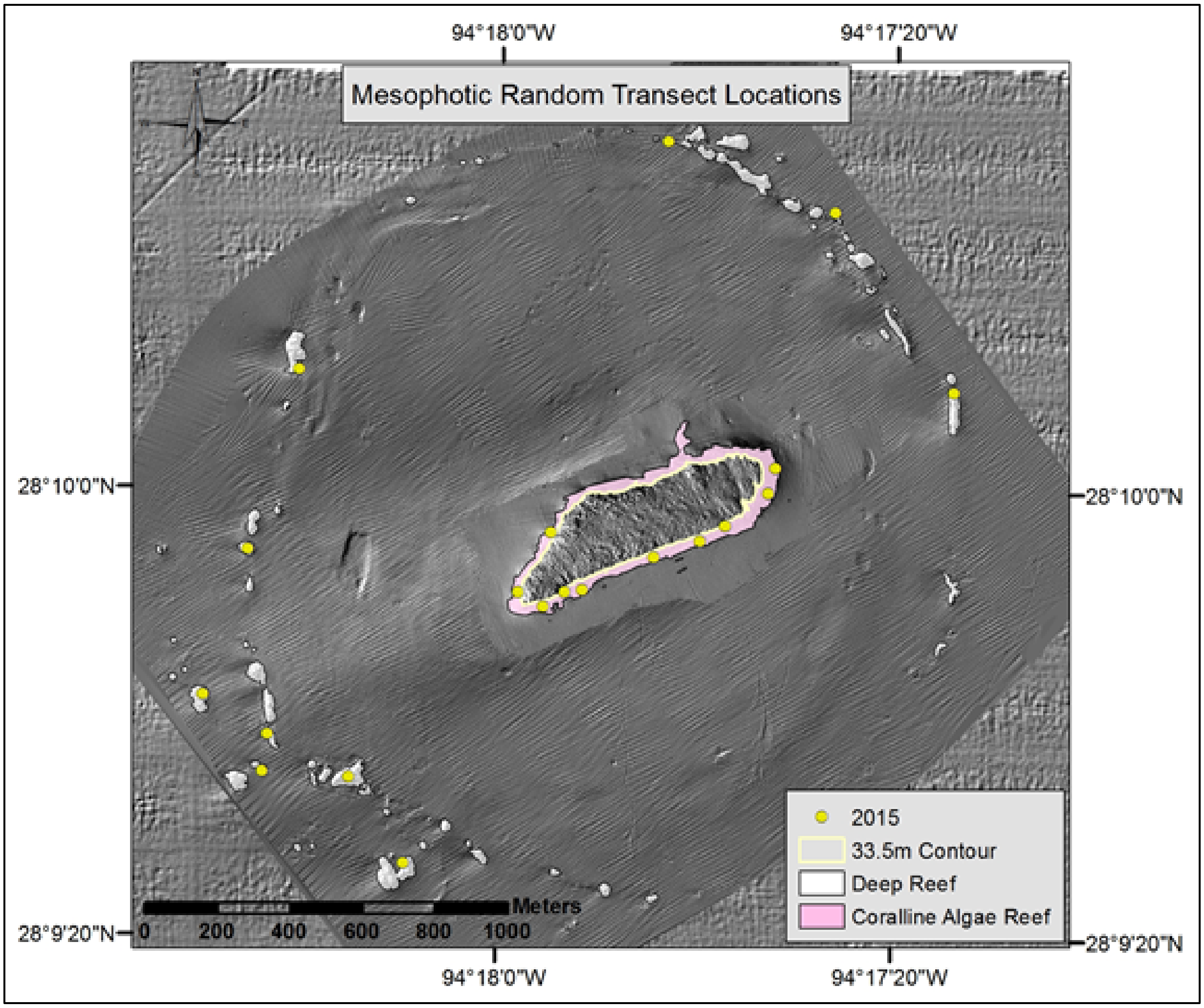


Figure 2.5. Location of mesophotic random transect surveys from 2015. Deep reef habitat is denoted with white and coralline algae reef with pink. Image: NOAA

Data Processing

All images were analyzed for percent cover using Coral Point Count (CPCe), provided by the National Coral Reef Institute (Kohler & Gill 2006). For stratified random transects (both bank crest and mesophotic), each transect was treated as a sample with a minimum of 500 spatially random points distributed evenly within the transect (for example, in a transect with 11 images, each image had 46 random points). Each image of bank crest repetitive photostations was treated as a sample with 30 spatially random points used for point count analysis. The number of random points selected to analyze each transect or sample was based on formulae presented in Hilliard and Cahn (1961), where organisms of interest were assumed to contribute to > 0.5% of the benthic cover.

For bank crest random transects and repetitive photostations, organisms positioned beneath each random point were identified to the lowest possible taxon. Bleaching was recorded as "notes" in CPCe. Data were summarized to ten functional groups, including: scleractinia, hydrocoral, Porifera (encrusting and erect), macroalgae (including algae longer than approximately 3 mm and thick algal turfs), colonizable substrate (formerly crustose coralline algae, fine turfs, and bare rock [CTB] [Aronson & Precht 2000]), other biotic (ascidians, fish, serpulids, and unknown species), rubble (coral and substrate rubble), soft substrate (sand and silt), other abiotic (tape measures, tags, research equipment, and marine debris), and no data (no data and shadows). Rubble, soft substrate, other abiotic, and no data classifications were excluded from data analysis.

For mesophotic surveys, organisms positioned beneath each random dot were identified to lowest possible taxonomic level for Cnidaria, Porifera, and macroalgae and other organisms were identified to the phyla level. Data were summarized to fourteen functional groups, including: Scleractinia, hydrocoral, Antipatharia, Octocorallia, Alcyonacea, Porifera (encrusting and free standing), macroalgae (including algae longer than approximately 3 mm and thick algal turfs), colonizable substrate (crustose coralline algae, fine turfs, and bare rock), other biotic (ascidians, fish, serpulids, and unknown species), rubble (coral and substrate rubble), silted hard bottom, soft substrate (sand and silt), other abiotic (tape measures, tags, research equipment, and marine debris), and no data (no data and shadows). Rubble, silted hard bottom, soft substrate, other abiotic, and no data classifications were excluded from data analysis as they do not represent significant benthic biota.

Random transects, when possible, were conducted to include two transects at each location, resulting in 95 transects. Before analysis percent cover data from these two transects was averaged by site, resulting in 50 samples. For repetitive photostations, two combinations of data were analyzed: all photostations and continuous stations. From 1993 to 2015, a total of 1228 samples, comprised of 96 unique stations, were collected (Figure 2.6; Appendix B: Table B.1; Table B.2). During years when multiple collections of repetitive photostation data occurred (1994, 1995, and 2009) stations were averaged by year, resulting in 1,114 samples. This comprised "all station" data. Of all these stations, 14 had a continuous record from 1994 to 2015, missing no more than one consecutive year (photostations 8, 16, 19, 20, 21, 22, 25, 26, 28, 31, 40, 49, 55, 70). Where one year was missing, to maintain the continuous data set the preceding and subsequent year were averaged to generate data for that station. This was done in order to maintain a statistically significant number of continuous samples and comprised the "continuous" data.

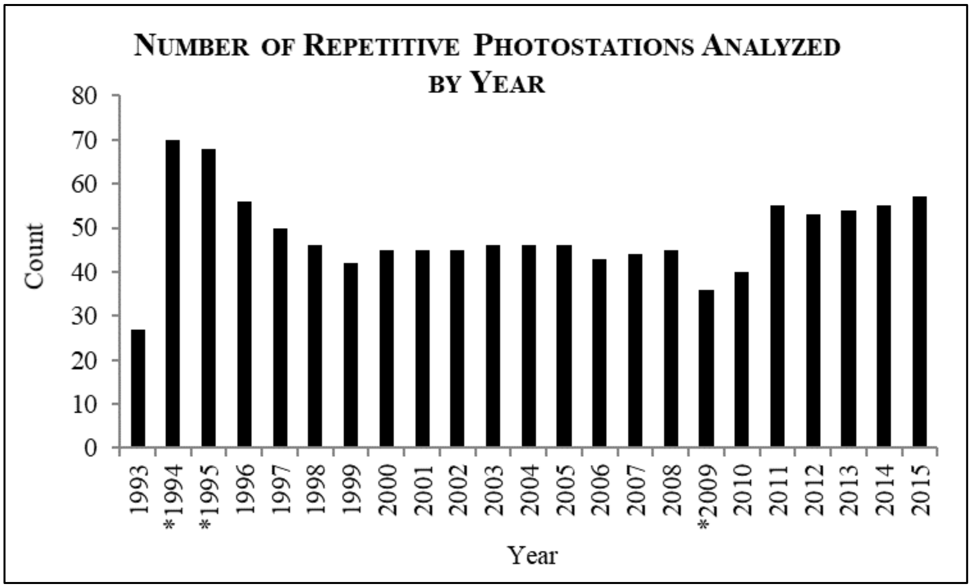


Figure 2.6. Repetitive photostations analyzed each year. Asterisks denotes years where two sample periods were averaged.

The long-spined sea urchin, *Diadema antillarum*, was counted in each random transect image and in the 14 continuous station images. In random transect images, counts were summed over the transect and density obtained per transect. On sites where two transects were conducted, urchin density was averaged between the two surveys, resulting in 50 samples. For continuous station data, urchin counts were performed on each image and density obtained per image. For years when multiple collections of repetitive photostation data occurred (1994, 1995, and 2009) sea urchin density was averaged by year. Density is presented as individuals per 1 m^2.

Since transects differed in area among mesophotic stratified random, weighted cover (percent cover multiplied by the area captured in the image) was used in analysis. In addition to point count analysis, colony counts for cnidarian species of interest (all cnidarians excluding hydroids) were conducted to the lowest possible taxonomic level for each image. Counts were summed across all images in a transect and presented as density per 1 m^2.

Statistical Analysis

Statistical analyses were only performed on bank crest data due to the limited sample size of mesophotic data. Functional category point count data from 2015 random transects were averaged between habitats types and presented as a summary of the current reef community. Details on the cover of species from the Scleractinia category in 2015 were presented as a summary of the current coral community at Stetson Bank.

Random transect data were tested for differences in benthic cover by habitat and year using non-parametric distance-based analyses. As percent cover data is non-binomial and in order to allow all variables to influence the analyses instead of being dominated by variables with the highest cover, data were fourth root transformed. Permutational multivariate analyses of variance

(PERMANOVA: Anderson et al. 2008) were based on Bray-Curtis similarity matrices. PERMANOVA represents a better alternative to ANOVA or MANOVA for ecological data as it does not assume Euclidean distance or normal distribution of the data. Habitat (2 levels: low relief and high relief) and year (3 levels: 2013-2015) were used as fixed orthogonal factors (sum of squares=Type 1, number of permutations=9999, permutation method= reduced model [crossed]). Where significant differences between years were found with PERMANOVA, contrasts between consecutive year were conducted and functional categories contributing to observed differences between years were examined using similarity percentages (SIMPER: Clarke 1993, Clarke et al. 2014) on fourth root transformed Bray-Curtis similarity matrices. SIMPER assesses the contribution of variables to the dissimilarity between groups.

Diversity measures (Shannon diversity [log base e], Pielou's evenness, and Margalef species richness) were calculated for each sample. These measures were analyzed together using a Euclidean distance similarity matrix, based on untransformed data, and tested for significant differences between year and habitat using PERMANOVA as described above, with contrasts for consecutive year testing.

Non-continuous data from 1993 and 2015 and continuous data from 1994 and 2015 were analyzed for significant differences in functional categories (excluding colonizable substrate) between 1994 and 2015 using PERMANOVA based on Bray-Curtis similarity matrices and fourth root transformed percent cover data. Year (2 levels: 1994 and 2015) was used as a fixed factor (Type 1 sum of squares, 9999 permutations, and unrestricted permutation method). Where significant differences were found, SIMPER was used to assess the contribution of each functional group to the observed dissimilarities.

Repetitive photostation data were averaged from 1994 to 2015 in both continuous and non-continuous datasets to reduce noise. Datasets were analyzed for year groupings using CLUSTER analysis, based on Bray-Curtis similarity matrices and square root transformed data (a less extreme transformation was used as the data were averaged and we did not want the analysis to be dominated by variables with the highest cover), and tested for significant clusters using similarity profile analysis (SIMPROF: Clarke et al. 2008, Clarke et al. 2014). Principal component ordination (PCO; Anderson et al. 2008) was used to visualize the data as the information could be projected well in low dimensional space and, unlike nMDS, percent variability is explained in each canonical axis. Multiple correlation vectors (correlation > 0.6) and temporal trajectories were overlaid on the PCO plot. Where significant clusters were found with SIMPROF, functional categories contributing to observed differences were examined using SIMPER on square root transformed Bray-Curtis similarity matrices.

Diversity measures (Shannon diversity [log base e], Pielou's evenness, and Margalef species richness) were calculated for each sample. These measures were analyzed together using a Euclidean distance similarity matrix, based on untransformed data, and tested for significant differences between year (fixed) and station (random) using PERMANOVA as described above, with contrasts for consecutive year testing.

Monotonic trends were tested for within datasets with >4 years of data. For parametric data (diversity measures), temporal trends were tested for using linear regression. For non-parametric data (percent cover), the non-parametric Mann-Kendall trend test was used.

Coherent species curves (Somerfield & Clarke 2013) were used to conduct r-mode analyses (an analysis of patterns among variables) and examine if species vary in significantly similar patterns through samples ordered naturally as a time series. Coherent species curves were analyzed for all datasets at species level, where data were averaged by year to reduce noise and square root transformed to prevent few species with the highest cover from dominating the analysis. Random transect data were not analyzed due to the limited number of sample years (3).

D. antillarum densities from both continuous repetitive stations and random transects were tested for differences between year and habitat/station using PERMANOVA. Continuous repetitive stations were tested for yearly trends using the Mann-Kendall trend test and both continuous station data and random transect urchin density data were tested for correlation with the Bray-Curtis resemblance matrix of untransformed macroalgal cover using PERMANOVA.

PERMANOVA, SIMPER, CLUSTER, SIMPROF, and PCO were performed in PRIMER version 7 with PERMANOVA+ add-in (Anderson et al. 2008, Clarke & Gorley 2015) while ANOVA, Student's t-tests, simple linear regression and correlation, and Mann-Kendall trend test were performed in R version 3.2.0 (R Development Core Team 2015). In the text, percent cover summaries are presented as the value ± standard error.

Where significantly different year groupings were observed in repetitive photostation data, qualitative comparisons were made for each photostation from the previous year, when available. Comparisons included notes on the loss, reduction, expansion, or gain of macroalgae and coral and sponge colonies and changes in their general condition.

Spatial interpolation of percent cover data from bank crest and mesophotic stratified random transects were mapped using inverse distance weighting (IDW). Interpolations were created without separating data by habitat, using a variable search radius and 12 points. Analyses were performed in ESRI's ArcMap version 10.4.

Results

Bank Crest Surveys – Random Transects

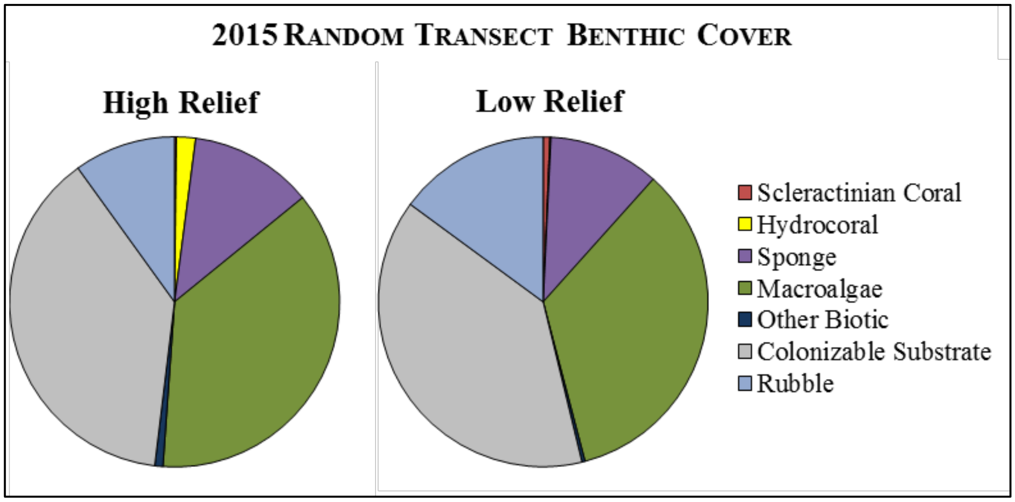


Figure 2.7. Percent composition of the six functional groups of benthic taxa.

A snapshot of the reef in 2015 indicates that the sessile benthic community in both high and low relief habitat was dominated by colonizable substrate and macroalga (high relief: 38.4% and 35.6%, respectively; low relief: 38.4% and 35.1%, respectively), while the main animal component was sponges (high relief: 12.5%; low relief: 10.4%) (Figure 2.7). Hydrocorals comprised a greater component of benthic cover in high relief habitat (1.9%) than low relief habitat (0.1%) and scleractinian corals contributed very low benthic cover in both habitats (high relief: 0.2%; low relief: 0.5%). One species contributed to hydrocoral cover, *Millepora alcicornis*, and five species contributed to scleractinian coral cover: *Stephanocoenia intersepta*, *Siderastrea radians*, *Madracis brueggemanni*, *Pseudodiploria strigosa*, and *Madracis decactis* (Figure 2.8).

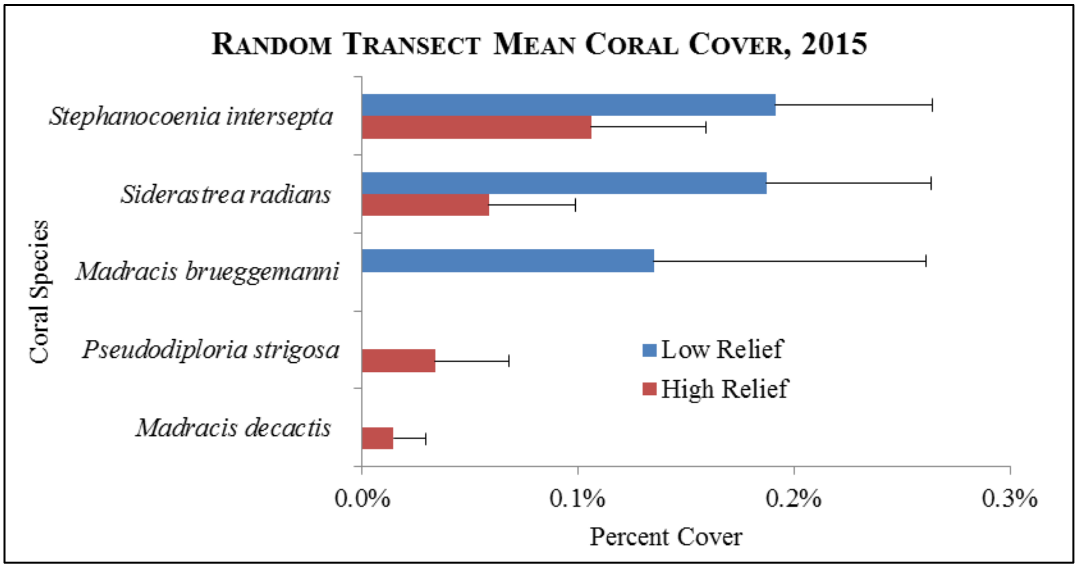


Figure 2.8. Mean percent cover of scleractinian corals in 2015 with standard error bars.

Random transect data (Appendix B: Table B.4; Table B.5) shows significant differences between habitat type (high and low) and year (2013-2015), with no significant interaction (Table 2.1). Across all years, hydrocoral and scleractinian cover contributed to over 50% of the observed dissimilarity between low and high relief habitats, where high relief habitat had greater cover of hydrocorals and lower cover of scleractinian corals than low relief habitat. Contrast comparisons between consecutive years showed significant differences between 2013-2014 and 2014-2015 (Table 2.2). Between 2013 and 2014, scleractinian coral and other biotic cover contributed most to the observed dissimilarity, where both variables declined between years. From 2014 to 2015, colonizable substrate and macroalgal cover contributed most to the observed dissimilarity, where colonizable substrate increased and macroalgae decreased between years.

Table 2.1. Main test PERMANOVA and SIMPER results for random transect major categories. Bold denotes significant values.

PERMANOVA				SIMPER			
Test	Pseudo-F	P(perm)	Unique Perms.	Functional Group	Average % Cover High Relief	Average % Cover Low Relief	% Contrib.
Habitat	7.06	**<0.001**	9966	Hydrocoral	1.39	0.12	32.19
				Scleractinian	0.18	0.42	21.51
Year	12.30	**<0.001**	9941	See Table 1.2			
Habitat x Year	1.14	0.370	9966	No Analysis			

Table 2.2. PERMANOVA yearly contrasts and SIMPER test results for random transects major categories. Bold denotes significant values.

PERMANOVA				SIMPER			
Test	Pseudo-F	P(perm)	Unique Perms.	Functional Group	Average % Cover Y1	Average % Cover Y2	% Contrib.
2013, 2014	3.27	**0.032**	9981	Scleractinian Coral	0.31	0.29	23.11
				Other Biotic	0.22	<0.01	21.52
2014, 2015	14.80	**<0.001**	9951	Colonizable Substrate	14.02	38.59	26.01
				Macroalgae	66.25	35.28	19.69

Diversity measure (Shannon diversity [H'loge], Margalef species richness [d], and Pielou's evenness [J']) similarities for each random transect were significantly different between year (p=0.02, pseudo-F=3.67) and habitat (p=0.05, pseudo-F=3.82), with no significant interaction. Contrasts between consecutive years found significant differences between 2014 and 2015 (p=0.01, pseudo-F=7.32) only.

Bank Crest Surveys – Repetitive Photostations

Repetitive photostation data for 1994 and 2015 were compared to examine changes in the benthic community between the beginning of monitoring and 2015. Significant differences in the community were observed in functional categories in both station and year (p<0.001, pseudo-f=1.81 and p<0.001, pseudo-f=35.51, respectively). No analyses for interactions were undertaken as there was insufficient replication at the lowest level in the design. Between years, hydrocoral and colonizable substrate cover contributed to over 50% of the dissimilarity (33.91% and 16.18%, respectively), where hydrocoral cover declined from 21.12% in 1994 to 1.74% in 2015 and colonizable substrate cover increased from 16.30% in 1994 to 48.77% in 2015 (Figure 2.9). Continuous station data revealed the same results and significant differences. Additional detail on yearly trends and covariation of variable is presented in the next paragraph.

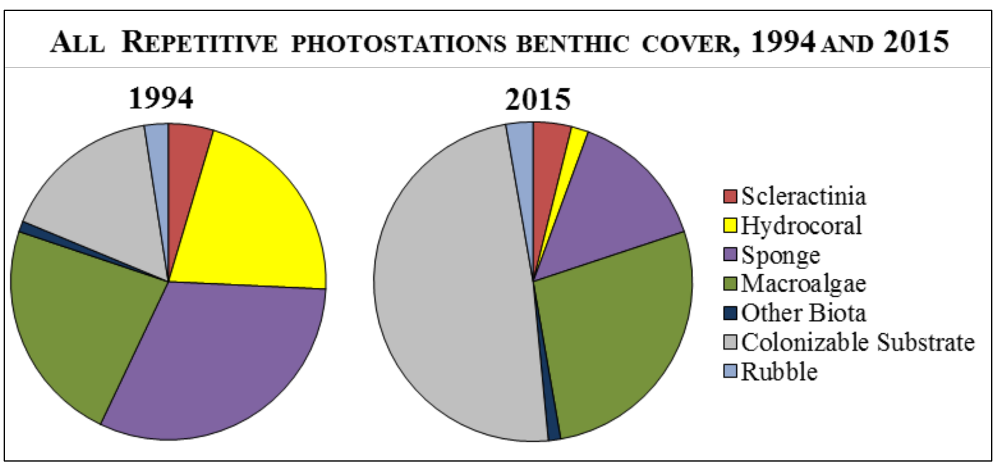


Figure 2.9. Percent composition of the six functional groups of benthic taxa in 1994 and 2015 from all repetitive photostations.

In yearly averaged data, all station data had two significant year clusters from SIMPROF analysis (A: 1993-2005; B: 2005-2015) (Figure 2.10). Between clusters A and B, hydrocoral and macroalgal cover contributed to over 50% of the dissimilarity (45.17% and 27.02%, respectively), where hydrocoral cover decreased from 22.58% in cluster A to 3.13% in cluster B and macroalgal cover increased from 28.76% in cluster A to 53.82% in cluster B.

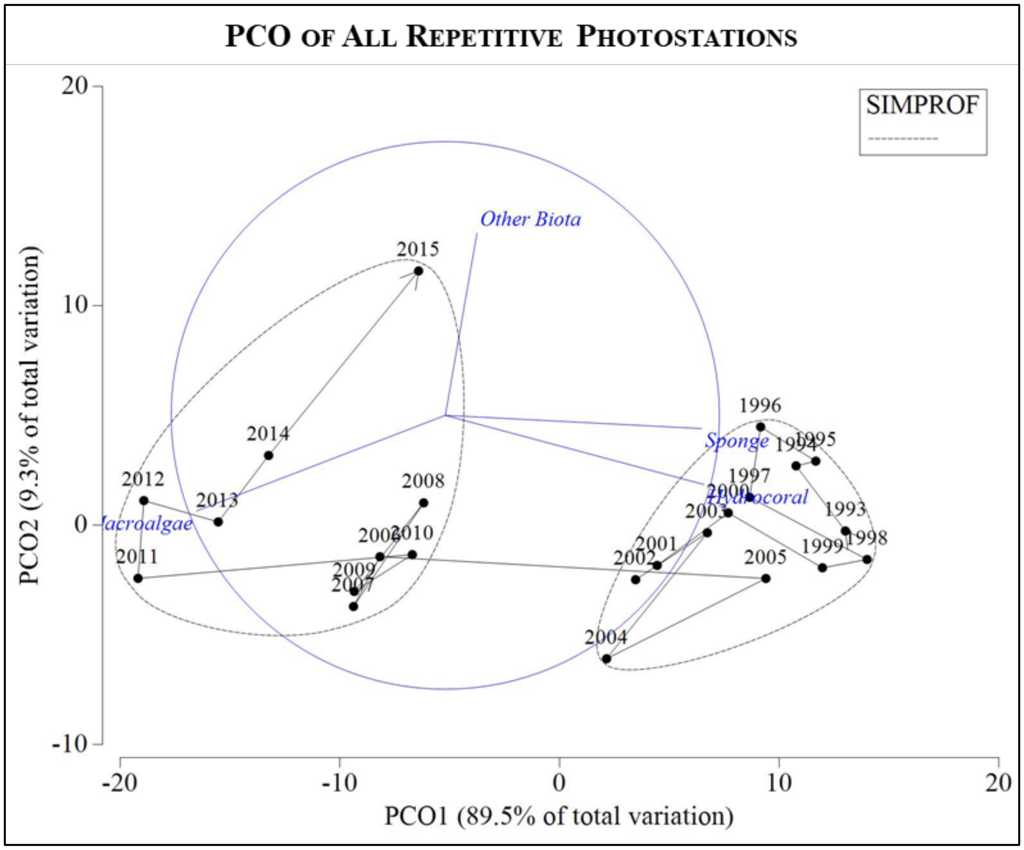


Figure 2.10. PCO for all repetitive photostations from 1993 to 2015. Blue lines represent variable vectors, with length and direction repressing direction of increase. The circle indicates the length of vector if 100% of that variable's variability is represented in the ordination plane. Convex hulls (contours) indicate groups of samples within which SIMPROF does not detect significant multivariate structure.

Continuous station data found additional significant clusters, significant changes in community composition (A: 1994-1996 and 1999; B: 1997 and 2000-2005; C: 2006-2010; D: 2011-2014; and E: 2015) and decreasing hydrocoral and increasing macroalgal cover contributed most to observed dissimilarities (Appendix B: Table B.6). Continuous station data also identified 2015 as significantly different from the other clusters (E: 2015) (Figure 2.11). Between clusters D and E, macroalgal cover contributed to 60% of the dissimilarity, where macroalgal cover in cluster D was 60.48% and in cluster E was 26.62%.

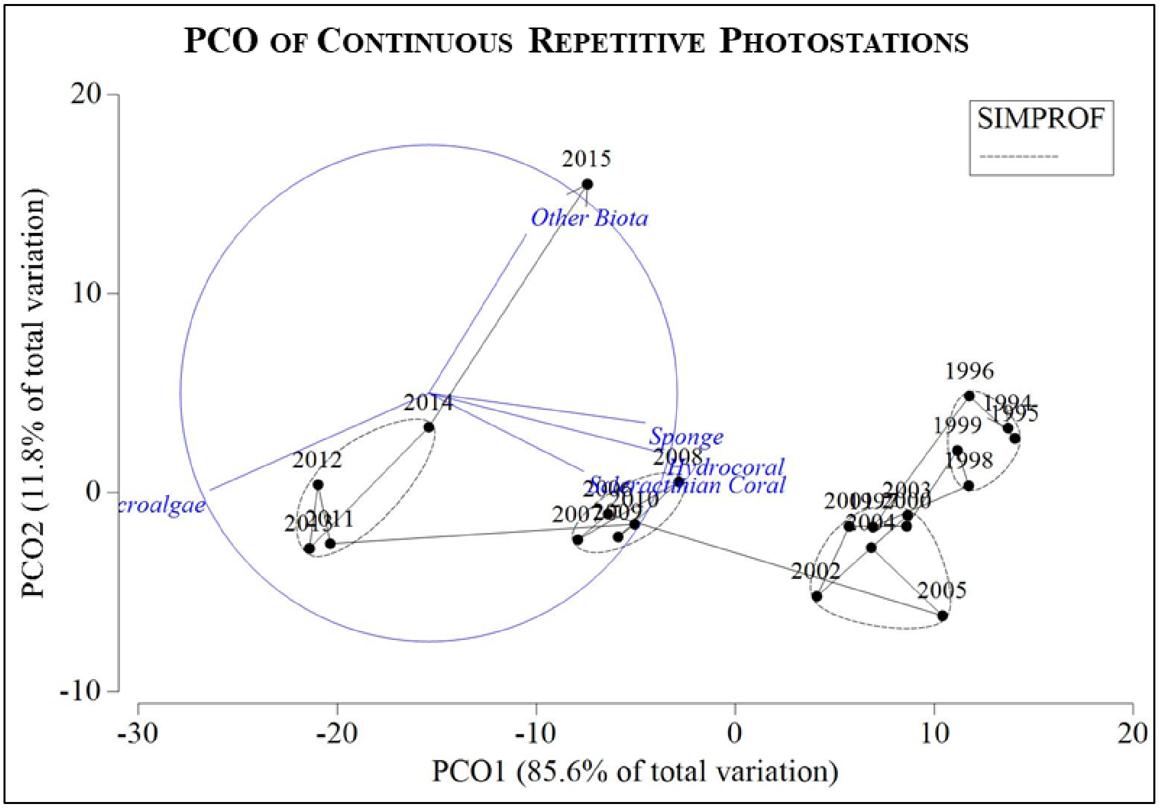


Figure 2.11. PCO for continuous repetitive photostations from 1994 to 2015. Blue lines represent variable vectors, with length and direction repressing direction of increase. The circle indicates the length of vector if 100% of that variable's variability is represented in the ordination plane. Convex hulls (contours) indicate groups of samples within which SIMPROF does not detect significant multivariate structure.

When comparing all station and continuous station data, significant differences occurred in diversity measures between year ($p<0.001$, pseudo-$F=3.98$ and $p=0.001$, pseudo-$F=2.09$, respectively) and station ($p<0.001$, pseudo-$F=5.01$ and $p<0.001$, pseudo-$F=9.39$, respectively). Contrasts were used to examine significant differences between consecutive years, finding different results between all station and continuous station data (Table 2.3).

Table 2.3. PERMANOVA yearly contrasts for all and continuous repetitive photostation diversity measures Bold denotes significant values.

Contrast	Continuous Stations			All Stations		
	Pseudo-F	P(perm)	Unique Perms.	Pseudo-F	P(perm)	Unique Perms.
1994-1995	1.41	0.249	9941	0.05	0.942	9951
1995-1996	0.68	0.448	9930	15.17	**<0.001**	9937
1996-1997	5.79	**0.029**	9920	3.33	0.067	9924
1997-1998	4.85	**0.043**	9929	12.30	**0.001**	9948
1998-1999	0.27	0.682	9935	1.27	0.269	9942
1999-2000	1.14	0.304	9923	0.09	0.834	9930
2000-2001	2.87	0.106	9939	4.11	**0.040**	9939
2001-2002	1.52	0.236	9936	2.75	0.092	9943
2002-2003	3.99	0.054	9941	5.54	**0.019**	9944
2003-2004	5.31	**0.023**	9942	3.59	0.055	9934
2004-2005	2.15	0.157	9932	1.56	0.203	9946
2005-2006	0.31	0.652	9938	1.43	0.237	9943
2006-2007	0.73	0.430	9933	0.26	0.698	9943
2007-2008	1.76	0.194	9938	2.48	0.108	9941
2008-2009	10.73	**0.004**	9926	3.17	0.072	9942
2009-2010	1.82	0.195	9926	1.68	0.194	9937
2010-2011	1.97	0.176	9925	0.54	0.495	9940
2011-2012	2.42	0.141	9920	0.06	0.906	9931
2012-2013	0.24	0.747	9949	6.42	**0.010**	9938
2013-2014	2.02	0.165	9929	0.88	0.363	9942
2014-2015	0.33	0.611	9934	4.63	**0.025**	9959

Linear regression showed no significant monotonic temporal trends in diversity measures in either all station data or continuous station data. Mann-Kendall trend analyses for all station and continuous station data identified significant negative monotonic trends in hydrocoral and sponge cover over time, as well as a significant positive monotonic trend in macroalgal cover over time. All station data identified one additional significant negative monotonic trend in Scleractinia cover (Figure 2.12; Table 2.4), potentially influenced by the loss of stations over time as this was not observed in the continuous station data.

Table 2.4. Mann-Kendall monotonic trend test results. Bold denotes significant values.

Functional Category	All Station Data		Continuous Station Data	
	τ	p	τ	p
Scleractinia	-0.37	**0.015**	-0.21	0.189
Hydrocoral	-0.44	**0.004**	-0.42	**0.007**
Sponge	-0.79	**<0.001**	-0.70	**<0.001**
Macroalgae	0.64	**<0.001**	0.61	**<0.001**
Other Biota	-0.51	0.751	-0.18	0.270
Colonizable Substrate	0.20	0.187	0.17	0.284

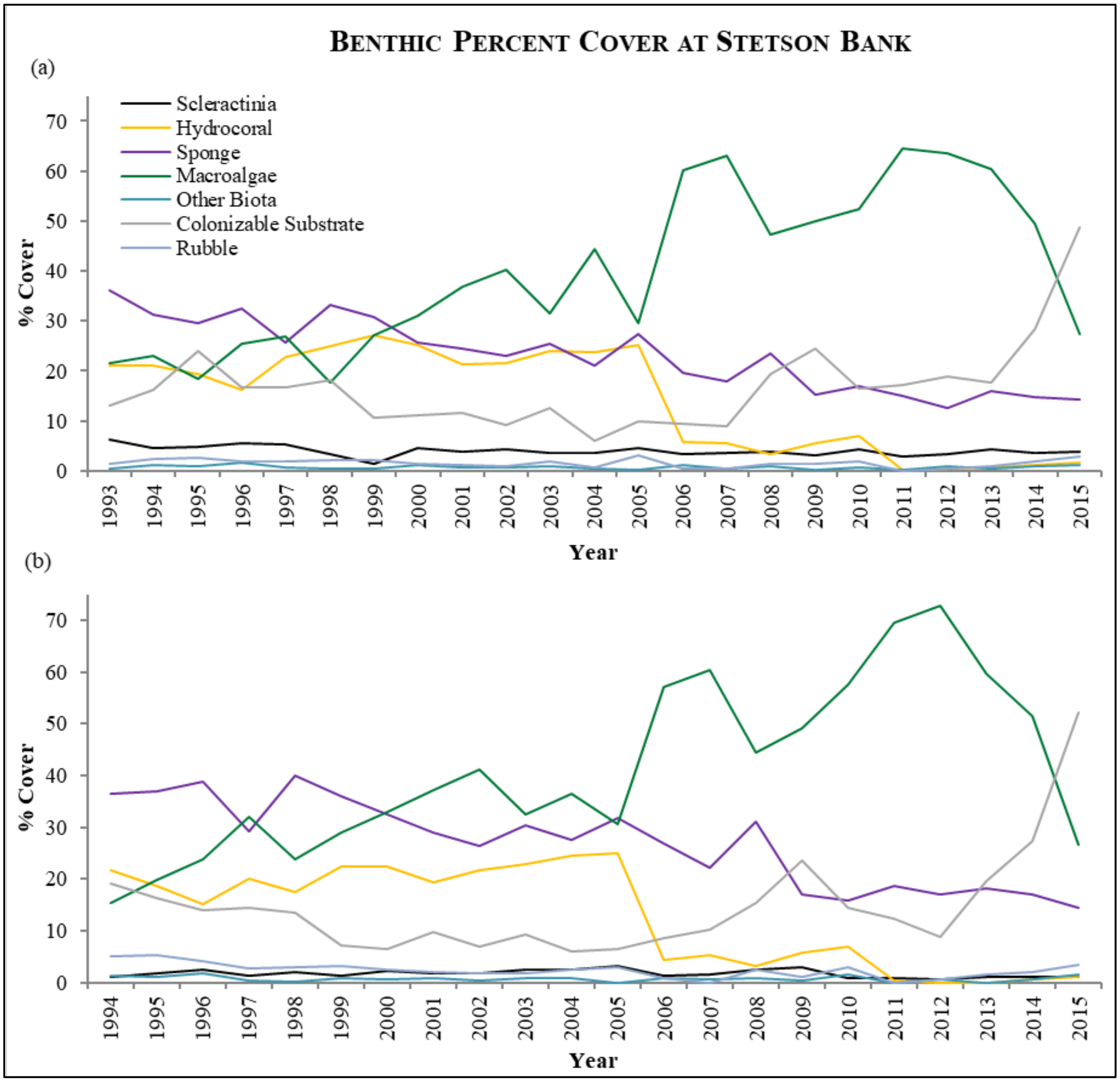


Figure 2.12. Benthic cover at Stetson Bank. A shows data from of all stations, from 1993 to 2015, and B shows data from continuous stations, from 1994 to 2015.

When all repetitive station data were analyzed, five coherent species groupings (A-E), containing two or more species, were found (Figure 2.13, with single species in Appendix B: Figure B.1). Group A contained species that appear to have increased in cover temporarily since 2007 with subsequent decline at different times, this group included a miscellaneous alga category and bare substrate/fine turf algae. Group B contained species that increased in cover following the 1998/2000 event through the 2005/2006 event and underwent decline shortly after, followed by a short-lived secondary increase and subsequent decline of two algal species (*Dictyota* spp. and turf algae matrix) and the sponge *Neofibularia nolitangere*. Group C contained species that underwent sudden and steep decline during the 2005/2006 event and have continued to slowly decline over time; this included the hydrocoral *M. alcicornis* and the sponge *Chondrilla nucula*. Group D contained species that tend to be present throughout the study period and did not show any significant temporal patterns; this included the scleractinian corals *Pseudodiploria strigosa* and *Madracis decactis,* several sponges (*Agelas clathrodes*, *Aiolochroia crassa*, and *Ircinia strobilina*), and crustose coralline algae. Group E contained species that were present sporadically throughout the monitoring period with no clear pattern. Continuous repetitive station data were more generalized but did not provide any additional coherent group information.

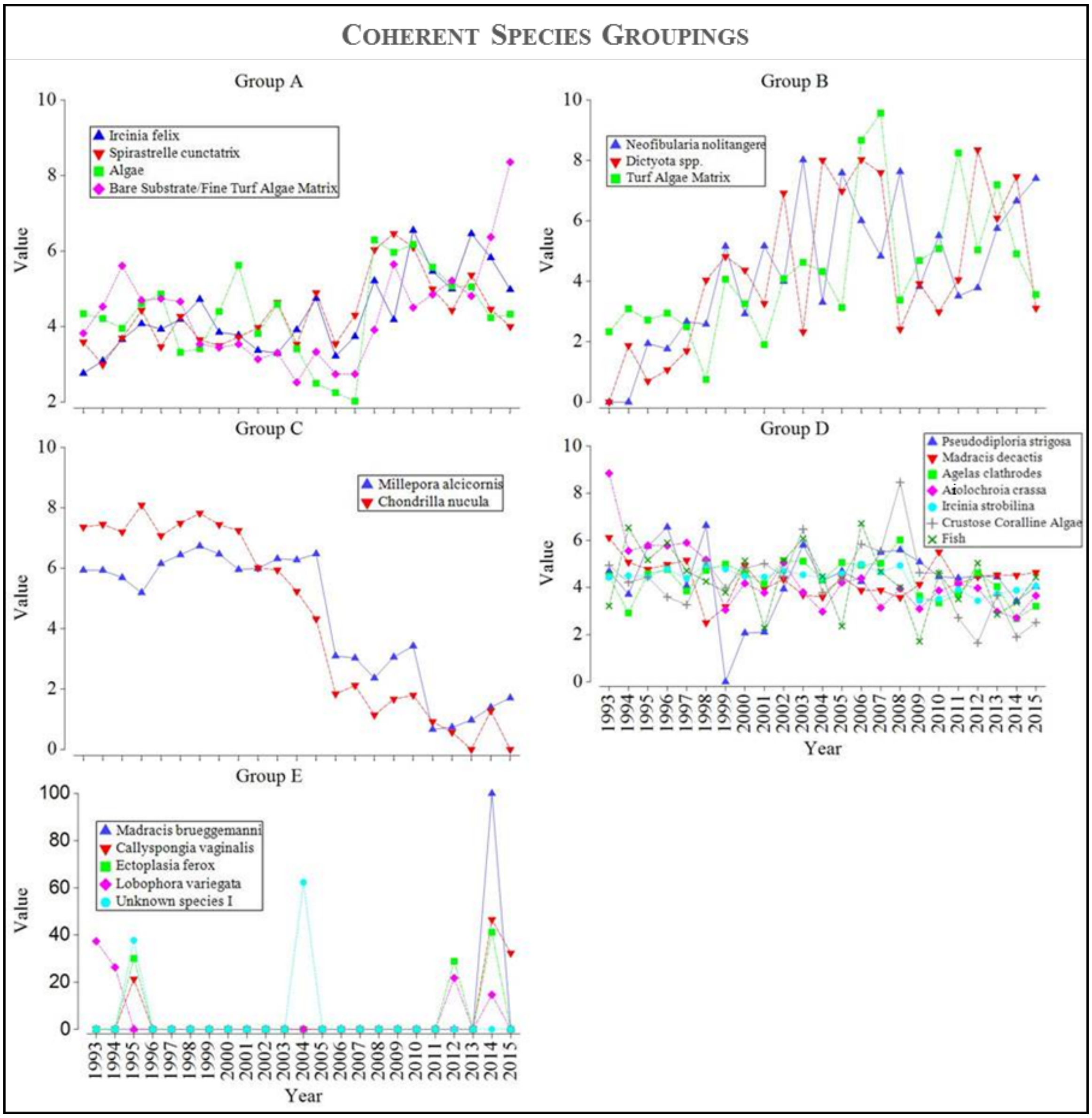


Figure 2.13. Coherent species groupings. Line plots for coherent species groupings A-E, with single species groups omitted. The y axis represents relative species abundance (square root transformed and standardized) and the x axis is sequential time.

Bleaching events have been anecdotally documented at Stetson Bank by divers in the fall of 2005 and 2010. Annual monitoring typically does not occur during peak bleaching times, and therefore these events are not always well captured in monitoring data. However, bleaching is recorded during processing of monitoring data and captured low levels of bleaching annually between 1993 and 2008 (Figure 2.14).

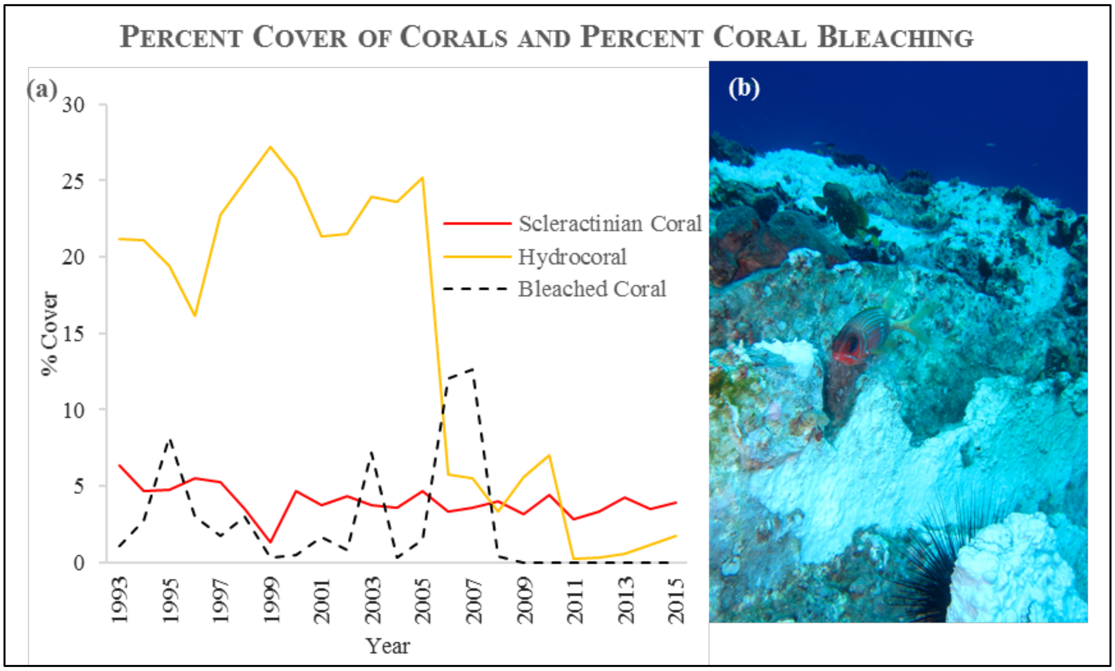


Figure 2.14. Percent cover of corals and percent coral bleaching. A: Graph of percent cover of corals and percent coral bleaching from all repetitive photostations, between 1993 and 2015. B: Bleaching of *Millepora alcicornis* at Stetson Bank in September, 2010. Photo: Emma Hickerson/NOAA

D. antillarum density in continuous station data (Appendix B: Table B.7) indicated that the population has varied over time with significant differences between years overall ($p<0.001$, Pseudo-F=4.68), but not between consecutive years. Urchin density peaked in 1996 and 2014 (1.38 and 2.72 individuals per 1 m^2, respectively) and showed no significant monotonic trends or significant correlation to macroalgal cover (Figure 2.15). Random transect data revealed a significant difference in urchin density between both year and habitat (year: $p=0.009$, Pseudo-F=4.76 and habitat: $p=0.028$, Pseudo-F=4.50), with no significant interaction. Contrasts between consecutive years showed a significant increase in urchin density between 2014 and 2015 ($p=0.0078$, Pseudo-F=2.7769). The largest density of *D. antillarum* in random transects was recorded in 2015 at 1.27 individuals per 1 m^2. Between habitat types, high relief habitat had greater average urchin density, with 1.29 individuals per 1 m^2, than low relief habitat, with 0.57 individuals per 1 m^2. In addition, a significant negative correlation was found between urchin density and macroalgal cover ($p<0.001$, pseudo-F=31.84 (Figure 2.16).

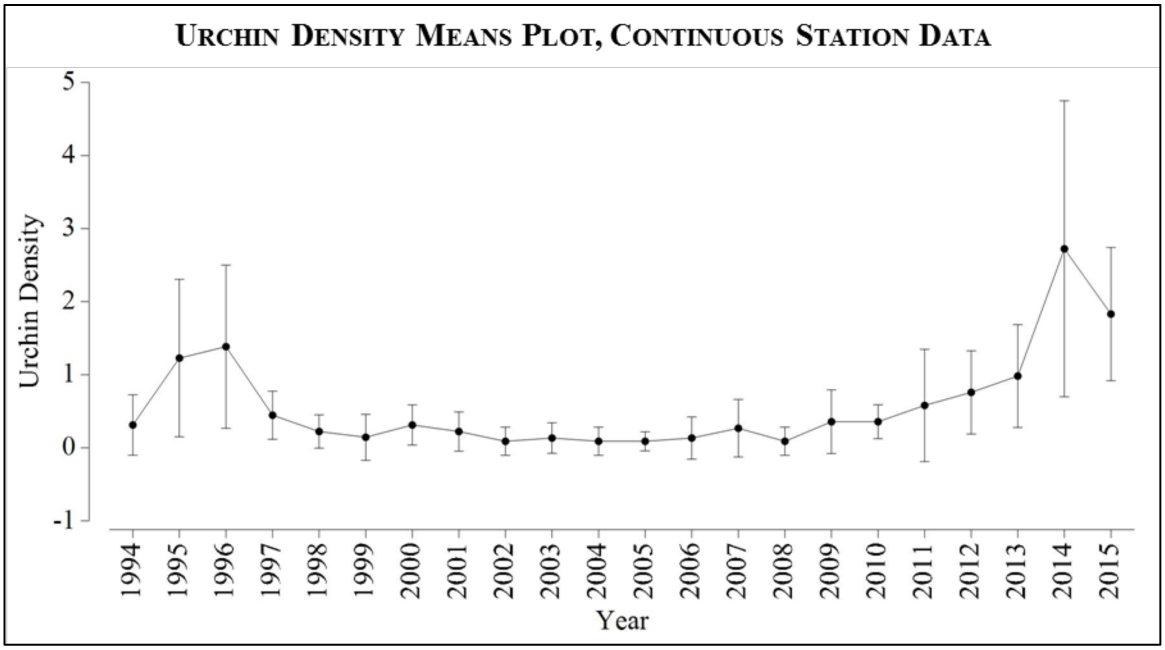


Figure 2.15. Means plot of *D. antillarum* density from continuous repetitive stations with standard error bars.

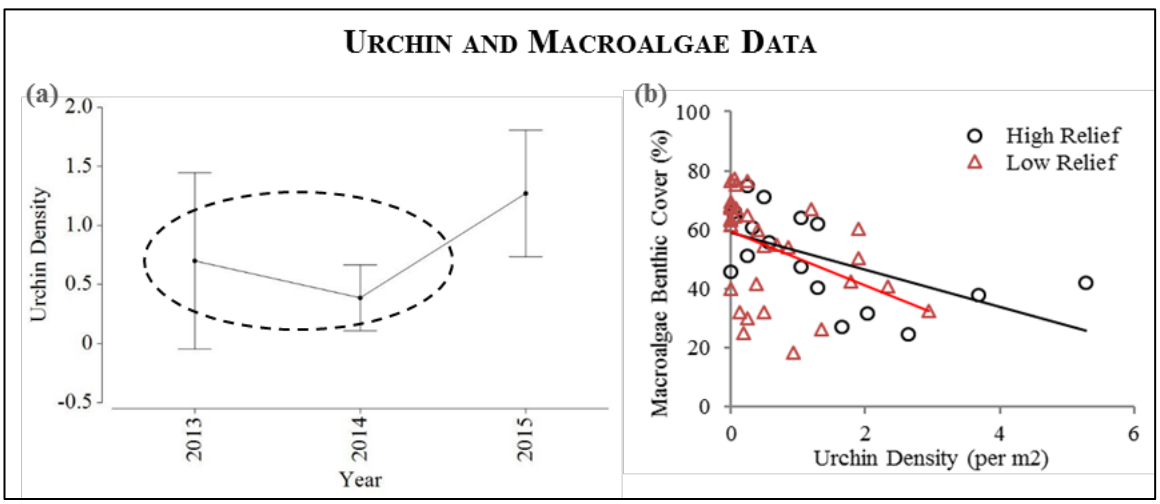


Figure 2.16. Urchin and macroalgae data. A: Means plot of *D. antillarum* density from random transect data with standard deviation bars, where dashed lines group consecutive years that are not significantly different. B: Plot illustrating negative correlation of *D. antillarum* density and macroalgal cover in high relief (black) and low relief (red) random transect data, where lines denote correlation, with black representing high relief habitat and red representing low relief habitat.

Qualitative comparisons (Appendix B: Table B.3) between years where significant changes in year clusters were observed indicate that, between 1998 and 1999, increased macroalgal cover was noted to be primarily due to increased abundance of *Dictyota* sp. In 2005-2006, increased macroalgal cover was primarily noted to have been a result of *Dictyota* sp. and turf algae overgrowth of former *M. alcicornis* colonies. Between 2014 and 2015, reduced macroalgal cover indicated declining cover of *Dictyota* sp. Notes on sponge colonies between the evaluated time frames are similar, documenting complete colony losses between years in addition to partial wasting in a variety of species.

Mesophotic Surveys

In 2015, the mesophotic benthic community in coralline algae habitat was predominantly colonizable substrate (39.8%) and deep reef habitat was predominantly silted hard bottom (37.2%). The main biotic component in coralline algae habitat was sponges (8.4%), primarily due to the abundance of encrusting sponges. The primary deep reef habitat biotic component was "other biotic" due to the abundance of hydroids (Figure 2.17).

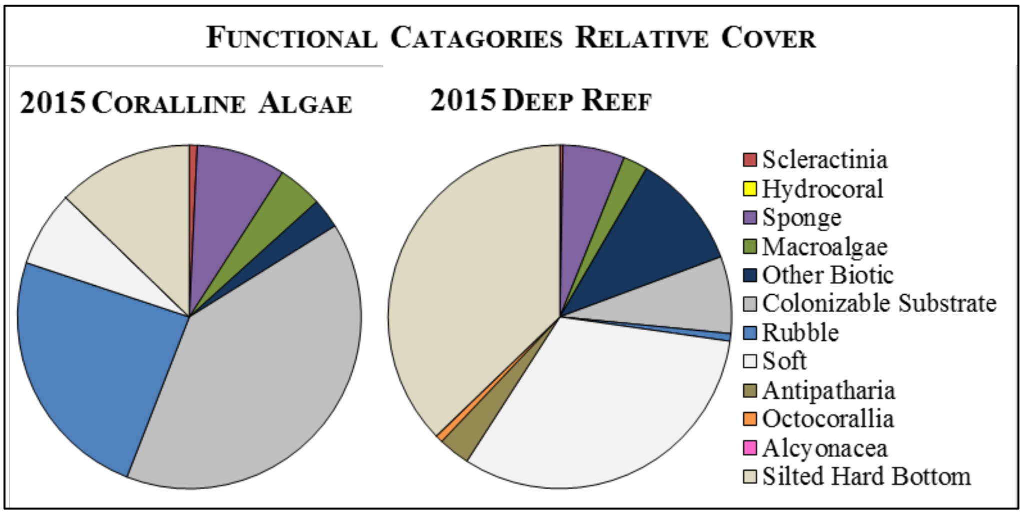


Figure 2.17. Relative percent cover of functional categories from coralline algae and deep reef habitats in 2015.

Of the cnidarian families (excluding hydroids), the densest family in deep reef habitat was Antipathidae with a mean of 3.57 individuals per m^2, due to the abundance of a black coral sea fan (potentially *Antipathes atlantica/gracilis*), which was entirely absent from coralline algae reefs. The densest colonies in coralline algae reef habitat were Astrocoeniidae at 1.35 individuals per m^2 (Figure 2.18), primarily due to the abundance of *Madracis brueggemanni* and *Stephanocoenia intersepta*.

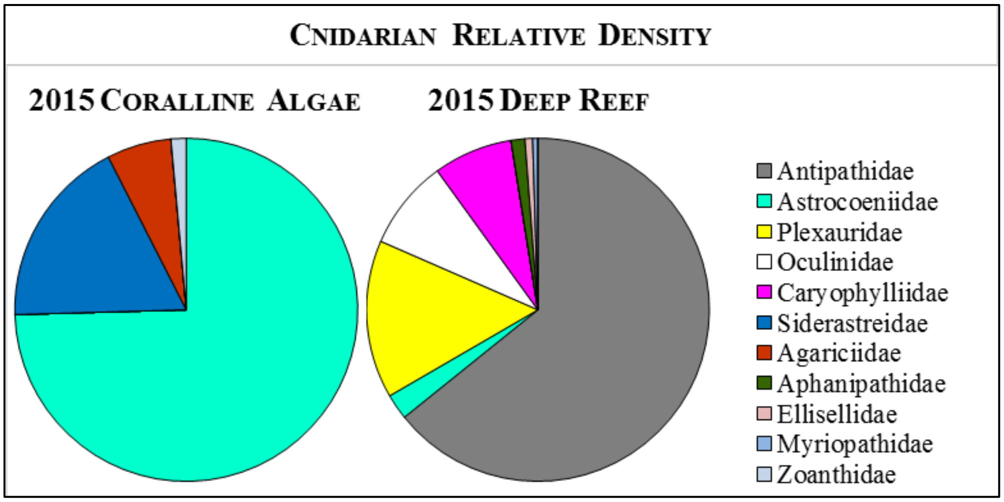


Figure 2.18. Relative density of cnidarian families of interest from coralline algae and deep reef habitats in 2015.

Spatial Interpolation

Spatial projection of benthic cover highlighted spatial patterns in several functional categories (Figure 2.19). In 2015 macroalgae and colonizable substrate were strongly associated with the bank crest and coralline algae reef habitat rather than deep reef habitat. Hydrocoral distribution was limited to the western portion of the bank crest, whereas Scleractinia had patchy distribution among all habitats. Sponges were found throughout all habitats. Antipatharia and Octocorallia were restricted to deep reef habitat, with Octocorallia being more prevalent in northeastern deep reef habitat.

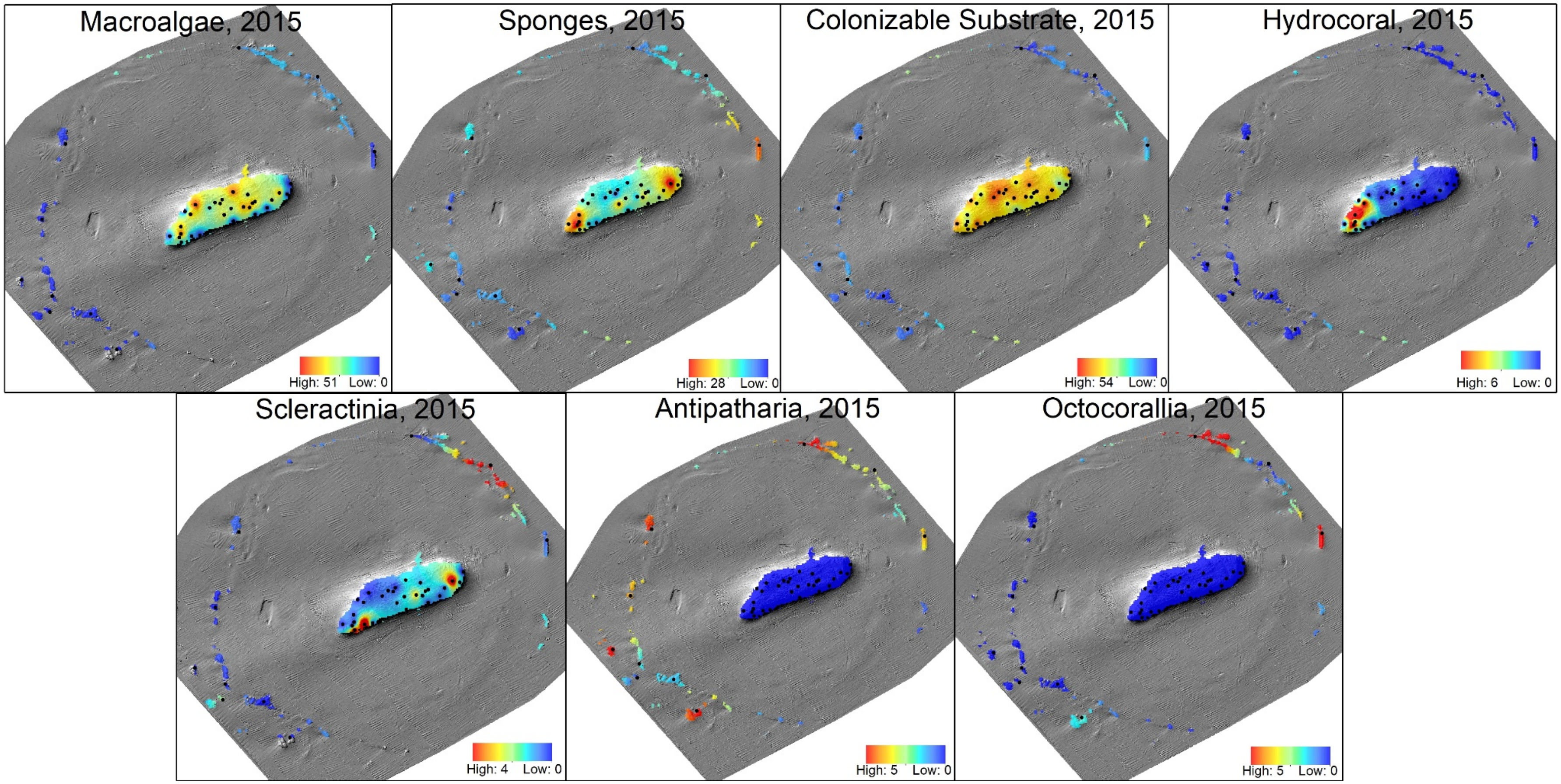


Figure 2.19. Inverse distance weighted benthic cover of select functional categories. Red represents high cover and blue represents low cover, and black dots mark the survey locations. Image: NOAA

Discussion

Periodic censuses can only provide snapshots of the community over time; however, various datasets presented in this report document that the sessile benthic community on the bank crest at Stetson Bank have undergone multiple significant changes since the initiation of the monitoring program. Repetitive photostation data show a habitat of predominantly hydrocoral (*M. alcicornis*) and sponges at the onset of monitoring in 1993, shifting to one of predominantly macroalgae in 2015. While the datasets show some variation, all agree a significant community shift occurred between 1998 and 2000, changing the reef community from one of predominantly *Millepora* and sponges to one of largely comprised of macroalgae. A distinct increase in algal cover was also noted during this time frame at the nearby East and West FGBs (Johnston et al. 2016a). Additionally, a large change in both scleractinian and hydrocoral cover was also documented at many reefs sites in the Florida Keys (Somerfield et al. 2008), the western Caribbean, and the Bahamas, which have been linked to the effects of bleaching and the 1998 El Niño-Southern Oscillation (ENSO) (Kramer 2003). Following 2000, the community appeared to stabilize rather than return to pre-1998 community structure, indicating underlying stressors were impacting the reef. This new community stability lasted through 2005, when a Caribbean-wide bleaching event caused extensive bleaching of hydrocorals at Stetson Bank. The resulting community underwent further increases in macroalgal cover and a sharp decline in hydrocoral cover. Following 2005, the community again did not recover to pre-2005 community structure and appeared to have increased variability between annual observations. Debose et al. (2013) also documented this year as a critical change in the community at Stetson Bank and suggested coastal runoff due to hurricane activity and extended elevated water temperatures contributed to reef stress and inhibited full recovery. One dataset (continuous repetitive photostations) supports a third significant change in the benthic community between 2010 and 2011 due to increasing macroalgae and declining sponges. Finally, two of the three datasets (continuous repetitive photostation and random transect) support a significant community difference between 2014 and 2015, due to reduced macroalgal cover and increased colonizable substrate cover. In 2015, macroalgae was still the predominant biotic component; however, colonizable substrate comprised the greatest benthic cover, representing the first time since 1999 that macroalgae was not the predominant benthic component.

The loss of hydrocoral cover at Stetson Bank has been observed to be linked to bleaching events, most notably in 2005 and 2010. While monitoring data were typically collected in early summer (May-July) before the full effects of bleaching were observed, bleaching information from monitoring data suggests that the corals at Stetson Bank exhibited low levels of bleaching (1-10% of coral showing bleaching) annually through 2008. Following 2008, these low levels of bleaching have not been documented on the reef. The absence of low-level bleaching in recent years is likely linked to the reduction in hydrocoral cover. *Millepora alcicornis* is the only hydrocoral and it is particularly susceptible to bleaching (Hagman & Gittings 1992, Marshall & Baird 2000, Fitt 2012). Similarly, low levels of bleaching have been reported annually at East and West FGBs in long term monitoring data, from 1989 to 2015 (Dokken et al. 1999, Dokken et al. 2002, Dokken et al. 2003, Precht et al. 2006, Zimmer et al. 2010, Johnston et al. 2013, Johnston et al. 2015b).

Significant declining trends were observed in multiple datasets for hydrocorals and sponges at Stetson Bank since the initiation of the monitoring program. While hydrocorals contributed to the significant community shift observed between 2005 and 2006, sponges have not been linked to a particular event, and have instead declined steadily over time. Also supported by multiple datasets, macroalgae showed a significant increasing trend since 1993, and was identified as contributing largely to the significant community changes in both 1998-2000, 2005-2006, 2010-2011 and 2014-2015. Despite significant changes and trends in the benthic cover, repetitive photostation data did not detect significant differences in community diversity measures, suggesting that changes in benthic cover were largely influenced by a single, or few, species. Conversely, random transect data detected significant changes in community diversity measures between 2013 and 2014, noting a reduction in diversity and evenness, where species diversity declined and benthic cover was dominated by fewer species.

Macroalgal cover at Stetson Bank has increased since the initiation of the monitoring program, with the highest cover reached in 2012, followed by a rapidly declining trend through 2015. Although algal cover is known to be highly dynamic and vary enormously by location (Diaz-Pulido & Garzon-Ferreira 2002, Bruno et al. 2009, Jackson et al. 2014), and season (Bertolino et al. 2016), the abundance of macroalgae is considered a key, typically negative, reef health metric (Steneck & Sala 2005) and plays an important ecological role in shallow reef ecosystems (Vroom et al. 2006). Over time, a significantly increasing trend of macroalgal cover has been documented at Stetson Bank, with coherence plots and qualitative observations highlighting *Dictyota* spp. and turf algae cover increases following both the 1998-2000 and 2005-2006 events, in addition to coherence plots highlighting the increase in miscellaneous algae following the 200-/2006 event. To better evaluate the recent decline in macroalgal abundance at Stetson Bank, continued monitoring over subsequent years, with increased temporal frequency, is needed. While the cause for the macroalgal decline and subsequent colonizable substrate increase has not been control tested, a potential factor is the existence of a robust herbivorous community supported by recent increasing density of the keystone grazer *D. antillarum* at Stetson Bank and a regional fishing focus on piscivorous fish, leaving herbivorous fish unaffected by direct fishing activities (Graham et al. 2003). Differences in *D. antillarum* density were observed between continuous repetitive station data and random transect data. As *D. antillarum* are considered patchily distributed organisms with their patchiness related to the complexity of the substrate (Tuya et al. 2004, Lessios 2016), the differences observed between datasets is potentially due to the larger area captured in random transect data compared to the small area, focused around high relief features, captured in continuous station data. Since 1994, *D. antillarum* density on Stetson Bank has been variable, with two notable peaks, one in 1996 and one larger peak recently between 2014/2015. As found in other studies in the Caribbean (Carpenter 1981, Sammarco 1982, Hay 1984, Lessios et al. 2001), a significant negative correlation between urchin density and macroalgal cover was found in both high and low relief habitats, highlighting the value of these species as keystone grazers and their potential top down control of macroalgal cover.

The benthic cover of sponges at Stetson Bank has shown significant steady decline since the start of the monitoring program. Sponges play multiple roles within the reef ecosystem, including increasing habitat complexity (Bell 2008). At the start of the long-term monitoring program at Stetson Bank, the community was sponge dominated. While species richness has remained similar over time, the predominant sponge from 1994, *C. nucula*, is no longer detected in repetitive photostation analysis. Coherence plots identify a similar declining trend of *C. nucula* to *M. alcicornis*, with a steady decline since 1998, followed by a steep decline following the 2005/2006 event and a steady decline thereafter. This sponge is a common dietary component of the hawksbill sea turtle (*Eretmochelys imbricata*), a species commonly observed at Stetson Bank (Meylan 1984, Rincon-diaz et al. 2011). The sponge *N. nolitangere*, found in high cover at Stetson Bank in 2015, has been noted to occur in disturbed habitats, and is occasionally included as a member of the fouling community. At Stetson Bank, *N. nolitangere* has shown similar cover trends to *Dictyota* spp. and turf algae, opportunistically filling available habitat following the 1998/2000 and 2005/2006 events. In comparison to other large Caribbean sponges (*I. strobilina* and *Agelas clathrodes*), *N. nolitangere* has the most rapid tissue regeneration rates but is also the most vulnerable to physical damage (Hoppe 1988). The sponge creates complex habitat for the sponge worm (*Haplosyllis spongicola*) and several fish species (Colin 1988) and is preyed upon by spongivorous angelfishes (Hoppe 1988).

The results presented in this report highlight the value of both continuous repetitive photostations and maintaining a large array of repetitive stations. Both datasets provided similar information; however, continuous stations emphasized small scale community changes that were not evident in the larger, all-station, dataset. This is due, in part, to the increased variability between stations when additional stations are used. Conversely, all-station data detected additional temporal trends that were not found in continuous station data, highlighting changes that have occurred due to lost photostations. These differences must be carefully interpreted because stations are often lost following mechanical impact to the reef (e.g., tropical weather systems and anchoring) or algal overgrowth, which impacts key biotic features that are characteristic of the individual station. Therefore, the benthic cover changes of these species may be real, not just an effect of station marker loss. The analysis of both datasets in this report demonstrate that while each can identify additional small changes, both datasets tell the same overall story. As there is a tendency to lose stations over time due to various events, this analysis supports that benthic monitoring information gathered from repetitive photostations is not lost as repetitive stations are lost.

The benthic cover reported in repetitive photostations and random transects differ greatly. The difference between the habitats and the different methods used to obtain the photographs for each of these datasets contributed to these differences. Repetitive photostations were located amongst the highest relief outcroppings (some with relief upward of 3 m) and selected because they had interesting biota (primarily coral and sponge) while random transects where delineated between high (>1.1 m) and low (<1 m) relief habitat and did not specifically target any biota. The primary difference reflected in the datasets is the reduced benthic cover of scleractinian coral in high relief random transect data in comparison to repetitive photostation data. Targeting scleractinian corals in repetitive photostations artificially increased their benthic cover when compared to randomized sampling. Repetitive photostations have been invaluable for documenting temporal changes at Stetson Bank and random transects will be continued in order to provide a greater representation of the entire community.

Random transects on the bank crest identified that the difference between high and low relief habitats at Stetson Bank was primarily due to the abundance of hydrocorals and scleractinian corals. High relief habitat has a higher cover of hydrocorals than low relief habitat, and low relief habitat has a higher cover of scleractinian corals than high relief habitat. Scleractinian coral cover in high relief habitat tends to be patchy, taking the form of isolated attached coral heads (*Porites astreoides*, *Pseudodiploria strigosa*, *Siderastrea radians*, and *S. intersepta*) or mounds of branching coral (*Madracis decactis*). Low relief habitat scleractinian coral cover tends to be more uniform with tiny attached coral heads (*S. radians*), and small free-living plating (*Agaricia fragilis*, *S. intersepta*) or branching corals (*M. brueggemanni*). These free-living colonies lie on the surface of the substrate and are moved by fish activity and currents (Pichon 1974). The presence of free-living colonies is considered to be an adaptive response to ecological conditions (Latypov 2007).

Recent monitoring efforts have expanded to characterize and monitor the mesophotic ecosystems surrounding Stetson Bank. Two distinct habitat types were encountered in this area, each with different communities. Coralline algae reef habitat was found primarily on the deeper flanks of the main bank feature and was defined by the presence of abundant crustose coralline algae, reflected in the relative cover of Rhodophtya. Cnidarians in coralline algae habitat were the third most dominant phylum, of which the family Astrocoeniidae comprised the highest cover and greatest density. Similar to low relief bank crest habitat, scleractinian coral species comprising the greatest cover were *M. brueggemanni* and *S. intersepta*, from the Astrocoeniidae family, and *S. radians*, from the Siderastreidae family. The hydrocoral *M. alcicornis* was not detected in mesophotic surveys. Deep reef habitat, defined by the presence of low/no light tolerant corals, was dominated by the Cnidarian phylum, of which both cover and density were dominated by Antipathidae, primarily due to the abundance of a black coral sea fan, potentially *A. atlantica/gracilis.* These results were similar to unpublished historical surveys exploring of the ring of outcrops surrounding Stetson Bank and characteristic of mid-shelf banks in the region (Sammarco et al. 2016). This data serves as a baseline to identify benthic community differences between habitats and from which to continue monitoring for temporal changes.

Overall Conclusions

Stetson Bank harbors a high latitude coral community with mesophotic coral community, existing along the northern limit for conditions for coral development and growth. In 1985, the reef was characterized as a "*Millepora*-Sponge" community (Rezak et al. 1985), essentially the community documented at the onset of the monitoring program in 1993. Over the past 23 years, the bank crest community became algae dominated, with the gradual pervasive decline in sponges and steep step-wise declines in hydrocoral abundance. Interestingly, despite existing in a marginal environment, which may have contributed to the decline in hydrocoral and sponge cover while macroalgal cover increased, scleractinian coral cover, while low, has remained stable throughout the study period in repetitive photostations. The past decade has documented phase-shifts of once coral-dominated to algae-dominated reefs throughout the Caribbean, leading researchers to debate whether these communities represent an alternative stable state or reversible phase changes (Côté et al. 2005, Rogers & Miller 2006, Mumby et al. 2007, Somerfield et al. 2008, Mumby 2009, Norström et al. 2009). Although the community at Stetson Bank has never been considered to be scleractinian coral dominated, prior to 1999 it was dominated by sponges and questions about the stability of these phase-shifted communities are still relevant. Although macroalgae is still the predominant biotic benthic component at Stetson Bank, 2015 saw the first time since 1999 that macroalgae did not compose the majority of benthic cover. While only continued monitoring will reveal what this means for the benthic community at Stetson Bank, the prevalence of bare colonizable substrate in 2015 may be a cautiously optimistic sign of a reversing phase change.

Chapter 3
FISH COMMUNITY

Large schools of juvenile vermilion snapper, *Rhomboplites aurorubens*, and brown chromis, *Chromis multilineata*, school at Stetson Bank. Photo: G.P. Schmahl/NOAA

Introduction

The fish community composition at Stetson Bank is similar to other Caribbean reefs, although overall diversity is lower and abundance is higher (Pattengill 1998). Absent or low-diversity families include hamlets (*Hypoplectrus* sp.), grunts (Haemulidae), and snapper (Lutjanidae) (Pattengill et al. 1997).

Fish populations within coral reef environments are known to be critical to ecosystem function (Kennedy et al. 2013, Holmlund & Hammer 1999). Since the late 1990s, reef fish density throughout the Caribbean has been in decline (Paddack et al. 2009), potentially due to habitat complexity loss (Alvarez-Filip et al. 2015) and over-exploitation (Jackson 1997, Pandolfi et al. 2003, Jackson et al. 2014). Within the designated boundaries of FGBNMS, which encompasses the bank crest at Stetson Bank, only traditional hook and line fishing activities are permitted.

Reef fish populations on the reefs of East and West FGBs, approximately 30 miles southeast of Stetson Bank, have been monitored since the late 1980s, documenting a relative stasis in fish abundance; however, consistent temporal variation in local reef fish populations have been observed (Zimmer et al. 2010, Johnston et al. 2013, Johnston et al. 2015b). On the bank crest of Stetson, a variety of fish surveys have been conducted since the initiation of the monitoring program. These surveys have been limited to non-decompression scuba operations, restricting surveys depths to < 33.5 m (110 ft). Starting in 1994, and continued on a non-scheduled basis, REEF [Reef Environmental Education Foundation] citizen science program surveys, using the roving diver technique (RDT; REEF 2016a), were conducted on both monitoring and recreational cruises using a combination of sanctuary researchers, REEF personnel, and volunteer divers. In 2010, belt and stationary point count methods were employed to quantify community metrics (abundance, biomass, and size frequency). In 2010 only, as part of NOAA's Natural Resource Damage Assessment following the *Deepwater Horizon* oil spill, belt transects were conducted following NOAA's National Centers for Coastal Ocean Science (NCCOS) Center for Coastal Monitoring and Assessment (CCMA) Biogeography Branch methods (Caldow et al. 2009). These data were omitted from this report due to the limited sample size. From 2012 to 2015, modified Bohnsack-Bannerot stationary visual fish censuses (Bohnsack et al. 1986) were conducted annually. In 2012, surveys started from permanent mooring buoys and from 2013 onward they were conducted from both permanent mooring buoys and within a stratified random design in high and low relief habitat. In 2015 BSEE and FGBNMS entered into an interagency agreement (E14PG00052) to fund the continuation and expansion of the monitoring effort. In addition to continuing reef fish censuses on the bank crest, the agreement expanded fish community monitoring to include mesophotic reef characterization and monitoring (Appendix A: Table A.1).

To date, 22 years of RDT sampling and four years of modified Bohnsack-Bannerot sampling have been conducted on the bank crest of Stetson Bank, with the latter distinguishing between high and low relief habitat for three years. In mesophotic habitat surrounding the bank crest, six exploratory ROV cruises were conducted over a nine-year timeframe (2001-2009) and one year of ROV belt transects were conducted to capture and characterize the mesophotic fish communities.

Methods

Bank Crest Surveys

From 2012 through 2015, fishes were visually assessed by trained scuba divers using a modified Bohnsack-Bannerot stationary visual fish census technique (Bohnsack et al. 1986) (Table 3.1; Appendix C: Table C.13). Divers were either highly experienced or extensively trained prior to completing surveys. Observations of fishes were restricted to an imaginary cylinder with a radius of 7.5 m, extending to the surface. All fish species observed within the first five minutes of the survey were recorded as the diver slowly rotated in place. Immediately following this five-minute observation period, one rotation was conducted for each species noted in the original five-minute period to record abundance (number of individuals per species) and fork length (within size bins). Size was binned into eight groups; <5 cm, 5 cm-10 cm, 10 cm-15 cm, 15 cm-20 cm, 20 cm-25 cm, 25 cm-30 cm, 30 cm-35 cm, and >35 cm, where each individual's size was recorded. Each survey required 15 minutes to complete. Transitory or schooling species were counted and measured at the time the individuals moved through the cylinder during the initial five-minute period. Surveys began in the early morning (after sunrise), and were repeated throughout the day until dusk. Each survey represented one sample. These surveys were conducted in the vicinity of permanent mooring buoys and at stratified random sites on the bank crest shallower than 33.5 m (110 ft) (Figure 3.1). Surveys near mooring buoys were conducted annually from 2012 to 2015, with a minimum of four surveys per buoy and 15 surveys per year conducted. Mooring buoys #1, #2, and #3, selectively located in flat habitat, near high relief habitat, served as starting locations for these surveys, from which a random heading, of 0°-360° and a random number of kick cycles from 0-40 kicks, was used to arrive at the survey start location. It was estimated that 40 kick cycles moved the diver approximately 50 m, with no current. A third number was generated to provide a random heading, from 0° to 360°, along which the tape was laid to mark the 7.5 m radius of the survey. Subsequent survey starting points were determined with additional sets of randomly generated numbers with the first number providing a heading, from 0° to 360°, and the second providing the number of fin kicks, from 12 to 40, to ensure the starting point was at least 15 m away from the previous location. A third number was generated to provide a random heading, from 0° to 360°, along which a tape was laid to mark the 7.5 m radius of the survey.

Stratified random fish surveys were conducted in conjunction with stratified random benthic transects from 2013 to 2015, where the survey start location was selected using a stratified random sampling design (see Chapter 2, Random Transect Methods). Surveys were stratified by habitat type (high or low relief) and randomly distributed over the bank. Modified Bohnsack-Bannerot surveys from 2012 to 2013 conducted from mooring buoys were treated as high relief habitat following testing for similarities, while 2014-2015 surveys conducted from mooring buoys included a habitat component that was used to classify high or low relief. The habitat component documented maximum relief and binned the percent composition of the sample area into five categories: <0.2 m, 0.2-0.5 m, 0.5-1.0 m, 1.0 m-1.5 m, and >1.5 m. Samples were entered directly into a Microsoft® Excel database by each surveyor. All data were checked for quality and accuracy.

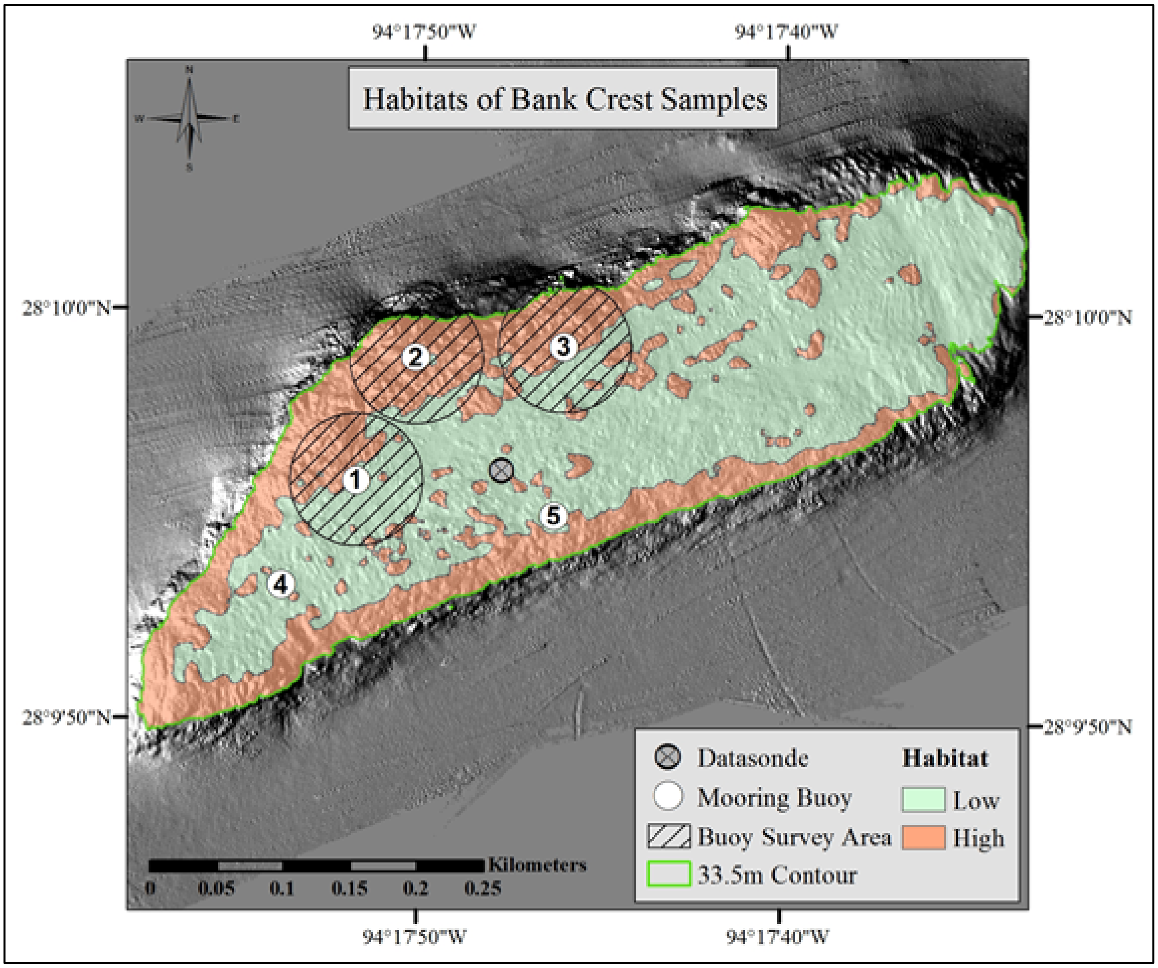


Figure 3.1. Habitats of bank crest fish surveys. Green highlights low relief habitat, orange highlights high relief habitats, and black hash overlay shows areas potentially captured in "Buoy" surveys. Image: NOAA

Table 3.1. Number of modified Bohnsack-Bannerot surveys conducted, 2012-2015.

Survey	Year			
	2012	2013	2014	2015
Buoy - High Relief	17	15	19	18
Buoy-Low Relief	-	-	0	5
Random - High Relief	-	9	5	8
Random - Low Relief	-	18	11	13

From 1994 through 2015, the REEF roving diver technique was employed to conduct a visual assessment of fish species and relative abundance scores on the bank crest, shallower than 33.5 m (110 ft) (Table 3.2) (REEF 2016a). Two levels of surveyor experience rankings (novice and experienced), as established by REEF training requirements, were used in this analysis (Pattengill-Semmens & Semmens 1998). Observations of fish were not restricted to a survey area, and divers swam freely through the site recording every observed species that they could identify with the goal to find as many species as possible. The approximate abundance of each recorded species was captured into four categories: single (1), few (2-10), many (11-100), and abundant (>100). Samples were conducted with no sampling design from permanent mooring buoys (#1-#5), selectively located in flat habitat near high relief features. REEF samples were entered into the REEF database via online form or mailed scantron by each surveyor. All available data for Stetson Bank were downloaded from the REEF database on February 10, 2016, as a text file (REEF 2016b). Data were not identified or averaged between diver teams.

Table 3.2. Number of REEF samples conducted, 1994-2015.

Year	Novice Samples	Experienced Samples	All Surveys
1994	0	8	8
1995	42	23	65
1996	58	16	74
1997	50	20	70
1998	28	8	36
1999	56	12	68
2000	53	23	76
2001	104	14	118
2002	62	6	68
2003	34	38	72
2004	61	44	105
2005	43	25	68
2006	16	33	49
2007	11	25	36
2008	0	0	0
2009	2	2	4
2010	0	0	0
2011	18	0	18
2012	18	0	18
2013	17	0	17
2014	19	0	19
2015	19	2	21

Mesophotic Transects

Exploratory ROV surveys from 2001 to 2009 had no sampling design. Standard definition video was reviewed and clipped into non-overlapping 10 minute transects with >75% hard bottom, post-hoc. In 2015, transect start location was selected using a stratified random sampling design, sampling two habitats: coralline algae reef and deep reef (Figure 2.5) and conducted for 10 minutes along hard bottom habitat with the ROV maintaining a speed of ~0.5 kph. High definition video was collected on each transect. Where visibility restricted the field of view to <5 m^2 or where >25% soft bottom habitat was encountered, transects were removed from analyses. Observations of fishes were restricted to the field of view of the ROV's video camera. All fish species observed were recorded, counted, and sized using mounted scale lasers in the field of view of the ROV. Fork length was binned into eight groups; 5 cm, 5 cm-10 cm, 10 cm-15 cm, 15 cm-20 cm, 20 cm-25 cm, 25 cm-30 cm, 30 cm-35 cm, and 35 cm, where the fork length of each individual was recorded. Transect data were entered directly into a Microsoft® Excel database by each surveyor. The area of each transect was calculated by importing ROV track data, recorded every two seconds, into ArcMap®. Line data were smoothed using the PAEK algorithm and a smoothing tolerance of 10 m. Length was then calculated in WGS83 UTM15. Distance was multiplied by the maximum horizontal distance in the field of view, scaling the image based on the lasers at the furthest point in the field of view. Measurements were calculated using ImageJ. All data were checked for quality and accuracy.

From 2003 to 2015, a total of 37 fish transects were processed (Table 3.3, Appendix C: Table C.14). Surveys prior to 2015 were only conducted in deep reef habitat and not conducted annually, resulting in non-continuous samples. Thirty of these samples had tracking data and scale lasers and were processed for area and density.

Table 3.3. ROV cruise details, 2003-2015 (NURC=National Undersea Research Center; UNCW=University of North Carolina at Wilmington).

Cruise Name	Vessel	Date (Month-Year)	ROV	Number of Transects
DFH1	M/V *Spree*	Feb-01	NURC Phantom S2	2
DFH4	M/V *Spree*	Feb-03	NURC Phantom S2	4
DFH8	NOAA Ship *Nancy Foster*	Sep-04	NURC Phantom S2	3
DFH9	USA/NASA Ship *Liberty Star*	Oct/Nov. 04	NURC Phantom S2	2
DFH11	M/V *Spree*	Sep-05	NURC Phantom S2	1
DFH13	R/V *Manta*	May-09	NURC Phantom S2	8
DFH27	R/V *Manta*	Jul-15	UNCW MOHAWK	17

Data Processing

Within all the databases, family and trophic guild were recorded for each entry. Species were classified into four primary trophic guilds: herbivores (H), piscivores (P), invertivores (I), and planktivores (PL) using available data (Froese & Pauly 2016). Biomass estimates were computed from modified Bonhsack-Bannerot surveys using the allometric length-weight conversion formula (Bohnsack et al. 1986, Froese & Pauly 2016), and expressed as grams per 100 m².
Observations of manta rays, stingrays, and sharks were removed from all biomass analyses due to their rare nature and large size.

Abundance biomass curves (ABC; Warwick 1986) were generated for each sample and *w* statistics and diversity measures (Shannon diversity [log base e], Pielou's evenness, and Margalef species richness) were calculated for each sample. Curves and calculations were conducted in PRIMER version 7 (Clarke & Gorley 2015).

In REEF data, sighting frequency was calculated for each species by year.

Statistical Analysis

Due to the limited sample size and number of replicates of mesophotic fish surveys, only bank crest surveys were analyzed. However, summaries of mesophotic data are presented. Modified Bohnsack-Bannerot and mesophotic ROV transects data from 2015 were averaged between habitat types and presented as a summary of the current reef fish community at Stetson Bank.

Bank crest modified Bohnsack-Bannerot data coefficient of variation percentages were calculated (Appendix C: Table C.1) and used in dispersion weighting transformations. Data were tested for differences in community by habitat and year using non-parametric distance-based analyses using species presence/absence, species density, species biomass, trophic richness, trophic density, trophic biomass, relative trophic abundance, and size frequency. Species density and biomass data were dispersion weighted and square root transformed to reduce the impact of large schooling species on the analysis. Trophic density, trophic biomass, and size frequency data were square root transformed. Relative trophic abundance was calculated by standardizing trophic density data and used with no additional transformations. PERMANOVA (Anderson et al. 2008) was based on Bray-Curtis similarity matrices. PERMANOVA represents a better alternative to ANOVA or MANOVA for ecological data as it does not assume normal distribution of the data. Habitat (2 levels: low relief and high relief) and year (4 levels: 2012-2015) were used as fixed orthogonal factors (sum of squares=Type 1, number of permutations=9999, permutation method= reduced model [crossed]). Where significant differences between years were found with PERMANOVA, variables contributing to observed differences between years and habitat were examined using SIMPER (Clarke 1993, Clarke et al. 2014). SIMPER analysis in species level data were based on dispersion weighted and square root transformed Bray-Curtis similarity matrices and analysis on trophic level and size frequency data were based on square root transformed Bray-Curtis similarity matrices. SIMPER assesses the contribution of variables to the dissimilarity between groups. Further evaluation of the species contributing to observed differences were conducted through Type III SIMPROF (Clarke et al. 2008, Clarke et al. 2014) and presented as shade plots.

Diversity measures (Shannon diversity [log base e], Pielou's evenness, and Margalef species richness) were calculated for each sample. These measures were analyzed together using a Euclidean distance matrix, based on untransformed data, and tested for significant differences between year and habitat using PERMANOVA as described above.

Abundance-biomass curve *w* values were calculated for each sample. As data were normally distributed, year and habitat were tested using ANOVA, with pairwise examination using Student's t-tests.

No temporal trend testing was conducted on modified Bohnsack-Bannerot survey data due to the limited timeframe (4 years). However, means plots with standard deviation error bars were generated by year for total density, biomass, and *w* values.

From REEF data, sighting frequency and trophic group species richness were analyzed. Datasets were tested for significant year groupings using CLUSTER analysis, based on Bray-Curtis similarity matrices, and tested for significant clusters using (Clarke et al. 2008, Clarke et al. 2014). Non-metric multidimensional scaling (nMDS) was used to visualize the data as it was not well represented in the low dimensional space of PCO (Anderson et al. 2008). Coherent species curves (Somerfield & Clarke 2013) were used to analyze patterns in trophic guild richness ordered naturally as a time series, where data were averaged by year to reduce noise. The relationship between average species richness and the proportion of expert samples to total samples was examined using Kendall rank correlation as the data were not normally distributed.

For families of interest, species data were extracted and analyzed separately. For modified Bohnsack-Bannerot samples, data were transformed as described above and density and biomass were analyzed for differences between year and habitat using a two-way PERMANOVA based on Bray-Curtis similarity matrices with a dummy variable. Habitat (2 levels: low relief and high relief) and year (4 levels: 2012-2015) were used as fixed orthogonal factors (sum of squares=Type 1, number of permutations=9999, permutation method= reduced model [crossed]). Where significant differences between years were found with PERMANOVA, variables contributing to observed differences between years were examined using SIMPER. Relative abundance of individuals in each size bin was graphed for each species, where relative abundance was calculated by dividing the number of individuals in a size bin by the total abundance of that species. In REEF data, species that do not occur on current FGBNMS species lists (flowergarden.noaa.gov/about/fishlist.html), and were only observed by novice surveyors, were removed from analysis. Sighting frequency data were clustered using hierarchical agglomerative clustering, based on Bray-Curtis similarity matrices, and the significance of the divisions were tested using SIMPROF. Results from this analysis were graphed using nMDS, including SIMPROF clusters, year trajectory, and vector overlay of variables with > 0.8 correlation. Due to the relationship of parrotfish and angelfish with benthic biota (coral cover and sponge cover, respectively), coherent species curves were used to examine patterns of sighting frequency and benthic cover, using average annual benthic cover from repetitive photostation data (see Chapter 2). Monotonic trends in sighting frequency of these families of interest were tested for using the non-parametric Mann-Kendall trend test. Due to the lack of data for lionfish, data from Johnston et al. (2016b) are presented.

PERMANOVA, SIMPER, CLUSTER, SIMPROF, and nMDS were performed in PRIMER version 7 (Clarke & Gorley 2015) with the PERMANOVA+ add-in (Anderson et al. 2008) while Mann-Kendall trend tests and Kendall rank correlation were performed in R version 3.2.0 (R Development Core Team 2015).

Spatial interpolation of richness, biomass, and density data from bank crest and mesophotic stratified random transects were mapped using inverse distance weighting (IDW). Interpolations were created without separating data by habitat, using a variable search radius and 12 points. Analyses were performed in ESRI's ArcMap version 10.4.

Results

Bank Crest Surveys

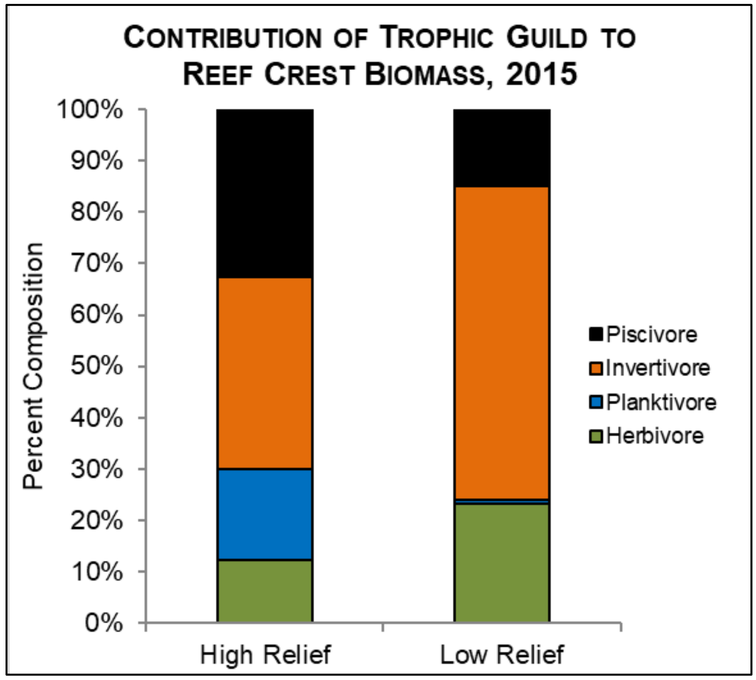


Figure 3.2. Bank crest trophic biomass contribution.

In 2015, the most frequently sighted species was sharpnose puffer (*Canthigaster rostrata*), the average density of bluehead (*Thalassoma bifasciatum*) was the greatest of any species, and the average biomass of great barracuda (*Sphyraena barracuda*) was the greatest of any species. In low relief habitat, the most frequently sighted species included the sharpnose puffer, bluehead, doctorfish (*Acanthurus chirurgus*), and cocoa damselfish (*Stegastes variabilis*). Average density of bluehead was the greatest of any species, and average biomass of French angelfish (*Pomacanthus paru*) was the greatest of any species (Appendix C: Table C.2). Trophic biomass in both habitats was predominately invertivores, while piscivores contributed a greater proportion to biomass in high relief habitat and herbivores contributed a greater proportion to biomass in low relief habitat (Figure 3.2).

From 2012 to 2015, a total of 133 modified Bohnsack-Bannerot surveys were conducted on the bank crest, with 69 originating from permanent mooring buoys (#1, #2, and #3), and 64 conducted at stratified random locations (22 in high relief habitat and 42 in low relief habitat). Total species richness from all surveys was 113 and total family richness from all surveys was 35. Average species richness per survey was 18, and average family richness per survey was 11. Significant differences were found in species presence/absence, species density, species biomass, trophic richness, trophic density, trophic biomass, size frequency between year and habitat, and relative trophic abundance between years. Significant interactions were found in all data except species trophic richness, trophic biomass, and relative trophic abundance (Table 3.4).

Table 3.4. PERMANOVA main test results. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Presence/Absence	Year	6.47	**<0.001**	9864
	Habitat	6.32	**<0.001**	9921
	Year*Habitat	2.08	**<0.001**	9889
Species Density	Year	5.94	**<0.001**	9853
	Habitat	5.64	**<0.001**	9893
	Year*Habitat	2.15	**<0.001**	9866
Species Biomass	Year	4.34	**<0.001**	9812
	Habitat	3.78	**<0.001**	9892
	Year*Habitat	2.01	**<0.001**	9852
Trophic Richness	Year	7.19	**<0.001**	9933
	Habitat	6.91	**<0.001**	9959
	Year*Habitat	1.49	0.184	9951
Trophic Density	Year	6.90	**<0.001**	9928
	Habitat	8.94	**<0.001**	9949
	Year*Habitat	2.34	**0.026**	9943
Trophic Biomass	Year	3.26	**<0.001**	9931
	Habitat	4.69	**<0.001**	9958
	Year*Habitat	1.44	0.170	9922
Relative Trophic Abundance	Year	5.87	**<0.001**	9944
	Habitat	1.46	**0.235**	9968
	Year*Habitat	1.18	**0.323**	9943
Size Frequency	Year	5.84	**<0.001**	9912
	Habitat	6.48	**<0.001**	9946
	Year*Habitat	2.10	**0.014**	9923

High relief habitat possessed significantly different species composition than low relief habitat, with multiple species contributing to the dissimilarity (Appendix C: Table C.3), including brown chromis (*Chromis multilineata*) and sailfin blenny (*Emblemaria pandionis*). The fish community, based on both density and biomass, was significantly different between habitats, with greater overall density, but lower biomass, in high relief habitat than in low relief habitat (density: 179.95 and 107.24 individuals/100 m^2, respectively; biomass: 10959.71 and 12162.06 g/100 m^2, respectively). Multiple species contributed to the observed dissimilarity between habitat based on species density (Appendix C: Table C.4) and species biomass (Appendix C: Table C.5). Reef butterflyfish (*Chaetodon sedentarius*) density and French angelfish biomass contributed the most to the observed dissimilarity between high and low relief habitats.

The richness of herbivores and invertivores was greater in high relief habitat and planktivore and piscivore richness was greater in low relief habitat. The differences in invertivore species richness contributed the greatest to the dissimilarity between habitats (Table 3.5). Significant differences in trophic composition between habitats were seen in both density and biomass, with a greater density of invertivores observed in high relief habitat than low relief and a greater biomass of piscivores observed in low relief habitat. The invertivore and piscivore guilds each represented the trophic guild with the greatest dissimilarity in density and biomass, respectively, between habitats (Table 3.6).

Table 3.5. Richness of trophic guilds by habitat with SIMPER results.

Trophic Group	High Relief Habitat Richness	Low Relief Habitat Richness	% Contribution to Richness Dissimilarity
Invertivore	10.51	9.23	43.21
Herbivore	5.02	4.60	27.56
Piscivore	1.65	1.28	16.22
Planktivore	1.57	1.62	13.02

Table 3.6. Density and biomass of trophic guilds by habitat with SIMPER results.

Trophic Group	High Relief Habitat		Low Relief Habitat		% Contribution to Density Dissimilarity	% Contribution to Biomass Dissimilarity
	Density (#/100m^2)	Biomass (g/100m^2)	Density (#/100m^2)	Biomass (g/100m^2)		
Herbivore	38.97	1295.01	30.83	3597.91	23.27	23.25
Invertivore	118.38	3918.69	63.72	1660.53	39.09	25.93
Planktivore	18.68	2282.47	9.68	1607.82	21.76	15.56
Piscivore	4.41	4037.52	7.92	4117.62	15.89	35.26

Between high and low relief habitat, the abundance of fish <5 cm and 5-10 cm in length contributed over 40% of the observed dissimilarity, with high relief habitat having greater average abundance of <5 cm and 5-10 cm length fish (Table 3.7). In low relief habitat there was a greater abundance of 30-35 cm and >35 cm length fish than in high relief habitat.

Table 3.7. Size frequency abundance by habitat with SIMPER results.

Size Class	High Relief Abundance	Low Relief Abundance	% Contribution to Dissimilarity
<5	208.84	127.57	26.58
5-10	51.59	32.57	18.30
10-15	13.67	4.79	8.17
15-20	12.83	6.68	10.42
20-25	13.15	2.55	9.23
25-30	6.00	4.38	8.88
30-35	9.24	15.55	11.01
>35cm	3.48	4.17	7.41

Pairwise comparisons between years indicated that each subsequent year was significantly different in species presence/absence, species density, species biomass, trophic richness, and size frequency data. Pairwise comparison of consecutive years in trophic density, trophic biomass, and relative trophic abundance data identify significant differences only between 2012-2013 and 2014-2015 (Table 3.8).

Table 3.8. PERMANOVA pairwise tests for year results. Bold denotes significant values. Main test results are in Table 2.4.

Data	Pairwise Comparison	t	P(perm)	Unique Permutations
Presence/Absence	2012-2013	2.28	**<0.001**	9929
	2013, 2014	2.00	**<0.001**	9929
	2014, 2015	2.72	**<0.001**	9938
Species Density	2012-2013	2.09	**<0.001**	9910
	2013, 2014	2.05	**<0.001**	9916
	2014, 2015	2.76	**<0.001**	9909
Species Biomass	2012-2013	2.01	**<0.001**	9911
	2013, 2014	1.82	**<0.001**	9892
	2014, 2015	2.26	**<0.001**	9881
Trophic Richness	2012-2013	2.19	**0.004**	9950
	2013, 2014	1.70	**0.031**	9949
	2014, 2015	3.81	**<0.001**	9951
Trophic Density	2012-2013	2.12	**0.003**	9971
	2013, 2014	1.06	0.327	9955
	2014, 2015	3.40	**<0.001**	9947
Trophic Biomass	2012-2013	2.09	**0.003**	9936
	2013, 2014	0.98	0.425	9956
	2014, 2015	2.28	**0.003**	9956
Relative Trophic Abundance	2012-2013	2.45	**0.001**	9958
	2013, 2014	0.59	0.761	9955
	2014, 2015	3.57	**<0.001**	9966
Size Frequency	2012-2013	2.17	**<0.001**	9956
	2013, 2014	1.79	**0.006**	9950
	2014, 2015	2.63	**<0.001**	9939

Species composition was significantly different between years, with different species contributing to the differences every year (Appendix C: Table C.6). Multiple species contributed to the observed dissimilarity between years in density (Appendix C: Table C.7) and biomass (Appendix C: Table C.8).

Differences in invertivore species richness contributed almost 40% of the dissimilarity between years (Table 3.9). While mean richness of herbivores were variable over time, invertivore and planktivore guilds increased in 2015 (Figure 3.3). Invertivore density and piscivore biomass contributed to over 30% of the dissimilarity between years (Table 3.10). Mean density of invertivores were variable over time, while herbivore density increased (Figure 3.4). Mean piscivore biomass varied over time and planktivore biomass decreased over time (Figure 3.5). While clear differences were present between years, trends were not tested for significance due to the limited temporal scale.

Table 3.9. Richness of trophic guilds by year with SIMPER results.

Trophic Guild	% Contrib.	Cum.%
2012-2013		
Invertivore	39.46	39.46
Herbivore	29.36	68.82
Piscivore	19.21	88.03
Planktivore	11.97	100.00
2013-2014		
Invertivore	40.00	40.00
Herbivore	28.49	68.49
Piscivore	19.35	87.84
Planktivore	12.16	100.00
2014-2015		
Invertivore	40.17	40.17
Herbivore	26.39	66.55
Piscivore	17.75	84.30
Planktivore	15.70	100.00

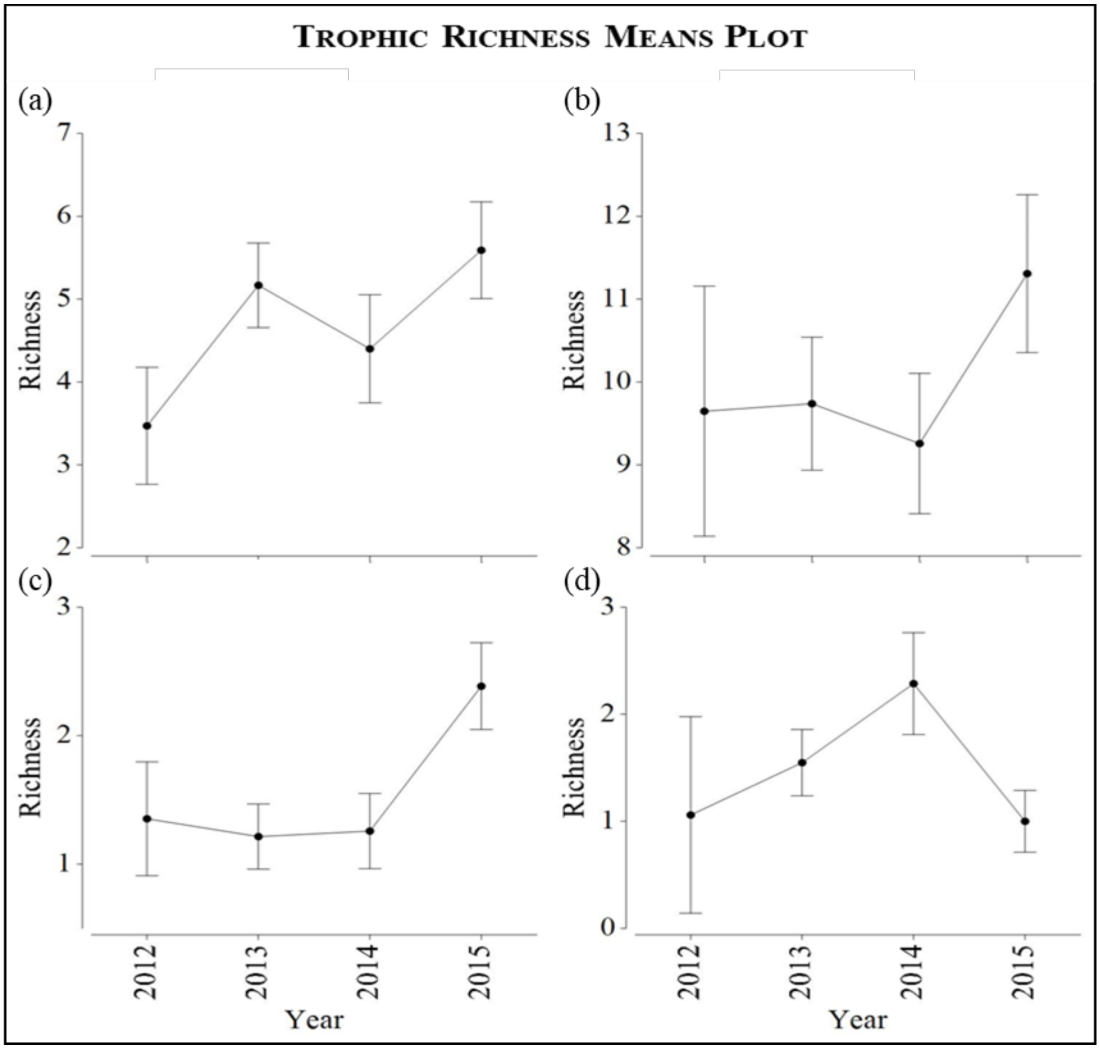


Figure 3.3. Trophic richness means plots over time. A: herbivores, B: invertivores, C: planktivores, and D: piscivores. Error bars represent standard deviation.

Table 3.10. Contribution of trophic guilds to dissimilarity in density and biomass between years. SIMPER results were based on square root transformed data. I = Invertivore, PL = Planktivore, H = Herbivore, and P = Piscivore. Density in #/100m^2 and biomass in g/100m^2.

Trophic Group	Density		Trophic Group	Biomass	
	% Contrib.	Cum.%		% Contrib.	Cum. %
2012-2013					
I	34.25	34.25	P	31.38	31.38
PL	30.77	65.02	I	25.56	56.94
H	20.7	85.72	PL	23.22	80.16
P	14.28	100	H	19.84	100

Table 3.10. Cont'd.

2013-2014					
I	31.98	31.98	P	37.61	37.61
PL	25.73	57.7	I	24.12	61.73
H	24.36	82.06	H	20.67	82.4
P	17.94	100	PL	17.6	100
2014-2015					
I	45.24	45.24	P	39.11	39.11
PL	20.91	66.15	I	26.99	66.1
H	20.64	86.79	H	18.43	84.53
P	13.21	100	PL	15.47	100

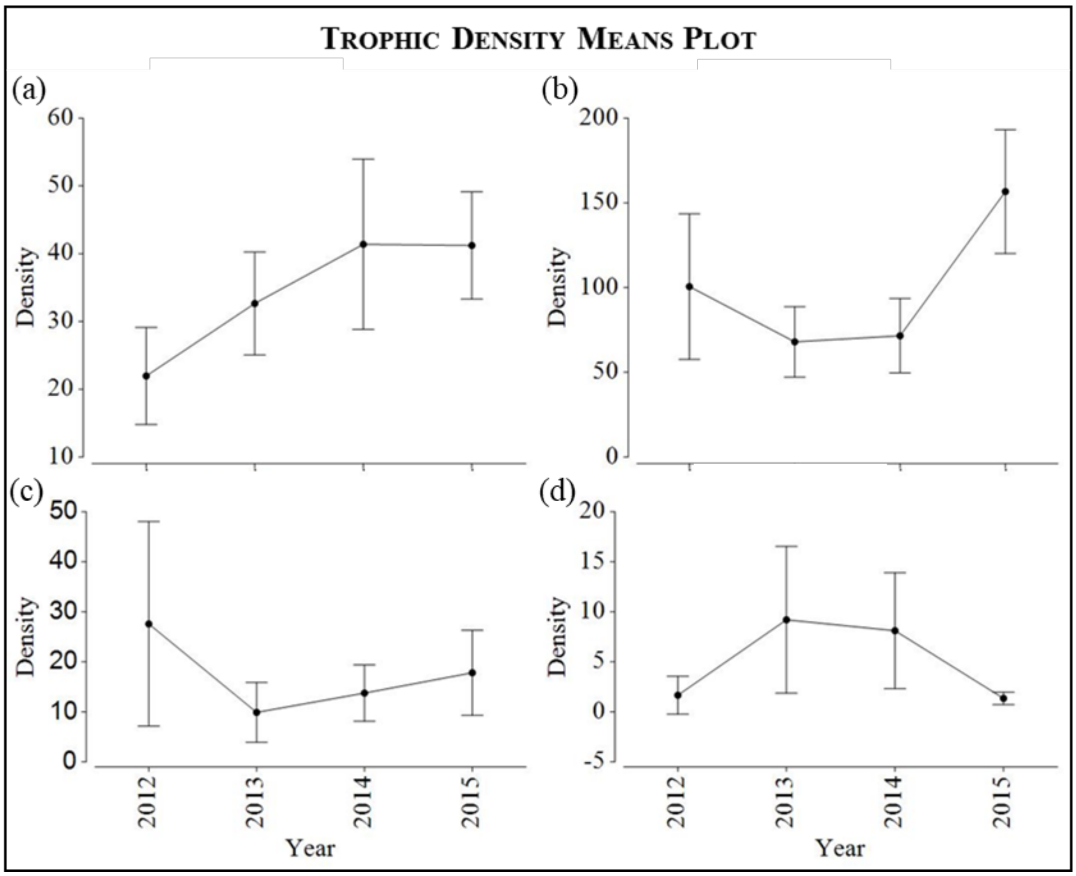


Figure 3.4 Means plots of relative trophic density (individuals/100m^2). A: herbivores, B: invertivores, C: planktivores, and D: piscivores. Error bars represent standard deviation.

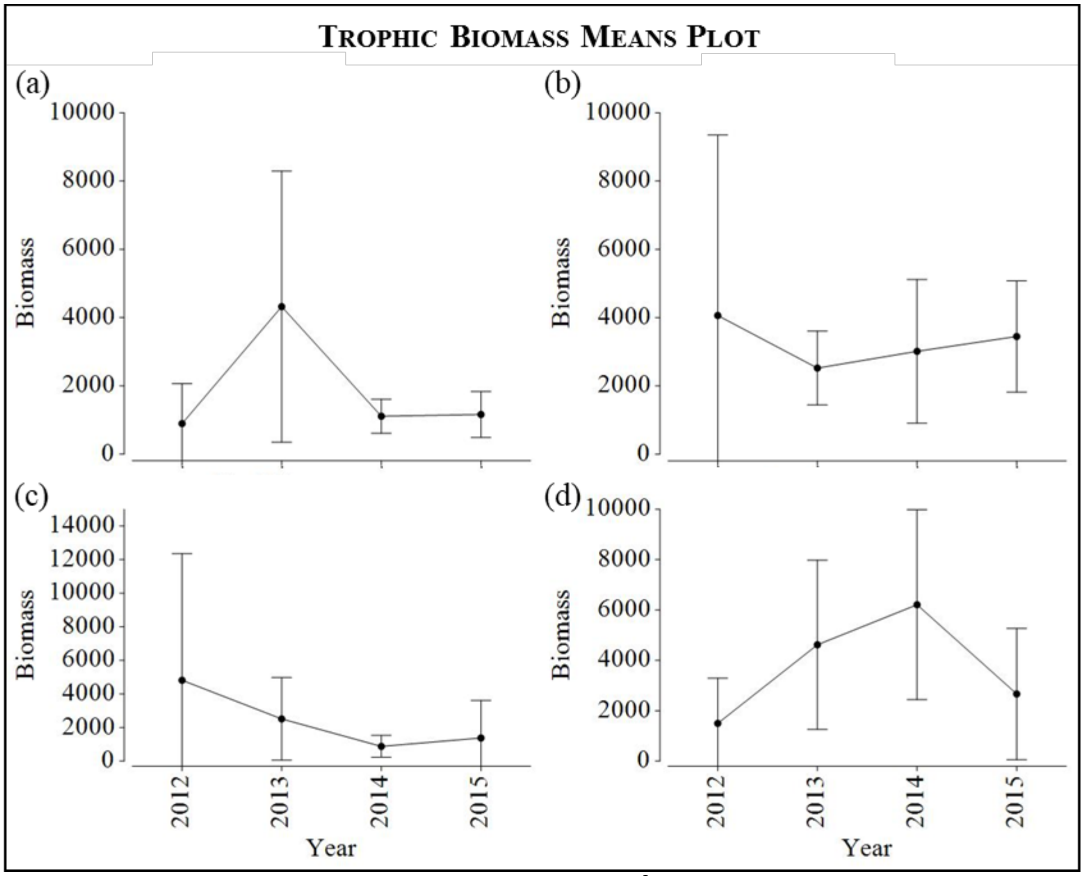


Figure 3.5. Means plots of relative trophic biomass (g/100m^2). A: herbivores, B: invertivores, C: planktivores, and D: piscivores. Error bars represent standard deviation.

The relative abundance of invertivores contributed the greatest to the dissimilarity between all years (Table 3.11). Similar to trophic density results, herbivore relative abundance appeared to increase from 2012 to 2014, but declined in 2015, while planktivore relative abundance decreased over time (Figure 3.6). The contribution of each trophic guild to biomass by year and habitat were graphed, showing an inverted biomass pyramid (greater biomass of piscivores than herbivores) in all years in high relief habitat (Figure 3.7).

Table 3.11. Contribution of trophic guilds to dissimilarity in relative abundance between years. SIMPER results were based on untransformed data.

Trophic Guild	% Contrib.	Cum.%
2012-2013		
Invertivore	34.57	34.57
Herbivore	33.37	67.94
Planktivore	25.3	93.24
Piscivore	6.76	100
2013-2014		
Invertivore	34.91	34.91
Herbivore	30.22	65.13
Planktivore	20.1	85.23
Piscivore	14.77	100
2014-2015		
Invertivore	40.62	40.62
Herbivore	28.68	69.3
Planktivore	20.11	89.41
Piscivore	10.59	100

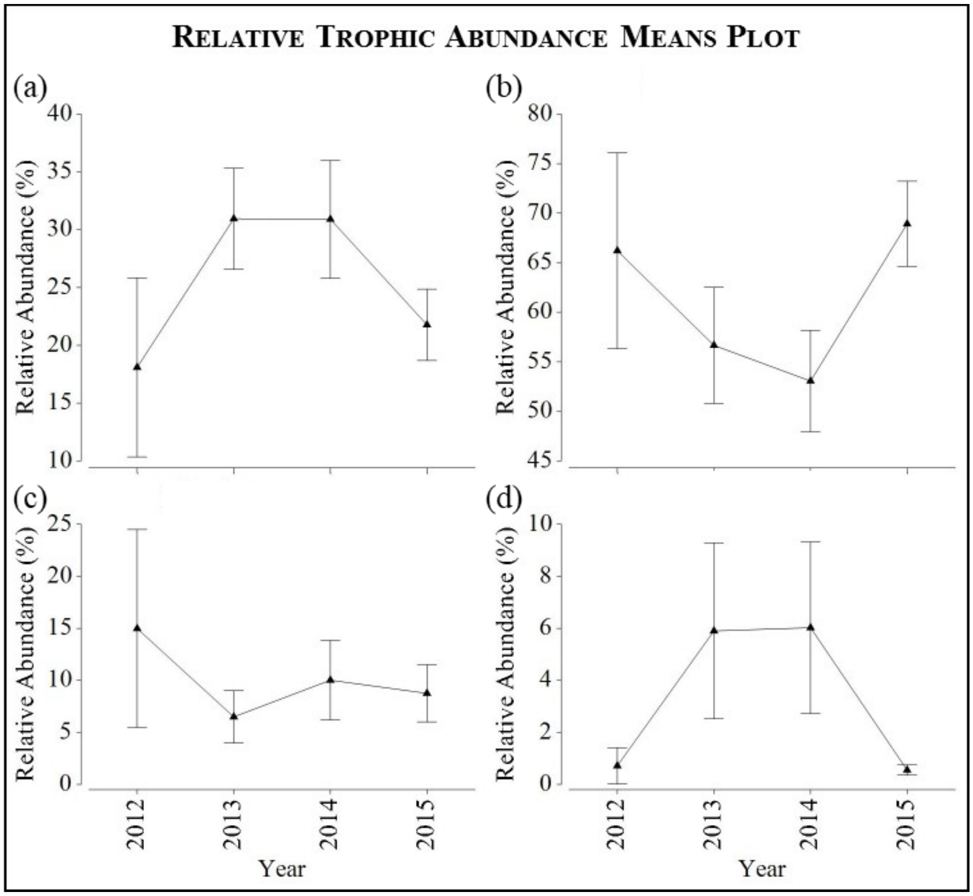


Figure 3.6. Means plots of relative trophic abundance. A: herbivores, B: invertivores, C: planktivores, and D: piscivores. Error bars represent standard deviation.

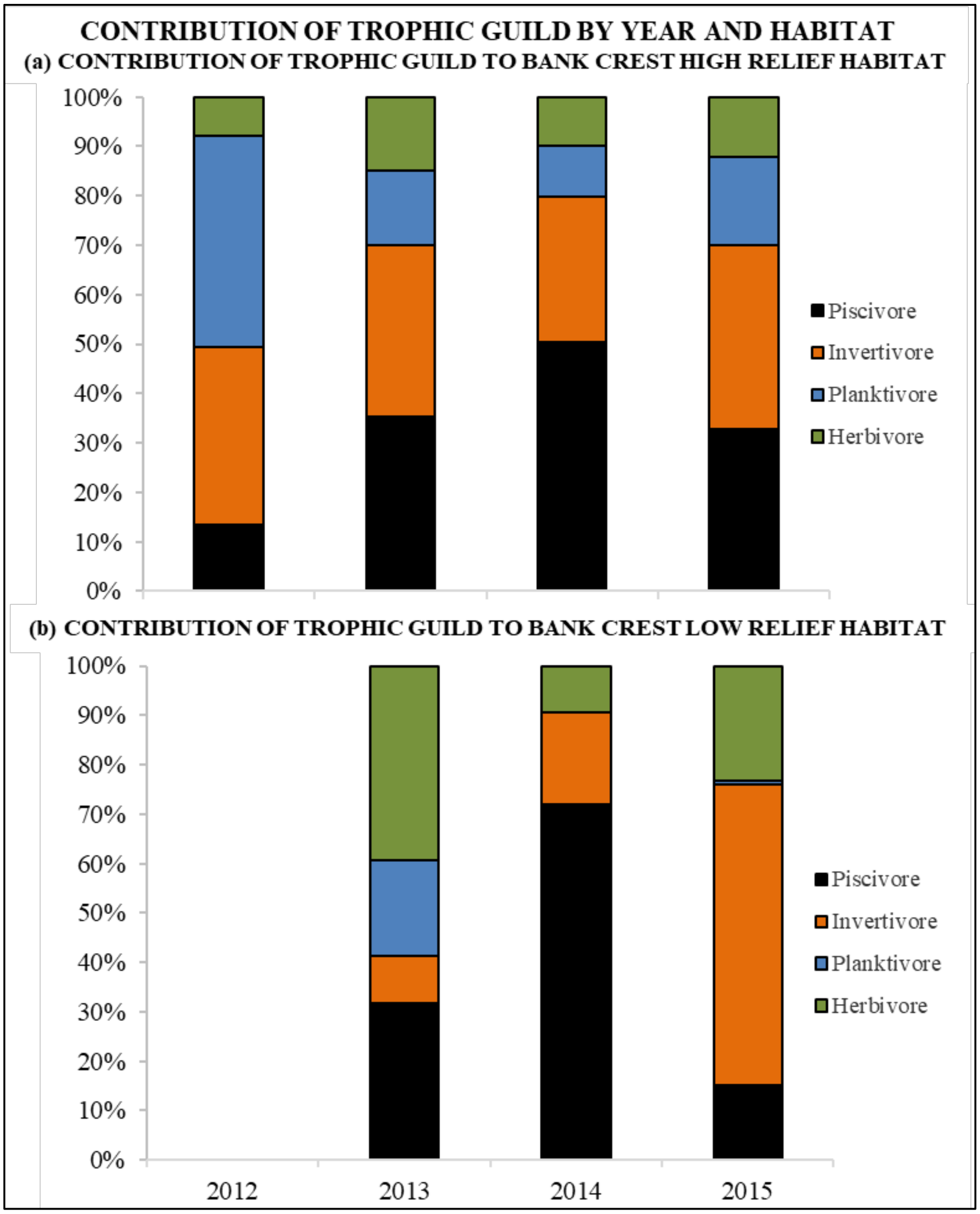


Figure 3.7. Contribution of trophic guild biomass by year and habitat. A: Trophic biomass for high relief habitat and B: trophic biomass for low relief habitat.

Size frequency differences between all consecutive years was primarily due to differences in abundance of fish <5 cm and 5-10 cm in length (Table 3.12). Abundance of <5 cm fish appeared to increase over time, while the abundance of 5-10 cm fish has been variable over time (Figure 3.8). While clear differences were present between years, trends were not tested for significance due to the limited temporal scale.

Table 3.12. Contribution of size class to dissimilarity in abundance between years. SIMPER results were based on square root transformed data.

Size Class	% Contrib.	Cum.%
2012-2013		
<5	21.99	21.99
5-10	17.1	39.09
20-25	14.38	53.47
10-15	13.3	66.77
15-20	11.71	78.48
30-35	8.53	87.01
25-30	6.6	93.62
>35cm	6.38	100
2013-2014		
<5	23.82	23.82
5-10	18.12	41.94
15-20	12.08	54.03
20-25	10.94	64.97
30-35	9.3	74.27
25-30	9.08	83.34
10-15	9.07	92.41
>35cm	7.59	100
2014-2015		
<5	34.12	34.12
5-10	15.21	49.33
15-20	10.5	59.83
10-15	9.07	68.9
30-35	8.84	77.74
25-30	8.5	86.24
20-25	7.99	94.23
>35cm	5.77	100

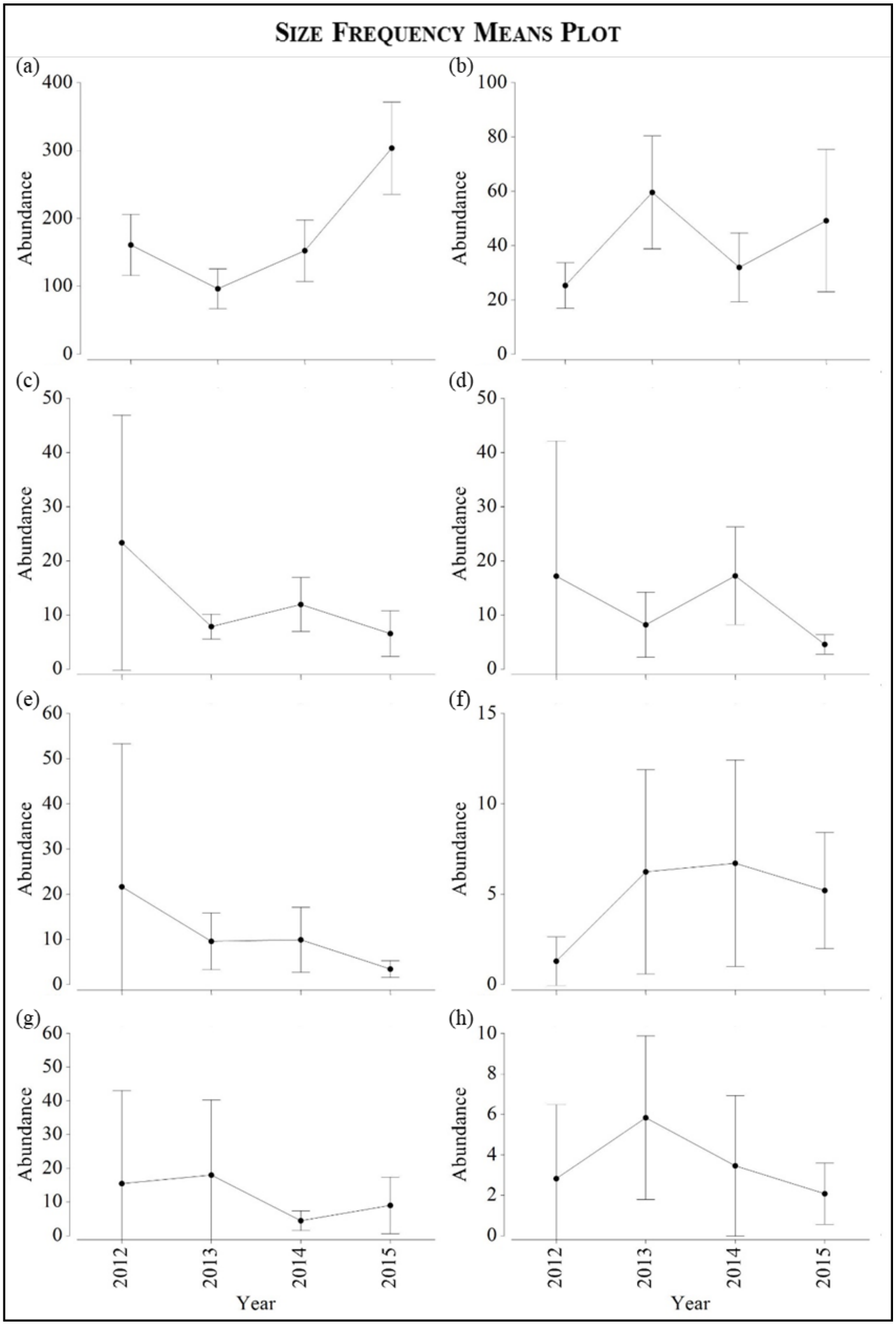


Figure 3.8. Abundance means plot for each size bin. A: <5 cm, B: 5 cm-<10 cm, C: 10 cm-<15 cm, D: 15 cm-<20 cm, E: 20 cm-<25 cm, F: 25 cm-<30 cm, G: 30 cm-<35 cm, H: ≥35 cm. Error bars represent standard deviation.

Since 2013, overall density increased while overall biomass decreased, and *w* values declined, potentially indicating increased recruitment (Figure 3.9). However, trends were not tested for significance due to the limited temporal scale. ABC analysis showed no significant differences between years or habitats, with no significant interaction. The average *w* value was slightly positive overall (0.15), good ecological condition, with a minimum value of -0.19 and maximum value of 0.62 (Appendix C: Table C.9).

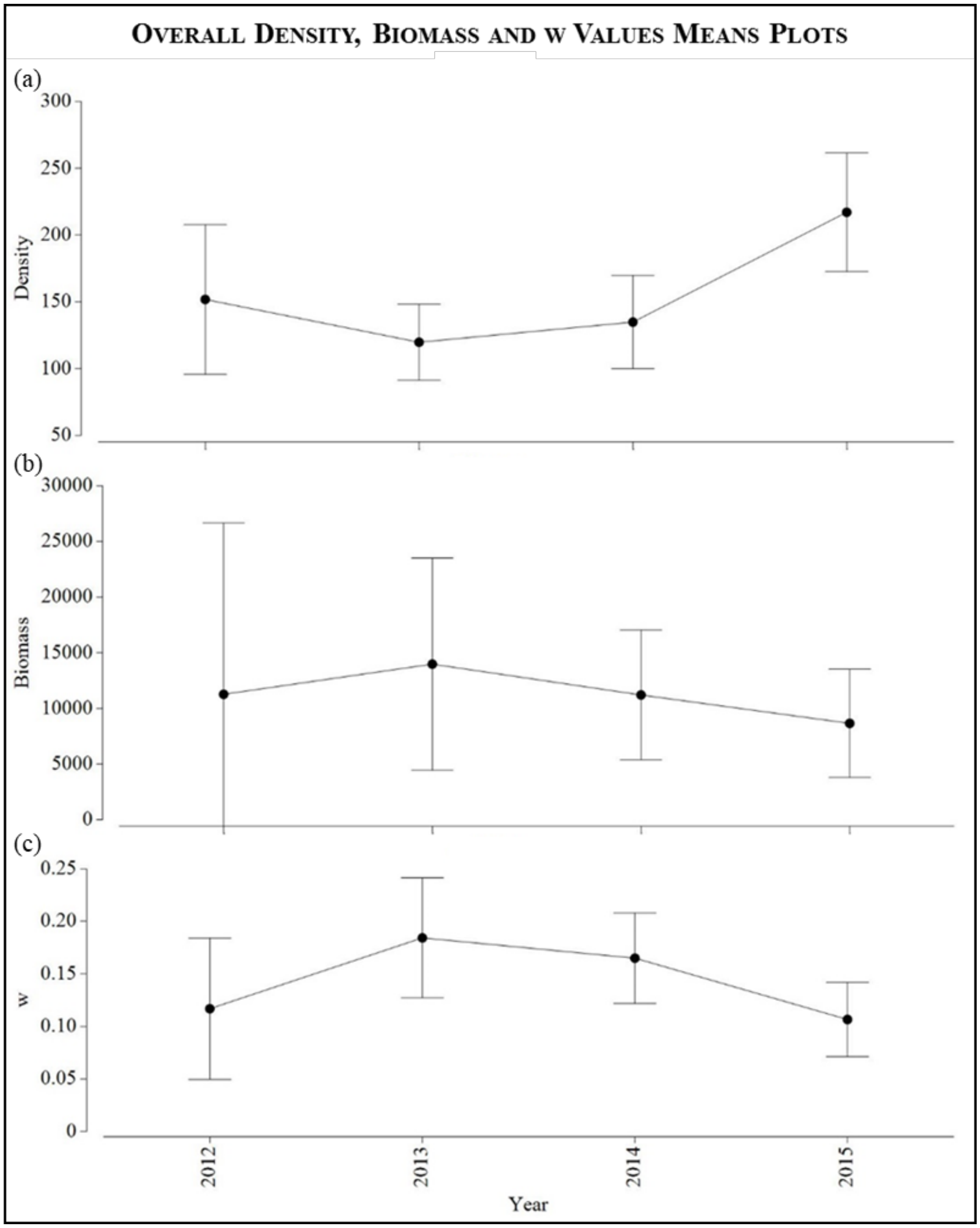


Figure 3.9. Means plots of (a) density (individuals/100m^2), (b) biomass (g/100m^2), and (c) w-values by year. Error bars represent standard deviation.

Of the three diversity measures calculated for each sample, significant differences were found between habitat and year (p=0.05, pseudo-F=3.63 and p<0.001, pseudo-F=5.93, respectively), with no significant interaction. Pairwise analysis of years found significant differences in diversity measures only between 2012 and 2013 (p<0.001, pseudo-F=3.94).

In REEF data, trophic group species richness was examined and three significant clusters (A: 2012; B: 1994-2000, 2003-2009; C: 2011-2002, 2011-2015) were found (Figure 3.10). Coherence plots indicated that all trophic guilds richness covaried over time, with richness declining in all trophic guilds 2011 onward (Figure 3.11). Further examination highlighted that the significant clusters and changes in trophic guild richness correlated with years of predominately novice surveys. A significant positive correlation was found between average species richness and proportion of samples conducted by experts, indicating high proportions of novice surveys reduced species richness (Table 3.13). Some similar trends were observed in species sighting frequency data, with 10 significant year clusters found in SIMPROF analysis (A: 1994 and 2009; B: 1995; C: 1996-1997; D: 1998-1999; E: 2000; F: 2001-2002; G: 2003-2006; H: 2007; I: 2011-2015; J: 2012). As observed in trophic richness data, total sighting frequency decreased as the proportion of expert surveys decreased.

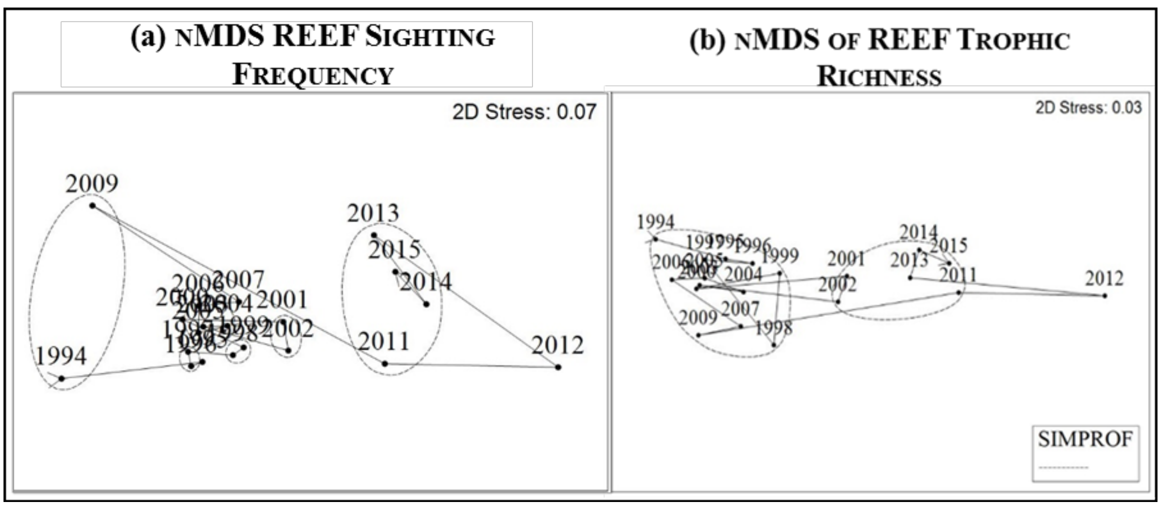


Figure 3.10. nMDS plots of REEF data. A: nMDS with 10 significant clusters in sighting frequency between 1994 and 2015. B: nMDS with three significant clusters in trophic group richness between 1994 and 2015.

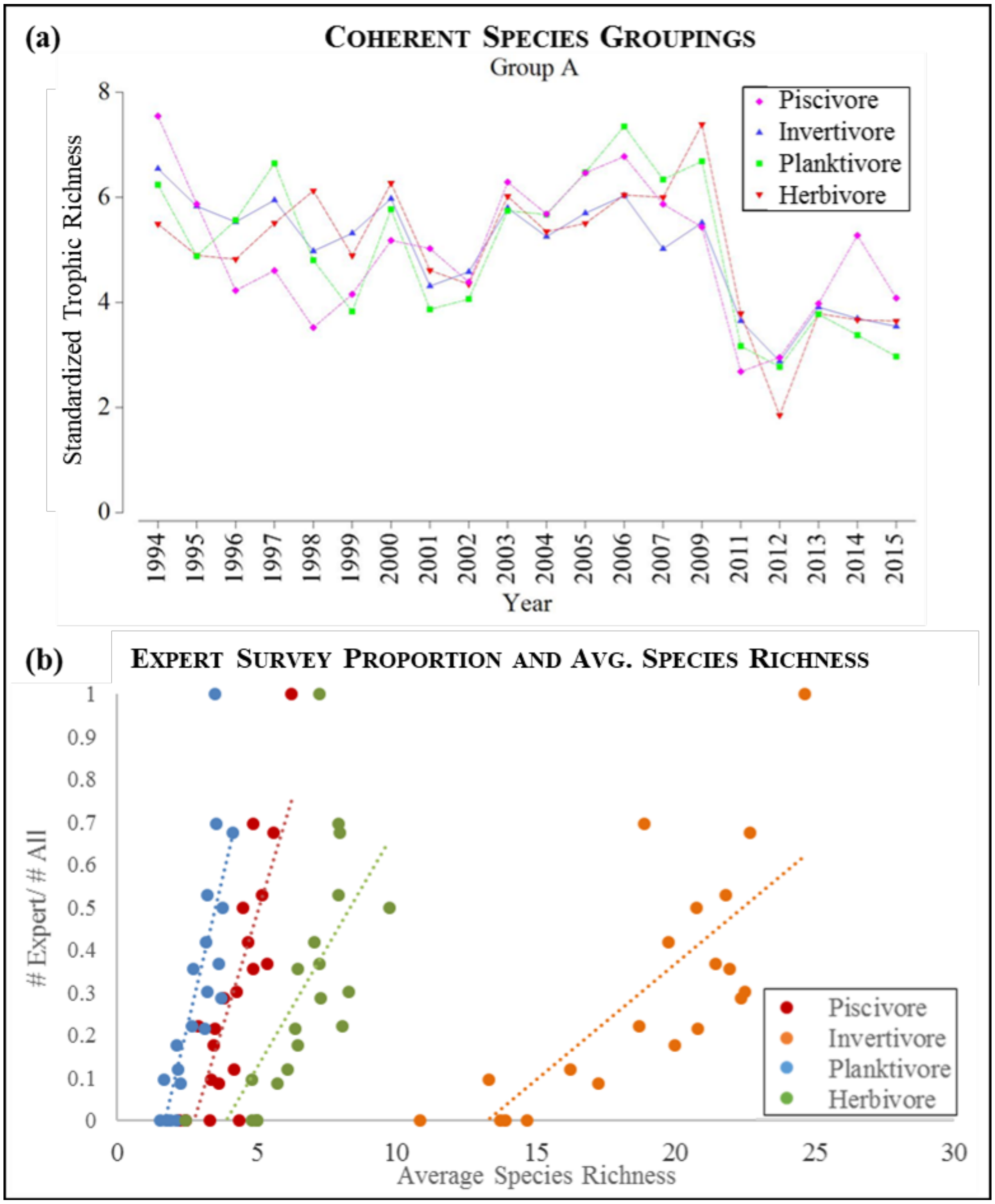


Figure 3.11. REEF data coherent variables and correlation plots. A: Single coherence plot of trophic guild richness, based on average yearly, variable standardized, and trophic richness data. B: correlation of average species richness with proportion of expert surveys to the total number of surveys in all trophic guilds.

Table 3.13. Kendall rank correlation results for trophic richness and proportion of expert surveys.

Trophic Guild	τ	P-value
Piscivore	0.66	**<0.001**
Invertivore	0.60	**<0.001**
Planktivore	0.67	**<0.001**
Herbivore	0.61	**<0.001**

Mesophotic Surveys

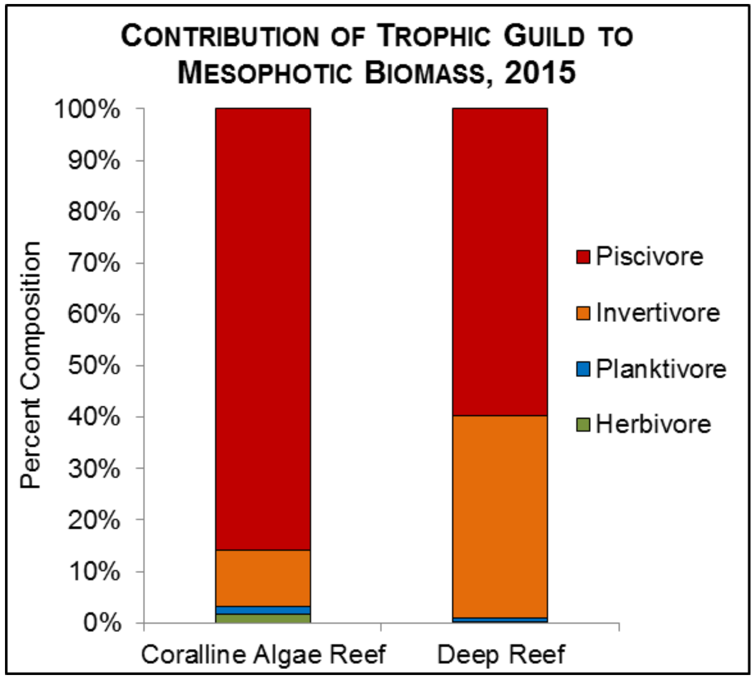


Figure 3.12. Mesophotic trophic biomass contribution.

In 2015, sighting frequency in coralline algae reef habitat was predominantly seven species (spotfin hogfish [*Bodianus pulchellus*], yellowtail reeffish [*Chromis enchrysura*], reef butterflyfish, rock hind [*Epinephelus adscensionis*], sharpnose puffer, cocoa damselfish, and purple reeffish [*Chromis scotti*]). Average density was predominantly composed of yellowtail reeffish, and average biomass was predominantly greater amberjack (*Seriola dumerili*). The average species richness in coralline algae reef habitat was 20. In deep reef habitat, red snapper (*Lutjanus campechanus*) were the most frequently observed species while both average density and biomass were predominantly tomtate (*Haemulon aurolineatum*) (Appendix C: Table C.1). Trophic biomass in both habitats was predominantly piscivores, with invertivores comprising the second highest biomass of all trophic guilds (Figure 3.12). The average species richness in deep reef habitat was 13.

When evaluating the entire dataset (2001-2015), total species richness in coralline algae reef habitat was 45 and in deep reef habitat was 62. The additional surveys lowered the average species richness in deep reef habitat to 11 species per survey. Trophic richness in both habitats was predominantly comprised of invertivores while herbivores had the lowest richness value (Table 3.14).

Table 3.14. Trophic richness of the mesophotic fish community.

Trophic Guild	Coralline Algae Reef	Deep Reef
Herbivore	4	2
Planktivore	6	5
Invertivore	24	21
Piscivore	10	11

Shade plots of fish presence/absence and density in deep reef habitat highlighted the 20 most important species for each dataset. In presence/absence data, species that were present throughout most surveys included rock hind, reef butterflyfish, yellowtail reeffish, red snapper, and tomtate. Regarding density data, the species highlighted in presence/absence data were present in varying density throughout the years, although shifting importance between species between years was observed. Lionfish and vermilion snapper density only became important in 2015 (Figure 3.13).

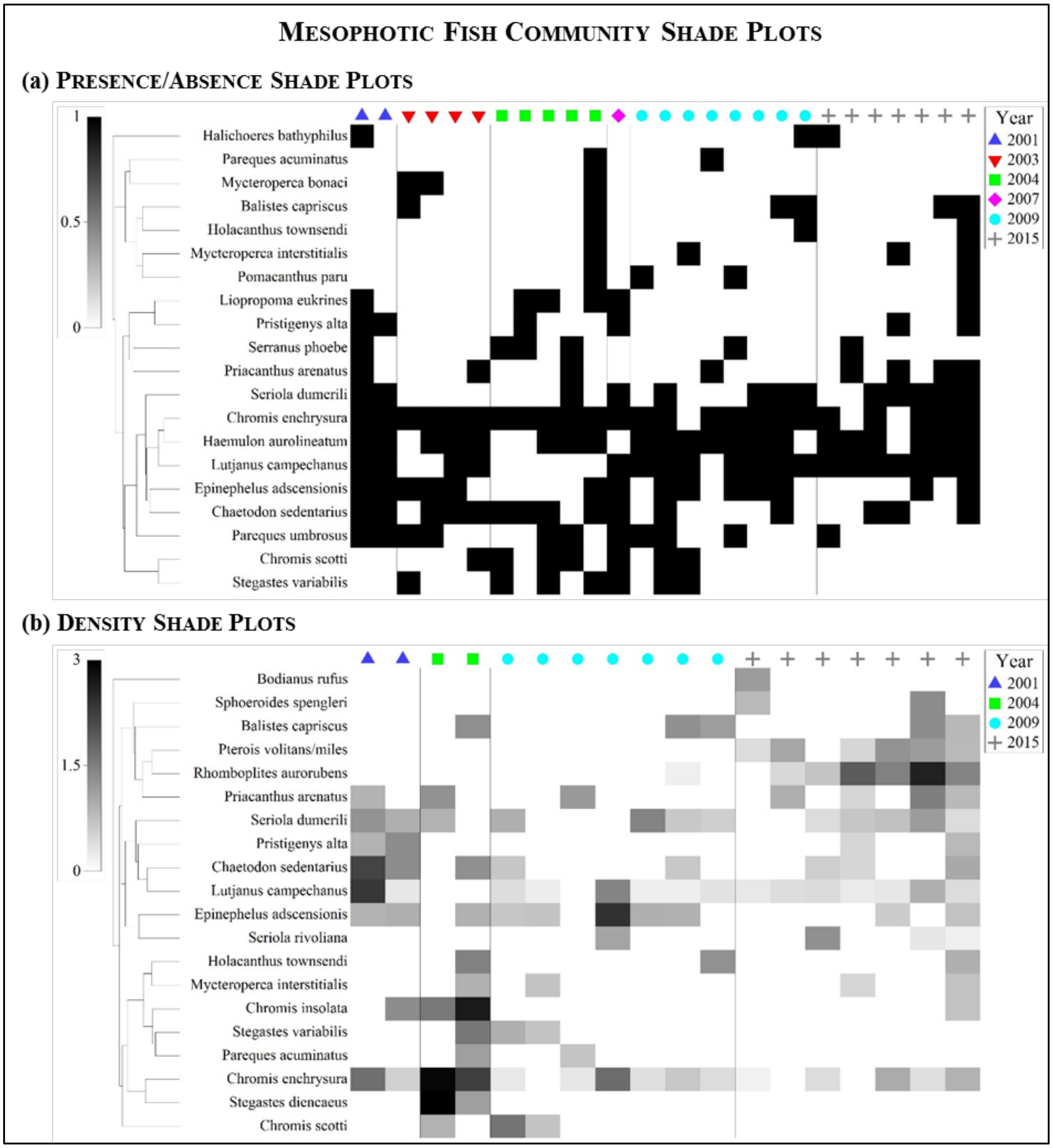


Figure 3.13. Mesophotic fish community shade plots. A: Presence/absence of the 20 most important species. B: density of the 20 most important species.

Spatial Interpolation

Spatial projection of fish data highlighted spatial patterns (Figure 3.14), although it should be noted that different methods were used to collect data on the bank crest (<33.5 m, scuba) and the mesophotic habitats (>33.5 m, ROV). Herbivore richness was tied strongly to the bank crest, where the majority of algae is found, while piscivore richness was greatest in deep reef habitat. Overall, richness was greatest on the bank crest, coralline algae reefs, and the southwestern portion of deep reef habitat. A feature in the southwestern portion of deep reef habitat was also highlighted as a biomass hotspot, where large schools of piscivorous grunts were found. The greatest density of fish was found on the northwestern portion of the bank crest and within coralline algae reefs.

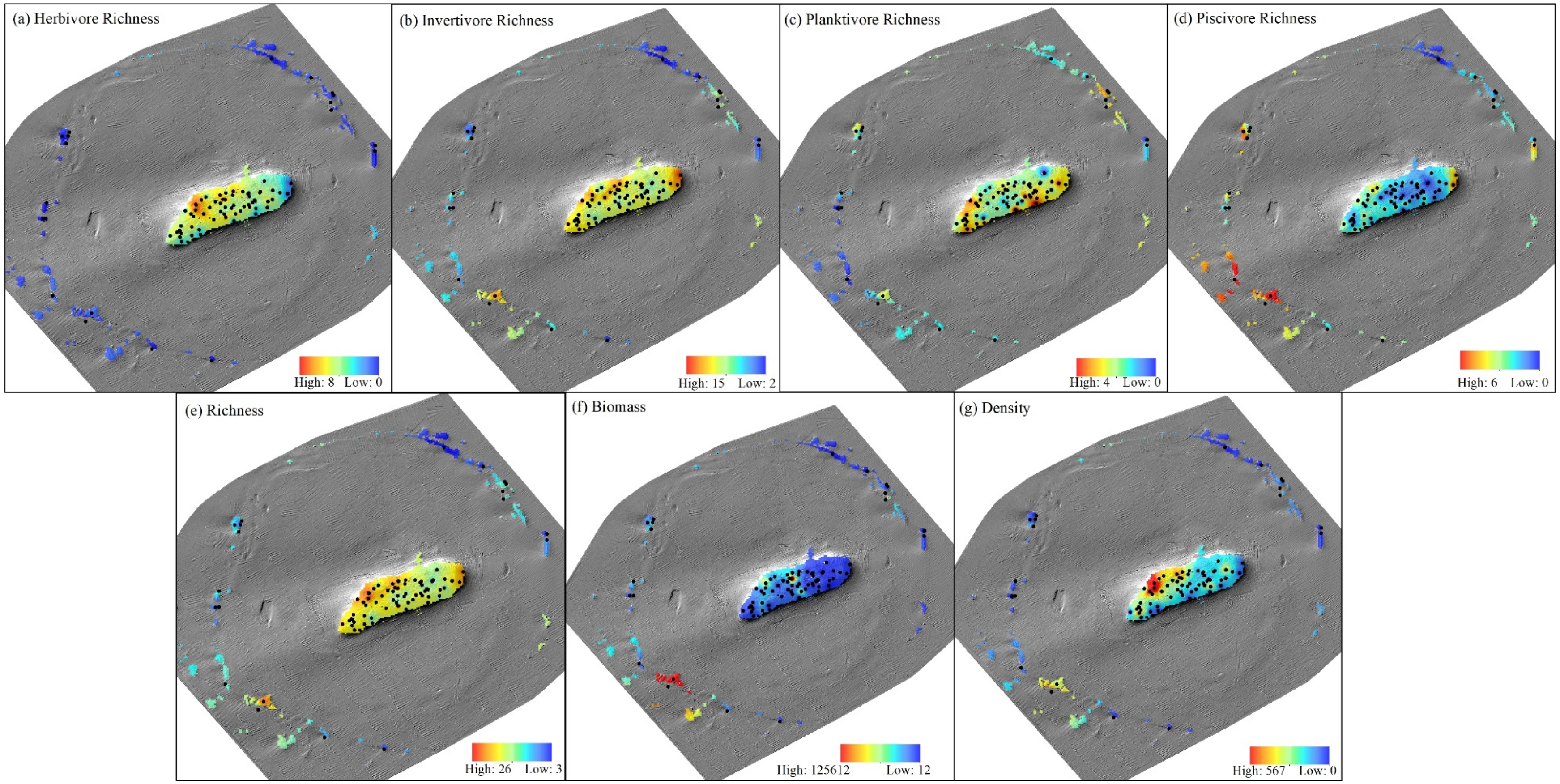


Figure 3.14. Inverse distance weighted fish richness, density, and biomass. A: Herbivore richness, B: invertivore richness, C: planktivore richness, D: piscivore richness, E: overall species richness, F: overall biomass, and G: overall density. Red represents high values, blue represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Groups of Interest

Due to particular interest in species including the grouper (*Mycteroperca*, *Cephalopholis, Epinephelus*, and *Dermatolepis* genera only), snapper (*Lutjanidae* genus only), grunt (*Haemulon* genera), parrotfish (*Sparisoma* and *Scarus* genera only), and angelfish (*Holacanthus* and *Pomacanthus* genera only) families, as well as the invasive lionfish species (*Pterois volitans*), separate analyses were conducted on these species.

Grouper

On the bank crest, the grouper family was comprised of six species in modified Bohnsack-Bannerot surveys, including graysby (*Cephalopholis cruentata*), rock hind (*Epinephelus adscensionis*), red hind (*Epinephelus guttatus*), yellowmouth grouper (*Mycteroperca interstitialis*), yellowfin grouper (*Mycteroperca venenosa*), and scamp (*Mycteroperca phenax*). Density and biomass indicated significant differences between habitats, but not between years, with no significant interactions (Table 3.15). Between habitats, differences were primarily due to a greater density and biomass of rock hind in high relief habitat, contributing over 60% to the dissimilarity in both comparisons (Table 3.16). Relative abundance size distribution and size at maturity were graphed for each species by habitat (Figure 3.15).

Table 3.15. PERMANOVA main test results for grouper density and biomass. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Grouper Density	Year	0.53	0.841	9937
	Habitat	3.66	**0.020**	9960
	Year*Habitat	0.87	0.503	9941
Grouper Biomass	Year	1.35	0.225	9943
	Habitat	6.98	**0.001**	9962
	Year*Habitat	0.38	0.845	9945

Table 3.16. Contribution of grouper species to dissimilarity in density and biomass between habitats. SIMPER results were based on data that were dispersion weighted and square root transformed.

	Density				Biomass			
Species	% Contrib.	Cum.%	Mean Density High Relief	Mean Density Low Relief	% Contrib.	Cum. %	Mean Biomass High Relief	Mean Biomass Low Relief
Epinephelidae: *Epinephelus adscensionis* (rock hind - I)	60.43	60.43	1.72	0.67	62.37	62.37	84.10	67.84
Epinephelidae: *Mycteroperca phenax* (scamp - P)	22.39	82.82	0.14	0.19	19.45	81.82	0.77	0.75
Epinephelidae: *Cephalopholis cruentata* (graysby - P)	7.27	90.08	0.14	0.04	8.65	90.47	8.77	2.05
Epinephelidae: *Mycteroperca interstitialis* (yellowmouth grouper - P)	7.10	97.18	0.06	0.05	6.04	96.51	13.28	19.94
Epinephelidae: *Epinephelus guttatus* (red hind - I)	2.45	99.63	0.02	0.01	2.99	99.51	3.27	0.56
Epinephelidae: *Mycteroperca venenosa* (yellowfin grouper - P)	0.37	100.00	0.01	0.00	0.49	100.00	86.52	0.00

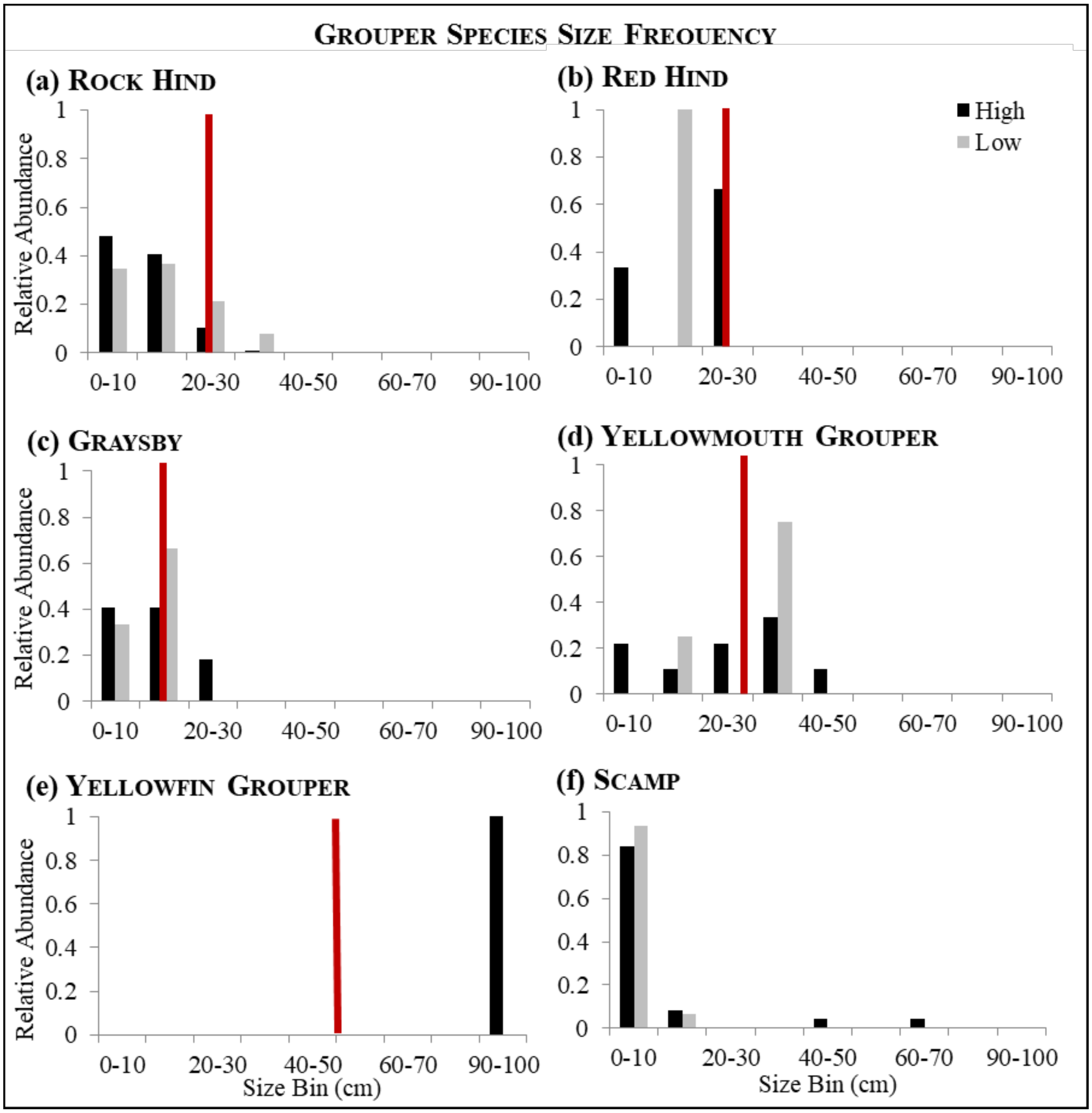


Figure 3.15. Size frequency of each grouper species. Graphs A-C are small-bodied grouper species and D-F are large-bodied grouper species. When data were available, red lines represent estimated size of female maturity (A: Bullock & Smith 1991; B, C, E: Froese & Pauly 2016; D: SAFMC 2005).

REEF surveys on the bank crest documented all the species in the modified Bohnsack-Bannerot surveys in addition to five other grouper species, including coney (*Cephalopholis fulva*), black grouper (*Mycteroperca bonaci*), marbled grouper, comb grouper, and gag grouper. REEF sighting frequency data found two significant year clusters (A: 1994-2011, 2013; B: 2012, 2014-2015), with rock hind and yellowmouth grouper driving the x distribution of the nMDS plot (0.80 correlation), where both species have seen reduced sighting frequencies over time (greater sighting frequency in cluster A than in cluster B) (Figure 3.16).

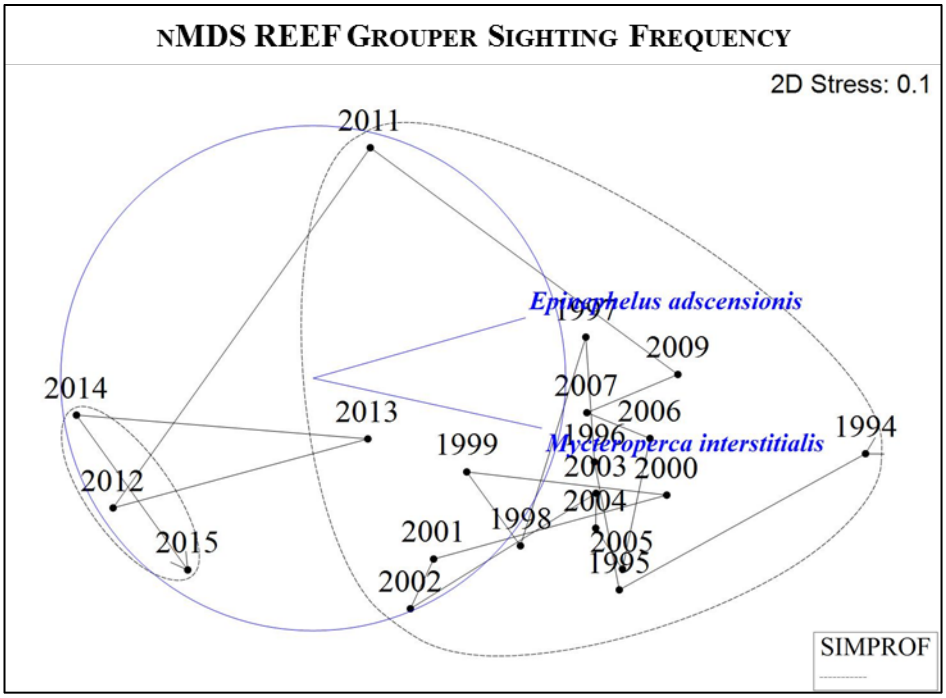


Figure 3.16. nMDS of REEF grouper sighting frequency. Blue lines represent a vector overlay of variables with >0.80 correlation.

In coralline algae reef and deep reef habitat, three additional species were observed: red hind, black grouper (*Mycteroperca bonaci*), and marbled grouper (*Dermatolepis inermis*).

When spatially projected, grouper density appeared greatest on the bank crest. However, biomass was greatest in deep reef habitat, indicating that while more grouper were seen on the bank crest, larger grouper were found in deep reef habitat (Figure 3.17).

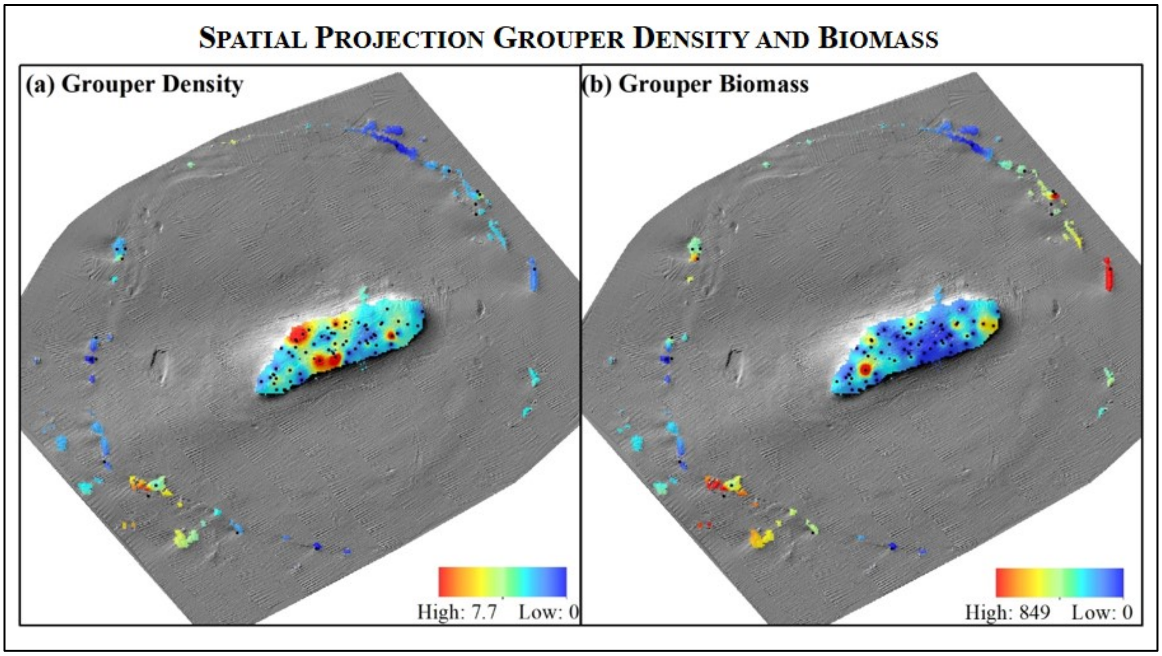


Figure 3.17. Inverse distance weighted grouper density and biomass. A is grouper density and B is grouper biomass. Red represents high values and green represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Snapper

The snapper family on the bank crest was composed of four species in modified Bohnsack-Bannerot surveys, including gray snapper (*Lutjanus griseus*), vermilion snapper (*Rhomboplites aurorubens*), dog snapper (*Lutjanus jocu*), and red snapper (*Lutjanus campechanus*). Density and biomass indicated significant differences between habitat and no significant interactions. However, between years, significant differences in snapper density, but not biomass, were found (Table 3.17). Between habitats, differences were primarily due to a greater density and biomass of gray snapper in high relief habitat, contributing over 60% to the dissimilarity in both comparisons (Table 3.18). Pairwise comparisons indicated that significant differences in consecutive years occurred in density between 2012-2013 and 2013-2014, but not between 2014 and 2015 (Table 3.18). The varying density of gray snapper between years was the greatest contributor to dissimilarity in all year comparisons (Appendix C: Table C.10). Relative abundance size distribution and size at maturity were graphed for each species by habitat, with red snapper size frequency in low relief habitat on the bank crest notably larger than in high relief habitat (Figure 3.18).

Table 3.17. PERMANOVA main test results for snapper density and biomass. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Snapper Density	Year	2.73	**0.012**	9937
	Habitat	4.16	**0.016**	9957
	Year*Habitat	1.21	0.293	9942
Snapper Biomass	Year	1.88	0.075	9945
	Habitat	7.52	**0.001**	9942
	Year*Habitat	1.65	0.152	9943

Table 3.18. Contribution of snapper species to dissimilarity in density and biomass between habitats. SIMPER resulted were based on data that were dispersion weighted and square root transformed.

Species	Density (#/100m^2)				Biomass (g/100m^2)			
	% Contrib.	Cum.%	Mean Density High Relief	Mean Density Low Relief	% Contrib.	Cum.%	Mean Biomass High Relief	Mean Biomass Low Relief
Lutjanidae: *Lutjanus griseus* (gray snapper - I)	63.30	63.30	5.61	0.71	64.61	64.61	1855.35	36.92
Lutjanidae: *Rhomboplites aurorubens* (vermilion snapper - P)	17.20	80.50	0.93	5.42	16.58	81.18	191.21	2057.33
Lutjanidae: *Lutjanus campechanus* (red snapper - P)	15.11	95.61	0.17	0.10	15.65	96.84	12.46	108.32
Lutjanidae: *Lutjanus jocu* (dog snapper - P)	4.39	100.00	0.03	0.00	3.16	100.00	94.66	0.00

Table 3.19. Pairwise PERMANOVA for snapper density by year. Bold denotes significant values.

Pairwise Test	Pseudo-F	P(perm)	Unique Permutations
2012, 2013	1.71	**0.046**	9915
2013, 2014	2.17	**0.005**	9964
2014, 2015	1.64	0.064	9969

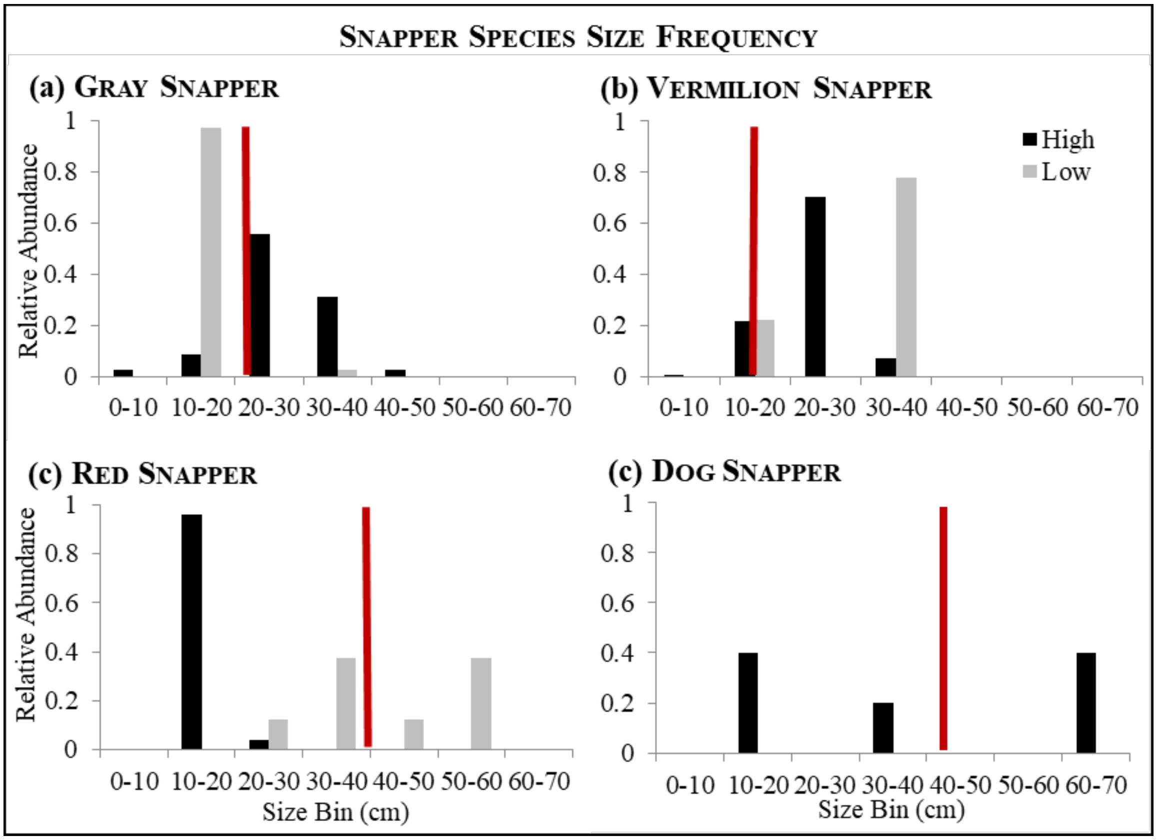


Figure 3.18. Size frequency of each snapper species. Red lines represent estimated size of female maturity (A, D: García-Cagide et al. 1994; B: Collins & Pinckney 1988; C: Froese & Pauly 2016).

REEF surveys on the bank crest documented all the species found in the modified Bohnsack-Bannerot surveys in addition to four other snapper species, including mutton snapper (*Lutjanus analis*), yellowtail snapper (*Ocyurus chrysurus*), blackfin snapper (*Lutjanus buccanella*), and cubera snapper (*Lutjanus cyanopterus*). REEF sighting frequency data found two significant year clusters (A: 1994-2008, 2010-2015; B: 2009), with gray snapper and yellowtail snapper exhibiting greater than 0.80 correlation, where both species had greater sighting frequency in cluster B than in cluster A (Figure 3.19).

In coralline algae reefs, two of the same snapper species seen on the bank crest were observed (red snapper and gray snapper). In deep reef habitat, the same species seen in coralline algae reef habitats were observed as well as one additional species observed on the bank crest (vermilion snapper).

When spatially projected, there are localized high densities of snapper on the bank crest with more widespread moderate density in the southwestern deep reef habitat. Similar localized high biomass was found on the bank crest, but biomass in deep reef was highest in the northeastern portion (Figure 3.20).

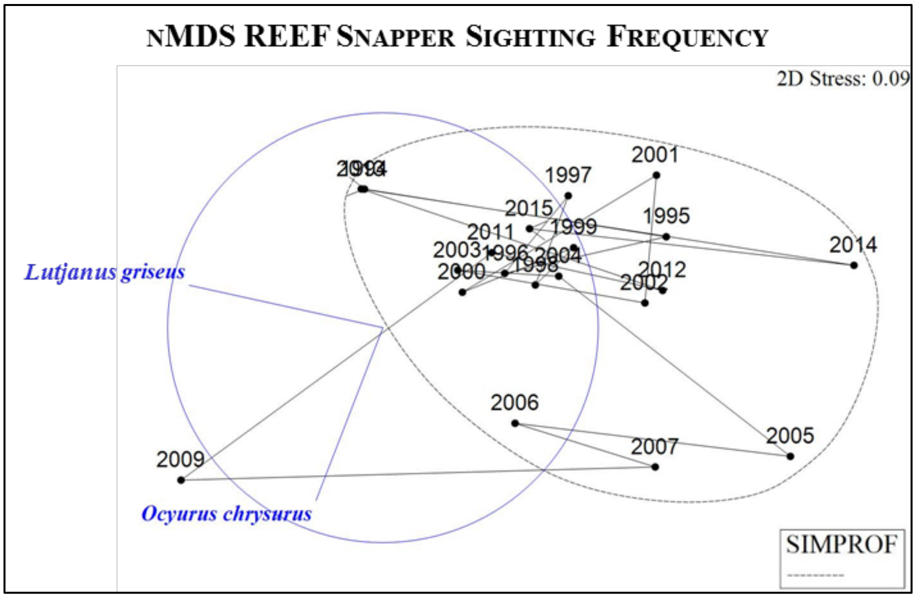


Figure 3.19. nMDS of REEF snapper sighting frequency. Blue lines represent a vector overlay of variables with >0.80 correlation.

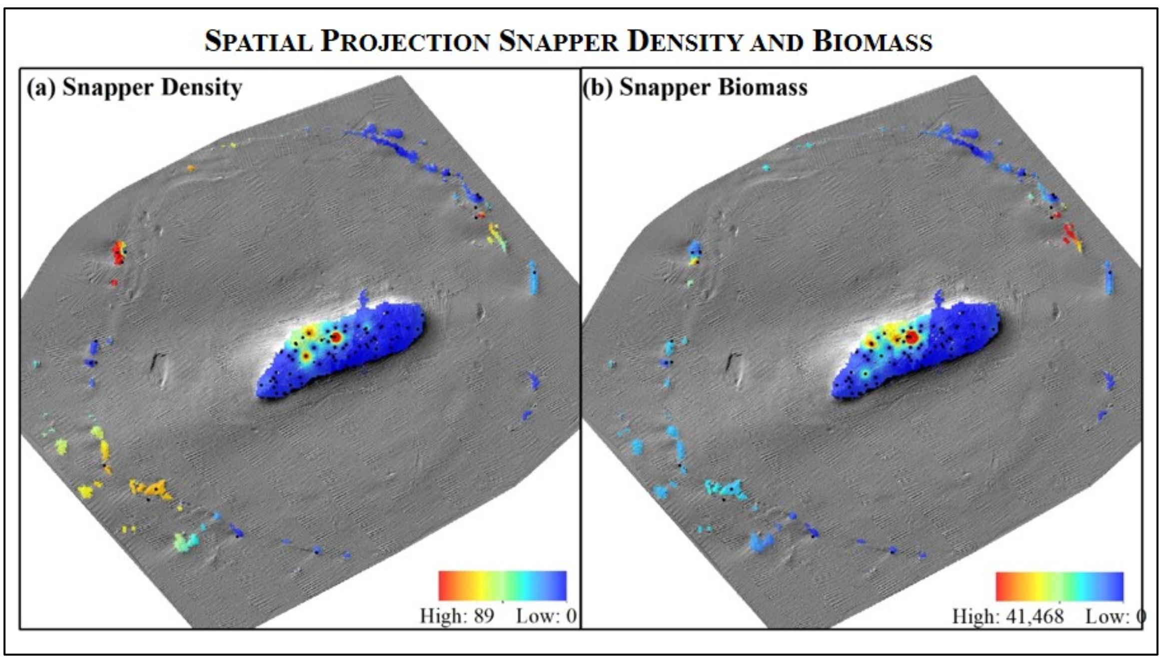


Figure 3.20. Inverse distance weighted snapper density and biomass. A is snapper density and B is snapper biomass. Red represents high values and green represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Grunt

The grunt family on the bank crest was composed of one species from modified Bohnsack-Bannerot surveys: cottonwick (*Haemulon melanurum*). Density and biomass indicated significant differences between years, but not between bank crest habitats, with no significant interactions (Table 3.20). Pairwise comparisons indicated no significant differences in consecutive years (Table 3.21). However, the increase in individuals in 2015 indicates a potential recruitment event for the species. Relative abundance size distribution with size at maturity and biomass means plots were graphed, highlighting the small size of individuals and supporting the theory of a potential recruitment event in 2015 (Figure 3.21).

Table 3.20. PERMANOVA for grunt main test results. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Grunt Density	Year	2.25	**0.045**	9932
	Habitat	0.57	0.496	9921
	Year*Habitat	0.18	0.860	9947
Grunt Biomass	Year	3.28	**0.018**	9952
	Habitat	2.00	0.176	9902
	Year*Habitat	1.30	0.280	9939

Table 3.21. Pairwise PERMANOVA for grunt density and biomass by year. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Pairwise Test	Pseudo-F	P(perm)	Unique Permutations
Grunt Density	2012, 2013	0.64	0.549	238
	2013, 2014	0.91	0.623	9606
	2014, 2015	1.67	0.083	9785
Grunt Biomass	2012, 2013	0.64	0.551	235
	2013, 2014	0.91	0.622	9593
	2014, 2015	1.97	0.071	9731

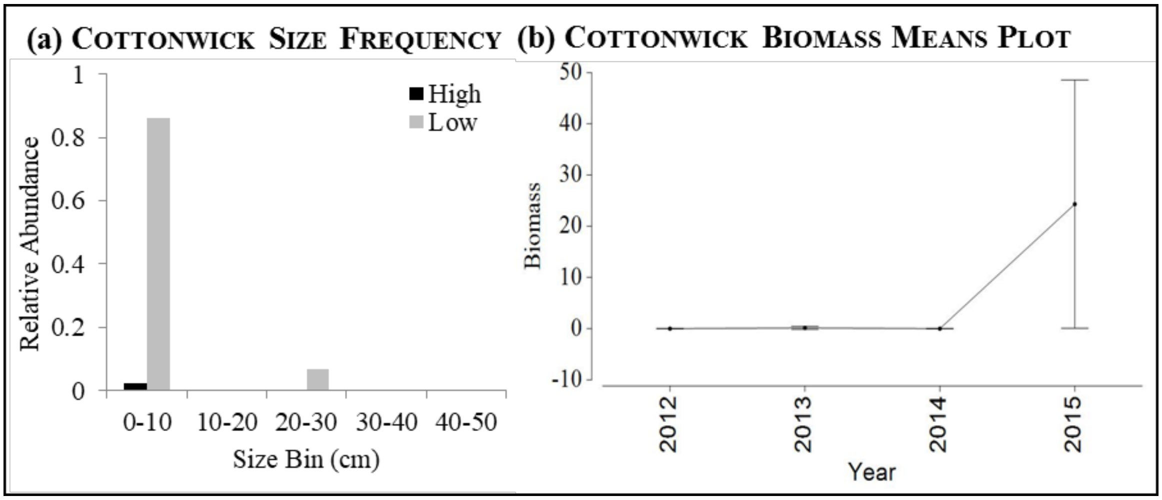


Figure 3.21. Size frequency and biomass of cottonwick.

REEF surveys on the bank crest documented cottonwick, seen in the modified Bohnsack-Bannerot surveys, in addition to two other grunt species: tomtate (*Haemulon aurolineatum*) and sailor's choice (*Haemulon parra*). No significant year clusters were observed in sighting frequency data.

On coralline algae reefs, only cottonwick were observed, and in deep reef habitat, only tomtate were observed. When spatially projected, the southwestern portion of the deep reef habitat had the greatest densities and biomass of grunts on the bank (Figure 3.22).

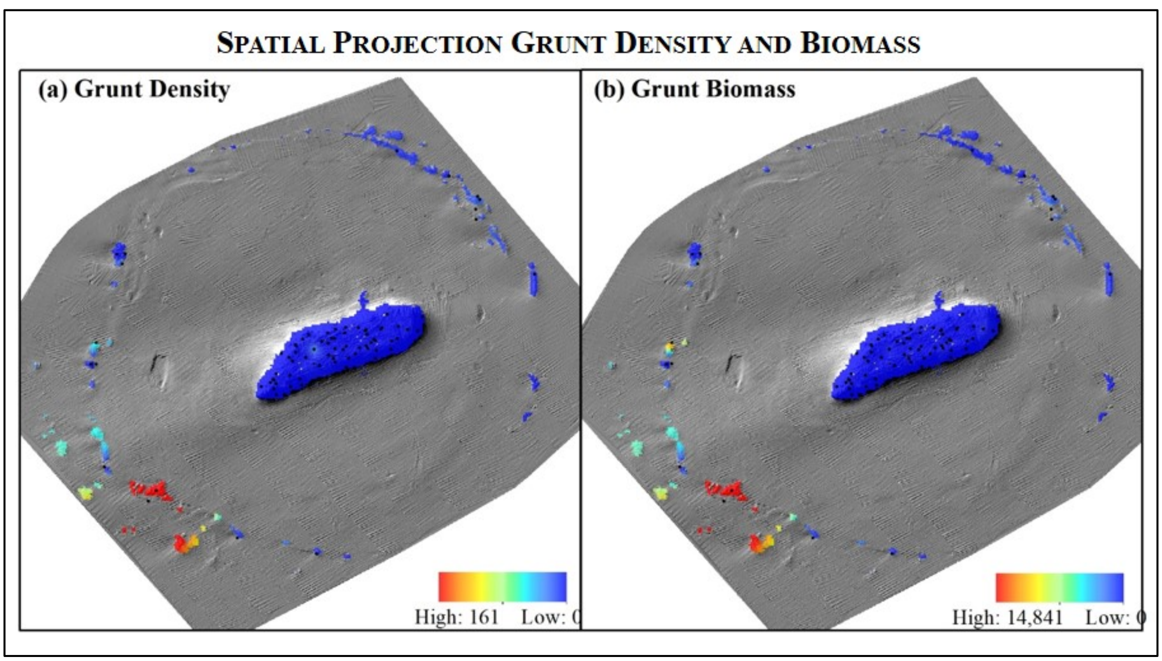


Figure 3.22. Inverse distance weighted grunt density and biomass. A is grunt density and B is grunt biomass. Red represents high values and green represents low values, and black dots mark survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Parrotfish

Parrotfishes have been identified as an important species group on coral reefs (Jackson et al. 2014). Seven species have been documented on the bank crest at Stetson Bank: striped parrotfish (*Scarus iseri*), princess parrotfish (*Scarus taeniopterus*), queen parrotfish (*Scarus vetula*), greenblotch parrotfish (*Sparisoma atomarium*), redband parrotfish (*Sparisoma aurofrenatum*), bucktooth parrotfish (*Sparisoma radians*), and stoplight parrotfish (*Sparisoma viride*). Density and biomass indicated significant differences between years and no significant interactions. Between habitats, significant differences in parrotfish density, but not biomass, were found (Table 3.22). Between habitats, differences were primarily due to a greater density of redband parrotfish and lower density of greenblotch parrotfish in high relief habitat compared to low relief habitat (Table 3.23). Pairwise comparisons indicated that significant differences in consecutive years occurred between 2012-2013 and 2013-2014, but not between 2014-2015, in both density and biomass (Table 3.24). The varying density and biomass of greenblotch parrotfish between years was the greatest contributor to the dissimilarity between years (Appendix C: Table C.11 and Table C.12). The population of parrotfish was dominated by small individuals ≤10 cm in both habitats and biomass has varied over the years with no obvious trend (Figure 3.23).

Table 3.22. PERMANOVA for parrotfish main test results. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Parrotfish Density	Year	6.92	**<0.001**	9923
	Habitat	3.19	**0.014**	9960
	Year*Habitat	1.44	0.189	9959
Parrotfish Biomass	Year	4.28	**<0.001**	9920
	Habitat	1.99	**0.093**	9941
	Year*Habitat	1.37	0.205	9931

Table 3.23. Contribution of parrotfish species to dissimilarity in density between habitats. SIMPER results were based on data that were dispersion weighted and square root transformed.

Species	Density (#/100m^2)			
	% Contrib.	Cum.%	Average Density High Relief	Average Density Low Relief
Labridae: *Sparisoma aurofrenatum* (redband parrotfish - H)	34.50	34.50	0.76	0.18
Labridae: *Sparisoma atomarium* (greenblotch parrotfish - H)	29.04	63.54	2.13	4.21
Labridae: *Scarus taeniopterus* (princess parrotfish - H)	12.37	75.91	0.47	0.28
Labridae: *Sparisoma viride* (stoplight parrotfish - H)	8.50	84.41	0.11	0.07
Labridae: *Sparisoma radians* (bucktooth parrotfish - H)	8.10	92.51	0.09	0.59
Labridae: *Scarus iseri* (striped parrotfish - H)	6.30	98.81	0.56	0.18
Labridae: *Scarus vetula* (queen parrotfish - H)	1.19	100.00	0.00	0.01

Table 3.24. Pairwise PERMANOVA for parrotfish density and biomass by year. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Pairwise Test	Pseudo-F	P(perm)	Unique Permutations
Parrotfish Density	2012, 2013	2.00	**0.005**	9944
	2013, 2014	2.71	**<0.001**	9953
	2014, 2015	1.58	0.065	9962
Parrotfish Biomass	2012, 2013	1.84	**0.011**	9946
	2013, 2014	1.88	**0.009**	9942
	2014, 2015	1.20	0.213	9936

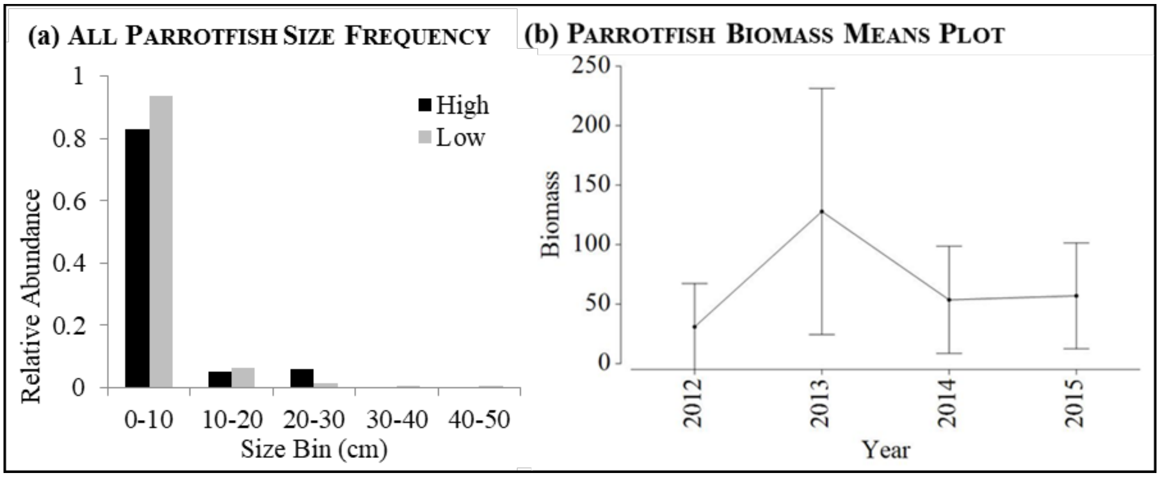


Figure 3.23. Parrotfish family size frequency and biomass, where A is parrotfish size frequency for all species together and B is a means plot of parrotfish biomass for all species together.

REEF surveys on the bank crest documented all the species identified in the modified Bohnsack-Bannerot surveys, except bucktooth parrotfish, and documented three additional parrotfish species: blue parrotfish (*Scarus coeruleus*), redtail parrotfish (*Sparisoma chrysopterum*), and rainbow parrotfish (*Scarus guacamaia*). No significant clusters in parrotfish sighting frequency between 1994 and 2015 were found. Coherence plots composed of parrotfish sighting frequency and average yearly hydrocoral and scleractinian coral cover show that the sighting frequencies of princess parrotfish, queen parrotfish, stoplight parrotfish, and greenblotch parrotfish co-varied with hydrocoral and scleractinian coral cover, where sighting frequency and coral cover declined between 2006 and 2015 (Figure 3.24), but no significant trend was found. While an overall decline in species richness was observed in REEF data from 2011 to 2015 as a result of a reduced proportion of expert surveys, co-varying declines in benthic cover and parrotfish sighting frequency occurred prior to 2011, suggesting that the decline in proportion of expert surveys did not drive the finding of covariance.

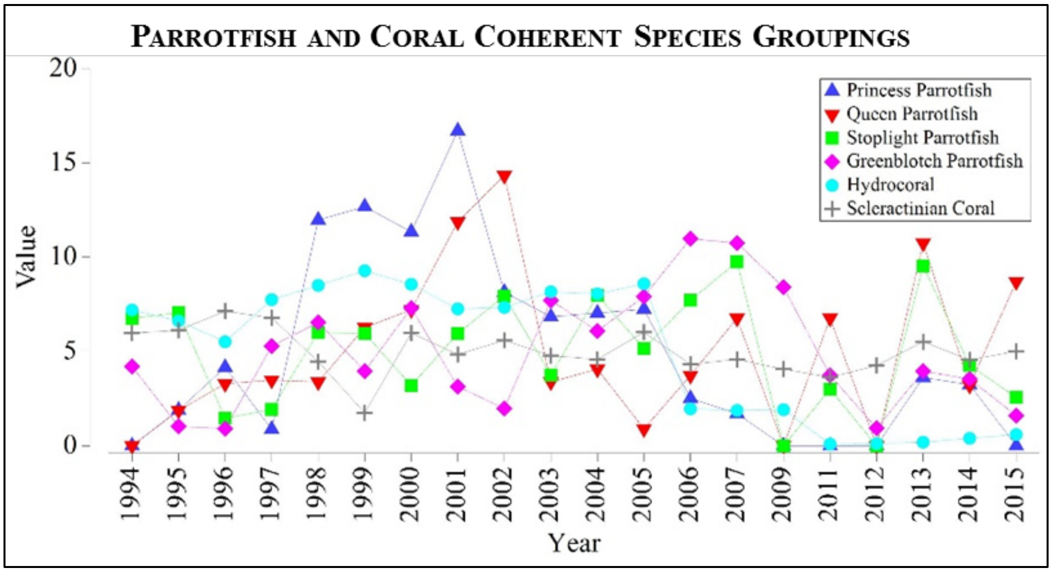


Figure 3.24. Line plot of co-varying parrotfish sighting frequency and coral cover. Four of nine species of parrotfish were found to co-vary with hydrocoral and scleractinian coral cover.

No parrotfish species were observed in coralline algae reef habitat or deep reef habitat. Spatial projection of parrotfish density and biomass confirms this observation, that parrotfish were restricted to the bank crest, where areas on the central southern bank crest possessed the greatest density and biomass (Figure 3.25).

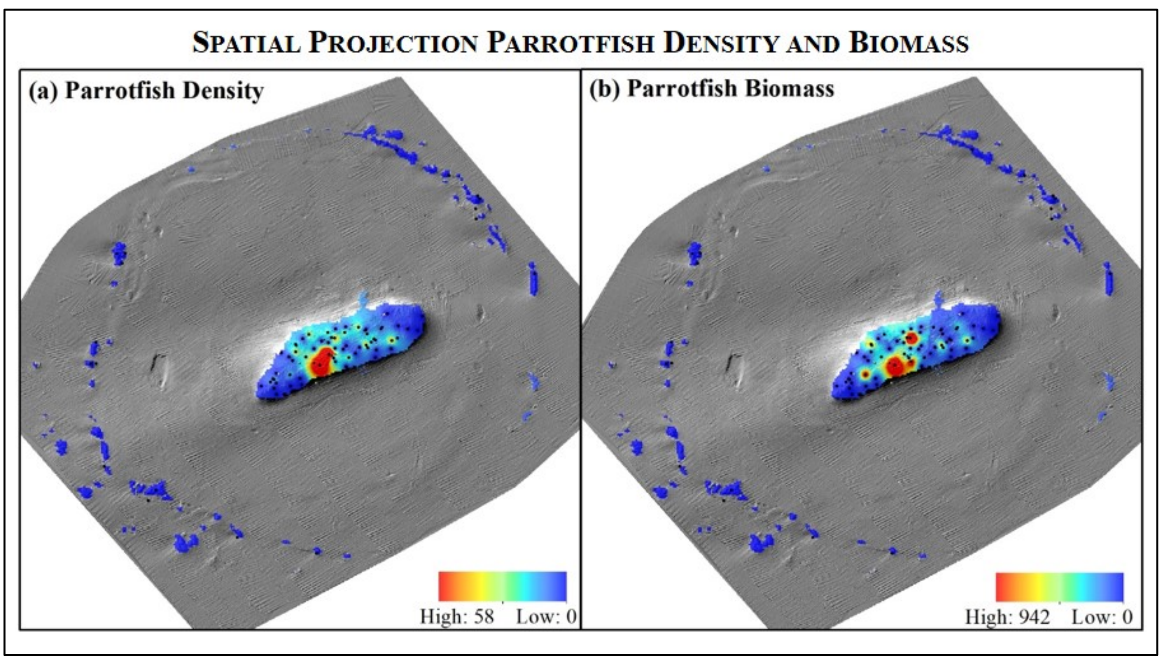


Figure 3.25. Inverse distance weighted parrotfish density and biomass. A is parrotfish density and B is parrotfish biomass. Red represents high values and green represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Angelfish

Several species of angelfish are known spongivores, making them of particular interest where sponge cover is a major component of the benthic biota, like at Stetson Bank. While not all spongivores, five species of angelfish were documented on the bank crest in modified Bohnsack-Bannerot surveys, including blue angelfish (*Holacanthus bermudensis*), queen angelfish (*Holacanthus ciliaris*), Townsend angelfish (*Holacanthus townsendi*), rock beauty (*Holacanthus tricolor*), and French angelfish (*Pomacanthus paru*). Density and biomass indicated significant differences between years, with significant differences between habitat only in density, in addition to a significant interaction between year and habitat in density (Table 3.25). Between habitats, differences were primarily due to a greater density of French angelfish in high relief habitat compared to low relief habitat (Table 3.26). Pairwise comparisons indicated that no significant differences occurred in consecutive years for density and biomass, except between 2014 and 2015 for density (Table 3.27). The reduced density of French angelfish between 2014 and 2015 contributed over 40% to the dissimilarity between years (Table 3.8). The population of angelfish was spread among size bins, with the majority of individuals in low relief habitat in the 10-20 cm size range and the majority of individuals in high relief habitat in the 30-40 cm size range. Biomass appears to have increased over time, although trends were not tested for significance (Figure 3.26).

Table 3.25. PERMANOVA for angelfish main test results. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Main Test	Pseudo-F	P(perm)	Unique Permutations
Angelfish Density	Year	3.14	**0.004**	9942
	Habitat	3.46	**0.025**	9960
	Year*Habitat	2.35	**0.040**	9949
Angelfish Biomass	Year	2.40	**0.021**	9937
	Habitat	2.26	0.084	9948
	Year*Habitat	1.81	0.107	9938

Table 3.26. Contribution of angelfish species to dissimilarity in density between habitats. SIMPER results were based on data that were dispersion weighted and square root transformed.

Species	Density (#/100m^2)			
	% Contrib.	Cum.%	Average Density High Relief	Average Density Low Relief
Pomacanthidae: *Pomacanthus paru* (French angelfish - I)	42.89	42.89	1.07	0.89
Pomacanthidae: *Holacanthus ciliaris* (queen angelfish - I)	24.07	66.96	0.26	0.18
Pomacanthidae: *Holacanthus bermudensis* (blue angelfish - I)	19.95	86.91	0.15	0.35
Pomacanthidae: *Holacanthus tricolor* (rock beauty - I)	9.66	96.57	0.22	0.07
Pomacanthidae: *Holacanthus townsendi* (Townsend angelfish - I)	3.43	100.00	0.03	0.06

Table 3.27. Pairwise PERMANOVA for angelfish density and biomass by year. Bray-Curtis similarity matrices were based on data that were dispersion weighted and square root transformed. Bold denotes significant values.

Data	Pairwise Test	Pseudo-F	P(perm)	Unique Permutations
Angelfish Density	2012, 2013	0.50	0.831	9946
	2013, 2014	1.66	0.057	9947
	2014, 2015	1.82	**0.031**	9959
Angelfish Biomass	2012, 2013	0.79	0.617	9957
	2013, 2014	1.06	0.327	9961
	2014, 2015	1.53	0.090	9955

Table 3.28. Contribution of angelfish species to dissimilarity in density between 2014 and 2015. SIMPER results were based on data that were dispersion weighted and square root transformed.

Species	Density (#/100m^2)			
	% Contrib.	Cum.%	Average Density 2014	Average Density 2015
Pomacanthidae: *Pomacanthus paru* (French angelfish - I)	42.69	42.69	1.46	1.18
Pomacanthidae: *Holacanthus ciliaris* (queen angelfish - I)	23.60	66.29	0.40	0.15
Pomacanthidae: *Holacanthus bermudensis* (blue angelfish - I)	23.05	89.34	0.11	0.49
Pomacanthidae: *Holacanthus tricolor* (rock beauty - I)	10.04	99.38	0.13	0.22
Pomacanthidae: *Holacanthus townsendi* (Townsend angelfish - I)	0.62	100.00	0.03	0.00

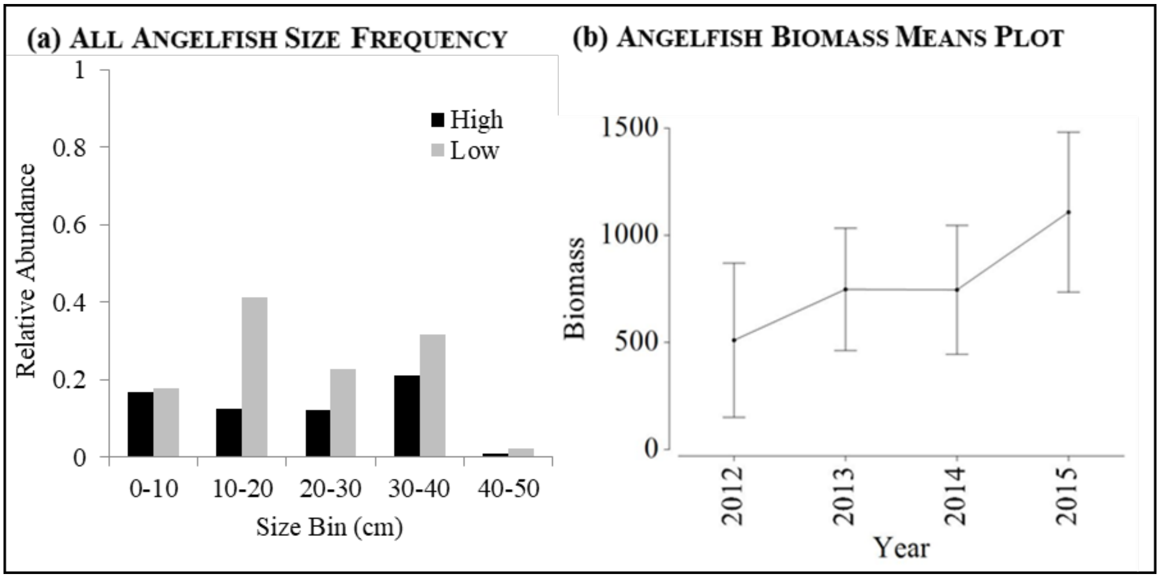


Figure 3.26. Angelfish family size frequency and biomass, where A is angelfish size frequency for all species together and B is a means plot of angelfish biomass for all species together.

REEF surveys on the bank crest documented all the species in the modified Bohnsack-Bannerot surveys in addition to the gray angelfish (*Pomacanthus arcuatus*). Seven significant clusters in angelfish sighting frequency were found (A: 1994-1997 and 1999; B: 1998 and 2002-2007; C: 2000-2001; D: 2009; E: 2011; F: 2012; and G: 2013-2015). Blue angelfish, rock beauty, French angelfish, and queen angelfish exhibited greater than 0.80 correlation, with most species generally declining over time (Figure 3.27). Coherence plots of angelfish sighting frequency with average total sponge cover showed that the sighting frequencies of rock beauty and blue angelfish co-varied with total sponge cover. When angelfish sighting frequency was evaluated with cover of each sponge species, blue angelfish were found to co-vary with *Ircinia strobilina* cover (Figure 3.28). The sighting frequency of rock beauty and blue angelfish exhibited a significantly declining trend over time (τ=-0.50, p=0.003 and τ=-0.59, p<0.001, respectively), similar to the trend seen in sponge cover (τ=-0.79, p<0.001 [Chapter 2]).

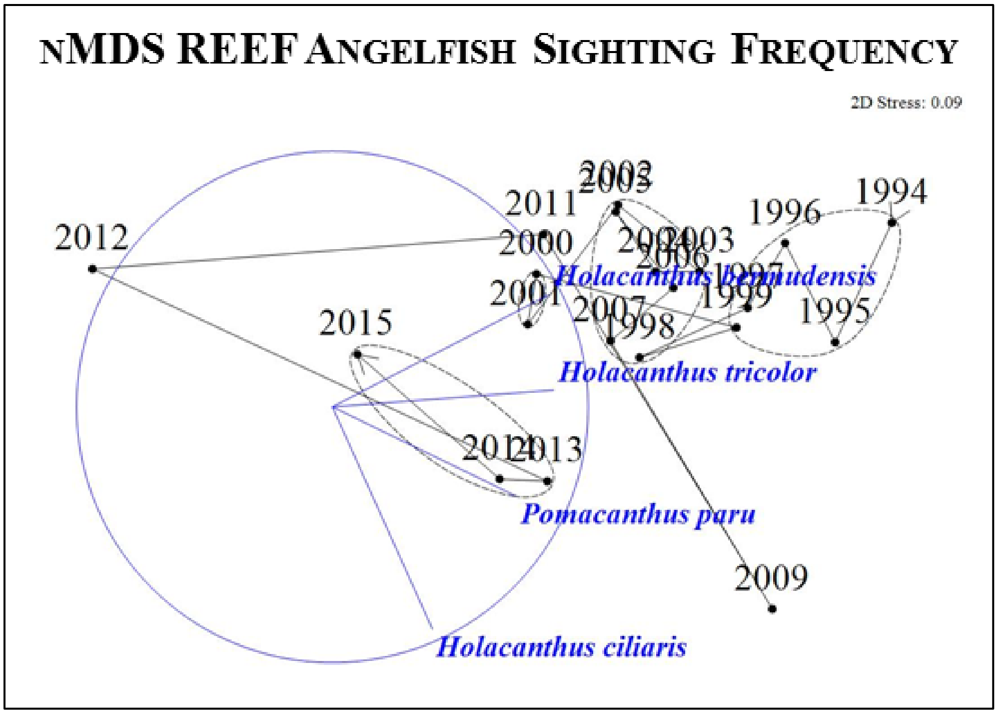


Figure 3.27. nMDS of REEF angelfish sighting frequency. Blue lines represent a vector overlay of variables with >0.80 correlation.

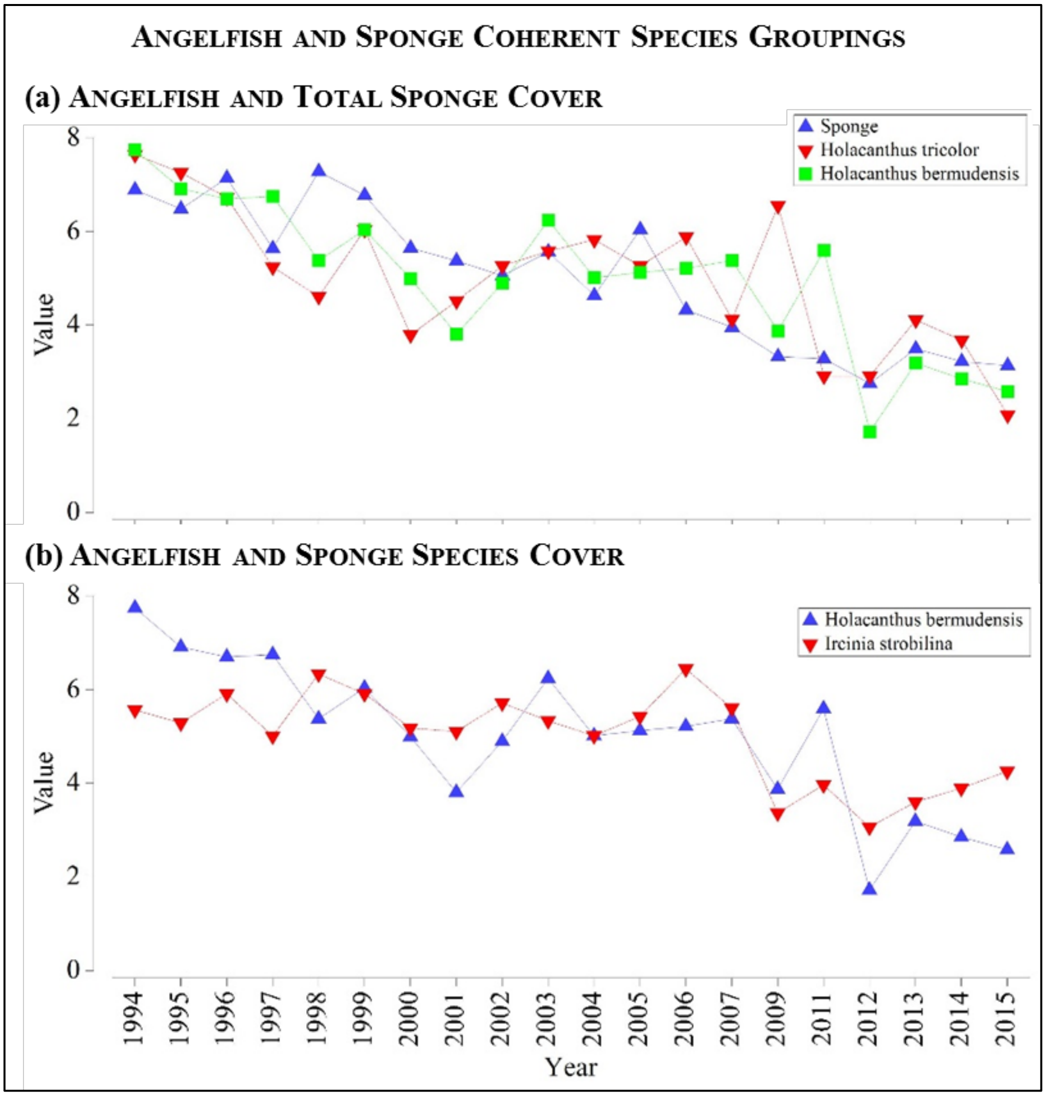


Figure 3.28. Line plot of co-varying angelfish sighting frequency and sponge cover, where A shows angelfish species co-varying with total sponge cover and B shows a single angelfish species co-varying with a sponge species.

In coralline algae reefs, five of the same angelfish species seen on the bank crest were observed (blue angelfish, queen angelfish, Townsend angelfish, rock beauty, and French angelfish). In deep reef habitat, these same species were observed with the exception of rock beauty and the addition of gray angelfish.

When spatially projected, the northeastern and southwestern portions of deep reef habitat had noticeably greater angelfish densities. Biomass of angelfish on the bank crest had a similar spatial pattern to sponge cover (Figure 2.19). However, the greatest angelfish biomass was seen in the southwestern deep reef habitat (Figure 3.29).

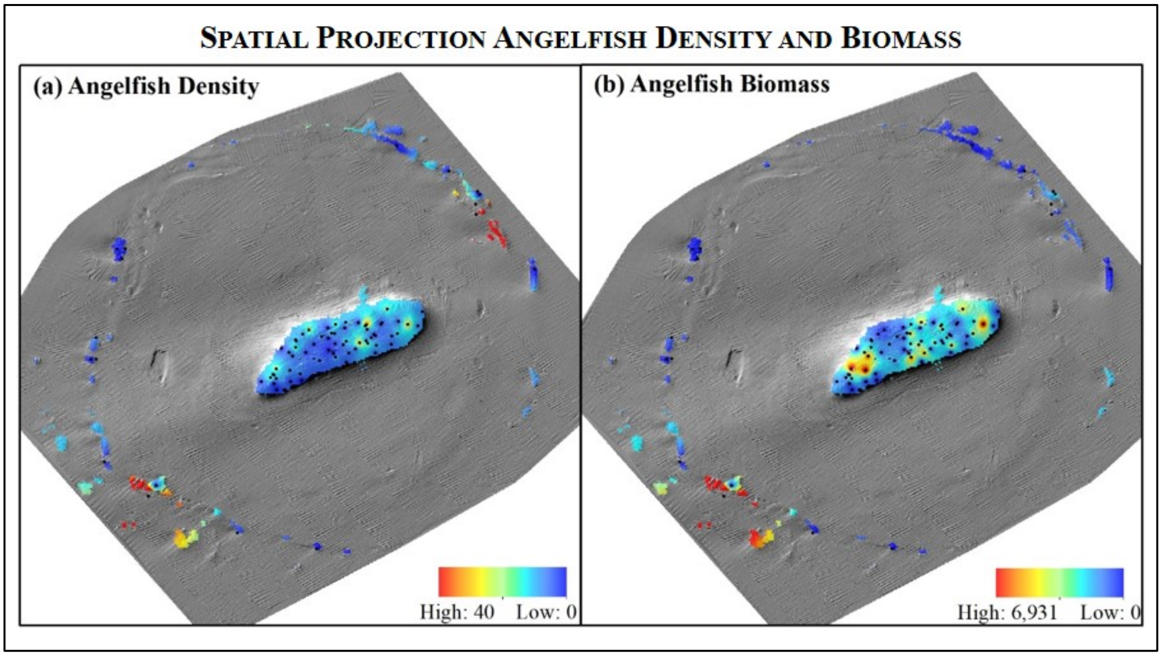


Figure 3.29. Inverse distance weighted angelfish density and biomass. A is angelfish density and B is angelfish biomass. Red represents high values and green represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Lionfish

Lionfish on the bank crest were only recorded in modified Bohnsack-Bannerot data three times (twice in 2013 and once in 2014) and once in REEF data in 2013. Due to the limited size of this dataset, additional analysis was not conducted. However, Johnston et al. (2016b) obtained lionfish observations from general scientific dives at Stetson Bank between 2011 and 2014, where each dive was treated as a sample and the observation of lionfish noted. The study documented the first observation of lionfish at Stetson Bank in 2011, followed by annually increasing observations, mean total length, and mean body weight until 2013, when observations declined but mean total length and mean body weight continued to increase (Figure 3.30).

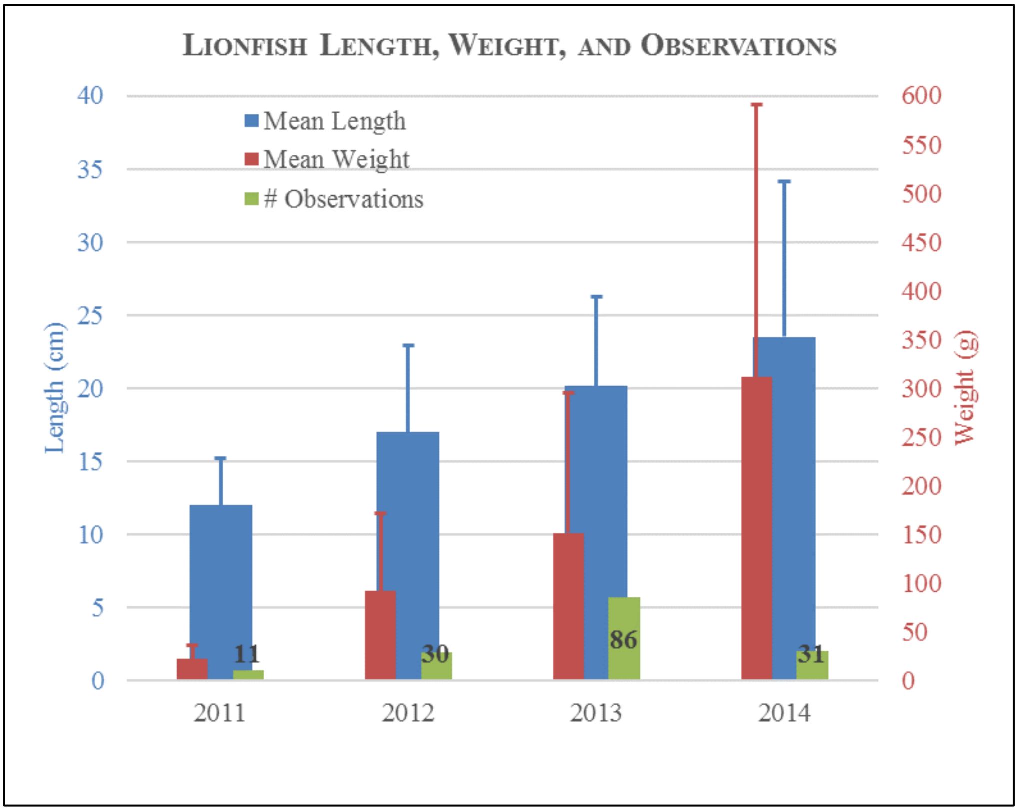


Figure 3.30. Mean length, weight, and total observations of lionfish. Data from Johnston et al. 2016b. Errors bars are standard error. Blue represents mean length, red represents mean weight, and green represents the total observations, with the number of observations labeled for year.

While observed in gradually increasing frequency on the bank crest between 2011 and 2013, lionfish were more prevalent in both coralline algae and deep reef habitat than bank crest surveys in 2015 (Table 3.29). Survey methods between the bank crest and coralline algae/deep reef habitats were different, with bank crest surveys conducted using scuba divers and modified Bohnsack-Bannerot and roving diver surveys while coralline algae/deep reef surveys were conducted using ROV transects.

Table 3.29. Lionfish sighting frequency, density, and biomass in all habitats for 2015. Density is in #/100m^2 and biomass is in g/100m^2.

Habitat	Sighting Frequency	Sighting Frequency Rank	Average Density	Average Biomass
Bank Crest	0.0	-	0.00	0.0
Coralline Algae Reef	40.0	6	0.07	53.9
Deep Reef	85.7	2	0.55	83.6

When spatially projected, the low densities of lionfish on the reef crest was evident, with a concentration of lionfish density and biomass in the southwestern portion of deep reef habitat (Figure 3.31).

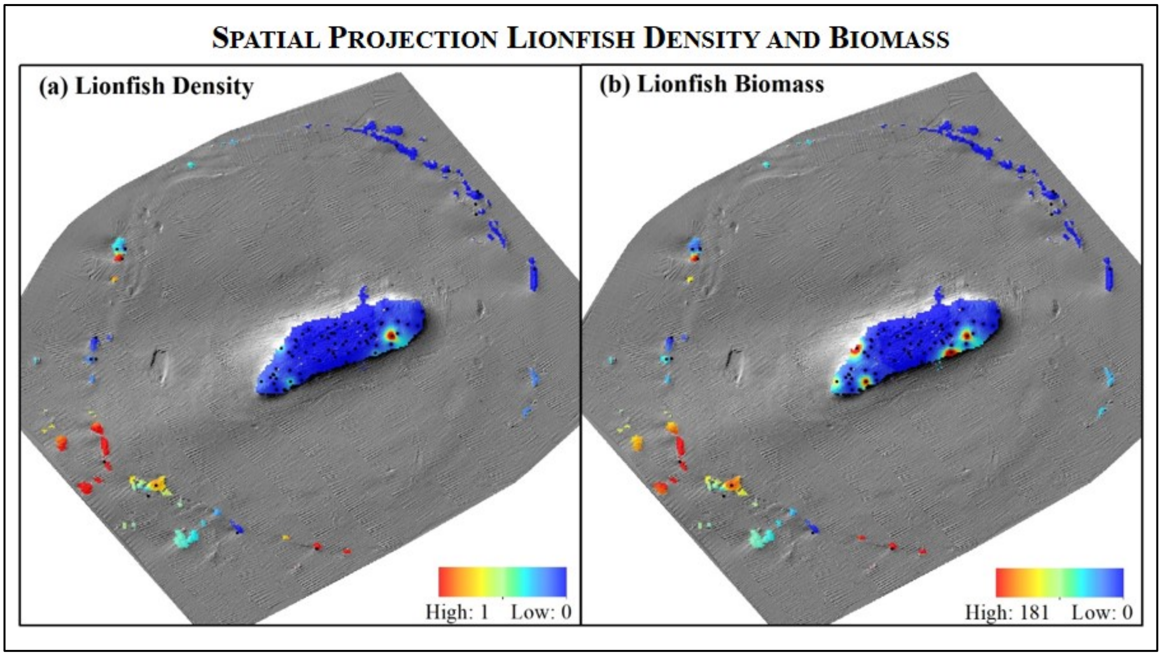


Figure 3.31. Inverse distance weighted lionfish density and biomass. A is lionfish density and B is lionfish biomass. Red represents high values and green represents low values, and black dots mark the survey locations. Biomass is in g/100m^2 and density is in #/100m^2. Image: NOAA

Shark and Ray

Two species of shark and ray were documented on the bank crest in modified Bohnsack-Bannerot surveys; sandbar shark (*Carcharhinus plumbeus*) and southern stingray (*Dasyatis americana*). Sightings of these species were consistently low throughout the years and density indicated no significant differences between year or habitat, and no significant interactions. Relative abundance size distribution and size at maturity were graphed for each species by habitat (Figure 3.32).

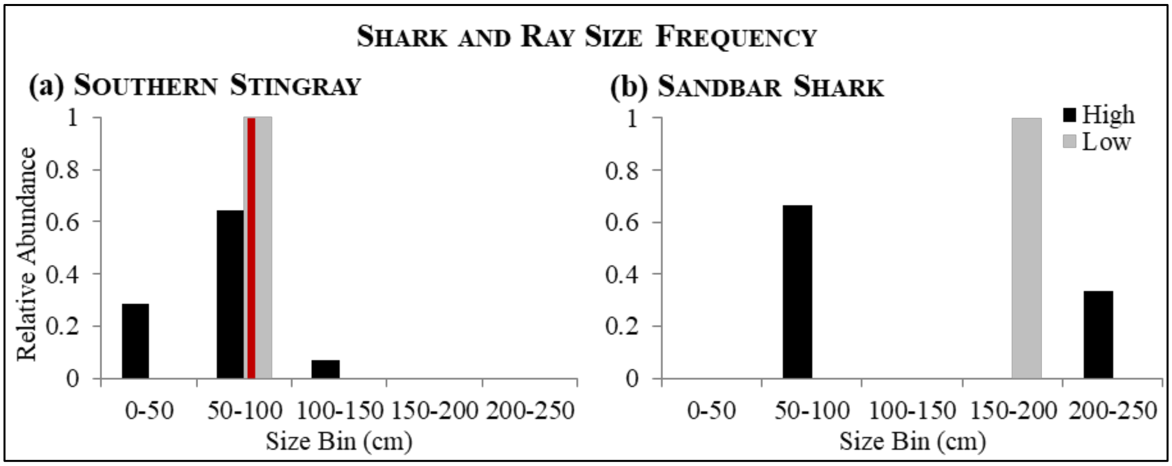


Figure 3.32. Size frequency of each shark and ray species. Red lines represent estimated size of female maturity (A: Froese & Pauly 2016).

REEF surveys on the bank crest documented the two species found in the modified Bohnsack-Bannerot surveys and 13 other shark and ray species, including spinner shark (*Carcharhinus brevipinna*), silky shark (*Carcharhinus falciformis*), bull shark (*Carcharhinus leucas*), blacktip shark (*Carcharhinus limbatus*), dusky shark (*Carcharhinus obscurus*), reef shark (*Carcharhinus perezi*), nurse shark (*Ginglymostoma cirratum*), scalloped hammerhead (*Sphyrna lewini*), great hammerhead (*Sphyrna mokarran*), spotted eagle ray (*Aetobatus narinari*), roughtail stingray (*Dasyatis centroura*), manta ray (*Manta birostris*), and mobula ray (*Mobula tarapacana*). Sighting frequency data showed no significant year clusters between 1993 and 2015.

In coralline algae reef habitat, sandbar shark and southern stingray were recorded on ROV surveys. No sharks or rays were recorded in deep reef habitat.

When spatially projected, the densities of sharks and rays are concentrated around the central feature of the bank, with localized increased densities near the edges of the bank crest (Figure 3.33).

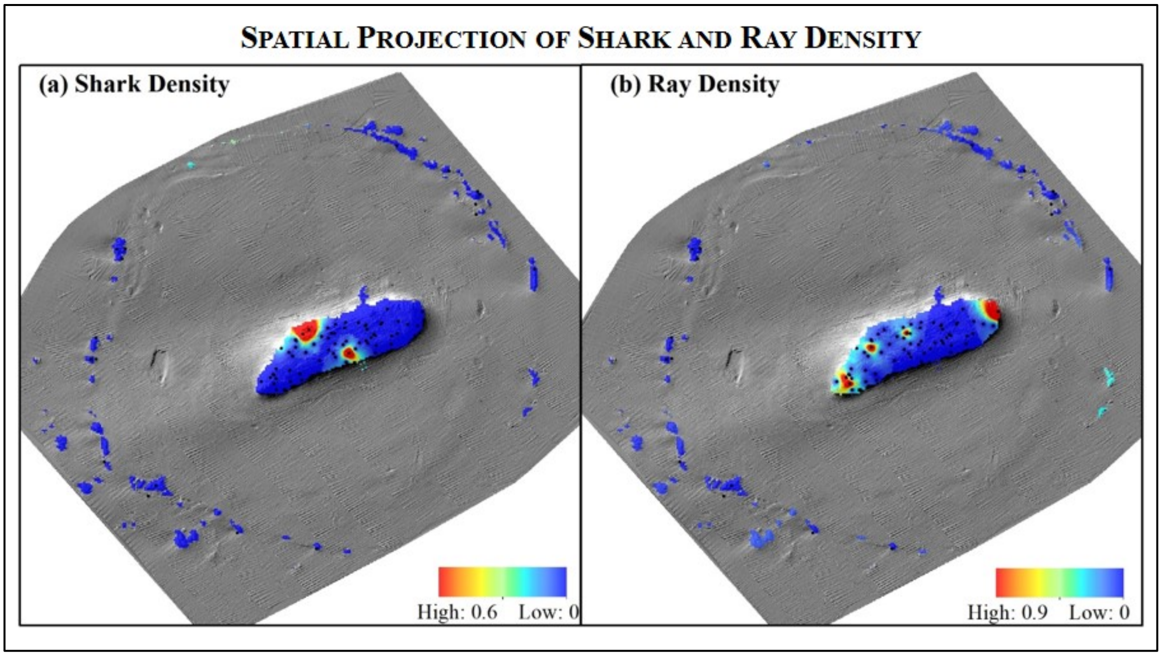


Figure 3.33. Inverse distance weighted A shark and B ray density. Red represents high values and green represents low values, and black dots mark the survey locations. Density measured in #/100m^2. Image: NOAA

Discussion

Fish communities are considered indicators of ecosystem health (Sale 1991) and are therefore an important component in long-term monitoring programs. Monitoring fish community changes over extended periods of time is valuable in detecting changes beyond the normal variations in the community.

PERMANOVA results found two significantly different fish communities between low and high relief habitat on the bank crest, with low relief habitat having lower densities, greater biomass, lower richness of invertivores, and larger size fish than high relief habitat. While not tested against the shallow data due to differing sampling techniques, clear differences were also apparent between the bank crest communities and those found in coralline algae and deep reef habitat. Between coralline algae and deep reef habitat, while no analysis was conducted to evaluate the importance of these differences, coralline algae reef habitat had greater overall species richness, greater piscivore and herbivore richness, and lower invertivore richness. Limited surveys were conducted in mesophotic habitats during the course of this study; however, recent additions to the monitoring program provided a baseline for future analysis and monitoring of these habitats.

The fish community on the bank crest was significantly variable between years exhibiting significant interactions between year and habitat in many analyses thereby indicating that the community differed depending on both year and habitat. From 2012 to 2015, varying densities of invertivorous fishes and variable biomass of piscivorous fishes were the primary variables contributing to differences between all years, with no apparent trends. The size frequency of fish during this timeframe was also significantly different, with increasing abundance of individuals <5 cm and variable abundance of individuals 5-10 cm contributing the most to these differences between all years. Greater numbers of fish <5 cm indicate a potential increase in recruitment; however, as surveys were conducted in different months each year (from late May to early July), this finding may also reflect recruitment seasonality. Additionally, no significant difference between years was found in abundance biomass curves. Several significant year clusters between 1994 and 2015 based on community composition often contained only a few years, implying that the fish community between years was variable at Stetson Bank. However, the significant cluster found from 2011 to 2015 (excluding 2012) is not biologically significant as it is potentially due to sampling methods causing reduced species richness during that timeframe. While Pattengill (1998) documented that the use of novice surveys still provided meaningful information, they also noted a positive correlation with survey experience and the power of the data. As only novice surveys were conducted from 2011 to 2015, and we also found positive correlation with survey experience and species richness, we do not consider this cluster meaningful.

Trophic biomass was significantly different between both year and habitat on the bank crest. In high relief habitat from 2012 to 2015, an inverted biomass pyramid was found. In low relief habitat in 2013 and 2015, herbivore biomass outweighed piscivore biomass. However, in 2014, low relief habitat also exhibited an inverted biomass pyramid. In an inverted biomass pyramid, piscivore dominance is associated with minimal detrimental environmental impacts, particularly from fishing (Friedlander & DeMartini 2002, DeMartini et al. 2008, Knowlton & Jackson 2008, Sandin et al. 2008, Singh et al. 2012). Typically, inverted biomass pyramids are associated with healthy reef systems with high coral cover. However, coral cover at Stetson Bank throughout the 2012-2015 timeframe was low, as compared to other Caribbean reefs (Jackson et al. 2014), comprising less than 3% of the benthic cover. Despite the overall lack of coral cover, the high relief environment at Stetson Bank is composed of geologically and biologically complex habitat that provides structure for schooling behavior as well as potential refuges for prey fishes to shelter from predators, which is nearly absent in low relief habitat. The observed inverted biomass pyramid in the high relief habitat is potentially due to the availability of refuges, rapid turnover rates of prey items, slow growth rates of predators, and potential food subsidies from the surrounding pelagic environment (Odum et al. 1971, DeMartini et al. 2008, Wang et al. 2009). The lack of an inverted biomass pyramid in the low relief habitat during most years may be attributed to the lack of refuge available for prey, highlighting the importance of refuge (Hixon & Beets 1993). Overall, mean species richness was lower and mean biomass was greater, with greater variability, than other reef locations, including the East and West FGBs (Table 3.30). The topography at Stetson Bank may account for this higher observation of biomass, as fish have less complex structure to hide within in comparison to East and West FGBs.

Table 3.30. Mean biomass and richness of Caribbean and Gulf reefs, ± standard error. [1]Caldow et al. 2015, [2]Bauer et al. 2015b, [3]Bauer 2015, [4]Roberson et al. 2015, [5]Pittman et al. 2015, [6]Clark et al. 2015, [7]Bauer et al. 2015a, [8]Johnston et al. 2018.

Region	Mean Biomass ($g/100\ m^2$)	Mean Richness ($richness/100\ m^2$)
Puerto Rico[1,2,3]	3,830.25 ± 188.51	18.19 ± 0.19
U.S. Virgin Islands[4,5,6,7]	6,355.38 ± 172.60	20.70 ± 0.12
East and West Flower Garden Banks[8]	7,176.25 ± 857.88	18.90 ± 0.61
Stetson Bank	11,339.40 ± 2020.75	17.86 ± 0.40

Spatial analysis demonstrated the importance of the bank crest and coralline algae reefs to herbivorous fishes. Herbivores were almost completely lacking in deep reef habitat. This was supported by reduced macroalgal cover in deep reef habitat (Chapter 2). Deep reef habitat at Stetson Bank is not as valuable to herbivorous fishes although many piscivorous species were documented there, with the southwest corner of the habitat supporting overall high density and biomass.

On the bank crest from 2012 to 2015, grouper did not vary significantly between years in density or biomass. However, significant differences between habitat on the bank crest were due to the greater abundance of rock hind in high relief habitat. In deep reef habitat, additional grouper species and greater biomass of grouper were observed. Snapper on the bank crest exhibited similar trends in density and biomass as grouper; however, snapper density was also significantly different between years. Between habitats on the bank crest, gray snapper were more abundant in high relief and, between years, the variable density of gray snapper in 2012-2013 created significant differences. In deep reef habitat, one additional species of snapper was observed, and both the bank crest and deep reef habitats were found to support, in localized areas, high densities of snapper. No significant difference was found between years or bank crest habitats for grunts. Although the similar species were observed throughout the bank crest and deep reef habitat, the southwest portion this habitat harbored very high densities and biomass of grunts. Parrotfish varied between both year and habitat on the bank crest, with multiple species contributing to the differences. The parrotfish population on the bank crest was predominantly small individuals (<5 cm), with an abundance of greenblotch parrotfish. Parrotfish appeared to be restricted to the bank crest as no observations were recorded in deep reef habitat.

The angelfish community on the bank crest was significantly different between habitats primarily because the abundance of French angelfish contributed to greater total density within high relief habitat. A significant difference was also found between years among the bank crest community, with a reduction in French angelfish density and biomass in 2015 (although overall angelfish biomass has increased since 2012). In deep reef habitat, fewer species of angelfish were seen; however, they were sufficiently abundant to be considered an important component of the community. The sponge community on Caribbean reefs is primarily influenced by local environmental stressors as well as predation by fish and hawksbill sea turtles (Pawlik et al. 2013, Lorders et al. 2018, Pawlik et al. 2018). Many species of angelfish, including rock beauty and blue angelfish, are known spongivores that consume sponges as a major component of their diet

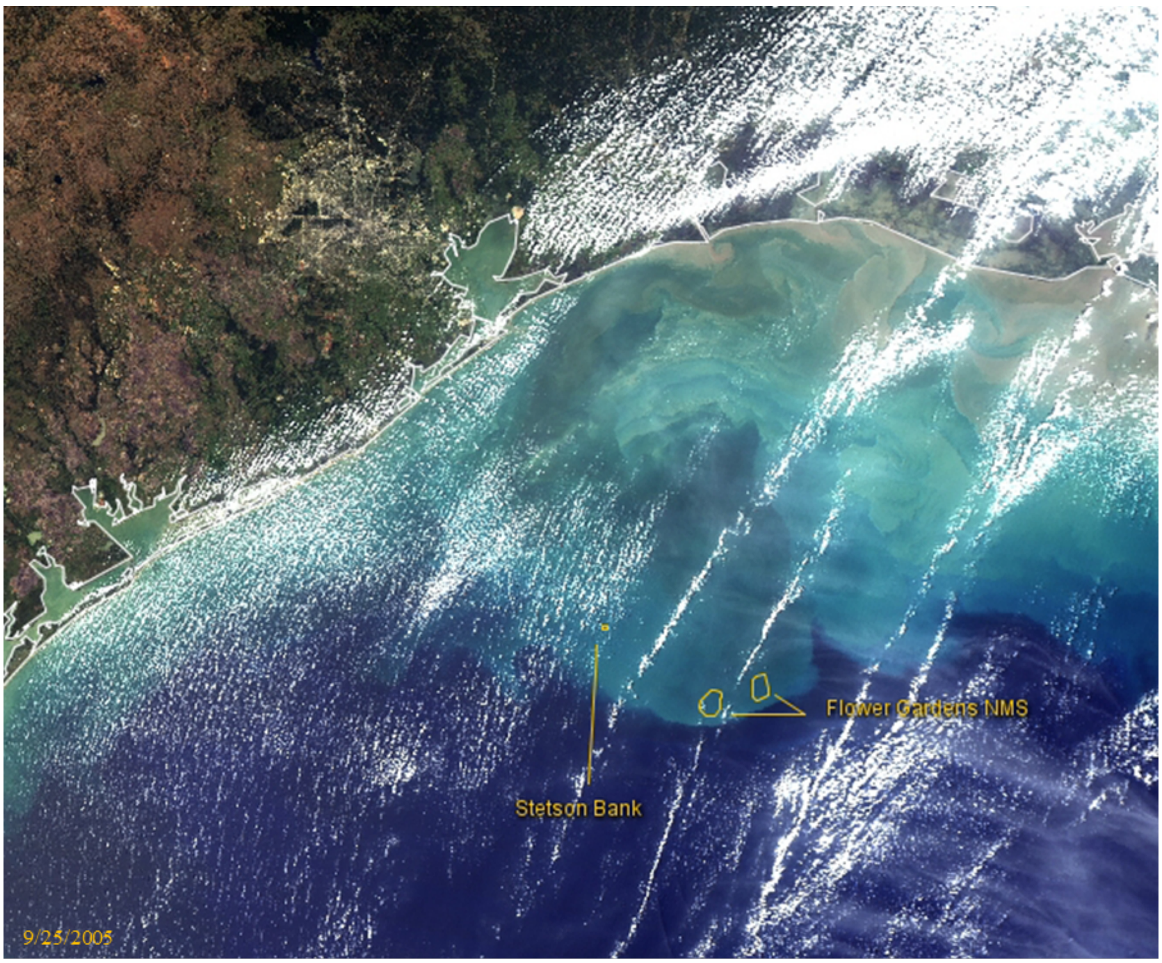
(Randall & Hartman 1968, Dunlap & Pawlik 1996). On the bank crest, rock beauty and blue angelfish were found to co-vary with total sponge cover, and blue angelfish were found to co-vary with the sponge *I. strobilina*, all exhibiting a significant and gradual decline over time. *I. strobilina* produces a deterrent to deter predation (Pawlik et al. 1995) and has been reported to have temporary negative effects when force fed to various species of angelfish (Hoppe 1988); however, Randall & Hartman (1968) documented *I. strobilina* in the gut content of wild queen angelfish, a close relative of blue angelfish. At Stetson Bank, *I. strobilina* was the predominant sponge species in repetitive photostations, following the decline of *C. nucula* in 2005-2006. Rock beauties, known spongivores, have also been documented in association with hydrocorals, perhaps for habitat (Lieske & Myers 1994), which, like sponge cover at Stetson Bank, has declined over time. These connections demonstrate predator-prey dynamics at Stetson Bank and their indirect effects.

Although lionfish have been reported at Stetson Bank by recreational scuba divers since 2011, very few lionfish were recorded in surveys during this study period. However, Johnston et al. (2016b) documented increasing sighting frequency from 2011 to 2013 with a subsequent decline in 2014, and increasing size and biomass from 2011 to 2014, on the bank crest. The cause for the decline in sighting frequency in 2014 is unknown. On the bank crest, lionfish occurrence is lower at Stetson Bank than East and West FGBs (Johnston et al. 2016b). While the cause of these lower numbers is likely complex, a contributing factor may be the abundance of moray eels found at Stetson Bank. In their native range, large moray eels (Muraenidae) not only prey on lionfish, but locations inhabited by these eels are actively avoided by lionfish (Bos et al. 2017). In deep reef habitat in 2015, lionfish were among the most frequently sighted species, despite low detection in bank crest surveys. Though survey methods differed, this large difference in sighting frequency suggests that lionfish are preferentially using deep reef habitat, potentially due to buffered thermal variations in deeper water and available refuge. Despite having a wide thermal tolerance, lionfish thermal preference is dependent on the typical environmental conditions (Barker 2015). These conditions, including water temperature variations and depth, are discussed in Chapter 4. Additionally, removal efforts on the bank crest have been successful at Stetson Bank; however, removals from habitats deeper than 33.5 m is challenging and has not been conducted at Stetson Bank to-date. This has allowed the establishment of communities in the mesophotic habitat as potential source populations for the bank crest. The invasion of this exotic species is of particular concern due to their voracious appetite, high fecundity, and apparent lack of predators. Additionally, the presence of lionfish has been documented to suppress recruitment of other fishes (Albins & Hixon 2008). On the bank crest, low recorded lionfish numbers, coupled with increasing abundance of small fishes (<5 cm) since their initial invasion, suggest that lionfish have had minimal influence on fish recruitment at Stetson Bank, but additional data are needed to assess their true impact to the fish community. In the mesophotic fish community, where lionfish sighting frequency is greater, there is limited historical data to provide baseline information on the fish community prior to the lionfish invasion making assessing impacts more challenging.

Overall Conclusions

The fish community at Stetson Bank is naturally variable, both within the various habitats found at the bank and between years. The community is diverse, composed of reef associated and pelagic species, including some commercially and recreationally valuable species. Despite having lower total species richness and lower species richness per sample than the East and West Flower Garden Banks, biomass at Stetson Bank is more variable but greater. The fish community as a whole exhibited variability between years and most families of interest revealed no trends over time. However, declines seen in the cover of major benthic components of the community at Stetson Bank were reflected in declines in species of fish that depend on those particular resources for food, thereby illustrating the trophic connections between the communities.

Chapter 4
LOCAL WATER QUALITY AND ENVIRONMENTAL CONDITIONS



Satellite image of the north western Gulf of Mexico following Hurricane Rita. Image: NASA/GSFC MODIS, processed by NOAA CoastWatch

Introduction

This chapter explores the environmental conditions at Stetson Bank during the study period, including the physical, biological, and chemical characteristics of the water column. The bank's location ~130 km offshore provides some separation from turbid, brackish, coastal waters; however, pockets of mixed coastal and oceanic waters have been documented reaching Stetson Bank annually between May and July, increasing turbidity and potentially conveying pollutants and particulates (Deslarzes & Lugo-Fernández 2007).

Anomalously high river discharge combined with ocean currents can convey coastal water towards the outer shelf and to the vicinity of Stetson Bank. Obvious major river outfalls including the Atchafalaya and Mississippi river basins, deliver on average upwards of 650,000 ft^3/s of water into the Gulf of Mexico. Due to the large volume of water flowing from the Mississippi/Atchafalaya river basins, draining over 40% of the contiguous United States, increased flow rates from these rivers can alter water conditions on the continental shelf, including the area of Stetson Bank (Morey et al. 2003, Bianchi et al. 2010). Additionally, the net effect of anomalously high discharge from smaller Texas Rivers may also transport coastally influenced water to the offshore environment.

The periodic impacts of tropical weather systems on the environmental conditions of reefs in the tropical biotope have been documented for decades (Woodley et al. 1981, Scoffin 1993, Harmelin-Vivien 1994, Aronson et al. 2001, Riegl 2007, among others). However, the impact of these systems on habitat greater than 20 m deep, like Stetson Bank, are not well documented. Tropical weather systems can impact a reef in a variety of ways, including mechanical damage from waves, currents, or projectiles, reduced water clarity from sediment resuspension and runoff, and stress from reduced salinity due to rain and runoff. However, these events can also positively influence reefs through storm-induced water cooling and aiding in larvae dispersal (Lugo-Fernández et al. 2001, Manzello et al. 2007, Lugo-Fernandez & Gravois 2010).

Various parameters to evaluate the environmental conditions were collected throughout the study period, including a variety of water quality data. However, continuous datasets have been developed in recent years whereas many of these parameters were collected periodically (Appendix A: Table A.1). Between 1993-1995 and 2002-2005, water temperature on the crest of Stetson Bank (24 m) was collected sporadically. Since 2006, a continuous record of water temperature at 24 m exists (with the exception of periods in 2008). Other water quality parameters measured sporadically at this station include salinity, pH, dissolved oxygen concentration (DO), and turbidity. Starting in 2014, a 30 m water temperature station was established, followed by a 40 m water temperature station in 2015.

Water sampling for nutrient loading measurements have been collected on a quarterly basis at three depths in the water column since 2009 (bank crest, mid water, and surface), with ocean carbonate measurements added in 2013.

Methods

Field Methods

Water Quality

Water temperature on the crest of Stetson Bank (24 m), was collected sporadically between 1993-1995, using a HOBO® thermistor, and 2002-2008, using a YSI multiparameter sonde. Since 2009, a continuous record of water temperature at 24 m was collected, using a Sea-Bird® Electronics Inc. 37 MicroCAT® logger until 2015 (Appendix: Table A.1). The logger was installed on a large railroad wheel in the midsection of the bank crest (Figure 4.1). The instrument recorded temperature and salinity hourly throughout the year, with quarterly downloading and maintenance. Maintenance and factory service of the instrument were performed annually. In November 2015, a Sea-Bird® Electronics, 16plus V2 CTD was deployed to replace the MicroCAT® 37, equipped with a WET Labs ECO NTUS turbidity meter.

Figure 4.1. Sea-Bird® Electronics Inc. 37 MicroCAT® logger mounted to a railroad wheel at Stetson Bank. Photo: G.P. Schmahl/NOAA

Onset® Computer Corporation HOBO® Pro v2 U22-001 thermographs were used as a backup to the Sea-Bird instruments and recorded temperature on an hourly basis from 2009 forward. In October 2015, a HOBO® thermograph was deployed at a 30 m station, located on the northerly edge of the bank crest, to record temperature hourly. In June 2015, another HOBO® thermograph was installed at 40 m, also along the northern edge of the bank crest, to record temperature hourly. The loggers were downloaded and maintained on a quarterly basis. The

HOBO® thermistors were attached either directly to the primary SEABIRD instrument at the 24 m station or to eyebolts embedded in the substrate at the 30 m and 40 m stations. In 2015, two loggers were deployed in deep reef habitat at 44 m and 54 m using an acoustic release system. However, the acoustic release system failed and the instruments have not been recovered to date.

Water samples for nutrient and ocean carbonate analyses were collected each quarter starting in 2009 and 2013, respectively. Water samples were initially collected using a manually triggered handheld Niskin bottle, lowered on a measured line. Starting in 2011, samples were collected using a sampling carousel equipped with a Sea-Bird® Electronics 19plus V2 CTD and six OceanTest® Corporation 2.5 liter Niskin bottles, with bottles activated at specific depths (Figure 4.2). Each quarter, three nutrient samples, with one replicate for each depth, were collected near the seafloor (approximately 20 m depth), mid-water (10 m depth) and near the surface (1 m depth). Ocean carbonate samples were collected at identical depth intervals, with only one replicate collected with the surface (1 m) sample.

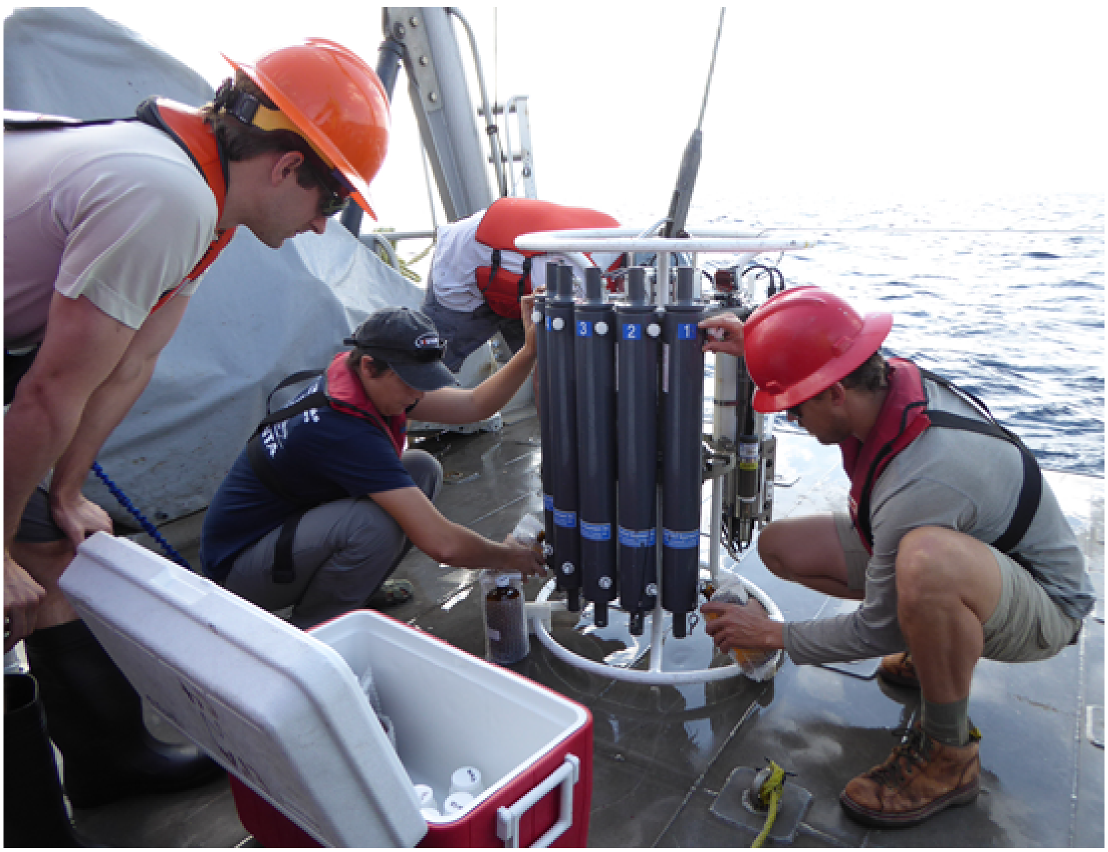


Figure 4.2. A carousel is used to collect water samples and vertical profiles. Photo: G.P. Schmahl/NOAA

Once samples were collected, subsamples were transferred as follows: chlorophyll-*a* subsamples were transferred to 1000 ml brown glass containers with no preservatives; reactive soluble phosphorous subsamples were placed in 250 ml white plastic bottles with no preservatives; and ammonia, nitrate, nitrite, and total nitrogen subsamples were transferred in 1000 ml white plastic bottles and preserved with sulfuric acid. Within minutes of sampling, labeled sample containers were stored on ice at 4 °C and a chain of custody was initiated for processing at an EPA-certified laboratory. The samples were transported and delivered to A&B Laboratories in Houston, TX within twenty-four hours of collection.

Water samples for ocean carbonate measurements were processed following methods requested by the Carbon Cycle Laboratory (CCL) at TAMU-CC. Samples in 2013 were transferred to Pyrex 250 ml borosilicate bottles with polypropylene caps. From 2014 to 2015, samples were collected in ground glass bottles sealed with Apiezon® grease and a rubber band. Bottles were filled using a 30 cm plastic tube that connected from the spout of the Niskin. Bottles were rinsed three times using the sample water, filled carefully to reduce bubble formation, and overflowed by at least 200ml. 100 µl of $HgCl_2$ was added to each bottle before inverting vigorously. Samples were then stored at 4°C and sent to CCL at TAMU-CC, in Corpus Christi, Texas. Each sample was analyzed for pH, alkalinity, dissolved inorganic carbon (DIC), $\Omega_{aragonite}$, and pCO_2.

Data Processing

Tropical weather systems, sea surface temperature (SST), significant wave height, and degree heating weeks (DHW) were obtained from external sources and processed.

While each tropical weather system varies extensively in reach and impacts, the following assumptions were made to focus the selection of storm systems to examine in this report. The average storm has a radius of 3° latitude (Merrill 1984) and, following reasoning in Lugo-Fernandez & Gravois (2010) and Debose et al. (2013), storms that passed within 200 km of Stetson Bank had the potential to impact the bank. This selection criteria represents a continuation of the data presented in DeBose et al. (2013). Type, track, and maximum wind speed of hurricanes and tropical storms that passed within 200 km of Stetson Bank were obtained from NOAA's Office for Coastal Management (NOAA Office of Coastal Management 2017) for 1993-2015. Storm types were classified using the Saffir-Simpson scale.

SST, salinity, wave height (WVHT), and average waver period data were obtained from the National Data Buoy Center (NDBC) Station 42019, located 107 km west-southwest of Stetson Bank, in a water depth of 82.3 m (Figure 4.3). Data were collected hourly and averaged daily. Anomaly calculations were conducted by subtracting each daily value from the daily average value for all years. Average wave period was converted to wavelength using the deep-water wave-relation equation (Dean and Dalrymple 1991), where L=wavelength (m) and T=wave period (sec):

$$L_o = \frac{9.8}{2\pi} T^2$$

Figure 4.3. Location of NDBC 42019 and examined river systems in relation to Stetson Bank. Image: NOAA

DHW data were obtained from NOAA's Coral Reef Watch Program from 2001 to 2015 (NOAA Coral Reef Watch 2017). This data provides a measurement of the accumulated thermal stress based on sea surface temperatures. One DHW is equal to one week of SSTs 1ºC above the expected summertime maximum (Wellington et al. 2001). Data were obtained at 50 km resolution.

Major Texas and Louisiana River basins draining into the western and northwestern GOM with available discharge data were selected (Table 4.1). Discharge, in ft^3/s, of the lower Atchafalaya River USGS station number 07381600, Brazos River USGS station number 08116650, Colorado River USGS station number 08162500, Neches River USGS station number 08041000, Nueces River USGS station number 08211500, Mississippi River USACE station number 01100Q (1/1/199-10/28/2008), and USGS station number 07374525 (10/29/2008-12/31/2015), Sabine River USGS station number 08030500, and Trinity River USGS station number 08067000, were obtained from USGS National Water Information System (USGS 2018) and USACE River Gauge (USACE 2018). While this report examines January 1993 through December 2015, data from historical river gauges typically covered only part of this timeframe. Daily data for the timeframe available were obtained and converted to percent discharge based on the maximum average discharge or anomaly for each river system.

Table 4.1. Major Texas and Louisiana River basins draining into the western and northwestern Gulf of Mexico with discharge data. *Kammerer 1987 and [ŧ]TWDB 2019.

River	River Length (miles)	Total Drainage Basin Area (square miles)	Annual Discharge (acre-feet/year)
Atchafalaya*	1,420	951,000	41,990,145
Mississippi*	2,340	1,150,000	429,313,000
Brazos[ŧ]	840	45,573	6,074,000
Colorado[ŧ]	865	42,318	1,904,000
Neches[ŧ]	476	9,937	4,323,000
Nueces[ŧ]	315	16,700	539,700
Sabine[ŧ]	360	9,756	5,864,000
Trinity[ŧ]	550	17,913	5,727,000

The diffuse attenuation coefficient for downwelling irradiance at 490 nm (Kd490) in m^{-1} was obtained from the Moderate Resolution Imaging Spectroradiometer (MODIS) aboard NASA's Earth Observing System (EOS) Aqua satellite, using NASA's KD2M algorithm (NASA 2017). This coefficient indicates light (at the specified wavelength) attenuation through the water column and is directly related to water clarity and the presence of particles in the water column. Higher coefficients mean lesser attenuation depths and lower water clarity. Data were obtained in 2 km resolution on a daily basis and averaged to weekly values, resulting in 52 data points a year. One large attenuation coefficient outlier (6/7/2010 = 0.74 m^{-1}) was removed from analysis.

Temperature data from SeaBird and HOBO loggers at each station were averaged by day. For temperature data, a historical average of data from the previous 10 years (2006-2015) was used for anomaly calculations. Salinity and turbidity from the SeaBird instrument were also averaged by day. DO, pH, and turbidity data from 2004 to 2007 exhibited problems with drift and were therefore excluded from analyses.

Chlorophyll-a and nutrient analyses results were obtained quarterly from A&B Laboratories and compiled. Ocean carbonate analyses results were compiled and received as an annual report from the CCL at TAMU-CC.

Statistical Analysis

SST from NDBC Station 42019, bank crest temperature data from the 24 m station on Stetson Bank, and Atchafalaya River discharge rates from 2003 to 2015 were averaged by month and tested for long-term trends using a seasonal Mann-Kendall trend test and decomposed into trend, seasonality, and remainder using Season Trend Decomposition using Loess (STL; Cleveland et al. 1990). To minimize data gaps, data were subset as necessary: SST data from 1993 to 2015; and bank crest (24 m) temperature data from 2003-2015. Missing data were excluded from the STL analysis. Data were averaged by week for use in correlation tests. Kendall rank correlation was used to test for relations between salinity and Atchafalaya River discharge from 2005 to 2015, as salinity was not normally distributed, and Pearson's correlation was used to test for relationships between Kd490 and Atchafalaya River discharge from 2003 to 2015, as data were normally distributed. Analyses were performed in R version 3.2.0 (R Development Core Team 2015).

Results

Tropical Weather Systems

A total of twelve storms were documented within 200 km of Stetson Bank between 1993 and 2015 (Table 4.2, Figure 4.4). Annually, there was a 52% incidence of a tropical weather system coming within 200 km of the bank and the majority of storms (42%) occurred in the area during September (Figure 4.5). The majority of storms (58%) were classified as tropical storms on the Saffir-Simpson scale.

Table 4.2. Tropical weather systems that passed within 200 km of Stetson Bank between 1993 and 2015.

Name	Date	Max. Saffir-Simpson Scale within 200km of Stetson Bank	Max. Wind Speed within 200 km of Stetson Bank (mph)	Passed with 200 km of NDBC 42019
Dean	Jul-1995	Tropical Storm	45	x
Allison	Jun-2001	Tropical Storm	60	x
Bertha	Aug-2002	Tropical Depression	30	x
Fay	Sep-2002	Tropical Storm	60	x
Claudette	Jul-2003	Cat. 1	85	x
Grace	Aug-2003	Tropical Storm	40	x
Ivan	Sep-2004	Tropical Storm	60	
Rita	Sep-2005	Cat. 3	125	
Humberto	Sep-2007	Cat. 1	90	x
Edouard	Aug-2008	Tropical Storm	65	
Ike	Sep-2008	Cat. 2	110	x
Bill	Jun-2015	Tropical Storm	55	x

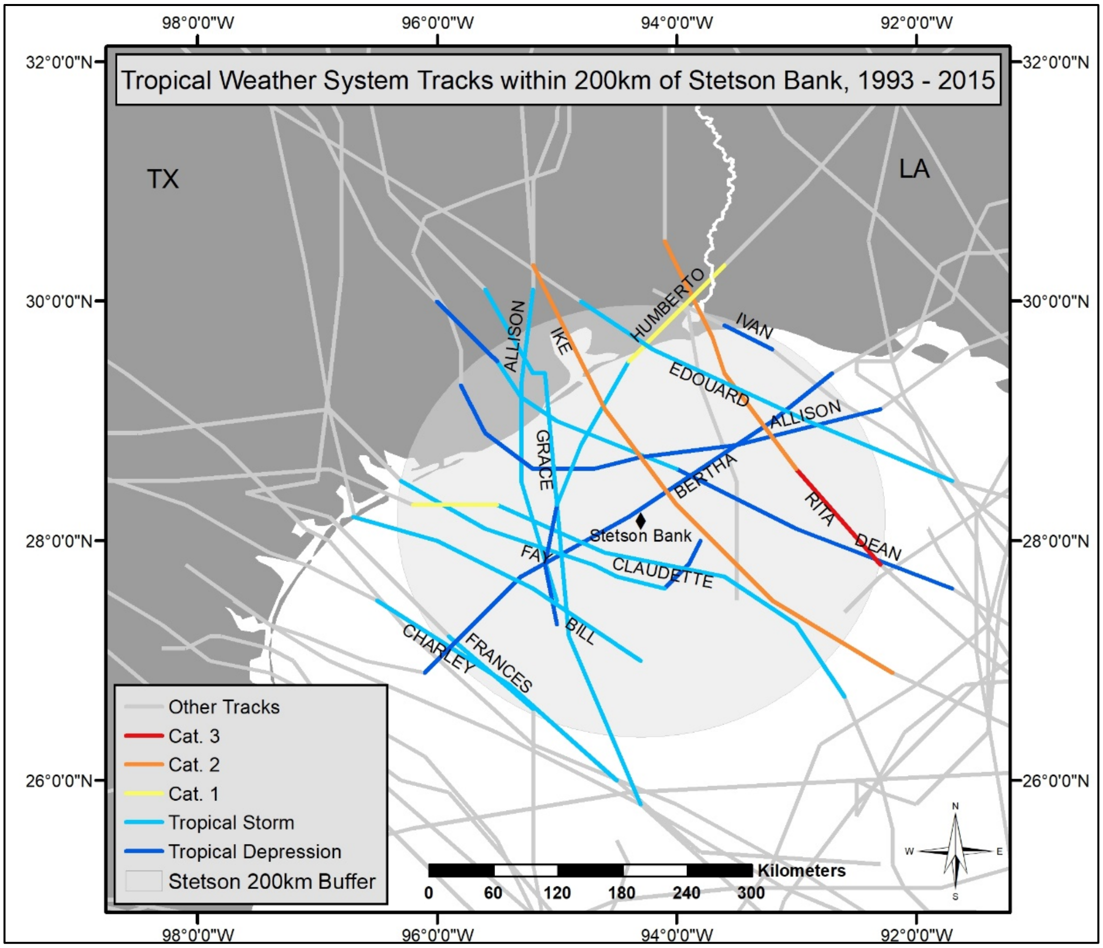


Figure 4.4. Map of tropical weather systems that passed within 200 km of Stetson Bank between 1993 and 2015. Color denotes storm classification based on the Saffir-Simpson scale. Image: NOAA

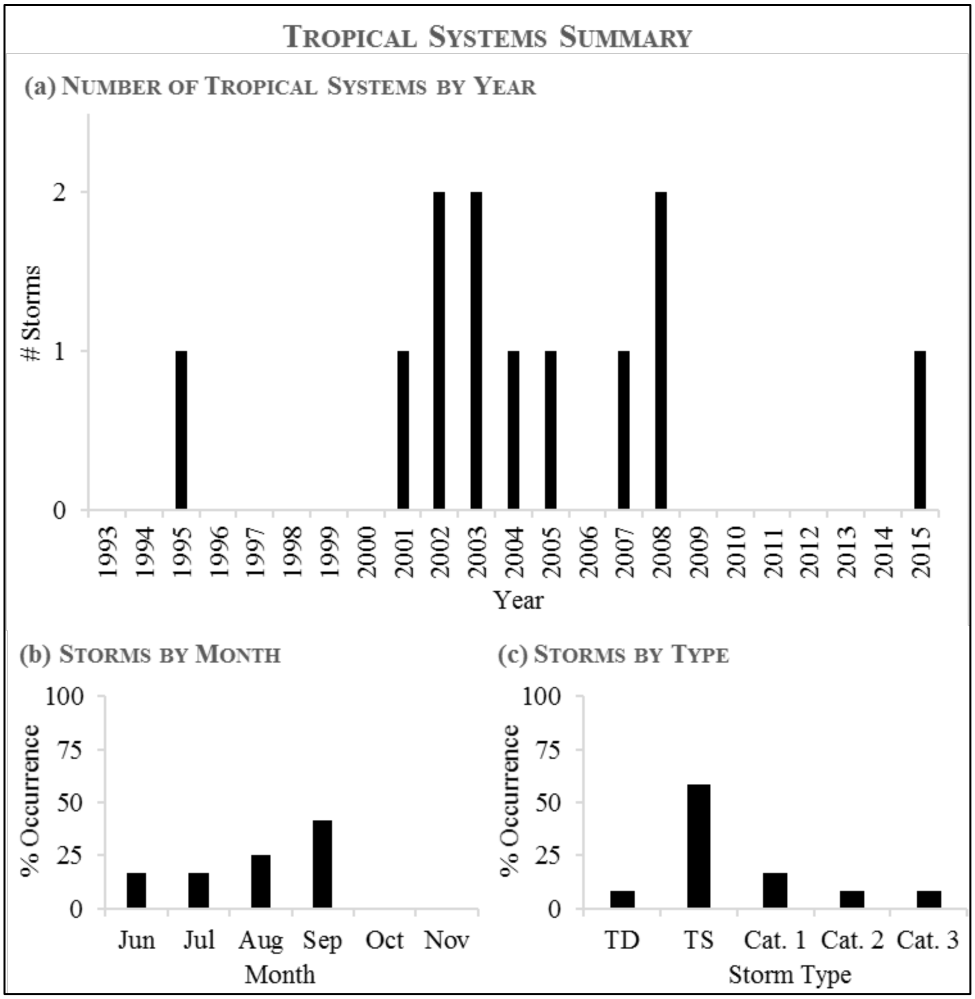


Figure 4.5. Tropical weather systems summary data. A shows the number of storms by year, B shows the percent occurrence of all storms by month during the hurricane season, and C shows the percent occurrence of all storms by storm type.

Wave Impact

Wave height anomalies (WVHTA) and wavelength were obtained to explore wave energy impacts. Wave height was average by day over the entire data set and subtracted from the daily wave height, highlighting days with higher or lower than average wave action. Overall, the maximum WVHTA varied by year, and were not consistently tied to years when tropical weather systems passed within 200 km of Stetson Bank (Figure 4.6). However, the two maximum WVHTAs were seen in 2005 (4.70 m) and in 2008 (5.12 m), when Hurricane Rita (category 3, 2005) and Hurricane Ike (category 2, 2008) passed within 200 km of Stetson Bank. Average

WVHTA were close to zero but positive, indicating greater than average wave heights, for 70% of years, and negative, indicating lower than average wave heights, for 30% of years.

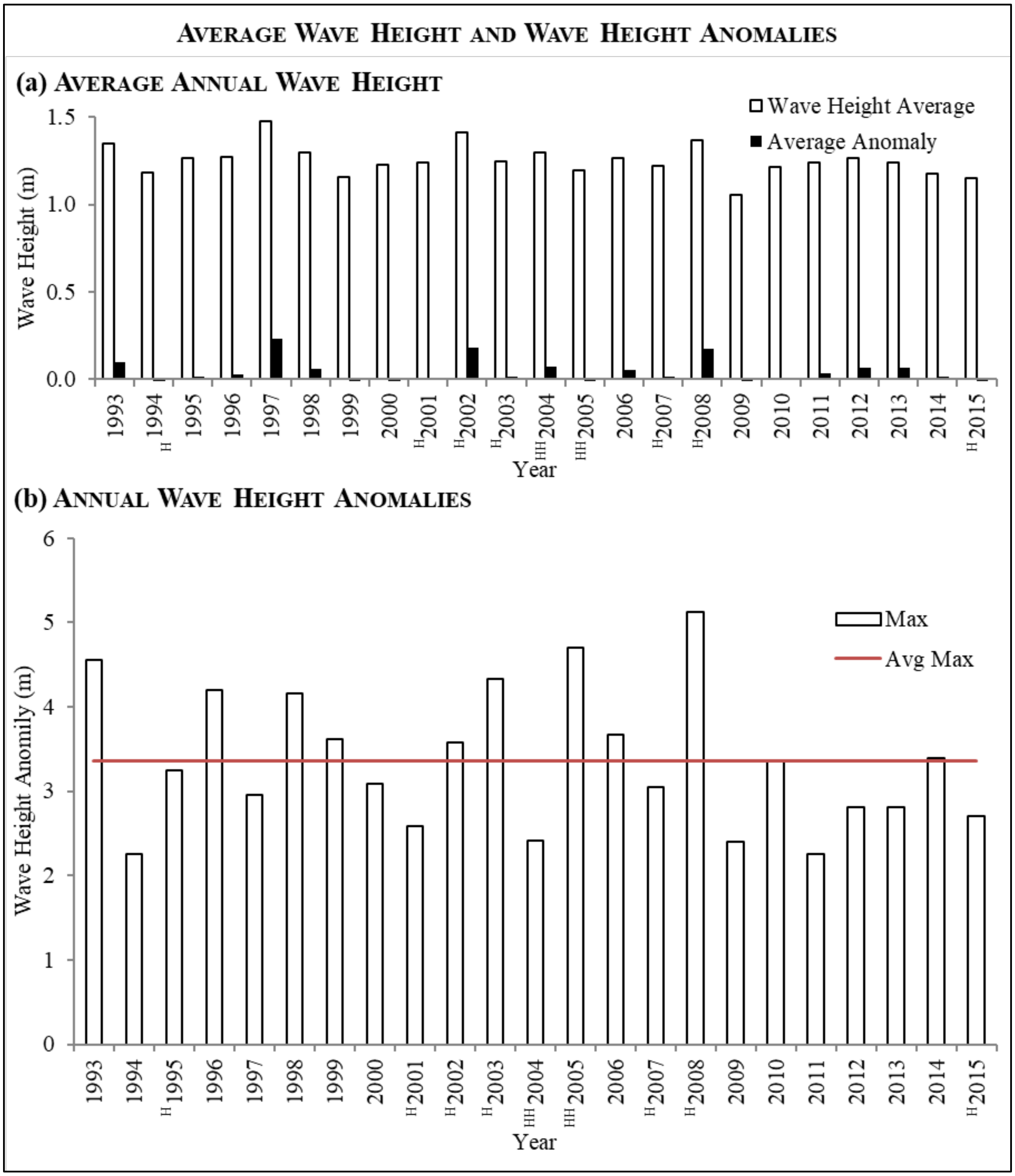


Figure 4.6. Annual wave height anomalies from NDBC Station 42019. Years with H denote years where tropical weather systems passed with 200 km of Stetson Bank and within 200km of NDBC 42019 (located 107 km to the west-southwest of Stetson Bank), and years with HH denotes years where tropical weather systems passed with 200 km of Stetson Bank but not within 200 km of NDBC Station 42019.

Wavelength is indicative of wave impact at depth, where a wavelength of >40 m has the potential energy to impact the crest of Stetson Bank at 20 m. Maximum daily wavelength data indicated that every year there was sufficient wave energy to impact the bank crest, with the maximum wave length recorded on 9/23/2005 in association with Hurricane Rita (Figure 4.7). When wavelength was averaged by year, 2008 had the greatest mean wavelength (38.9m).

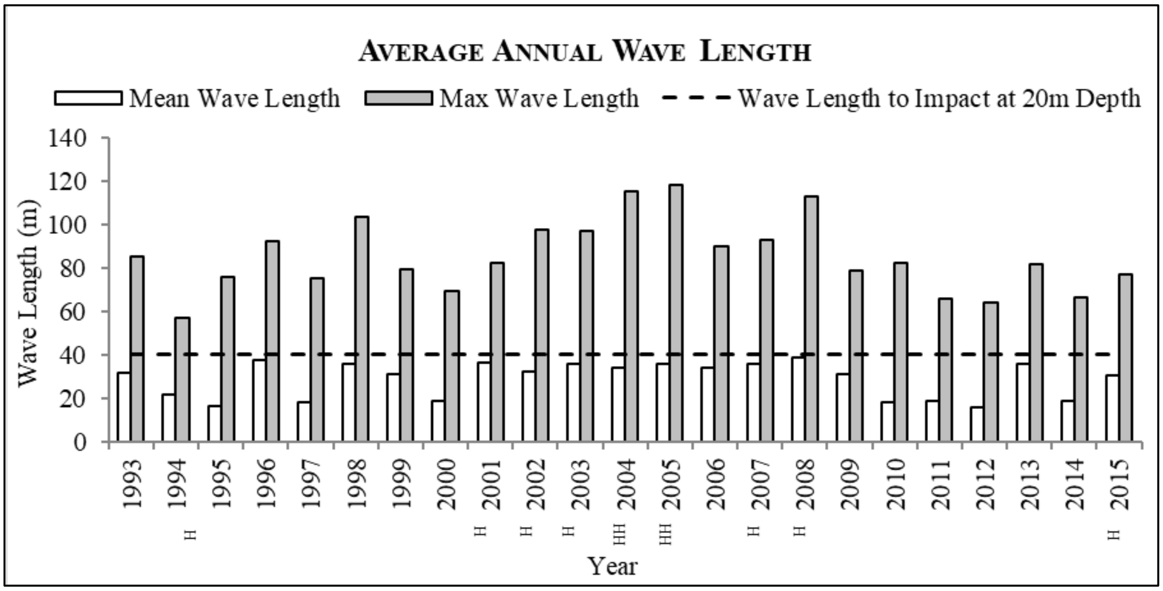


Figure 4.7. Annual wavelength from NDBC Station 42019. Years with H denote years where tropical weather systems passed with 200 km of Stetson Bank and within 200km of NDBC 42019 (located 107 km to the west-southwest of Stetson Bank), and years with HH denotes years where tropical weather systems passed with 200 km of Stetson Bank but not within 200 km of NDBC Station 42019.

Temperature

Most years observed DHWs of <1 ºC-week annually. The maximum observed was 10.5 ºC-week in 2010, followed by 6 ºC-week in 2005. Bleaching-level thermal stress events are defined as DHW ≥ 4 ºC-weeks, where DHW of ≥ 4 ºC-weeks have resulted in ecologically significant bleaching and 8 ºC-weeks have resulted in significant coral mortality (Eakin et al. 2010, Heron et al. 2016) (Figure 4.8).

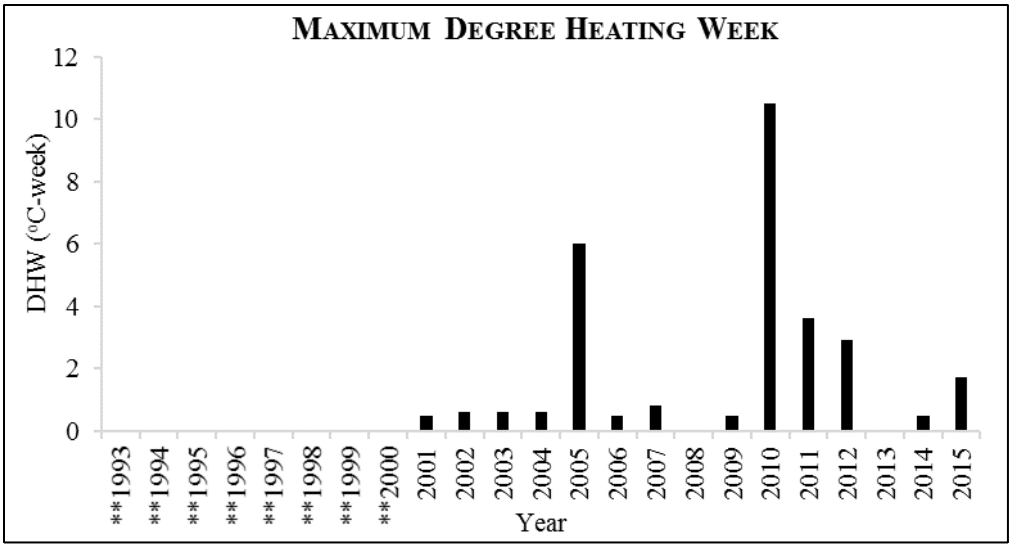


Figure 4.8. Degree Heating Weeks in the 50 km area of Stetson Bank. Years with asterisks denote no data.

Mean temperatures from the NDBC Station 42019 and the bank crest station at 24 m showed similar average data throughout the year (Figure 4.9 and Figure 4.10). According to NDBC Station 42019 data, the two maximum SST anomalies were found in 2014 (+8.54 ºC higher than the average daily temperature) and in 2005 (+8.07 ºC higher than the average daily temperature), while minimum anomalies occurred in 2010 (-11.49 ºC lower than average daily temperature) and 2004 (-9.34 ºC lower than average daily temperature). A significant increasing trend in SST was found from 1993-2015 (τ=0.138, p-value=0.008). Bank crest data at 24 m had a reduced range of thermal anomalies, meaning less deviation from the average daily temperature value, than SST, which exhibited maximum anomalies in 2006 (+3.20 ºC) and 2012 (+2.72 ºC), and minimum anomalies in 2005 (-5.39 ºC) and 2011 (-4.90 ºC). Although no significant trend was found in the shorter duration bank crest temperature data at 24 m (2003-2015), the trend line was positive (τ=0.04, p-value=0.828).

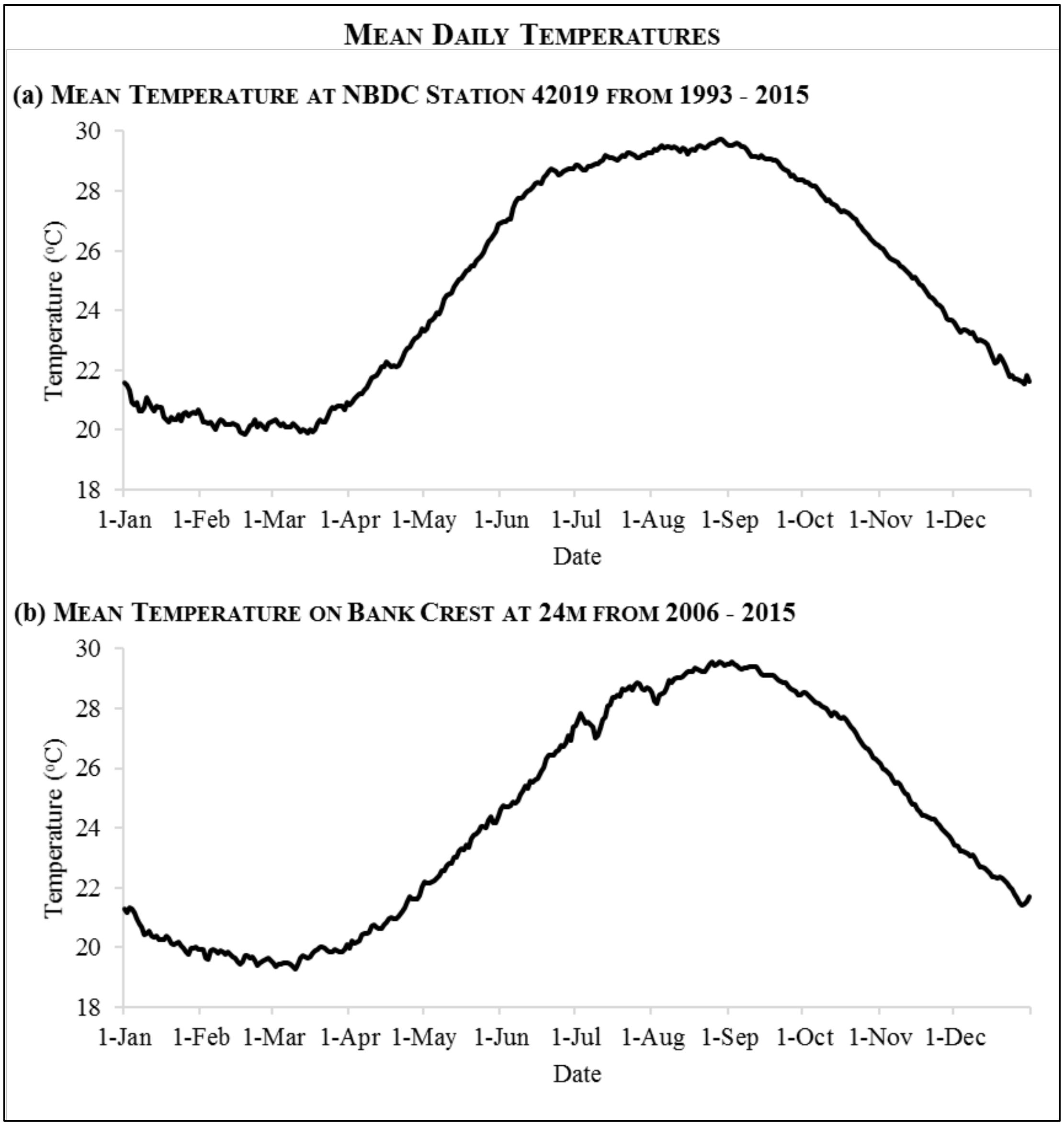


Figure 4.9. Mean temperatures. A: NDBC Station 42019 from 1993 to 2015 and B: Stetson Bank crest at 24 m from 2006 to 2015.

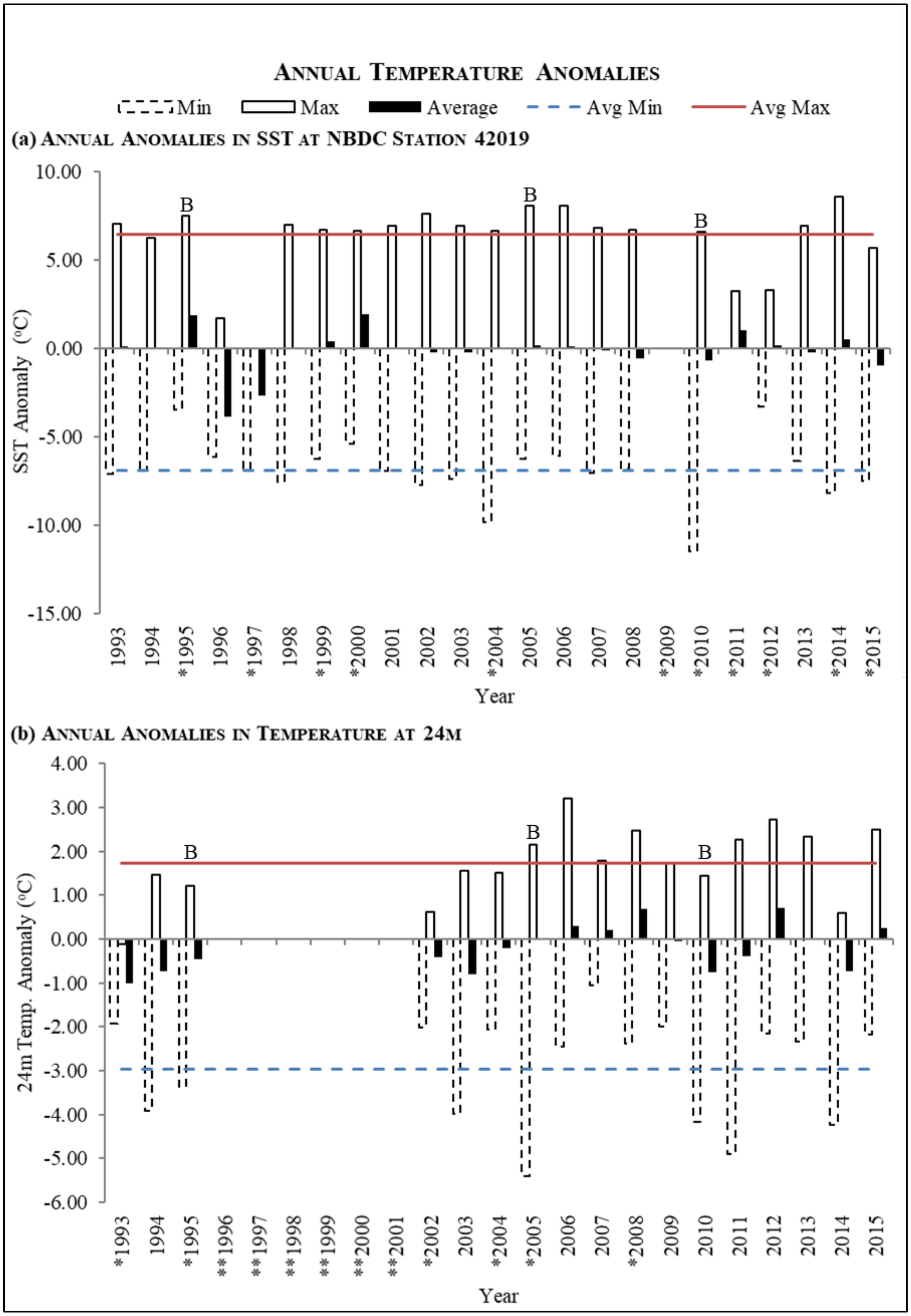


Figure 4.10. Annual temperature anomalies from 1993 to 2015. A shows SST data from the NDBC Station 42019 and B shows temperature data from the 24 m station on the bank crest at Stetson Bank. Years with a single asterisk (*) denote incomplete data and years with double asterisks (**) denote no data. B denotes years where bleaching was documented at FGBNMS (FGBNMS 2016).

According to 2015 data alone, SST was on average 1.12 ºC warmer than the bank crest (24 m). However, in comparison to 24 m, 30 m was an average of 0.57 ºC cooler and 40 m was an average of 1.83 ºC cooler. Thermal variability also declined with increased station depth (Figure 4.11).

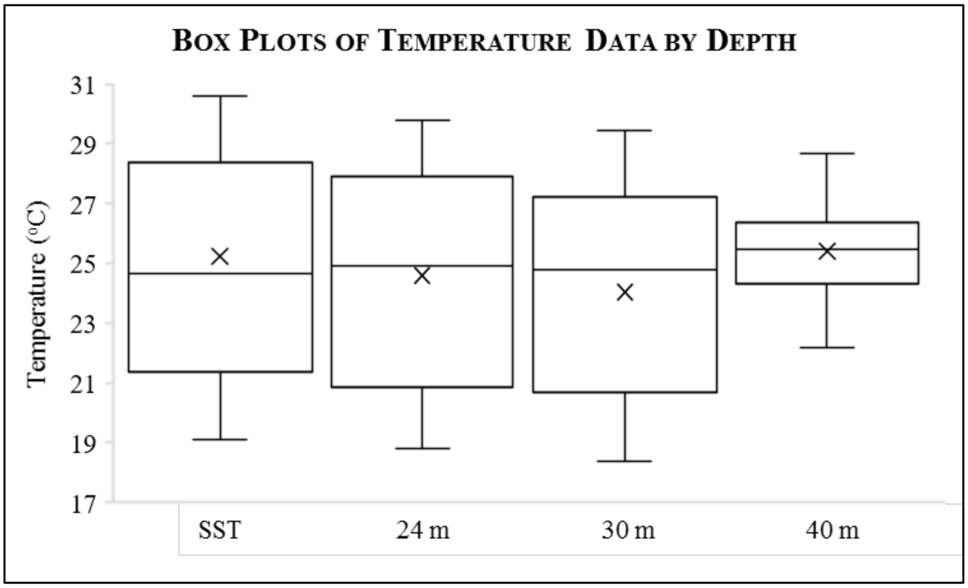


Figure 4.11. Box plot of 2015 temperature data by station depth. The 40m station was installed on 6/25/2015, and therefore presents an incomplete dataset of 2015.

For two of the three largest tropical weather systems, temperature data from the surface (from NDBC 42019) and at 24 m (at the bank crest) showed that water temperature temporarily declined following the passage of the storms (Figure 4.12). Limited water temperature data is available for the passage of Hurricane Ike in 2008, as the hurricane damaged the data buoy and the bank crest temperature instrument at Stetson Bank was inoperative during the month of September.

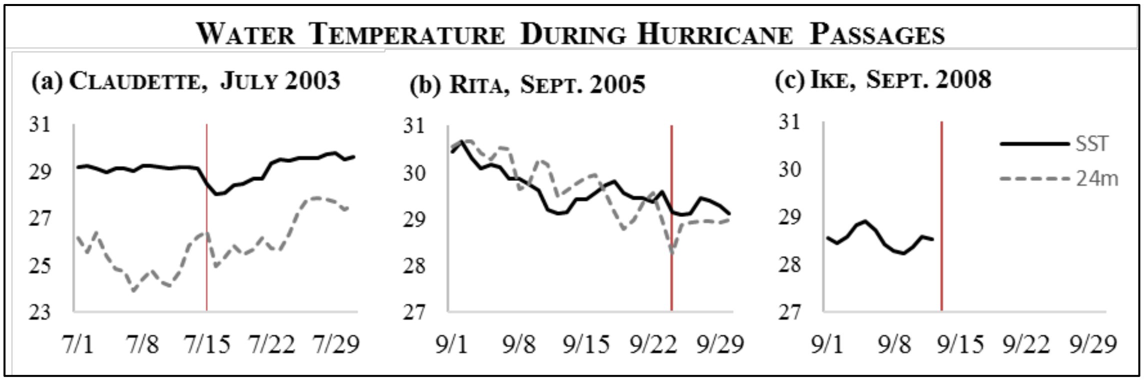


Figure 4.12. SST data from NDBC Station 42019 during the passage of Hurricanes. Red lines denote the date of hurricane passage. A shows data for the month of July, 2003, when hurricane Claudette passed on 7/15; B shows data for the month of September 2005, when hurricane Rita passed on 9/24; and C shows data for the month of September, 2008, when Hurricane Ike passed on 9/13.

Salinity

Averaged data demonstrate lower salinity, with wider variation, from late March through mid-August, where the lowest average salinity was 34.82 psu. The highest average salinity was 36.52 psu. The two minimum salinity anomalies, where salinity was unusually low, were documented in 2005 (-8.39 psu) and 2006 (-5.05 psu) (Figure 4.13).

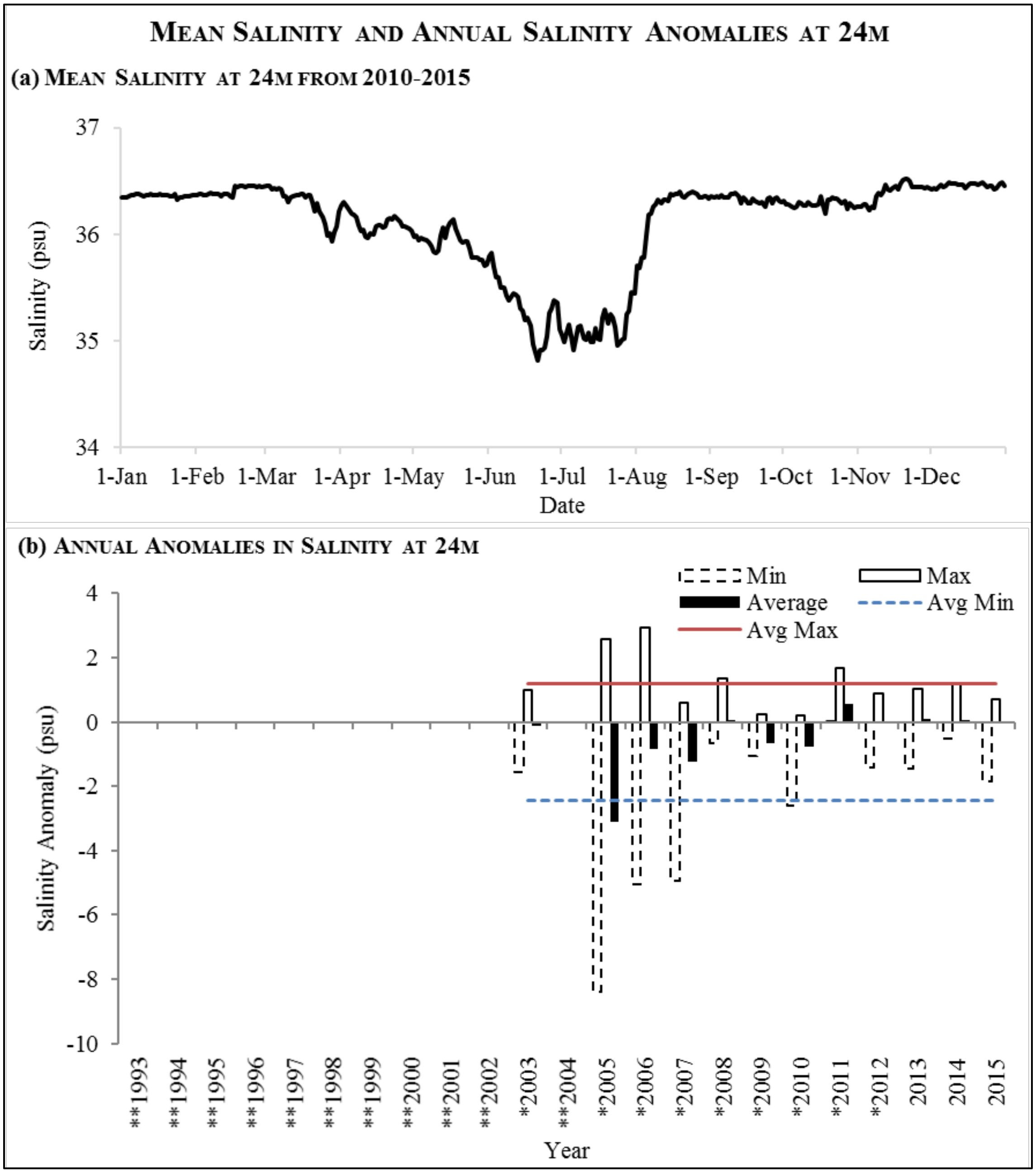


Figure 4.13. Mean salinity and annual anomalies. A: Mean salinity measurements from 2010 to 2015 from the 24 m station on the bank crest. B: annual salinity anomalies from the 24 m station on the bank crest. Years with a single asterisk (*) denote incomplete data and years with double asterisks (**) denote no data.

River Discharge

Average discharge for the Atchafalaya and Mississippi rivers peaked between April and June, followed by a gradual decline in discharge through September. However, major Texas rivers showed different trends in peak flow, exhibiting high discharge pulses during the same time period that the Atchafalaya and Mississippi rivers exhibited decreased average discharge (Figure

4.14). Annual discharge anomalies of each river system indicate cyclic wet and dry years but do not reveal similar trends between these river systems. However, in 2015, all rivers experienced higher than average anomalous discharge that resulted in increased average discharge for the entire year (Figure 4.15).

The Atchafalaya River showed no significant trend in river discharge over time (τ=0.01, p-value=0.914) and no significant correlation between Atchafalaya River discharge and bank crest salinity were found (τ=0.02, p-value=0.578).

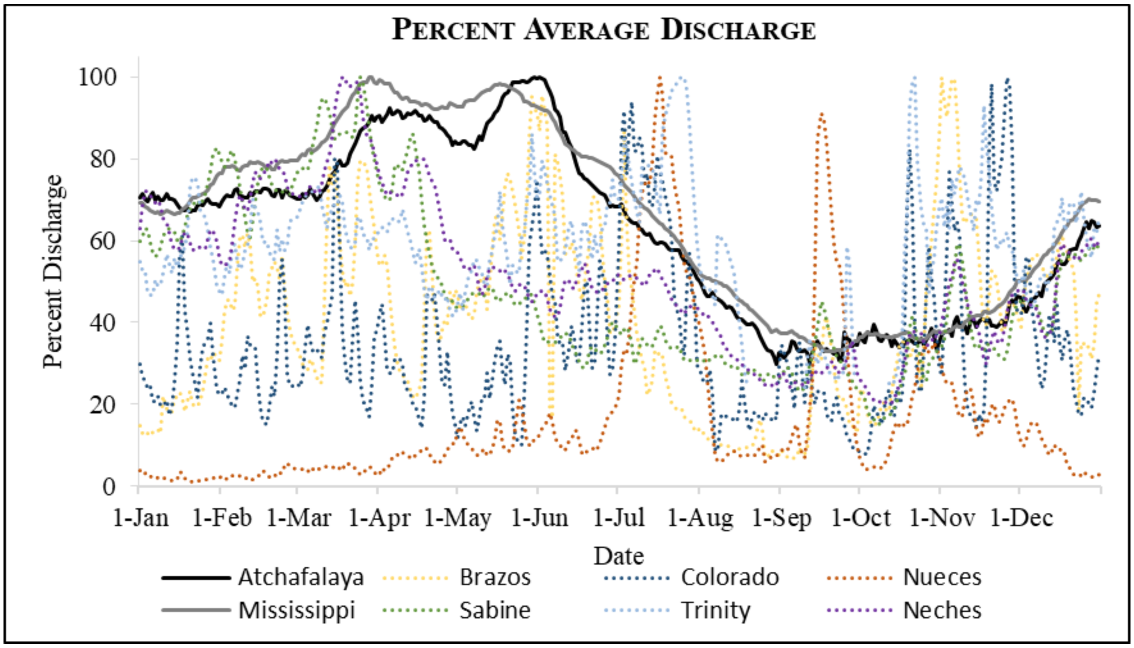


Figure 4.14. Mean percent discharge of major river systems. Solid lines represent major US rivers and dashed lines represent major Texas rivers.

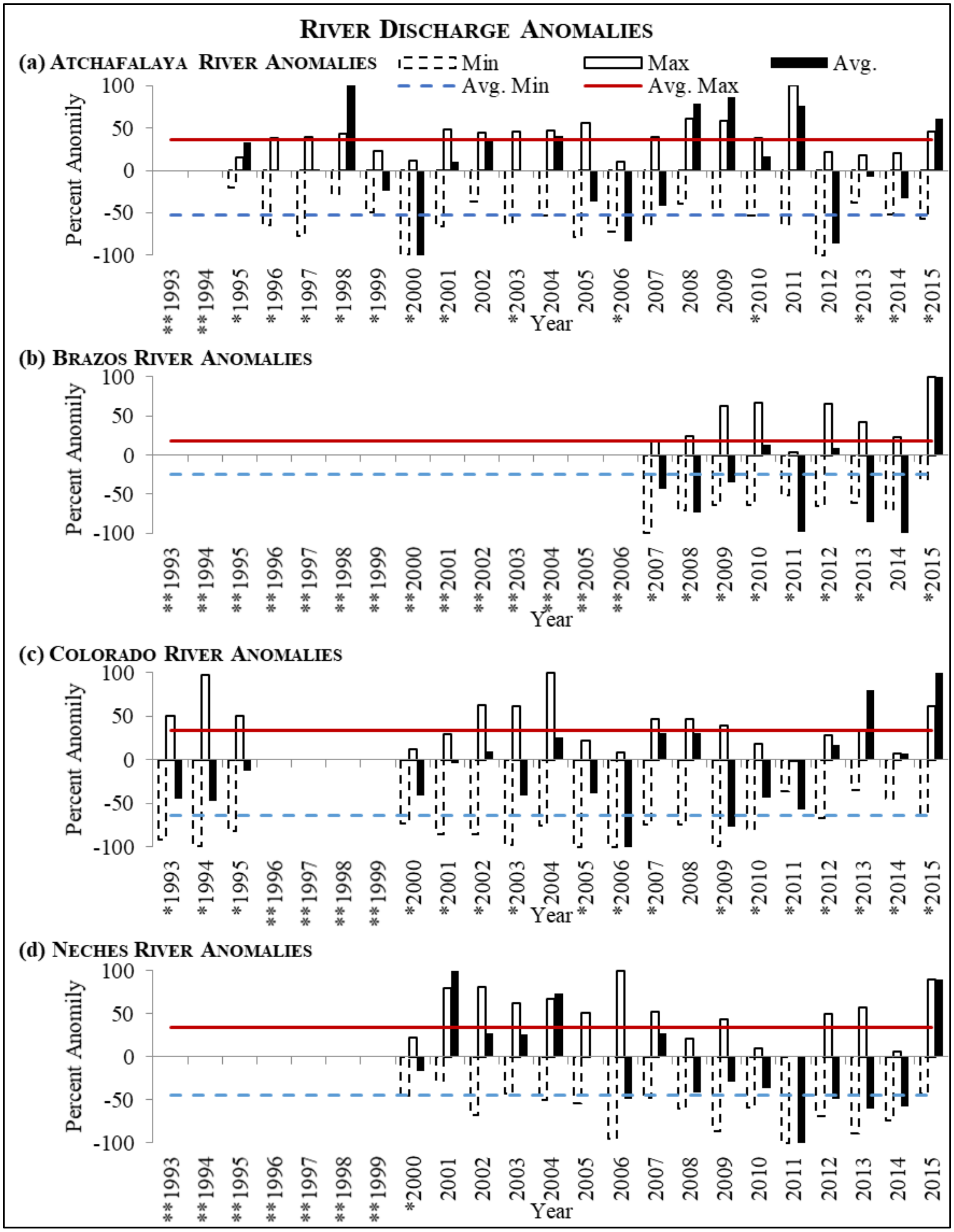


Figure 4.15. River discharge anomalies. Percent discharge anomaly presents for (a) Atchafalaya, (b) Brazos, (c) Colorado, (d) Neches, (e) Nueces, (f) Mississippi, (g) Sabine, and (h) Trinity Rivers.

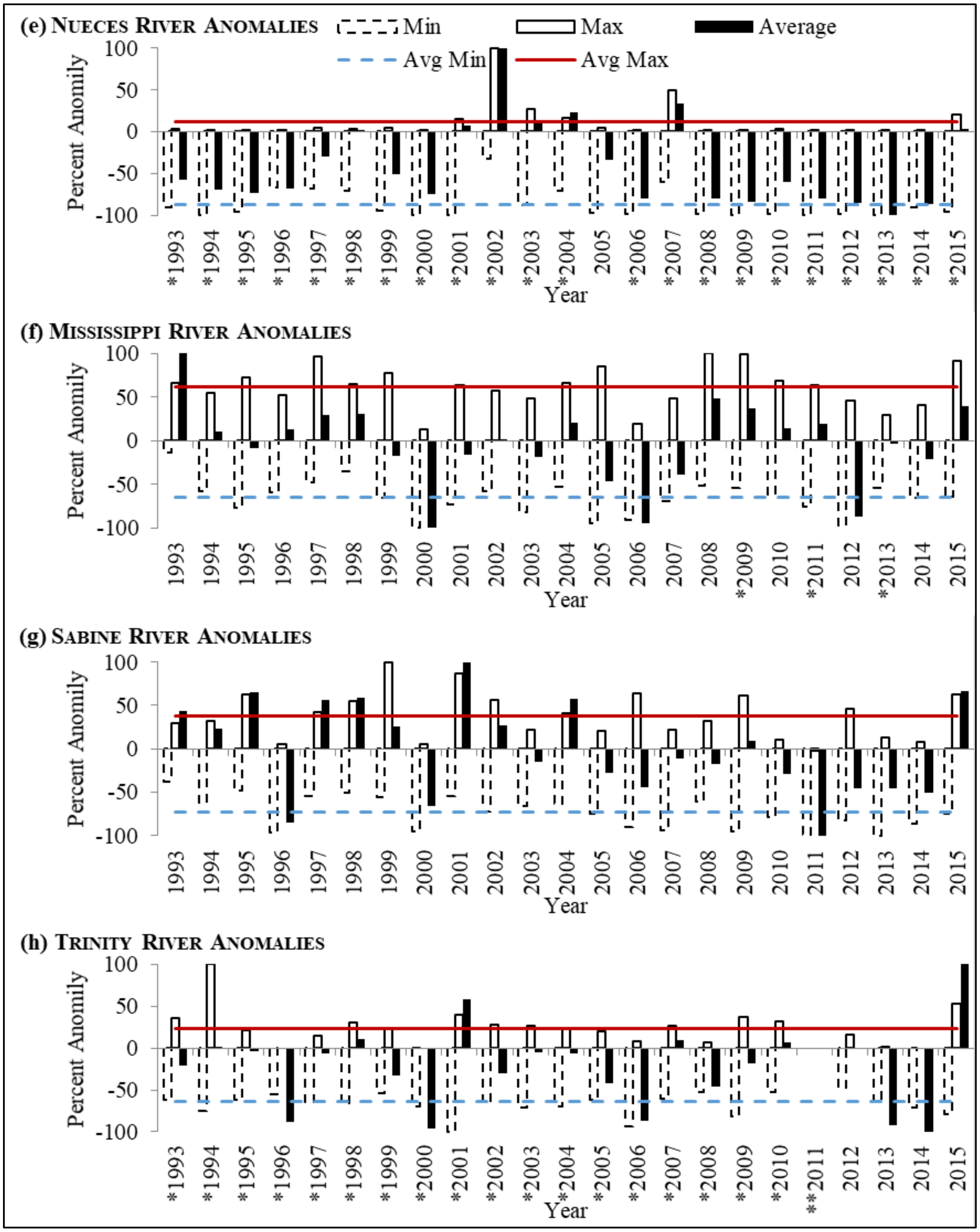


Figure 4.15. Cont'd.

Diffuse Attenuation

Kd490 indicated that the lowest surface water clarity occurred between November and June, and the greatest surface water clarity developed between July and October. The two maximum anomalies occurred in 2005 (0.09 m^{-1}) and 2008 (0.08 m^{-1}) (Figure 4.16). A significant positive correlation was found between Atchafalaya River discharge and surface Kd490 (t=3.94, p-value<0.001), with a correlation coefficient of 0.16.

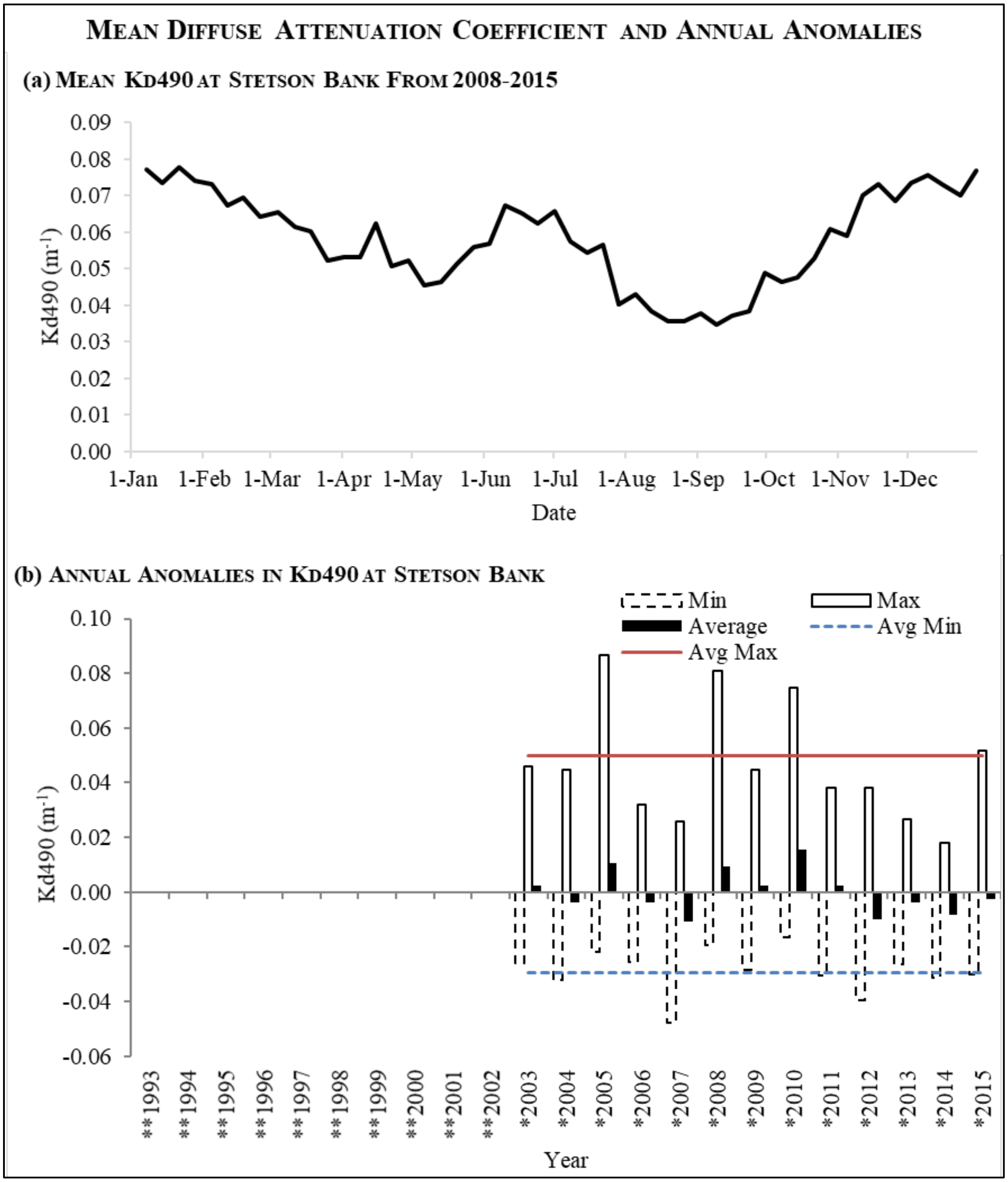


Figure 4.16. Mean Kd490 and annual anomalies at Stetson Bank. A: Mean Kd490 from 2008 to 2015. B: annual anomalies. Years with single asterisks (*) denote incomplete data and years with double asterisks (**) denote no data.

Summary

Two particular years are prominent due to the multitude of distressing oceanographic impacts occurring at the bank during the year (Table 4.3). In 2005, a hurricane passed within 200 km of the bank, ecologically significant degree heating weeks were recorded, both extreme high and low temperature anomalies were documented on the bank crest, low salinity anomalies were documented on the bank crest, and high diffuse attenuation anomalies were recorded. In 2010, a magnitude of degree heating weeks was recorded that could cause ecologically significant changes, extreme low temperature anomalies were documented on the bank crest, low salinity anomalies were documented on the bank crest, and high diffuse attenuation anomalies were recorded.

Table 4.3. Summary of oceanographic events at Stetson Bank. “ND” indicates no data and “X” indicates a year with the listed impact. Years where no events were documented and a limited amount of data were available are represented with a dash (-).

Year	Tropical Activity	Significant DHW	> Avg. High Temp. Anomaly @ 24m	< Avg. Low Temp. Anomaly @ 24m	< Avg. Low Sal. Anomaly @ 24m	> Avg. High Kd490 Anomaly @ 24m	Total
1993		ND			ND	ND	-
1994		ND		X	ND	ND	1
1995	X	ND		X	ND	ND	2
1996		ND	ND	ND	ND	ND	-
1997		ND	ND	ND	ND	ND	-
1998		ND	ND	ND	ND	ND	-
1999		ND	ND	ND	ND	ND	-
2000		ND	ND	ND	ND	ND	-
2001	X		ND	ND	ND	ND	1
2002	X				ND	ND	1
2003	X			X	ND		2
2004	X				ND		1
2005	X	X	X	X	X	X	6
2006			X		X		2
2007	X		X		X		3
2008	X		X			X	3
2009							0
2010		X		X	X	X	4
2011			X	X			2
2012			X				1
2013			X				1
2014				X			1
2015	X		X			X	3

The majority of nutrient samples analyzed have been below detectable limits for phosphorus, ammonia, nitrite, nitrate, phosphorus, nitrogen, and chlorophyll a. While further investigation is needed, isolated pulses in nutrient measurements may suggest that exposures to increased levels of nutrients are acute rather than chronic at Stetson Bank (Appendix D: Table D.5). Phosphorous has not reached detectable levels throughout the study period. In three sampling periods, ammonia levels reached detectable limits, with no stratification through the water column (7/8/2009, 8/21/2011, and 3/26/2012). On 5/18/2011, ammonium levels reached the maximum detected levels (0.30 mg/L) in surface water at Stetson Bank and declined with depth. While nitrite has never reached detectable levels, two samples had detectable levels of nitrate (2/16/2011 and 5/18/2011), but these levels were not detected throughout the water column. The greatest number of detections occurred in total nitrogen where eight samples throughout the water column were above detectable levels: 7/8/2009, 11/24/2009, 2/16/2011, 5/18/2011, 8/21/2011, 10/24/2011, 3/26/2012, and 11/9/2012. The highest recorded level was 35 mg/L in surface water on 8/21/2011. In one sample (11/9/12), chlorophyll a reached detectable levels throughout the water column.

Ocean carbonate samples were collected beginning in November, 2013. Samples included pH, pCO_2, alkalinity, and total dissolved CO_2 (DIC) (Appendix D: Table D.6), and indicate well buffered seawater with small annual pH and $\Omega_{aragonite}$ fluctuations <0.1 pH units (overall 8.073±0.024) and <0.8 (overall 3.70±0.20), respectively. Surface seawater *p*CO2 did not appear to significantly deviate from the atmospheric value and with the largest deviations (±50 μatm) occurred in late winter-early spring as well as late summer-early fall. In all other seasons when samples were collected and analyzed, air-sea *p*CO2 gradient was mostly <20 μatm. The distribution of ΔpCO_2 on an annual basis suggests that this area had a small net air-sea CO_2 flux.

Discussion

Hurricanes, typically sweeping across the Gulf of Mexico from the east or south, pose a 52% chance of passing within 200 km of Stetson Bank annually, similar to the probability at East and West FGBs (Lugo-Fernandez & Gravois 2010). Stetson Bank, situated deep in the water column (>17 m), can be somewhat insulated from direct physical impacts from all but the strongest storms; however, the fragile claystone/siltstone substrate that comprises the bank is particularly susceptible to mechanical damage (Hickerson & Schmahl 2005). NDBC Station 42019, located 107 km to the west-southwest of Stetson Bank, recorded the two greatest WVHTAs in 2005 and 2008, along with sufficient wavelength to indicate wave energy reaching the bank crest, when two of the largest tropical weather systems documented in this monitoring time frame (Hurricane Rita and Hurricane Ike, respectively) passed within 200 km of Stetson Bank. Coastal runoff can impose short term changes to the water column at Stetson Bank via anomalous discharge from rivers systems. However, as Stetson Bank is located ~130 km offshore in the middle of the continental shelf, local current conditions at the time of the high flow event are critical in determining the extent of coastal runoff impacts. Of the 12 tropical weather systems documented throughout the study period, two hurricanes, Rita in 2005 (Cat. 3) and Ike in 2008 (Cat. 2), were documented to affect the bank crest directly (both substrate and sessile biota damage) and indirectly (observed turbidity increases potentially due to coastal runoff). Hurricane Katrina, which made landfall in Louisiana in 2005, did not pass within 200 km of Stetson Bank but was followed three weeks later by Hurricane Rita. Combined, these two major hurricanes produced extensive coastal runoff that appears to have extended to Stetson Bank (Figure 4.17). However, the passage of Hurricane Rita may have helped mitigate an active coral bleaching event in the region by quickly reducing water temperatures as seen in Figure 3.11 (b).

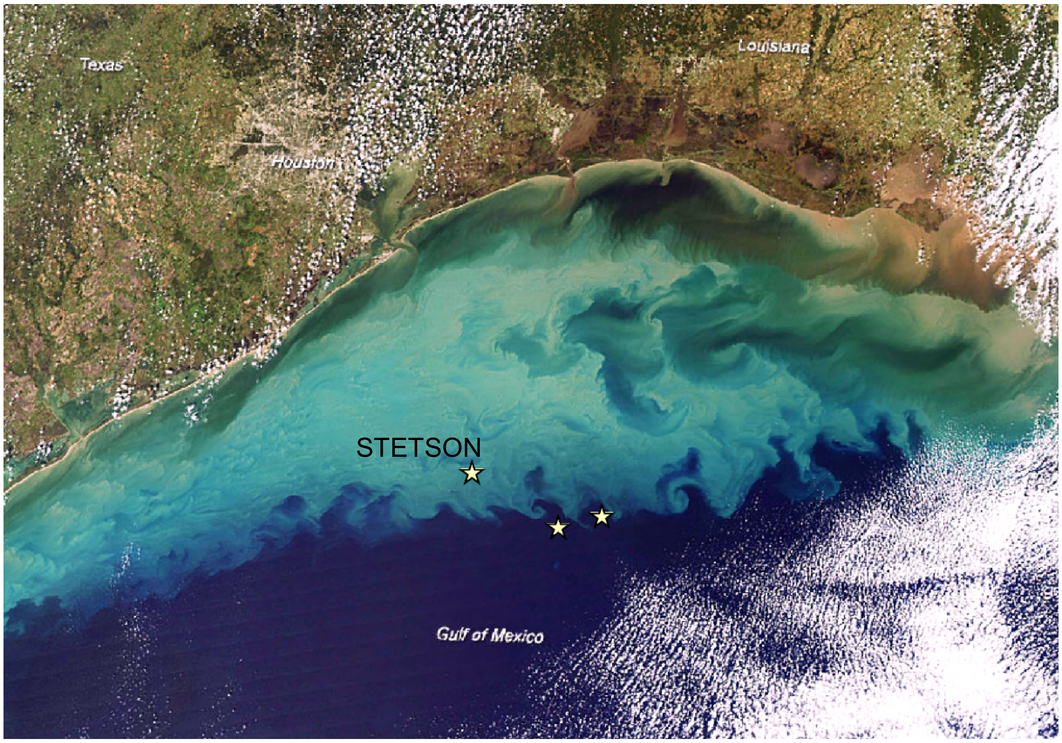


Figure 4.17. True color satellite imagery of the northwestern Gulf of Mexico. Image was taken on 9/25/2005, following the passage of Hurricanes Katrina and Rita. Image: NASA/GSFC MODIS, processed by NOAA CoastWatch

Several coral bleaching events have been documented at Stetson Bank since the initiation of the monitoring program. Calculating DHW can serve as a useful tool for predicting and tracking thermal stress that can lead to coral bleaching. Since 2001, 2005 and 2010 have reached thermal stress levels indicative of significant bleaching at Stetson Bank. These impacts were observed in benthic cover changes (see Chapter 2), primarily of the thermally sensitive hydrocoral *M. alcicornis*. While DHW calculations rely on sea surface temperature data, the depth of the bank crest at Stetson (>17 m) can help insulate benthic organisms from short-term temperature fluctuations due to the reduced temperature variability observed with increased depth. Annual temperature anomaly data do not take into account the duration of the anomaly, but do highlight the cyclic nature of El Niño (warmer than average) and La Niña (cooler than average) events as well as demonstrate that those impacts can be seen at both the sea surface and at depth. Both heat and cold can stress corals and extreme variability can have particularly detrimental effects on coral reefs (Hoegh-Guldberg 1999, Hoegh-Guldberg et al. 2005, Baker et al. 2008). Heat induced stress is characterized by reduced growth rates, bleaching, and ultimately the death of the coral over extended time periods. Cold-induced stress inhibits coral growth at a greater rate than heat stress; however, it has been documented that some corals can acclimate to a colder environment over extended periods of time (Roth et al. 2012). Sponges are another important component of the benthic community at Stetson Bank. Similar effects of heat and cold stress have been documented for sponge communities. Select species of zooxanthellate sponges can exhibit heat stress from sustained high temperatures through bleaching, and the subsequent death of the colony (Vicente 1990, Fromont & Garson 1999), whereas cold stress decreased sponge size, but did not lead to sponge mortality (Storr 1964). Approximately 45% of years within this reporting period saw warmer than average sea surface anomalies occur within the same year as cooler than average anomalies and, on the bank crest, only two years saw the same trend (2005 and 2011). Generally, greater thermal variability was seen on the sea surface than on the bank crest (+8.5 ºC to-11.5 ºC and +3.2 ºC to -5.4 ºC, respectively), with 2015 data from 30 m and 40 m stations supporting reduced thermal variability with increased depth. Long term trend analysis documented a significant increasing trend in SST at Stetson Bank over 22 years. A similar trend was not observed in the smaller 12-year sample period on the bank crest, potentially due to the need for multi-decadal data to effectively observe these trends.

Typically, the water column over Stetson Bank can be considered oceanic: salinity ~35 psu, low organic nutrient levels, annual pCO_2 fluctuations, and minimal terrestrial input. However, acute impacts from shore-based runoff are observed, primarily between April and August. Water from 31 states flows into the Gulf of Mexico, making it the largest watershed in the continental U.S. The Atchafalaya and Mississippi rivers have been identified as potential major input sources for waters that influence Stetson Bank (Dodge & Lang 1983, Rabalais et al. 1996, DeBose et al. 2013), with peak average discharge observed from March through June. However, anomalous discharge from smaller river systems, such as the Texas rivers included in this report, may also have the potential to impact the waters around Stetson Bank. While transporting much smaller quantities of water, these smaller systems reach discharge peaks at different times, often reaching peak discharge while the major U.S. river systems are outputting their lowest average discharge. Flow anomalies highlight years with major land flooding events. However, depth, spatial offset, and currents greatly influence how coastal run off mixes with other oceanic waters. Further studies on the river systems that influence the area of Stetson Bank are needed to help identify the relationship between coastal runoff and the localized low salinity events observed at the bank. A study by Le Hénaff et al. (2018) documented that the cumulative discharge of small rivers (Sabine, Neches, Village Creek, Trinity, San Jacinto, Brazos, Lavaca, Guadalupe, and San Antonio rivers) flowing into the northwestern Gulf of Mexico can equal that of the Atchafalaya River during intense rains and floods. Surface water clarity, represented by diffuse attenuation, can also be used to indicate potential coastal influences as water clarity is diminished by sediment particles suspended in coastal water as it moves offshore. Diffuse attenuation at Stetson Bank showed a significant correlation between reduced water clarity and increased flow rate from the Atchafalaya River, with high anomalies recorded following major hurricane events (2005 and 2008). Coastal water influences can cause increased levels of nutrients and pollutants, as well as low salinity events, in typically oceanic waters. While not tested during the timeframe of this study, additional assays to evaluate the presence of pollutants in the water column at Stetson Bank would provide additional insight into potential contaminants transport to the bank during acute coastal runoff events. Strong and rapid onset low salinity events have been documented to cause similar responses in corals as thermal stress, where corals exhibit bleaching, tissue sloughing, and subsequent death (Coles 1993, Titlyanov et al. 2000, Kerswell & Jones 2003). Similarly, Storr (1964) documented tropical marine sponge mortality following low salinity shock events. While salinity data from the bank crest are limited, the greatest anomaly was observed in 2005, where bank crest salinity was reduced to an all-time low of 27.4 psu in June.

Nutrient levels in water samples at Stetson Bank support a hypothesis that coastal impacts are not chronic, as nutrients remained below detectable limits for the majority of samples. High levels of nutrients indicate poor water quality conditions that can impact the organisms on the bank. Ammonia, a natural byproduct of decomposition and protein metabolism (excreted as a waste by animals), can also be introduced to a system through anthropogenic sources, including pollution from fertilizers and organic matter. Ammonia serves as a nitrogen source for plant growth; however, in high concentrations, it can be toxic to a variety of marine life (in relation to pH and temperature) (USEPA 1989). Nitrogen and phosphorous, naturally occurring nutrients that can also be introduced through anthropogenic sources such as pollution from fertilizers, support the growth of algae and plants. However, persistent high levels of these nutrients fuel algal blooms that can smother other benthic organisms and deplete oxygen in the water. Although samples at Stetson Bank did occasionally present detectable values, they were below levels considered dangerous for marine organisms (USEPA 1989).

Seasonal and spatial distribution of seawater carbonate chemistry demonstrates that seawater in the FGBNMS area (including East Flower Garden Bank, West Flower Garden Bank, and Stetson Bank), despite its proximity to land, behaved similar to an open ocean setting (such as the Bermuda Atlantic Time-series Study, or BATS) (Bates et al. 2012) in terms of its annual pCO_2 fluctuation and minimal terrestrial influence. Carbonate analysis indicated a thermal control on the carbonate system (carbonate saturation state and CO_2 partial pressure, or pCO_2) in this region. Seasonal patterns in $npCO_2$ may correspond to a shift in the balance between respiration and production. In 2010, concern over potential contamination from the *Deepwater Horizon* Macondo well oil spill led to a Natural Resource Damage Assessment that tested for hydrocarbons at Stetson Bank. While polynuclear aromatic hydrocarbons were detected in low concentrations in most samples (DIVER 2018), the source of the detected hydrocarbons is unknown. Low concentrations and a lack of apparent physical damage to the biota on the bank suggest that the hydrocarbons had no significant lethal impact on the biota. Additional testing to identify the isotopic source of the hydrocarbons in samples collected from Stetson Bank would provide further insight into the source of carbon in this region.

Overall Conclusions

Stetson Bank is typically bathed in oceanic waters with high salinity and low nutrient levels. However, periodic acute coastal events can impact the area. These factors, combined with the high latitude of the bank and thermal variability seen in the water column make Stetson Bank a marginal environment for coral recruitment and growth. Despite challenging environmental conditions, the bank supports a benthic community of coral and sponges and the presence of warm tropical waters brought to the area from the Caribbean via the Loop Current and spin-off eddies combined with oligotrophic waters maintain sufficient conditions for coral and sponge community growth and larval transport (Biggs 1992, Schmahl et al. 2008). However, dramatic changes in the benthic community have occurred when a multitude of oceanographic stressors impacted the bank (notably 2005 and 2010). The synergistic effect of these multiple stressors potentially increased their impact on the benthic community (Coles & Jokiel 1978), and is likely a factor driving the changes observed on the bank crest.

Chapter 5
CONCLUSIONS

This report presents a historical review and reanalysis of available data from 1993 through 2015 for Stetson Bank, an uplifted claystone/siltstone feature within Flower Garden Banks National Marine Sanctuary, located in the northwestern Gulf of Mexico. Federal and state agencies, as well as scientific institutions, have provided a wide array of information including studies, publications, and reports regarding the geology and biology of the bank crest. Due to its location in a region of well-developed offshore activity (including oil and gas exploration, commercial shipping, and commercial fishing) and the presence of diverse and dense benthic and fish assemblage, the local community, as well as researchers and legislators, lobbied to add Stetson Bank to FGBNMS, marking it as a place of national significance.

Four benthic habitats were documented at Stetson Bank, each with characteristic communities (high and low relief bank crest, coralline algae reef, and deep reef). Since 1993, the high relief benthic community at Stetson Bank has had periods of stability and periods of significant change. While significant changes have occurred over short time periods, such as the dramatic decline of hydrocoral (*M. alcicornis*) and sponge (*C. nucula*) cover between 2005 and 2006, long term trends were also observed, such as the steady decline of sponges and increase of macroalgae. The data presented and reanalyzed in this report supports previously documented significant changes in the benthic community between 1998-2000 and 2005-2006 (DeBose et al. 2013) and found additional significant changes between 2010-2011 and 2014-2015. Changes in the community were primarily due to declining hydrocoral and sponge cover while macroalgal cover increased. However, the significant changes in 2014-2015 were primarily due to declining macroalgae cover and the increased availability of substrate for potential colonization. Although macroalgal cover is highly dynamic and dependent upon location and season (Diaz-Pulido & Garzon-Ferreira 2002, Bruno et al. 2009, Jackson et al. 2014, Bertolino et al. 2016), a significant increasing trend of macroalgal cover was documented at Stetson Bank from 1998 to 2012, followed by a rapid decline through 2015. During this period, a corresponding increase in the keystone urchin grazer *D. antillarum* occurred, thereby having a top down control effect on macroalgal cover.

Similar to benthic habitats, four distinct fish communities were documented between the habitats, each supporting a diverse and variable community. When bank crest data were compared with results presented in Pattengill et al. (1997), the trophic structure of the fish community at Stetson Bank has changed since the mid 1990s, with piscivores and herbivores comprising a greater proportion of the community and planktivores representing less in 2015. A longer-term dataset is needed to evaluate recruitment trends as current data suggests that recruitment increased between 2013 and 2015 and divers have noted distinct recruitment events of fish species occurring at Stetson Bank. This potentially contributes to the high temporal variation seen in the community and can make interpreting significant changes in monitoring data difficult. Long term trends were also found in historical sighting frequency data, including the co-variance of two species of spongivorous angelfish with total sponge cover, highlighting potential predator-prey dynamics on the bank crest and documenting impacts through the food chain of declining sponge cover. On a species level, the blue angelfish (*H. bermudensis*) co-varied with *I. strobilina*, the current predominant sponge species found in repetitive photostations at Stetson Bank. Both species have been in slow decline since 1993.

A variety of exotic and invasive species have been documented at Stetson Bank. Of particular concern, due to their invasive nature and negative impacts throughout the Caribbean, are lionfish (*P. volitans*) (Arias-González et al. 2011, Albins & Hixon 2013). For the timeframe of this report, observations were too infrequent for in-depth analysis of impacts and spatial analysis indicates their density and biomass were greater in the mesophotic habitat surrounding the main reef feature. Andradi-Brown et al. (2017) documented that lionfish inhabiting mesophotic habitat in Honduras had greater biomass and higher fecundity than those on shallower reefs, indicating that these individuals using deeper habitat can disproportionately contribute to the recruitment of lionfish in shallower habitat. Additionally, as it is more difficult to conduct targeted removals below recreational scuba diving limits, the mesophotic lionfish will potentially serve as a primary source population for the bank crest.

Data presented in this report show that Stetson Bank typically experiences at least one environmental stressor annually (such as tropical storm activity, significant DHW, anomalously high or low water temperatures, anomalously low salinity, or anomalously high diffuse attenuation), but the benthic community has remained stable during those years without multiple stressors. Although environmental data were lacking for the 1998/2000 event, other significant changes in the benthic community occurred in years where multiple environmental stressors were documented in the same year (2005/2006, 2010/2011, and 2014/2015). Similar to other coral reef ecosystems (D. Vinebrooke et al. 2004, Yakob & Mumby 2011, Darling et al. 2013) the benthic community at Stetson Bank can be resilient when impacted by a few stressors annually, but the synergistic effect of multiple stressor interactions can result in impaired recovery and significant community changes. Stressors considered in this report are primarily linked to water temperature and quality, where coastal water runoff is a primary concern in this otherwise typically oceanic environment. Historically, sources for coastal runoff for the region have focused on major U.S. river systems but anomalously high discharge from major Texas river systems may play a larger role, in concert with currents and wind, than previously considered (Le Hénaff et al. 2018).

Historical impacts and their frequency are important to examine as they influence how subsequent events impact the biota. At Stetson Bank, the impact to bleaching-sensitive hydrocoral cover appeared greater in 2005/2006 (25% to 6%) than in 2010/2011 (7% to <1%). While several environmental stressors occurred at the bank in 2010/2011, most of the hydrocoral cover was lost in 2005/2006, leaving a much smaller hydrocoral cover to be affected 2010/2011. The frequency of stressful environmental events is predicted to increase as the climate changes (Easterling et al. 2000, Hoegh-Guldberg et al. 2008), providing little time for recovery between events and leading to dramatic and long-term shifts in communities (Done 1992, Knowlton 2015).

Stetson Bank occurs along a high latitude (above 28º North) in marginal environmental conditions to support coral reef growth. The dynamic pattern of stressors presented in this report highlights that these high latitude reefs may not be suitable for long term coral reef development, but may experience years of coral reef growth and stable environmental conditions followed by a year of multiple stressors, leading to dramatic changes in the benthic community. However, the variable temperature regimes that corals at these locations experience are thought to increase their thermal tolerance (Oliver & Palumbi 2011), driving the evaluation of high-latitude reefs as potential coral refuges in the face of climate change. However, additional site-specific factors, including larval dispersal, environmental conditions, and geographic location, should be considered in evaluating these communities. Climate experts suggest that the protection of high latitude reefs, in order to reduce stressors and support resilient communities, is the best current course of action for resource managers (Beger et al. 2014).

The monitoring program at Stetson Bank represents one of the longest-running monitoring efforts of a northern latitude coral community. These historical datasets help document changes in the community, increase our understanding of the community change, examine environmental interactions over time, and monitor the drivers of ecosystem change in the northern Gulf of Mexico, thereby guiding research initiatives and management decisions in the region.

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GLOSSARY OF ACRONYMS

ABC – Abundance biomass curves
BATS – Bermuda Atlantic Time-series Study
BLM – Bureau of Land Management
BOEM – Bureau of Ocean Energy Management
BSEE – Bureau of Safety and Environmental Enforcement
CCL – TAMU-CC Carbon Cycle Laboratory
CCMA – NCCOS Center for Coastal Monitoring and Assessment
CMECS – Coastal and Marine Ecological Classification Scheme
CPC – Cardinal Point Captains
CPCe – Coral Point Count® with Excel® extensions
DHW – Degree heating weeks
DIC – Total dissolved CO_2
DO – Dissolved oxygen concentration
EFH – Essential Fish Habitat
ENSO – El Niño-Southern Oscillation
EOS – NASA's Earth Observing System
EPA – Environmental Protection Agency
FGBs – Flower Garden Banks
FGBNMS – Flower Garden Banks National Marine Sanctuary
GMFMC – Gulf of Mexico Fishery Management Council
GREAT – Gulf Reef Environmental Action Team
H – Herbivores
HAPC – Habitat Area of Particular Concern
I – Invertivores
MMS – Minerals Management Service
MODIS – Moderate Resolution Imaging Spectroradiometer
NASA – National Aeronautics and Space Administration
NMSF – National Marine Sanctuary Foundation
NCCOS – National Centers for Coastal Ocean Science
NDBC – National Data Buoy Center
nMDS – Non-metric multidimensional scaling
NOAA – National Oceanic and Atmospheric Administration
NRDA – Natural Resource Damage Assessment
NURC – National Undersea Research Center
ONMS – NOAA Office of National Marine Sanctuaries
P – Piscivores

PCO – Principal component ordination
PERMANOVA – Permutational multivariate analyses of variance
PL – Planktivores
RDT – Roving diver technique
REEF – Reef Education and Environmental Foundation
ROV – Remotely operated vehicle
SIMPER – Similarity percentages
SIMPROF – Similarity profile analysis
SST – Sea surface temperature
STL – Seasonal Trend Decomposition using Loess
TAMU-CC – Texas A&M University – Corpus Christi
TAMUG – Texas A&M University at Galveston
TWDB – Texas Water Development Board
UNCW – University of North Carolina at Wilmington
UNCW-UVP – University of North Carolina at Wilmington - Undersea Vehicle Program
USACE – U.S. Army Corps of Engineers
US BGN ACUF – United States Board on Geographic Names Advisory Committee on Undersea Features
USCGS – U.S. Coast & Geodetic Survey
USF – University of South Florida
USGS – U.S. Geological Survey
UTRGV – University of Texas Rio Grande Valley
WVHT – Wave height
WVHTA – Wave height anomalies

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