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Assessing Long-Term Coral Coverage at East and West Flower Garden Banks

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ASSESSING LONG-TERM CORAL COVERAGE AT EAST AND WEST FLOWER GARDEN BANKS

A Thesis

by

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ABSTRACT

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Coral reefs provide critical habitat for diverse benthic communities; however, more than 10% of existing reefs have been lost as a result of natural and anthropogenic stressors. While there has been a decline in reefs around the world, East and West Flower Garden Banks (FGBs), part of Flower Garden Banks National Marine Sanctuary (FGBNMS), have maintained >50% live coral cover for over 27 years as revealed by the FGBNMS long-term monitoring (LTM) program. In the 1980's, the LTM program at East and West FGBs was developed in response to the expansion of oil and gas exploration, anchoring, and fishing impacts to assess coral and reef-associated benthic organism health. As part of the LTM program, repetitive photostations were established in 1989 to detect and evaluate long-term changes and changes within the benthic community at East and West FGB. However, the assertion of maintenance of over 50% live coral coverage has been questioned because of changes in LTM methods used for image acquisition and data analysis throughout the study period. To address this concern, I used a standardized methodology to reanalyze repetitive photostation data and found a significant increase in repetitive photostation coral coverage at West FGB from 1989 to 2017 ($P = 4.3154\ e^{-05}$) and no significant change at East FGB ($P = 0.36783$). This is in contrast to the recorded declines at other coral reefs in the Gulf of Mexico region.

DEDICATION

The completion of my master's work would not have been possible without the love and support of my family and friends. To my parents Ricardo and Connie: thank you both for raising me to become a headstrong, brave & hard-working young woman. I know pursing my dreams of studying the ocean was something you didn't understand, but with your love and support, you both gave me the confidence to attain my dreams. To my brothers Richie and Rolie: thank you for always believing in me and supporting me throughout this journey, even when I didn't have that same confidence in myself. Thank you to my friends, old and new, who have given me endless support. For my grandma Concepcion: She was the one who pushed me to follow my dream and gave me the strength to run with it, even when others doubted me. She always loved me unconditionally and believed in me. Although she is no longer with us, I want to dedicate this work to her.

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Most importantly, I would like to thank the Flower Garden Banks National Marine Sanctuary Office in Galveston, Texas, for making my work possible. Thank you to all professionals I had the pleasure and opportunity of working with, Dr. Raven Blakeway, Kelly O'Connell, and Emma Hickerson. I would also like to extend my utmost gratitude to Grace McDermott, for working with and helping me with analysis of West FGB work.

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"Any opinions, findings, conclusions, or recommendations expressed in this publication are those of the author(s) and do not reflect the view of the U.S. Department of Commerce, National Oceanic and Atmospheric Administration."

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CHAPTER I

INTRODUCTION

Located in the northwestern Gulf of Mexico, Flower Garden Banks National Marine Sanctuary (FGBNMS) consists of 17 banks supporting coral reefs and coral communities. East Flower Garden Bank (East FGB) and West Flower Garden Bank (West FGB), known together as the Flower Garden Banks (FGBs), are located 190 km off the coast of the Texas-Louisiana border (Figure 1). The FGBs are found on top of ancient salt domes (diapirs) that formed 160-170 million years ago when the seawaters evaporated, leaving an abundant layer of salt on the seafloor (Schmahl et al. 2008; Johnston et al. 2021). In time, as the Gulf of Mexico deepened, pressures from dense overlying sediments caused the uplift of these salt layers (Gardner et al. 1998; Brooke & Schroeder 2007). Coral communities began to colonize the tops of the salt domes approximately 10,000-15,000 years ago (Hickerson et al. 2008).

East FGB is approximately 65.8 km^2, of which 1.02 km^2 is coral reef, and West FGB is approximately 77.2 km^2, of which 0.4 km^2 is coral reef (Hickerson et al. 2008). These banks range in depth from 16-150 m and the shallowest portions (< 40 m) are covered by well-developed coral reef (Johnston et al. 2016). These coral reefs consist of 21 different species of scleractinian corals, 18 of which are hermatypic (reef-building) (Bright et al. 1984).

Reef-building corals are a key component of the ocean environment because they provide habitat for diverse fauna, including commercially important fish species (Hoegh-Guldberg 1999; Cole et al. 2008; Coker et al. 2013). For example, the FGBs support a high abundance of fish, are a seasonal habitat for whale sharks, and are an important habitat for juvenile manta rays (Gitting

& Hickerson 1998; Childs 2001; Pattengill-Semmens et al. 2003; Stewart et al. 2018). Coral reefs not only provide habitat but also protect coastlines and are important for attracting tourism (Hoegh-Guldberg 1999; Cole et al. 2008; Coker et al. 2014). Diving can be an important tourist activity for coastal cities, including those in Texas, where the FGBs attract divers not only for tourism but also to aid scientific research and special-permitted events such as invasive lionfish removals (Thailing et al. 2003; Johnston et al. 2016). For example, the FGBNMS hosts an annual Lionfish Invitational, a permitted four-day event to remove as many invasive lionfish as possible (FGBNMS 2021).

Recognizing the importance of the FGBs, the U.S. Department of Interior, (starting with the Bureau Land Management, then the Minerals Management Service [MMS]) began monitoring the reefs in the early 1970s after the first quantitative data was collected in 1972 (Johnston et al. 2018a, 2021). In 1978, offshore drilling activities near the East FGB prompted the initiation of the first regular monitoring efforts, which continued until drilling ceased in 1983 (Gittings 1998; Aronson et al. 2005). The official start of the current long-term monitoring (LTM) program at the FGBs began in 1989. The goal of the program was to assess coral health, detect biotic community changes over time, and provide baseline data in the event of natural or human-induced impacts. This program was funded through a partnership between the MMS and Texas A&M University (Johnston et al. 2018a, 2021). The FGBs were designated a national marine sanctuary in 1992, and Stetson Bank was added to the sanctuary boundaries in 1996 (Gittings & Hickerson 1998; Cancelmo 2008). The motivation for their national marine sanctuary designation was to protect them from expanding oil and gas exploration as well as from anchoring and fishing impacts. In 2021, FGBNMS expanded to protect 14 additional reefs and banks, extending the sanctuary from ~145 km to ~414 km. The FGBNMS is one of 15 national marine sanctuaries in the United States (Schmahl et al. 2008; NOAA Sanctuaries 2021).

The FGBNMS LTM program consists of non-destructive monitoring of the benthic community within designated 1-hectare study sites via random transect photostations and repetitive photostations on the coral caps of East FGB and West FGB. The study sites on each bank were delineated using corner markers and the installation of eyebolts at 25 m markers along each 100 m perimeter and crosshair line (Figure 2 & 3; Johnston et al. 2018a, 2020, 2021). Random transects are a series of non-overlapping randomly initiated 10 m photo transects within study sites that are used to document the benthic reef community at East FGB and West FGB (Johnston et al. 2016, 2018a, 2021). Repetitive photostations follow specific colonies over time and document changes in benthic assemblages within study sites (17-27 m) (Johnston et al. 2013, 2016, 2020). Selection of permanent (repetitive) long-term monitoring sites was based on several factors, such as habitat type, size (coral colonies within the specified area), history (previous monitoring within the specified area), management relevance, depth, location (to coral colonies and other study sites), and accessibility and safety during sampling (Johnston et al. 2018a, 2018a). Permanent (repetitive) study sites at East FGB and West FGB were designated in 1988-1989 within the reef-building zones (Gittings et al. 1992; Johnston et al. 2018a). The data generated from the repetitive photostations allows for the comparison of the habitats and benthic communities within and between East and West FGBs (Johnston et al. 2017). Other components of the LTM program that are important to assess the overall health of the reef include water quality sampling, coral demographic surveys, planimetry techniques, videotaped transects, fish surveys, and sea urchin and lobster surveys (Johnston et al. 2018a).

The monitoring program at the East and West FGBs have documented over 50% live coral cover during the last 30 years (Johnston et al. 2021). This percentage of coral cover indicates that these reefs support a healthy, thriving coral reef ecosystem (Hickerson et al. 2012; Johnston et al. 2016, 2021). Such healthy coral ecosystems are uncommon considering there has been a significant decline in living corals worldwide in recent decades in response to climate change and anthropogenic activities, which have increased physical stressors that negatively affect corals (Gardener et al. 2003; Mumby et al. 2011; Jackson et al. 2014; Johnston et al. 2016, 2019, 2021).

Some effects of climate change that negatively impact corals include rising ocean temperatures and ocean acidification. Such effects are caused by increasing concentrations of atmospheric greenhouse gases and subsequent increases in the concentration of absorbed carbon dioxide in the ocean (Hoegh-Guldberg & Bruno 2010). The effects of these changes differ with depth; reefs that are located at shallower depths (< 30 m) are often more heavily affected than reefs found at deeper depths (Richmond 1993; Hickerson et al. 2008; Bongaerts et al. 2010). Although the FGBs are between 16-150 m deep, the reef-building coral cap ranges in depth from 16-46 m, potentially exposing them to disturbance (Johnston et al. 2013; Johnston et al. 2019). For example, a bleaching event occurred in late September and early October of 2016 when seawater temperatures were above 30°C (86°F) for 36 days at East FGB and 21 days at West FGB (Johnston et al. 2017, 2019). Bleaching was observed from repetitive photostations at 17-27 m at East FGB, where 24% of the corals exhibited bleaching, and at 18-25 m at WFGB, where 10% of the colonies exhibited bleaching (Johnston et al. 2017, 2019). Most bleached corals had recovered by January 2017, with only 4% of corals at East FGB still showing signs of bleaching (Johnston et al. 2017, 2019). Although these corals appeared to have recovered, severe effects from disturbances, such as prolonged coral bleaching events, can lead to substantial reef die offs (Hughes et al. 2003). Die offs have occurred in shallower reefs, such as the Great Barrier Reef and the Florida Keys, where bleaching events have been a common reoccurrence (Berkelmans et al. 2004).

In addition to climate change, tourism may also have negative impacts on coral reefs. Although tourism provides economic value and brings appreciation of coral reefs to communities, direct (e.g., coral mining, digging of canals and access into islands and bays, diver interactions, and blast fishing) and indirect (e.g., pollution, overfishing, sedimentation) tourism-related human interaction with coral reefs often causes negative impacts (Pollnac 2007). These activities also result in loss of coral reef habitat but the extent to which coral reefs are impacted differs by reef location (e.g., inshore vs. offshore), proximity to human activity and interaction, and frequency and type of anthropogenic activity (Hodgson 1999; Halpern et al. 2008). Loss of coral reef habitat can have detrimental effects on the marine environment by reducing habitat and thus reducing fish abundance and diversity, nursery habitat, and tourism revenue (Hoegh-Guldberg 1999; Cole et al. 2008; Bridge et al. 2013; Coker et al. 2014). Because the FGBs have been protected by bottom-contact restrictions since 1992 and are 190 km from shore-based anthropogenic activities, they may experience fewer negative effects from tourism and fisheries activities and thus may not have experienced declines in coral reef habitat from these threats. However, despite their remote locations and bottom-contact restrictions, the FGBs have experienced fishing threats, such as illegal fishing within the sanctuary as well as discarded fishing gear (Office of National Marine Sanctuaries 2008).

Although the FGBNMS LTM program does indicate a stable and healthy coral ecosystem over the last 30 years (Johnston et al. 2021), the assertion of maintenance of over 50% live coral coverage has been questioned because of changes in LTM repetitive photostations methods used for image acquisition and data analysis throughout the monitoring period (Table S1). At repetitive photostations, images are taken by divers, who use maps to locate photostations (Figure 5 & 6) and then use a t-frame camera setup to take the photograph in the same orientation annually (Johnston et al. 2016, 2021). From 1989 to 1995, images were analyzed using mylar traces and a calibrated planimeter (Gittings et al. 1992). From 1996 to 2003, cover estimates were determined by projecting 35 mm slides onto a flat surface containing three sets of 100 randomly positioned markers (Dokken et al. 1999; Dokken et al. 2003; Johnston et al. 2021). Within the same time frame, 1996-2003, coverage was also estimated by another point-count method that electronically distributed 250 random points onto the photo in Photoshop 7.0 (Precht et al. 2005; Johnston et al. 2021). It was not until 2004 that the current method of projecting images in Coral Point Count with Microsoft Excel extensions (CPCe) became the standardized method (Johnston et al. 2021). Using different techniques over the years could have induced variation unrelated to actual changes in the coral community. To address these concerns relating to the methodology and to validate the findings of Johnston et al. (2021), this study will analyze repetitive photostation images from 1989 to 2017 using standardized methodology and modern taxonomy to determine the state of living corals at the FGBs (Figure 4).

To accomplish this goal, the following objectives were pursued:

(1) Reanalyze repetitive photostation data from 1989 to 2017 using a repetitive photostation key to determine coral species coverage over the 27-year period (1989-2017). This assessment will ultimately result in a coral cover time-series analysis from 1989-2017 from the East and West FGB repetitive photostations using standardized methodology.

(2) Utilizing the dataset produced from the reanalysis of repetitive photostation images, determine which coral species are the most reoccurring at East and West FGBs.

(3) identify potential drivers of the coral coverage trends observed at East and West FGBs repetitive photostations.

I anticipate that this standardized approach will improve accuracy of species identification, coral cover estimates, and monitoring of coral reef health at East and West FGBs. For example, the development of the photostation keys will allow for better species identification of older photographs. Applying a standardized method when analyzing repetitive photostation data from East and West FGB will allow for validation of past coral coverage percentages.

I hypothesize that based on repetitive photostation coral coverage data, a significant increase in coral coverage will be observed at both East and West FGBs and that *Orbicella franksi* and *Pseudodiploria strigosa* will be the biggest contributors to coral coverage at East and West FGBs.

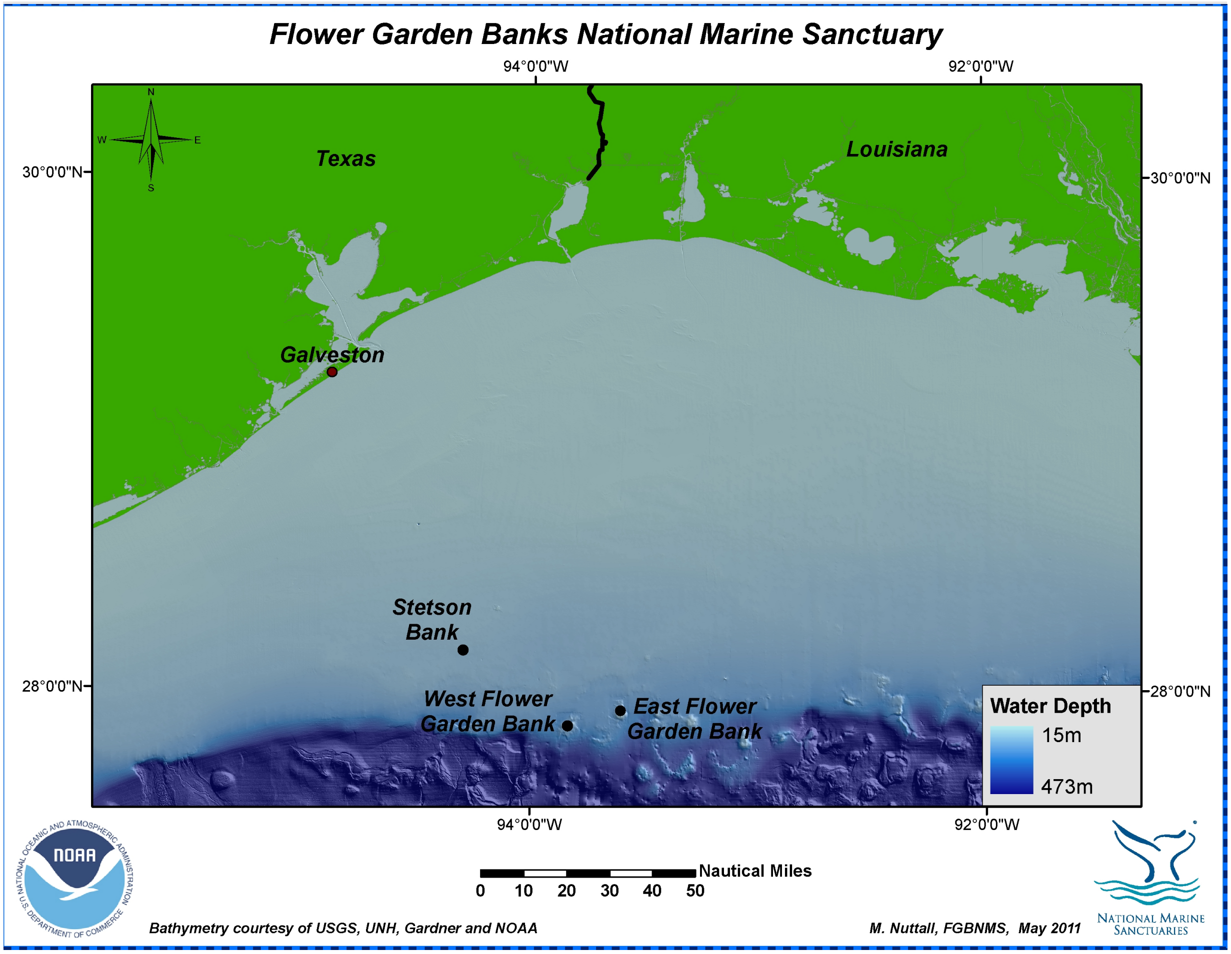

Figure 1: Locations of East and West Flower Garden Banks and Stetson Bank located within the boundaries of Flower Garden Banks National Marine Sanctuary (FGBNMS). Image is available from the FGBNMS website.

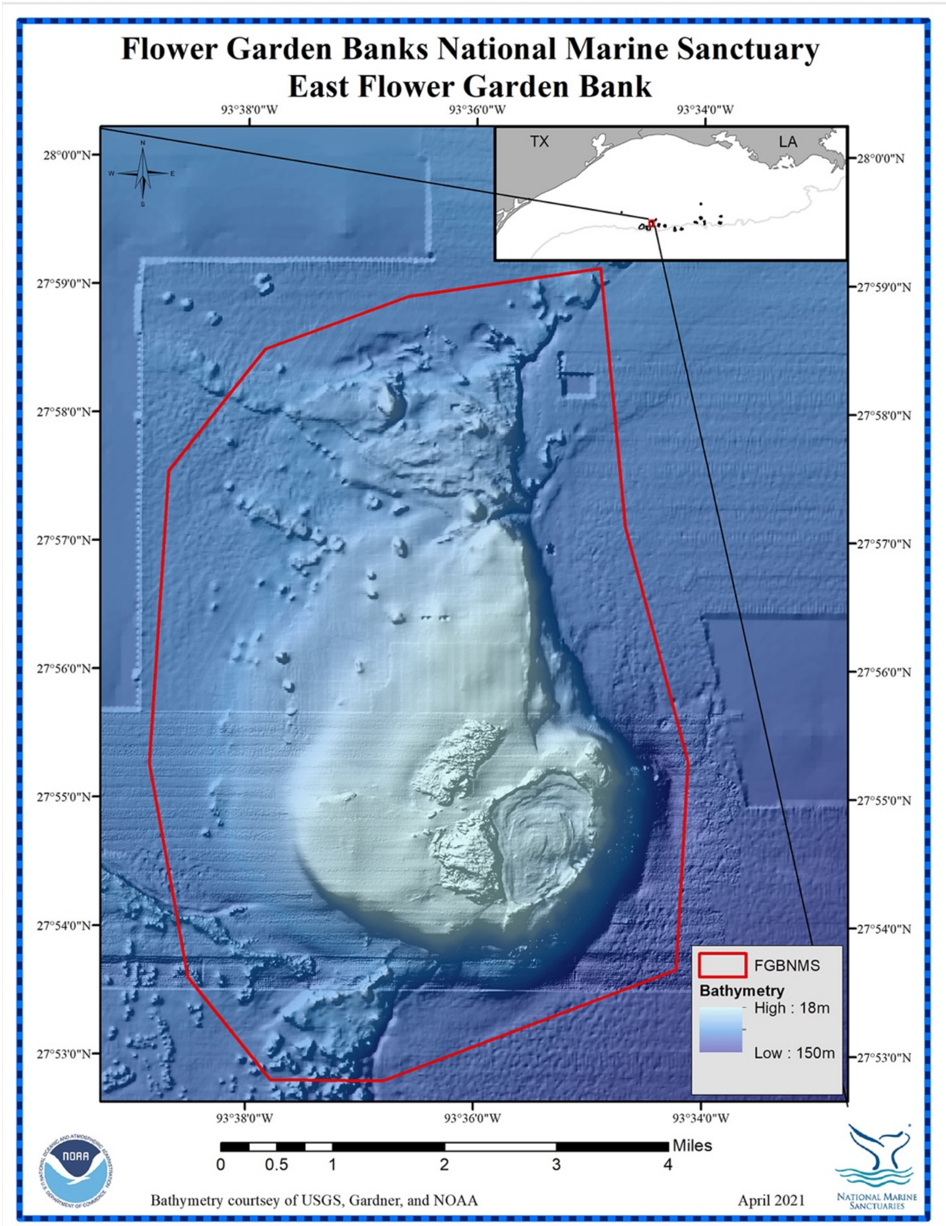

Figure 2: Bathymetry map of East Flower Garden Bank, located within the boundaries of Flower Garden Banks National Marine Sanctuary (FGBNMS). Image is available from the FGBNMS website.

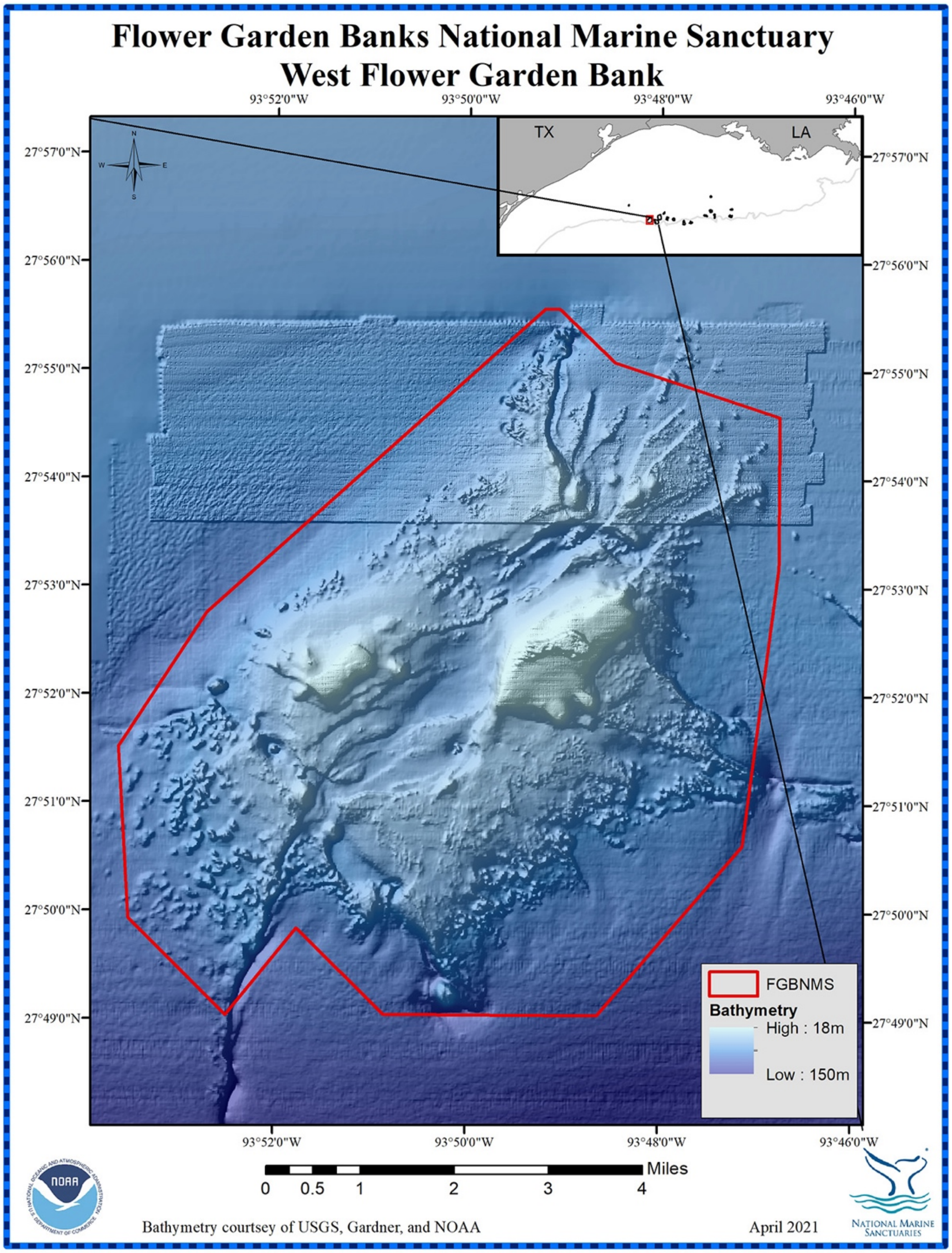

Figure 3: Bathymetry map of West Flower Garden Bank, located within the boundaries of Flower Garden Banks National Marine Sanctuary (FGBNMS). Image is available from the FGBNMS website.

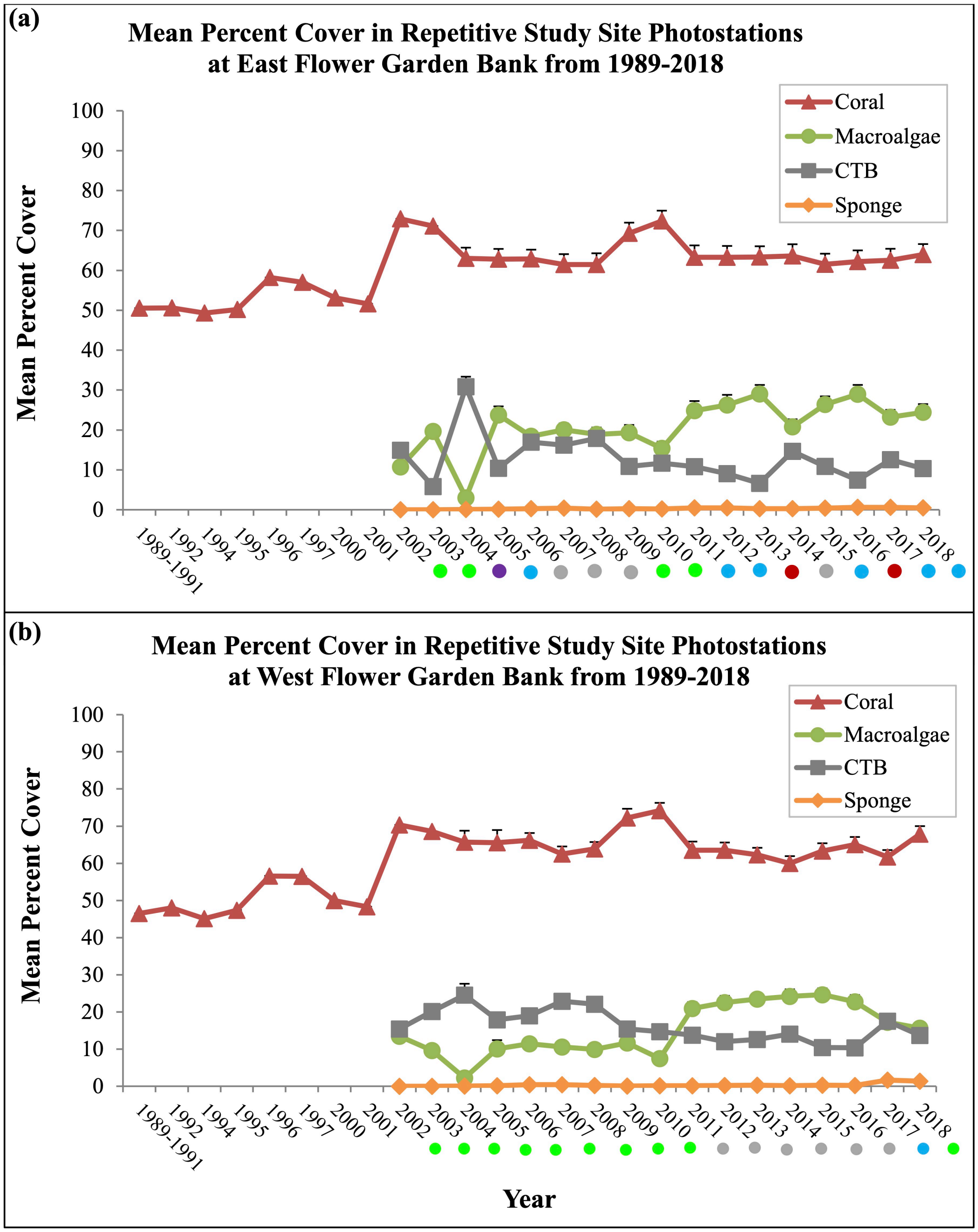

Figure 4: Mean percent benthic cover + SE of repetitive photostations of functional groups within (a) East Flower Garden Bank and (b) West Flower Garden Bank study sites from 1989 to 2018 (Images from Johnston et al. 2020 publication).

CHAPTER II

METHODOLOGY

Repetitive Photostations

The FGBNMS's LTM program includes 37 permanent (repetitive) photostations at East FGB and 41 at West FGB, within a 1-hectare study site at each bank. Repetitive photostations were marked by permanent pins with numbered cattle tags. East FGB LTM photostations occur at 17-27 m depth on the eastern side of the bank (Johnston et al. 2021). West FGB LTM photostations are at 18-25 m depth in the central region of the bank (Johnston et al. 2021). Repetitive photostation data were collected during annual monitoring trips that occurred between June and August, starting in 1989, and coincided with water-quality assessments and faunal censusing of other key reef taxa (e.g., lobsters, urchins, and fishes). Repetitive photostations were located by divers using detailed underwater maps (Figure 5 & 6). Once located, photostations were photographed using a high-quality camera housed in an underwater housing compartment, camera specifics varied over the 27-year period as technology changed and advanced (Table S1; Johnston et al. 2021). The camera was mounted 2 m from the substrate on the center of an aluminum t-shaped frame (Johnston et al. 2021). To ensure correct orientation each year, the aluminum t-shaped frame was positioned in a north-facing direction using a compass and was kept vertical with an attached bulls-eye bubble level (Johnston et al. 2021). This setup captured a seafloor area of 5 m^2 (Johnston et al. 2021)

Coral Identification Key/ Identification of 'Unidentified' Corals

To increase coral identification accuracy, in particular for the older photos with low-quality resolution, and to standardize coral identifications, a coral identification key was created for each repetitive photostation in Adobe Illustrator (AI) with archived FGBNMS images from 2015. This year was chosen because the images were the most recent, high-quality photographs obtained before bleaching events occurred in 2016. To create the coral identification key, photos were imported into AI. All living scleractinians within each image were outlined in designated species-specific colors using a pencil tool and a Wacom Cintiq tablet to trace coral colonies. Colonies were labeled and assigned a specimen number (Figure 7). For example, all *Orbicella franksi* colonies identified within a photo were outlined in orange, labeled, and numbered from the top left to the bottom right of each photograph. Numbering allowed for a count of total specimens within each of the images.

Potential 'unknown' species, or those that could not be identified because of shadow or camera angle, were outlined in red, labeled as an 'unidentified' or to the lowest possible taxon, and logged into an excel sheet to track them for future ground truthing. *Orbicella* colonies that potentially could be hybrids of *Orbicella franksi, Orbicella faveolata,* and *Orbicella annularis* were labeled *Orbicella* species (Figure 8). Species confirmation of *Orbicella* spp. colonies were done by ground truthing by the FGBNMS dive team during the 2018 field season. After ground truthing, coral identification keys and the excel sheet were updated to show the correct identification of the flagged coral.

Benthic Quantification

Corals and other benthic organisms in repetitive photostation images were identified and quantified using CPCe. These data were used to calculate coral and benthic coverage for a 27-

year time-series analysis of East and West FGB from 1989-2017. A project-specific code file with a complete list of identification labels (Table S2) was used for all CPCe analyses. This list contains labels for all corals that can be found at the FGBs, as well as categories for sponge, macroalgae, and composite substrate for crustose coralline algae (CCA), turf algae, and bare rock, among others (Aronson et al., 2005; Johnston et al., 2013, 2021). Each image was first uploaded into CPCe and 100 random points were generated across each image (Fig. 9). The object underneath each point was assigned to one of the identification labels. All identifications were quality assessed and quality checked for misidentifications by a FGBNMS scientist. This analysis was done for each image taken over the last 27 years for the 37 photostations at East FGB and 41 at West FGB. CPCe files were exported to Excel as a comma separated file (.csv).

Statistical Analysis

Coral Abundance

The dataset that was originally exported from CPCe was condensed to focus on major taxa. Corals with 10 or more observations across the 27 y at East FGB or at West FGB were kept as individual species categories (*Millepora alcicornis, Montastarea cavernosa, Mussa angulosa, Orbicella annularis, Orbicella Pseudodiploria strigosa, Orbicella franksi, Orbicella* species, *Pseudodiploria strigosa,* and *Stephanocoenia intersepta*). Groups at species level that were not represented by 10 or more observations were grouped together by genus if their combined observations for that group (e.g., *Millepora* spp) were ≥ 10. For those species with < 10 observations for the genus (e.g., *Scolymia cubensis, Rhizosmilia maculata, Eusimilia* spp, *Dichocoenia stokesi,* and *Tubastraea coccinea*), they were assigned to the category 'Other Identified Corals'. As a result, the dataset was condensed from 33 (Table S2) to 16 coral groups (Table S2). Data were condensed because numerous groups had less than 100 observations across the ~2000 images analyzed, with 17 species categories having less than 10 observations. Because analysis was based on the analysis of overall coral coverage at East and West FGBs, condensing would not affect the ability to identify trends in coral coverage.

For each bank, three datasets were created from the raw CPCe output: bank-averaged coral data (averaged over all photostations and years per bank), year-averaged coral data (averaged over photostations per year for each bank), and photostation-averaged coral data (averaged over years for each photostation for each bank).

Bank-Averaged Coral Dataset, East and West FGB

To visualize the contribution of each coral species or genus to the bank-averaged coral dataset, shade plots were created (Primer-E version 7.0 software). Based on the results of the shade plots, a fourth-root transformation was performed on the raw CPCe bank-averaged coral dataset for both East and West FGB to correct for overrepresentation of highly abundant species categories. To visualize the similarities among coral coverage at the 37 photostations at East FGB and 41 photostations at West FGB, a non-metric multidimensional scaling analysis (nMDS) based on a Bray-Curtis similarity matrix was performed with the following settings: 100 bootstrap averages, minimum rho of 0.99, photostations as a factor, and 95% bootstrap regions. To determine significant long-term trends in mean percentage coral cover at East and West FGB, a Mann-Kendall trend test in R (version 4.0.4) was conducted using the package “Kendall” and the command “MannKendall”.

To investigate if initial total percentage cover (i.e., low vs. high) of a photostation affected the overall rate of coral cover increase or decrease at a photostation, a linear regression was run using categorized data from initial photostation total percentage cover. Initial total percentage cover (first appearance within the dataset) for each photostation were calculated and photostations were categorized based on their initial total percentage cover rank within the dataset to one of three categories (Table 1): high (above 75th percentile), medium (between the 25th and 75th percentile), and low (in 25th percentile). Once photostations were categorized by initial total percentage cover, all total percentage cover data for photostations were imported into JMP (16.0.0) for linear regression analysis by bank with year as the predictor variable and percentage cover as the response variable.

A two-way permutational MANOVA (PERMANOVA) was conducted on East and West FGB dataset to determine the significance of the relationship of year and bank to coral coverage. The PERMANOVA was run on the fourth-root overall dataset for East and West FGB with the following parameters: Type III (Sums of squares type: partial), fixed effects sum to zero for mixed terms, permutation of residuals under the full model, 999 permutations, and fixed factors year and bank.

Year-Averaged and Photostation-Averaged Coral Datasets, East and West FGB

Based on shade plots, it was determined that a fourth-root transformation would provide the best representation of the photostation coral community. A Bray-Curtis similarity matrix was constructed from the datasets containing mean coral cover by year and mean coral cover by photostation for each bank. A Cluster analysis with SIMPROF tests was performed on the year-averaged and photostation-averaged datasets, to identify *a posteriori* year and photostation groupings for subsequent analyses. Species contributions (SIMPER) analysis was run with a 70% cutoff for low contributions on “year” groups for each bank to determine which fauna were the greatest contributors to the similarities and differences within and among years; “photostation” as a factor was not further considered because no significant clusters were identified for either bank. A nMDS analysis was conducted for year-averaged and photostation-averaged datasets. Year-averaged coral dataset nMDS was performed from the similarity matrix for that dataset with 100 restarts, minimum stress of 0.01, and year trajectories to visualize the relationship in coral coverage over time for each of the banks, as well as SIMPROF cluster overlays to identify significant groups. Photostation-averaged coral datasets nMDS was conducted with the same parameters that were used on the year-averaged dataset, however instead of the utilization of trajectories, a cluster overlay was utilized (60%). A bar plot was created for each dataset within PRIMER to further visualize species contribution to overall coral coverage based on year and photostation. PERMANOVA analysis on the fourth root transformed year-averaged dataset was conducted with the following parameters: Type III (sums of squares type: partial), fixed effects sum to zero for mixed terms, permutation of residuals under the full model, fixed factor year, and 999 permutations. These same parameters were utilized for the fourth root transformed photostation-averaged dataset, except the fixed factor examined was photostation.

Table 1: Distribution of the first recorded percentage cover from photostations at East Flower Garden Bank and West Flower Garden Bank used to categorize all repetitive photostation percentage coral coverage data for all years (1989-2017).

Bank	Minimum	25^{th} percentile	50^{th} percentile	75^{th} percentile	Maximum
East FGB	16.49	46.355	58.76	70.67	87.63
West FGB	24.73	42.965	54.35	67.01	83.16

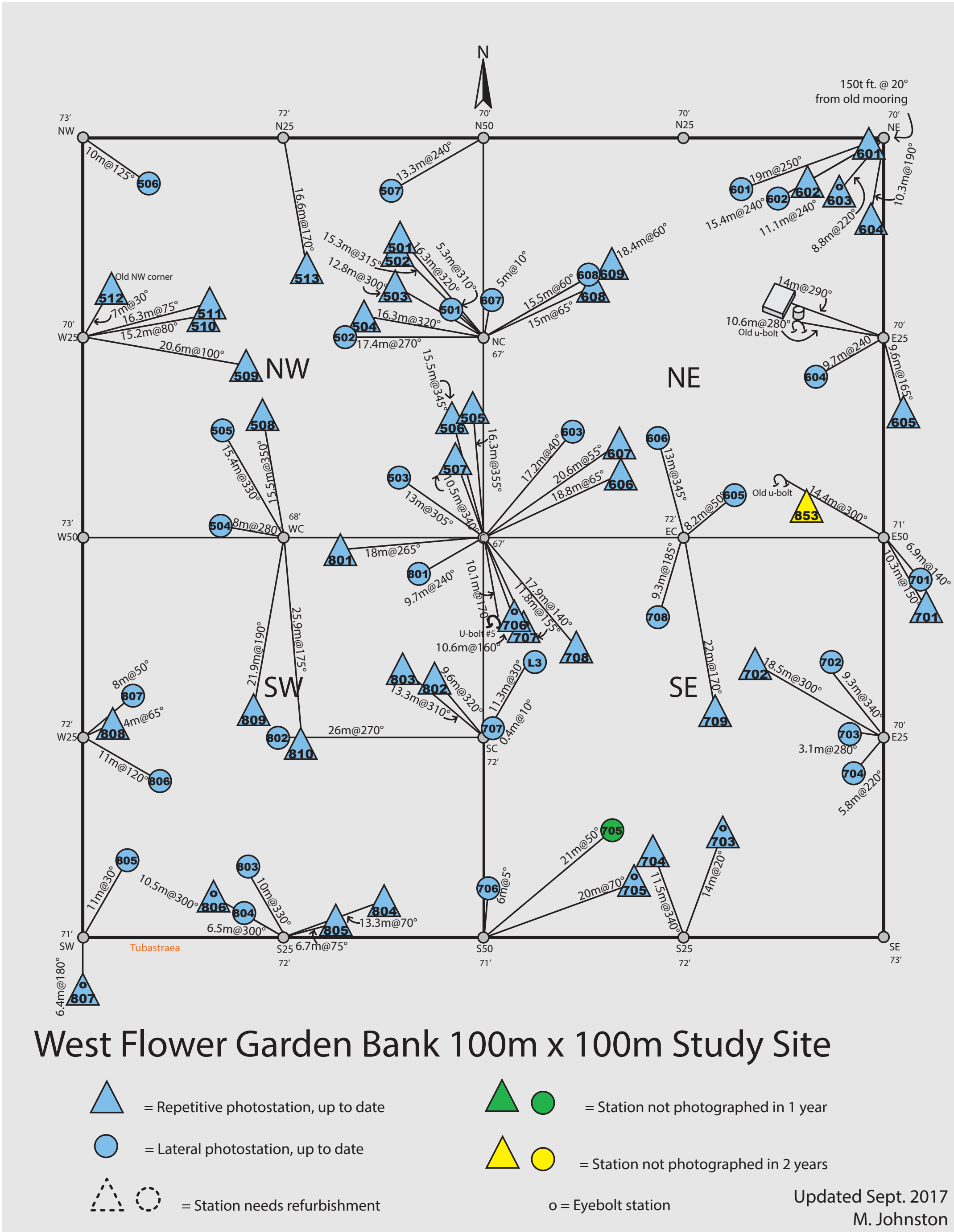

Figure 5: West Flower Garden Bank study site (100 m x 100 m) depicting the locations of repetitive photostations (From Johnston et al. 2017).

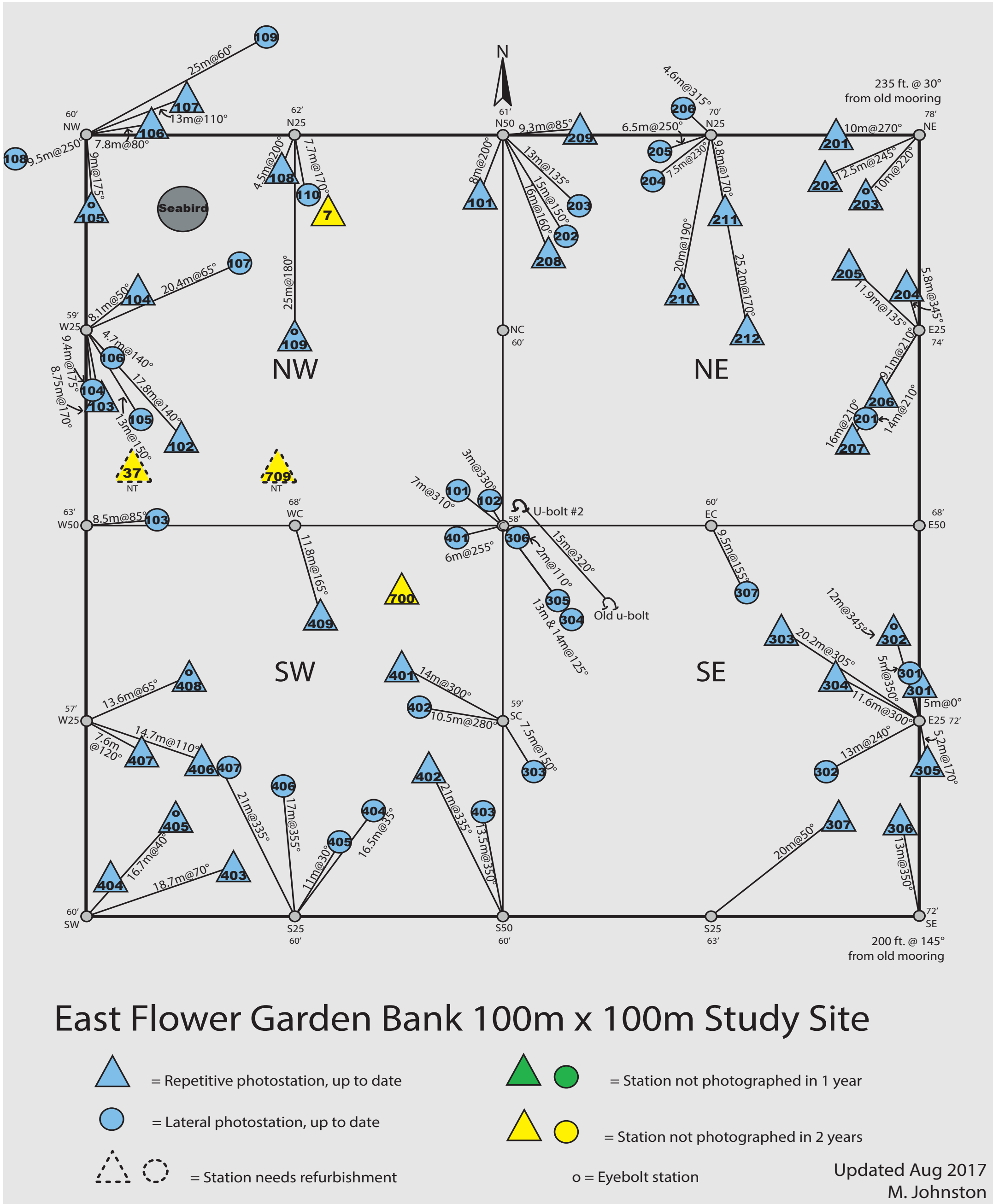

Figure 6: East Flower Garden Bank study site (100 m x 100 m) depicting the locations of repetitive photostations (From Johnston et al. 2017).

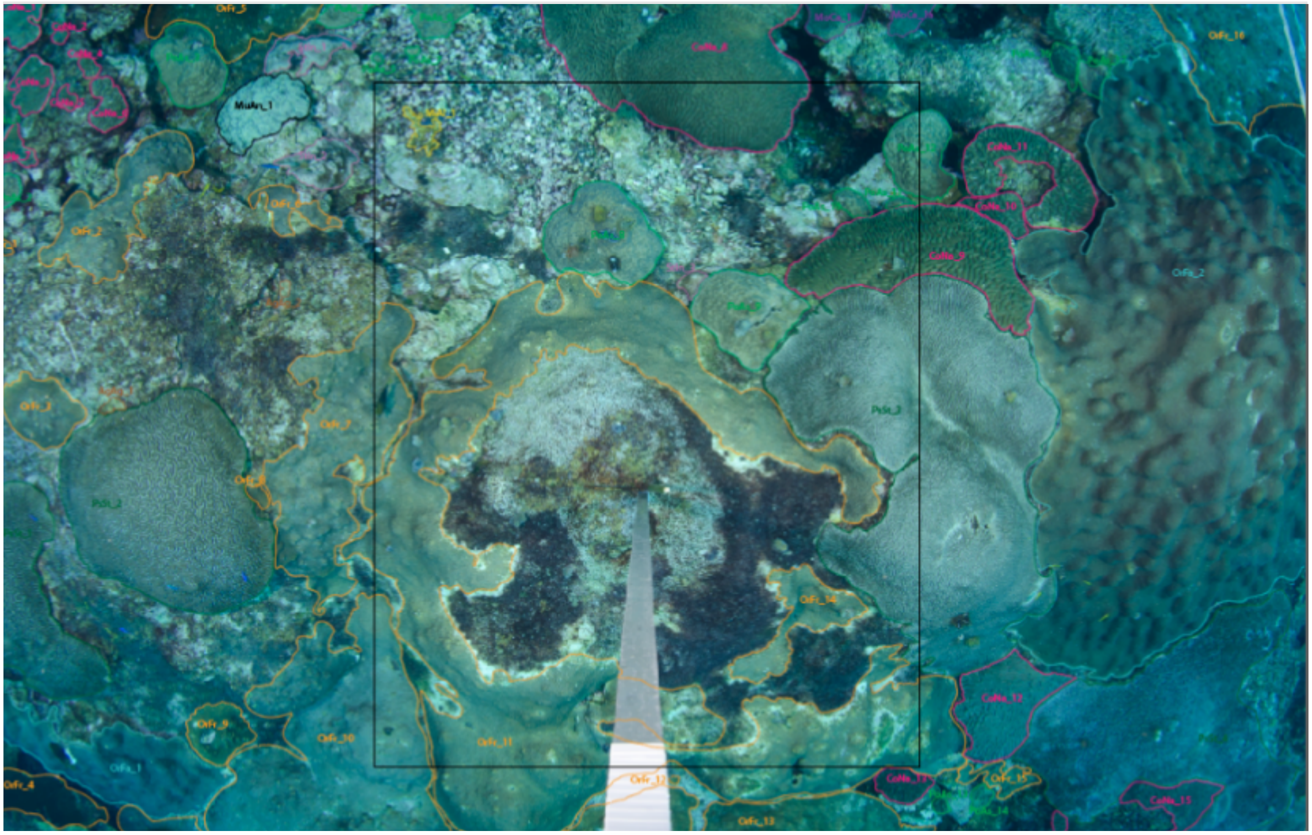

Figure 7: Example of outlining, identifying, and labeling coral species using Adobe Illustrator (AI) (ver. 23.0.2.567). This 2015 photo is from East Flower Garden Bank, repetitive photostation #206.

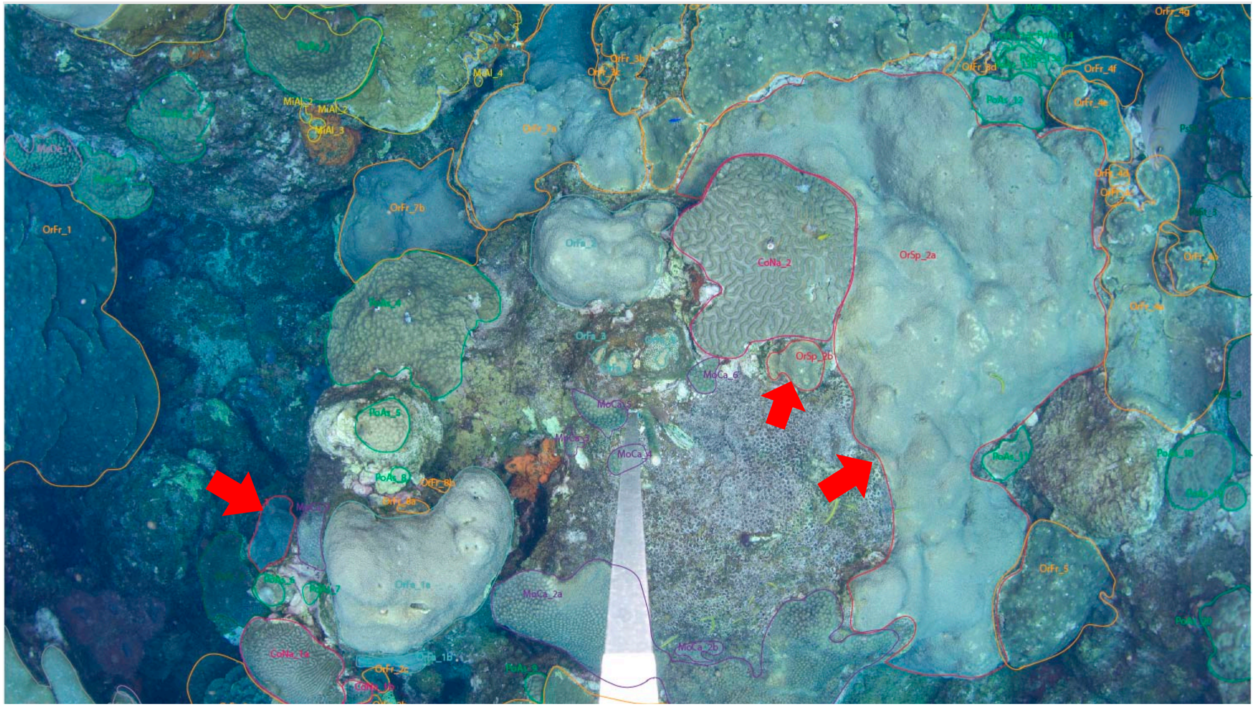

Figure 8: An example from Adobe Illustrator of the potential 'unidentified' *Orbicella* species that are labeled *Orbicella spp* and outlined in red. The bright red arrows identify the potential 'unidentified' *Orbicella spp* colonies.

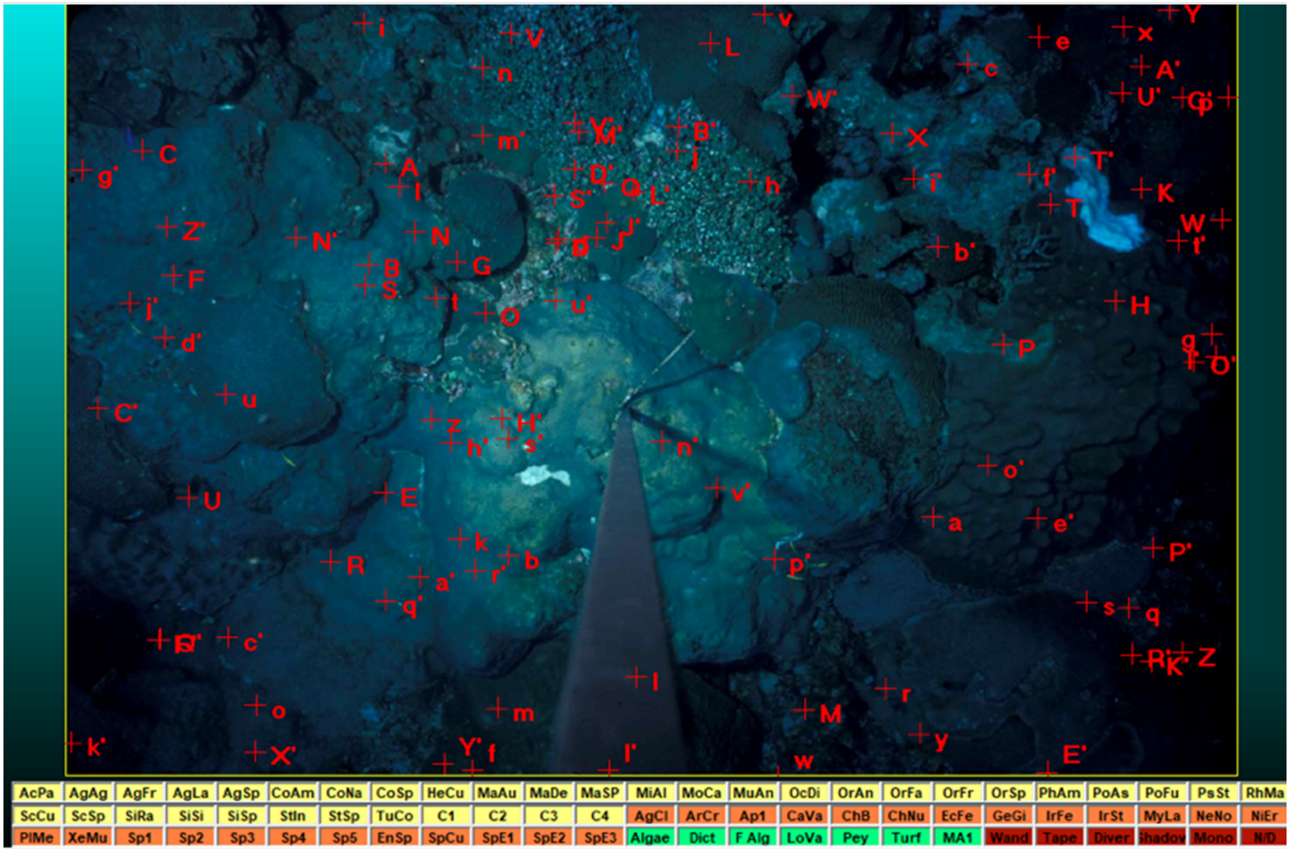

Figure 9: A 2002 photo from repetitive photostation #206 from East Flower Garden Bank with 100 random points (red letters) assigned to coral or substrate category in Coral Point Count. Percentage cover was determined by tallying the number of points falling on each species as a percentage of the total 100 points (Table S2).

CHAPTER III

RESULTS

Bank-Averaged Coral Dataset, East and West FGB

Sixteen coral species groups were observed at repetitive photostations from the condensed dataset at East FGB and West FGB. *Orbicella franksi* had the highest cover at East FGB (33.06% ± 0.678 SE), followed by *Psuedodiploria strigosa* (8.02% ± 0.316) (Figure 10). Bank-averaged percentage coral cover at East FGB exhibited an apparent increase in mean percentage coral cover over 27 y (1989-2017) (Figure 11 and 12), but the trend was not significant (Mann-Kendall: tau = 0.121, 2-sided p-value = 0.36783; Figure 13a). The bootstrap-averaged nMDS (Fig. 14) revealed that the photostation coral communities at East FGB are similar to one another with low intrabank variability indicating similar photostation coral communities across the 1-hectare study area. There was a significant increase in total percentage coral cover across years among photostations with low initial total cover at EFGB (linear regression: y = -601.48 + 0.3257*Year, R^2= 0.0248 P = 0.0219; Table 2) with a predicted 0.32 percentage point increase (e.g., 55.00 to 55.32%) in cover each year. No significant pattern was observed in repetitive photostations that displayed a medium initial total percentage coral coverage (linear regression: y= -6.832 + 0.03567*Year, R^2=0.00063 P= 0.6067; Table 2). At photostations with a high initial total percentage coral cover, a significant decrease was observed in percentage coral cover each year (linear regression: y = 393.92 – 0.1595*Year, R^2=0.0225 P=0.0389;Table 2) with a predicted 0.15 percentage point decrease in total coral cover each year.

Orbicella franksi had the highest percentage cover at WFGB photostations (31.25% $\pm$ 0.502 SE) followed by *Pseudodiploria strigosa* (7.69% $\pm$ 0.262 SE) (Figure 10). Bank-averaged coral percentage cover at West FGB displayed an increase in mean total percentage coral cover from repetitive photostations over a 27-year period (1989-2017) (Figure 11 & 12). Mean total percentage coral cover from repetitive photostations from 1989 to 2017 at WFGB ranged from 50.82-61.67%, displaying a significant increasing trend (Mann-Kendall: tau = 0.54, 2-sided pvalue = 4.3154 e-05) (Figure 13b). The bootstrap-averaged nMDS (Figure 15) revealed that the photostation coral communities at West FGB are similar to one another with low intrabank variability indicating similar photostation coral communities across the 1-hectare study area. There was a significant increase in total percentage coral cover across years among photostations with low initial total percentage coral cover (linear regression: y = -1393.09 + 0.7196*Year, R^2=0.1866 P= <0.0001; Table 3) with a predicted coral cover increase of 0.71 percentage points per year. A significant increase was observed in percentage total coral cover each year among photostations that displayed medium initial total cover (linear regression: y = -487.75 + 0.2725*Year, R^2=0.0489 P=<0.0001; Table 3). These stations show a predicted total coral cover increase of 0.27 percentage points per year. There was no trend observed in percentage total coral cover each year among photostations with high initial percentage total coral cover (linear regression: y = -119.37 + 0.0939*Year, R^2=0.005764 P=0.2781; Table 3).

PERMANOVA analysis revealed that bank and year have a significant effect on coral cover with no significant interaction between bank and year at East and West FGB (Table 4).

Year-Averaged Coral Dataset, East and West FGB

Coral communities among 37 photostations at EFGB were similar to one another in terms of coral coverage (Table 7 & 9, Figure 16 & 17). CLUSTER analysis resulted in three significant clusters based on year: Cluster A (1995, 1996, 1997, 1998, 2001, 2002), Cluster B (1989, 1990, 1991, 1992, 1994, 1999, 2000, 2003, 2004, 2005, 2006, 2007, 2008, 2010), and Cluster C (2009, 2011, 2012, 2013, 2014, 2015, 2016, 2017). The SIMPER analysis indicated that average similarity and dissimilarity within significant year cluster groups at East FGB was primarily due to *Orbicella franksi* (Table 7). The most reoccurring species within Clusters A (average similarity: 93.79) and B (average similarity: 93.40) included, *Orbicella franksi, Orbicella faveolata,* and *Pseudodiploria strigosa. Orbicella franksi* contributed 32.6% - 33.73% to coral cover (14.26% - 17.35% contribution to similarity), *Orbicella faveolata* contributed 6.71% - 6.07% to coral cover (9.6% - 6.89% contribution to similarity), and *Pseudodiploria strigosa* contributed 6.39% - 6.07% to coral cover (9.43% - 11.49% contribution to similarity). The most reoccurring species within Cluster C (average similarity: 91.48) included *Orbicella franksi, Pseudodiploria strigosa* and *Porites spp*. *Orbicella franksi* contributed 32.08% to coral cover (15.90% contribution to similarity), *Pseudodiploria strigosa* contributed 10.26% to coral cover (11.91% contribution to similarity) and *Porites spp* contributed 4.54% to coral cover (9.70% contribution to similarity) (Table 7, Figure 16 & 17). Differences between year clusters were due primarily to the high contribution of *Mussa angulosa, Orbicella annularis, Orbicella spp, Millepora alcicornis,* and Other Identified Coral to overall coral coverage at East FGB (Table 7). The nMDS analysis of the year-averaged dataset with overlayed year trajectories showed the similarity among years and direction of change of coral coverage with the three clusters resulting from the Cluster analysis with SIMPROF test being well separated (Figure 17). Figure 16 displays coral species that carried the most weight in overall coral coverage among years at East FGB. The most reoccurring species identified within the bar plot are *Orbicella franksi, Pseudodiploria strigosa, Orbicella faveolata,* and *Porites spp,* which are the species identified within the SIMPER analysis as contributing the most to similarities within year clusters (Table 7; Figure 17). PERMANOVA analysis revealed, clustering by year has a significant relationship with coral coverage among years at East FGB study sites (P = 0.001; Table 5).

Coral communities among 41 photostations at West FGB were similar to one another in terms of overall coral coverage and the species that drove the similarity. Cluster analysis on the bank-averaged coral dataset for WFGB resulted in seven significant clusters: Cluster A (2004), Cluster B (2002), Cluster C (1991, 1992, 1994, 1995, 1996, 1997, 1998, 1999. 2001, 2003, 2005, 2006, 2007, 2008, 2009, 2010, 2011), Cluster D (2012), Cluster E (1989, 1990, 2000), Cluster F (2013), and Cluster G (2014, 2015, 2016, 2017). The SIMPER analysis indicated that average similarity and dissimilarity within significant year cluster groups at West FGB was primarily due to *Orbicella franksi* (Table 8). The most reoccurring species within Clusters C (average similarity: 94.93) and E (average similarity: 95.55) included *Orbicella franksi, Pseudodiploria strigosa,* and *Orbicella faveolata* (Table 8). *Orbicella franksi* had an average percent cover of 54.73% - 53.14% (15.96% - 14.81% contribution to similarity), *Pseudodiploria strigosa* had an average percent cover of 13.30% - 13.30% (11.05% - 10.37% contribution to similarity) and *Orbicella faveolata* had an average percent cover of 6.55% - 7.05% (9.19% - 8.43% contribution to similarity). The most reoccurring species within Cluster G (average similarity: 96.78) included *Orbicella franksi, Pseudodiploria strigosa,* and *Porites spp. Orbicella franksi* contribute 50.06% to coral cover (14.01% contribution to similarity), *Pseudodiploria strigosa* contribute 13.30% to coral cover (9.98% contribution to similarity) and *Porites spp* contribute 7.59% to coral cover (8.75% contribution to similarity) (Table 8). Species driving dissimilarities within these clusters are *Orbicella spp, Agaricia spp, Mussa angulosa, Madracis spp, Siderastraea spp, Stephanocoenia intersepta, Orbicella faveolate, Millepora alcicornis*, Unknown Coral and Other

Identified Coral to overall coral coverage at West FGB (Table 8). The nMDS analysis of the year-averaged dataset with overlayed year trajectories showed the similarity among years and direction of change of coral coverage with the three clusters resulting from the Cluster analysis with SIMPROF test being well separated (Figure 19). The most reoccurring species at West FGB are *Orbicella franksi, Pseudodiploria strigosa, Porites* spp., and *Montastraea cavernosa*, which were the species identified within the SIMPER analysis as contributing the most to similarity within groups (Table 8; Figure 19). PERMANOVA analysis revealed, clustering by year has a significant relationship with coral coverage among years at West FGB study area (P = 0.001; Table 6).

Photostation-Averaged Coral Dataset, East and West FGB

The Cluster analysis with SIMPROF test found no significant clustering of photostations (Figure 20). The SIMPER analysis showed that the photostation-averaged dataset coral communities at East FGB are similar to one another in terms of coral coverage across the study area (average similarity: 75.26) (Table 9). *Orbicella franksi* carried the most weight in overall coral coverage at East FGB (Table 9). The most species contributing the most to similarity within repetitive photostations were *Orbicella franksi, Pseudodiploria strigosa* and *Porites* spp. (Table 9). Likewise, the nMDS analysis of the photostation-averaged coral dataset with overlayed clusters (80%) showed the similarities among photostations across the study area (Figure 21).

West FGB SIMPER analysis displayed similar results as to East FGB (Figure 22). The SIMPER analysis showed that photostation-averaged coral dataset coral communities at West FGB are similar to one another in terms of coral coverage across the study area (similarity: 74.55) (Table 10). *Orbicella franksi* was the species that drove similarity among photostations and was the species that carried the most weight in overall coral coverage (Table 10). The most common species contributing the most to similarity within repetitive photostations were *Orbicella franksi, Pseudodiploria strigosa,* and *Porites spp.* Like East FGB, the nMDS analysis of the photostation-averaged coral dataset with overlayed clusters (80%) displayed similarities among photostations across the study area (Figure 23).

Table 2: Linear regression analysis output for initial coral coverage at East Flower Garden Bank repetitive photostations.

Low (15% - 46%)	
Linear Fit Rsquare P Sample Size	y = -601 + 0.3205*Year 0.0248 0.0219 9
Medium (47% - 70%)	
Linear Fit Rsquare P Sample Size	y = -6.832 + 0.0357*Year 0.00063 0.6067 18
High (71%- 100%)	
Linear Fit Rsquare P Sample Size	y = 393.92 - 0.1595*Year 0.0225 0.0389 10

Table 3: Linear regression analysis output for initial coral coverage at West Flower Garden Bank repetitive photostations.

Low (23% - 42%)	
Linear Fit	y = -1393.09 + 0.7196*Year
Rsquare	0.1866
P	<.0001
Sample Size	10
Medium (43% - 66%)	
Linear Fit	y = -487.75 + 0.2725*Year
Rsquare	0.0489
P	<.0001
Sample Size	21
High (67% - 100%)	
Linear Fit	y = -119.36 + 0.0939*Year
Rsquare	0.0057
P	0.2781
Sample Size	10

Table 4: Two-way PERMANOVA analysis for percentage coral cover, comparison of repetitive photostations at East and West Flower Garden Bank with bank and year as fixed factors, including degrees of freedom (df), sum of squares (SS), mean squares (MS) and number of unique permutations (perms).

Source	df	SS	MS	Pseudo-F	P (perm)	Unique Permutations	P (MC)
Bank	1	29075	29075	33.144	0.001	998	0.001
Year	27	77590	2873.7	3.2759	0.001	996	0.001
Bank x Year	27	1608	689.18	0.78563	0.969	996	0.958
Residual	1701	1.49E+06	877.23				
Total	1756	1.62E+06					

Table 5: PERMANOVA analysis for percentage coral cover among years at East Flower Garden Bank repetitive photostations with year as the fixed factor, including degrees of freedom (df), sum of squares (SS), mean square (MS) and number of unique permutations (perms).

Source	df	SS	MS	Pseudo-F	p(perm)	unique perms	p(MC)
cluster_year	2	914.46	457.23	16.481	0.001	999	0.001
Residuals	25	693.59	27.744				
Total	27	1608.1					

Table 6: PERMANOVA analysis for percentage coral cover among years at West Flower Garden Bank repetitive photostations with year as the fixed factor, including degrees of freedom (df), sum of squares (SS), mean square (MS) and number of unique permutations (perms).

Source	df	SS	MS	Pseudo-F	P(perm)	unique perms	p(MC)
cluster_year	6	620.56	103.43	8.44	0.001	998	0.001
Residuals	21	257.34	12.254				
Total	27	877.9					

Table 7: One-way analysis of similarity (SIMPER) with a 70% cut off for low contributing species to coral community structure at East Flower Garden Bank based on year, including average abundance (Av. Abund), average similarity (Av. Sim), similarity standard deviation (Sim/SD), percent contribution (Contrib%), cumulative contribution (Cum.%), average dissimilarity (Av. Diss), and dissimilarity standard deviation (Diss/SD).

Group A, average similarity: 93.79

Group A (1995, 1996, 1997, 1998, 2001, 2002)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.39	13.38	40.68	14.26	14.26
Orbicella faveolata	1.61	9.01	61.72	9.6	23.87
Pseudodiploria strigosa	1.59	8.84	23.14	9.43	33.3

Group B, average similarity: 93.40

Group B (1989, 1990, 1991, 1992, 1994, 1999, 2000, 2003, 2004, 2005, 2006, 2007, 2008, 2010)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.41	16.21	33.82	17.35	17.35
Pseudodiploria strigosa	1.62	10.73	22.6	11.49	28.84
Orbicella faveolata	1.57	10.46	27.59	11.19	40.04

Group C, average similarity: 91.48

Group C (2009, 2011, 2012, 2013, 2014, 2015, 2016, 2017)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.38	14.55	21.09	15.9	15.9
Pseudodiploria strigosa	1.79	10.9	27.64	11.91	27.82
Porites spp	1.46	8.87	17.27	9.7	37.51

Groups B & A, average dissimilarity: 12.54

Species	Group B Av. Abund.	Group A Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Other identified coral	0	1.05	3.25	16.61	25.94	25.94
Mussa angulosa	0.32	1.22	2.79	3.5	22.21	48.15
Orbicella annularis	0.77	0.74	0.87	0.89	6.91	55.05

Groups B & C, average dissimilarity: 11.16

Species	Group B Av. Abund.	Group C Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Siderastraea spp	0	0.59	1.91	4.5	17.15	17.15
Orbicella spp	0.57	0.93	1.64	1.67	14.71	31.86
Millepora alcornis	0.99	0.69	1.05	1.6	9.37	41.24

Groups A & C, average dissimilarity: 13.69

Species	Group A Av. Abund	Group C Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Mussa angulosa	1.22	0.38	2.5	3.62	18.29	18.29
Other identified coral	1.05	0.24	2.44	2.34	17.8	36.09
Orbicella spp	0.67	0.93	1.29	1.62	9.45	45.55

Table 8: One-way analysis of similarity (SIMPER) with a 70% cut off for low contributing species to coral community structure at West Flower Garden Bank based on year, including average abundance (Av. Abund), average similarity (Av. Sim), similarity standard deviation (Sim/SD), percent contribution (Contrib%), cumulative contribution (Cum.%), average dissimilarity (Av. Diss), and dissimilarity standard deviation (Diss/SD).

Group A (2004) Less than 2 samples in group	
Group B (2002) Less than 2 samples in the group	

Group C, average similarity: 94.93

Group C (1991, 1992, 1994, 1995, 1996, 1997, 1998, 1999, 2001, 2003, 2005, 2006, 2007, 2008, 2009, 2010, 2011)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum. %
Orbicella franksi	2.72	15.15	44.31	15.96	15.96
Pseudodiploria strigosa	1.91	10.49	36.31	11.05	27.01
Orbicella faveolata	1.6	8.72	22.35	9.19	36.2

Group D (2012) Less than 2 samples in the group	

Group E, average similarity: 95.55

Group E (1989, 1990, 2000)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.7	14.15	201.54	14.81	14.81
Pseudodiploria strigosa	1.91	9.91	59.8	10.37	25.18
Montastraea cavernosa	1.63	8.06	9.3	8.43	33.61

Group F (2013) Less than 2 samples in the group	

Group G, average similarity: 96.78

Group G (2014, 2015, 2016, 2017)					
Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.66	13.56	71.79	14.01	14.01
Pseudodiploria strigosa	1.91	9.66	51.26	9.98	23.99
Porites Sp	1.66	8.46	60.81	8.75	32.74

Groups E & C, average dissimilarity: 8.96

Species	Group E Av. Abund	Group C Av. Abund	Av. Diss	Diss/Sd	Contrib%	Cum.%
Orbicella spp	1.27	0.07	3.27	4.42	36.44	36.44
Agaricia spp	0	0.45	1.21	1.46	13.51	49.94
Mussa angulosa	0.79	0.57	0.6	1.26	6.72	56.67

Groups E & B, average dissimilarity: 9.50

Species	Group E Av. Abund	Group B Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Mussa angulosa	0.79	0	2.2	20.47	23.12	23.12
Madracis spp	0.71	0	1.98	126.09	20.86	43.98
Orbicella spp	1.27	0.65	1.75	2.82	18.4	62.38

Groups C & B, average dissimilarity: 9.51

Species	Group C Av. Abund	Group B Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Madracis spp	0.69	0	1.99	7.12	20.94	20.94
Orbicella spp.	0.07	0.65	1.67	2.86	17.54	38.48
Mussa angulosa	0.57	0	1.63	3.27	17.13	55.61

Groups E & A, average dissimilarity: 9.67

Species	Group E Av. Abund	Group A Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Siderastea spp	1.14	0	3.18	10.78	32.95	32.95
Orbicella spp	1.27	0.67	1.68	2.72	17.36	50.3
Stephanocoenia intersepta	1.07	0.75	0.9	8.29	9.35	59.66

Groups C & A, average dissimilarity: 10.83

Species	Group C Av. Abund	Group A Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Siderastrea spp	1.12	0	3.23	17.1	29.86	29.86
Orbicella spp	0.07	0.67	1.73	2.97	15.98	45.83
Agaricia spp	0.45	0	1.28	1.43	11.85	57.68

Groups B & A, average dissimilarity: 11.90

Species	Group B Av. Abund	Group A Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Siderastrea spp	1.15	0	3.4	Undefined!	28.6	28.6
Mussa angulosa	0	0.84	2.51	Undefined!	21.06	49.66
Madracis spp	0	0.8	2.37	Undefined!	19.95	69.61

Groups E & D, average dissimilarity: 8.91

Species	Group E Av. Abund	Group D Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	1.27	1.83	1.42	2.51	15.91	15.91
Agaricia spp	0	0.54	1.37	419.06	15.36	31.3
Other Identified Coral	0	0.46	1.18	419.06	13.21	44.51

Groups C & D, average dissimilarity: 12.16

Species	Group C Av. Abund	Group D Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.07	1.83	4.62	8.39	38.03	38.03
Unknown Coral	0.91	1.44	1.4	3.25	11.51	49.54
Other Identified Coral	0	0.46	1.21	81.85	9.97	59.51

Groups B & D, average dissimilarity: 14.32

Species	Group B Av. Abund	Group D Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.65	1.83	3.19	Undefined!	22.3	22.3
Madracis spp	0	0.77	2.07	Undefined!	14.47	36.78
Mussa angulosa	0	0.64	1.72	Undefined!	12.01	48.79

Groups A & D, average dissimilarity: 15.39

Species	Group A Av. Abund	Group D Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.67	1.83	3.12	Undefined!	20.29	20.29
Siderastrea spp	0	1.11	3	Undefined!	19.53	39.82
Agaricia spp	0	0.54	1.45	Undefined!	9.41	49.23

Groups E & F, average dissimilarity: 8.85

Species	Group E Av. Abund	Group F Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Agaricia spp	0	0.76	2.01	406.44	22.66	22.66
Orbicella faveolata	1.4	0.86	1.41	3.03	15.99	38.65
Orbicella spp	1.27	1.72	1.16	1.99	13.1	51.75

Groups C & F, average dissimilarity: 10.98

Species	Group C Av. Abund	Group F Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.07	1.72	4.46	7.86	40.65	40.65
Orbicella faveolata	1.6	0.86	2	10.03	18.21	58.85
Agaricia spp	0.45	0.76	0.9	1.1	8.18	67.03

Groups B & F, average dissimilarity: 14.42

Species	Group B Av. Abund	Group F Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.65	1.72	2.98	Undefined!	20.66	20.66
Agaricia spp	0	0.76	2.13	Undefined!	14.77	35.43
Orbicella faveolata	1.62	0.86	2.1	Undefined!	14.57	50

Groups A & F, average dissimilarity: 14.49

Species	Group A Av. Abund	Group B Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum%
Orbicella spp	0.67	1.72	2.9	Undefined!	19.91	19.91
Siderastrea spp	0	0.96	2.66	Undefined!	18.23	38.14
Orbicella faveolata	1.7	0.86	2.32	Undefined!	15.91	54.06

Groups D & F, average dissimilarity: 7.52

Species	Group D Av. Abund	Group F Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Unknown Coral	1.44	0.93	1.31	Undefined!	17.36	17.36
Other Identified Coral	0.46	0	1.18	Undefined!	15.65	33
Orbicella faveolata	1.28	0.86	1.05	Undefined!	14.03	47.03

Groups E & G, average dissimilarity: 6.92

Species	Group E Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Agaricia spp	0	0.75	1.95	9.29	28.16	28.16
Millepora alcicornis	1.09	0.83	0.68	3.02	9.76	37.92
Orbicella spp	1.27	1.35	0.57	1.51	8.21	46.12

Groups C & G, average dissimilarity: 7.74

Species	Group C Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella spp	0.07	1.35	3.43	4.85	44.36	44.36
Agaricia spp	0.45	0.75	0.88	1.11	11.33	55.68
Millepora alcicornis	0.99	0.83	0.5	1.4	6.41	62.09

Groups B & G, average dissimilarity: 10.53

Species	Group B Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Agaricia spp	0	0.75	2.07	8.39	19.65	19.65
Orbicella spp	0.65	1.35	1.94	3.6	18.39	38.04
Madracis spp	0	0.68	1.88	16.14	17.84	55.88

Groups A & G, average dissimilarity: 11.73

Species	Group A Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Siderastea spp	0	1.01	3.12	26.3	26.62	26.62
Agaricia spp	0	0.75	2.06	8.39	17.57	44.19
Orbicella spp	0.67	1.35	1.87	3.48	15.9	60.09

Groups D & G, average dissimilarity: 7.35

Species	Group D Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Unknown Coral	1.44	0.93	1.28	7.94	17.47	17.47
Orbicella spp	1.83	1.35	1.21	2.46	16.51	33.98
Other Identified Coral	0.46	0	1.16	439.87	15.85	49.83

Groups F & G, average dissimilarity: 5.39

Species	Group F Av. Abund	Group G Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Orbicella faveolata	0.86	1.51	1.69	7.2	31.44	31.44
Orbicella spp	1.72	1.35	0.95	1.87	17.68	49.11
Stephanocoenia intersepta	1.14	0.95	0.49	3.5	9.12	58.23

Table 9: One-way SIMPER with a 70% cut off for low contributing coral species based on 37 repetitive photostations at East Flower Garden Bank including average abundance (Av. Abund), average similarity (Av. Sim), similarity standard deviation (Sim/SD), percent contribution (Contrib%) and cumulative contribution (Cum.%).

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.31	16.3	5.39	21.65	21.65
Pseudodiploria strigosa	1.58	10.3	4.66	13.69	35.34
Porites spp	1.4	9.92	6.7	13.19	48.53

Table 10: One-way SIMPER with a 70% cut off for low contributing coral species based on 41 repetitive photostations at West Flower Garden Bank including average abundance (Av. Abund), average similarity (Av. Sim), similarity standard deviation (Sim/SD), percent contribution (Contrib%) and cumulative contribution (Cum.%).

Species	Av. Abund	Av. Sim	Sim/SD	Contrib%	Cum.%
Orbicella franksi	2.64	16.94	5.37	22.72	22.72
Pseudodiploria strigosa	1.78	10.4	3.38	13.95	36.68
Porites spp	1.56	9.55	9.33	12.81	49.49

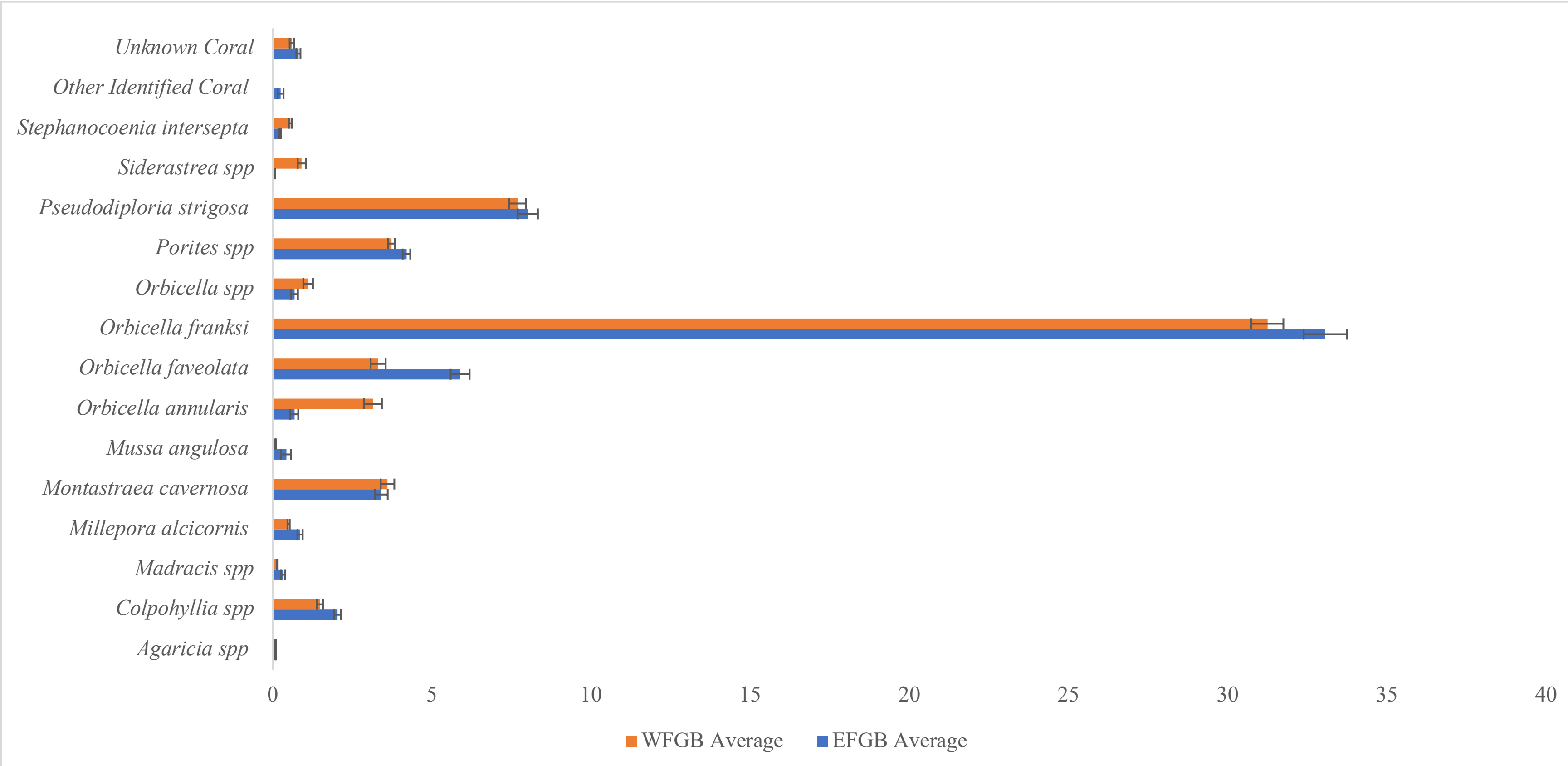

Figure 10: Averaged coral species coverage (± SE) from repetitive photostations at East and West Flower Garden Banks from 1989-2017.

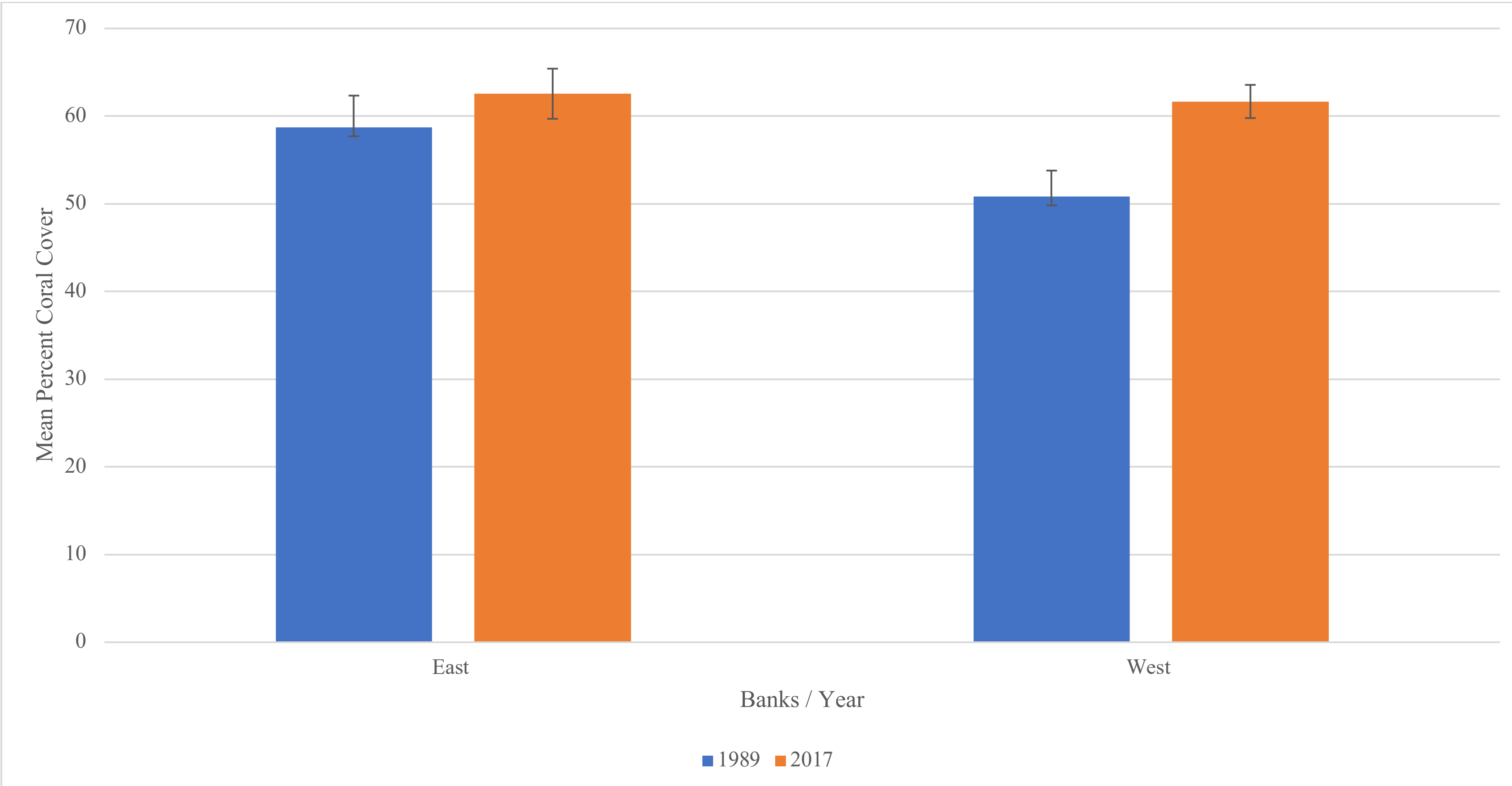

Figure 11: Mean percentage coral cover (± SE) from repetitive photostations at East and West Flower Garden Banks for 1989 and 2017.

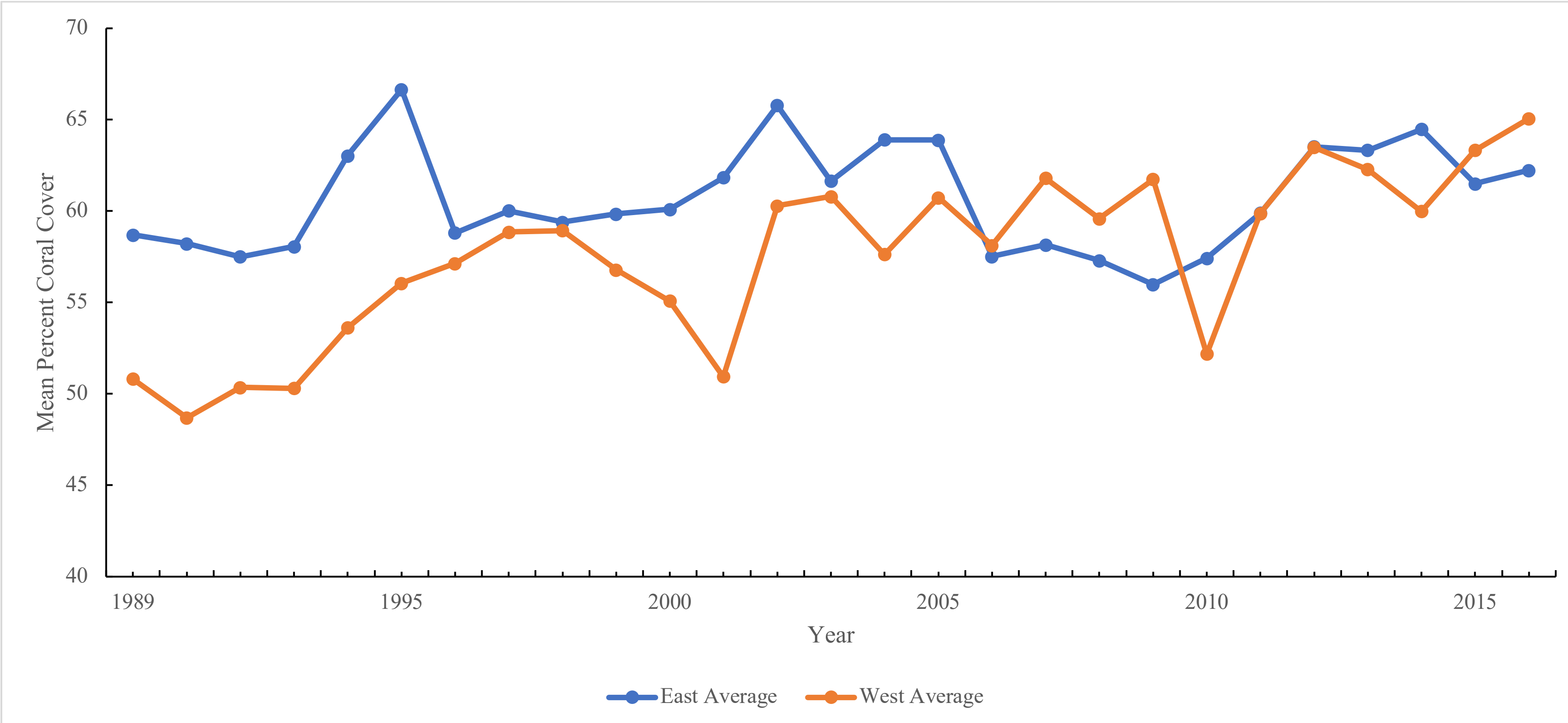

Figure 12: Mean percentage coral coverage from repetitive photostations at East and West Flower Garden Banks for the 27-year time frame (1989-2017).

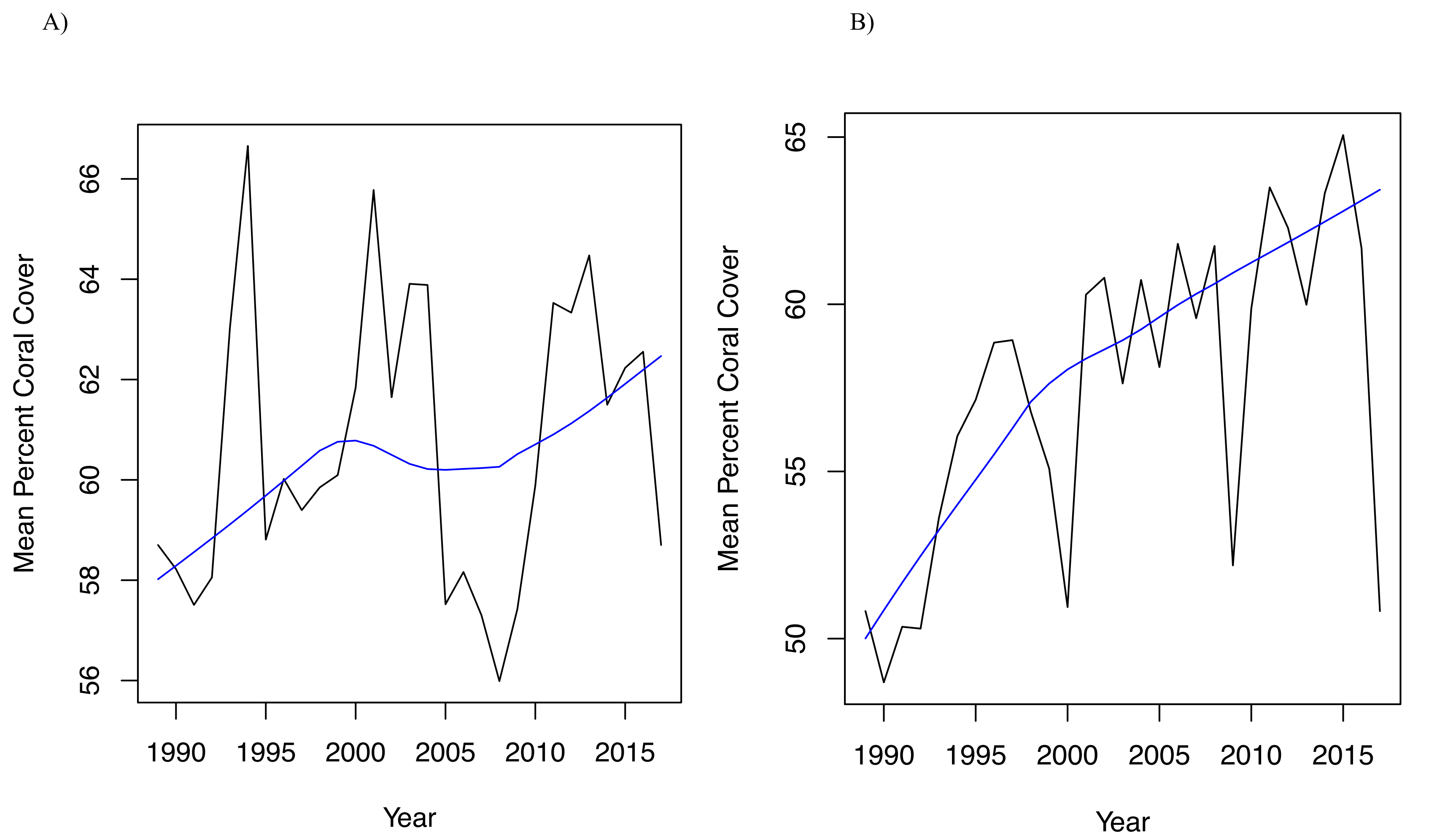

Figure 13: Mann Kendall trend test displaying an overall increasing trend for (A) East Flower Garden Bank and (B) West Flower Garden Bank. The black represents the trend in repetitive photostation mean percentage coral coverage among years and the blue line represents a smooth line to visualize the trend.

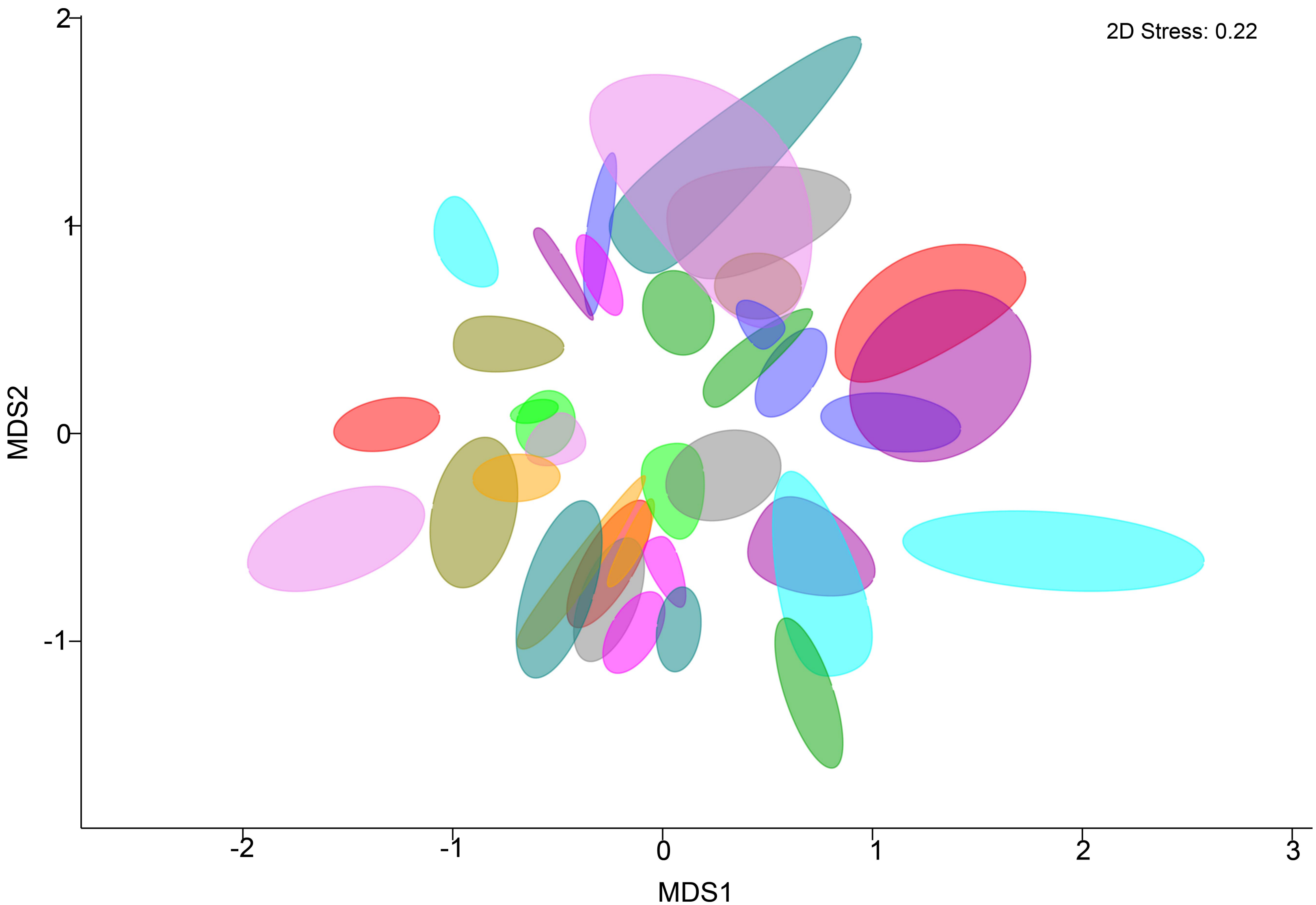

Figure 14: nMDS based on 100 bootstrap averages of coral community samples at 37 repetitive photostations at East Flower Garden Bank. Ellipses represent 95% bootstrap regions generated from 100 bootstrap averages for each photostation.

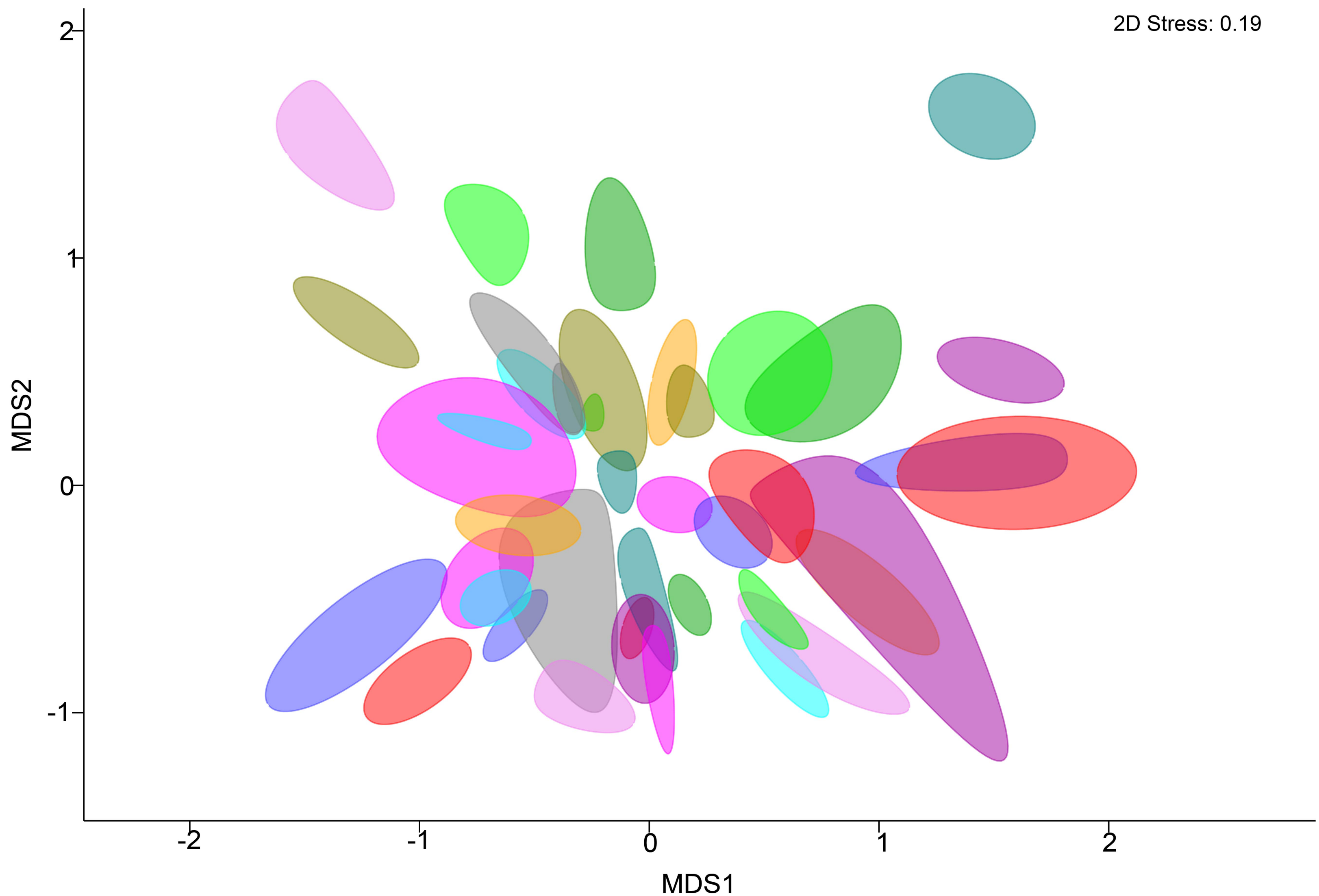

Figure 15: nMDS based on 100 bootstrap averages of coral community samples at 41 repetitive photostations at West Flower Garden Bank. Ellipses represent 95% bootstrap regions generated from 100 bootstrap averages for each photostation.

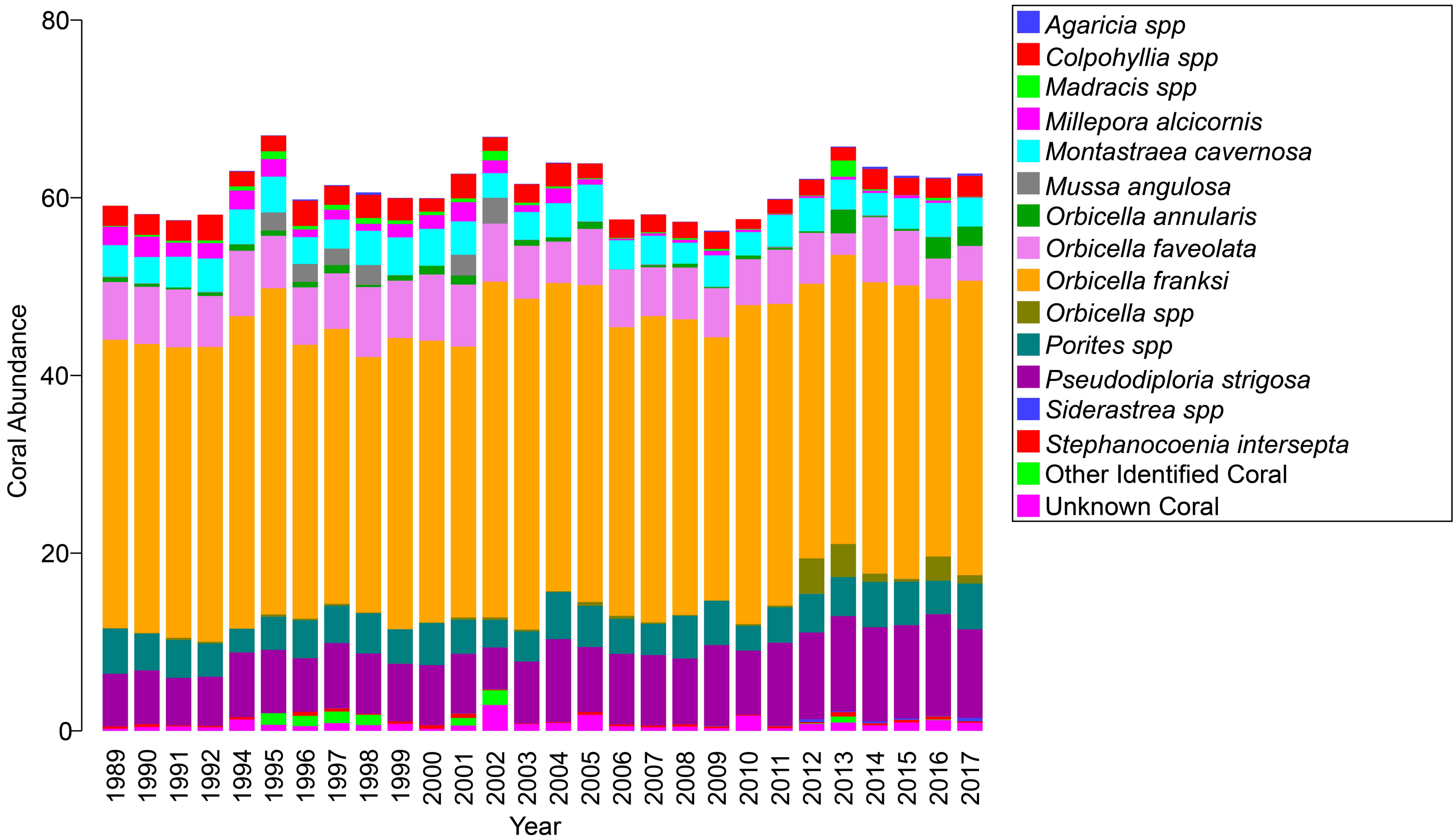

Figure 16: Bar plot representing species coral coverage for each year over the last 27 year for East Flower Garden Bank for each year (1989-2017).

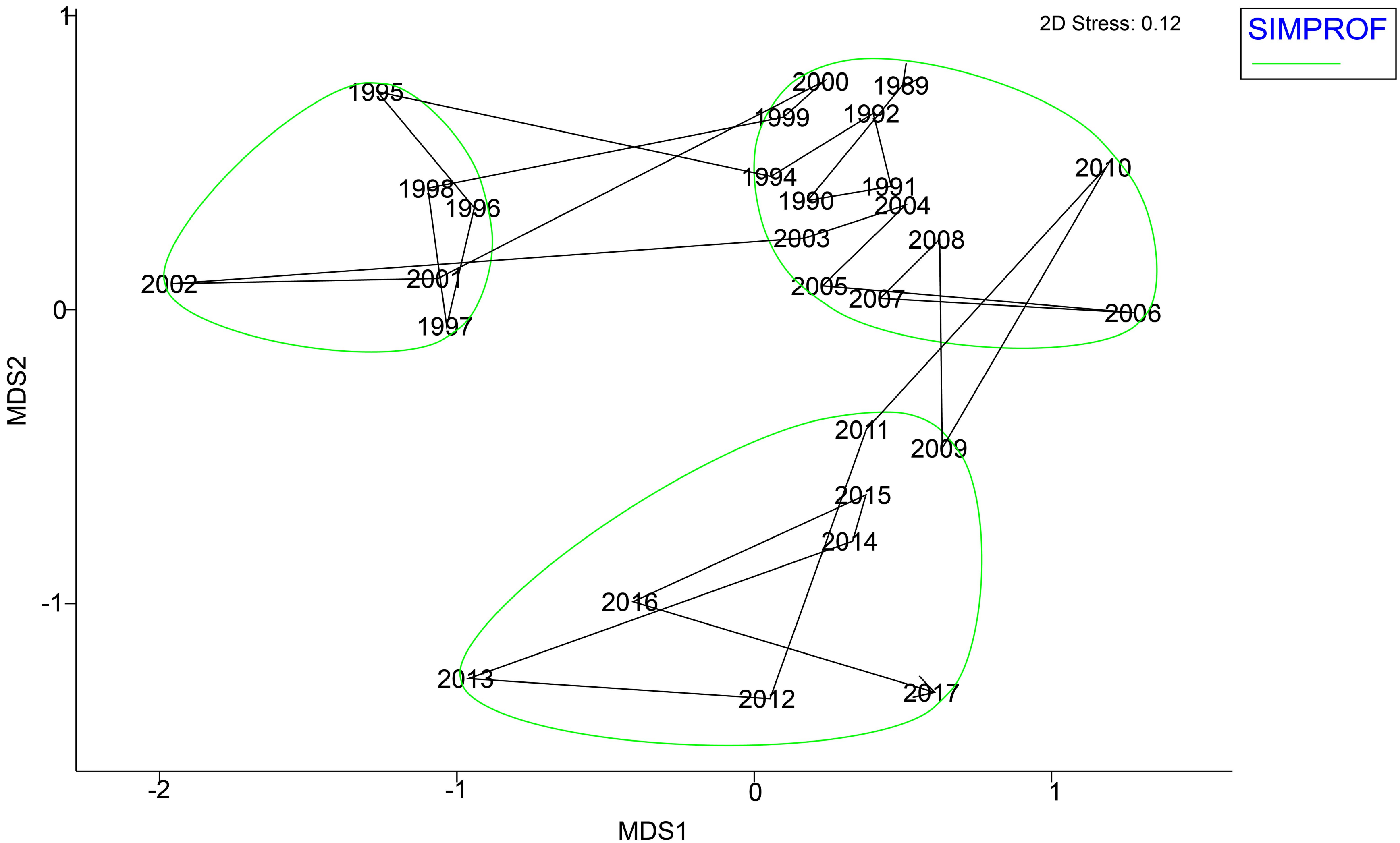

Figure 17: nMDS based on annual Bray-Curtis similarities among percentage coral coverage from East Flower Garden Bank repetitive photostations for 27 years. The trajectory represents the time progression of the dataset and SIMPROF clusters represent significant year groupings.

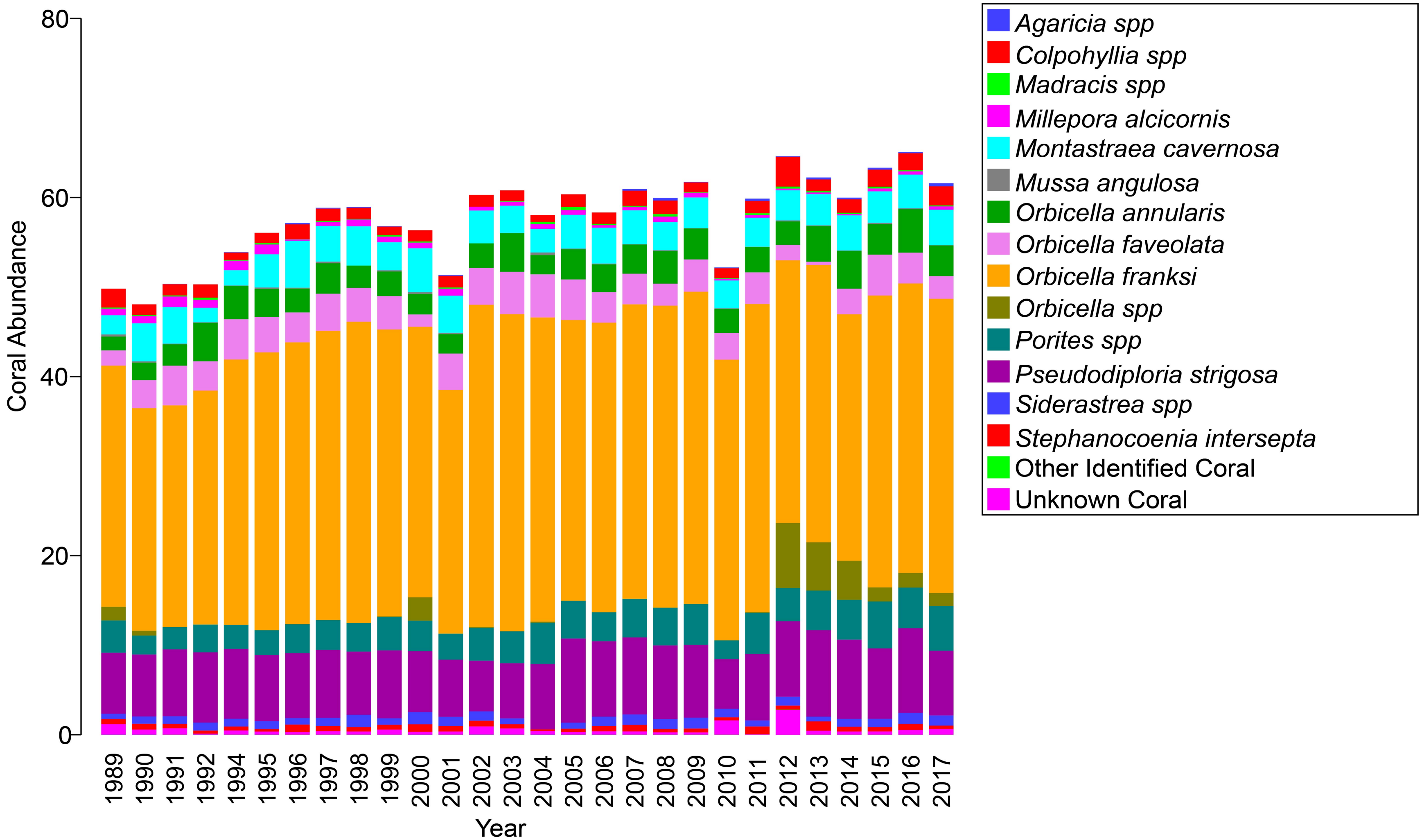

Figure 18: Bar plot displaying species coral coverage for each year over the last 27 years for West Flower Garden Bank (1989-2017).

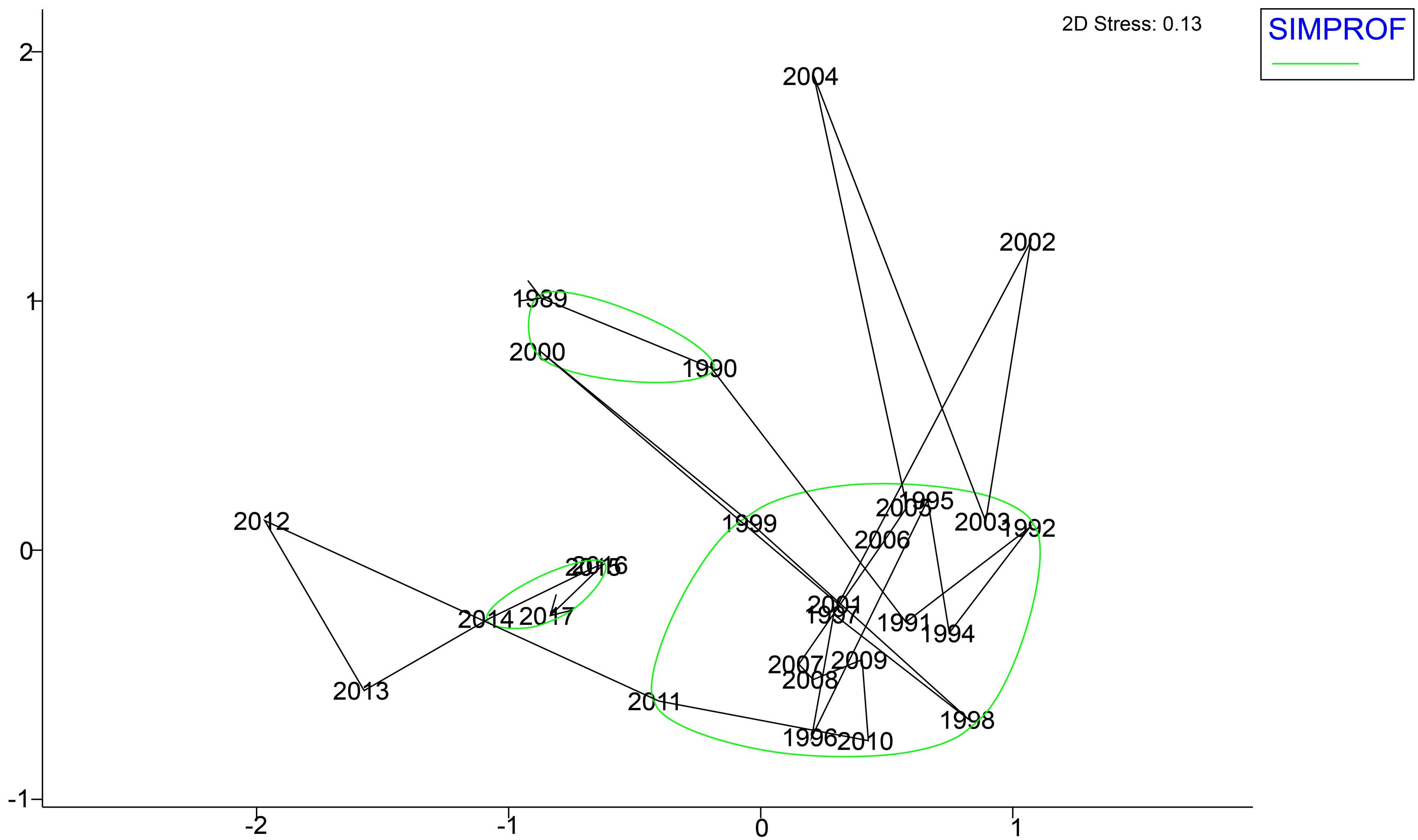

Figure 19: nMDS based on annual Bray-Curtis similarities among percentage coral coverage at West Flower Garden Bank repetitive photosations for 27 years. The trajectory represents time progression of the dataset and SIMPROF clusters represent significant year groupings.

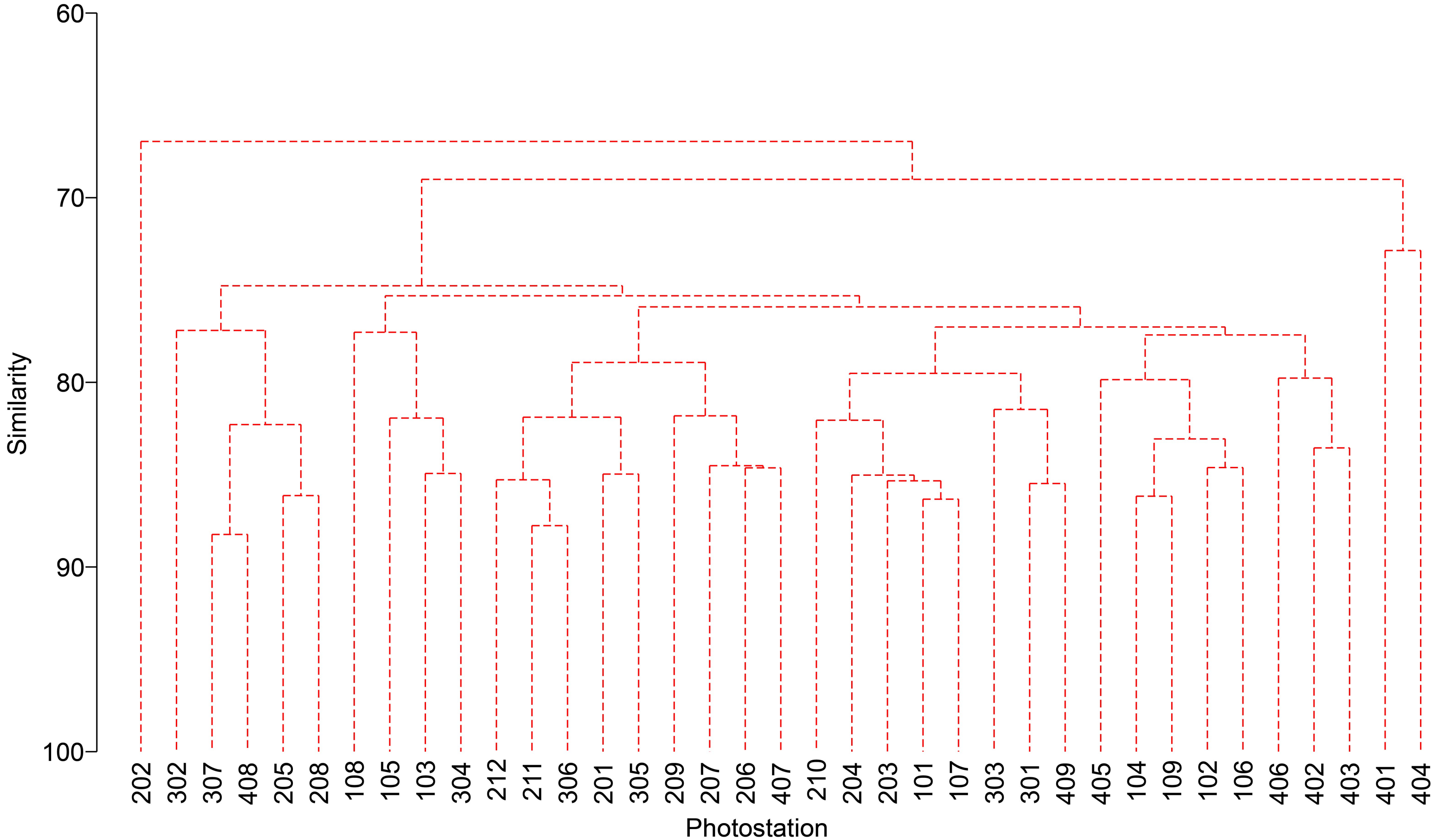

Figure 20: Cluster analysis for East Flower Garden Bank repetitive photostations, indicating a singular grouping for all photostations.

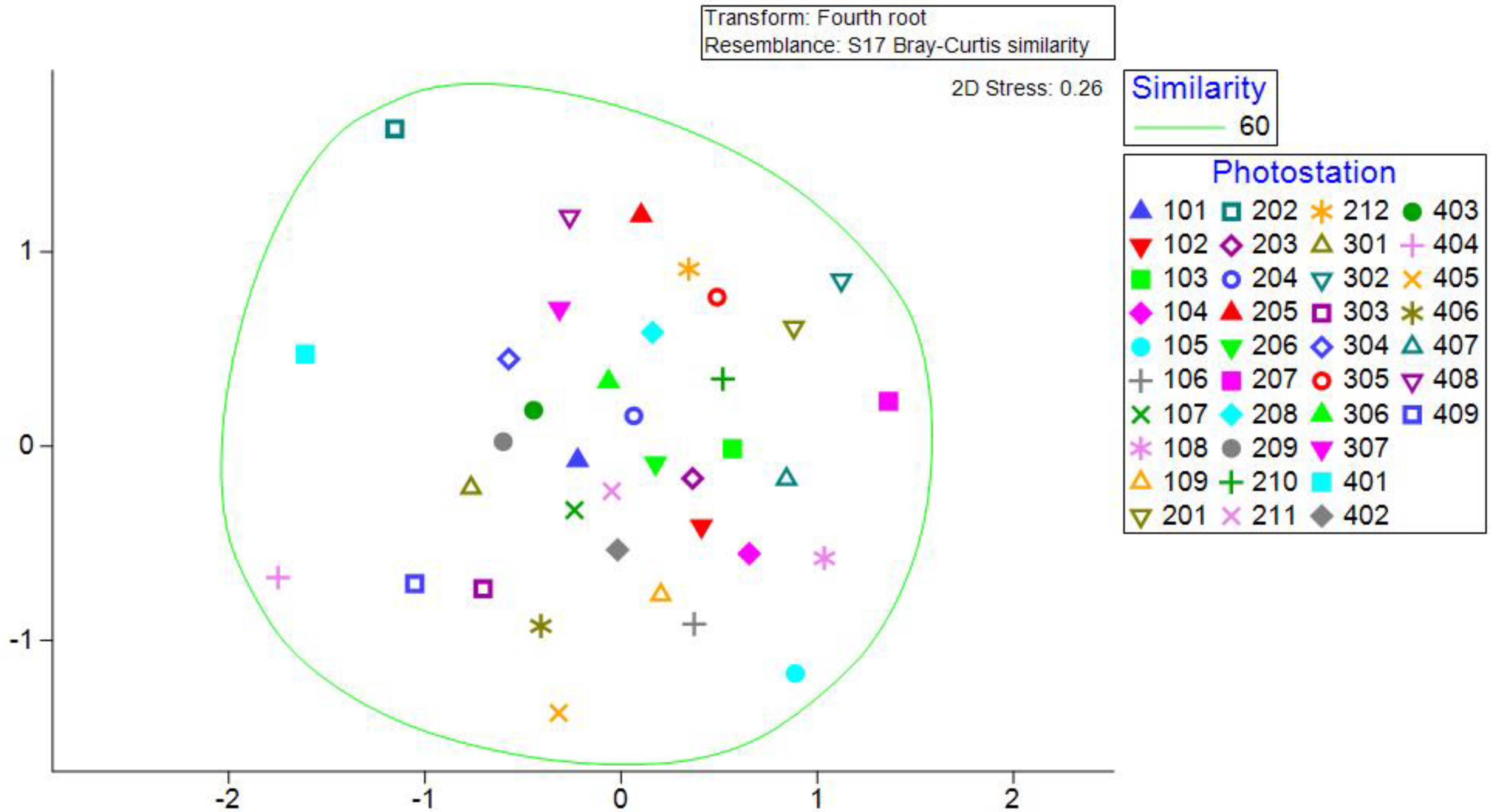

Figure 21: nMDS based on Bray-Curtis similarities for percentage coral coverage at 37 repetitive photostations at East Flower Garden Bank with cluster overlay (60% similarity).

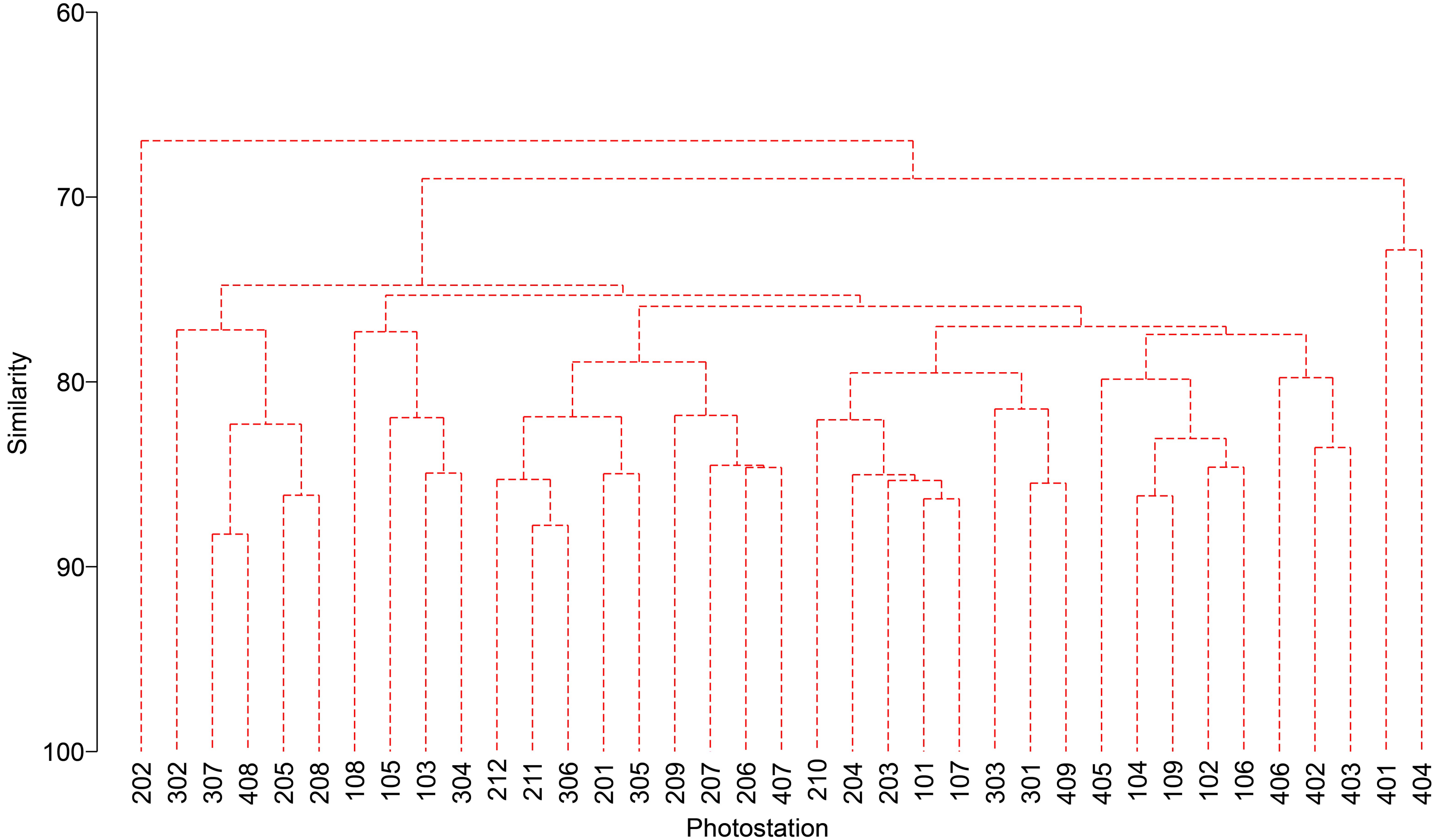

Figure 22: Cluster analysis for West Flower Garden Bank repetitive photostations, indicating a singular grouping for all photostations.

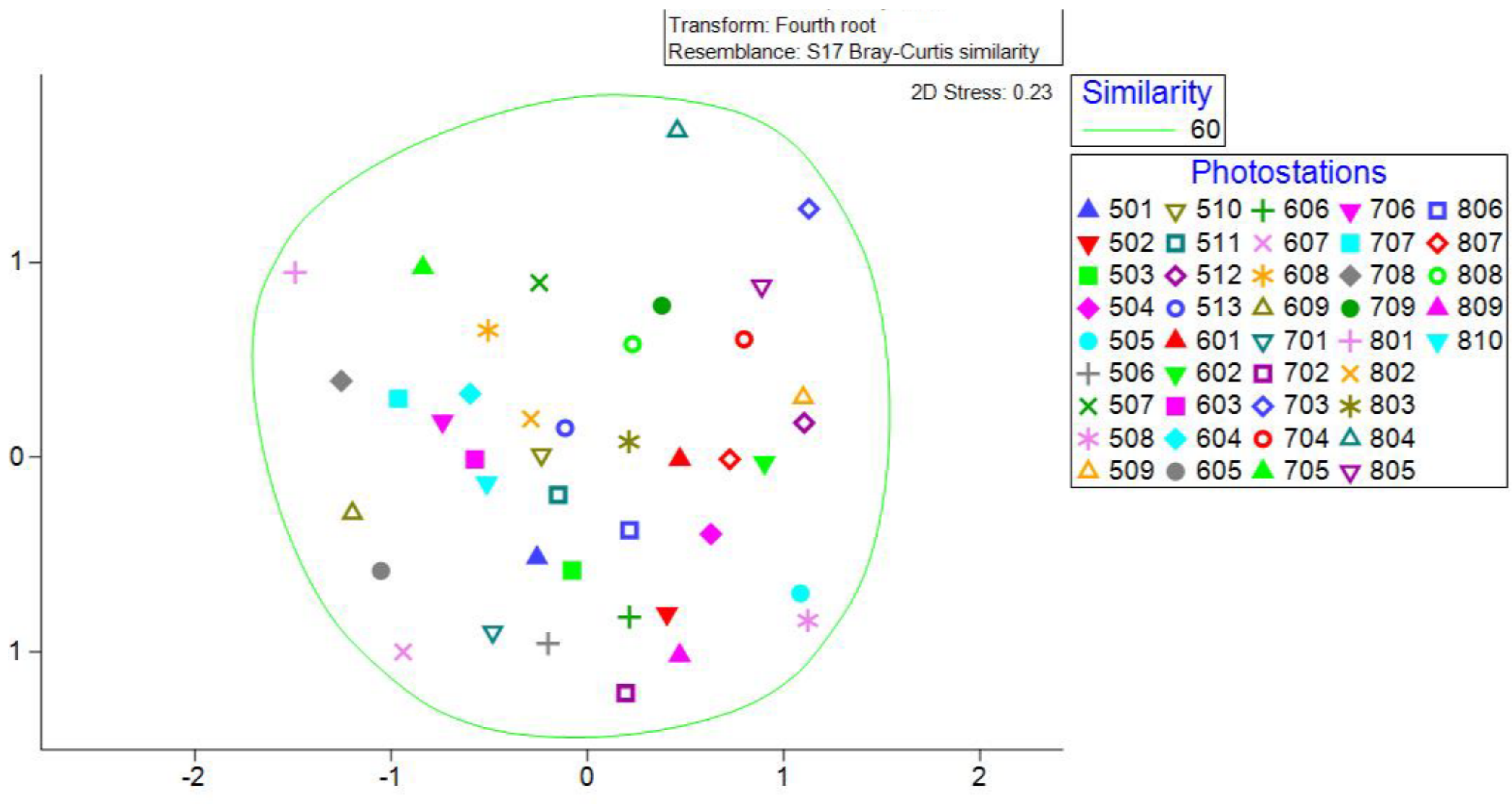

Figure 23: nMDS based on Bray-Curtis similarities for percentage coral coverage at 41 repetitive photostations at West Flower Garden Bank with cluster overlay (60% similarity).

CHAPTER IV

DISCUSSION

Coral Coverage Trends and Coral Species Contributions

Long-term monitoring at East and West FGBs began in the early 1980s and have been continuously monitored since. The official start of the long-term monitoring program at East and West FGBs was 1989, leading to a series of camera setups, point count analysis, data analysts and utilization of repetitive photostation coral identification key. As years have progressed, camera setups and data analysis for LTM repetitive photostations has varied, progressing from 35-mm slides and film to digital still images (Table S1; Dokken et al. 2003; Precht et al. 2005; Johnston et al. 2018a, 2021). Based on previous studies coral coverage at East and West FGBs has exhibited an ability to maintain a high percentage of coral coverage (>50%) over 30 years of monitoring (1989-2019) (Johnston et al. 2016, 2018a, 2020, 2021). Due to the various camera setups, data analysis and data analysts, a reanalysis of the historical repetitive photostation image dataset was conducted to determine if coral coverage at East and West FGBs has truly maintained or increased high coral coverage rates. Because of the utilization of various camera and light setups, image quality and clarity progressed and bettered as technology advanced, allowing for possible discrepancies within the image analysis. In reanalyzing repetitive photostation images utilizing a standardized methodology allows for a more thorough analysis by a single individual, utilizing a repetitive photostation coral identification key to allow for more accuracy in the identification of fauna within images. I hypothesized that coral coverage at East and West FGBs would increase significantly over the 27-year time period, however this was incorrect. Reanalysis resulted in only West FGB exhibiting a significantly increasing trend in coral coverage based on repetitive photostation data. This could have occurred because of varying proportions of low to high initial percentage coral cover across the two banks with low and medium exhibiting the highest increases. West FGB had a greater proportion of low and medium initial percentage coral coverage which likely resulted in the observed significant trend (Figure 13b, Table 3). However, at East FGB no significant trend was observed, rather it maintained high rates of coral coverage. This could be due to repetitive photostations that had low initial percentage coral cover exhibited a significant increasing trend, while those that had high initial percentage coral cover exhibited a significant decreasing trend and those that had a medium initial percentage coral cover displayed no trend (Figure 13a, Table 2). Other attributable factors leading my study to differ include, varying light and camera setups, as well as different data analysis utilized, progressing from mylar traces to a point count analysis and also the differing data analysts who processed the image analysis (Johnston et al. 2018a, 2021). Because monitoring at East and West FGBs has occurred for 30 years, various entities assumed responsibility for conducting LTM (Johnston et al. 2018 a, 2021). Monitoring was first conducted through a contract between the MMS and Texas A&M University (TAMU), establishing the 1-hectare study sites (Johnston et al. 2018a, 2021). LTM was conducted by TAMU and environmental consulting groups until 2009, when BOEM and NOAA formed an interagency agreement, allowing FGBNMS to take over monitoring efforts (Johnston et al. 2021). Community composition of the FGB coral reefs may also contribute to the stability and increasing coral coverage trends. The FGBs consists primarily of reef-building corals (stony corals), with *Orbicella franksi* carrying the most weight in overall coral coverage at both East FGB and West FGB (Tables 9, 10, 11, & 12; Figures 13, 16, & 17). *Orbicella franksi* displayed an ability to adapt to local climate change in Panama and Bermuda (Silberger et al. 2019). However, thermal tolerances varied among the two areas; corals in Panama had a higher heat tolerance as opposed to those in Bermuda (Silberger et al. 2019). The difference in thermal tolerance is most likely due to the different climates in the area and potentially different symbiont types (Silberger et al. 2019). Because of the maintained high coral coverage of the FGBs, it could be inferred that coral composition plays a key role in the success of the FGBs (Middlebrook et al. 2008; Silberger et al. 2019). Species most observed at East and West FGBs based on previous reports (Johnston et al. 2021) include *Orbicella franksi* and *Pseudodiploria strigosa* as the most reoccurring species. Based on these reports, I hypothesized that these species would remain the most reoccurring species observed at the FGBs. My reanalysis on the repetitive photostation dataset revealed *Orbicella franksi* and *Pseudodiploria strigosa* as the most reoccurring species among East and West FGBs.

Potential Drivers of the Trends Observed at East and West FGBs

The FGB has the benefit of being over 204 km offshore and therefore are typically buffered from anthropogenic stressors thereby reducing some negative effects. Part of their success is likely due to their designation as a national marine sanctuary and subsequent protection and intensive monitoring (Hickerson et al. 2008, 2012; Johnston et al. 2016, 2021). Mean percentage coral coverage at repetitive photostations at West FGB exhibited a significant increase over the 27 y (Figure 12a) and coverage in repetitive photostations at East FGB remained stable (Figure 12b), which is contrary to many patterns observed for many other reefs in the Caribbean. However, anthropogenic, and natural-induced stressors are still known to occur at the FGBs, although it seems that they are not as impactful as compared to other coral reefs in the Caribbean region (Hickerson et al. 2012; Johnston et al. 2016). Nevertheless, they have been subjected to other natural-induced stressors, such as hurricanes and environmental stressors caused by precipitation and current events. For example, East FGB experienced a coral mortality event outside of the LTM study sites in July 2016 that is believed to have been caused primarily by localized low dissolved oxygen because of the accumulation of turbid coastal waters along the Texas coast after precipitation and excessive river discharge and possible upwelling at East FGB (Johnston et al. 2018b; Le Henaff et al. 2019; Kealoha et al. 2020).

Despite their importance, coral reefs around the world have been experiencing high mortality rates (Pandolfi et al. 2003; Bellwood et al. 2004; Hoegh-Guldberg et al. 2017; Madin & Madin 2019) associated with seawater temperatures above 30°C, which results in corals bleaching. Coral bleaching events affect coral reefs differently based on a variety of factors. Typically, reefs can recover if the event does not have a high duration. However, if it has a high duration or recurrence within a certain time period, the coral's ability to recover decreases. This leaves them highly susceptible to coral disease or natural disturbances and anthropogenic stressors (Baker et al. 2008; Gil-Agudelo et al. 2019). Bleaching events have had detrimental effects on coral reefs around the globe, while some seemingly had little or no effect at all on the reef’s health (Baker et al. 2008; Johnston et al. 2019). Bleaching severity could differ because of reef depth and proximity to locales affected by anthropogenic stressors (e.g., whether reefs are inshore or offshore) (Berkelmans et al. 1999; Hickerson et al. 2008; Smith et al. 2016a & 2016b). Bleaching events have progressively intensified within the last few decades because of climate change, revealing that coral reefs are among the most vulnerable to the effects of climate change, anthropogenic stressors, and coral disease (Donner et al. 2005; Wilson et al. 2006; Hoegh-Guldberg et al. 2017). For example, the Great Barrier Reef (GBR) experienced some of its worst coral bleaching events in 1998 and 2002 (Berkelmans et al. 1999; 2004). The coral reefs of the Florida Keys have also experienced mass coral bleaching events since 1987, with them progressively worsening as years progress (Manzello 2015). Shallow-water coral reefs are the most affected by coral bleaching, likely due to seawater temperatures at the surface being higher or more UV rays are able to penetrate shallow reefs (Putnam et al. 2017; Johnston et al. 2019). Unlike these shallower reefs, the East FGB and West FGB experience fewer coral bleaching events. However, one occurred in 2016 that had a more pronounced effect on East FGB (Johnston et al. 2019). The severity of the bleaching event occurred because seawater temperatures were above 30 C longer at East FGB compared to West FGB, because of this, the effects of coral bleaching were much more severe at East FGB. However minimal mortality was observed at both banks and most corals were able to fully recover by the next year. The absence of prolonged bleaching is likely a contributor to the consistent and increasing coral coverage observed at the FGBs. Depth and location of the FGBs likely contributed to reduced coral bleaching as compared to other coral reefs within the Caribbean Ocean (Smith et al. 2008, 2015, 2016a & 2016b; Bongaets et al. 2010; Loya et al. 2016; Lesser et al. 2018; Rocha et al. 2018; Belter et al. 2020).

The FGBs depth is believed to aid them in their ability to avoid many different factors that are affecting other coral reefs around the world (Cancelmo 2008; Hickerson et al. 2012). Reefs that are further offshore, away from human interaction and those deeper than traditional shallow depths have been observed to exhibit a healthier ecosystem (Hickerson et al. 2008; Gouezo et al. 2019). Offshore coral reefs still experience natural induced stressors (hurricanes, coral species, diseases, etc.), as well as some anthropogenic stressors (scuba divers, invasive species, pollution, and overfishing). Coral reef depth also has an effect on coral recruitment composition, with size of coral recruits increasing with depth (Turner et al. 2018). The FGBs reef crest is considerably deeper than most shallow-water reefs.

Stressed corals, either due to bleaching or other stressors such as pollution, may be more susceptible to disease, which can also lead to declines in coral reefs over time in response to climate change (Estrada-Saldivar et al. 2020; Estrada-Saldivar et al. 2021). The coral reefs found within the Caribbean are among the most affected by coral diseases (Bruno et al. 2007; Alvarez-Filip et al. 2019; Estrada-Saldivar et al. 2020). Diseases that affect scleractinians (reef-building) include yellow-band disease, white pox, black-band disease, red-band disease (Hickerson et al. 2008; Bruno et al. 2007; Estrada-Saldivar et al. 2020), and stony coral tissue loss disease (SCTLD), which is an emergent white-plague type coral disease that has caused large scale die-offs in reef building coral communities in the Caribbean (Alvarez-Filip et al. 2019; Estrada-Saldivar et al. 2020; Meiling et al. 2020). It has affected 20 or more scleractinian corals, including reef-building species, as well as species that are listed as endangered (Hickerson et al. 2008; Aeby et al. 2019; Meyer et al. 2019; Meiling et al. 2020; Estrada-Saldivar et al. 2021). SCTLD was first recorded off the Florida coast in 2014 and has continued to spread throughout the Caribbean, infecting corals causing multifocal lesions that can quickly progress throughout the colony often resulting in total mortality of the colony (Meiling et al. 2020; Estrada-Saldivar et al. 2021). Unlike at Caribbean and other Gulf of Mexico coral reefs, the FGBs have not had any SCTLD or other coral disease outbreaks (Hickerson et al. 2008; Belter et al. 2020). Therefore, the lack of coral disease on the FGBs may contribute to the stable and increasing trends at these reefs relative to other reefs. The relative lack of disease presence could be partly because the coral reef is living within a national marine sanctuary at deeper depths and thus corals are not exposed to stressors that might allow for opportunistic pathogens to infect them. However, there is no definitive answer, which emphasizes the importance of conducting further research, such as comparative surveys of disease affected reefs and the FGB.

Other anthropogenic stressors that plague coral reefs around the globe include pollution, human interaction, overfishing, and invasive species. For example, oceans around the world have experienced increasing amounts of pollution, including oil to microplastics. Oil spills have occurred around the globe, some being worse than others. The worst in United States was the 2010 Deepwater Horizon Oil Spill, which affected all marine life in the spill zone (Joye 2015; Kujawinski et al. 2020). This spill occurred over an 87-day period in the northern Gulf of Mexico, 520 km from East FGB, from the Mancondo well, spilling at least 5 million barrels of oil and 250,000 metric tons of natural gas, primarily methane (Joye 2015). While the spill did cause damage to wildlife and marine life within the Gulf of Mexico, no effects were documented at the FGBs, most likely because the spill zone did not reach the FGBs (Johnston et al. 2013; Kujawinski et al. 2020). However, if the Deepwater Horizon oil spill had impacted the FGBs, long-term monitoring conducted within the sanctuary would have provided valuable data, emphasizing the importance of long-term monitoring among coral reefs (Lubchenco et al. 2012; Johnston et al. 2013; Murawski et al. 2016).

As human populations have increased, the demand on coral reefs as source of food has also increased significantly (Roberts 1995) resulting in overfishing effects on coral reefs (Roberts 1995; Shantz et al. 2020). Overfishing takes a toll on coral reef health because all organisms living within the reef rely on each other; when a part of that food chain is either removed or numbers are significantly depleted, it will cause changes in species abundance and composition within the system (Roberts 1995; Valentine & Heck 2005; Shantz et al. 2020). Overfishing can reduce species diversity, leading to local extinctions of not only the species being targeted but also other organisms living within the reef (Roberts 1995). For example, overfished reefs are susceptible to algae growth due to the loss of herbivorous fish (Loh et al. 2015; Shantz et al. 2020). With the loss of herbivorous fish, algal overgrowth of coral occurs, primarily affecting mound/boulder corals, such as *Pseudodiploria strigosa* (Loh et al. 2015). Despite its designation as a national marine sanctuary, recreational fishing and illegal fishing have been observed at the FGBs, as well as lost or discarded fishing gear (Office of National Marine Sanctuaries 2008). Nevertheless, these activities are not common occurrences at the FGBs, which may contribute to the stability and increase in coral coverage at the FGBs.

Marine protected areas, such as, National Marine Sanctuaries (NMS) have played an important role in the survival and overall health of coral reef ecosystems around the world. NMS are designated areas of the US marine systems that are protected to preserve the marine life within that area. There are 15 NMS and 2 national monuments in the United States (https://sanctuaries.noaa.gov). The Florida Keys National Marine Sanctuary (FKNMS), one of the largest NMS, protects over 6,000 species of marine life, the nation's only bank-barrier coral reef and the largest seagrass communities are included in this NMS (Donahue et al. 2011). FKNMS experienced detrimental declines in marine life before its designation as a national marine sanctuary in 1990 (Donahue et al. 2011; Toth et al. 2014). After its designation as a national marine sanctuaries, protective measures were immediately executed to protect and preserve the remaining marine life in the area, such as prohibiting mining, oil exploration or anything that would cause damage to the seafloor, also enacting a restriction on large ship traffic through the area (Shivlani et al. 2008; https://floridakeys.noaa.gov/about/fknmsp_act.html). However, their designation as a national marine sanctuary has not allowed for an increase or stability in coral coverage to occur (Somerfield et al. 2008; Toth et al. 2014). While the designation as a national marine sanctuary has benefited the FGBs, enforcing regulations to all who visit, while also running a long-term monitoring (LTM) program to keep track of the benthic community living within the sanctuary, it cannot be considered as the only factor driving the high coral coverage trends observed.

Although national marine sanctuaries have the advantage of being protected from various anthropogenic stressors that are causing significant declines in coral reef health, the adjacent areas, e.g., at the border of the sanctuaries, can still be adversely affected if an oil spill, originates in the adjacent area. National marine sanctuaries reefs are also still subjected to human induced stressors. For example, SCUBA diving is a recreational activity that is permitted within national marine sanctuaries and that can, result in direct and indirect stressors to the coral reef ecosystem (DM Lawrence; Roche et al. 2016). SCUBA is an important part of tourism at coastal destination sites, creating a large portion of income for local areas, it causes stress to the reefs that utilized (Hawkins et al. 2002; Roche et al. 2016). As years have progressed, the impact that scuba has on coral reefs has become clearer (Roche et al. 2016). For example, it is now known that damage to corals (such as removal, hitting or breaking) living within dive sites that are frequently visited occurs more than previously thought (Roche et al. 2016). While scuba diving can have adverse effects, the FGB is well regulated, and no significant diving based damaged was observed in this study.

The trend of maintaining high coral coverage and potentially being resistant to different natural stressors, such as coral diseases and coral bleaching, is not something that is seen globally within coral reefs. East and West FGBs have retained high coral coverage that has remained stable or increased throughout the time it has been monitored (Bellwood et al. 2004; Bruno & Selig 2007; De’ath et al. 2012; Bridge et al. 2013; Madin & Madin 2019). Given that this trend is not common globally, further studies of the FGBs ability to maintain high coral coverage rates is required. The success of the FGBs is not truly known, it is assumed by many that its success is due in part to their depth (17-25 m), offshore location, and designation as a national marine sanctuary which buffers them from common stressors.

Further analysis should be conducted to determine the drivers behind the FGBs ability to increase and maintain high levels of coral coverage at each bank and if these trends are predicted to continue. It has been observed that depth and location most likely play a significant role in the success of maintaining coral coverage at coral reefs found within mesophotic depths since they are believed to be somewhat shielded from localized anthropogenic and natural induced stressors (Lesser et al. 2009; Smith et al. 2015). Although the photostations from this study are not found within mesophotic depths (30-150 m), a better understanding of reefs found within mesophotic depths would allow for a better understanding of East and West banks.

REFERENCES

Aeby GS, Ushijima B, Campbell JE, Jones S, Williams GJ, Meyer JL, Hase C, & Paul VJ (2019). Pathogenesis of a Tissue Loss Disease Affecting Multiple Species of Corals Along the Florida Reef Tract. *Frontiers in Marine Science.* 6:678. doi: 10.3389/fmars.2019.00678

Alvarez-Filip L, Estrada-Saldivar N, Perez-Cervantes E, Molina-Hernandez A, & Gonzales-Barrios FJ (2019). A rapid spread of the stony coral tissue loss disease outbreak in the Mexican Caribbean. *Peer J* 7:e8069 https://doi.org/10.7717/peerj.8069

Aronson RB, Precht WF, Murdoch TJT, & Robbart ML (2005). Long-Term Persistence of Coral Assemblages on the Flower Garden Banks, Northwestern Gulf of Mexico Implications for Science and Management. *Gulf of Mexico Science, 23*(1). doi:10.18785/goms.2301.06

Baker AC, Glynn PW, & Riegel B (2008). Climate-change and coral reef bleaching: An ecological assessment of long-term impacts, recovery trends and future outlook. *Estuarine, Coastal, and Shelf Science.* 80(2008) 435-471. doi: 10.1016/j.ecss.2008.09.003

Bellwood D, Hughes T, Folke C, & Nystrom M (2004). Confronting the coral reef crisi. *Nature* 429, 827-833. https://doi.org/10.1038/nature02691

Belter M, Blondeau J, Donovan C, Edwards K, Enochs I, Formel N, Geiger E, Gittings S, Grove S, Hickerson E, Jonhston M, Kelsey H, Lohr K, Miller N, Nuttall M, Schmahl GP, Towle E, & Viehman S (2020). Coral reef condition: A status report for the Flower Garden Banks. *NOAA Coral Reef Conservation Program.* https://doi.org/10.25923/5662-r141

Berkelmans R & Oliver JK (1999). Large-scale bleaching of corals on the Great Barrier Reef. *Coral Reefs* 18: 55-60. https://doi.org/10.1007/s003380050154

Berklemans R, De'ath G, Kininmonth S (2004). A comparison of the 1998 and 2002 coral bleaching events on the Great Barrier Reef: spatial correlation, patterns, and predictions. *Coral Reefs* 23: 74-83. doi: 10.1007/s00338-003-0353-y

Bongaerts P, Ridgway T, Sampayo EM, & Hoegh-Guldberg O (2010). Assessing the 'deep reef refugia' hypothesis: focus on Caribbean reefs. *Coral Reefs* 29: 309-327 doi: 10.1007//s00338-009-0581-x

Borneman EH & Wellington GM (2005). Pathologies Affecting Reef Corals at the Flower Garden Banks, Northwesterm Gulf of Mexico. Gulf of Mexico Science 23(1). doi: 10.18785/goms.2301.07

Bridge TCL, Hughes TP, Guinotte JM, & Bongaerts P (2013). Call to protect all coral reefs. *Nature Climate Change,* 3(6), 528-530. doi:10.1038/nclimate1879

Bright TJ, Kraemer GP, Minnery GA, and Viada ST (1984). Hermatypes of the Flower Garden Banks, Northwestern Gulf of Mexico: A Comparison to other Western Atlantic Reefs. Coral Reef Paper, Bulletin of Marine Science. 34(3):461-476.

Brooke S & Schroeder WW (2007). State of Deep Coral Ecosystems in the Gulf of Mexico Region: Texas to the Florida Straits. State Deep Coral Ecosystem United States 271-306

Brown BE (1997). Coral bleaching: causes and consequences. *Coral Reefs* 16, S129-S138. https://doi.org/10.1007/s003380050249

Bruno JF & Selig ER (2007). Regional Decline of Coral Cover in the Indo-Pacific: Timing, Extent, and Subregional Comparisons. *PLoS ONE* 2(8): e711. doi:10.1371/journal.pone.0000711

Cancelmo J (2008). Texas Coral Reefs. Texas A&M Univ. Press, Corpus, Christi, Tx.

Childs JN (2001). The Occurrence, habitat use and behavior of sharks and rays associating with topographic highs in the Northwestern Gulf of Mexico. Texas A&M University. https://oasktrust.library.tamu.edu/handle/1969.1/ETD-TAMU-2001-THESIS-C45.

Coker DJ, Wilson SK, & Pratchett MS (2014). Importance of live coral habitat for reef fishes *Rev Fish Biol Fisheries* 24: 89. doi: 10.1007/s11160-013-9319-5

Cole AJ, Pratchett MS, and Jones GP (2008), Diversity and functional importance of coral-feeding fishes on tropical coral reefs. Fish and Fisheries, 9: 286-307. doi:10.1111/j.1467-2979.2008.00290.x

De'ath G, Fabricius KE, Sweatman H, Puotinen M (2012). The 27-year decline of coral cover on the Great Barrier Reef and its causes. *PNAS* 109(44): 17995-17999. https://doi.org/10.1073/pnas.1208909109

Dokken QR, MacDonald IR, Tunnell JW Jr, Beaver CR, Boland GS, & Hagman DK (1999). Long-term monitoring at the East and West Flower Garden Banks 1996-1997. OCS Study MMS 99-0005, US Department of Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Dokken QR, MacDonald IR, Tunnell JW Jr, Wade T, Withers K, Dilworth SJ, Bates TW, Beaver CR, & Rigaud CM (2003). Long-term monitoring at the East and West Flower Garden

Banks National Marine Sanctuary, 1998-2001. OCS Study MMS 2003-031, US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Review, New Orleans, Louisiana.

Donahue S, Acosta A, Akins L, Ault J, Bohnsack J, Boyer J, Callahan M, Causey B, Cox C, Delaney J, Delgado G, Edwards K, Garrett G, Keller B, Kellison GT, Leeworth VR, MacLaughlin L, McClenachan L, Miller MW, Miller SL, Ritchie K, Rohmann S, Santavy D, Pattengill-Semmens C, Sniffen B, Werndli S, & William DE (2008). The State of Coral Reef Ecosystems in the Florida Keys. In: Waddell JE, Clarke AM (eds) The state of coral ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73, NOAA/NCCOS Center for Coastal Monitoring & Assessment Biogeography Team, Silver Spring, MD. pp 161-187

Eakin CM, Morgan JA, Heron SF, Smith TB, Liu G, et al. (2010). Caribbean Corals in Crisis: Record Thermal Stress, Bleaching, and Mortality in 2005. *PLos ONE* 5(11): e13969. doi:10.1371/journal.pone.0013969

Estrada-Saldivar N, Molina-Hernandez A, Perez-Cervantes E, Medellin-Maldonado F, Gonzalez-Barrios FJ, & Alvarez-Filip L (2020). Reef-scale impacts of the stony coral tissue loss disease outbreak. *Coral Reefs* 39:861-866. https://doi.org/10.1007/s00338-020-01949-z

Estrada-Saldivar N, Quiroga-Garcia BA, Perez-Cervantes E, Rivera-Garibay OO, & Alvarez-Filip L (2021). Effects of the Stony Coral Tissue Loss Disease Outbreak on Coral Communities and the Benthic Composition of Cozumel Reefs. *Front. Mar. Sci.* 8:632777. doi: 10.3389/fmars.2021.632777

Flower Garden Banks website (flowergarden.noaa.gov/about/history.html)

Gardener JV, Mayer LA, Hughes Clark JA, and Kleiner A (1998) High-Resolution Multibeam Bathymetry of East and West Flower Garden Banks, Stetson Banks, Gulf of Mexico. Gulf of Mexico Science 16 (2). DOI: 10.18785

Gardner TA, Cote IM, Gill JA, Grant A, & Watkinson AR (2003). Long-term region-wide declines in Caribbean corals. *Science* 301: 958-960

Gardner TA, Cote IM, Gill JA, Grant A, & Watkinson AR (2005). Hurricanes and Caribbean Coral Reefs: Impacts, Recovery Patterns, and Role in Long-Term Decline. *Ecology* 86(1), pp. 174-184. https://doi.org/10.1890/04-0141

Gil-Agudelo DL, Cintra-Buenrostro CE, Brenner J, Gonzalez-Diaz P, Kiene W, Lustic C & Perez-Espana H. (2020) Coral Reefs in the Gulf of Mexico Large Marine Ecosystem: Conservation Status, Challenges, and Opportunities. *Front. Mar. Sci.* 6:807. doi: 10.3389/fmars.2019.00807

Gittings SR, Boland GS, Deslarzes KJP, Hagman DK, & Holland BS (1992). Long-term monitoring at the East and West Flower Garden Banks. OCS Study MMS 92-0006, US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, Louisiana.

Gittings SR (1998). Reef Community Stability on the Flower Garden Banks, Northwest Gulf Mexico. *Gulf of Mexico Science,* 16(2). doi: 10.18785/goms.1602.05

Gittings SR & Hickerson EL (1998). Flower Garden Banks National Marine Sanctuary: Introduction. *Gulf of Marine Science* 16(2). Retrieved from https://aquila.usm.edu/goms/vol16/iss2/6

Goodbody-Gringley G & Manfrino C (2020). Coral Reef Resilience Over Two Decades at Little Cayman Island. Technical Report from the Central Caribbean Marine Institute. 20201:11 pp.

Gouezo M, Golbuu Y, Fabricius K, Olsudong D, Mereb G, Nestor V, Wolanksi E, Harrison P, & Doropulos C. (2019). Drivers of recovery and reassembly of coral reef communities. *Proc. R. Soc. B.* 286: 20182908. http:// dx.doi.org/ 10.1098/rspb.2018.2908

Halpern B, Walbridge S, Selkoe K, Kappel C, Micheli F, D'Agrosa C, Bruno J, Casey K, Ebert C, Fox H, Fujita R, Heinemann D, Leninan H, Madin E, Perry M, Selig E, Spalding M, Steneck R, & Watson R (2008) A Global Map of Human Impact on Marine Ecosystems. *Science* 319(5865), 948-952. DOI: 10.1126/science.1149345

Hawkins JP, Roberts CM, Van'T Hof T, De Meyer K, Tratalos J, & Aldam C (2002). Effects of Recreational Scuba Diving on Caribbean Coral and Fish Communities. *Conservation Biology* 13(4): 888-897. https:// doi.org/10.1046/j.1523-1739.1999.97447.x

Hickerson E, Schmahl G, Robbart M, Precht WF, & Caldow C (2008). The State of Coral Reef Ecosystems of the Flower Garden Banks, Stetson Bank, and other banks in the Northwestern Gulf of Mexico. *State Coral Reef Ecosys United States Pacific Free Assoc States* 2008 NOAA Tech Memo NOS NCCOS 189-217.

Hickerson EL, Schmahl GP, Johnston MA, Nuttal MF, Embesi JA, & Eckert RJ (2012) Flower Garden Banks-A Refuge in the Gulf of Mexico? in *Proceedings of the 12th International Coral Reef Symposium* (Cairns, QLD).

Hodgson G (1999) A Global Assessment of Human Effects on Coral Reefs. *Science Direct* Marine Pollution Bulletin 38(5), 345-355. https:// doi.org/10.1016/S0025-32X(99) 00002-8

Hoegh-Guldberg O (1999). Climate change, coral bleaching and the future of the world's coral Reefs. Marine and Freshwater Research 50, 839-866.

Hoegh- Guldberg O & Bruno JF (2010). The Impact of Climate Change on the Worlds Marine Ecosystems. Science, 328(5985), 1523-1528. doi:10.1126/science.1189930

Hoegh-Guldberg O, Poloczanska ES, Skirving W, & Dove S (2017). Coral Reef Ecosystems under Climate Change and Ocean Acidification. *Frontiers Marine Science* 4:158. doi: 10.3389/fmars.2017.00158

Hughes TP, Baird AH, Bellwood DR, Card M, Connolly SR, Folke C, Grosberg R, Hoegh-Guldberg O, Jackson JBC, Kleypas J, Lough JM, Marshall P, Nystrom M, Palumbi SR, Pandolfi JM, Rosen B, & Roughgarden J (2003). Climate Change, Human Impacts, & the Resilience of Coral Reefs. *Science* 301(5635), 929-933. DOI:10.1126/science.1085046

Hughes TP, Bellwood DR, Baird AH, Brodie J, Bruno JF, & Pandolfi JM (2011). Shifting base-lines, declining coral cover, and the erosion of reef resilience: comment on Sweatman et al. *Coral Reefs* 30, 653-660. https://doi.org/10.1007/s00338-011-0787-6

Jackson JBC, Donovan MK, Cramer KL, & Lamm VV (eds) (2014). Status and trends of Caribbean coral reefs: 1970-2012. *Global Coral Reef Monitoring Networking*, IUCN, Gland, Switzerland.

Johnston MA, Nutall MF, Eckert RJ, Embesi JA, Slowey NC, Hickerson EL, & Schmahl GP (2013). Long-Term Monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 2009-2010, Volume 1: Technical Report. U.S. Dept. of Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, Louisiana. OCS Study BOEM 2013-214. 202 pp.

Johnston MA, Embesi JA, Ecket RJ, Nuttall MF, Hickerson EL, & Schmahl GP (2016). Persistence of coral assemblages at East and West Flower Garden Banks, Gulf of Mexico. *Coral Reefs,* 35(3), 821-826. doi:10.1007/s00338-016-1452-x

Johnston MA, Sterne TK, Eckert RJ, Nuttall MF, Embesi JA, Walker RD, Hu X, Hickerson EL & Schmahl GP (2017). Long-Term Monitoring at East and West Flower Garden Banks: 2016 Annual Report. Marine Sanctuaries Conservation Series ONMS-17-09. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX. 132 pp.

Johnston MA, Sterne TK, Blakeway RD, MacMillan J, Nuttall X, Hu X, Embesi JA, Hickerson EL, & Schmahl GP (2018a). Long-Term Monitoring at East and West Flower Garden Banks: 2017 Annual Report. Marine Sanctuaries Conservation Series ONMS-18-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX. 124 pp.

Johnston MA, Nuttall MF, Eckert RJ, Blakeway RD, Sterne TK, Hickerson EL, Schmahl GP, Lee MT, MacMillan J, Embesi JA (2018b). Localized coral reef mortality event at East Flower Garden Bank, Gulf of Mexico. *Bulletin of Marine Science* 95(2): 239-250. https://doi.org/10.5343/bms.2018.0057

Johnston MA, Hickerson EL, Nuttall MF, Blakeway RD, Sterne TK, Eckert RJ, & Schmal GP (2019). Coral Bleaching and Recovery from 2016 to 2017 at East and West Flower Garden Banks, Gulf of Mexico. *Coral Reefs* **38**, 787-799 https://doi.org/10.1007/s00338-019-01788-7

Johnston MA, Blakeway RD, O'Connell K, MacMillan J, Nuttall MF, Hu X, Embesi JA, Hickerson EL, & Schmahl GP (2020). Long-Term Monitoring at East and West Flower

Garden Banks: 2018 Annual Report. National Marine Sanctuaries Conservation Series ONMS-20-09. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX. 124 pp.

Johnston MA, O'Connell K, Blakeway RD, MacMillan J, Nuttall MF, Hu X, Embesi JA, Hickerson EL, & Schmahl GP (2021). Long-Term Monitoring at East and West Flower Garden Banks: 2019 Annual Report. National Marine Sanctuaries Conservation Series ONMS-21-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX. 88 pp.

Joye SB (2015). Deepwater Horizon, 5 years on. *Science* 349 (6248), 595-593. doi: 10.1126/science.aab4133

Kealoha AK, Doyle SH, Shamberger KEF, Sylvan JB, Hetland RD, DiMarco SF (2019). Localized hypoxia may have caused coral reef mortality at the Flower Garden Banks. *Coral Reefs* 39:119-132. https://doi.org/10.1007//s00338-019-01883-9

Kohler KE & Gill SM (2006). Coral Point Count with Excel extensions (CPCe): A Visual Basic Program for the determination of coral and substrate coverage using random point Count methodology. *Computers & Geosciences, 32*(9), 1259-1269.doi:10.1016/j.cageo. 2005.11.009

Kujawinski EB, Reddy CM, Rodgers RP, Thrash JC, Valentine DL, White HK (2020). The first decade of scientific insights from the Deepwater Horizon oil release. *Nat Rev Earth Environ* 1, 237-250. https://doi.org/10.1038/243017-020-0046-x

Lawrence DM Scuba Diver Access and Stressed-Reef Ecosystems in Florida Keys National Marine Sanctuary.

Le Henaff M, Muller-Karger FE, Kourafalou VH, Otis D, Johnson KA, McEachron L, & Kang H (2019). Coral mortality event in the Flower Garden Banks of the Gulf of Mexico in July 2016: Local hypoxia due to cross-shelf transport of coastal flood waters? *Continental Shelf Research* 190, 2019, 103988. https://doi.org/10.1016/j.csr.2019.103988

Lesser MP, Slattery M, & Leichter JJ (2009). Ecology of mesophotic coral reefs. *Journal of Experimental Marine Biology and Ecology* 375 (2009) 1-8. doi: 10.1016/j.jembe.2009.05.009

Lesser MP, Slattery M, & Mobley CD (2018). Biodiversity and Functional Ecology of Mesophotic Coral Reefs. *Annual Review of Ecology, Evoloution, and Systematics.* 49:49-71. https://doi.org/10.1146/annurev-ecolsys-110617-062423

Levesque JC (2011). Commercial fisheries in the northwester Gulf of Mexico: possible implications for conservation management at the Flower Garden Banks National Marine Sanctuary. *ICES Journal of Marine Science* 68(10): 2175-2190. https://doi.org/10.1093/icesjms/fsr155

Loh T, McMurray SE, Henkel TP, Vicente J, & Pawlik JR (2015). Indirect effects of overfishing on Caribbean reefs: sponges overgrow reef-building corals. *PeerJ* 3:e901. https://doi.org/10.7717/peerj.901

Loya Y, Eyal G, Tribitz T, Lesser MP, & Appledoorn R (2016). Theme section on mesophotic coral ecosystems: advances in knowledge and future perspectives. *Coral Reefs.* 35:1-9. doi: 10.1007/s00338-016-1410-7.

Lubchenco J, McNutt MK, Dreyfus G, Murawski AS, Kennedy DM, Anastas PT, Chu S, & Hunter T (2012). Science in support of the *Deepwater Horizon* reponse. PNAS vol. 109 no. 50. doi: 10.1073/pnas.1204729109

Madin JS & Madin E (2015). The full extent of the global coral reef crisis. *Conservation Biology,* 29(6), 1724-1726. http://www.jstor.org/stable/24761085

Manzello DP (2015). Rapid Recent Warming of Coral Reefs in the Florida Keys. *Scientific Reports* 5:16762. doi: 10.1038/srep16762

Meiling S, Muller EM, Smith TB, & Brandt ME (2020). 3D Photogrammetry Reveals Dynamics of Stony Coral Tissue Loss Disease (SCTLD) Lesion Progression Across a Thermal Stress Event. *Frontier Marine Science* 7:597643. doi: 10.3389/fmars.2020.597643

Menza C, Kendall M, & Hile S (2008). The deeper we go the less we know. *Rev. Biol. Trop.* (Int. J. Trop. Biol. ISSN-0034-7744) 56(1): 11-24.

Meyer JL, Castellanos-Gell J, Aeby GS, Hase CC, Ushijima B, & Paul VJ (2019). Microbial Community Shifts Associated with the Ongoing Stony Coral Tissue Loss Disease Outbreak on the Florida Reef Tract. *Frontiers in Microbiology.* 10:2244. doi: 10.3389/fmicb.2019.02244

Middlebrook R, Hoegh-Guldberg O, & Leggat W (2008). The effect of thermal history on the susceptibility of reef-building corals to thermal stress. *The Journal of Experimental Biology* 211, 1050-1056. doi: 10.1242/jeb.013284

Mora C (2008). A clear human footprint in the coral reefs of the Caribbean. *Proc. R. Soc. B* 275: 767-773. http://doi.org/10.1098/rspb.2007.1472

Moran R (1998). Balancing Resource Utilization with Resource Protection: The Flower Garden Banks Marine Sanctuary - A Success Story. *Gulf of Mexico Science* 16 (2). doi: 10.18785/goms.1602.14

Mumby PJ & Steneck RS (2011). The resilience of coral reefs and its implications for reef management. In: Dubinsky Z, Stambler N (eds) Coral Reefs: an ecosystem in transistion.

Springer, Netherlands, pp 509-519

Office of National Marine Sanctuaries (2008). Flower Garden Banks National Marine Sanctuary Condition Report 2008. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 49 pp.

Office of National Marine Sanctuaries (2011). Florida Keys National Marine Sanctuary Condition Report 2011. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 105 pp.

Office of National Marine Sanctuaries (2012). Flower Garden Banks National Marine Sanctuary Regulations Summary. Office of National Marine Sanctuaries, National Oceanic and Atmospheric Administration.

Pandolfi JM, Bradbury RH, Sala E, Hughes TP, Bjorndal KA, Cooke RG, McArdle D, McClenachan L, Newman MJH, Paredes G, Warner RR, & Jackson JBC (2003). Global Trajectories of the Long-Term Decline of Coral Reef Ecosystems. *Science* 301: 955-958. doi: 10.1126/science.1085706

Pattengill-Semmens CV & Gittings SR (2003). A Rapid Assessment of the Flower Garden Banks National Marine Sanctuary (Stony Corals, Algae, & Fishes) *Atoll Research Bulletin* 496: 500-510.

Pollnack RB (2007). Theme section on the "Aspects of interactions between humans and coral reefs." *Coral Reefs* 26, 913-914. https://doi.org/10.1007/s00338-007-0308-9

Precht WF, Robbart ML, Boland GS, & Schmahl GP (2005). Establishment and Initial Analysis of Deep Reef Stations (32-40 M) at the East Flower Garden Bank. Gulf of

Mexico Science, 23(1). Doi:10.18785/goms.2301.10

Putnam HM, Barott KL, Ainsworth TD, & Gates RD (2017). The Vulnerability and Resilience of Reef-Building Corals. *Current Biology* 27(11): R528-R540. https://doi.org/10.1016/j.cub.2017.04.047

Richmond RH (1993). Coral Reefs: Present Problems and Future Concerns Resulting from Anthropogenic Disturbance, *Integrative and Comparative Biology,* Volume 33, Issue 6, December 1993, Pages 524-536, https://doi.org/10.1093/icb/33.6.524

Roberts CM (1995). Effects of Fishing on the Ecosystem Structure of Coral Reefs. *Conservation Biology* 9(5): 988-995. https://doi.org/10.1046/j.1523-1739.1995.9051332.x-i1

Rocha LA, Pinheiro HT, Shepherd B, Papastamatiou YP, Luiz OJ, Pyle RL & Bongaerts P (2018). Mesophotic coral ecosystems are threatened and ecologically distinct from shallow water reefs. *Science* 361(6399), 281-284. doi: 10.1126/science.aaq.1614

Roche RC, Harvey CV, Harvey JJ, Kavanagh AP, McDonald M, Stein-Rostaing R & Turner JR (2016). Recreational Diving Impacts on Coral Reefs and the Adoption of Environmentally Responsible Practices within the SCUBA Diving Industry. *Environmental Management* 58, 107-116. https://doi.org/10.1007/s00267-016-0696-0

Rogers CS (1993). Hurricanes and coral reefs: the intermediate disturbance hypothesis revisited. *Coral Reefs* 12: 127-137. doi: 10.1007/BF00334471

Rogers CS (2013). Coral Reef Resilience through Biodiversity. *Hindawi Publishing Corporation ISRN Oceanography* 2013: 18. https://dx.doi.org/10.5402/2013/739034

Schantz AA, Ladd MC, & Burkepile DE (2020). Overfishing and the ecological impacts of extirpating large parrotfish from Caribbean coral reefs. *Ecological Monographs* 90(2):e01403. https://doi.org/10.1002/ecm.1403

Schmahl GP, Hickerson EL, & Precht WF (2008). Biology and Ecology of Coral Reefs and Coral Communities in the Flower Garden Banks Region, Northwestern Gulf of Mexico. In B.M. Riegel & R.E. Dodge (Eds.), *Coral Reefs of the USA* (pp. 221-261). Dordrecht: Springer Netherland.

Shivlani M, Leeworthy VR, Murray TJ, Suman DO, & Tonioli f (2008). Knowledge, Attitudes and Perceptions of Management Strategies and Regulations of the Florida Keys National Marine Sanctuaries by Commercial Fishers, Dive Operators, and Environmental Group Memebers: A Baseline Characterization and 10-year Comparison. Marine Sanctuaries Conservation Series ONMS-08-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sancturaries, Silver Spring, MD. 170 pp.

Silbiger NJ, Goodbody-Gringley G, Bruno JF, & Putnam HM (2019). Comparative thermal performance of the reef-building coral *Orbicella franksi* at its latitudinal range limits. *Marine Biology* 166:126. doi: 10.1007/s00227-019-3573-6

Smith TB, Blondeau NJ, Calnan JM, Kadison E, & Herzlieb S (2008). Assessing coral reef health across onshore to offshore stress gradients in the US Virgin Islands. *Marine Pollution Bulletin* 56: 1983-1991. doi: 10.1016/j.marpolbul.2008.08.015

Smith TB, Glynn PW, Mate JL, Toth LT, & Gyory J (2014). A depth refugium from catashtrophic coral bleaching prevents regional extinction. *Ecology* 95(6): 1663-1673. doi:10.1890/13-0468.1

Smith TB, Gyory J, Brandt ME, Miller WJ, Jossart J, & Nemeth RS (2016a). Caribbean mesophotic coral ecosystems are unlikely climate change refugia. *Global Change Biology* 22(8): 2756-2765. https://doi.org/10.1111/gcb.13175

Smith TB, Brantneris VW, Canals M, Brandt ME, Martens J, Brewer RS, Kadison E, Kammann M, Keller J & Holstein DM (2016b). Potential Structuring Forces on a Shelf Edge Upper Mesophotic Coral Ecosystem in the US Virgin Islands. *Front. Mar. Sci.* 3:115. doi: 10.3389/fmars.2016.00115

Somerfield PJ, Jaap WC, Clarke KR, Callahan M, Hackett K, Porter J, Lybolt M, Tsokos C, & Yanev G (2008). Changes in coral reef communities among the Florida Keys, 1996-2003. *Coral Reefs* 27: 951-965. doi:10.1007/s00338-008-0390-7

Stefanski SF & Shimshack JP (2016). Valuing Marine Biodiversity in the Gulf of Mexico: Evidence from the Proposed Boundary Expansion of the Flower Garden Banks National Marine Sanctuary. *Marine Rescource Economics* 31(2): 211-232. https://doi.org/10.1086/685109

Stewart JD, Nuttall M, Hickerson EL, & Johnston MA (2018). Important juvenile manta ray habitat at Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico. *Marine Biology* 165:111. doi:10.1007/s00227-018-3364-5

Thailing CE & Ditton RB (2003). Demographics, Motivations, and Participation Patterns of Sport Divers in the Flower Garden Banks National Marine Sanctuary, *Department of Wildlife & Fisheries Sciecnes Texas A&M University,* 338-348.

Toth LT, van Woesik R, Murdoch TJT, Smith SR, Ogden JC, Precht WF, & Aronson RB (2014). Do no-take reserves benefit Florida's corals? 14 years of change and stasis in the Florida

Keys National Marine Sanctuary. *Coral Reefs* 33:565-577. doi: 10.1007/s00338-014-1158-x

Turner JA, Thomson DP, Cresswell AK, Trapon M, & Babcock RC (2018). Depth-related patterns in coral recruitment across a shallow to mesphotic gradient. *Coral Reefs* 37: 711-722. doi: 10.1007/s00338-018-1696-8

Valentine JF & Heck KL (2005). Perspective review of the impacts of overfishing on coral reef food web linkages. *Coral Reefs* 24: 209-213. doi: 10.1007/s00338-004-0468-9

White HK, Hsing PY, Cho W, Shank TM, Cordes EE, Quattrini AM, Nelson RK, Camilli R, Demopoulos AWJ, German CR, Brooks JM, Roberts HH, Shedd W, Reddy CM & Fisher CR (2012). Impact of the Deepwater Horizon oil spill on a deep-water coral community in the Gulf of Mexico. *PNAS* 109(50): 20303-20308. https://doi.org/10.1073/pnas.1118029109

Wilson SK, Graham NAJ, Pratchett MS, Jones GP, & Polunin NVC (2006). Multiple disturbances and the global degradation of coral reefs: are reef fishes at risk or resilient? *Global Change Biology* 12: 2220-2234. doi: 10.1111/j.1365-2486.2006.01252.x

Wilson SK, Fisher R, Pratchett MS, Graham NAJ, Dulvy NK, Turner RA, Cakacakas A, Polunin NVC, & Rushton SP (2008). Exploitation and habitat degradation as agents of change within coral reef fish communities. *Global Change Biology* 14: 2796-2809. doi: 10.1111/j.1365-2486.2008.01696.x

Wilson SK, Robinson JPW, Chong-Seng K, Robinson J, & Graham NAJ (2019). Boom and bust of keystone structure on coral reefs. *Coral Reefs* 38:625-635. doi: 10.1007/s00338-019-01818-4

Wright RM, Correa AMS, Quigley LA, Santiago-Vasquez LZ, Shamberger KEF, & Davies SW (2019). Gene Expression of Endangered Coral (*Orbicella spp.*) in Flower Garden Banks National Marine Sanctuary After Hurricane Harvey. *Front. Mar. Sci.* 6:672. doi :10.3389/fmars.2019.00672

APPENDIX A

APPENDIX A

Table displaying the various light and camera setups and data analysis utilized for the long-term monitoring program at East and West FGB.

Year	Equipment
1989	Nikonos V camera, 15 mm wide angle lens, two 225 watt-second strobes, 36 exposure, 100 ASA Kodacolor VRG or Fujicolor print fim
1991	Nikonos V camera, 15 mm wide angle lens, two 225 watt-second strobes, 36 exposure, 100 ASA Kodacolor VRG or Fujicolor print fim
1992	Nikonos V camera, 15 mm wide angle lens, two 225 watt-second strobes, 36 exposure, 100 ASA Kodacolor VRG or Fujicolor print fim
1994	Nikonos V camera, 15 mm lens and two Ikelite 225 watt-second strobes
1995	Nikonos V camera, 15 mm lens and two Ikelite 225 watt-second strobes
1996	Nikonos V camera, 15 mm lens and two Ikelite 225 watt-second strobes
1997	Nikonos V camera, 15 mm lens and two Ikelite 225 watt-second strobes
1998	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
1999	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
2000	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
2001	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
2002	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
2003	Nikonos V camera, Kodak Ektachrome 100, 36-exposure slide film, 15 mm lens, and two Ikelite 225 watt-second strobes
2004	Nikonos V camera, Kodak Ektachrome or EliteChrome 200 ASA, 36-exposure slide film, 15 mm lens, and two Ikelite 75 watt-second strobes
2005	Nikonos V camera, Kodak Ektachrome or EliteChrome 200 ASA, 36-exposure slide film, 15 mm lens, and two Ikelite 75 watt-second strobes
2006	Nikonos V camera, Kodak Ektachrome or EliteChrome 200 ASA, 36-exposure slide film, 15 mm lens, and two Ikelite 75 watt-second strobes
2007	Nikonos V camera, Kodak Ektachrome or EliteChrome 200 ASA, 36-exposure slide film, 15 mm lens, and two Ikelite 75 watt-second strobes
2008	Sea&Sea DX-1G digital camera and underwater housing, two Inon D-2000S strobes and a Sea&Sea Wide Angle Lens
2009	Nikon Coolpix P5000 camera in Ikelite housing, Inon UWL-100 Type 2 wet mount wide-angle converter lens, and two Ikelite DS125 strobes
2010	Canon Power Shot G11 digital camera, FIX Fish-Eye housing with 165 dome port, two Inon Z240 strobes
2011	Nikon D300 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2012	Nikon D300 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2013	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2014	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2015	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2016	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2017	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2018	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes
2019	Nikon D7000 SLR digital camera, 16 mm lens in Sea&Sea housing with small dome port, and two Inon Z240 strobes

Year	Analysis
1989	images were analysed using mylar traces and calibrated planimeter
1991	images were analysed using mylar traces and calibrated planimeter
1992	paper templates of photos with 100 randomly located dots
1994	paper templates of photos with 100 randomly located dots
1995	paper templates of photos with 100 randomly located dots
1996	projecting image onto a flat surface, then using 24 transparent overlays containg 100 randomly positions crosses
1997	projecting image onto a flat surface, then using 24 transparent overlays containg 100 randomly positions crosses
1998	slides were projected onto a flat surface and each was overlain with 3 sets of 100 randomly positions markers
1999	slides were projected onto a flat surface and each was overlain with 3 sets of 100 randomly positions markers
2000	slides were projected onto a flat surface and each was overlain with 3 sets of 100 randomly positions markers
2001	slides were projected onto a flat surface and each was overlain with 3 sets of 100 randomly positions markers
2002	CPCe overlaid with 300 random dots on the image
2003	CPCe overlaid with 300 random dots on the image
2004	CPCe overlaid with 300 random dots on the image
2005	CPCe overlaid with 300 random dots on the image
2006	CPCe overlaid with 300 random dots on the image
2007	CPCe overlaid with 300 random dots on the image
2008	CPCe overlaid with 300 random dots on the image
2009	CPCe overlaid with 100 random dots on the image
2010	CPCe overlaid with 100 random dots on the image
2011	CPCe overlaid with 100 random dots on the image
2012	CPCe overlaid with 100 random dots on the image
2013	CPCe overlaid with 100 random dots on the image
2014	CPCe overlaid with 100 random dots on the image
2015	CPCe overlaid with 100 random dots on the image
2016	CPCe overlaid with 100 random dots on the image
2017	CPCe overlaid with 100 random dots on the image
2018	CPCe overlaid with 100 random dots on the image
2019	CPCe overlaid with 100 random dots on the image

APPENDIX B

APPENDIX B

Original dataset compared the condensed dataset used to run statistical analysis. The seven different colors display the species that were condensed down to a genus level, other identified coral and the unknown corals.

Original Dataset	**Condensed Dataset**
Agaricia agaricites	*Agaricia spp*
Agaricia fragilis	*Colpohyllia spp*
Agaricia lamarcki	*Madracis spp*
Agaricia sp	*Millepora alcicornis*
Colpophyllia amaranthus	*Montastraea cavernosa*
Colpophyllia natans	*Mussa angulosa*
Colpophyllia sp	*Orbicella annularis*
Madracis auretenra	*Orbicella faveolata*
Madracis decactis	*Orbicella franksi*
Madracis sp	*Orbicella spp*
Millepora alcicornis	*Porites spp*
Montastraea cavernosa	*Pseudodiploria strigosa*
Mussa angulosa	*Siderastrea spp*
Orbicella annularis	*Stephanocoenia intersepta*
Orbicella faveolata	Other Identified Coral
Orbicella franksi	Unknown Coral
Orbicella spp	
Porites astreoides	
Porites furcata	
Pseudodiploria strigosa	
Siderastrea radians	
Siderastrea siderea	
Siderastrea sp	
Stephanocoenia intersepta	
Scolymia cubensis	
Rhizosmilia maculata	
Dichocoenia stokesi	
Tubastraea coccinea	
Unknown Coral	
Unknown Coral I	
Unknown Coral II	
Unknown Coral III	

BIOGRAPHICAL SKETCH

Rebekah A. Hernandez attended high school at Edinburg High School in Edinburg, Texas from 2009 to 2013, where she earned her high school diploma. After graduating high school, Rebekah attended The University of Texas Rio Grande Valley, formerly known as The University of Texas Pan American from 2013-2016, where she earned her Bachelor of Science in Biology. Rebekah began work at an Emergency Veterinary clinic in McAllen, Texas, where she worked as veterinary technician. After working at the clinic for 5 months, Rebekah started work on her Master of Science in Ocean, Coastal, and Earth Sciences at The University of Texas Rio Grande Valley, earning her degree in August 2021. Rebekah can be reached by mail at P.O. Box 1852, Edinburg, TX 78540 or by email at rebekah.hernandez@hotmail.com.