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National Oceanic and Atmospheric Administration
National Marine Sanctuaries Conservation Science Series

Long-Term Monitoring at East and West Flower Garden Banks: 2019 Annual Report

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A manta ray (*Mobula cf. birostris*) swims over the long-term monitoring study site at East Flower Garden Bank. Photo: G.P. Schmahl/NOAA

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The Office of National Marine Sanctuaries, part of the National Oceanic and Atmospheric Administration, serves as the trustee for a system of underwater parks encompassing more than 600,000 square miles of ocean and Great Lakes waters. The 14 national marine sanctuaries and two marine national monuments within the National Marine Sanctuary System represent areas of America's ocean and Great Lakes environment that are of special national significance. Within their waters, giant humpback whales breed and calve their young, coral colonies flourish, and shipwrecks tell stories of our nation's maritime history. Habitats include beautiful coral reefs, lush kelp forests, whale migration corridors, spectacular deep-sea canyons, and underwater archaeological sites. These special places also provide homes to thousands of unique or endangered species and are important to America's cultural heritage. Sites range in size from less than one square mile to almost 583,000 square miles. They serve as natural classrooms and cherished recreational spots, and are home to valuable commercial industries.

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

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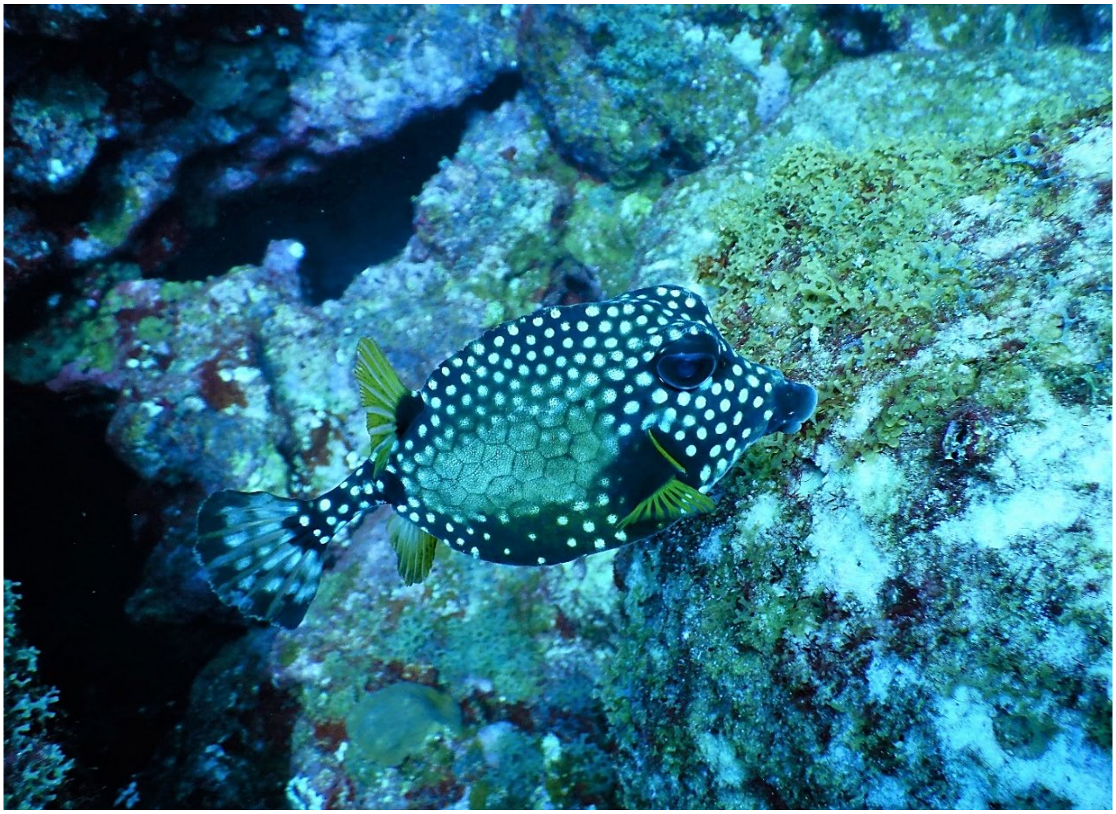
Abstract

This report summarizes fish and benthic community observations and water quality data collected from East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB) in 2019, along with nearly 30 years of historical monitoring data. EFGB and WFGB are part of Flower Garden Banks National Marine Sanctuary (FGBNMS), located in the northwestern Gulf of Mexico. The annual long-term monitoring program began in 1989 and is funded by the National Oceanic and Atmospheric Administration's FGBNMS and the Bureau of Ocean Energy Management, with support from the National Marine Sanctuary Foundation. In 2019, mean coral cover was 55% within the EFGB one-hectare study site and 60% within the WFGB one-hectare study site. Mean macroalgae cover was 33% within the EFGB one-hectare study site, which differed significantly from the 21% mean macroalgae cover within the WFGB one-hectare study site. Mean coral cover has increased significantly at WFGB and remained stable at EFGB since 1989. Mean macroalgae cover has increased significantly at both banks since 1999. Mean coral cover within repetitive photostations has increased significantly since 1989 at both banks. The *Orbicella* spp. complex, listed as threatened under the Endangered Species Act, accounted for the majority of the coral cover within the one-hectare study sites. Sea urchin density was 1.25 and 28.08 individuals per 100 m^2 at the EFGB and WFGB one-hectare study sites, respectively. The reef fish community was comprised primarily of the families Labridae and Pomacentridae. For commercially and recreationally important species, grouper density was higher within the EFGB one-hectare study site while snapper density was higher within the WFGB one-hectare study site. During 2019, water temperatures on the reef exceeded 30°C for six non-continuous days at EFGB and ten non-continuous days at WFGB. Coral bleaching at both banks was less than 1% at the time of surveys. A significant monotonic increasing trend in seawater temperature was detected at both banks from 1990 to 2019, indicating ocean temperatures have risen at FGBNMS over the past three decades. The results of this report highlight the importance of long-term monitoring efforts by providing one of the longest records of coral reef health in the Gulf of Mexico and Caribbean region.

Key Words

benthic community, coral ecosystem, coral reef, fish community, long-term monitoring, Flower Garden Banks National Marine Sanctuary, Gulf of Mexico, marine protected area, water quality

Executive Summary



A juvenile smooth trunkfish (*Lactophrys triqueter*) swims over the reef at East Flower Garden Bank. Photo: Jimmy MacMillan/CPC

Since 1989, a federally supported long-term coral reef monitoring program has focused on two one-hectare study sites at East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB) in the northwestern Gulf of Mexico (Figure ES.1). In 30 years of nearly continuous monitoring, mean live coral cover has, on average, oscillated around 52% within the one-hectare study sites at both banks, with a low of 37% at WFGB in 1992 and a high of 66% at WFGB in 2010. Despite global coral reef declines in recent decades, EFGB and WFGB have suffered minimally from hurricanes, recovered from coral bleaching events, and shown no signs of disease, with the exception of a localized mortality event at EFGB in 2016.

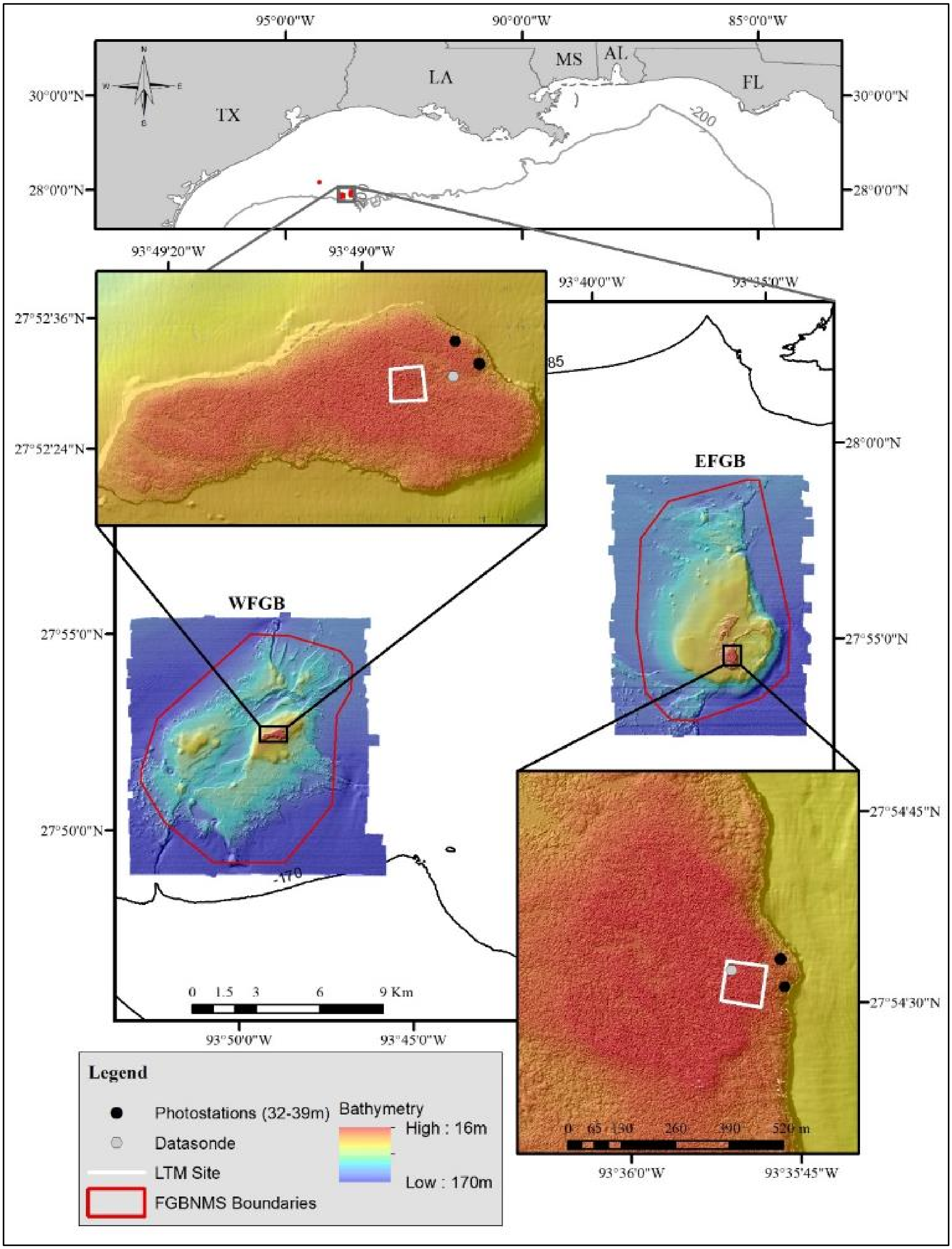


Figure ES.1. Bathymetric map of EFGB and WFGB, with inset of the Gulf of Mexico coastline, long-term monitoring (LTM) one-hectare study sites, and repetitive photostation locations that range in depth from 32–39 m.

This report summarizes fish and benthic community observations and water quality data from 2019, as well as nearly 30 years of historical monitoring data. The benthic and fish community surveys were conducted within one-hectare study sites at EFGB and WFGB by a team of multidisciplinary scientists using random transects to document components of benthic cover, surveys for sea urchins and lobster, and reef fish visual census surveys to examine fish population composition. Repetitive photostations (ranging in depth from 18–39 m) documented changes in the composition of benthic assemblages at specific locations. The annual long-term monitoring program is jointly funded by the National Oceanic and Atmospheric Administration's Flower Garden Banks National Marine Sanctuary (FGBNMS) and the Bureau of Ocean Energy Management (BOEM), with support from the National Marine Sanctuary Foundation. Key findings from 2019, as well as historical trends from 1989 to 2019, are described herein.

Living coral, followed by macroalgae, is the principal benthic community component on the coral reef cap at EFGB and WFGB. In 2019, living coral cover was 55% and 60% in the EFGB and WFGB one-hectare study sites (17–27 m), respectively (57% for both sites combined). Mean coral cover has increased significantly at WFGB and remained stable at EFGB since 1989. Mean macroalgae cover has increased significantly at both banks since 1999, averaging 30% of the benthic cover since 2009 (Figure ES.2).

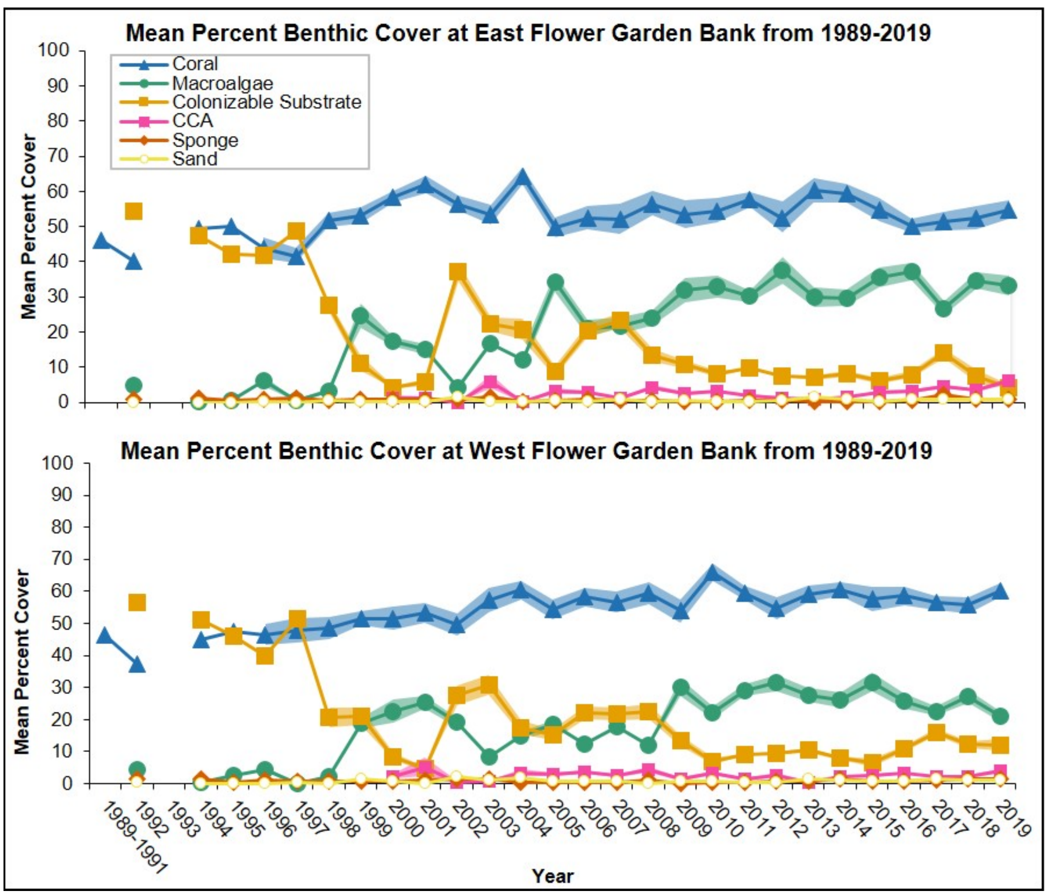


Figure ES.2. Mean percent benthic cover ± SE bands from random transect surveys within EFGB and WFGB one-hectare study sites from 1989 to 2019.

A total of 12 coral species were documented in one-hectare study site surveys at EFGB and 13 at WFGB. *Orbicella franksi* had the highest mean coral cover within EFGB (28%) and WFGB (35%) one-hectare study sites, followed by *Pseudodiploria strigosa* (8% for both EFGB and WFGB), *Porites astreoides* (6% for both EFGB and WFGB), *Orbicella faveolata* (EFGB 5%, WFGB 1%), and *Colpophyllia natans* (2% for both EFGB and WFGB). The *Orbicella* species complex, including *O. franksi, O. faveolata*, and *Orbicella annularis* (all of which are listed as threatened species under the Endangered Species Act), made up 60% of the observed coral species within the EFGB one-hectare study site and 66% of the observed coral species within the WFGB one-hectare study site.

Orbicella franksi had the highest mean coral cover within EFGB (28%) and WFGB (35%) one-hectare study sites, followed by *Pseudodiploria strigosa* (8% for both sites), *Porites astreoides* (6% for both sites), *Orbicella faveolata* (EFGB 5%, WFGB 1%), and *Colpophyllia natans* (2% for both sites). The *Orbicella* species complex, including *O. franksi, O. faveolata*, and *Orbicella annularis* (all of which are listed as threatened species under the Endangered Species Act), made up 60% of the total coral cover within EFGB surveys and 66% of the total coral cover within WFGB surveys.

Percent coral cover increased with depth in permanent repetitive photostations (18–39 m). Percent coral cover ranged from 24–94% at EFGB photostations (n=60) and 42–96% at WFGB photostations (n=65) in 2019. Of the 25 coral species common to the reef caps, 15 coral species were observed in EFGB repetitive photostations and 14 were observed in WFGB repetitive photostations. Coral species composition within photostations changed slightly with depth (e.g., *O. franksi* and *Montastraea cavernosa* were the most abundant species at increased depths). Less than 1% of the coral cover analyzed was pale or bleached, although surveys occurred at a time of year when coral bleaching is not typically noted at EFGB and WFGB, and water temperatures and exposure times were lower than threshold levels known to trigger bleaching. Twenty-four EFGB photostations and 27 WFGB photostations have been in place in the one-hectare study sites since the beginning of the monitoring program. Significant increases in coral cover at photostations were observed from 1989 to 2019 (Figure ES.3). Despite higher coral cover in repetitive photostations compared to random transect surveys, these sites are critical in enabling researchers to track individual colonies and specific sites over time (especially during extreme events), are indicative of broad, reef-wide trends, and complement random benthic survey data to provide a more complete long-term monitoring program.

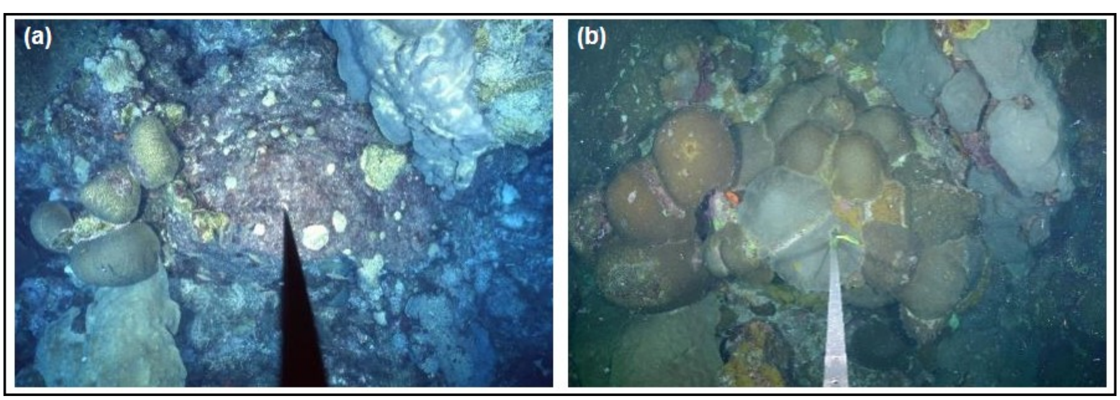


Figure ES.3. Time series of EFGB repetitive photostation #102 at a depth of 20 m from (a) 1989 to (b) 2019. Photos: (a) MMS (Gittings et al. 1992), (b) NOAA

Long-spined sea urchin (*Diadema antillarum*) populations within the EFGB one-hectare study site have remained low (ranging from 0–2 per 100 m^2) since sea urchin monitoring surveys were first conducted in 2004, but densities within the WFGB one-hectare study site (1–28 per 100 m^2) were significantly higher than EFGB through 2019. Since lobster surveys began in 2004, Caribbean spiny lobster (*Panulirus argus*) and spotted spiny lobster (*Panulirus guttatus*) counts ranged from zero to two individuals per 100 m^2.

A total of 30 families and 84 fish species were recorded in 2019 surveys. These surveys indicated that Labridae (wrasses and parrotfish) and Pomacentridae (damselfish) were the predominant fish families observed. Bonnetmouth (*Emmelichthyops atlanticus*) were the most abundant species at both banks in 2019. Mean fish density and biomass were greater within the EFGB one-hectare study site (327.33 ± 77.74 individuals/100 m^2 and 7,091.15 ± 1,923.40 g/100 m^2, respectively) than the WFGB one-hectare study site (312.05 ± 47.18 individuals/100 m^2 and 6,035.98 ± 1,090.06 g/100 m^2, respectively), and piscivores had the greatest mean biomass in all surveys. For commercially and recreationally important species, mean grouper biomass was 123.40 ± 73.21 g/100 m^2 within the EFGB one-hectare study site and 58.94 ± 25.07 g/100 m^2 within the WFGB one-hectare study site. Mean snapper biomass was 70.00 ± 50.51 g/100 m^2 within the EFGB one-hectare study site and 353.32 ± 167.14 g/100 m^2 within the WFGB one-hectare study site. For the first time since 2013, when invasive lionfish (*Pterois volitans*) were first documented in surveys, lionfish were not observed in surveys in 2019; however, divers did observe them on the reef. The non-native regal demoiselle (*Neopomacentrus cyanomos*) was observed in surveys (0.72 individuals/100 m^2 at both banks) for the second consecutive year.

Water quality instruments were located in sand flats (23 m depth at EFGB and 27 m depth at WFGB) at each bank. At these sites, mean seawater temperatures ranged from 20.73°C to 30.21°C at EFGB and 20.41°C to 30.19°C at WFGB. Significantly increasing monotonic trends at both banks from 1990 to 2019 were detected, indicating warming ocean temperatures at FGBNMS over the past three decades (Figure ES.4).

Daily mean salinity ranged from 32.64 psu to 36.54 psu at EFGB (23 m) and 33.85 psu to 36.56 psu at WFGB (27 m). Lower salinity values occurred in June and July, most likely due to freshwater river runoff extending to the outer continental shelf. Nutrients from seawater sampled quarterly (chlorophyll-*a*, ammonia, nitrate, nitrite, phosphorous, and total Kjeldahl nitrogen) were below detectable limits at both banks. Carbonate chemistry indicated that the carbonate system was thermally controlled with clear seasonality (highest dissolved inorganic carbon values in February, highest $p CO_2$ values in August, and highest $\Omega_{aragonite}$ values in November) within the water column around FGBNMS.

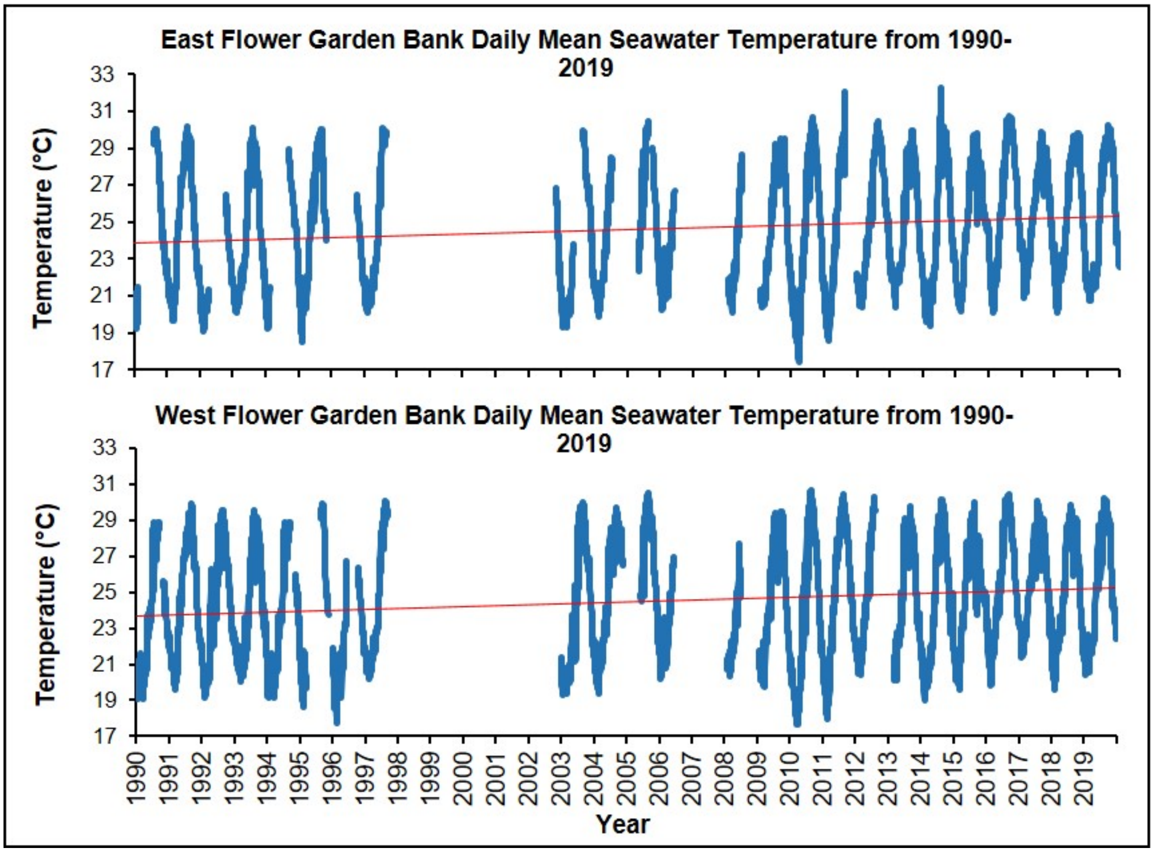


Figure ES.4. Daily mean seawater temperature (ºC) demonstrating seasonal variation at EFGB (23 m depth) and WFGB (27 m depth) and a significant increase over time (red trend line) from 1990 to 2019.

Overall, some of the most important trends documented since monitoring began in 1989 have been stable coral cover at EFGB and significantly increasing coral cover at WFGB, significantly increasing macroalgae cover at both banks, significantly increasing sea urchin populations at WFGB, and significantly increasing seawater temperatures at reef depth. In contrast to many other reefs in the Gulf of Mexico and Caribbean region, while macroalgae at EFGB and WFGB has increased, coral cover has not declined. The high coral cover documented at EFGB and WFGB since the beginning of the monitoring program makes these banks unique among the region's coral reefs and justifies the need for continued protection. Sustained monitoring will allow researchers to document changes in reef community condition, link changes to oceanographic events, and compare to historical baselines. This level of monitoring enables resource managers to make informed decisions regarding management and research amid threats such as climate change, invasive species, water quality degradation, and natural disturbances such as storms.

Chapter 1: Long-Term Monitoring at East and West Flower Garden Banks

The spiral plumes from Christmas tree worms (*Spirobranchus giganteus*) on a symmetrical brain coral (*Pseudodiploria strigosa*) colony at West Flower Garden Bank. Photo: Kelly Drinnen/NOAA

Habitat Description

The coral reef-capped East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB) are part of a discontinuous arc of reef environments along the outer continental shelf in the northwestern Gulf of Mexico (Bright et al. 1985) (Figure 1.1). These reefs occupy elevated salt domes located approximately 190 km south of the Texas and Louisiana border, containing several distinct habitats ranging in depth from 16–150 m (Bright and Rezak 1976; Schmahl et al. 2008) (Figure 1.1).

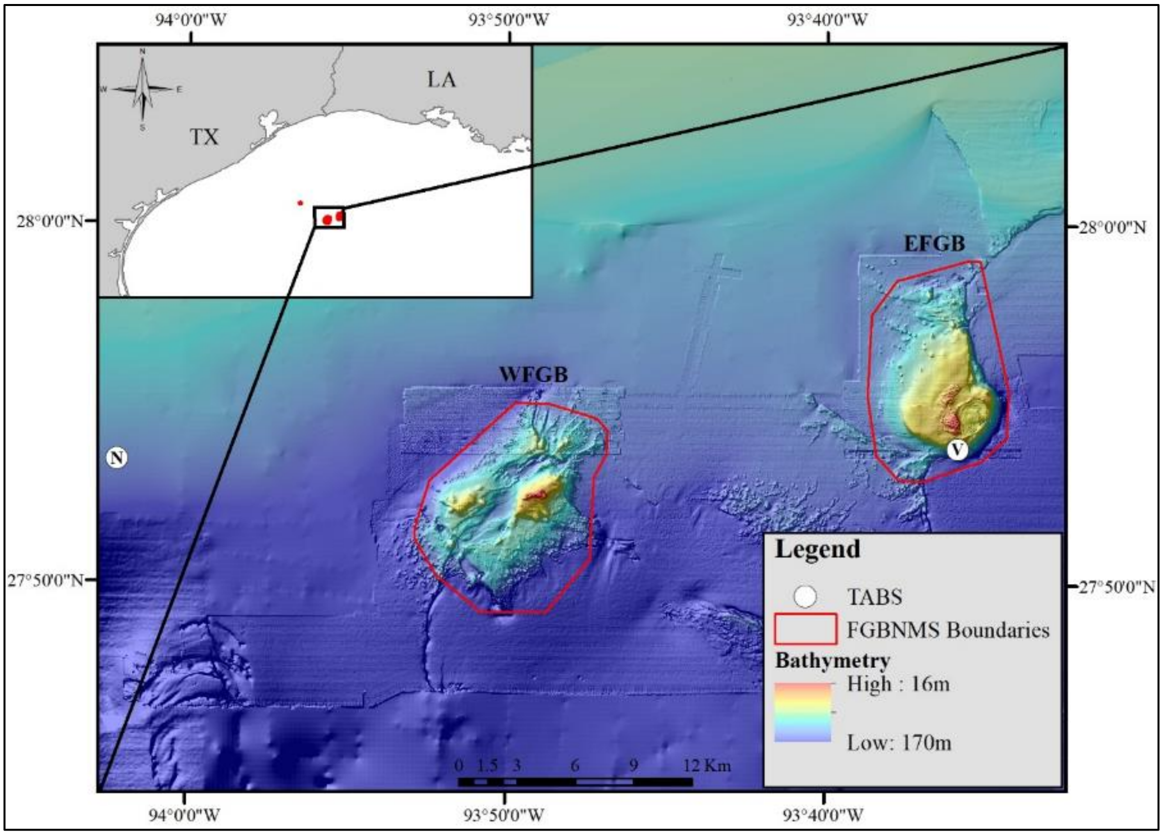


Figure 1.1. Map of EFGB and WFGB, Texas automated buoy system (TABS) buoys, and inset of the Texas-Louisiana border with banks and other topographic features along the continental shelf of the northwestern Gulf of Mexico.

The caps of the banks are approximately 20 km apart and within the photic zone, where conditions are ideal for colonization by species of corals, algae, invertebrates, and fish that are also found in the Caribbean region (Goreau and Wells 1967; Schmahl et al. 2008; Clark et al. 2014; Johnston et al. 2016a). The shallowest portions of each bank are topped by well-developed coral reefs in depths ranging from 16–40 m. Although the coral species found on the EFGB and WFGB reef caps are the same as species on Caribbean reefs, octocorals are absent and scleractinian corals of the genus *Acropora* are rare. These differences are likely due to depth and the latitude of the banks; FGBNMS is near the northernmost limit of the coral distribution range and is distanced from source populations (Bright et al. 1985; CSA 1989).

Long-Term Monitoring Program History

In the 1970s, due to concerns about potential impacts from offshore oil and gas development, the Department of Interior (initially through the Bureau of Land Management, then the Minerals Management Service [MMS], and now the Bureau of Ocean Energy Management [BOEM]) has supported monitoring at EFGB and WFGB to collect data and determine if the reefs are impacted by nearby oil and gas activities (Figure 1.2).

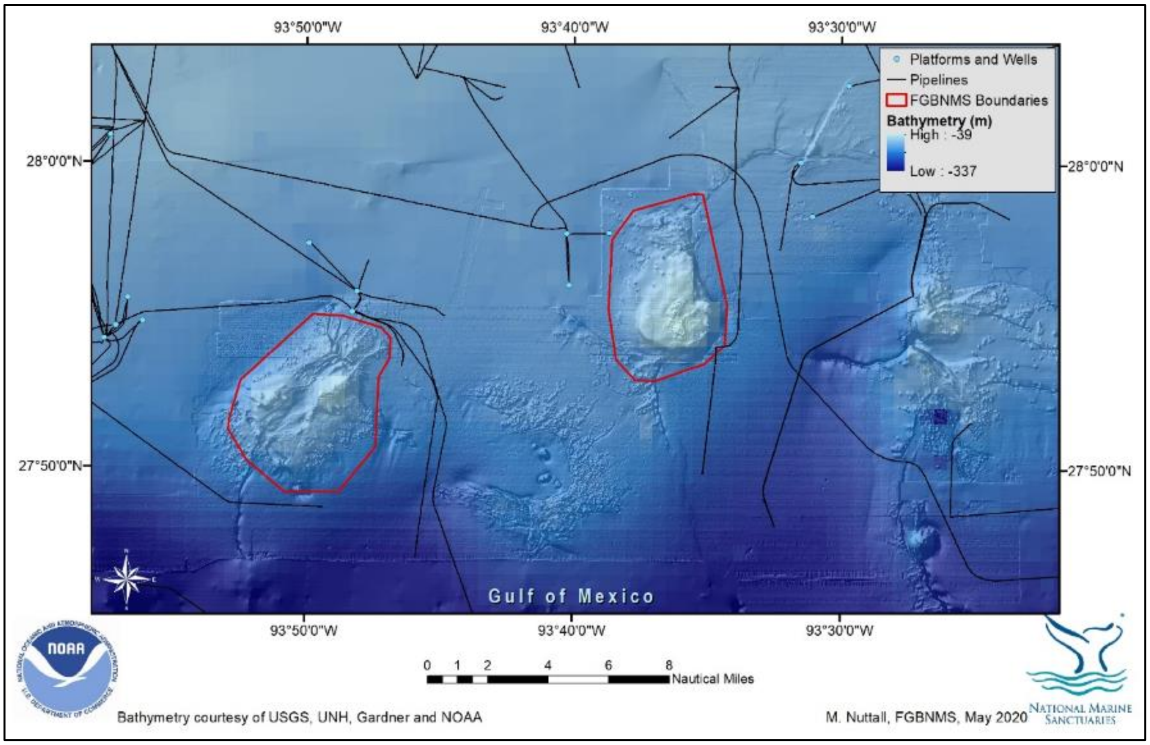


Figure 1.2. Map of oil and gas platforms, wells, and pipelines near EFGB and WFGB. FGBNMS boundaries are outlined in red.

First under industry funding, then MMS funding and a contract with Texas A&M University (TAMU), one-hectare long-term monitoring study sites were established in 1989, marking the official start of the Flower Garden Banks Long-Term Monitoring (LTM) program (CSA 1989; Gittings et al. 1992) (Figure 1.3). Flower Garden Banks National Marine Sanctuary (FGBNMS) was established in 1992 (15 CFR Part 992 § 922.120). Monitoring was conducted by both TAMU and environmental consulting groups through competitive contracts until 2009, at which time BOEM and the National Oceanic and Atmospheric Administration (NOAA) established an interagency agreement for FGBNMS to carry out the LTM program.

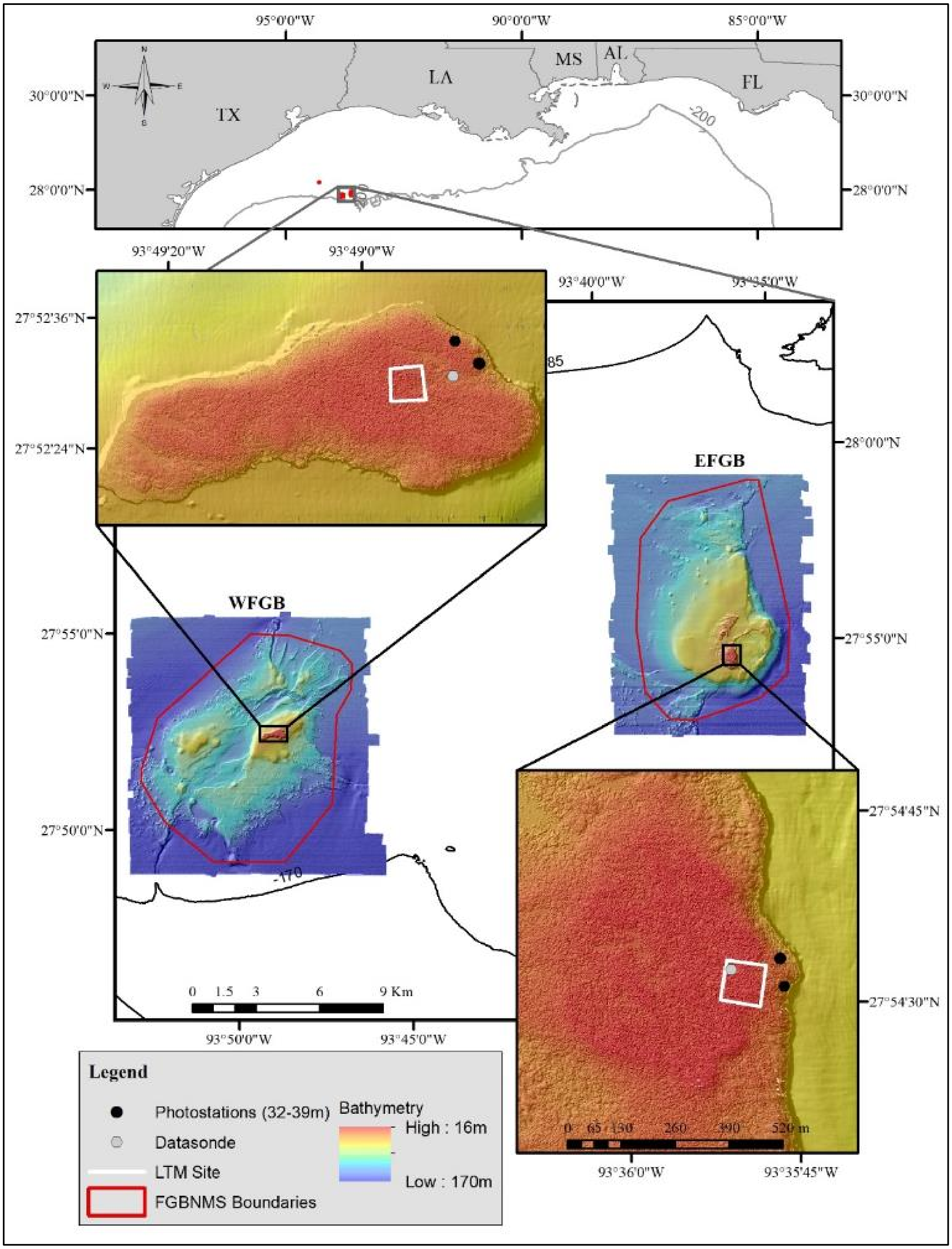


Figure 1.3. Bathymetric map of EFGB and WFGB, with inset of the Gulf of Mexico coastline, long-term monitoring (LTM) one-hectare study sites, and repetitive photostation locations that range in depth from 32–39 m.

Long-Term Monitoring Program Objectives

Priorities of FGBNMS include managing natural resources as stated in the National Marine Sanctuaries Act and identifying coral reef threats and potential sources of impacts, including: overfishing, pollution, runoff, visitor impacts, disease, bleaching, invasive species, hurricanes, and oil and gas exploration and extraction. Knowing the condition of natural resources within the national marine sanctuary and providing scientifically credible data is fundamental to NOAA's ability to protect and manage these areas and evaluate management actions.

Through the interagency agreement, the LTM program is of significant interest to both NOAA and BOEM, who share responsibility to protect and monitor these important marine resources. The five objectives and subsequent indicators of the FGBNMS LTM program include:

- Monitor and evaluate environmental changes and variability in abundances of reef-associated organisms across multiple time scales
    - Indicators: Benthic percent cover, fish community dynamics, water quality, and coral demographic analyses
- Identify changes in coral reef health resulting from both natural and human-induced stressors to facilitate management responses
    - Indicators: Bleaching, disease, and invasive species
- Provide a resource to facilitate adaptive management of activities impacting reef-related resources
    - Indicators: Baseline data and image archive of damage to resources if observed
- Identify and monitor key species that may be indicative of reef and ecosystem health
    - Indicators: Sea urchin and lobster density
- Provide a consistent and timely source of data monitoring environmental conditions and the status of living marine sanctuary resources
    - Indicators: Published, peer-reviewed annual reports

Long-Term Monitoring Program Components

The LTM program was designed to assess the health of the coral reefs, detect change over time, and provide baseline data in the event that natural or human-induced activities endanger the integrity of EFGB and WFGB coral communities. The high coral cover and robust fish populations compared to other reefs in the region, combined with historical data collection and the proximity to oil and gas infrastructure development, make EFGB and WFGB ideal sentinel sites for continued monitoring. The following techniques are used in this monitoring program to evaluate coral reef diversity, growth rates, and community health in designated monitoring areas at each bank:

- Random photographic transects document benthic cover;
- Repetitive photostations detect and evaluate long-term changes at the stations and in individual coral colonies while controlling for small-scale environmental heterogeneity;
- Biennial coral demographic surveys provide information on recruitment, coral density, and coral colony size;
- Stationary reef fish visual census surveys assess community structure of coral reef fishes;

- Long-spined sea urchin (*Diadema antillarum*) and lobster (*Panulirus argus* and *Panulirus guttatus*) surveys establish current population levels and trends;
- Water quality datasondes record salinity, temperature, and turbidity at depth; and
- Quarterly nutrient sampling documents chlorophyll *a*, ammonia, nitrate, nitrite, total Kjeldahl nitrogen, and phosphorous levels.

Long-Term Monitoring Data Collection

The long-term monitoring study area consists of several locations on the EFGB and WFGB coral reef cap where benthic, fish, and water quality data are collected. Long-term monitoring data have been collected annually during summer months since 1989 in permanent 10,000 m² study sites (100 m x 100 m or 1 hectare; hereafter referred to as "one-hectare study sites") at EFGB and WFGB. The corners and centers of the one-hectare study sites are marked by large eyebolts as reference markers. Within the one-hectare study sites, depths range from 17–27 m at EFGB and 18–25 m at WFGB (Figures 1.4 and 1.5). Permanent mooring buoy anchors (mooring buoy#2 at EFGB and mooring buoy#5 at WFGB) have been established near the one-hectare study site centers to facilitate field operations (Table 1.1). Additionally, permanent repetitive photostations were installed at each bank beyond the one-hectare study site boundaries to capture benthic cover in depth ranges of 32–39 m: twenty-three repetitive photostations at EFGB are located east of buoy#2 and twenty-four repetitive photostations at WFGB are located north of buoy#2 (Figures 1.4 and 1.5). Water quality datasondes are located near buoy#2 at EFGB and buoy#2 at WFGB (Figures 1.4 and 1.5). Additional temperature loggers at 30 m and 40 m are paired with repetitive photostations at these depths at EFGB and WFGB (Figures 1.4 and 1.5).

Table 1.1. Coordinates and depths for permanent moorings within one-hectare study sites at each bank.

Mooring	Lat (DDM)	Long (DDM)	Depth (m)
EFGB Mooring #2	27° 54.516 N	93° 35.831 W	19.2
WFGB Mooring #5	27° 52.509 N	93° 48.900 W	20.7

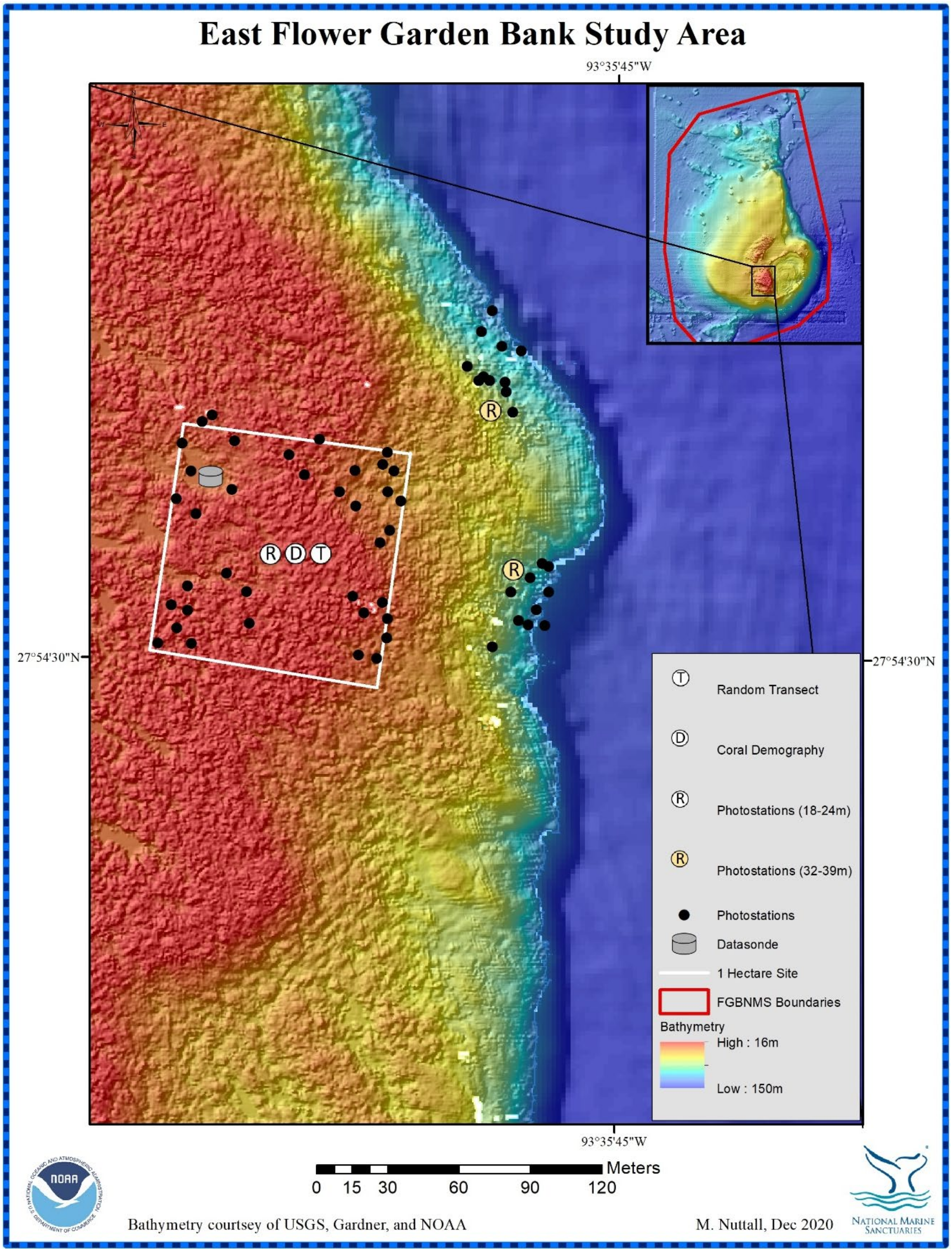


Figure 1.4. Bathymetric map of EFGB showing the location of the long-term monitoring one-hectare study site, within which repetitive photostations (18–24 m) and random transect and coral demographic surveys are conducted. Also included is the water quality datasonde and 32–39 m repetitive photostation locations.

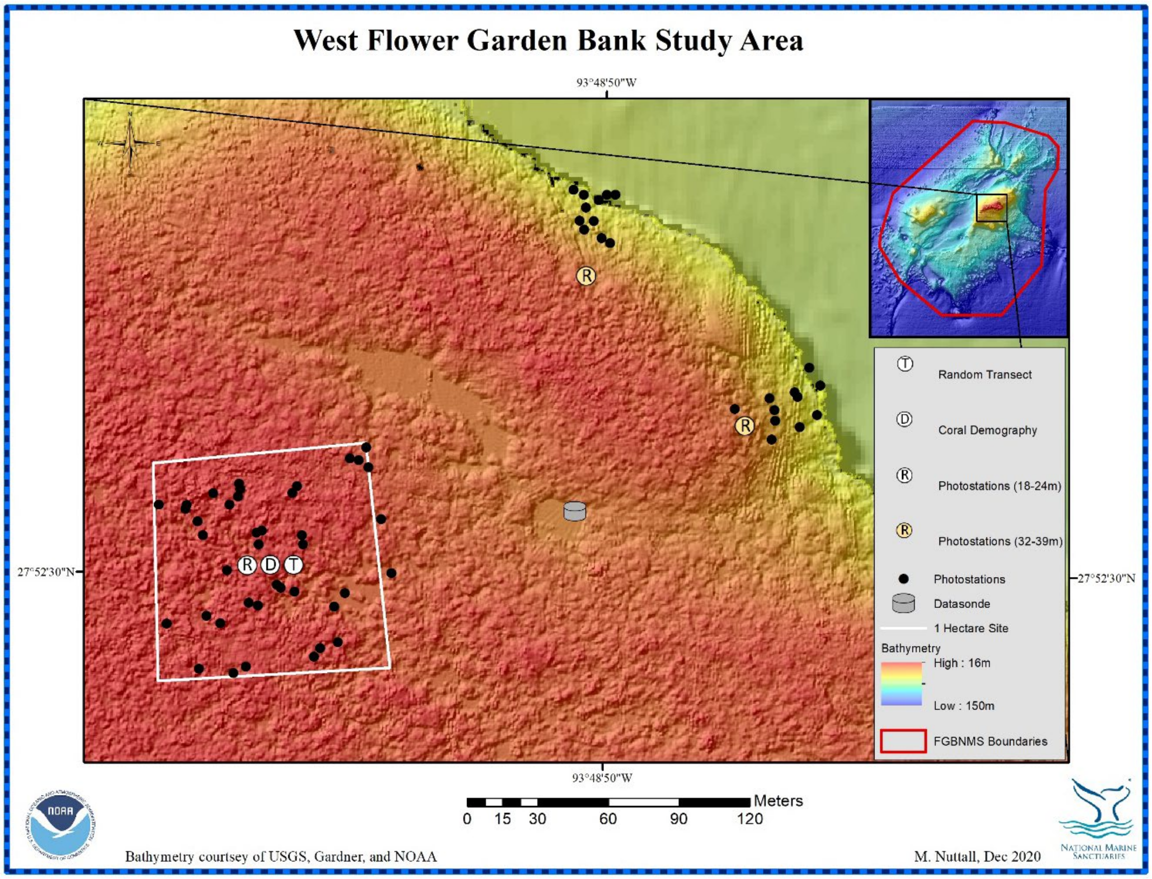


Figure 1.5. Bathymetric map of WFGB showing the location of the long-term monitoring one-hectare study site, within which repetitive photostations (18–24 m) and random transect and coral demographic surveys are conducted. Also included is the water quality datasonde and 32–39 m repetitive photostation locations.

Field Operations

Long-term monitoring data were collected at both EFGB and WFGB in 2019 and SCUBA operations were conducted off the NOAA R/V *Manta* (Table 1.2). Water samples were collected, water quality instruments were exchanged, and data were downloaded by FGBNMS staff during all four quarters in 2019 (Table 1.2). See each respective chapter for detailed field operation methodology.

Annual fieldwork at EFGB was conducted July 23–26, 2019 (Table 1.2), but strong surface and bottom currents (0.5 kt), poor water column visibility (less than 15 m, with patches less than 8 m), 1–2 m seas, and thunderstorms prevented the completion of all tasks. Annual at WFGB was conducted July 30–August 2, 2019 (Table 1.2), and all surveys were completed, along with the remaining surveys from EFGB. Water quality samples were collected and instruments were exchanged, but strong currents did not allow for the photography of WFGB repetitive photostations within the 32–39 m depth range or the exchange of the temperature loggers at 30 m and 40 m.

A one-day cruise in December 2019 was conducted to complete the WFGB repetitive photostation photography within the 32–39 m depth range and HOBO water quality instrument exchange at the 30 m and 40 m depth locations (Table 1.2).

Coral demographic surveys are conducted biennially and therefore were not collected in 2019.

Table 1.2. Monitoring and response cruises completed at EFGB and WFGB in 2019.

Date	Cruise and Tasks Completed
02/05/2019	Water quality cruise: Instrument exchange
02/28/2019	Water quality cruise: Water sample collection
05/16/2019	Water quality cruise: Instrument exchange and water sample collection
07/23/2019–07/26/2019	Long-term monitoring cruise: EFGB annual monitoring
07/30/2019–08/02/2019	Long-term monitoring and water quality cruise: EFGB and WFGB annual monitoring, water quality instrument exchange, and water sample collection
11/19/2019	Water quality cruise: Instrument exchange and water sample collection
12/04/2019	Water quality instrument exchange and WFGB repetitive photostation photography (32–39 m depth range)

Chapter 2:
Benthic Community

NOAA diver with camera and strobes mounted on an aluminum t-frame takes random transect photographs at EFGB.
Photo: G.P. Schmahl/NOAA

Introduction

Benthic cover, including components such as corals, sponges, crustose coralline algae (CCA), and macroalgae, was determined through analysis of a series of randomly located 8-m photo transects within EFGB and WFGB one-hectare study sites. These surveys were used to compare habitat and document the benthic reef community within EFGB and WFGB one-hectare study sites, as well as temporal changes within each one-hectare study site.

Permanent repetitive photostations were photographed to document changes in the composition of benthic assemblages at select locations ranging in depth from 18–39 m at EFGB and WFGB. The photographs were analyzed to measure percent benthic cover using random-dot analysis. All comparisons within this category are intended solely to assess differences among groups of repetitive photostations, as they were not randomly selected. While these stations can help identify directions and causes of change, they are not intended to estimate reef-wide populations or communities.

Methods

Random Transect Field Methods

In 2019, sixteen non-overlapping random transects were completed within each one-hectare study site in depths ranging from 17–27 m. Divers were given a randomly generated start location and heading for each survey. A Canon Power Shot® G11 digital camera in an Ikelite® housing and 28 mm equivalent wet mount lens adapter, mounted on a 0.65-m t-frame with bubble level and two Inon® Z240 strobes was used to capture images along the transects. The bubble level mounted to the t-frame center ensured images were taken in a vertical orientation. The mounted camera was placed at pre-marked intervals 80 cm apart on a spooled 15-m measuring tape, producing 17 non-overlapping images along the transect (Figure 2.1). Each still frame image captured a 0.8 x 0.6 m area (0.48 m^2). This produced a total photographed area of 8.16 m^2 per transect, and a minimum of 130.56 m^2 photographed area per one-hectare study site per year. For more detailed methods, reference Johnston et al. (2017a).

It should be noted that during the entirety of the monitoring program, a variety of underwater camera setups were used as technology advanced from 35-mm slides (1989 to 2001), digital videography using video still frame grabs (2002 to 2009), and digital still images (2010 to 2019) (Gittings et al. 1992; CSA 1996; Dokken et al. 1999, 2003; Precht et al. 2006; Zimmer et al. 2010; Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020). Prior to the use of Coral Point Count with Microsoft® Excel® extensions (CPCe), percent cover was calculated with mylar traces and a calibrated planimeter from 1989 to 1995 (Gittings et al. 1992; CSA 1996). From 1996 to 2003, random-dot layers were generated manually in photo software programs (Dokken et al. 1999, 2003).

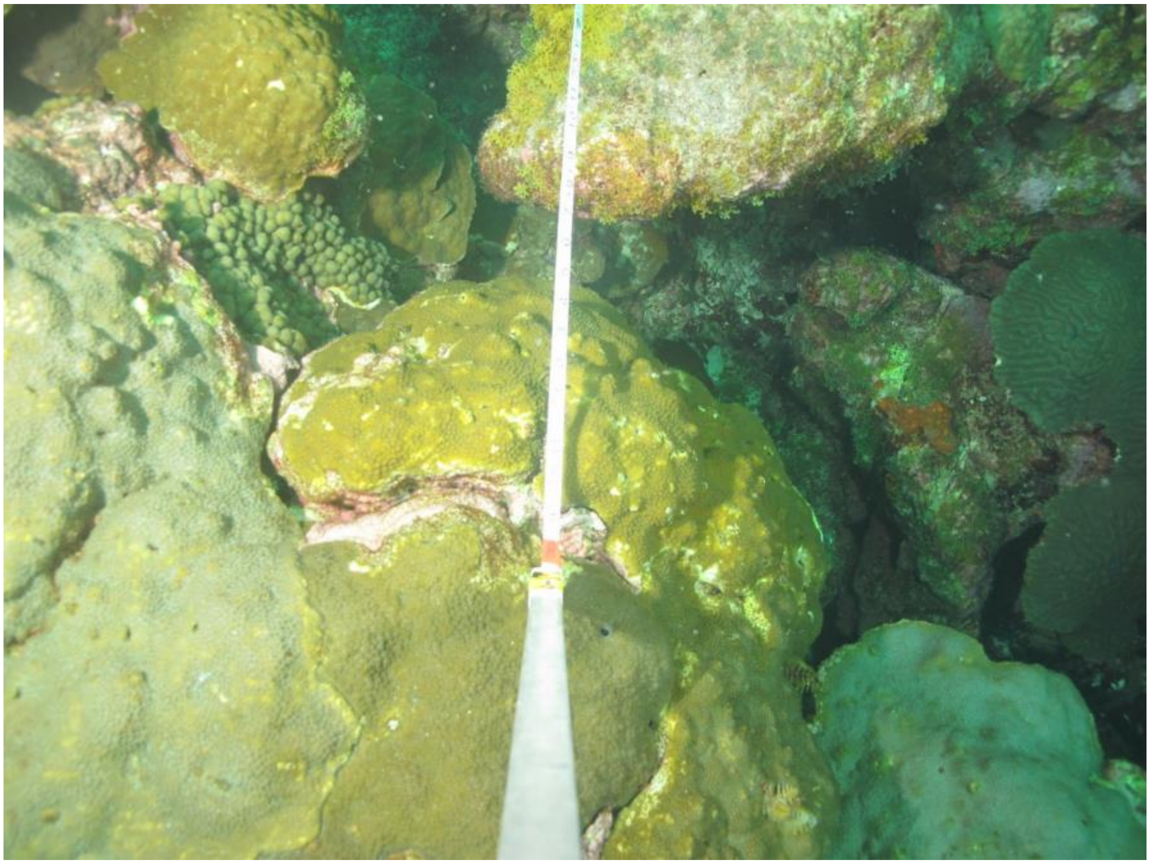


Figure 2.1. Photo taken at marked interval along random transect with camera mounted to aluminum t-frame at EFGB. Photo: Raven Blakeway/CPC

Random Transect Data Processing

Mean percent benthic cover from random transect images was analyzed using CPCe version 4.1 with a 500-point overlay randomly distributed among all images within a transect (30 spatially random points per image) (Aronson et al. 1994; Kohler and Gill 2006). Organisms positioned beneath each random point were identified to the lowest possible taxonomic level, and cover was categorized into seven groups: 1) coral, 2) sponges (including encrusting sponges), 3) CCA, 4) macroalgae (algae longer than approximately 3 mm and thick algal turfs covering underlying substrate), 5) colonizable substrate (including fine turf algae, rubble, and bare rock) (Aronson and Precht 2000; Aronson et al. 2005), 6) sand, and 7) an "other" category (biotic components such as sea urchins, ascidians, fish, serpulids, and unknown species). Additional features (photostation tags, tape measures, scientific equipment) and points with no data (shadows) were excluded from the analysis. Points on corals that could not be differentiated because of camera angle or camera distortion were labeled as "unidentified coral." *Orbicella* colonies that could not be identified to the species level were labeled as *Orbicella* spp. Point count analysis was applied on photos within a transect and mean percent cover for all groups was determined by averaging all transects per one-hectare study site. Results are presented as mean percent cover ± standard error (SE).

Incidences of coral bleaching, paling, concentrated and isolated fish biting, and mortality were also recorded as "notes" in CPCe, providing additional data for each random point. Any point that landed on a portion of coral that was white in color was characterized as "bleached." Any point that landed on coral that was pale relative to what is considered "normal" for the species was characterized as "paling" (AGRRA 2012). If the colony displayed some bleaching or paling, but the point landed on a healthy area of the organism, the point was "healthy" and no bleaching or paling was noted in CPCe. To classify fish biting, any point that landed where fish biting occurred on a coral head more than once was classified as concentrated fish biting, and any point where there was only one occurrence of fish biting was classified as isolated fish biting. Mortality included any point on recently dead coral (exposed bare skeleton) with little to no algae growth that could still be identified to the species level.

Consistency for photographic random transect methods was ensured by using multiple, scientific divers all trained on the same camera systems for correct camera operation. Camera settings and equipment were standardized so that consistent transect images were taken annually, and equipment checklists were provided in the field to ensure divers had all equipment and were confident with tasks assigned. Random transect photographs were reviewed promptly after images were taken, in the field, to ensure the quality was sufficient for analysis. After all benthic components were identified in CPCe files, quality assurance/quality control (QA/QC) consisted of an independent review by a separate, trained researcher, different from the CPCe analyzer, to ensure all identified points from the random transect photographs were accurate. Any mistakes were corrected before percent cover analysis was completed.

Random Transect Statistical Analysis

Benthic community interactions in EFGB and WFGB random transects were evaluated with distance-based analyses using Primer® version 7.0 (Anderson et al. 2008; Clarke et al. 2014). Euclidean distance resemblance matrices were calculated using untransformed percent cover data from random transect benthic groups. Data were left untransformed so that the significance of non-dominant groups was not overinflated. Permutational multivariate analysis of variance (PERMANOVA) was based on Euclidean distance resemblance matrices and used to test for benthic community differences and estimate components of variation between one-hectare study sites (Anderson et al. 2008). If significant differences were found, groups or species contributing to observed differences were examined using similarity percentages (SIMPER) to assess the percent contribution of each variable to dissimilarity between groups (Clarke et al. 2014).

Coral species composition was compared between one-hectare study sites using PERMANOVA on square-root transformed coral species percent cover data with Euclidean distance similarity matrices. Diversity indices for coral species, including Margalef's species richness (d), Pielou's evenness (J'), and Shannon diversity (H'), were calculated to make comparisons between one-hectare study sites in 2019. Similarity matrices from diversity indices, based on square-root-transformed data and Euclidean distance, were tested for significant dissimilarities using analysis of similarity (ANOSIM) (Clarke et al. 2014).

To assess trends in historical random transect mean percent cover data (1992 to 2019), benthic groups by year and one-hectare study site were visualized using principal coordinates ordination (PCO), based on Euclidean distance similarity matrices, with percent variability explained on each canonical axis. A time series trajectory with correlation vectors (correlation >0.2) was overlaid on PCO plots to represent the direction of the variable gradients for the plot (Anderson et al. 2008; Clarke et al. 2014). Cluster analyses for year groups were performed on Euclidean distance similarity matrices with SIMPROF tests to identify significant (α=0.05) clusters within the data (Clarke et al. 2008). One-hectare study site communities were compared using PERMANOVA. SIMPER identified groups contributing to observed dissimilarities (Clarke et al. 2014).

Mean percent benthic cover from random transect surveys was analyzed from 1989 to 2019. Monotonic trends in mean percent cover data were assessed using the Mann-Kendall trend test in R version 2.13.2 (Hipel and McLeod 1994; Helsel and Hirsch 2002). Tests for significant correlation among benthic cover groups were completed in R version 2.13.2 with Pearson's correlation (Helsel and Hirsch 2002). It should be noted that the range of data collected has varied slightly over the years. From 1989 to 1991, only mean percent coral cover data were collected; other major benthic groups were added in 1992. No data were collected in 1993.

Repetitive Photostation Field Methods

Repetitive photostations, marked by permanent pins with numbered tags on the reef, were located by SCUBA divers using detailed underwater maps displaying compass headings and distances to each station. Thirty-seven photostations were located within the EFGB one-hectare study site and 41 photostations within the WFGB one-hectare study site. Twenty-three photostations at EFGB were located outside the one-hectare study site (east of buoy#2) at depths ranging from 32–39 m (Figure 1.3). Twenty-four photostations at WFGB were located outside the one-hectare study site (near buoy #2) at depths ranging from 32–39 m (Figure 1.4).

After photostations were located, divers photographed each station using a Nikon® D7000® SLR camera with 16-mm lens in a Sea&Sea® housing with small dome port and two Inon® Z240 strobes (1.2 m apart). The camera was mounted in the center of a T-shaped camera frame, at a distance of 2 m from the substrate (Figure 2.2). To ensure that the stations were photographed in the same manner each year, the frame was oriented in a north-facing direction and kept vertical using an attached bullseye bubble level and compass (for more detailed methods, reference Johnston et al. 2017a). This set-up produced images covering 5 m^2. In 2019, all repetitive photostations were photographed.

It should be noted that during the entirety of the monitoring program, underwater camera setups used to capture benthic cover in the repetitive photostations changed as technology advanced from 35-mm slides and film (1989 to 2007) to digital still images (2008 to 2019) (Gittings et al. 1992; CSA 1996; Dokken et al. 1999, 2003; Precht et al. 2006; Zimmer et al. 2010; Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020). From 1989 to 2009, photographs for each repetitive photostation encompassed an 8 m^2 area, but changed to a 5 m^2 area in 2009, a 9 m^2 area in 2010, and back to a 5 m^2 area from 2011 onward due to requirements for consistent image quality, changes in camera equipment, and updated technology. The total number of photostations changed over time as well, as new stations were established or old stations were lost. Approximately 40 photostations have been maintained within each one-hectare study site since 1989. Within the 32–39 m depth range, nine of the 23 EFGB photostations were established in 2003 and 12 of the 24 WFGB photostations were established in 2012. Two additional EFGB stations (30 m and 31 m) were added in 2013. The remaining 12 photostations in this depth range at each bank were added in 2017.

Figure 2.2. NOAA diver photographs a repetitive photostation with camera and strobes mounted to an aluminum t-frame. Photo: G.P. Schmahl/NOAA

Repetitive Photostation Data Processing

Mean percent benthic cover from repetitive photostation images was analyzed using CPCe version 4.1 (Aronson et al. 1994; Kohler and Gill 2006). A total of 100 random dots were overlaid on each photograph and benthic species lying under these points were identified and verified by QA/QC (see Benthic Community Methods: Random Transect Data Processing for detailed methods). Point count analysis was conducted for all photos and mean percent cover for functional groups was determined by averaging across all photostations per one-hectare study site. Results are presented as mean percent cover ± SE. All repetitive photostation comparisons were only made with other repetitive photostations. Because photostations were not randomly selected, they are not intended to estimate reef-wide populations or benthic communities.

Repetitive Photostation Statistical Analysis

Benthic community interactions were evaluated using distance-based analyses with Primer® version 7.0 (Anderson et al. 2008; Clarke et al. 2014) and PERMANOVA (see Benthic Community Methods: Random Transect Statistical Analysis). Percent coral cover was compared among repetitive photostations using PERMANOVA with photostation depth as a covariable on square-root-transformed coral species percent cover data with Euclidean distance similarity matrices. Mean percent coral cover from repetitive photostations was compared between 1989 and 2019 (n=24 at EFGB and n=27 at WFGB) using a paired t-test in R version 2.13.2.

Results

Random Transect Mean Percent Cover

Coral, followed by macroalgae, had the highest mean percent benthic cover at EFGB and WFGB in 2019 (Figure 2.3 and Table 2.1).

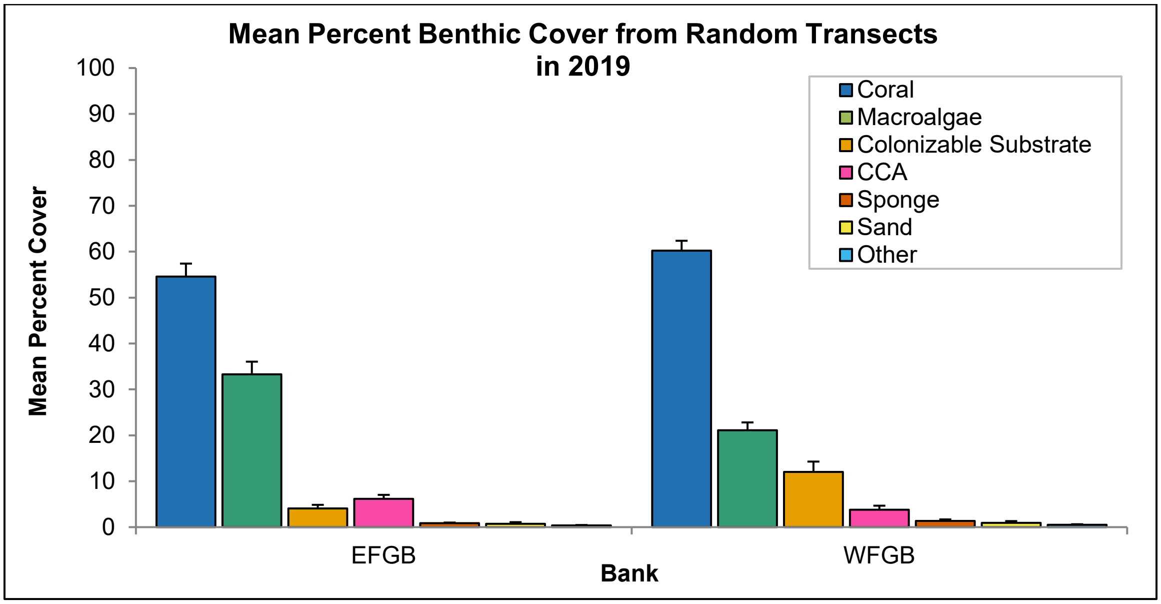


Figure 2.3. Mean percent benthic cover + SE from random transect surveys within EFGB and WFGB one-hectare study sites in 2019.

Table 2.1. Range of mean percent cover categories from random transect surveys at EFGB, WFGB, and both one-hectare study sites combined, in 2019.

Benthic Cover Type	EFGB	WFGB	Combined
Coral	35.64–73.55%	44.80–75.20%	35.64–75.20%
Macroalgae	16.60–53.19%	5.53–35.10%	5.53–35.10%
CCA	1.70–14.02%	0.89–13.69%	0.89–14.02%
Colonizable substrate	0.00–14.48%	0.00–36.76%	0.00–36.76%
Sponge	0.00–2.07%	0.20–5.31%	0.00–5.31%
Sand	0.00–5.84%	0.00–5.53%	0.00–5.84%
Other	0.00–0.88%	0.00–1.43%	0.00–1.43%

PERMANOVA analysis revealed significant differences in benthic community composition between EFGB and WFGB one-hectare study sites in 2019 (Table 2.2). SIMPER analysis identified the observed dissimilarity between one-hectare study sites was due to significantly higher macroalgae cover at EFGB and significantly higher coral cover at WFGB (contributing 44% and 32%, respectively).

Table 2.2. PERMANOVA results comparing 2019 mean percent benthic cover in EFGB and WFGB random transect surveys. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Benthic cover	1997	1	8.48	**0.001**
Res	7069	30		
Total	9067	31		

Twelve species of coral were observed within the EFGB random transect surveys and 13 species of coral were observed in the WFGB random transect surveys (13 coral species were observed at both one-hectare study sites combined) (Figure 2.4). *Orbicella franksi* was the most abundant coral species observed within EFGB (27.52% ± 3.00) and WFGB (35.40% ± 2.99) surveys. *Pseudodiploria strigosa* was the second most abundant species (EFGB 7.65% ± 1.12 and WFGB 7.78% ± 1.08) (Figure 2.4).

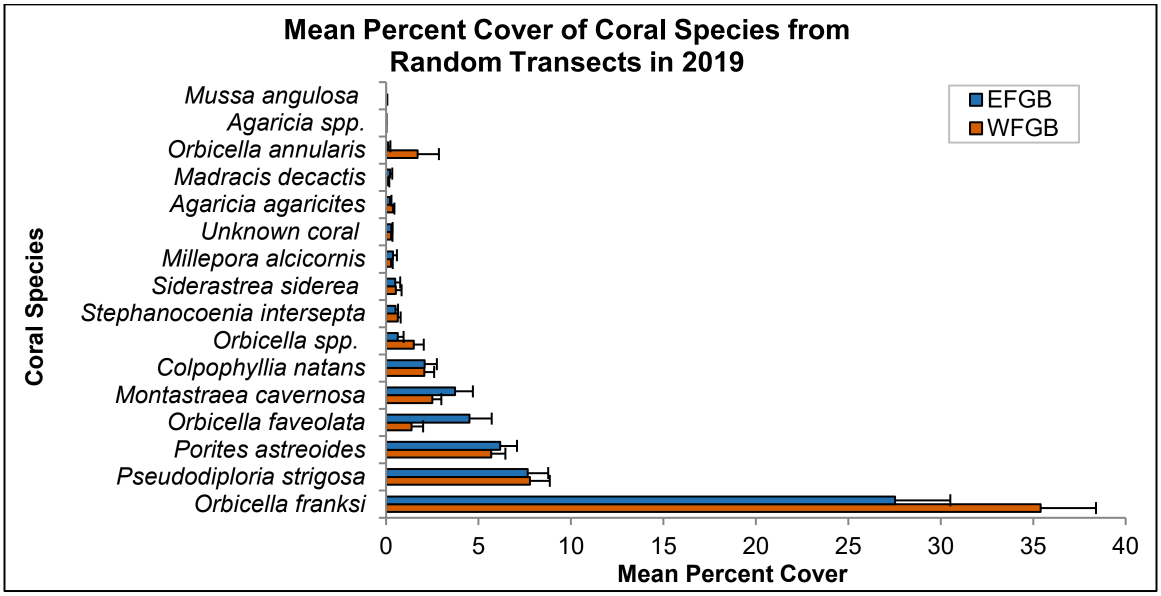


Figure 2.4. Mean percent cover + SE of observed coral species from random transect surveys within EFGB and WFGB one-hectare study sites in 2019.

The *Orbicella* spp. Complex, including *O. franksi, Orbicella faveolata*, and *Orbicella annularis* (listed as threatened species under the Endangered Species Act in 2014), made up 60.09% of the observed coral cover within EFGB random transects and 66.36% within WFGB random transect surveys (63.23% for both one-hectare study sites surveys combined). PERMANOVA analysis revealed significant differences in coral species composition between one-hectare study sites (Table 2.3). SIMPER analysis indicated that this difference was due to significantly higher *O. faveolata* cover in the EFGB one-hectare study site and significantly higher *O. franksi* cover in the WFGB one-hectare study site (contributing 21% and 18%, respectively).

Table 2.3. PERMANOVA results comparing 2019 coral species mean percent cover in EFGB and WFGB random transect surveys. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Coral species	17	1	2.14	**0.034**
Res	237	30		
Total	254	31		

Coral species diversity measures were averaged for each one-hectare study site in 2019 (Table 2.4). ANOSIM analysis revealed no significant differences between one-hectare study site coral communities.

Table 2.4. Mean coral species diversity measures ± SE within EFGB and WFGB one-hectare study sites in 2019.

Random Transect Coral Diversity Measures	EFGB	WFGB
Margalef's species richness (d)	1.46 ± 0.04	1.44 ± 0.04
Pielou's evenness (J')	0.79 ± 0.01	0.75 ± 0.02
Shannon diversity (H'(loge))	1.19 ± 0.03	1.13 ± 0.04

Less than 0.5% of the coral cover within the random transect surveys was bleached or pale in late July and early August of 2019. It is important to note that surveys occurred at the time of year when coral bleaching is not typically noted at FGBNMS and water temperatures and exposure times were lower than threshold levels known to trigger bleaching (Ogden and Wicklund 1988; Glynn and D'Croz 1990; Hagman and Gittings 1992; Johnston et al. 2019). In addition, less than 0.5% of coral cover was affected by fish biting and signs of mortality. Fish biting that resulted in the removal of coral polyps from affected areas is most likely the result of damselfish gardening or grazing by stoplight parrotfish (*Sparisoma viride*) (Bruckner and Bruckner 1998; Bruckner et al. 2000).

Random Transect Long-Term Trends

Mean percent coral cover from 1989 to 2019 ranged from 40–64% in the EFGB one-hectare study site and 37–66% in the WFGB one-hectare study site (depths ranging 17–27 m), and significantly increased in the WFGB one-hectare study site over the time period ($\tau=0.60$, $p<0.001$) (Figure 2.5) Mean percent coral cover in the EFGB one-hectare study site remained stable. Sponge, CCA, macroalgae, colonizable substrate, and sand data were not available until 1992. Prior to 1999, macroalgae cover was consistently below 5% within the one-hectare study sites; however, in 1999, macroalgae cover increased to approximately 20% and has averaged 30% for the past ten years. In general, macroalgae and colonizable substrate varied inversely and were significantly correlated at EFGB ($\tau=-8.76$, $p<0.001$) and WFGB ($\tau=-8.84$, $p<0.001$). While macroalgae colonized available substrate, it did not outcompete or displace coral. From 1992 to 2019, macroalgae significantly increased in EFGB ($\tau=0.65$, $p<0.001$) and WFGB ($\tau=0.52$, $p<0.001$) one-hectare study sites. Colonizable substrate significantly decreased in EFGB ($\tau=-0.58$, $p<0.001$) and WFGB ($\tau=-0.48$, $p<0.001$) one-hectare study sites (Figure 2.5).

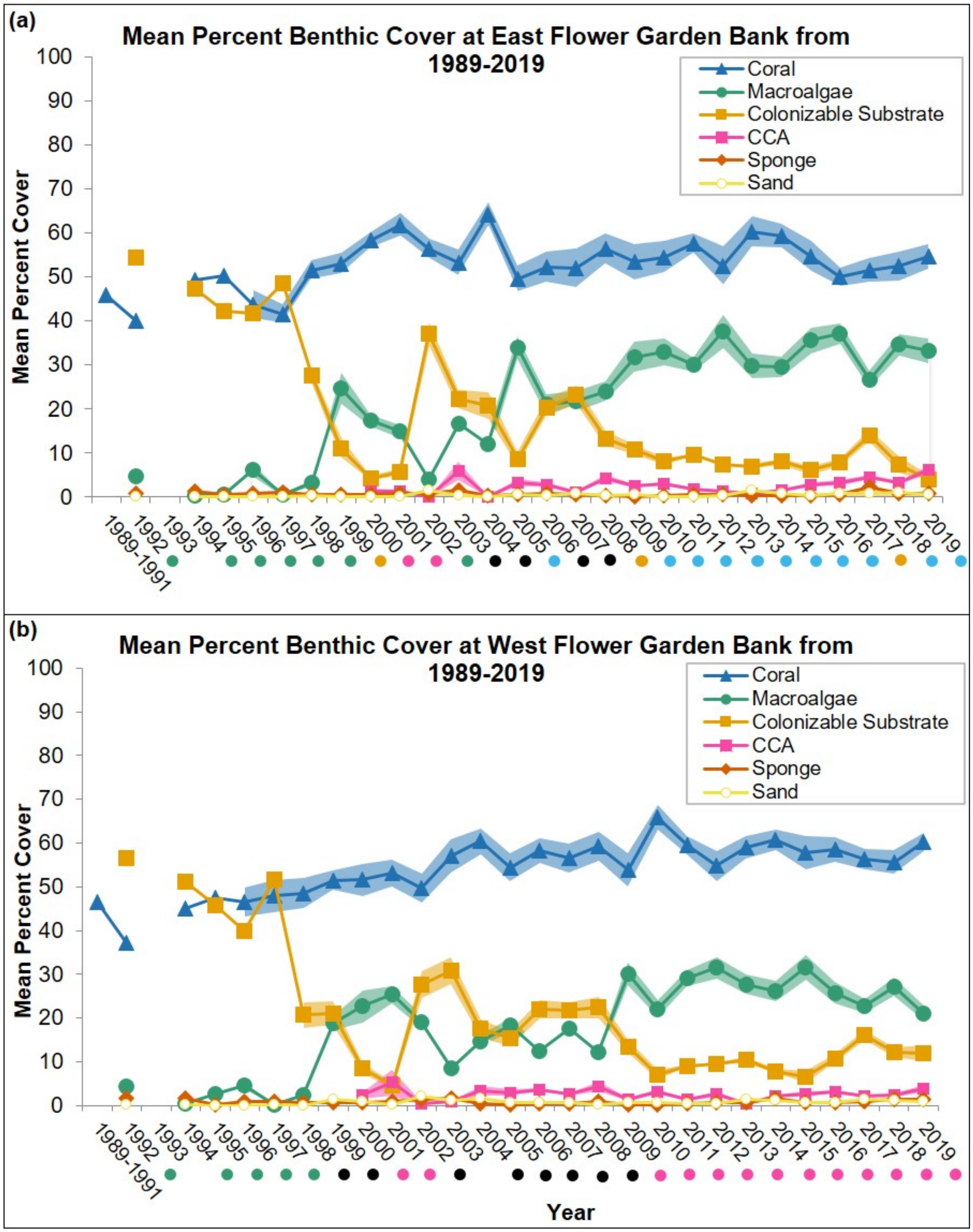


Figure 2.5. Mean percent benthic cover ± SE bands from random transect surveys within (a) EFGB and (b) WFGB one-hectare study sites from 1989 to 2019. The colored dots below the years on the x-axis represent significant year clusters corresponding to SIMPROF groups in Figures 2.6 and 2.7.

Only coral cover data were reported from 1989–1991 and no mean percent cover data were reported in 1993. Sources: 1989 to 1991, Gittings et al. (1992); 1992 to 1995, Continental Shelf Associates, Inc. (CSA 1996); 1996 to 2001, Dokken et al. (2003); 2002 to 2008, PBS&J (Precht et al. 2006; Zimmer et al. 2010); 2009 to 2018, FGBNMS (Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020)

For available complete yearly mean benthic percent cover data (1992 to 2019), SIMPROF analysis detected five significant year clusters in the EFGB one-hectare study site (A: 1992 to 1998 and 2002; B: 2003 to 2004 and 2006 to 2007; C: 2000 to 2001; D: 1999, 2008, and 2017; and E: 2005, 2009 to 2016 and 2018 to 2019) (Figures 2.5 and 2.6). Colonizable substrate and macroalgae mean percent cover contributed to over 85% of the dissimilarity (57.50% and 28.01%, respectively) between clusters A and B, corresponding to an increase in macroalgae and decrease in colonizable substrate cover after 1998 (Figure 2.5a). Colonizable substrate and macroalgae mean percent cover were also the primary contributors to the dissimilarity between clusters B and D (41.92% and 35.69%, respectively), A and D (61.91% and 32.64%, respectively), and A and E (55.36% and 40.47%, respectively). Colonizable substrate was the primary contributor to the dissimilarity between clusters B and C (78.30%), as well as clusters A and C (79.90%).

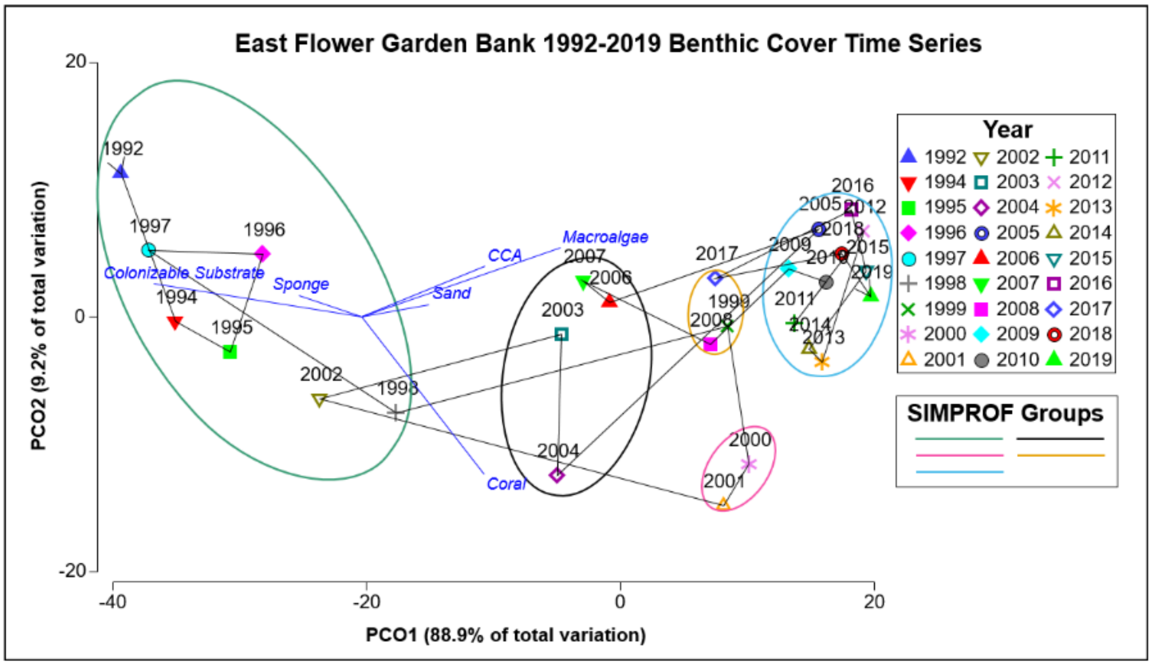


Figure 2.6. PCO plot for random transect benthic cover analysis within the EFGB one-hectare study site from 1992 to 2019. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

Yearly mean benthic percent cover from 1992 to 2019 at the WFGB one-hectare study site displayed a similar pattern to EFGB, resulting in three significant year clusters (A: 1992 to 1997; B: 1998 to 1999, 2002 to 2004, and 2006 to 2008; and C: 2000 to 2001, 2005, and 2009 to 2019) (Figure 2.7). Colonizable substrate mean percent cover contributed to over 70% of the dissimilarity between clusters A and B, corresponding to higher colonizable substrate cover (Figure 2.5b). Macroalgae and colonizable substrate mean percent cover also contributed to the dissimilarity between clusters B and C (46.90% and 43.51%, respectively), corresponding to an increase in macroalgae and decrease in colonizable substrate cover after 1998 (Figure 2.5b). Differences between clusters A and C were attributed to colonizable substrate and macroalgae mean percent cover (67.10% and 24.55%, respectively).

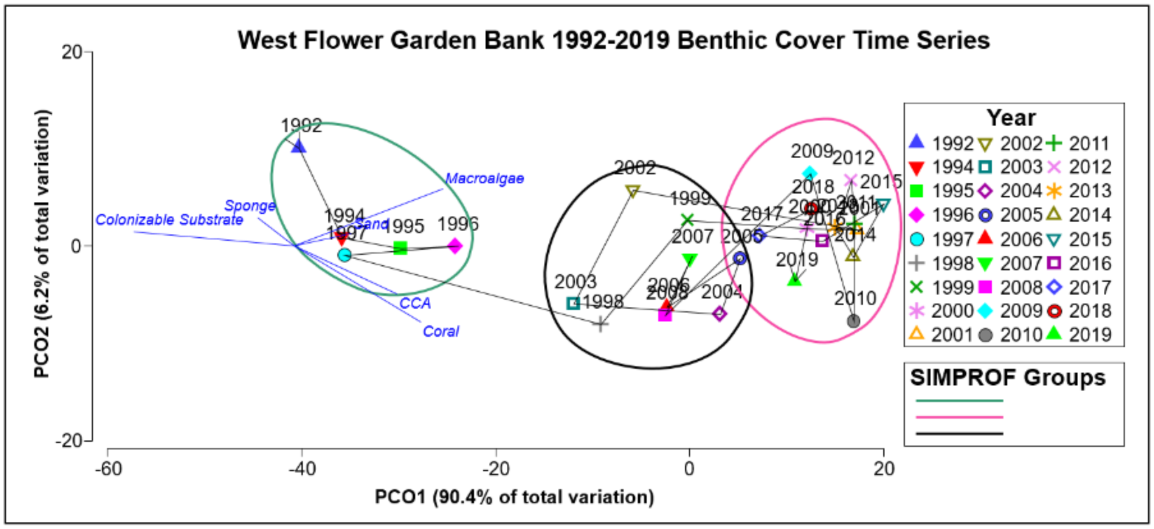


Figure 2.7. PCO plot for random transect benthic cover analysis within the WFGB one-hectare study site from 1992 to 2019. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

PERMANOVA results revealed no significant differences between one-hectare study sites, suggesting that EFGB and WFGB one-hectare study sites were similar to each other from 1992 to 2019 in overall benthic community composition, experiencing similar shifts though time.

Repetitive Photostation Mean Percent Cover

Coral and macroalgae were the dominant benthic cover categories in EFGB and WFGB repetitive photostations in 2019 (Figure 2.8 and Table 2.5).

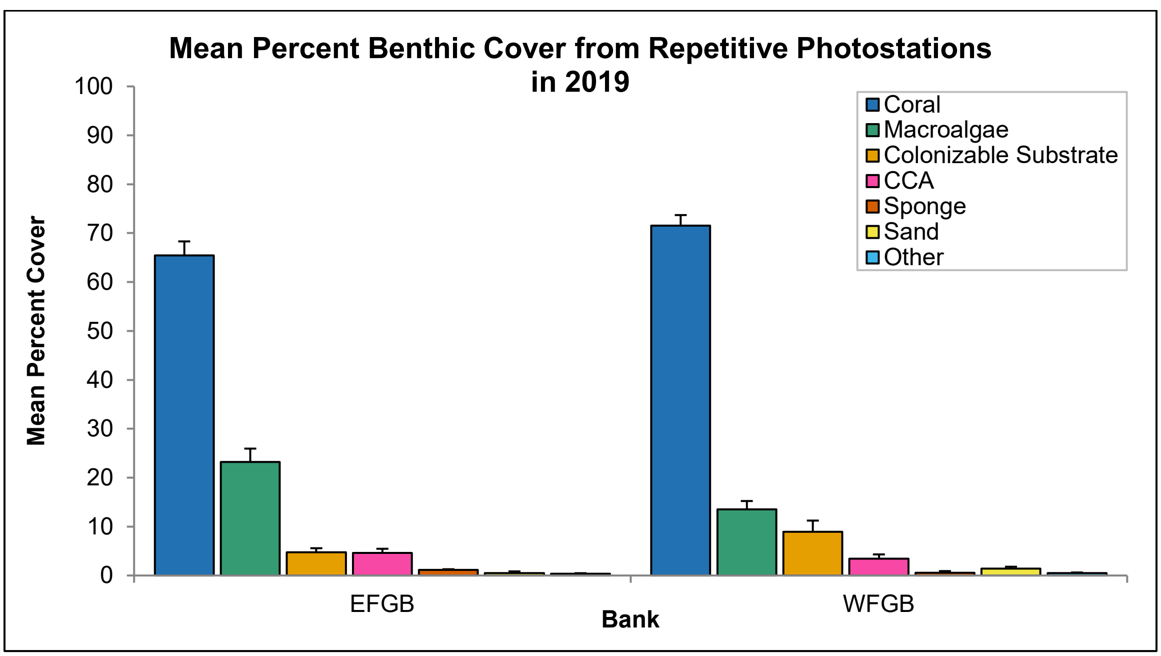


Figure 2.8. Mean percent benthic cover + SE within EFGB and WFGB repetitive photostations in 2019.

Table 2.5. Range of mean percent cover categories from EFGB and WFGB repetitive photostations, and all photostations combined, in 2019.

Range of Mean Percent Cover	EFGB	WFGB	EFGB and WFGB combined
Coral	24.21–93.88%	41.67–95.92%	24.21–95.92%
Macroalgae	2.04–48.42%	0.00–37.50%	0.00–48.42%
CCA	0.00–25.56%	0.00–19.79%	0.00–25.56%
Colonizable substrate	0.00–22.22%	0.00–30.21%	0.00–30.21%
Sponge	0.00–21.05%	0.00–6.12%	0.00–21.05%
Sand	0.00–7.37%	0.00–36.00%	0.00–36.00%
Other	0.00–3.16%	0.00–3.13%	0.00–3.13%

PERMANOVA analysis comparing benthic groups revealed significant differences, suggesting that the EFGB and WFGB repetitive photostations were dissimilar in benthic community composition in 2019 (Table 2.6). SIMPER analysis identified the observed dissimilarity among photostations was due to significantly higher macroalgae cover in the EFGB photostations and significantly higher coral cover in the WFGB photostations (contributing 30% and 37%, respectively).

Table 2.6. PERMANOVA results comparing 2019 mean percent benthic cover in EFGB and WFGB repetitive photostations. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
EFGB and WFGB photostations	4699	1	11.98	**0.001**
Res	48244	123		
Total	52943	124		

The repetitive photostations ranged in depth from 17.98–39.01 m at EFGB (averaging 25 m depth) and 19.81–38.10 m at WFGB (averaging 26 m depth). Mean percent benthic cover categories ranged widely among the photostations (an approximate 70% difference among photostations at EFGB and 54% difference at WFGB) (Table 2.5). Less than 1% of the coral cover analyzed was observed to be pale or bleached in the EFGB and WFGB repetitive photostations. In addition, less than 1% of old mortality was observed and signs of recent mortality were less than 2% in the repetitive photostations.

Fifteen coral species were observed in EFGB repetitive photostations (Figure 2.9) and fourteen were observed in WFGB repetitive photostations (Figure 2.10). *Orbicella franksi* was the predominant coral species observed in EFGB repetitive photostations (35.68% ± 2.49), followed by *P. strigosa* (7.98% ± 1.31) and *M. cavernosa* (5.69% ± 1.17) (Figure 2.9). *Orbicella franksi* was the predominant coral species observed in WFGB repetitive photostations (38.63% ± 2.50), followed by *M. cavernosa* (7.37% ± 1.38) and *P. strigosa* (6.87% ± 1.12) (Figure 2.10).

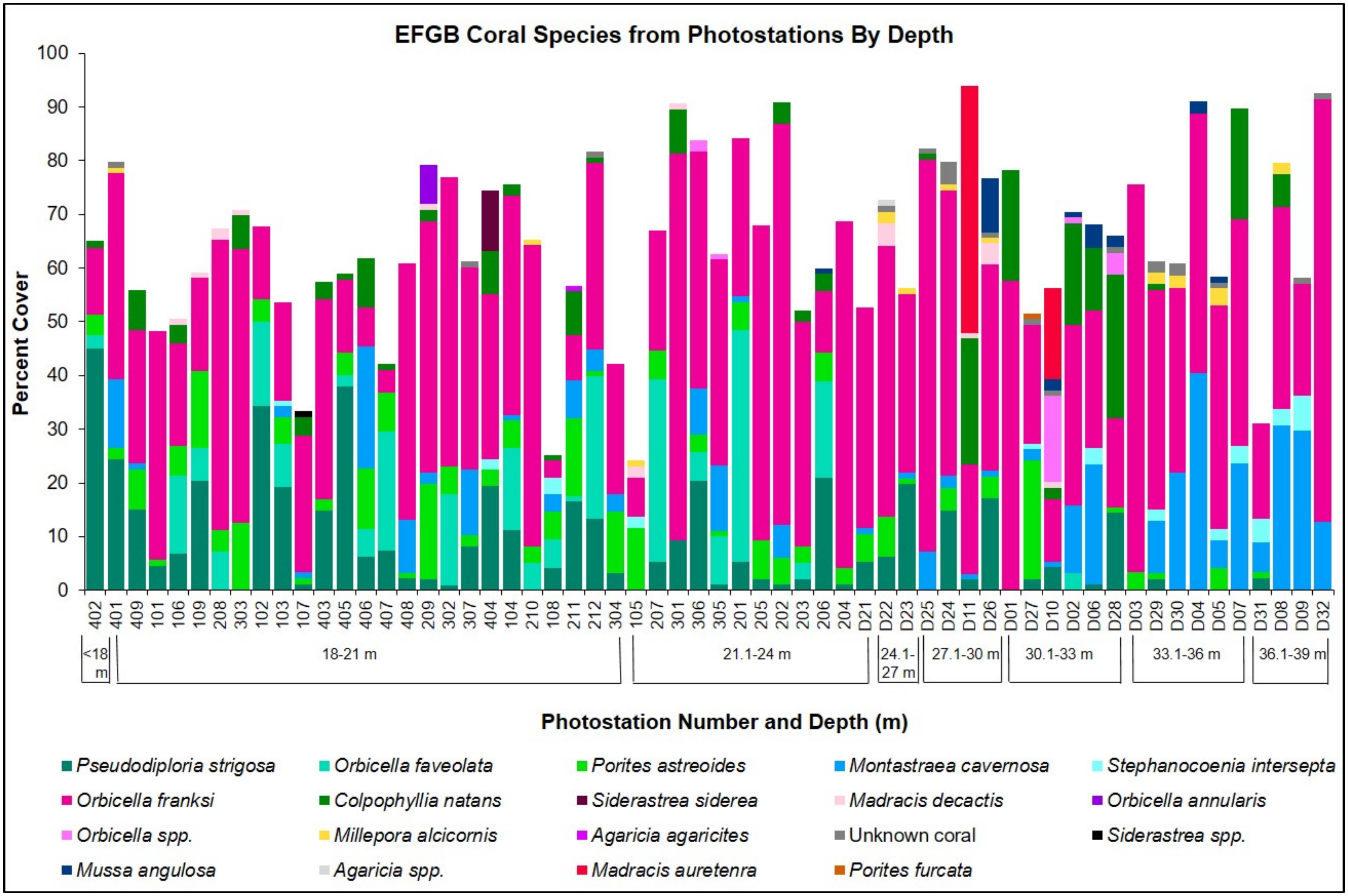


Figure 2.9. Mean percent cover of coral species within individual EFGB repetitive photostations (n=60) in 2019. Photostations are grouped in depth bins (m) from shallowest to deepest.

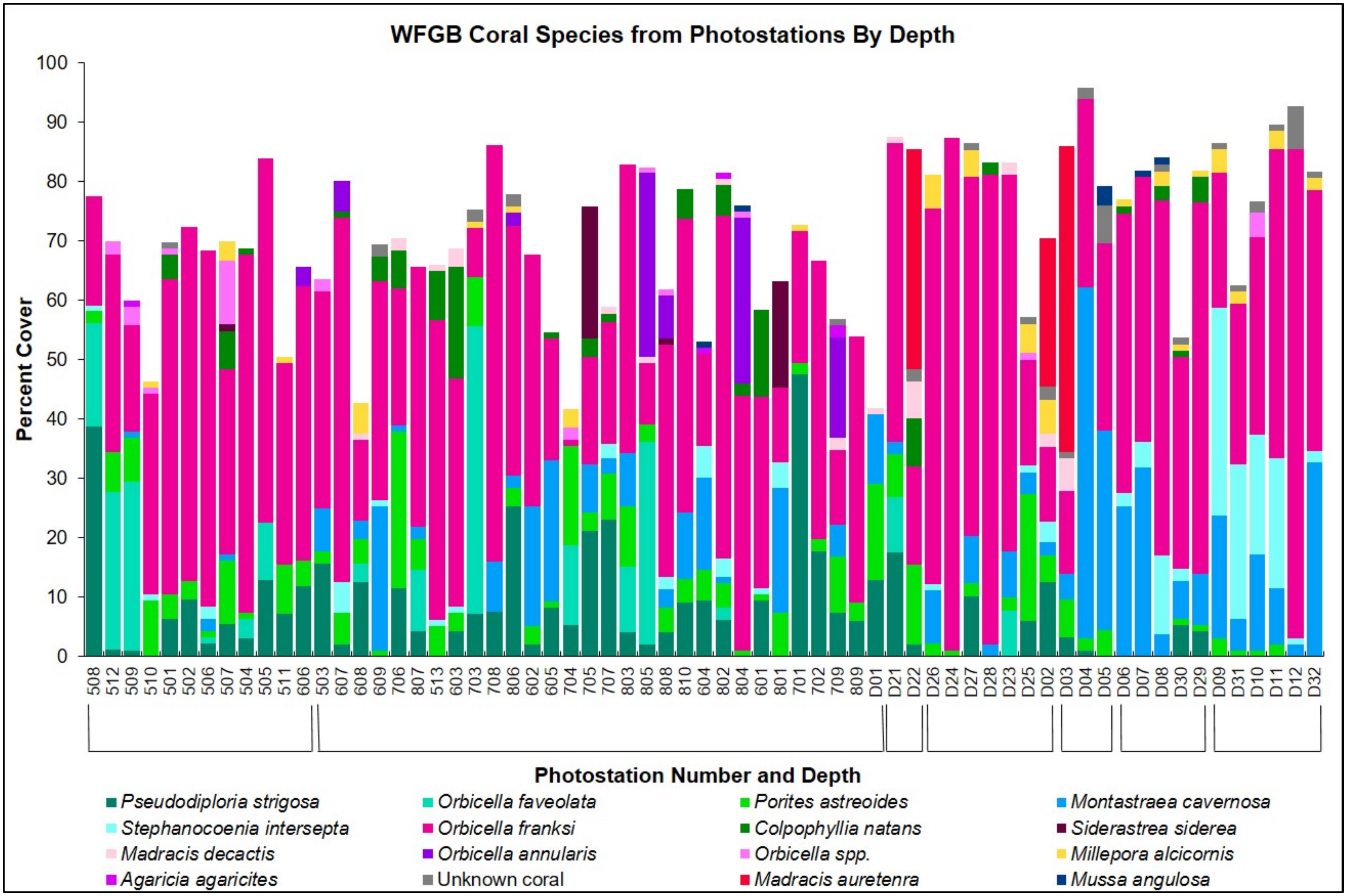


Figure 2.10. Mean percent cover of coral species within individual WFGB repetitive photostations (n=65) in 2019. Photostations are grouped in depth bins (m) from shallowest to deepest.

A significant correlation between percent coral cover and depth was found in the repetitive photostations at EFGB and WFGB in 2019, suggesting that coral species percent cover within the photostations changed with depth (Table 2.6).

Table 2.6. PERMANOVA results comparing 2019 coral species mean percent cover by depth in EFGB and WFGB repetitive photostations. Bold text denotes significant value.

EFGB	**Sum of Squares**	**df**	**Pseudo-F**	**P (perm)**
Depth	190	1	11.20	**0.001**
Photostation	553	33	0.99	0.527
Res	424	25		
Total	1168	59		
WFGB	**Sum of Squares**	**df**	**Pseudo-F**	**P (perm)**
Depth	204	1	10.54	**0.001**
Photostation	616	30	1.06	0.351
Res	639	33		
Total	1460	64		

Overall, percent coral cover in repetitive photostations increased with depth; however, individual species showed different trends. In EFGB repetitive photostations, *Colpophyllia natans* percent cover increased with depth, and in some photostations, almost doubled in percent cover (Figure 2.11a). *Colpophyllia natans* percent cover showed the opposite trend in WFGB photostations, where percent cover decreased with depth (Figure 2.12 a). *Montastraea cavernosa* percent cover increased with depth in both EFGB and WFGB repetitive photostations (Figure 2.11b and 2.12b). *Orbicella faveolata* and *P. strigosa* percent cover decreased as depth increased in EFGB and WFGB repetitive photostations (Figure 2.11 c and d; Figure 2.12 c and d). In general, *O. franksi* and *Porites astreoides* percent cover remained consistently high among all photostation depths, and *Stephanocoenia intersepta* increased with depth in EFGB and WFGB repetitive photostations (Figure 2.9 and 2.10).

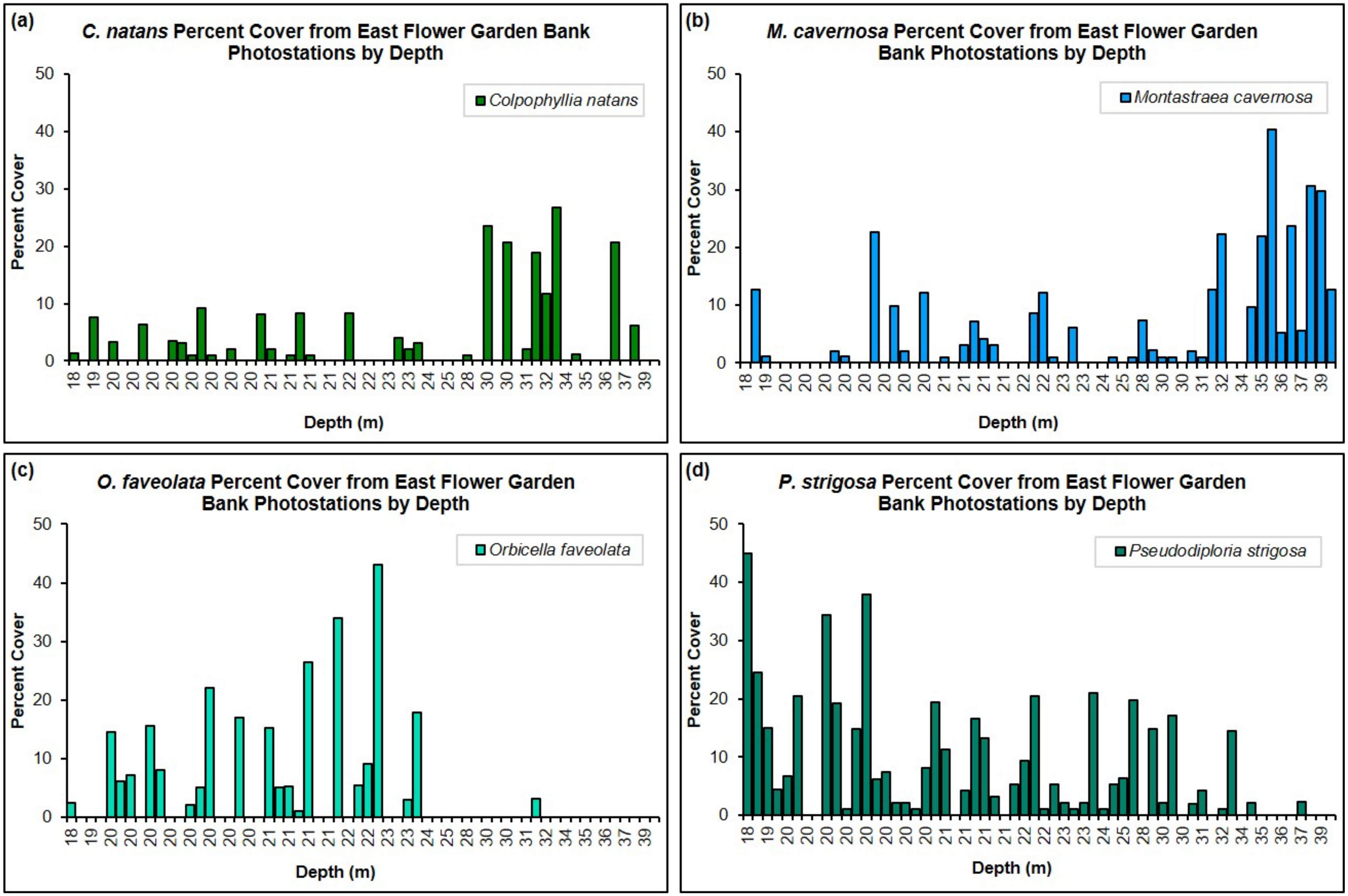


Figure 2.11. Mean percent cover of (a) *C. natans*, (b) *M. cavernosa*, (c) *O. faveolata*, and (d) *P. strigosa* within individual EFGB repetitive photostations (n=60) in 2019. Photostations are arranged by depth (m) from shallowest to deepest.

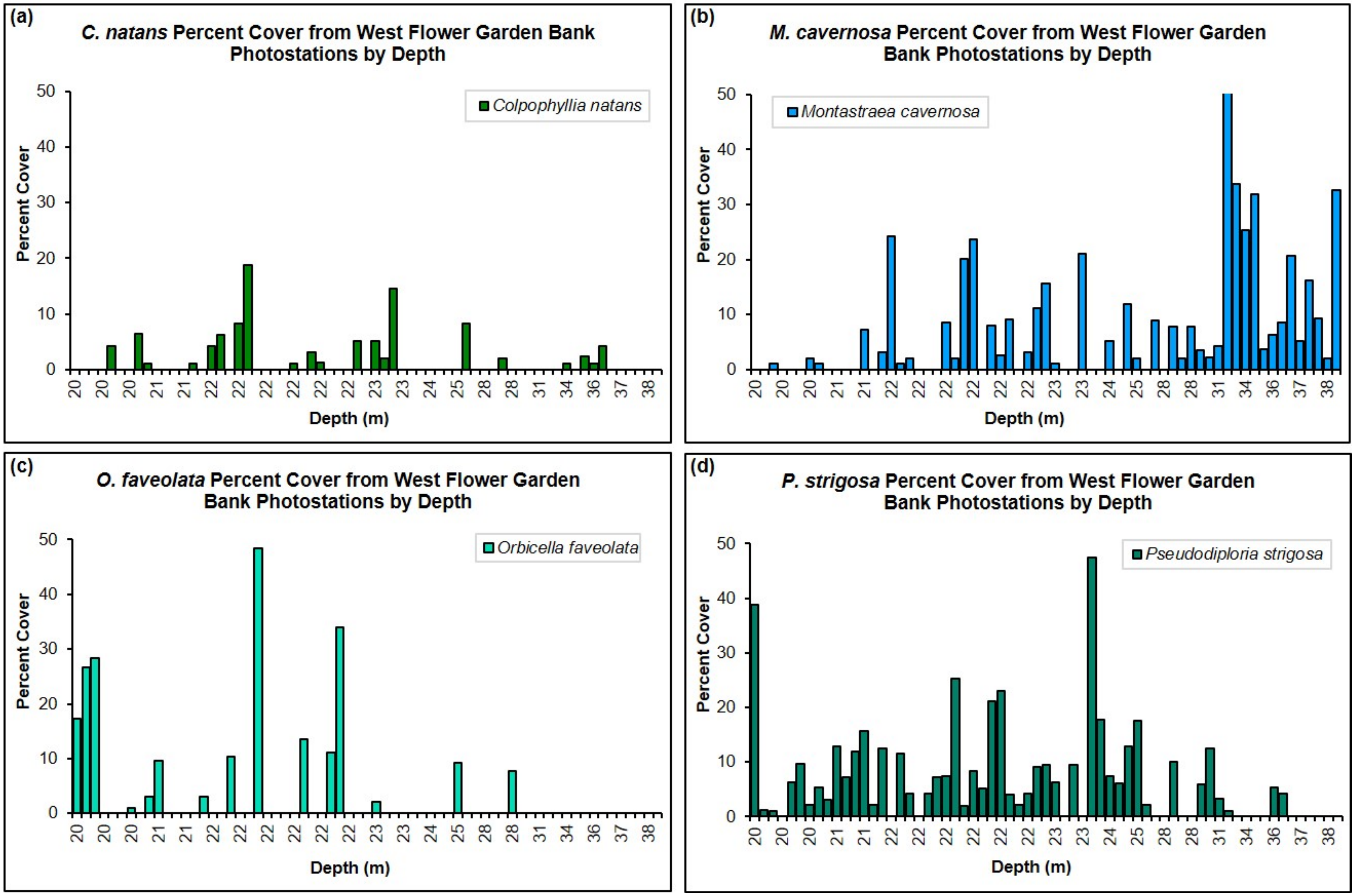


Figure 2.12. Mean percent cover of (a) *C. natans*, (b) *M. cavernosa*, (c) *O. faveolata*, and (d) *P. strigosa* within individual WFGB repetitive photostations (n=65) in 2019. Photostations are arranged by depth (m) from shallowest to deepest.

Repetitive Photostation Long-Term Trends

Twenty-four EFGB photostations and 27 WFGB photostations (ranging in depth from 20–24 m) have been in place since the beginning of the monitoring program, spanning from 1989 to 2019. Mean percent coral cover changed from 58.72% % ± 3.80 in 1989 to 64.81%% ± 3.50 in 2019 among the 24 EFGB photostations and 50.30% % ± 3.06 in 1989 to 67.68%% ± 2.62 in 2019 among the 27 WFGB photostations (Figure 2.13). Coral cover significantly increased from 1989 to 2019 in EFGB photostations (t-test, df=22, t=2.19, p=0.039) and WFGB photostations (t-test, df=24, t=6.94, p>0.001).

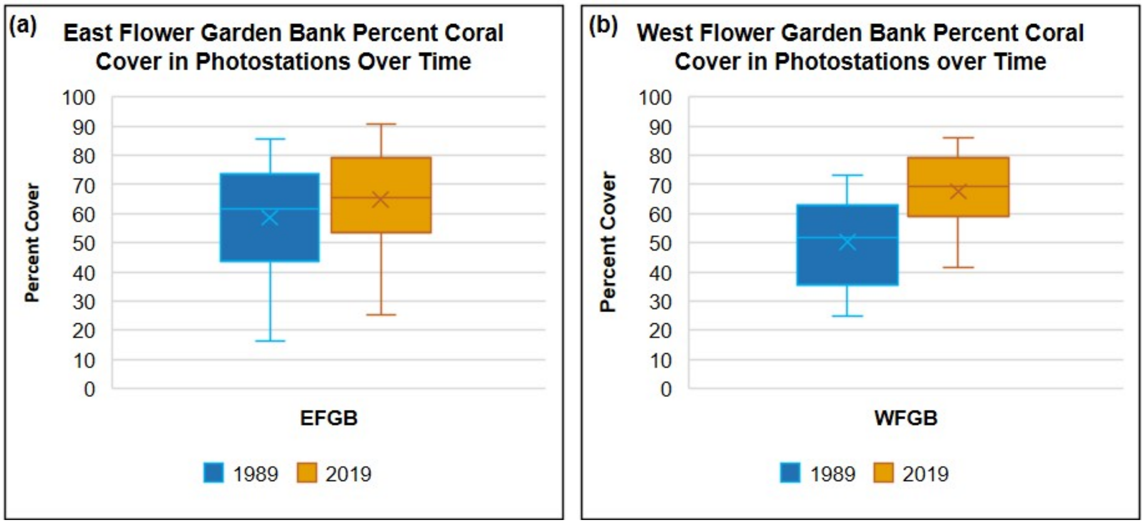


Figure 2.13. Box plot depicting percent coral cover in (a) EFGB (n=24) and (b) WFGB (n=27) repetitive photostations in 1989 and 2019.

As an example of the value of long-term repetitive photographs, Figure 2.14 documents changes in select photostations over time. It should be noted that some colonies appeared paler in certain years due to variations in photographic equipment (e.g., 35 mm slides, 35 mm film, and digital images), ambient conditions, and as colony health or condition changed. Furthermore, photo quality is affected by time of day, camera settings, and lighting. In EFGB repetitive photostation #102, changes over time include bare substrate colonization and overgrowth by *P. strigosa* and *P. astreoides* colonies in the center of the station from 1989 to 2019 and algal colonization on a *P. strigosa* colony in the lower left corner affecting approximately 50% of the colony in 2019 (Figure 2.14a and b). This photostation represents an extreme example of increased coral cover, but nevertheless, illustrates the value of a historic photographic record. In WFGB photostation #501, *O. franksi* cover increases from 1989 to 2019 and a black *Ircinia strobilina* sponge that was present in 1989 is absent in 2019 (Figure 2.14c and d). In WFGB photostation #503, a large *C. natans* colony present in 1989 is absent in 2019 (Figure 2.14e and f). In EFGB repetitive photostation #D07, large *M. cavernosa* colonies in the center of the station gained tissue over the years, and the margin of the *C. natans* colony on the left side of the station grew closer to the *M. cavernosa* colonies (Figure 2.14g and h).

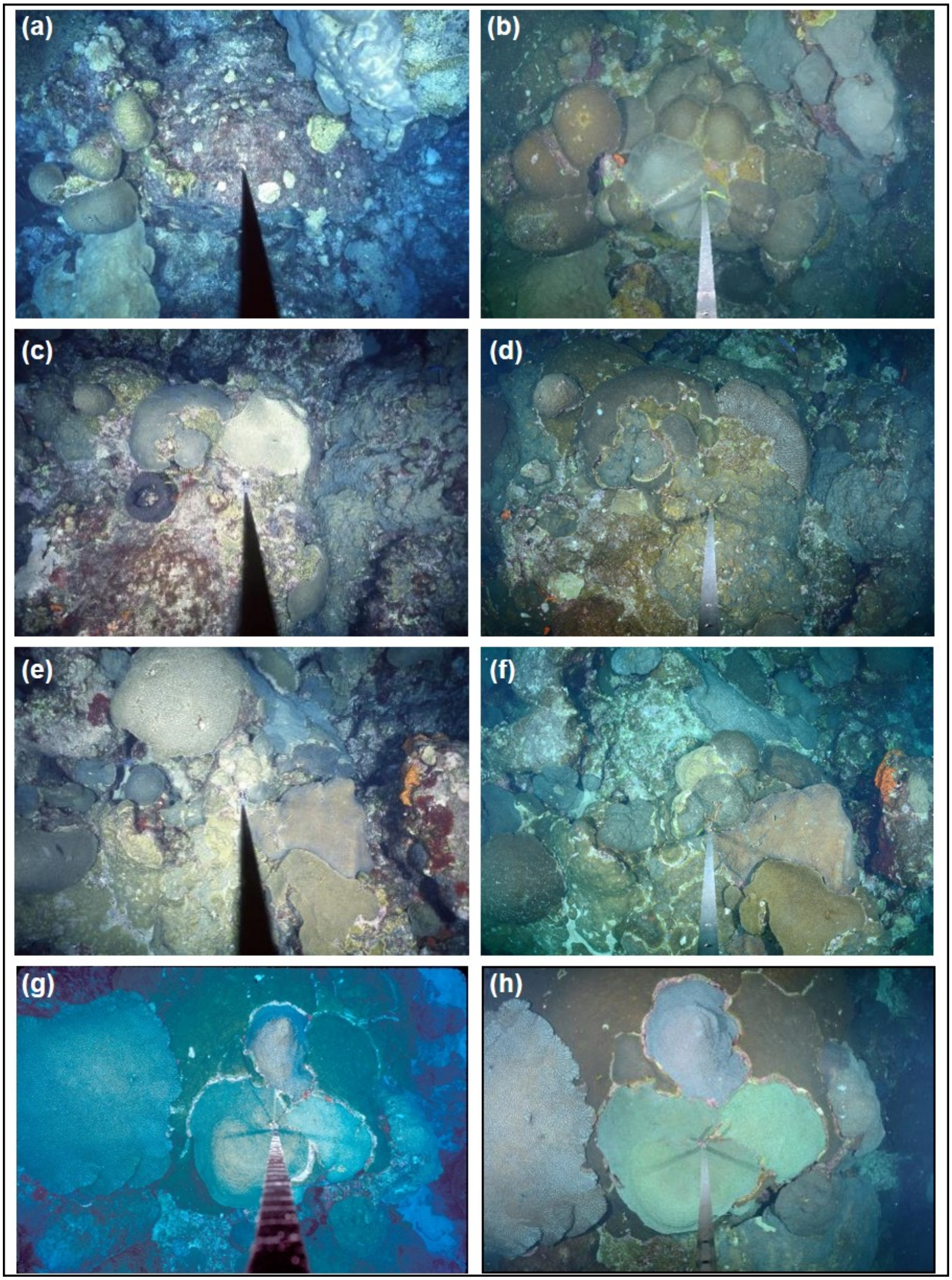


Figure 2.14. Time series of select repetitive photostations: EFGB repetitive photostation #102 (20 m) from (a) 1989 to (b) 2019; WFGB repetitive photostation #501 (20 m) from (c) 1989 to (d) 2019; WFGB repetitive photostation #503 (20 m) from (e) 1989 to (f) 2019; and EFGB repetitive photostation #D07 (36 m) from (g) 2005 to (h) 2019. Photos: (a, c, d) MMS (Gittings et al. 1992), (b, d, f, g, h) NOAA

Discussion

Despite global coral reef declines in recent decades, mean coral cover within EFGB and WFGB one-hectare study sites has remained near or above 50% for the combined 30 years of monitoring. Mean macroalgae percent cover increased from approximately 5% to 20% between 1998 and 1999, and increased to approximately 30% over the past ten years. The inverse relationship between macroalgae and colonizable substrate observed throughout the long-term monitoring program in random transect surveys reflects the tendency for macroalgae to grow over exposed hard bottom rather than outcompeting coral or sponges.

These general trends suggest that from 1992 to 1998 the reef community within the one-hectare study sites was stable, and from 1999 onward, there was a shift as colonizable substrate was populated by macroalgae. In contrast to other shallow water reefs in the Caribbean region and many worldwide, increases in macroalgae have not coincided with or caused a decline in coral cover in the EFGB and WFGB one-hectare study sites (Gardner et al. 2003; Mumby and Steneck 2011; DeBose et al. 2012; Jackson et al. 2014; Johnston et al. 2016a, 2017a, 2017b, 2018a, 2020). While a portion of EFGB outside the one-hectare study site was affected by a localized mortality event in July of 2016 (275 m from the EFGB one-hectare study site) and both banks were impacted by coral bleaching in the fall of 2016, neither event resulted in significant coral cover declines within the one-hectare study sites (Johnston et al. 2018b, 2019).

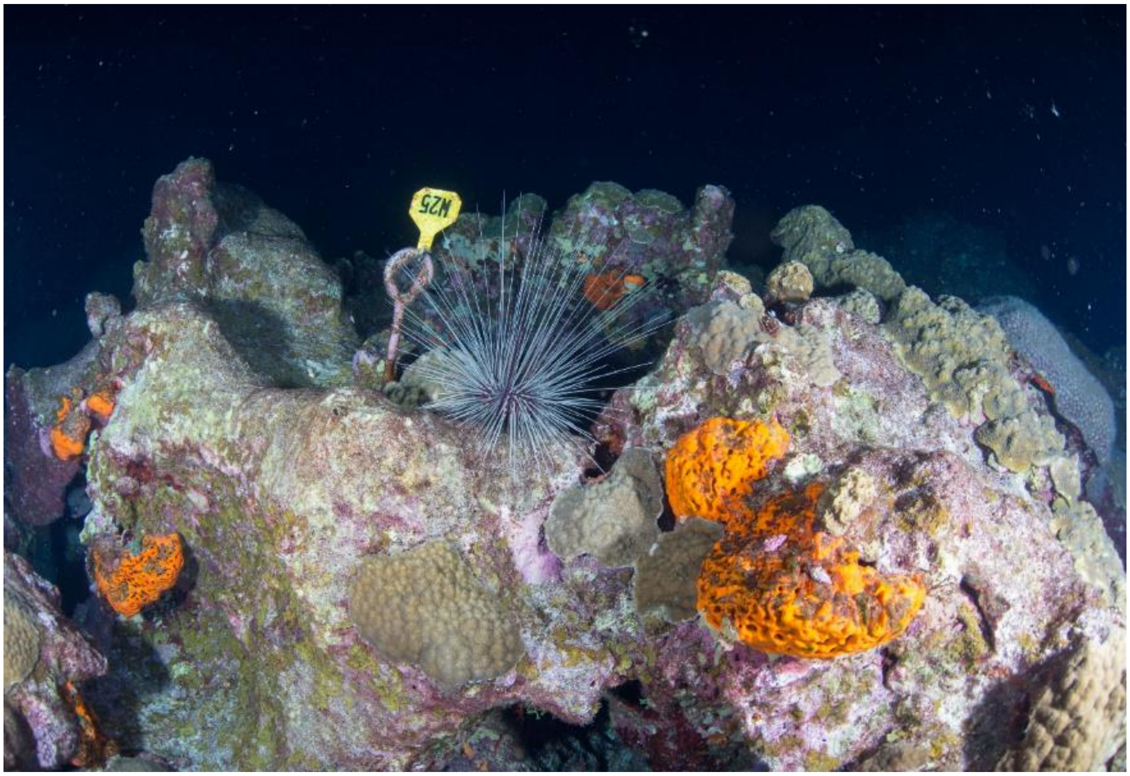
Increases in macroalgae have occurred on other reefs in the Gulf of Mexico and Caribbean region. For example, Stetson Bank, a series of claystone and siltstone pinnacles once covered by a low-diversity coral and sponge community (located 48 km northwest of WFGB), has shown a similar but more prominent trend of increasing macroalgae. Macroalgae cover peaked in 2012, and coincided with decreased sponge and coral cover (DeBose et al. 2012; Nuttall et al. 2018, 2020). Increasing macroalgae and significant coral declines have occurred within Florida Keys National Marine Sanctuary (Toth et al. 2014), which is also within the Gulf region. Mean coral cover sanctuary-wide declined from 13% in 1996 to 7% in 2008, and was as low as 3% in some areas of the Florida Keys in 2011 (Ruzicka et al. 2009; ONMS 2011; Toth et al. 2014). This decline in the Florida Keys was most likely due to a combined effect of disease, hurricane damage, and thermal stress (Toth et. al 2014). Coral reef growth in Florida has recently been identified as severely impacted, as coral bleaching from 2014 to 2015 and the rapid progression of Stony Coral Tissue Loss Disease since 2014 has caused substantial declines (NOAA CRCP 2020). Overfishing, bleaching, algae competition, coastal development, and coral disease have all been implicated in declines on reefs in the wider Caribbean (Gardner et al. 2003; Steneck et al. 2011; Jackson et al. 2014). At the time of this report, no coral disease was observed within EFGB and WFGB random transect surveys or repetitive photostations.

Within EFGB and WFGB random transect surveys, thirteen coral species were observed, while fifteen coral species were observed in repetitive photostations in 2019. The species observed in the repetitive photostations that were not observed in random transect surveys included *Madracis auretenra* and *Porites furcata*. Significantly higher mean coral cover estimates (69%) were obtained from the repetitive photostations (ranging in depth from 18–39 m) than the random transects (57%; ranging in depth from 17–27 m) in both EFGB and WFGB. This has been documented in previous reports (Precht et al. 2006; Zimmer et al. 2010; Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020), and it should be noted that the repetitive photostations were not intended to be representative of the coral reef communities within the one-hectare study sites and may not provide an accurate assessment of the cover of species within the habitats on the slope of the reef cap, as repetitive photostations were not randomly selected. The randomly selected benthic transects are the primary mechanism for community analysis for the one-hectare study sites, while the repetitive photostations provide a long-term dataset allowing for specific conclusions about individual corals and processes (competition, overgrowth, growth rates, etc.) over time while controlling for small-scale environmental heterogeneity.

In the repetitive photostations, coral cover significantly increased with depth. It is not unusual for corals on the slopes of reefs to grow flatter in order to cover additional bottom area to more efficiently capture sunlight than those at shallower depths (Hoeksema et al. 2017). The predominant coral in repetitive photostations at all depths was *O. franksi*; however, *M. cavernosa* ranked second within the 32–39 m depth range, whereas *P. strigosa* ranked second in the shallower areas. A noticeable difference within the 32–39 m depth range was the lack of *Orbicella annularis* and low cover of *P. strigosa*. *Stephanocoenia intersepta* and *Madracis* spp. were more abundant in photostations within the 32–39 m depth range. It should be noted that all comparisons within this category are intended solely to compare among the groups of repetitive photostations, as they were not randomly selected, and therefore have similar bias. While these stations can help identify directions and causes of change, they are not intended to estimate or compare reef-wide populations or communities. Despite higher coral cover in repetitive photostations than random transect surveys, these sites are critical in enabling researchers to track individual colonies over time, especially during extreme events, such as the 2016 bleaching event (Johnston et al. 2019), and as environmental conditions change (Knowlton and Jackson 2008; Heron et al. 2016; van Hooidonk et al. 2016; Hughes et al. 2017).

In summary, the EFGB and WFGB one-hectare study sites have not shown any decline in coral cover since 1989. In fact, as reported herein, the opposite has been documented, and the FGBNMS reefs have 6 to 11 times higher coral cover values than selected other locations in the Caribbean region (Caldow et al. 2009; Clark et al. 2014; Jackson et al. 2014; Johnston et al. 2017a, b). This may be due to the remote, offshore location and deep water surrounding the banks, which provides a more stable environment than shallower, coastal reefs (Aronson et al. 2005; Johnston et al. 2015). However, despite their remote location and deeper depth compared to other Caribbean reefs, EFGB and WFGB are not impervious to impacts, as seen with the 2016 localized mortality and bleaching events (Johnston et al. 2018b, 2019). Climate change, invasive species, and water quality degradation continue to threaten the resources of the FGBNMS (ONMS 2008; Nuttall et al. 2014; Johnston 2016b). As the environment in the Gulf of Mexico changes over time (Karnauskas et al. 2015), continued monitoring will be important to document ecosystem variation.

Chapter 3: Sea Urchin and Lobster Surveys



A long-spined sea urchin (*Diadema antillarum*) on the reef at WFGB. Photo: G.P. Schmahl/NOAA

Introduction

The long-spined sea urchin (*Diadema antillarum*) was an important herbivore on coral reefs throughout the Caribbean until 1983, when an unknown pathogen decimated populations throughout the region, including FGBNMS (Gittings and Bright 1987). This invertebrate is a significant marine herbivore and can substantially control macroalgae cover on coral reefs. Additionally, lobsters are commercially important species throughout much of the Caribbean and Gulf of Mexico; however, population dynamics of Caribbean spiny lobster (*Panulirus argus*) and spotted spiny lobster (*Panulirus guttatus*) at EFGB and WFGB are not well understood. Therefore, sea urchin and lobster surveys help document the abundance of these species within the one-hectare study sites.

Methods

Field Methods

Due to the nocturnal nature of these species, visual surveys were conducted at night, a minimum of 1.5 hours after sunset. Surveys for *D. antillarum*, *P. argus*, and *P. guttatus* were conducted along all six 100-m lines (four perimeter lines and two center crosshairs) in the one-hectare study sites. A 2-m-wide belt transect was surveyed along each of the six 100-m lines, thus totaling 1,200 m^2 per bank. The first diver began on the right side of the line and the second diver on the left.

Divers swam slowly along the boundary line, recording sea urchins and lobsters within a 1-m swath on their side of the line. Divers used flashlights to look into and under reef crevices and, if a sea urchin or lobster was seen, observations, including bank, boundary line, and the number observed, were recorded on a datasheet. In 2019, all lines were surveyed within the EFGB and WFGB one-hectare study sites.

Consistency for the survey method was ensured by using multiple, scientific divers trained to identify sea urchin and lobster species located at FGBNMS. Divers were experienced in the survey technique used, and equipment checklists were provided to ensure divers had equipment for assigned tasks. QA/QC procedures ensured surveyors reviewed and entered species count data in a Microsoft® Excel® database on the same date the survey took place. All datasheets were reviewed and compared to data entered in the database during field operations to check for entry errors, and mistakes were corrected before data analysis was completed.

Data Analysis

All sea urchins and lobsters observed on each 100-m line were summed per one-hectare study site. Density was calculated as number of individuals per 100 m^2 for each species ± SE. Statistical analyses were conducted on square-root-transformed density data using non-parametric distance-based analyses with Primer® version 7.0 (Anderson et al. 2008; Clarke et al. 2014). PERMANOVA examined differences in density between year and one-hectare study sites with a Bray-Curtis similarity matrix and added dummy variable (value of 1). Tests for significant correlation between sea urchin density and macroalgae percent cover were completed in R version 2.13.2 with Pearson's correlation (Helsel and Hirsch 2002).

Results

Density of *D. antillarum* was 1.25 individuals/100 m^2 ± 0.56 within the EFGB one-hectare study site and 28.08 individuals/100 m^2 ± 7.06 within the WFGB one-hectare study site in 2019 (Figure 3.1). One *P. guttatus* was observed in the EFGB one-hectare study site and no *P. argus* were observed (Figure 3.1).

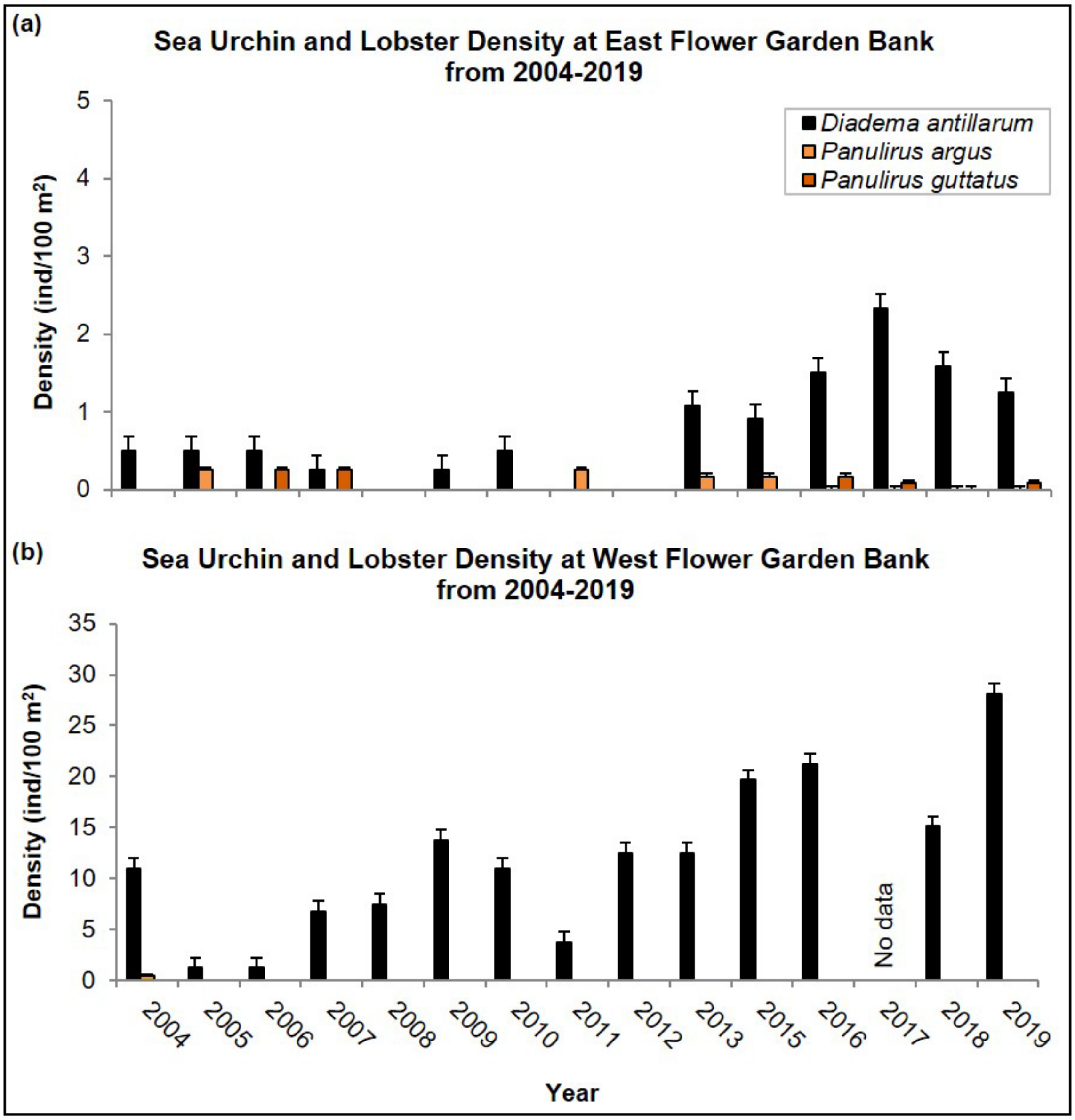


Figure 3.1. Sea urchin and lobster density (individuals/100 m2) + SE within EFGB and WFGB one-hectare study sites from 2004 to 2019 (note the difference in scale on the y-axis).

No data were available for either bank in 2014 and at WFGB in 2017. Sources: 2004 to 2008, PBS&J (Precht et al. 2006; Zimmer et al. 2010); 2009 to 2018, FGBNMS (Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020).

Since 2004, *D. antillarum* densities have ranged from 0–2.3 individuals/100 m² within the EFGB one-hectare study site and 1.25–28.08 individuals/100 m² within the WFGB one-hectare study site. Higher numbers of *D. antillarum* have been observed during surveys at the WFGB one-hectare study site throughout the monitoring program (Figure 3.1). Since 2004, lobster densities have ranged from 0–0.25 individuals/100 m² within the EFGB and WFGB one-hectare study sites combined.

When *D. antillarum* density was compared between one-hectare study sites and years, PERMANOVA analysis revealed that sea urchin density was significantly greater within the WFGB one-hectare study site (Table 3.1).

Table 3.1. PERMANOVA results comparing sea urchin densities between EFGB and WFGB one-hectare study sites and years 2004 to 2019. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	11168	1	55.94	**0.001**
Year	3666	14	1.31	0.263
Res	2595	13		
Total	17087	28		

Due to the importance of *D. antillarum* as herbivores on coral reefs, sea urchin density from 2004 to 2019 and macroalgae percent cover for EFGB and WFGB were tested for correlation; however, no significant correlation was found, despite higher *D. antillarum* densities and lower macroalgae percent cover at WFGB.

Discussion

Diadema antillarum are important herbivores on coral reefs, helping to reduce macroalgae cover through grazing, making room for coral growth and new recruits (Edmunds and Carpenter 2001; Carpenter and Edmunds 2006). After the mass die off in 1983, *D. antillarum* populations have not recovered to pre-1983 levels, which were at least 140 individuals/100 m^2 at EFGB and 50 individuals/100 m^2 at WFGB (Gittings and Bright 1986, 1987; Gittings 1998). Post-1983, *D. antillarum* densities dropped to near zero (Gittings and Bright 1987). Since then, patchy but limited recovery has been documented in the Caribbean region (Edmunds and Carpenter 2001; Kramer 2003; Carpenter and Edmunds 2006). *Diadema antillarum* densities at nearby Stetson Bank increased from 2009 to 2014 and have plateaued in recent years, averaging 105 individuals/100 m^2 in 2018 (Nuttall et al. 2020). No estimates of sea urchin abundance were made at Stetson Bank prior to the die off.

Diadema antillarum populations within the EFGB one-hectare study site remained low during the 2019 monitoring period and were similar to those reported in previous studies (Zimmer et al. 2010; Johnston et al. 2017a, 2017b, 2018a, 2020). Populations within the WFGB one-hectare study site have been consistently higher than EFGB. Continued monitoring will be required to track and compare temporal changes at both one-hectare study sites. Lobster densities within EFGB and WFGB one-hectare study sites have been historically low throughout the monitoring program. Lobsters are, however, occasionally observed by divers, and occur on the banks in low abundance.

Chapter 4:
Fish Surveys

A school of crevalle jack (Caranx hippos) swim over the reef at East Flower Garden Bank. Photo: Kelly Drinnen/NOAA

Introduction

Divers conducted stationary reef fish visual census surveys in EFGB and WFGB one-hectare study sites to examine fish population composition and changes over time. The surveys were used to characterize and compare fish assemblages between one-hectare study sites and years.

Methods

Field Methods

Fishes were assessed by divers using modified stationary reef fish visual census surveys based on methods originally described by Bohnsack and Bannerot (1986). Twenty randomly located surveys were conducted within the one-hectare study site at EFGB and 25 were conducted within the one-hectare study site at WFGB. Each survey represented one sample. Observations of fishes were restricted to an imaginary cylinder with a 7.5 m radius, extending from the substrate to the surface (for more detailed methods, reference Johnston et al. 2017a) (Figure 4.1).

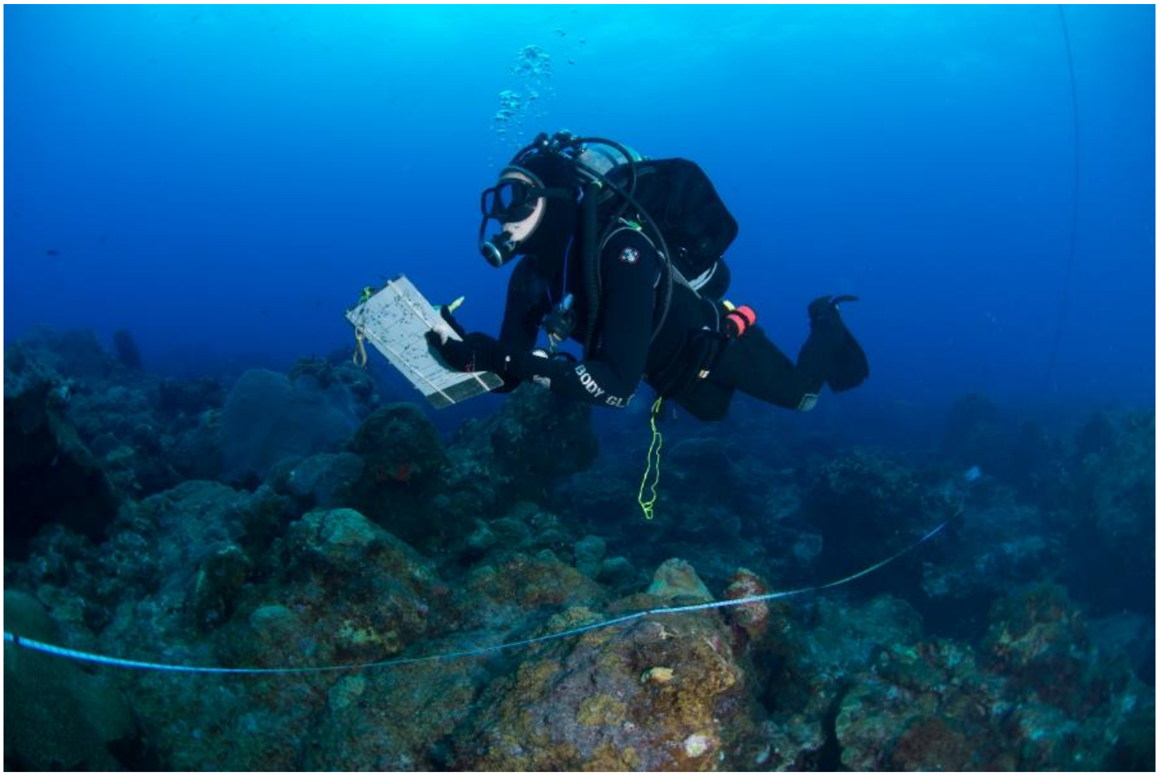


Figure 4.1. NOAA diver conducting a fish survey at EFGB. Photo: G.P. Schmahl/NOAA

All fish species observed within the first five minutes of the survey were recorded while the diver slowly rotated in place in the imaginary survey cylinder. Immediately following this five-minute observation period, one rotation was conducted for each species noted in the original five-minute period to record abundance (number of individuals per species) and fork length (within size bins). Size for each individual was estimated and binned into one of eight groups: <5 cm, ≥5 to <10 cm, ≥10 to <15 cm, ≥15 to <20 cm, ≥20 to <25 cm, ≥25 to <30 cm, ≥30 to <35 cm, and ≥35 cm. If fishes were greater than 35 cm in length, divers estimated the size to the nearest cm. Each survey required approximately 15 to 20 minutes to complete. Transitory or schooling species were counted and measured at the time the individuals moved through the cylinder during the initial five-minute period. After the initial five-minute period, additional species were recorded but marked as observed after the official survey period. These observations were excluded from the analysis, unless otherwise stated, except for reporting the total number of species observed in all 2019 surveys. Fish surveys began in the early morning (after 0700 CDT), and were repeated throughout the day until dusk (1900 CDT).

Consistency in the survey method was ensured with the use of scientific divers trained to identify FGBNMS fish species. Divers were experienced in the survey technique used, and equipment checklists were provided in the field to ensure divers had equipment for assigned tasks. All fish survey divers carried a pre-marked PVC measuring stick to provide a size reference.

Data Processing

Surveyors reviewed and entered fish survey data in a Microsoft® Excel® database on the same date the survey took place. Fish survey datasheets were retained and reviewed after fieldwork was completed for QA/QC. All datasheets were reviewed and compared to data entered in the database to check for entry errors, and any mistakes were corrected prior to data processing. For each entry, fish family, trophic guild, and biomass were automatically recorded in the database (Bohnsack and Harper 1988; Froese and Pauly 2019). Species were classified into four major trophic guild categories: herbivores (H), piscivores (P), invertivores (I), and planktivores (PL).

Statistical Analysis

Summary statistics of fish census data included abundance, density, sighting frequency, species richness, and biomass. Total abundance was calculated as the number of individuals per sample, and percent relative abundance was the total number of individuals for one species divided by the total of all species and multiplied by 100. Density was expressed as the number of individual fish per 100 m^2 ± SE, and calculated as the total number of individuals per sample by the area of the survey cylinder (176.7 m^2) and multiplied by 100. Sighting frequency for each species was expressed as the percentage of the total number of samples in which the species was recorded. Mean species richness was the average number of species represented per sample ± SE. Fish biomass was expressed as grams per 100 m^2 ± SE and computed by converting length data to weights using the allometric length-weight conversion formula (Bohnsack and Harper 1988) based on information provided by FishBase (Froese and Pauly 2019). As sizes less than 35 cm were binned, the median size in each size bin was used to calculate biomass (for example, fish in the ≥5 to <10 cm size bin were assigned the total length of 7.5 cm). Observations of manta rays and stingrays were removed from biomass analyses only, due to their rare nature and large size.

For family analysis, percent coefficient of variation (CV%) was calculated to determine the power of the analyses. CV% was calculated using the following formula:

$$CV\% = SE/\bar{X}$$

where SE = standard error and $\bar{X}$ = population mean. A CV% of 20% or lower is optimal, as it would be able to statistically detect a minimum change of 40% in the population within the survey period (Roberson et al. 2014).

Statistical analyses were conducted on square-root-transformed density and biomass data (reducing the influence of large schooling species on analyses) using distance-based Bray-Curtis similarity matrices with Primer® version 7.0 (Anderson et al. 2008; Clarke et al. 2014). Differences in the fish community based on species level resemblance matrices were investigated using PERMANOVA (Anderson et al. 2008). If significant differences were found, species contributing to observed differences were examined using SIMPER to assess the percent contribution of species to dissimilarity between one-hectare study sites (Clarke et al. 2014). Differences at the family level for key species were compared for dissimilarities using ANOSIM. For long-term density and biomass trends for which data were available (2011 to 2018), the distance between centroids was calculated from Bray-Curtis similarity matrices and visualized using metric multi-dimensional scaling plots with a time series trajectory overlay split between locations (Anderson et al. 2008).

Dominance plots were generated based on species abundance and biomass with Primer® version 7.0 (Anderson et al. 2008; Clarke et al. 2014). W-values (difference between the biomass and abundance curves) were calculated for each survey (Clarke 1990). W-values range between -1<w>1, where w=1 indicates that the population is dominated by a few large species, w=-1 indicates that the population is dominated by numerous small species, and w=0 indicates that accumulated biomass is evenly distributed between large and small species. Dissimilarities in w-values between one-hectare study sites were assessed using ANOSIM on untransformed data with Euclidean distance similarity matrices (Clarke et al. 2014).

Results

A combined total of 30 families and 84 species (77 at EFGB and 63 at WFGB, respectively) were observed in 2019 at EFGB and WFGB one-hectare study sites. Mean species richness was 19.80 ± 1.39 per survey at EFGB, 21.60 ± 0.84 per survey at WFGB, and 20.80 ± 0.78 per survey for both EFGB and WFGB one-hectare study sites combined. Bonnetmouth (*Emmelichthyops atlanticus*) had the highest relative abundance of all species in EFGB surveys (47.54%), followed by mackerel scad (*Decapterus macarellus*) (18.15%), bluehead (*Thalassoma bifasciatum)* (7.51%), brown chromis (*Chromis multilineata*) (7.35%), and creole wrasse (*Clepticus parrae*) (3.95%) (Figure 4.2). In WFGB surveys, bonnetmouth had the highest relative abundance (62.70%), followed by brown chromis (8.05%), Atlantic creolefish (*Paranthias furcifer*) (5.27%), creole wrasse (5.13%), and bluehead (4.91%) (Figure 4.2).

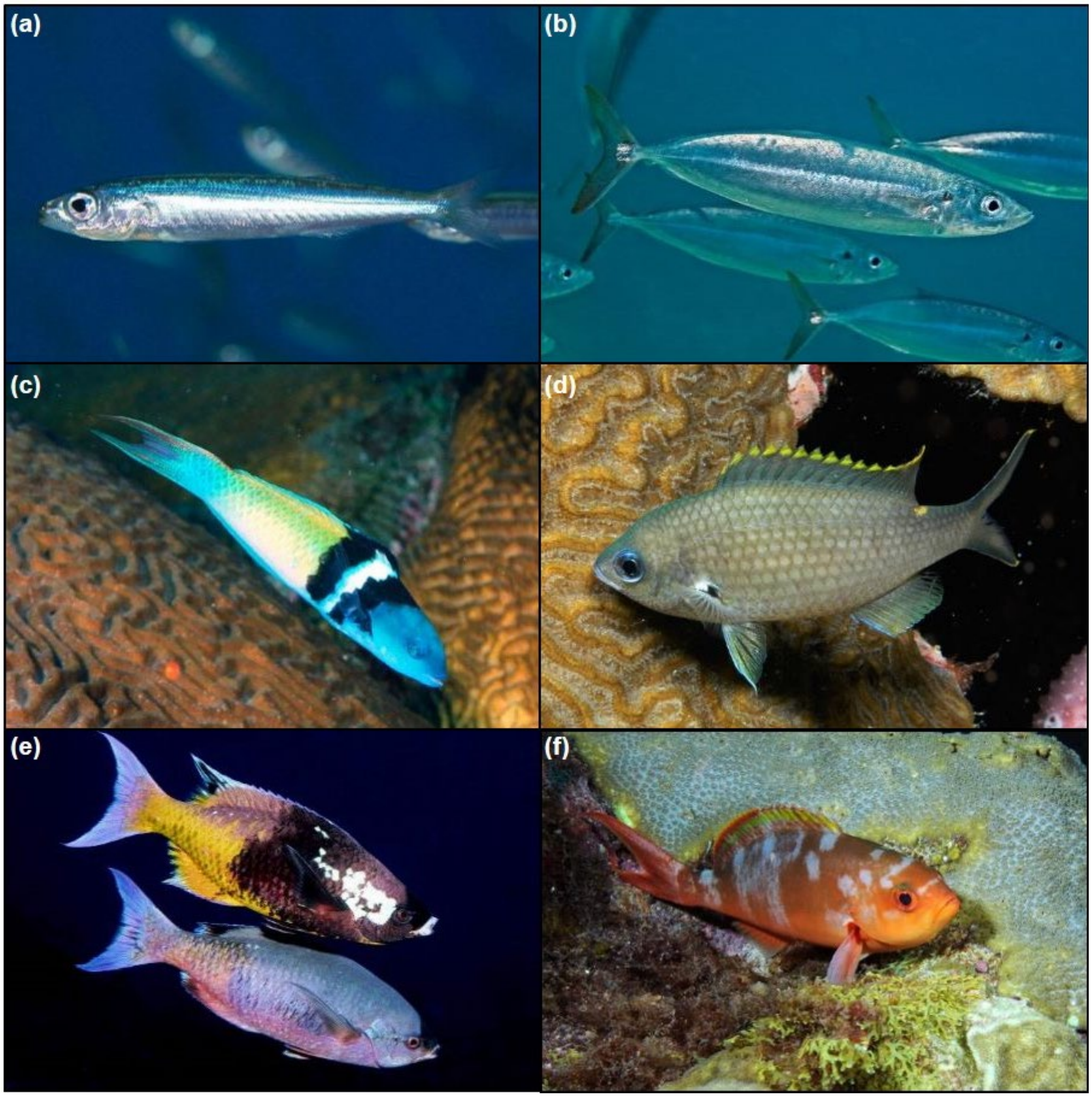


Figure 4.2. Most abundant fish species observed at EFGB and WFGB one-hectare study sites in 2019: (a) bonnetmouth, (b) mackerel scad, (c) bluehead, (d) brown chromis, (e) creole wrasse, and (f) Atlantic creolefish. Photos: (a, b) Carlos Estapé, (c, d, e, f) G.P. Schmahl/NOAA

Sighting Frequency and Occurrence

The most frequently sighted species was bluehead, observed in 95% of surveys at EFGB and 96% of surveys at WFGB. Other frequently sighted species included brown chromis, great barracuda (*Sphyraena barracuda*), and bicolor damselfish (*Stegastes partitus*) (Table 4.1). Two manta rays (*Manta* spp.) and no sharks were observed in surveys and are considered “rare,” typically occurring in <20% of all surveys (REEF 2014).

Table 4.1. Sighting frequencies for the 20 most frequently sighted species at EFGB and WFGB one-hectare study sites, including all surveys combined, in 2019.

Fish Species	EFGB	WFGB	Combined
Bluehead (*Thalassoma bifasciatum*)	95.00%	96.00%	95.56%
Brown chromis (*Chromis multilineata*)	85.00%	92.00%	88.89%
Great barracuda (*Sphyraena barracuda*)	85.00%	92.00%	88.89%
Bicolor damselfish (*Stegastes partitus*)	85.00%	88.00%	86.67%
Blue tang (*Acanthurus coeruleus*)	90.00%	84.00%	86.67%
Queen parrotfish (*Scarus vetula*)	85.00%	88.00%	86.67%
Reef butterflyfish (*Chaetodon sedentarius*)	85.00%	80.00%	82.22%
Cocoa damselfish (*Stegastes variabilis*)	85.00%	76.00%	80.00%
Sharpnose puffer (*Canthigaster rostrata*)	60.00%	88.00%	75.56%
Atlantic creolefish (*Paranthias furcifer*)	60.00%	84.00%	73.33%
Blue chromis (*Chromis cyanea*)	60.00%	84.00%	73.33%
Stoplight parrotfish (*Sparisoma viride*)	85.00%	64.00%	73.33%
Creole wrasse (*Clepticus parrae*)	50.00%	80.00%	66.67%
Spanish hogfish (*Bodianus rufus*)	55.00%	76.00%	66.67%
Black durgon (*Melichthys niger*)	60.00%	60.00%	60.00%
Threespot damselfish (*Stegastes planifrons*)	40.00%	76.00%	60.00%
Graysby (*Cephalopholis cruentata*)	60.00%	52.00%	55.56%
Bonnetmouth (*Emmelichthyops atlanticus*)	35.00%	68.00%	53.33%
Bar jack (*Caranx ruber*)	45.00%	56.00%	51.11%
Redband parrotfish (*Sparisoma aurofrenatum*)	50.00%	48.00%	48.89%

Density

Mean fish density (individuals/100 m²) ± SE was 327.33 ± 77.74 in EFGB surveys, 299.83 ± 59.18 in WFGB surveys, and 312.05 ± 47.18 for one-hectare study site surveys combined. Density was significantly greater in EFGB surveys (Table 4.2). SIMPER analysis identified greater abundance of bonnetmouth (21.23%) at WFGB and greater abundance of mackerel scad at EFGB (5.31%) as the main contributors to the differences (Table 4.3).

Table 4.2. PERMANOVA results comparing mean fish density between EFGB and WFGB one-hectare study sites from 2019. Bold text denotes significant value.

Source	**Sum of Squares**	**df**	**Pseudo-F**	**P (perm)**
Bank	3756	1	2.51	**0.015**
Res	64299	43		
Total	68054	44		

Table 4.3. Mean density (individuals/100 m^2 ± SE) of the 10 most abundant species from EFGB and WFGB one-hectare study site surveys, and all surveys combined, in 2019.

Fish Species	EFGB	WFGB	Combined
Bonnetmouth (*Emmelichthyops atlanticus*)	155.63 ± 63.49	188.00 ± 49.02	173.62 ± 38.83
Mackerel scad (*Decapterus macarellus*)	59.42 ± 34.22	2.83 ± 2.83	27.98 ± 15.66
Brown chromis (*Chromis multilineata*)	24.05 ± 3.32	24.13 ± 3.14	24.10 ± 2.26
Bluehead (*Thalassoma bifasciatum*)	24.59 ± 3.61	14.71 ± 2.61	19.10 ± 2.26
Atlantic creolefish (*Paranthias furcifer*)	12.51 ± 4.10	15.80 ± 8.65	14.34 ± 5.10
Creole wrasse (*Clepticus parrae*)	12.93 ± 5.17	15.39 ± 4.53	14.30 ± 3.37
Blue chromis (*Chromis cyanea*)	2.83 ± 1.03	10.25 ± 2.37	6.95 ± 1.49
Bicolor damselfish (*Stegastes partitus*)	4.10 ± 1.12	3.89 ± 0.59	3.99 ± 0.59
Threespot damselfish (*Stegastes planifrons*)	0.91 ± 0.38	3.10 ± 0.84	2.13 ± 0.52
Cocoa damselfish (*Stegastes variabilis*)	2.57 ± 0.65	1.72 ± 0.73	2.10 ± 0.49

Trophic Guild Analysis

Species were grouped by trophic guild into four major categories, as defined by NOAA's Center for Coastal Monitoring and Assessment BioGeography Branch fish-trophic level database: herbivores, piscivores, invertivores, and planktivores (Caldow et al. 2009). Size-frequency distributions using relative abundance were graphed for each trophic guild (Figure 4.3).

Piscivores dominated the small and large size classes at both one-hectare study sites. Herbivores dominated the mid-range size classes. Invertivores and planktivores were more variable across mostly small and mid-size classes (Figure 4.3).

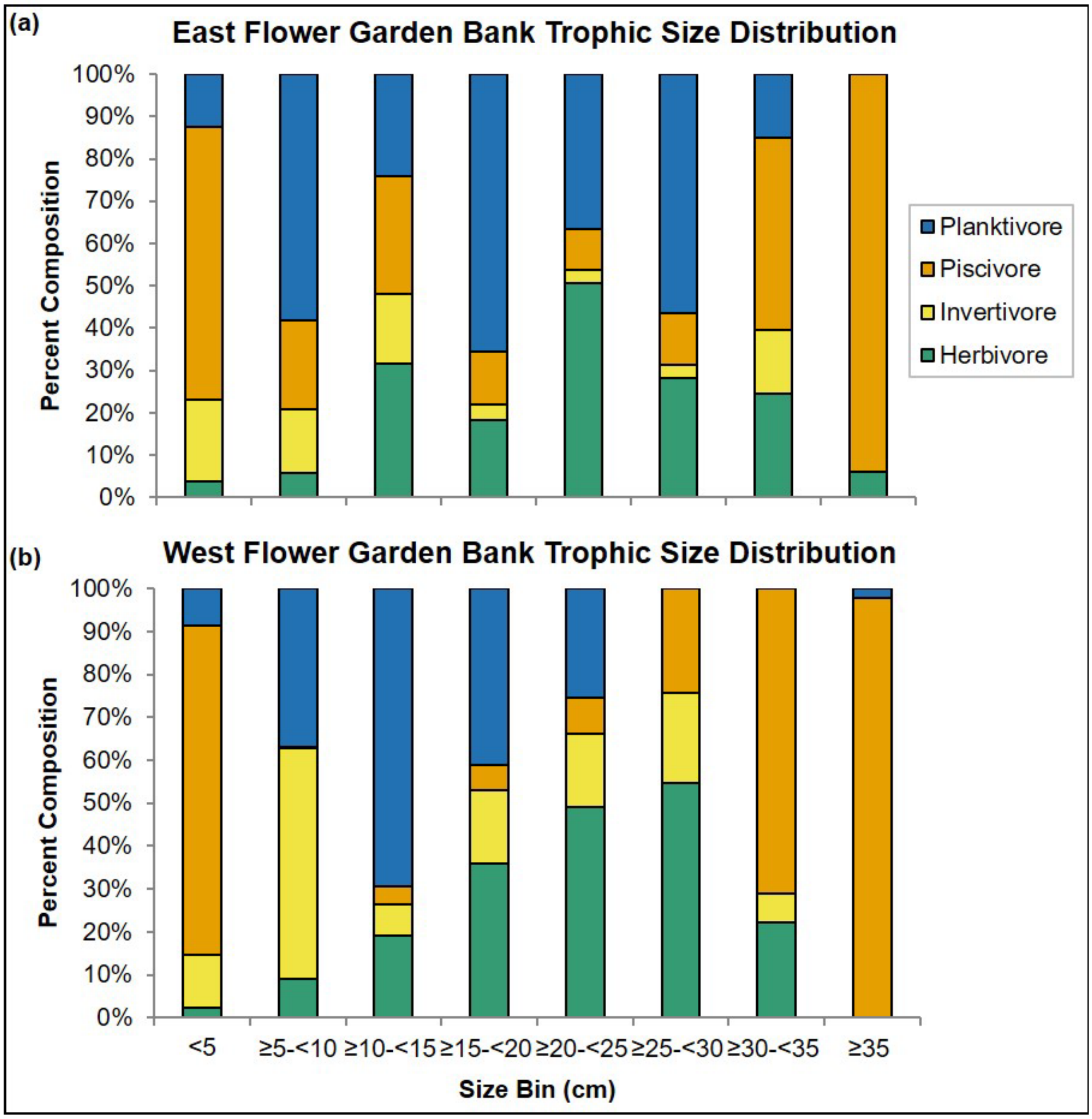


Figure 4.3. Fish size distribution by trophic guild at (a) EFGB and (b) WFGB one-hectare study sites in 2019.

Biomass

Mean biomass (g/100 m² ± SE) was 7,091.15 ± 1,923.40 in EFGB surveys, 6,035.98 ± 1,090.06 in WFGB surveys, and 6,504.95 ± 1,037.46 for one-hectare study site surveys combined in 2019. PERMANOVA analysis revealed that fish biomass was significantly greater in EFGB surveys (Table 4.4). SIMPER analysis identified the main contributor to higher fish biomass at the EFGB one-hectare study site was greater local abundance of Atlantic creolefish (12.77%).

Table 4.4. PERMANOVA results comparing mean fish biomass between EFGB and WFGB one-hectare study sites from 2019. Bold text denotes significant value.

Source	**Sum of Squares**	**df**	**Pseudo-F**	**P (perm)**
Bank	3867	1	1.77	**0.017**
Res	93757	43		
Total	97624	44		

When classified by trophic guild, piscivores possessed the highest mean biomass for all surveys; invertivores had the lowest mean biomass (Table 4.5). No significant differences were found among trophic guilds between one-hectare study sites. Overall, piscivores represented approximately 53% of biomass, followed by herbivores (23%), planktivores (17%), and invertivores (7%) for one-hectare study sites combined.

Table 4.5. Mean biomass (g/100 m^2) ± SE for each trophic guild from EFGB and WFGB one-hectare study site surveys, and surveys from both banks combined, in 2019.

Trophic Group	EFGB	WFGB	Combined
Herbivore	2,183.23 ± 488.19	902.62 ± 180.52	1,471.78 ± 219.40
Invertivore	538.33 ± 120.37	383.59 ± 76.72	452.36 ± 67.43
Planktivore	1,858.47 ± 415.57	565.17 ± 113.03	1,139.97 ± 169.94
Piscivore	2,511.12 ± 561.50	4,184.60 ± 836.92	3,440.83 ± 512.93

Mean biomass for each species, grouped by trophic guild, is presented in Table 4.6. At the EFGB one-hectare study site, 42% of herbivore biomass was contributed by black durgon (*Melichthys niger*). For invertivores, the greatest contribution was from French angelfish (*Pomacanthus paru*) (23%). For piscivores, great barracuda contributed the greatest biomass (55%). For planktivores, the greatest contribution was from Atlantic creolefish (84%) (Table 4.6).

At the WFGB one-hectare study site, 34% of herbivore biomass was contributed by stoplight parrotfish (*Sparisoma viride*). For invertivores, the greatest contribution was from gray snapper (*Lutjanus griseus*) (20%). For piscivores, horse-eye jack contributed the greatest biomass (44%). For planktivores, the greatest contribution was from Atlantic creolefish (73%) (Table 4.6).

Table 4.6a. Biomass (g/100 m^2) ± SE of each species of herbivore from EFGB and WFGB one-hectare study site surveys, and surveys from both banks combined, in 2019.

Fish Species	**EFGB**	**WFGB**	**Combined**
Black durgon (*Melichthys niger*)	913.38 ± 635.85	151.46 ± 44.78	490.09 ± 285.41
Stoplight parrotfish (*Sparisoma viride*)	525.69 ± 148.63	303.45 ± 71.84	402.22 ± 77.98
Queen parrotfish (*Scarus vetula*)	271.48 ± 61.96	154.79 ± 26.97	206.66 ± 32.15
Blue tang (*Acanthurus coeruleus*)	157.87 ± 80.26	99.74 ± 16.84	125.57 ± 36.62
Chub (Bermuda/yellow) (*Kyphosus saltatrix/incisor*)	146.50 ± 65.20	31.25 ± 22.70	82.47 ± 32.35
Princess parrotfish (*Scarus taeniopterus*)	55.14 ± 25.08	59.07 ± 25.44	57.32 ± 17.81
Redband parrotfish (*Sparisoma aurofrenatum*)	64.31 ± 28.77	42.08 ± 16.21	51.96 ± 15.53

Fish Species	EFGB	WFGB	Combined
Ocean surgeonfish (*Acanthurus tractus*)	5.97 ± 4.34	17.85 ± 5.65	12.57 ± 3.75
Midnight parrotfish (*Scarus coelestinus*)	0.00	15.63 ± 15.63	8.68 ± 8.68
Striped parrotfish (*Scarus iseri*)	19.47 ± 12.91	0.00	8.65 ± 5.84
Cocoa damselfish (*Stegastes variabilis*)	8.56 ± 4.28	6.49 ± 1.28	7.41 ± 2.01
Yellowtail damselfish (*Microspathodon chrysurus*)	3.25 ± 2.24	9.94 ± 4.79	6.97 ± 2.86
Bicolor damselfish (*Stegastes partitus*)	3.34 ± 1.33	5.16 ± 2.76	4.35 ± 1.64
Dusky damselfish (*Stegastes adustus*)	3.11 ± 1.74	2.01 ± 1.22	2.50 ± 1.02
Doctorfish (*Acanthurus chirurgus*)	0.00	1.42 ± 1.07	0.79 ± 0.60
Redlip blenny (*Ophioblennius macclurei*)	0.26 ± 0.22	0.01 ± 0.01	0.12 ± 0.10
Greenblotch parrotfish (*Sparisoma atomarium*)	0.03 ± 0.02	0.00	0.01 ± 0.01

Table 4.6b. Biomass (g/100 m^2) ± SE of each species of invertivore from EFGB and WFGB one-hectare study site surveys, and surveys from both banks combined, in 2019.

Fish Species	EFGB	WFGB	Combined
Gray snapper (*Lutjanus griseus*)	63.81 ± 50.77	75.36 ± 32.25	70.22 ± 28.47
French angelfish (*Pomacanthus paru*)	122.30 ± 71.53	0.00	54.36 ± 32.65
Ocean triggerfish (*Canthidermis sufflamen*)	91.31 ± 49.18	16.86 ± 16.86	49.95 ± 24.11
Queen angelfish (*Holacanthus ciliaris*)	39.02 ± 24.81	33.21 ± 21.33	35.80 ± 16.01
Brown chromis (*Chromis multilineata*)	26.47 ± 8.43	38.45 ± 8.85	33.12 ± 6.18
Spanish hogfish (*Bodianus rufus*)	22.27 ± 7.55	40.69 ± 10.19	32.51 ± 6.66
Reef butterflyfish (*Chaetodon sedentarius)*	35.10 ± 12.58	28.73 ± 9.00	31.56 ± 7.43
Bluehead (*Thalassoma bifasciatum*)	32.58 ± 16.71	14.51 ± 3.82	22.54 ± 7.74
Threespot damselfish (*Stegastes planifrons*)	8.58 ± 4.18	33.02 ± 9.85	22.16 ± 6.01
Rock beauty (*Holacanthus tricolor*)	0.00	36.56 ± 17.67	20.31 ± 10.11
Smooth trunkfish (*Lactophrys triqueter*)	10.72 ± 5.77	20.34 ± 9.87	16.07 ± 6.03
Jolthead porgy (*Calamus bajonado*)	35.13 ± 35.13	0.00	15.61 ± 15.61
Yellow goatfish (*Mulloidichthys martinicus*)	0.00	14.09 ± 6.33	7.83 ± 3.64

Fish Species	EFGB	WFGB	Combined
Gray triggerfish (*Balistes capriscus*)	14.50 ± 14.50	0.00	6.45 ± 6.45
Yellowhead wrasse (*Halichoeres garnoti*)	6.24 ± 4.48	5.48 ± 2.52	5.82 ± 2.41
Spotfin butterflyfish (*Chaetodon ocellatus*)	9.95 ± 9.20	0.29 ± 0.29	4.58 ± 4.10
Sergeant major (*Abudefduf saxatilis*)	1.03 ± 0.60	7.25 ± 4.07	4.49 ± 2.31
Sharpnose puffer (*Canthigaster rostrata*)	0.81 ± 0.32	4.57 ± 1.35	2.90 ± 0.80
Whitespotted filefish (*Cantherhines macrocerus*)	0.00	4.91 ± 4.91	2.73 ± 2.73
Orangespotted filefish (*Cantherhines pullus*)	3.86 ± 2.24	1.54 ± 1.05	2.57 ± 1.15
Puddingwife (*Halichoeres radiatus*)	1.13 ± 0.85	3.55 ± 2.45	2.48 ± 1.41
Longsnout butterflyfish (*Prognathodes aculeatus*)	0.78 ± 0.73	1.96 ± 1.39	1.43 ± 0.83
Honeycomb cowfish (*Acanthostracion polygonius*)	3.18 ± 2.96	0.00	1.41 ± 1.32
Squirrelfish (*Holocentrus adscensionis*)	3.15 ± 3.15	0.00	1.40 ± 1.40
Rock hind (*Epinephelus adscensionis*)	2.32 ± 2.32	0.00	1.03 ± 1.03
Yellowtail reeffish (*Chromis enchrysura*)	0.80 ± 0.80	0.92 ± 0.78	0.87 ± 0.55
Regal demoiselle (*Neopomacentrus cyanomos*)	1.43 ± 0.93	0.15 ± 0.11	0.72 ± 0.42
Balloonfish (*Diodon holocanthus*)	1.21 ± 1.21	0.00	0.54 ± 0.54
Clown wrasse (*Halichoeres maculipinna*)	0.55 ± 0.54	0.44 ± 0.26	0.49 ± 0.28
Foureye butterflyfish (*Chaetodon capistratus*)	0.00	0.68 ± 0.68	0.38 ± 0.38
Redspotted hawkfish (*Amblycirrhitus pinos*)	0.08 ± 0.07	0.00 ± 0.00	0.04 ± 0.03
Neon goby (*Elacatinus oceanops*)	0.01 ± 0.01	0.01 ± 0.01	0.01 ± 0.01
Jackknife fish (*Equetus lanceolatus*)	0.01 ± 0.01	0.00	0.01 ± 0.01

Table 4.6c. Biomass (g/100 m^2) ± SE of each species of invertivore from EFGB and WFGB one-hectare study site surveys, and surveys from both banks combined, in 2019.

Fish Species	EFGB	WFGB	Combined
Great barracuda (*Sphyraena barracuda*)	1388.82 ± 392.59	1394.29 ± 344.86	1391.86 ± 256.16
Horse-eye jack (*Caranx latus*)	585.49 ± 338.32	1831.19 ± 722.44	1277.55 ± 434.55
Crevalle jack (*Caranx hippos*)	209.17 ± 209.17	426.18 ± 426.18	329.73 ± 252.39

Fish Species	EFGB	WFGB	Combined
Dog snapper (*Lutjanus jocu*)	3.06 ± 3.06	276.59 ± 161.57	155.02 ± 91.28
Bonnetmouth (*Emmelichthyops atlanticus*)	138.87 ± 86.12	37.59 ± 9.80	82.60 ± 38.86
Black jack (*Caranx lugubris*)	2.41 ± 2.41	126.10 ± 49.14	71.12 ± 28.61
Graysby (*Cephalopholis cruentata*)	38.37 ± 11.70	35.19 ± 13.30	36.60 ± 8.94
Bar jack (*Caranx ruber*)	51.71 ± 18.99	24.40 ± 6.29	36.54 ± 9.24
Tiger grouper (*Mycteroperca tigris*)	71.58 ± 71.58	0.00	31.81 ± 31.81
Yellowmouth grouper (*Mycteroperca interstitialis*)	5.03 ± 5.03	23.75 ± 21.83	15.43 ± 12.30
Scamp (*Mycteroperca phenax*)	13.10 ± 13.10	0.00	5.82 ± 5.82
Spotted moray (*Gymnothorax moringa*)	0.00	9.34 ± 9.34	5.19 ± 5.19
Needlefish spp. (*Belonidae* spp.)	3.52 ± 3.52	0.00	1.56 ± 1.56

Table 4.6d. Biomass (g/100 m^2) ± SE of each species of planktivore from EFGB and WFGB one-hectare study site surveys, and surveys from both banks combined, in 2019.

Fish Species	EFGB	WFGB	Combined
Atlantic creolefish (*Paranthias furcifer*)	1554.72 ± 649.09	412.87 ± 181.84	920.36 ± 313.36
Creole wrasse (*Clepticus parrae*)	154.05 ± 99.55	143.29 ± 48.11	148.07 ± 51.03
Mackerel scad (*Decapterus macarellus*)	145.82 ± 102.67	0.29 ± 0.29	64.97 ± 46.28
Blue chromis (*Chromis cyanea*)	0.49 ± 0.18	4.61 ± 1.03	2.78 ± 0.65
Yellowtail snapper (*Ocyurus chrysurus*)	3.13 ± 2.60	1.37 ± 1.37	2.15 ± 1.37
Whitefin sharksucker (*Echeneis neucratoides*)	0.00	2.64 ± 2.64	1.46 ± 1.46
Sunshinefish (*Chromis insolata*)	0.26 ± 0.11	0.10 ± 0.05	0.17 ± 0.05
Purple reeffish (*Chromis scotti*)	0.02 ± 0.01	0.00	0.01 ± 0.01

Abundance-Biomass Curves

Mean w-values for both the EFGB and WFGB one-hectare study sites were 0.02 ± 0.01. For all samples at each one-hectare study site, mean w-values remained close to 0, indicating a balanced community where biomass was spread uniformly between large and small individuals (Figure 4.4). ANOSIM comparisons of w-values between one-hectare study sites revealed no significant dissimilarities between the dominance plot w-values.

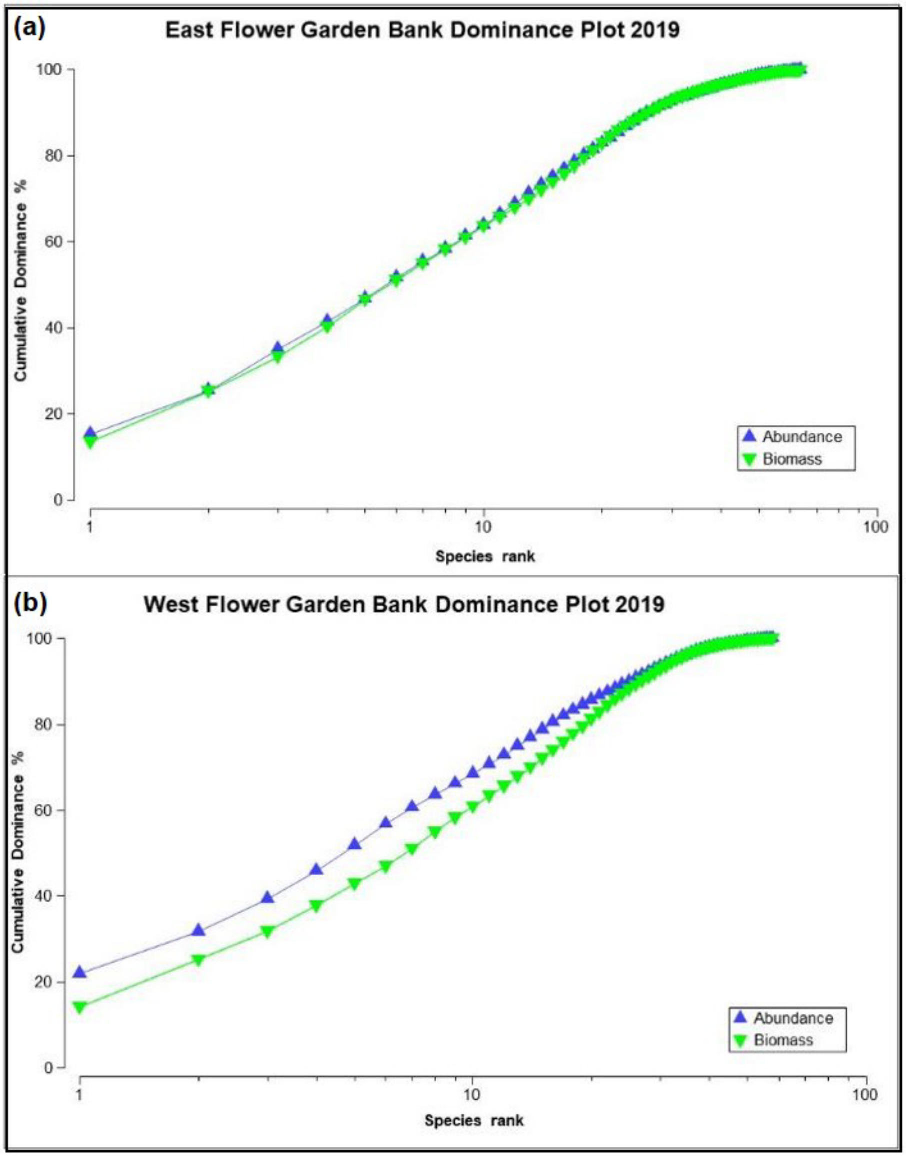


Figure 4.4. Abundance-biomass curves for (a) EFGB and (b) WFGB one-hectare study sites in 2019.

Family Level Analysis

Additional analyses were conducted for grouper and snapper families due to their importance in fishing, and parrotfish due to their role as important herbivores. Further analyses were also conducted for invasive lionfish (*Pterois volitans*) and the non-native regal demoiselle (*Neopomacentrus cyanomos*).

In 2019, five species of grouper were observed in all surveys combined: graysby (*Cephalopholis cruentata*), rock hind (*Epinephelus adscensionis*), scamp (*Mycteroperca phenax*), tiger grouper (*Mycteroperca tigris*), and yellowmouth grouper (*Mycteroperca interstitialis*). It should be noted that the coefficient of variation percentage for density (15.20%) had relatively good power to detect population change while the coefficient of variation percentage for biomass (38.95%) had poor power to detect population differences in 2019. Grouper mean biomass was 123.40 ± 73.21 in EFGB surveys and 58.94 ± 25.07 in WFGB surveys. Mean biomass of small-bodied grouper (graysby, rock hind, and scamp) was 53.79 ± 16.53 in EFGB surveys and 35.19 ± 13.30 in WFGB surveys. Mean biomass of large-bodied grouper (tiger and yellowmouth grouper) was 76.61 ± 71.49 in EFGB surveys and 23.75 ± 21.83 in WFGB surveys. Size distributions of observed grouper in 2019 varied by species (Figure 4.5). No significant differences in the grouper community were detected between one-hectare study sites in 2019.

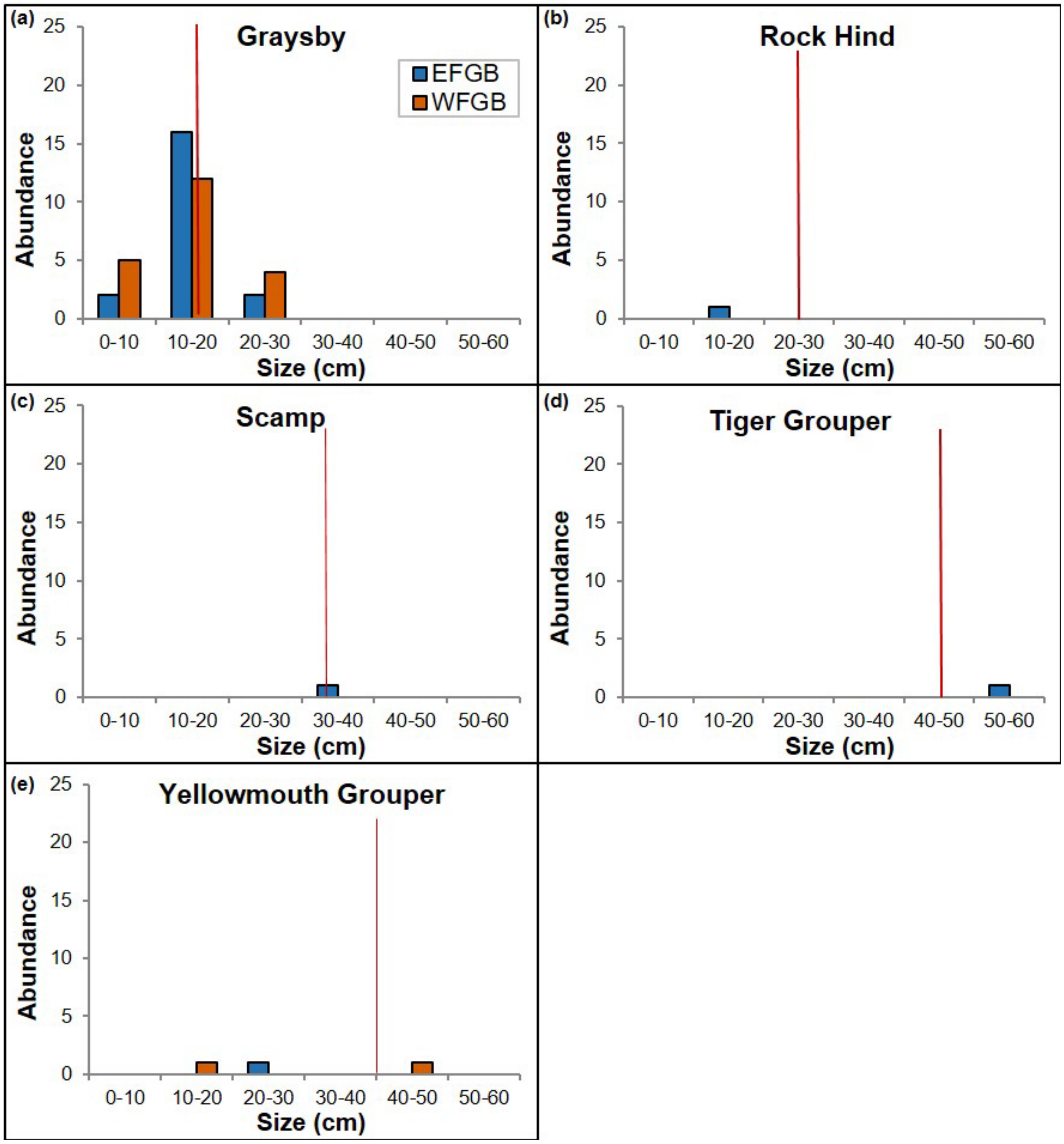


Figure 4.5. Size frequency of grouper species in EFGB and WFGB one-hectare study sites in 2019: (a) graysby, (b) rock hind, (c) scamp, (d) tiger grouper, and (e) yellowmouth grouper. Vertical solid red lines represent estimated size of female maturity (Froese and Pauly 2019).

Threee snapper (*Lutjanidae*) species were observed in 2019 surveys: dog snapper (*Lutjanus jocu*), gray snapper, and yellowtail snapper (*Ocyurus chrysurus*). Coefficient of variation percentages (23.19% for density, 41.88% for biomass) indicated that the data had poor power to detect population differences due to the low number of snapper observed. Mean snapper biomass was 70.00 ± 50.51 in EFGB surveys and 353.32 ± 167.14 in WFGB surveys. Dog snapper were all reproductively immature individuals, while gray snapper size distributions were a mix of both small, reproductively immature individuals and large, reproductively mature individuals. Yellowtail snapper were all reproductively immature individuals (Figure 4.6). No statistical tests were run on the snapper community due to poor statistical power.

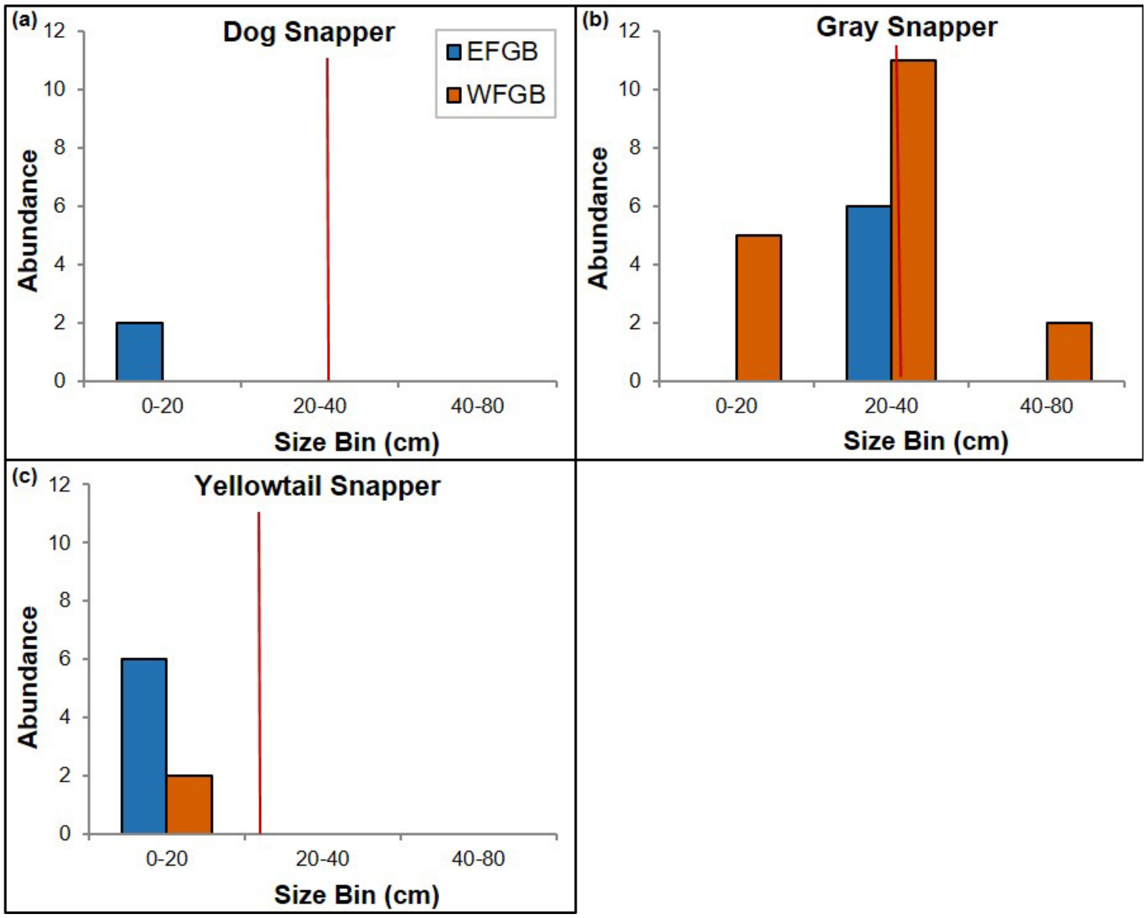


Figure 4.6. Size frequency of snapper species observed in EFGB and WFGB one-hectare study sites in 2019: (a) dog snapper, (b) gray snapper, and (c) yellowtail snapper. Vertical solid red lines represent estimated size of female maturity (Froese and Pauly 2019).

Parrotfishes are important herbivores on coral reefs because they are effective grazers (Jackson et al. 2014). Parrotfish observed in EFGB and WFGB 2019 surveys included seven species: striped parrotfish (*Scarus iseri*), princess parrotfish (*Scarus taeniopterus*), queen parrotfish (*Scarus vetula*), greenblotch parrotfish (*Sparisoma atomarium*), redband parrotfish (*Sparisoma aurofrenatum*), stoplight parrotfish, and midnight parrotfish (*Scarus coelestinus*). Coefficient of variation percentages (14.01% for density and 14.20% for biomass) indicated that the data had good power to detect population differences. Mean biomass of parrotfishes was 936.13 ± 194.58 in EFGB surveys and 577.29 ± 100.81 in WFGB surveys. No significant differences in parrotfish biomass were detected between one-hectare study sites in 2019. The parrotfish population in both one-hectare study sites had wide ranging size distributions, but were dominated by small- to moderate-sized individuals (<5 to <25 cm) (Figure 4.7).

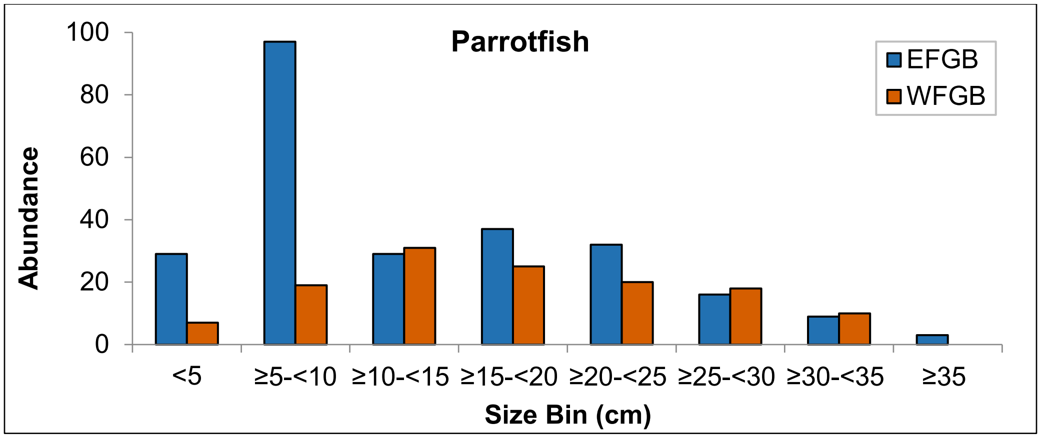


Figure 4.7. Size frequency of parrotfishes in EFGB and WFGB one-hectare study sites in 2019.

Invasive and Non-Native Species

Lionfish, an invasive species native to the Indo-Pacific, were first observed by scuba divers in FGBNMS in 2011 and in one-hectare study site surveys in 2013; however, 2019 is the first year since 2013 that lionfish have not been observed in surveys (Fig 4.8). Lionfish were still opportunistically observed by divers during long-term monitoring field work and during lionfish removal cruises held June 10 to 13, 2019 and August 26 to 29, 2019. Lower densities at EFGB and WFGB correlated with lionfish density declines in the northern Gulf of Mexico, which are potentially associated with the emergence of an ulcerative skin disease in late 2017 and 2018 (Harris et al. 2020).

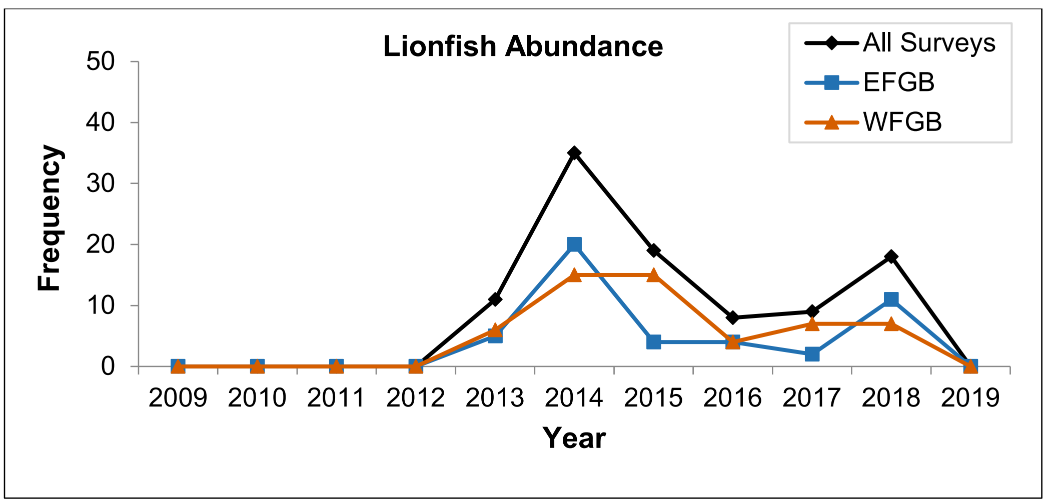


Figure 4.8. Lionfish abundance in EFGB and WFGB one-hectare study sites from 2013 to 2019.

For the second consecutive year, regal demoiselle, a non-native species from the Indo-Pacific region, were observed in surveys at both banks. Mean density for all surveys was 0.72 ± 0.42 and mean biomass for all surveys was 1.21 ± 0.57. No significant differences were detected between one-hectare study sites in 2019.

Fish Surveys Long-Term Trends

Since 2002, mean fish density ranged from 52.70 to 564.68 individuals/100 m² at EFGB one-hectare study sites, and 64.80 to 471.87 individuals/100 m² at WFGB one-hectare study sites (Figure 4.8). Fish community density was compared among years and one-hectare study sites when complete survey data were available (2011 to 2019). PERMANOVA analysis revealed significant differences between one-hectare study sites and among years (Table 4.7), demonstrating fish community, based on density, was highly variable between year and location from 2011 to 2019 (Figure 4.8). The observed dissimilarity in community based on density between one-hectare study sites from 2011 to 2019 was mainly attributable to bonnetmouth (11.67%) and brown chromis (8.42%).

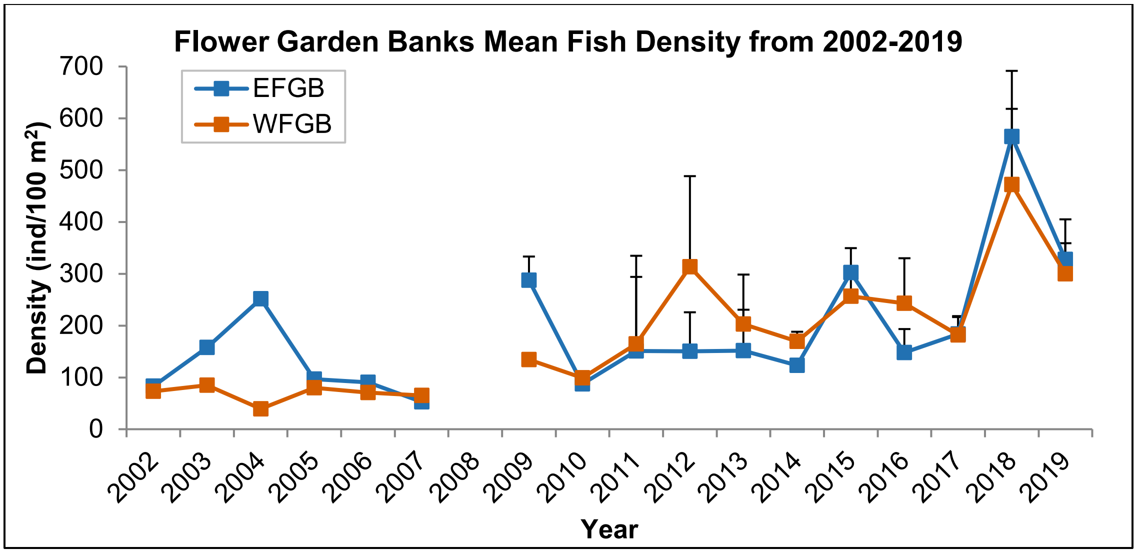


Figure 4.8. Mean fish density (individuals/100 m^2) + SE in EFGB and WFGB one-hectare study sites from 2002 to 2019. No data were collected in 2008 and SE was not available before 2009. Source: 2002 to 2008, PBS&J (Precht et al. 2006; Zimmer et al. 2010); 2009 to 2019, FGBNMS (Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020)

Table 4.7. PERMANOVA results comparing mean fish density in EFGB and WFGB one-hectare study sites and among years from 2011 to 2019. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	346	1	2.47	**0.046**
Res (Bank)	61519	439		
Total (Bank)	61865	440		
Year	12835	8	14.14	**0.001**
Res (Year)	49030	432		
Total (Year)	61865	440		

Community biomass data, first collected in 2006, was highly variable in the one-hectare study sites and ranged from 4,547.24 to 60,160.96 g/100 m² in EFGB surveys and 2,458.47 to 27,226.00 g/100 m² in WFGB surveys from 2006 to 2019 (Figure 4.9). PERMANOVA analysis revealed significant differences between one-hectare study sites and among years (Table 4.8). The observed dissimilarity in community based on biomass between one-hectare study sites from 2011 to 2019 was mainly attributable to great barracuda (10.98%) and Atlantic creolefish

(7.90%). The spike in biomass at EFGB in 2018 was attributable to greater local abundance of great barracuda and horse-eye jack (Johnston et al. 2020).

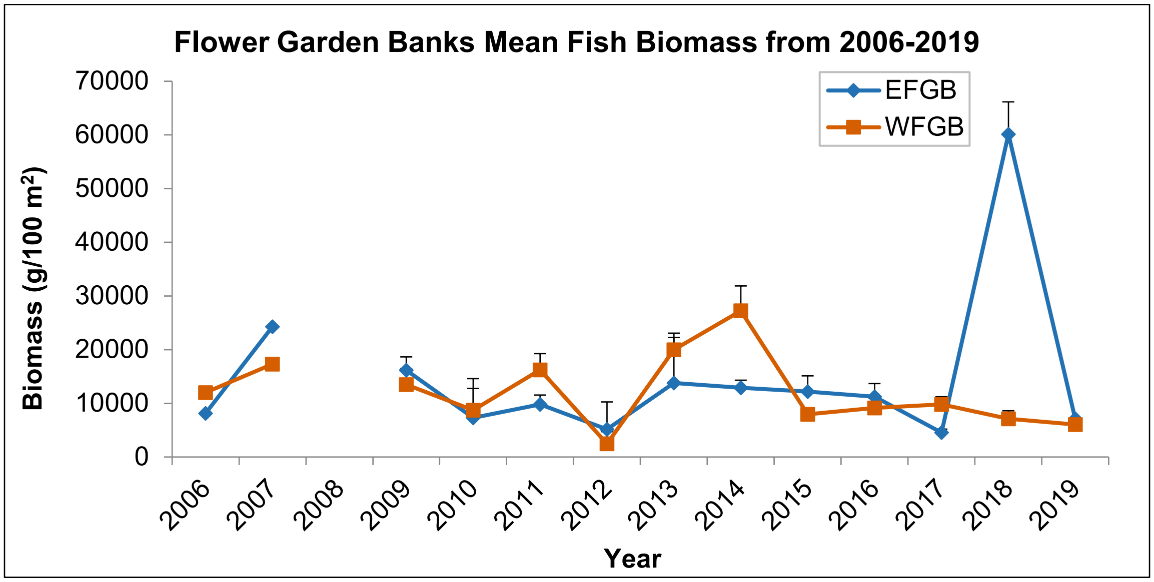


Figure 4.9. Mean fish biomass (g/100 m^2) + SE in EFGB and WFGB one-hectare study sites from 2006 to 2019. No data were collected in 2008 and SE was not available before 2009. Source: 2002 to 2008, PBS&J (Precht et al. 2006; Zimmer et al. 2010); 2009 to 2019, FGBNMS (Johnston et al. 2013, 2015, 2017a, 2017b, 2018a, 2020)

Table 4.8. PERMANOVA results comparing mean fish biomass in EFGB and WFGB one-hectare study sites and among years from 2011 to 2019. Bold text denotes significant values.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	8573	1	2.78	**0.006**
Res (Bank)	135280	439		
Total (Bank)	136130	440		
Year	422140	8	24.27	**0.001**
Res (Year)	939210	432		
Total (Year)	136130	440		

Additional analyses were conducted to investigate trends in grouper and snapper density at EFGB and WFGB one-hectare study sites over time (when complete survey data were available, 2011 to 2019). The most common grouper species at both EFGB and WFGB one-hectare study sites were graysby and yellowmouth grouper. Tiger grouper, scamp, coney, red hind, and rock hind were denser in EFGB surveys, and black grouper were denser in WFGB surveys (Figure 4.10).

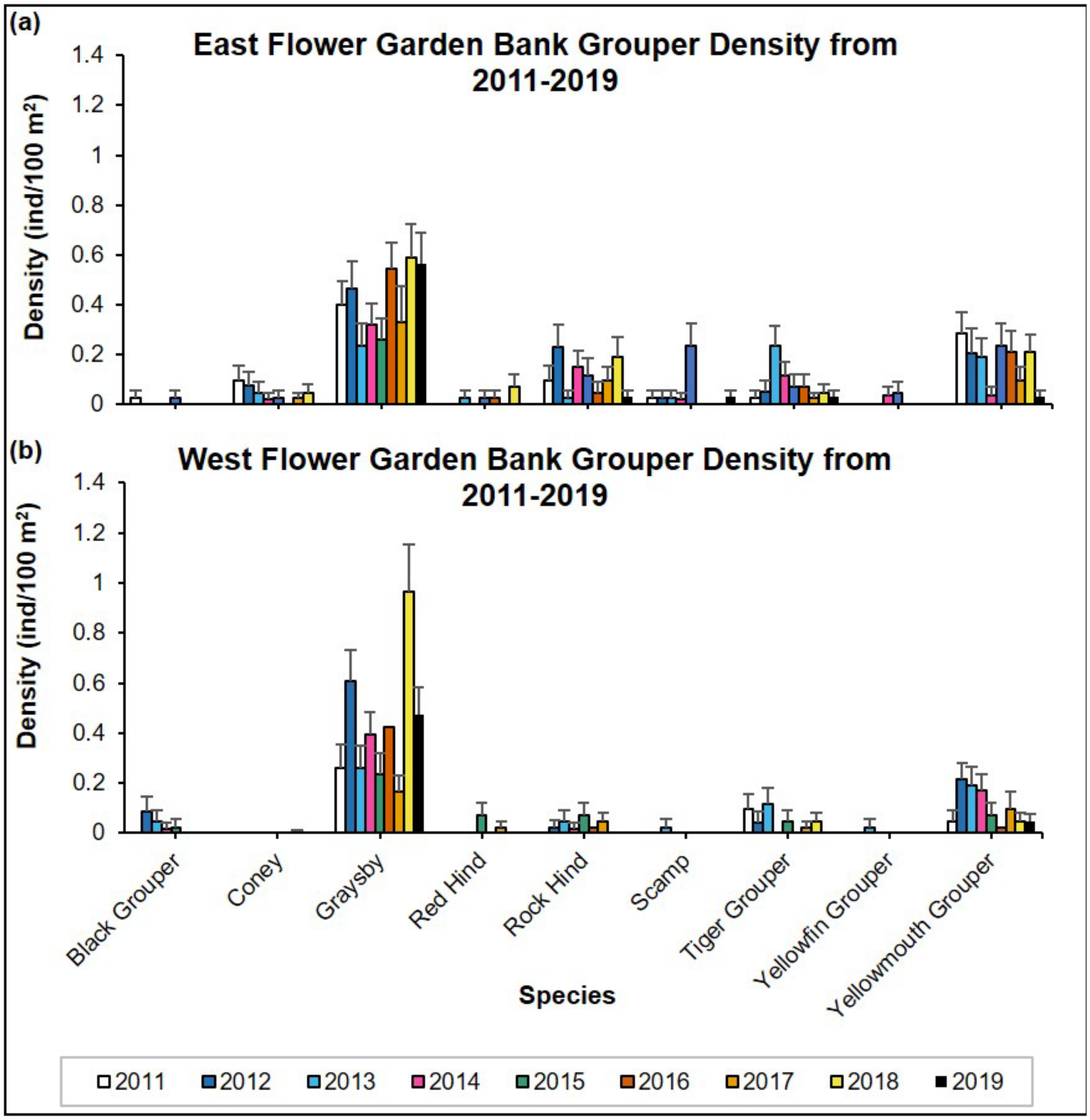


Figure 4.10. Mean density (individuals/100 m^2) + SE of grouper species within (a) EFGB and (b) WFGB one-hectare study site surveys from 2011 to 2019. Source: FGBNMS (Johnston et al. 2015, 2017a, 2017b, 2018a, 2020)

Grouper community density was compared among years and one-hectare study sites from 2011 to 2019. PERMANOVA analysis revealed that grouper density was significantly higher in EFGB surveys than in WFGB surveys, and also varied among years (Table 4.9). The observed dissimilarity between one-hectare study sites from 2011 to 2019 was mainly attributable to graysby (47.13%) and yellowmouth grouper (20.77%).

Table 4.9. PERMANOVA results comparing mean grouper density within EFGB and WFGB one-hectare study sites from 2011 to 2019. Bold text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	12	8	2.87	**0.001**
Bank	3	1	5.09	**0.002**
Year*bank	6	8	1.40	0.090
Res	220	423		
Total	240	440		

From 2011 to 2019, dog snapper and gray snapper were denser in WFGB surveys than EFGB surveys (Figure 4.11). PERMANOVA analysis revealed that snapper density was significantly higher at the WFGB one-hectare study site than the EFGB one-hectare study site (Table 4.10). The observed dissimilarity was mainly attributable to the greater abundance of dog snapper at WFGB (60.73%).

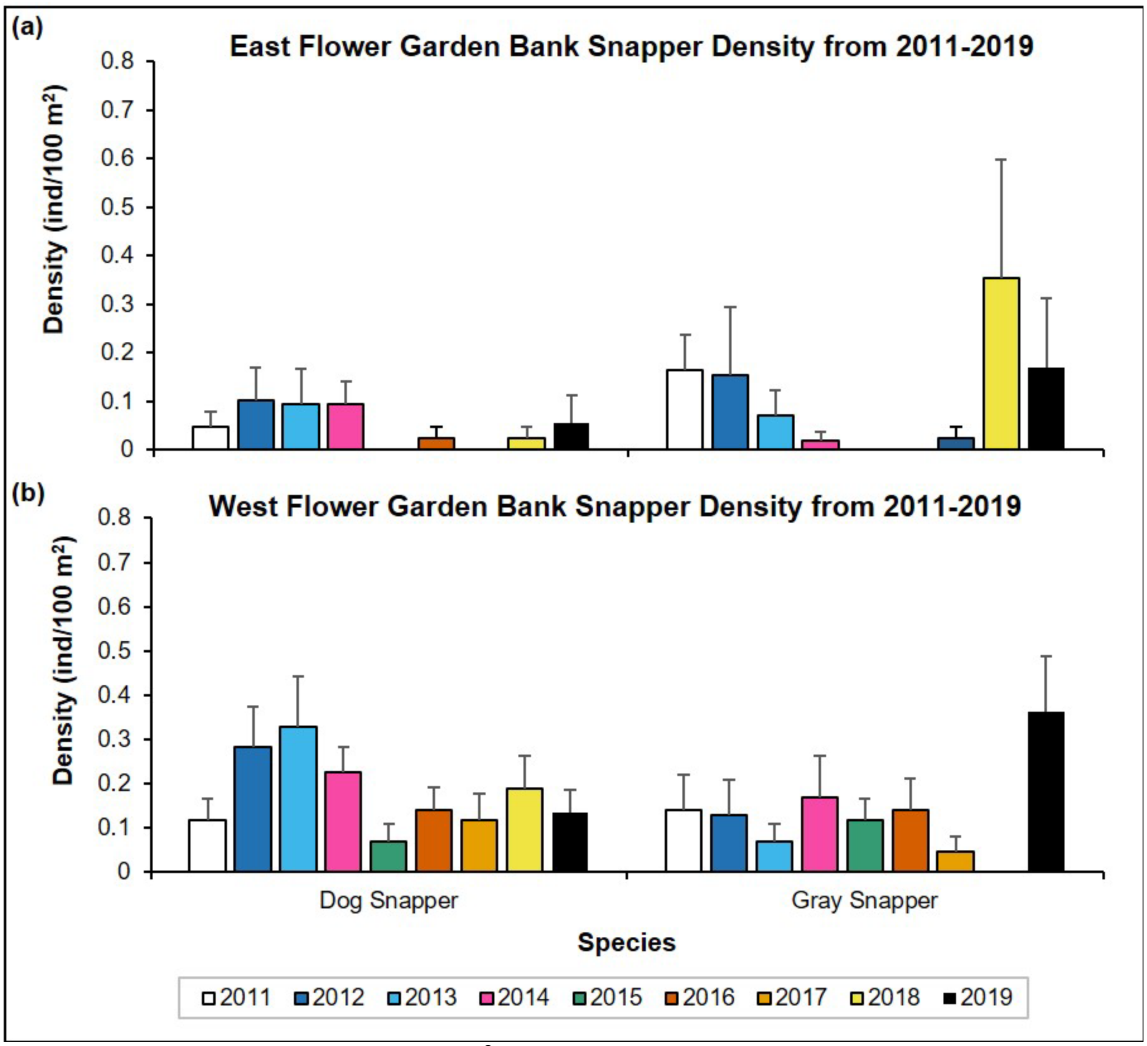


Figure 4.11. Mean density (individuals/100 m^2) + SE of snapper species within (a) EFGB and (b) WFGB one-hectare study sites from 2011 to 2019. Source: FGBNMS (Johnston et al. 2015, 2017a, 2017b, 2018a, 2020)

Table 4.10. PERMANOVA results comparing mean snapper density within EFGB and WFGB one-hectare study sites from 2011 to 2018. Bold text denotes significant values.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	3	8	1.65	0.053
Bank	3	1	15.36	**0.001**
Year*bank	2	8	1.06	0.368
Res	83	423		
Total	90	440		

Discussion

Fish communities are indicators of ecosystem health (Sale 1991; Knowlton and Jackson 2008; Jackson et al. 2014) and therefore an important component of long-term monitoring programs. Monitoring fish communities over time is valuable in detecting changes from normal variations that exist within the community. Historically, the fish communities at EFGB and WFGB have been considered low in species diversity but high in biomass (Zimmer et al. 2010). The fish assemblages of EFGB and WFGB occur near the northern latitudinal limit of coral reefs in the Gulf of Mexico, are remote from other tropical reef communities, and possess somewhat different fish assemblages than reef systems in the Caribbean, e.g., the limited presence of lutjanids (snappers) and haemulids (grunts) (Rooker et al. 1997; Precht et al. 2006; Johnston et al. 2017a). Approximately 150 reef fish species have been documented on the EFGB and WFGB reef caps (Pattengill 1998; Pattengill-Semmens and Semmens 1998).

EFGB and WFGB have lower overall abundance of herbivorous fishes than other Caribbean reefs (Dennis and Bright 1988; Bauer et al. 2015a; Bauer et al. 2015b; Bauer et al. 2015c; Caldow et al. 2015; Clark et al. 2015a, 2015b). Historically, low macroalgae cover was reported in annual monitoring surveys (Gittings et al. 1992), while recent data suggest a significant increase in mean macroalgae cover over time (Johnston et al. 2018a). During the 2019 study period, the herbivore guild possessed the second greatest mean biomass, contributing to 23% of the total biomass within one-hectare study site surveys. Herbivore biomass was also greater at EFGB, where macroalgae percent cover was higher in 2019. Within the herbivore guild, 33% of the total biomass was attributed to black durgon. Piscivores had the greatest mean biomass, with approximately 53% of the total biomass within one-hectare study sites. Within the piscivore guild, great barracuda contributed to over 40% of the total biomass, followed by horse-eye jack (37%). It is unknown how the presence of the research vessel might affect estimates of abundance and biomass for species like great barracuda, which often congregate below the R/V *Manta*. On one hand, the vessel concentrates the fish in an area directly over the one-hectare study sites, potentially inflating estimates; however, the fish tend to remain near the surface, outside survey sites where they might otherwise be if not for the presence of the vessel, thus decreasing estimates.

Abundance-biomass curves have historically been used to ascertain community health on shallow-water coral reefs; a community dominated by few large species is considered “healthy” and a community dominated by many small species is considered “impacted” (DeMartini et al. 2008; SOKI Wiki 2014). At EFGB and WFGB one-hectare study sites, results indicated that fish communities were evenly distributed (w-values close to 0), and the dominance plots for surveys were representative of a healthy population.

Bonnetmouth, followed by mackerel scad, had the highest relative abundance in EFGB surveys in 2019. In WFGB 2019 surveys, bonnetmouth also had the highest relative abundance. These species were observed as large schools of 100 to 2,000 individuals within the water column (Figure 4.12). It should be noted that mackerel scad and bonnetmouth are ephemeral species, but large transient schools have been documented in surveys at both banks from 2016 to present (Johnston et al. 2017b, 2018a, 2020).

Figure 4.12. School of bonnetmouth at WFGB. Photo: Michelle Johnston/NOAA

Commercially and recreationally important grouper and snapper density was low (<1 individual/100 m^2) at EFGB and WFGB one-hectare study sites in 2019. The grouper and snapper species observed consisted of both juvenile and mature individuals. It should be noted that typical recruitment/nursery habitat for snappers (mangroves and seagrasses) are not present at EFGB and WFGB, and the mechanism for recruitment of this family to the area is not well understood (Mumby et al. 2004; Clark et al. 2014). Due to the biogeographic isolation of EFGB and WFGB, the fish assemblage is thought to have a high rate of self-recruitment, as planktonic larval duration can limit larval supply and dispersal from other reefs in the southern Gulf of Mexico to EFGB and WFGB; however, the degree to which larval supplies from other reefs in the region are transported to FGBNMS is also dependent upon oceanographic conditions (i.e., Loop Current and associated eddies) (Wetmore et al. 2020).

Parrotfishes are important herbivores on coral reefs because they are effective algal grazers (Jackson et al. 2014). Parrotfish have been identified as key reef species, and their abundance and biomass are positively correlated with coral cover (Jackson et al. 2014). The mean biomass of parrotfish within study site surveys is considered low, but comparable to other Caribbean reefs (Jackson et al. 2014) (Table 4.11). However, low parrotfish biomass can be associated with high fishing pressure and low coral cover, neither of which are documented at EFGB or WFGB. Given the abundance of food for parrotfish at EFGB and WFGB, their low abundance is perplexing.

Table 4.11. Mean biomass (g/100 m^2) for parrotfish at EFGB, WFGB, and other Caribbean reefs. All data, with the exception of EFGB and WFGB data, are from AGRRA (2012).

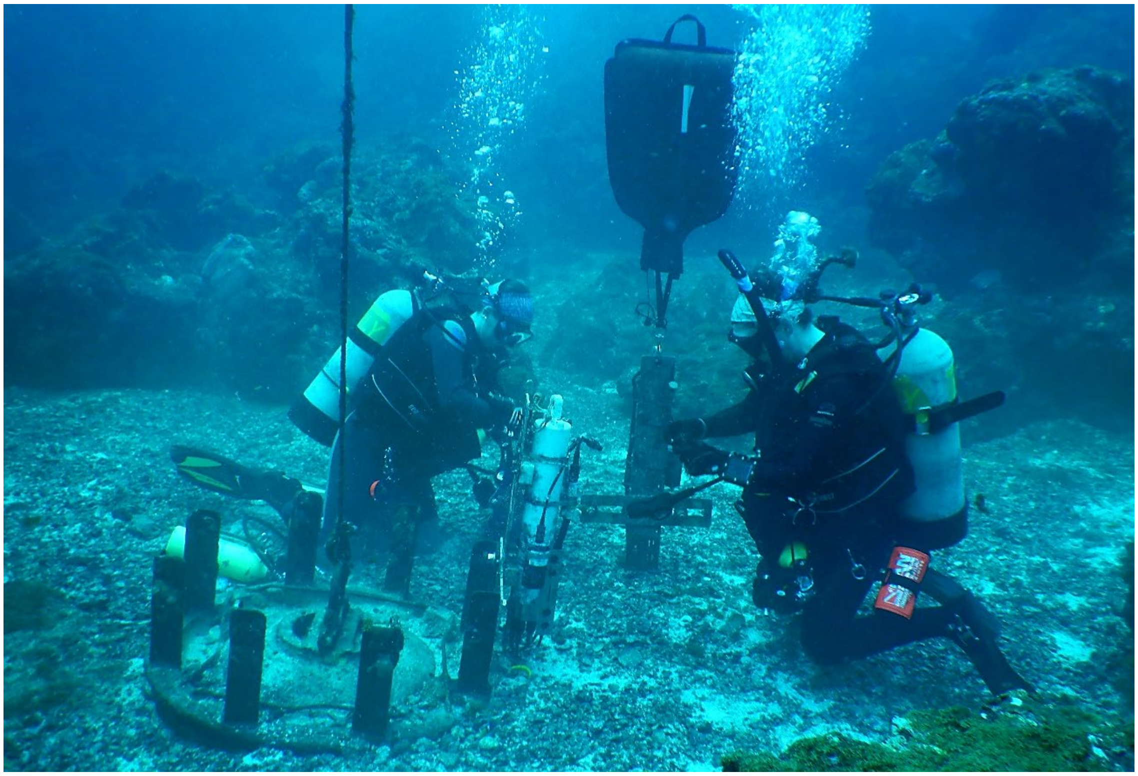
Location	Biomass (g/100 m2)
Mexico	1,710
Belize	1,200
East and West Flower Garden Banks one-hectare study site surveys combined (this report)	740
Guatemala	670
Honduras	440

Lionfish have been observed by divers consistently on the reefs since 2011, but they were not observed in 2019 surveys. Since their first observation, numbers rapidly increased through 2014, declined after 2015 (Johnston et al. 2016b), rose again in 2018, and sharply declined in 2019. It is believed that lionfish density declines in the northern Gulf of Mexico may be correlated with the emergence of an ulcerative skin disease in late 2017 and 2018, thus reducing recruitment in the region (Harris et al. 2020). Lionfish with skin ulcers were removed from FGBNMS in warm summer months of 2018 and 2019, consistent with information provided in Harris et al. (2020). Another hypothesis is that low density in 2019 may be the result of high numbers of lionfish removed from FGBNMS in June and August 2018 (364 and 771, respectively) during Lionfish Invitational removal cruises on the M/V *Fling*.

Lionfish are commonly seen during crepuscular feeding periods at dawn and dusk. Though fish surveys are conducted throughout the day, the fact that most surveys are not conducted when lionfish are most active may reduce the accuracy of estimates of lionfish densities. However, mean lionfish densities at EFGB and WFGB (approximately 4–40 lionfish ha^{-1}) (Johnston et al. 2016b) have yet to reach levels recorded elsewhere in the southeast U.S. and Caribbean region, such as North Carolina (150 lionfish ha^{-1}) (Morris and Whitfield 2009) and the Bahamas (100–390 lionfish ha^{-1}) (Green and Côté 2009; Darling et al. 2011), as well as on artificial reefs in the northern Gulf of Mexico (10–100 lionfish ha^{-1}) (Dahl and Patterson 2014). Since 2015, permitted lionfish removal cruises during summer months on the recreational dive vessel M/V *Fling* have been conducted to help suppress predation on native fish; however, divers are limited to the upper portion of the reef crest (<40 m) and focus around the mooring buoys (Green et al. 2014; Johnston et al. 2016b). Removals do not take place within the one-hectare study sites, ensuring sighting frequency, density, and biomass data are not directly affected. However, lionfish removals at nearby moorings are likely to result in lower abundances within the one-hectare study sites.

The regal demoiselle, a non-native species from the Indo-Pacific region, was observed in one-hectare study site surveys in 2019 at EFGB and WFGB for the second consecutive year. The suspected mode of introduction of this species was the inter-ocean transfer of oil rigs (Robertson et al. 2018). This species could compete with and displace native reef fish such as the brown chromis (Robertson et al. 2016). Sightings from EFGB and WFGB fish surveys were reported to the United States Geological Survey (USGS) invasive species sightings database, and FGBNMS will continue to monitor this species.

Chapter 5: Water Quality



NOAA divers exchange water quality instruments at EFGB. Photo: Jimmy MacMillan/CPC

Introduction

Several water quality parameters were continuously or periodically recorded at EFGB and WFGB. At a minimum, salinity, turbidity, and temperature were recorded every hour by data loggers installed in or near the one-hectare study sites at depths of approximately 24 m. Additionally, temperature loggers collected hourly readings at repetitive photostations at depths of 24 m, 30 m, and 40 m at each bank.

Water samples were collected quarterly throughout the year at three different depths within the water column and analyzed by an Environmental Protection Agency (EPA)-certified laboratory for select nutrient levels. Water column profiles were also acquired in conjunction with quarterly water sample collections. This chapter presents data from moored water quality instruments, water column profiles, and water samples collected in 2019.

Methods

Water Quality Field Methods

Temperature and Salinity Loggers

The primary instrument used at each bank for recording temperature, salinity, and turbidity was a Sea-Bird® Electronics 16plus V2 CTD (conductivity, temperature, and depth) (SBE 16plus) equipped with a WET Labs ECO NTUS turbidity meter at a depth of 23 m at EFGB and a depth of 27 m at WFGB. Loggers were secured to mounting anchors and located in sand flats at each bank (see Chapter 1, Figures 1.3 and 1.4). These instruments recorded temperature, salinity, and turbidity on an hourly basis. Instruments were exchanged by divers for downloading and maintenance in February, May, August, and November 2019. They were immediately exchanged with an identical instrument to avoid any interruptions in data collection. Data were then downloaded and reviewed, sensors were cleaned and confirmed to be operable, and battery duration was calculated. Maintenance, as well as factory service and calibration of each instrument, was performed according to an annual maintenance schedule.

Onset® Computer Corporation HOBO® Pro v2 U22-001 (HOBO) thermograph loggers were used to record temperature on an hourly basis. These loggers provided a highly reliable temperature backup for the primary SBE 16plus logging instruments located at the 23 m and 27 m stations at EFGB and WFGB. HOBO loggers were also deployed at 30 m and 40 m stations at EFGB and WFGB to record temperature hourly at deeper depths. The loggers were downloaded, maintained, and replaced in February, May, August, and December 2019. The instruments were attached directly to either the primary SBE 16plus instrument at the 24 m station or to permanent repetitive photostation markers at approximate 30 m and 40 m depths. Prior to re-installation, data were removed from the instruments and sufficient battery levels were verified for redeployment.

Water Column Profiles

Water column profiles were acquired in February and May of 2019 with a Sea-Bird® Electronics 19plus V2 CTD that recorded temperature, salinity, pH, turbidity, fluorescence, and dissolved oxygen (DO) every ¼ second to distinguish differences between three main depth gradients: the reef cap (~20 m), mid-water column (~10 m), and the surface (~1 m). A different carousel was used for the fourth quarterly water collections and profiles in November. The carousel package was a Sea-Bird® 55 Frame Eco water sampler equipped with six 4-liter Niskin bottles, a Sea-Bird® Electronics 19plus V2 CTD (last serviced 2019 February) capable of recording conductivity, depth, salinity, and temperature, and a Wet Labs C-Star Transmissometer measuring beam attenuation. The profiler lacked pH, DO, fluorescence, and turbidity data acquisition capabilities. Data were recorded following an initial three-minute soaking period after deployment and the resulting profile data were processed to include only downcast data. The CTD was lowered and returned to the surface at a rate <1 m/second. The water column profiles were attained on February 28, May 16, and November 19, 2019. Third quarter water samples were collected on August 2, 2019; however, a water column profile was not obtained due to instrument service at that time.

Water Samples

In conjunction with water column profiles, water samples were collected using a sampling carousel equipped with a Sea-Bird® Electronics *19plus* V2 CTD and a rosette of twelve OceanTest® Corporation 2.5-liter Niskin bottles for collections on February 28 and May 16, 2019. During the sample collections on August 2, 2019, FGBNMS's carousel was used without the Sea-Bird® Electronics *19plus* V2 CTD due to maintenance requirements. On November 19, 2019 a Sea-Bird® 55 Frame Eco water sampler equipped with six 4-liter Niskin bottles and a

Sea-Bird® Electronics *19plus* V2 CTD was used. The carousel was attached to the R/V *Manta* through a scientific winch cable, thereby allowing the operator to activate the bottles for sample collection at specific depths. Samples were collected on February 28, May 16, August 2, and November 19, 2019. Two Niskin bottles collected water samples near the reef cap on the seafloor (~20 m depth), midwater (~10 m depth), and near the surface (~1 m depth) for subsequent transfer to laboratory collection bottles.

Water samples were analyzed for chlorophyll *a* (chl *a*) and nutrients including ammonia, nitrate, nitrite, soluble reactive phosphorous (ortho phosphate), and total Kjeldahl nitrogen (TKN) (Table 5.1). Water samples for chl *a* analyses were collected in 1000-ml glass containers with no preservatives. Samples for soluble reactive phosphorous were placed in 250-ml bottles without preservatives. Ammonia, nitrate, nitrite, and total nitrogen samples were collected in 1000-ml bottles with a sulfuric acid preservative. An additional blind duplicate water sample was taken at one of the sampling depths for each sampling period. Within minutes of sampling, labeled sample containers were stored on ice at 0°C and a chain of custody was initiated for processing at an EPA-certified laboratory. The samples were transported and delivered for analysis to A&B Laboratories in Houston, Texas within 24 hours of collection.

Table 5.1. Standard EPA methods used to analyze water samples collected at the FGB.

Parameter	Test Method	Detection Limit
Chlorophyll *a*	SM 10200H	0.003-mg/l
Ammonia	SM 4500NH3D	0.10–mg/l
Nitrate	SM 4500NO3E	0.04–mg/l
Nitrite	SM 4500NO2B	0.02–mg/l
Soluble reactive phosphorous	SM 4500 P-E	0.02–mg/l
Total Kjeldahl nitrogen (TKN)	SM 4500NH3D	0.50–mg/l

Water samples for ocean carbonate measurements, including pH, alkalinity, CO_2 partial pressure (pCO_2), $\Omega_{aragonite}$, and total dissolved CO_2 (DIC), were collected following methods provided by the Carbon Cycle Laboratory (CCL) at Texas A&M University-Corpus Christi (TAMU-CC). Samples were collected in ground neck borosilicate glass bottles. Bottles were filled using a 30-cm plastic tube connected to the filler valve of a Niskin bottle. Bottles were rinsed three times using the sample water, filled carefully to reduce bubble formation, and overflowed by at least 200 ml. A total of 100 µl of saturated $HgCl_2$ was added to each bottle, which was then capped and the stopper was sealed with Apiezon® grease and secured with a rubber band. The bottles were then inverted vigorously to ensure homogeneous distribution of $HgCl_2$ and secured at ambient temperature for shipment. Samples and CTD profile data were sent to CCL at TAMU-CC. Ocean carbonate samples were obtained on February 28, May 16, August 2, and November 19, 2019.

Water Quality Data Processing and Analysis

Temperature, salinity, and turbidity data recorded on SBE *16plus* instruments and temperature data recorded on HOBO loggers were downloaded and processed in February, May, August and November of 2019. QA/QC procedures included a review of all files to ensure data accuracy and instruments were serviced based on manufacturer recommendations. The 24 hourly readings obtained each day were averaged into a single daily value and recorded in duplicate databases.

Each calendar day was assigned a value in the database. Separate databases were maintained for each logger type as specified in the standard operating procedures.

Previous reports used hourly sea surface temperature (SST) and sea surface salinity (SSS) data downloaded from Buoy V and Buoy N of the Texas Automated Buoy System (TABS) database; however, these buoys were removed in late April 2019 and January 2017, respectively. Therefore, surface buoy readings were unavailable or absent for 2019 analyses. In lieu of *in situ* surface data, satellite-derived SST and SSS data for 2019 were downloaded from the NOAA Environmental Research Division Data Access Program (ERDDAP) data server for comparison to reef cap data. The SST dataset used was “GHRSST Level 4 MUR Global Foundation Sea Surface Temperature Analysis (v4.1)” and the SSS dataset used was “Sea Surface Salinity, Near Real Time, Miras SMOS 3-Day Mean (smosSSS3Scan3DayAggLoM), CoastWatch v6.62, 0.25°, 2010-present” (JPL MUR MEaSUREs Project 2015; Simons 2019; NOAA Coral Reef Watch 2020). Satellite-derived one-day mean SST data utilized for WFGB and EFGB in 2019 were available as a level-4 global 0.01-degree grid produced at the NASA Jet Propulsion Laboratory Physical Oceanography Distributed Active Archive Center under support by the NASA MEaSUREs program. Satellite derived SSS data utilized for WFGB and EFGB in 2019 were available as a level-3 gridded three-day mean dataset from MIRAS satellite observations over the global ocean.

SBE *16plus* instruments and HOBO loggers located on the reef cap were exchanged during designated water quality cruises in February, May, and November and combined with the WFGB long-term monitoring cruise in August 2019. The 30-m and 40-m HOBO loggers at EFGB were exchanged on July 30 during the long-term monitoring research cruise and the WFGB 30-m and 40-m HOBO loggers were exchanged on December 4, 2019; therefore, 30-m and 40-m data for EFGB were updated and available to July 30, 2019 and WFGB 30-m and 40-m data were updated and available to December 4, 2019. The EFGB 24-m backup HOBO logger revealed a data interruption from May 18 through August 1, 2019.

For seawater temperature, salinity, and turbidity data, EFGB and WFGB SBE *16plus* 2019 daily mean data were compared using a paired t-test in R version 2.13.2. Monotonic trends over the course of seawater temperature and salinity long-term datasets were detected using the Seasonal-Kendall trend test in a Microsoft Windows® DOS executable program developed by USGS for water resource data (Hipel and McLeod 1994; Helsel and Hirsch 2002; Helsel et al. 2006). The Seasonal-Kendall trend test performed the Mann-Kendall trend test for each month and evaluated changes among the same months from different years over time, accounting for serial correlation in repeating seasonal patterns.

Results of chlorophyll *a* and nutrient analyses were obtained on March 12, May 29, August 14, and December 3, 2019 from A&B Labs and compiled into an Excel table. Results of ocean carbonate analyses were compiled and received as an annual report from CCL at TAMU-CC.

Results

Temperature

Surface temperature at EFGB ranged from 19.49°C to 31.06°C; at 24 m, it ranged from 20.73°C to 30.21°C (Figure 5.1). According to data from the 23-m SBE *16plus*, reef cap temperatures at EFGB exceeded 30°C for six nonconsecutive days in 2019. Reef cap temperatures averaged 30.13°C from August 16–18 and surpassed 30°C again on September 3, 10, and 11 in 2019 (Figure 5.1). The 23-m backup HOBO logger registered temperatures beyond 30°C for sixteen days in 2019: August 16–18 (mean of 30.26°C); August 24–26 (mean of 30.0°C); August 31 (30.06°C); September 2–3 (mean of 30.09°C); and September 9–15 (mean of 30.11°C) (Figure 5.1).

At the 30-m and 40-m EFGB stations, slightly cooler temperatures compared to the surface and 24-m locations (particularly during the summer months) were recorded. Data analysis was performed on a partial dataset due to the fact that the EFGB 30-m and 40-m loggers were last exchanged in July 2019; therefore, data were available from January 1 through July 30, 2019 at the writing of this report. At 30 m, temperature ranged from 20.80°C to 28.62°C; at 40 m, temperature ranged from 20.75°C to 25.82°C (Figure 5.1). The average temperature difference between the 23-m SBE *16plus* reef cap location and 30-m location from January 1 through July 30, 2019 was -0.46°C and the greatest temperature difference was -3.02°C on June 7, 2019. The average temperature difference between the 23-m and 40-m locations between January and July 2019 was -1.29°C and the greatest difference in temperature was -5.73°C on July 27, 2019.

Surface temperature at WFGB ranged from 19.92°C to 30.81°C; at 27 m, it ranged from 20.41°C to 30.19°C (Figure 5.1). At 30 m, temperature ranged from 20.43°C to 30.20°C (Figure 5.1). At 40 m, temperature ranged from 20.48°C to 30.17°C (Figure 5.1). At WFGB, the average temperature difference between the 27-m and 30-m locations was -0.11°C and the greatest temperature difference was -1.43°C on July 5, 2019. The average temperature difference between the 27-m and 40-m locations was -0.96°C, and the greatest difference in temperature between the 27-m and 40-m stations was 5.56°C on August 10, 2019.

According to data from the 27 m SBE *16plus*, reef cap temperatures at WFGB exceeded 30°C for 10 nonconsecutive days in 2019. Reef cap temperatures ranged 30.02°C to 30.19°C from August 14–17, 2019, reached 30.16°C on September 2, and surpassed 30°C from September 9–13, 2019. The 27-m backup HOBO logger registered temperatures at or above 30°C for 20 days in 2019: August 14–17 (mean of 30.22°C); August 23 and 29 (daily means of 30.00 °C and 30.11 °C, respectively); and September 1–14 (mean of 30.14 °C). The WFGB 30-m station exceeded 30°C for 10 nonconsecutive days on August 14–17 and September 2, 7, and 9–12, 2019. The WFGB 40-m station exceeded 30°C for eight nonconsecutive days on August 16–17, September 7, and September 9–13, 2019.

When comparing 2019 reef cap daily mean seawater temperatures, a significant difference occurred between EFGB 23-m and WFGB 27-m SBE *16plus* reef cap temperatures (t-test, df=364, t=2.74, p=0.006) due to warmer temperatures at EFGB.

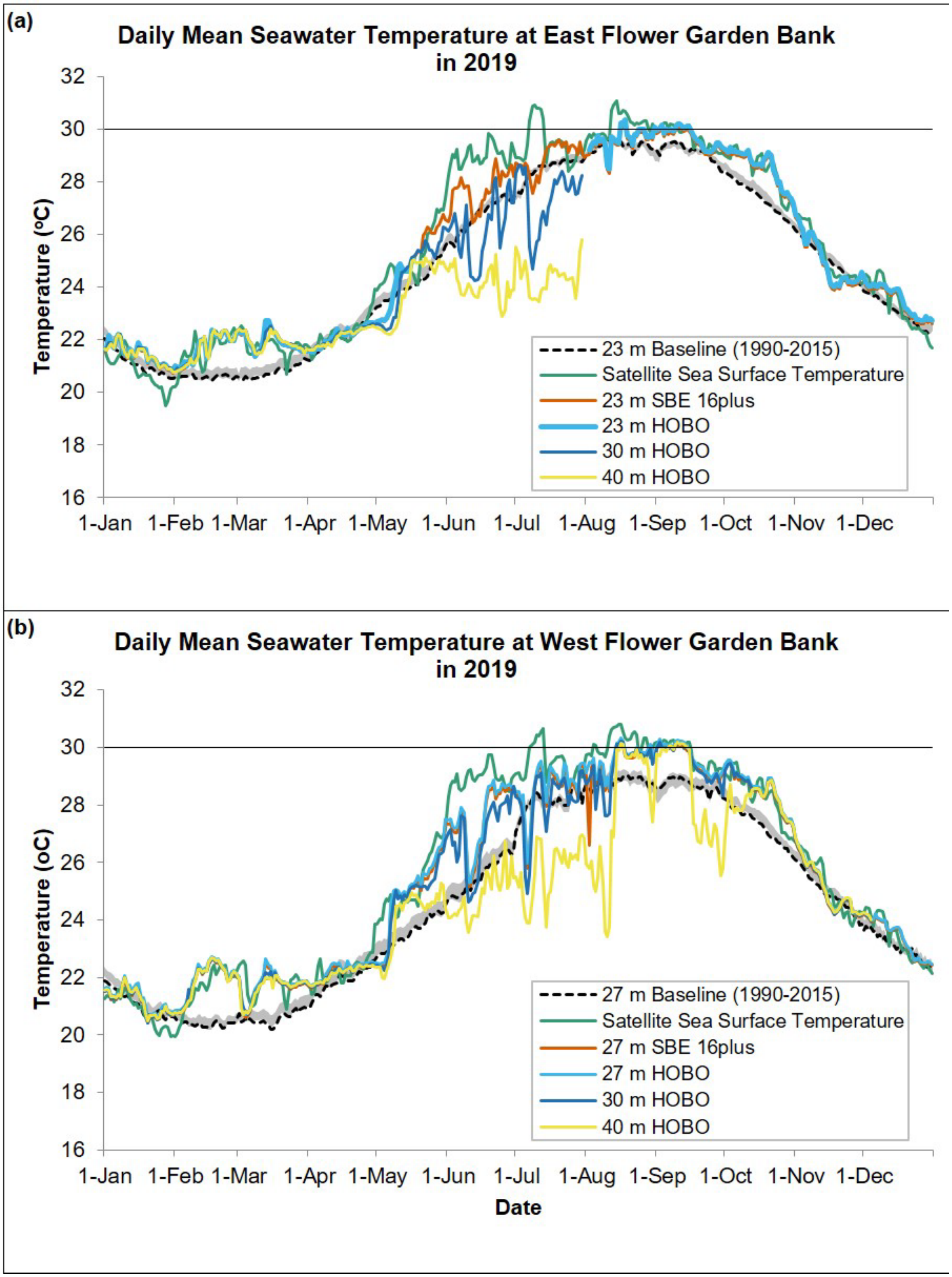


Figure 5.1. Daily mean seawater temperature (ºC) at (a) EFGB and (b) WFGB from various depths in 2019 and the 25-year daily mean water temperature baseline ± SE band in grey. The solid black line at 30ºC is a level known to trigger coral bleaching.

Seawater temperature data obtained from loggers at an approximate depth of 25 m from both banks combined have been collected since 1990. Though some data gaps occur due to equipment malfunction and changes in methods and/or instrumentation, long-term trends showed increasing temperature at EFGB and WFGB (Figure 5.2). The Seasonal-Kendall trend test on time-series daily mean seawater temperature data revealed significantly increasing monotonic trends from 1990 to 2019 at both EFGB and WFGB (τ=0.32, z=6.73, $p<0.001$ and τ=0.29, z=6.41, $p<0.001$, respectively) after adjusting for correlation among seasons (Figure 5.2).

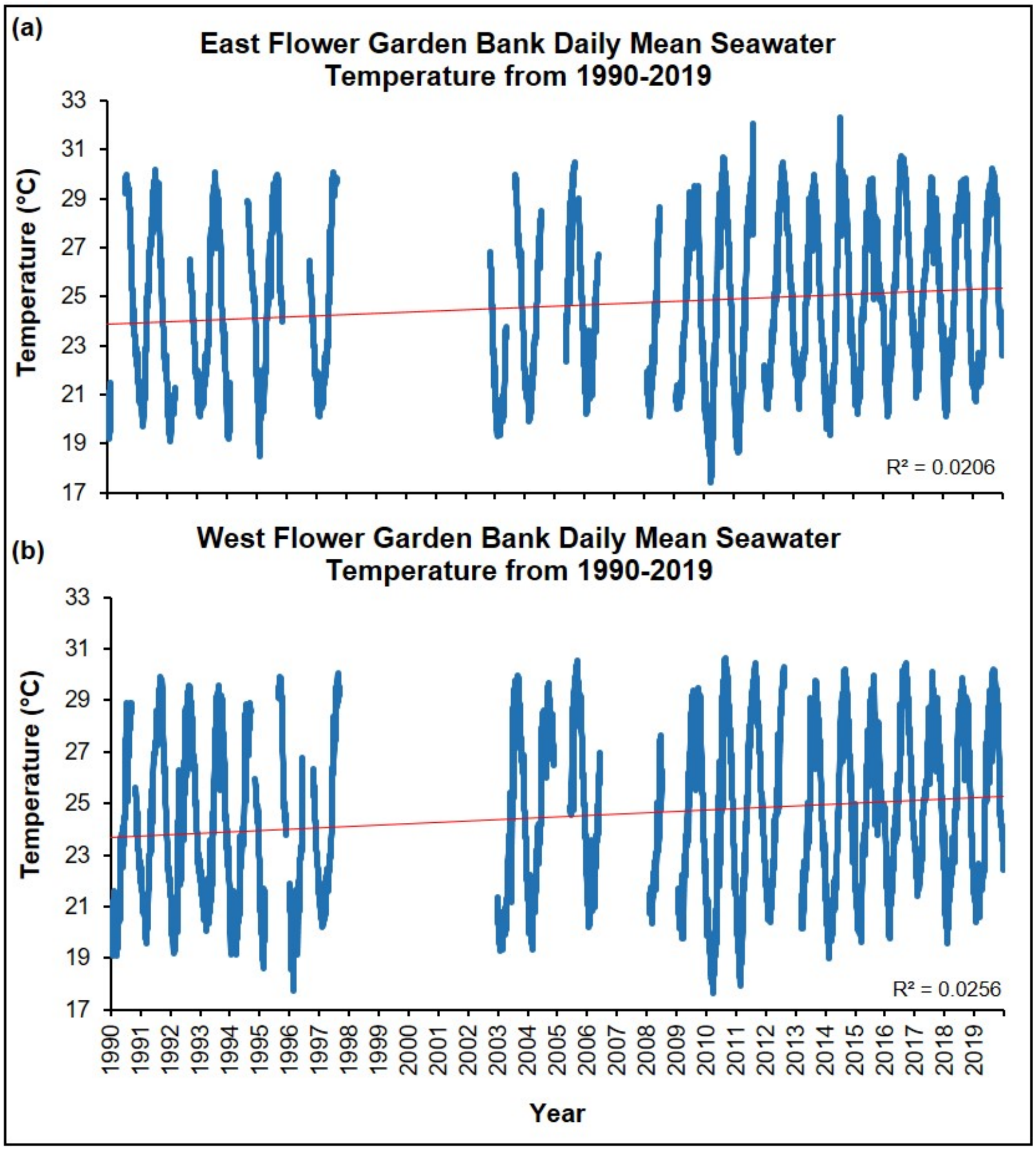


Figure 5.2. Daily mean seawater temperature (°C) demonstrating 12-month seasonal variation at (a) EFGB and (b) WFGB (approximate 25 m depth) from 1990 to 2019, as well as a significant increase over time (red trend line).

Salinity

Surface salinity at EFGB ranged from 28.99 to 37.94 psu; at 23 m, it ranged from 32.64 to 36.54 psu (Figure 5.3). At WFGB, surface salinity ranged from 29.30 to 38.01 psu; at 27 m, it ranged from 33.85 psu to 36.56 psu (Figure 5.3). There was a significant difference between EFGB and WFGB 2019 SBE *16plus* reef cap salinity daily means (t-test, df=364, t=-6.10, $p<0.002$), likely due to lower salinity levels at EFGB from late June to early August.

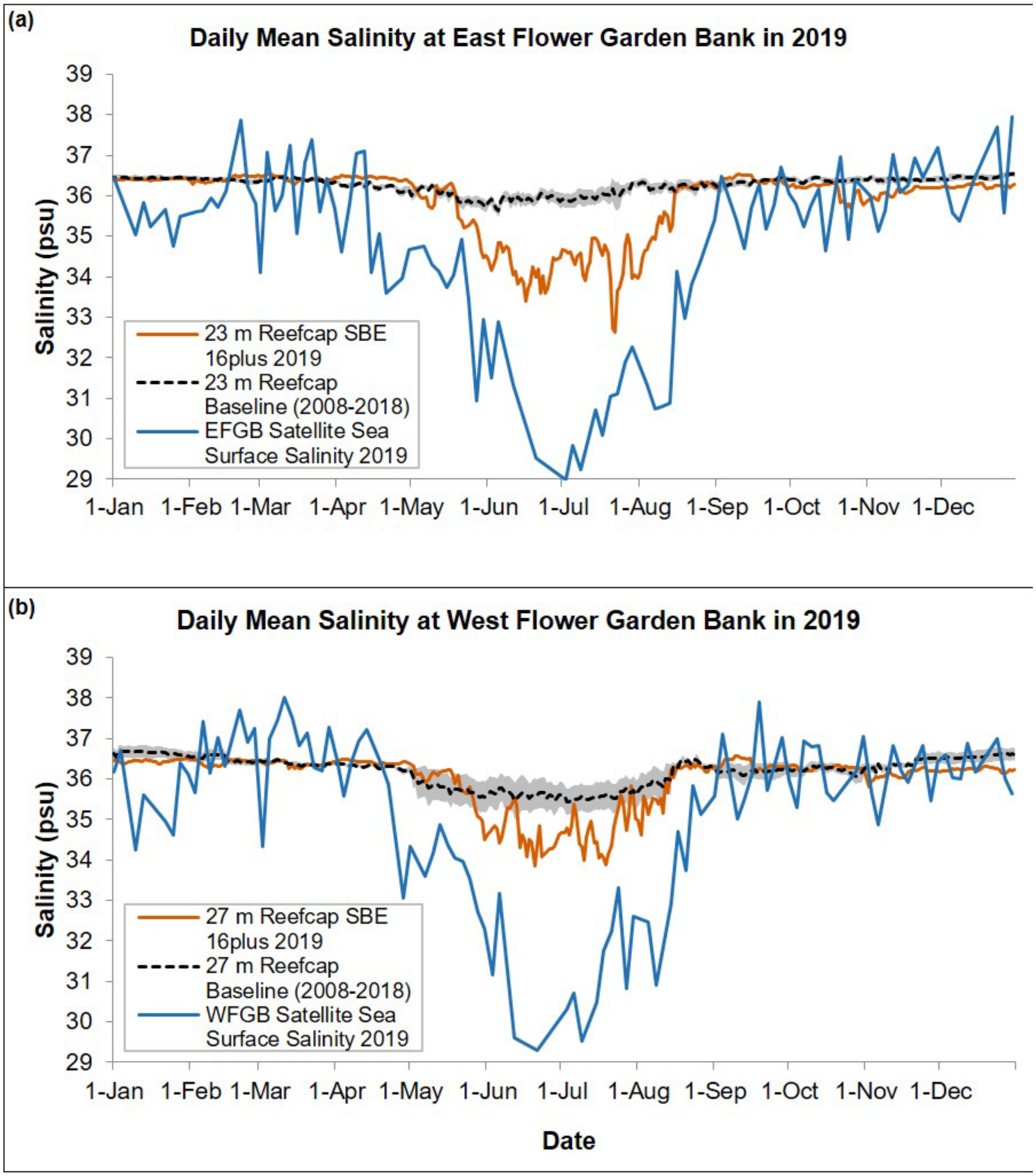


Figure 5.3. Daily mean salinity (psu) at the sea surface, SBE 16 plus reef cap station, and the reef cap station 10-year daily mean salinity baseline (2008–2018) ± SE band in grey at (a) EFGB and (b) WFGB in 2019.

Salinity data obtained from loggers at an approximate depth of 25 m from both banks combined have been collected throughout the monitoring program since 2008 with minimal disruptions in data acquisition (Figure 5.4). The Seasonal-Kendall trend test on time-series daily mean salinity data at EFGB resulted in a significantly decreasing monotonic trend from 2008 to 2019 (τ=-

0.22, z=-3.16, p=0.013) after adjusting for correlation among seasons (Figure 5.4). No significant trend was observed at WFGB.

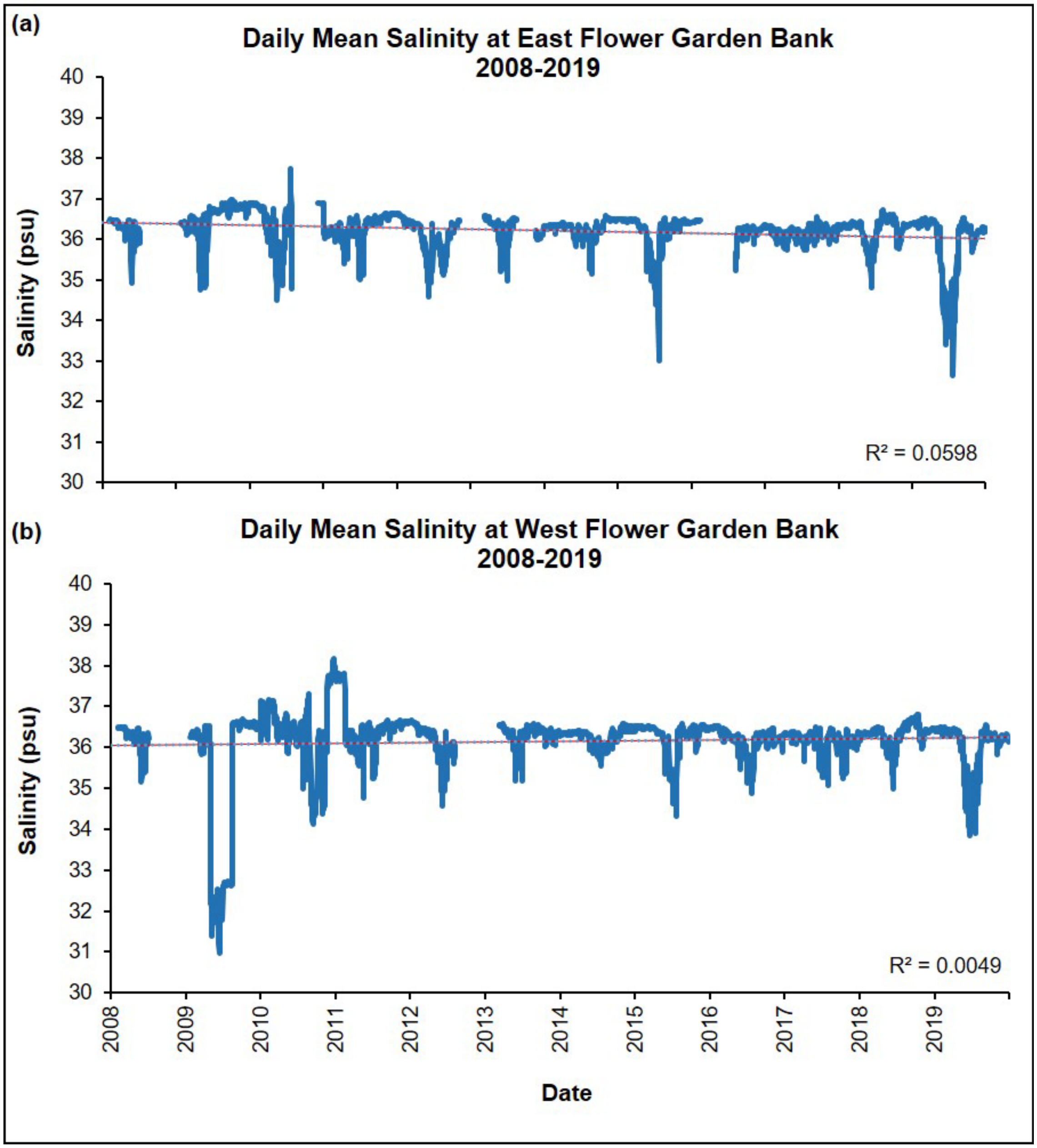


Figure 5.4. Daily mean salinity demonstrating 12-month seasonal variation at (a) EFGB (23 m) and (b) WFGB (27 m) from 2008 to 2019 (trend line in red).

Turbidity

Turbidity was added as a long-term monitoring data parameter in August 2016. Figure 5.5 shows turbidity data from EFGB (23 m) and WFGB (27 m) in 2019. The turbidity sensor at EFGB experienced a malfunction resulting in loss of data from May 24 through August 1, 2019; therefore, data from WFGB for this time period were removed for analysis to enable comparison between the two banks. A significant difference was observed between EFGB and WFGB (t-test, df=294 t=2.46, p=0.001). The 2019 turbidity values varied more (primarily during April and May) and were higher at EFGB than WFGB (0.15 versus 0.10 ntu, respectively).

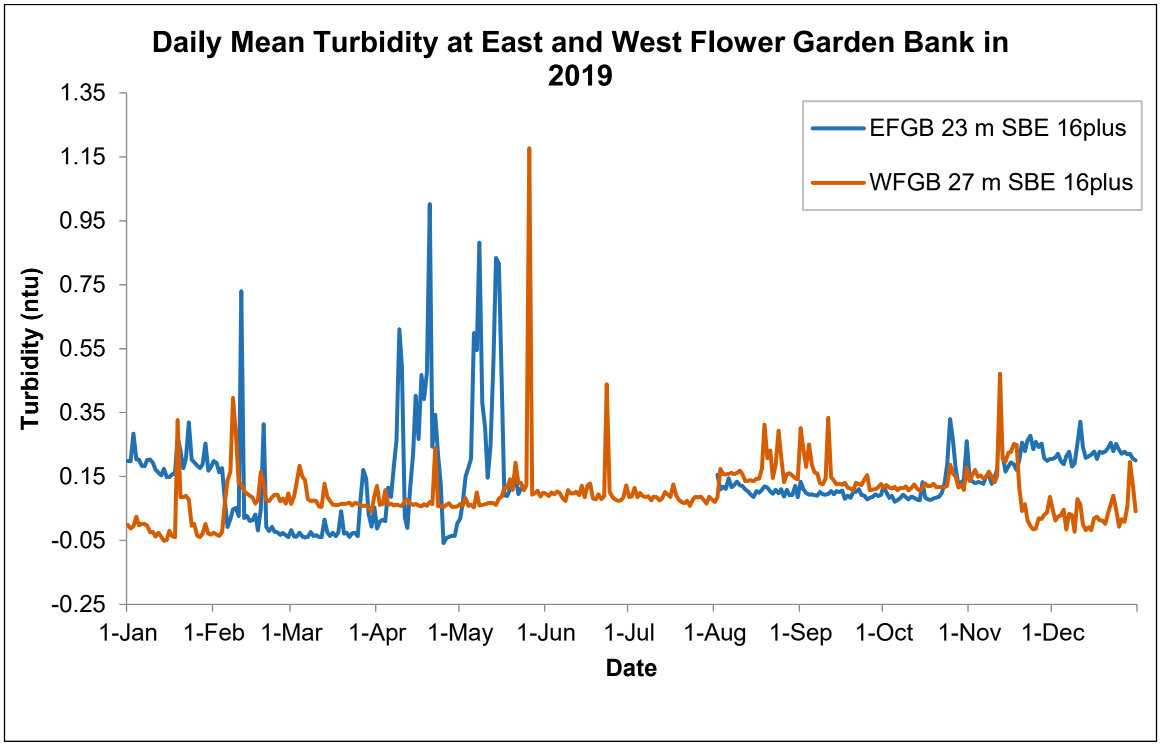


Figure 5.5. Daily mean turbidity (ntu) values in 2019 from EFGB (23 m) and WFGB (27 m).

Water Column Profiles

Water column temperatures at both banks were similar and fluctuated very little between the surface and the reef cap. No single profile varied more than 1°C from the surface to the reef cap (Figure 5.6 and 5.7). Salinity values between the two banks were similar, varying less than 0.09 psu on average over all three profiles. Salinity remained stable throughout the water column in February and November; however, the May profile displayed variability. DO values in February were similar at both banks; however, the May DO profile revealed considerable variability at both banks, shifting from 3.6 ml/L to 4.13 ml/L. The depth at which DO values stabilized during the May profile was different at EFGB and WFGB. Turbidity values were slightly higher at WFGB than EFGB in February and May. Fluorescence values at both EFGB and WFGB were higher during the February profile than during the May profile and fluorescence was generally higher at WFGB than EFGB. The EFGB profile taken on February 5, 2019 exhibits increased variation in fluorescence, peaking at a depth of 7.5 meters and decreasing with depth. According to the 2019 water column profile data, conditions such as turbidity and fluorescence appear more variable throughout the year at EFGB than at WFGB (Figure 5.6 and 5.7).

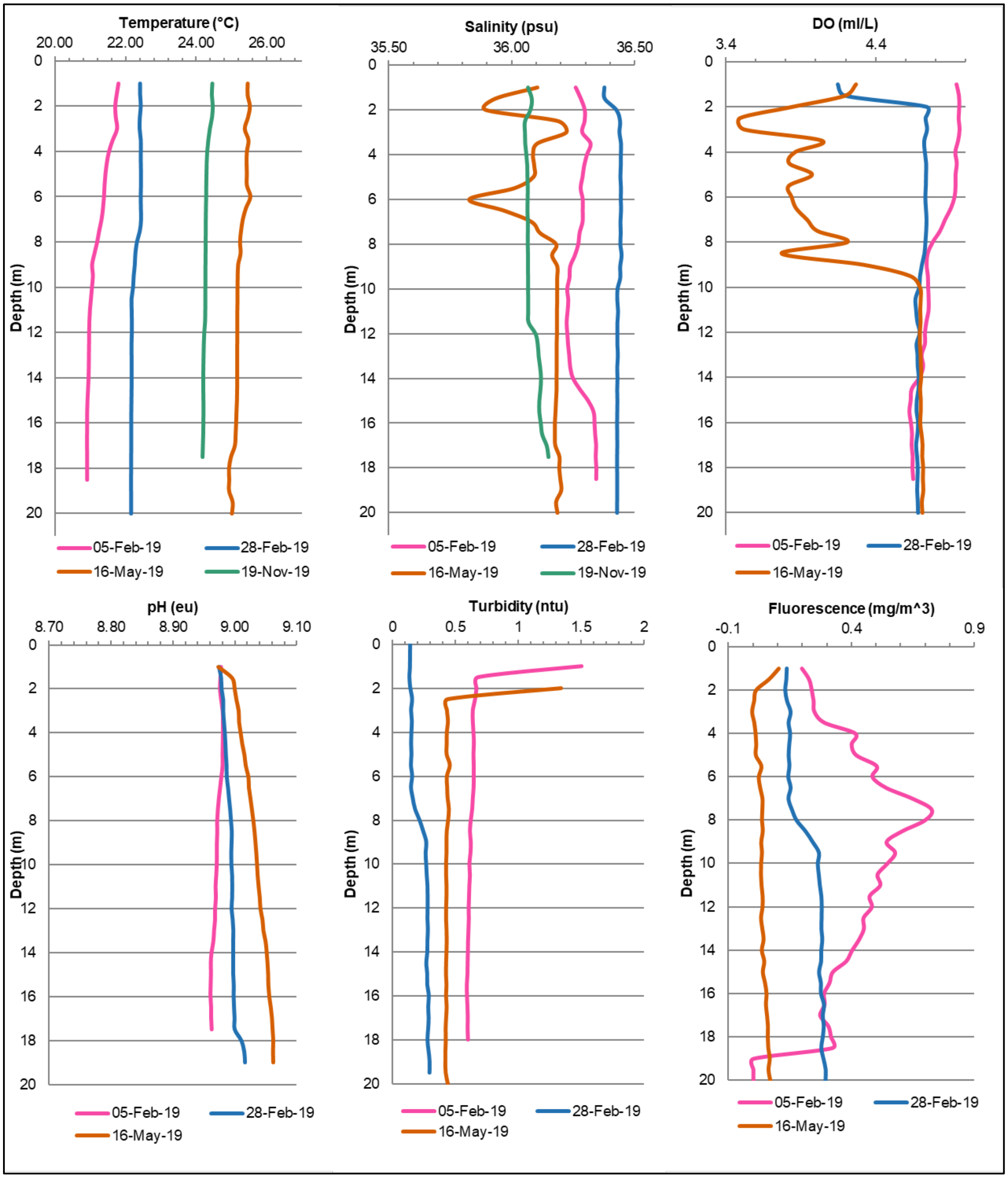


Figure 5.6. EFGB temperature, salinity, DO, pH, turbidity, and fluorescence water column profile data in February, May, and November 2019. The CTD used in November did not measure pH, fluorescence, turbidity, or DO.

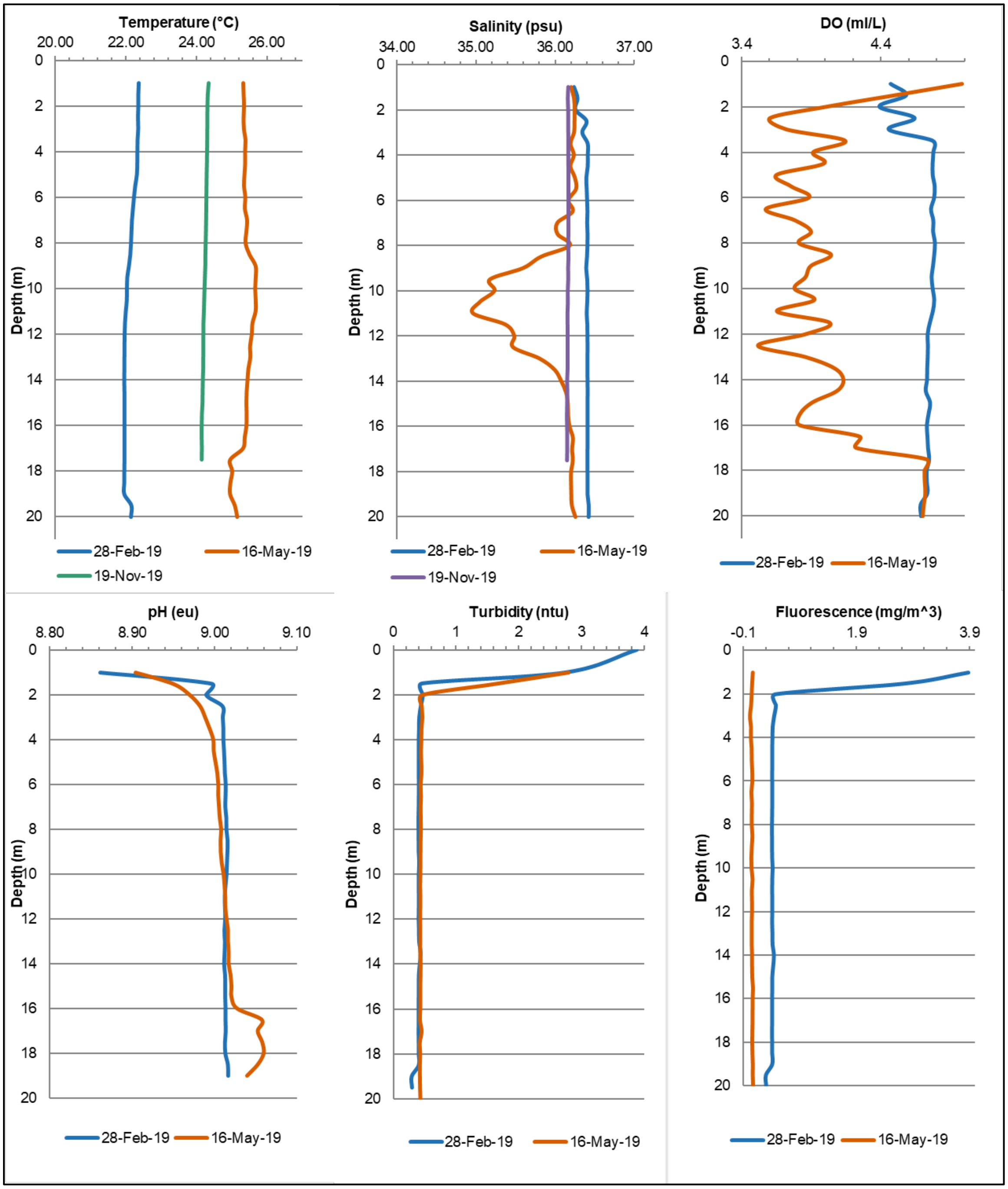


Figure 5.7. WFGB temperature, salinity, DO, pH, turbidity, and fluorescence water column profile data in February, May and November 2019. The CTD used in November did not measure pH, fluorescence, turbidity, or DO.

Water Samples

The first chl *a* and nutrient samples taken as part of the long-term monitoring program were in 2002 and since then, quarterly nutrient levels have typically been below detectable limits, with the exception of occasional ammonia and TKN detections prior to 2012 (Figures 5.8 and 5.9). The 2019 nutrient levels from each water column depth were below detectable limits in all samples.

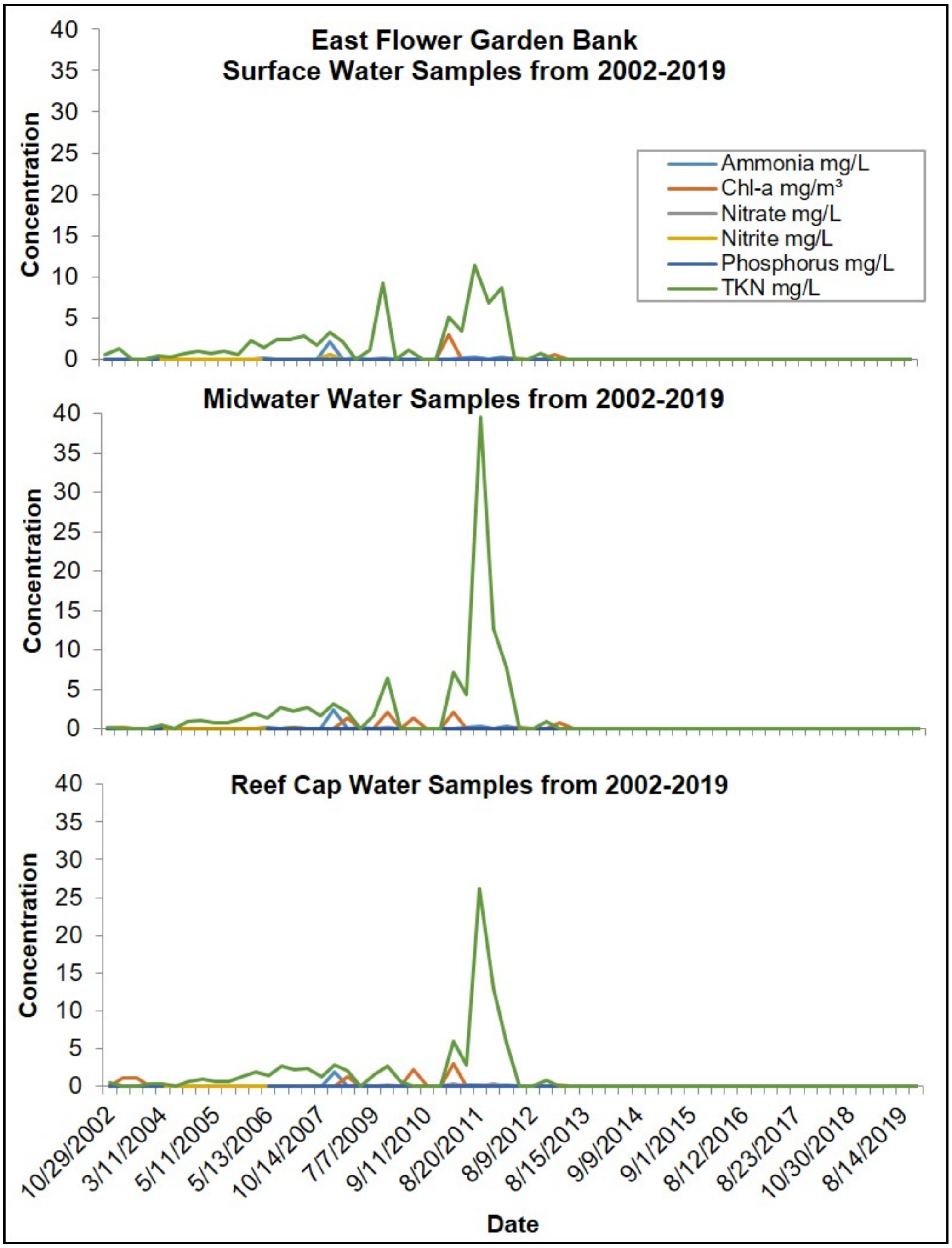


Figure 5.8. Nutrient concentrations from EFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2019.

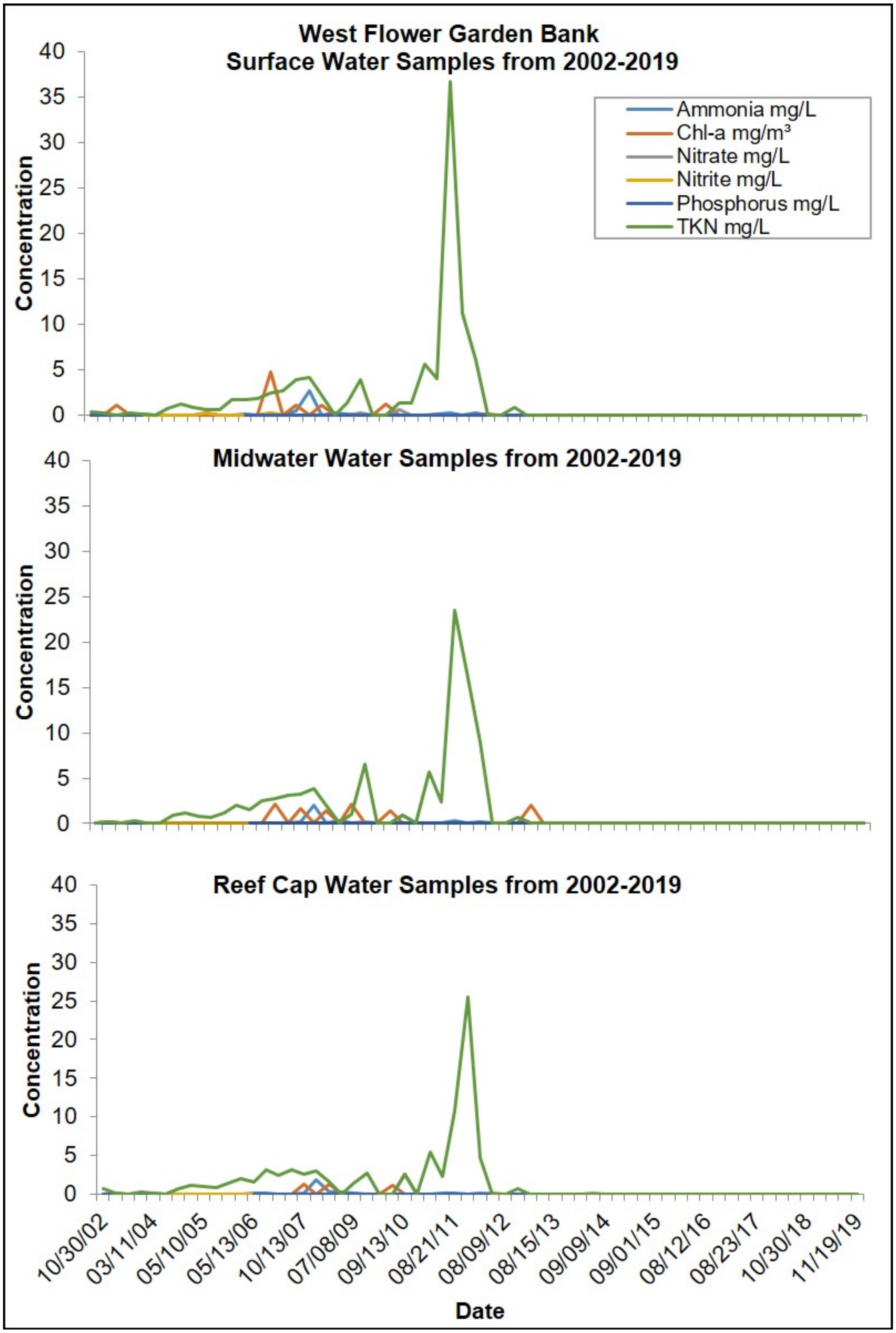


Figure 5.9. Nutrient concentrations from WFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2019.

Ocean carbonate measurements conducted by TAMU-CC are presented in Tables 5.2 and 5.3. Salinity in water samples exhibited marked lows in August 2019 across the system and a low of

29.26 ppt was recorded at EFGB in August (Table 5.2). Temperature trends remained typical of what has been observed since this type of analysis by TAMU-CC began in 2013.

Table 5.2. EFGB carbonate sample results for 2019 at three depths.

Sample Date	Depth (m)	Salinity (ppt)	Temp (°C)	pH Total	Alkalinity (μmol/kg)	DIC (μmol/kg)	pH in situ	$\Omega_{aragonite}$	pCO_2 (μatm)
2/5/2019	20	36.51	20.92	8.014	2397.9	2098.2	8.074	3.30	380.4
2/5/2019	10	36.47	21.05	8.025	2401.8	2095.8	8.083	3.38	371.6
2/5/2019	1	36.46	21.73	8.037	2400.7	2088.4	8.085	3.47	369.4
2/28/2019	1	36.61	22.43	8.039	2401.1	2087.0	8.077	3.50	377.1
5/16/2019	20	36.12	24.81	8.046	2398.5	2075.5	8.048	3.57	405.6
5/16/2019	10	36.27	25.28	8.047	2400.3	2076.8	8.042	3.58	413.3
5/16/2019	1	36.16	25.37	8.047	2395.0	2076.7	8.042	3.59	414.3
8/2/2019	20	32.58	29.00	8.080	2311.8	2012.0	8.020	3.66	432.9
8/2/2019	10	30.53	29.20	8.101	2266.6	1978.1	8.039	3.65	412.8
8/2/2019	1	29.26	30.00	8.107	2246.9	1966.8	8.033	3.54	425.7
11/19/2019	20	36.09	24.17	8.057	2396.4	2078.6	8.069	3.64	385.4
11/19/2019	10	35.95	24.25	8.066	2407.1	2073.0	8.077	3.70	376.6
11/19/2019	1	35.95	24.50	8.066	2398.4	2069.5	8.073	3.70	380.4

Table 5.3. WFGB carbonate sample results for 2019 at three depths.

Sample Date	Depth (m)	Salinity (ppt)	Temp (°C)	pH Total	Alkalinity (μmol/kg)	DIC (μmol/kg)	pH in situ	$\Omega_{aragonite}$	pCO_2 (μatm)
2/28/2019	20	36.55	21.94	8.031	2401.2	2091.0	8.076	3.44	377.7
2/28/2019	10	36.52	22.15	8.041	2402.3	2085.6	8.083	3.50	371.0
2/28/2019	1	36.50	22.35	8.039	2401.2	2085.8	8.078	3.50	376.2
5/16/2019	20	36.35	25.10	8.054	2400.5	2077.5	8.052	3.63	402.8
5/16/2019	10	36.36	25.34	8.052	2399.7	2076.9	8.046	3.63	408.6
5/16/2019	1	36.34	25.35	8.054	2396.0	2076.3	8.048	3.64	406.7
8/2/2019	20	35.12	29.5	8.088	2380.2	2053.0	8.021	3.88	437.6
8/2/2019	10	35.19	29.5	8.079	2363.3	2040.4	8.013	3.76	446.5
8/2/2019	1	32.99	29.4	8.081	2332.8	2026.7	8.016	3.67	444.4
11/19/2019	20	36.02	24.15	8.063	2399.8	2069.4	8.075	3.66	377.5
11/19/2019	10	36.04	24.22	8.068	2400.8	2073.3	8.079	3.71	374.7
11/19/2019	1	36.03	24.31	8.068	2400.6	2070.6	8.078	3.72	375.3

The pH and $\Omega_{aragonite}$ deviations remained fairly small in both 2019 and over the six-year period of carbonate chemistry monitoring. Carbonate chemistry indicated clear seasonality (highest dissolved inorganic carbon values in February, highest pCO_2 values in August, and highest $\Omega_{aragonite}$ values in November) within the water column around FGBNMS. The lowest pCO_2 values, where the air-sea pCO_2 gradients were greatest, corresponded with the lowest aragonite levels and the highest DIC records in February 2019 at EFGB and WFGB (Tables 5.2 and 5.3).

Discussion

Temperature, salinity, and turbidity were more variable at EFGB than WFGB in 2019. Satellite-derived SST exceeded 30°C in July and mid-August to September, generally following reef cap temperature trends. Data available for deep station sites at 30 m and 40 m surpassed 30°C for five consecutive days. Although water temperatures on the reef cap exceeded 30°C for six days at EFGB and ten days at WFGB, bleaching threshold curves calculated by Johnston et al. (2019) suggest that more than 50 days above 29.5°C would initiate a bleaching year at EFGB and WFGB, and this was not observed in 2019. However, significantly increasing monotonic seawater temperature trends from 1990 to 2019 were detected at both banks, suggesting that ocean temperatures at FGBNMS have risen over the past three decades and that more bleaching events will occur in the future.

Mean SSS fluctuated considerably at both banks in 2019 and reached a low in late June at WFGB and early July at EFGB according to satellite-derived data. Reef cap salinity values were above average from January through May and then decreased to a low of 33.85 psu at WFGB in mid-June and 32.64 psu in mid-July at EFGB. These values represent the lowest recorded reef cap salinity at EFGB since reliable salinity data acquisition was initiated in 2008 and the second lowest salinity recorded at WFGB. Average salinity values from May 15 through August 15 were 34.60 psu at EFGB and 35.0 psu at WFGB, indicating a prevalent freshwater presence that peaked in mid-July. Despite annual variation and a substantial increase in freshwater influence in 2019, salinity data collected at depth were within the accepted limits of salinity for coral reefs located in the Western Atlantic (31–38 psu; Coles and Jokiel 1992). The probable source of low-salinity water at the banks is a nearshore river-seawater mix that occasionally extends to the outer continental shelf, emanating principally from the Mississippi and Atchafalaya River watershed, subjecting the banks to nearshore processes and regional river runoff (Zimmer et al. 2010). In 2019, record flooding along the Mississippi River was recorded, and Hurricane Barry and Tropical Storm Imelda brought significant flooding to Louisiana and Texas (NOAA NCEI 2020). It could be hypothesized that corals at EFGB and WFGB are adapted to these periodic episodes of lower salinity coastal waters and are seemingly tolerant of (or simply unaffected by) such intermittent freshwater occurrences.

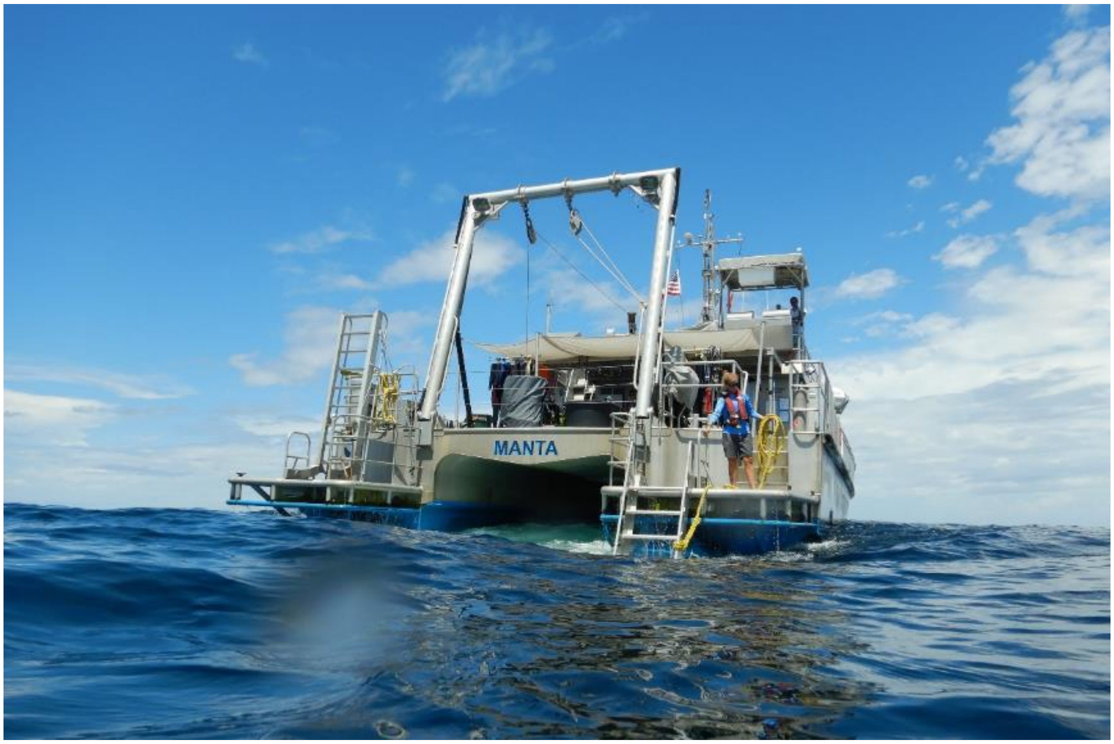
Comparatively high and more variable turbidity in April and May subsided during the summer and fall, although a turbidity sensor failure at EFGB created a gap in reef cap data from May 24 to August 1, 2019. Records indicated more frequent turbidity spikes at EFGB than at WFGB in 2019. Average turbidity values on coral reefs can range widely, but values of approximately 0.2 NTU are typical, and changes in turbidity or sustained spikes can be used as sentinels of environmental factors that may be impacting the reef ecosystem (Otero and Carbery 2005).

Water column profile data indicated that the sea water above the banks was generally well mixed; however, minor fluctuations in salinity and DO in the upper water column were observed in May at both banks. Despite reduced salinity levels, increased fluorescence in February, and indications of lightly stratified water in mid-May, laboratory analyses of nutrients remained below detectable limits. TKN concentrations, however, trended upwards from 2002 to 2011. This was likely due to organic nitrogen and ammonia forming in the water column through phytoplankton and bacteria cycling within the food chain. It is therefore subject to seasonal community fluctuations, but could also be affected by both point and non-point sources. When present, the probable sources of nutrients in the water column were nearshore waters (Nowlin et al. 1998), sediments (Entsch et al. 1983), or benthic and planktonic organisms (D'Elia and Wiebe 1990).

Carbonate analyses revealed that a period of freshwater was observed in August that likely contributed to decreased pCO_2. It should be noted that pH values from water column profiles differ by nearly a unit from the pH carbonate chemistry analysis from TAMU-CC (pH of 9 and pH of 8, respectively). Upon review of previous data points from earlier years, the Sea-Bird® Electronics *19plus* V2 CTD values have always been higher than carbonate analysis values by TAMU-CC, but values in 2019 were higher than expected, and may be the result of an electrode issue. In overall carbonate analysis, SST alone explained the majority of pH, Ω_{arag}, and pCO_2 variations, indicative of the overall oligotrophic nature of this area. The overall surface seawater pCO_2 did not significantly deviate from the atmospheric levels throughout annual cycles, although the largest air-sea gradient ($\Delta\ pCO_2$) occurred in early winter and late summer. Carbonate chemistry indicated clear seasonality within the water column around FGBNMS. The average $\Delta\ pCO_2$ suggests that this area had small net air-sea CO_2 flux. Seasonal and spatial distribution of seawater carbonate chemistry in 2019 demonstrates that seawater in the FGBNMS area, despite its relative proximity to land, behaved similar to an open ocean setting the majority of the time in terms of its annual pCO_2 fluctuation and minimal terrestrial influence. However, discharges from the Mississippi and Atchafalaya rivers, combined with many smaller Texas rivers, make EFGB and WFGB susceptible to pulses of freshwater that are more acidic than the ocean.

Overall, the water column is responsible for the connectivity among all the various coral reef habitats and acts as the medium between aquatic and terrestrial systems. Thus, water quality data are critical components of monitoring programs in order to provide indications of processes related coral reef ecosystem functions and provide information on the incursion of land-based materials.

Chapter 6:
Conclusions



A diver's view of the back deck of the R/V *Manta* upon surfacing from a dive at FGBNMS. Photo: Jimmy MacMillan/CPC

Despite coral cover declines on most coral reefs of the world in recent decades, mean coral cover within EFGB and WFGB long-term monitoring one-hectare study sites has ranged from 40–60% for the combined 30 years of monitoring. Even with macroalgae percent cover increasing significantly after the mass mortality of *D. antillarum* in the 1980s (with sustained cover of approximately 30% since 2009), unlike many other shallow reefs in the Caribbean region, increases in macroalgae cover have not been concomitant with reduced coral cover at EFGB or WFGB one-hectare study sites.

Coral species composition changed with depth and coral cover increased in repetitive photostations. Although coral cover in repetitive photostations and random transect surveys is not comparable, the former are critical in enabling researchers to track individual sites over time (especially during extreme events such as bleaching). The long-term monitoring program benefits from having both random benthic surveys and repetitive monitoring stations.

The reef fish community was comprised primarily of the families Labridae and Pomacentridae. Biomass was uniformly distributed between large and small individuals, and piscivores had the greatest mean biomass at both EFGB and WFGB. Groupers dominated the piscivores within EFGB surveys, while snapper density was higher at WFGB (but still low at both locations). Such differences may have to do with the location of one-hectare study sites on each bank relative to features such as the bank edges, but this has not been tested. For the first time since 2013, lionfish were not observed in fish surveys.

Although seawater temperatures on the reef cap exceeded 30°C for six days at EFGB and 10 days at WFGB in 2019, FGBNMS bleaching threshold curves suggest that more than 50 days above 29.5°C would initiate a bleaching year at EFGB and WFGB, and this was not observed during the sampling period. However, significantly increasing monotonic seawater temperature trends from 1990 to 2019 were detected at both banks, suggesting ocean temperatures have risen at FGBNMS over the past three decades and that bleaching events will most likely occur in the future. Salinity and nutrient loads on the reefs were nominal during 2019, and carbonate chemistry indicated that the area acted as a net CO_2 sink.

The most apparent changes since monitoring began in 1989 have been the increase in coral and macroalgae percent cover. EFGB and WFGB appear unusual compared to other reefs in the region because macroalgae has not yet affected coral cover as it has in many other places throughout the region (Jackson et al 2014). The monitoring program at EFGB and WFGB is critical to ensure data are available to understand and distinguish the drivers of ecosystem variation in the northern Gulf of Mexico (Karnauskas et al. 2015) and preserve the characteristics that sustain the health of this system. FGBNMS is an ideal sentinel site for the detection and tracking of conditions that are changing because of natural events and human threats, and has a robust historical baseline to which new data may be compared. This level of monitoring empowers resource managers to make educated decisions regarding management and research amid threats such as climate change, invasive species, storms, and water quality degradation.

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Glossary of Acronyms

ANOSIM – analysis of similarity

BOEM – Bureau of Ocean Energy Management

CCA – crustose coralline algae

CCL – Carbon Cycle Laboratory

Chl a – chlorophyll a

CPCe – Coral Point Count® with Excel® extensions

CTD – conductivity, temperature, and depth

CV% – coefficient of variation

DIC – total dissolved CO_2

DO – dissolved oxygen

EFGB – East Flower Garden Bank

EPA – Environmental Protection Agency

ERDDAP – Environmental Research Division Data Access Program

FGBNMS – Flower Garden Banks National Marine Sanctuary

LTM – long-term monitoring

MMS – Minerals Management Service

NOAA – National Oceanic and Atmospheric Administration

PCO – principal coordinates ordination

pCO_2 – CO_2 partial pressure

PERMANOVA – p*ermutational multivariate analysis of variance*

QA/QC – quality assurance/quality control

SIMPER – similarity percentages

SSS – sea surface salinity

SST – sea surface temperature

TABS – Texas Automated Buoy System

TAMU – Texas A&M University

TAMU-CC – Texas A&M University Corpus Christi

TKN – total Kjeldahl nitrogen

USGS – United States Geological Survey

WFGB – West Flower Garden Bank

Literature Cited

AGRRA (Atlantic and Gulf Rapid Reef Assessment). 2012. AGRRA Protocols, version 5.4. J.C. Lang, K.W. Marks, P.A. Kramer, P.R. Kramer, and R.N. Ginsburg, eds. 31 pp.

Anderson, M.J., R.N. Gorley, K.R. Clarke. 2008. PERMANOVA+ for PRIMER: guide to software and statistical methods. PRIMER-E Ltd. Plymouth, United Kingdom.

Aronson, R.B., P.J. Edmunds, W.F. Precht, D.W. Swanson, and D.R. Levitan. 1994. Large-scale, long-term monitoring of Caribbean coral reefs: simple, quick, inexpensive methods. Atoll Research Bulletin 421:1–19.

Aronson, R.B. and W.F. Precht. 2000. Herbivory and algal dynamics on the coral reef at Discovery Bay, Jamaica. Limnology and Oceanography 45:251–255.

Aronson, R.B., W.F. Precht, T.J. Murdoch, and M.L. Robbart. 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 23:84–94.

Bauer, L., A. Zitello, S.D. Hile, and T. McGrath. 2015a. Biogeographic characterization of fish and benthic communities, Jobos Bay, Puerto Rico 2009-06-08 to 2009-06-13 (NODC Accession 0125200). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Bauer, L., S.D.Hile, and T. McGrath. 2015b. Biogeographic characterization of fish and benthic communities, Vieques, Puerto Rico 2007-05-14 to 2007-05-24 (NODC Accession 0125235). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Bauer, L., S.D. Hile, and T. McGrath. 2015c. Biogeographic characterization of fish and benthic communities, St Thomas, US Virgin Islands 2012-06-12 to 2012-06-22 (NODC Accession 0125418). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Bohnsack, J.A. and S.P. Bannerot. 1986. A stationary visual technique for quantitatively assessing community structure of coral reef fishes. NOAA Technical Report NMFS 41. 15 pp.

Bohnsack, J.A. and D.E. Harper. 1988. Length-weight relationships of selected marine reef fishes from southeastern United States and the Caribbean. NOAA Technical Memorandum NMFS-SEFC-215. 31 pp.

Bright, T.J., D.W. McGrail, R. Rezak, G.S. Boland, and A.R. Trippett. 1985. The Flower Gardens: A compendium of information. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 85-0024. 103 pp.

Bright, T.J. and R. Rezak. 1976. A biological and geological reconnaissance of selected topographical features on the Texas continental shelf. Final Rept. to U.S. Dept. of Interior, Bureau of Land Management. Contract No. 08550-CT5-4. 377 pp.

Bruckner, A.W. and R.J. Bruckner. 1998. Destruction of coral by Sparisoma viride. Coral Reefs 17:350.

Bruckner, A.W., R.J. Bruckner, and P. Sollins. 2000. Parrotfish predation on live coral: “spot biting” and “focused biting.” Coral Reefs 19:50.

Caldow, C., R. Clark, K. Edwards, S.D. Hile, C. Menza, E. Hickerson, and G.P. Schmahl. 2009. Biogeographic characterization of fish communities and associated benthic habitats within the Flower Garden Banks National Marine Sanctuary: sampling design and implementation of SCUBA surveys on the coral caps. NOAA Technical Memorandum NOS NCCOS 81. Silver Spring, MD. 134 pp.

Caldow, C., K. Roberson, L. Bauer, C.F.G. Jeffrey, S.D Hile, and T. McGrath. 2015. Biogeographic characterization of fish and benthic communities, Parguera Region, Puerto Rico 2000-08-21 to 2010-09-21 (NODC Accession 0125202). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Carpenter R.C., and P.J. Edmunds. 2006. Local and regional scale recovery of *Diadema* promotes recruitment of scleractinian corals. Ecological Letters 9:268–277.

Clark, R., J.C. Taylor, C.A. Buckel, and L.M. Kracklet (eds). 2014. Fish and benthic communities of the Flower Garden Banks National Marine Sanctuary: science to support sanctuary management. NOAA Technical Memorandum NOS NCCOS 179. Silver Spring, MD. 317 pp.

Clark, R., C.A. Buckel, C. Taylor, S.D. Hile, and T. McGrath. 2015a. Biogeographic characterization of fish and benthic communities, Flower Garden Banks, Texas 2010-09-10 to 2012-10-02 (NODC Accession 0118358). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Clark, R., S.D. Hile, and T. McGrath. 2015b. Biogeographic characterization of fish and benthic communities, St Croix, US Virgin Islands 2012-05-07 to 2012-05-18 (NODC Accession 0125237). National Oceanographic Data Center, NOAA. Dataset. 12/30/2014.

Clarke, K.R. 1990. Comparison of dominance curves. Journal of Experimental Marine Biology and Ecology 138:143–157.

Clarke, K.R., P.J. Somerfield, and R.N. Gorley. 2008. Testing of null hypotheses in exploratory community analyses: similarity profiles and biota-environment linkage. Journal of Experimental Marine Biology and Ecology. 366:56–69.

Clarke, K.R., R.N. Gorley, P.J. Sommerfield, and R.M. Warwick. 2014. Change in marine communities: an approach to statistical analysis and interpretation, 3rd edition. PRIMER-E: Plymouth, Devon, UK. 260 pp.

Coles, S.L. and P.L. Jokiel. 1992. Effects of salinity on coral reefs. In Pollution in tropical aquatic systems, D.W. Connell and D.W. Hawker, eds. Boca Raton, FL: CRC Press. 147–166 pp.

CSA (Continental Shelf Associates). 1989. Environmental monitoring program for exploratory well #1, lease OCS-G 6264 High Island Area, South Extension, East Addition, Block A-401 near the Flower Garden Bank. Final Report. Jupiter, FL: CSA. 96 pp.

CSA (Continental Shelf Associates). 1996. Long-term monitoring at the East and West Flower Garden Banks. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 96-0046. 77 pp.

Dahl, K.A. and W.F. Patterson. 2014. Habitat-specific density and diet of rapidly expanding invasive red lionfish, Pterois volitans, populations in the northern Gulf of Mexico. PLoS One 9:e105852.

Darling, E.S., S.J. Green, J.K. O'Leary, and I.M. Côté. 2011. Indo-Pacific lionfish are larger and more abundant on invaded reefs: a comparison of Kenyan and Bahamian lionfish populations. Biological Invasions 13:2045–2051.

DeBose, J.L., M.F. Nuttall, E.L. Hickerson, and G.P. Schmahl. 2012. A high-latitude coral community with an uncertain future: Stetson Bank, northwestern Gulf of Mexico. Coral Reefs 32:255–267.

D'Elia, C.F. and W.J. Wiebe. 1990. Biogeochemical nutrient cycles in coral-reef ecosystems. In: Dubinsky, Z., ed. Coral reefs. Amsterdam: Elsevier. 49–74 pp.

DeMartini, E.E., A.M. Friedlander, S.A. Sandin, and E. Sala. 2008. Differences in fish-assemblage structure between fished and unfished atolls in the northern Line Islands, central Pacific. Marine Ecology Progress Series 365:199–215.

Dennis, G.D. and T.J. Bright. 1988. Reef fish assemblages on hard banks in the northwestern Gulf of Mexico. Bulletin of Marine Science 43:280–307.

Dokken, Q.R., I.R. MacDonald, J.W. Tunnell, C.R. Beaver, G.S. Boland, and D.K. Hagman. 1999. Long-term monitoring of the East and West Flower Garden Banks 1996–1997. U.S. Dept. of the Interior, Mineral Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 99-0005. 101 pp.

Dokken, Q. R., I.R. MacDonald, J.W. Jr. Tunnell, T. Wade, K. Withers, S.J. Dilworth, T.W. Bates, C.R. Beaver, and C.M. Rigaud. 2003. Long-term monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 1998–2001: final report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2003-031. 90 pp.

Edmunds, P.J. and R.C. Carpenter. 2001. Recovery of Diadema antillarum reduces macroalgal cover and increases abundance of juvenile corals on a Caribbean reef. Proceedings of the National Academy of Sciences 98:5067–5071.

Entsch, B., K.G. Boto, R.G. Sim, and J.T. Wellington. 1983. Phosphorous and nitrogen in coral reef sediments. Limnology and Oceanography 28:465–476.

Froese, R. and D. Pauly, eds. 2019. FishBase. World Wide Web electronic publication. Accessed 04/25/2019. www.fishbase.org

Gardner, T.A., I.M. Côté, J.A. Gill, A. Grant, and A.R. Watkinson. 2003. Long-term region-wide declines in Caribbean corals. Science 301:958–960.

Gittings, S.R. 1998. Reef community stability on the Flower Garden Banks, northwest Gulf of Mexico. Gulf of Mexico Science 1998:161–169.

Gittings, S.R. and T.J. Bright. 1986. Coral recovery following an incident of anchoring on the East Flower Garden Bank, northwestern Gulf of Mexico. Rep. to U.S. Dept. of Commerce, NOAA Sanctuary Programs Division, Silver Spring, MD. Final report contract No. NA85AA-H-CZ015. 45 pp.

Gittings, S.R. and T.J. Bright. 1987. Mass mortality of Diadema antillarum at the Flower Garden Banks, Northwest Gulf of Mexico: effect on algae and coral cover. Abstract. Benthic Ecology Meeting, Raleigh, NC.

Gittings S.R., G.S. Boland, K.J.P. Deslarzes, D.K. Hagman, and B.S. Holland. 1992. Long-term monitoring at the East and West Flower Garden Banks. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 92-0006. 206 pp.

Glynn, P.W. and L. D'Croz. 1990. Experimental evidence for high temperature stress as the case of El Nino-coincident coral mortality. Coral Reefs 8:181–191.

Goreau, T.F. and J.W. Wells. 1967. The shallow water Scleractinia of Jamaica: revised list of species and their vertical distribution range. Bulletin of Marine Science 17:442–454.

Green, S.J. and I.M. Côté. 2009. Record densities of Indo-Pacific lionfish on Bahamian coral reefs. Coral Reefs 28:107.

Green, S.J., N.K. Dulvy, A.L.M. Brooks, J.L. Akins, A.B. Cooper, S. Miller, and I.M. Côté. 2014. Linking removal targets to the ecological effects of invaders: a predictive model and field test. Ecological Applications 24:1311-1322.

Hagman, D.K. and S.R. Gittings. 1992. Coral bleaching on high latitude reefs at the Flower Garden Banks, NW Gulf of Mexico. Proceedings of the 7th International Coral Reef Symposium 1:38–43.

Harris, H.E., A.Q. Fogg, M.S. Allen, R.N.M. Ahrens, and W.F. Patterson, III. 2020. Precipitous declines in northern Gulf of Mexico invasive lionfish populations following the emergence of an ulcerative skin disease. Scientific Reports 10:1934.

Helsel, D.R. and R.M. Hirsch. 2002. Statistical methods in water resources. In: Techniques of water-resources investigations of the United States Geological Survey. Book 4, Hydrologic analysis and interpretation. Washington, DC: U.S. Geological Survey. 522 pp.

Helsel, D.R., D.K. Mueller, and J.R. Slack. 2006. Computer program for the Kendall family of trend tests: U.S. Geological Survey Scientific Investigations Report 2005–5275. 4 pp.

Heron, S.F., J.A. Maynard, R. van Hooidonk, and C. M. Eakin. 2016. Warming trends and bleaching stress of the world's coral reefs 1985–2012. Scientific Reports 6:3842.

Hipel, K.W. and A.I. McLeod. 1994. Time series modelling of water resources and environmental systems. http://www.stats.uwo.ca/faculty/aim/RPackages.htm

Hoeksema, B.W., P. Bongaerts, and C.C. Baldwin. 2017. High coral cover at lower mesophotic depths: a dense *Agaricia* community at the leeward side of Curaçao, Dutch Caribbean. Marine Biodiversity 47:67–70.

Hughes, T.P., J.T. Kerry, M. Álvarez-Noriega, J.G. Álvarez-Romero, et al. 2017. Global warming and recurrent mass bleaching of corals. Nature 543:373–377.

Jackson, J.B.C., M.K. Donovan, K.L. Cramer, V.V. Lam, eds. 2014. Status and trends of Caribbean coral reefs: 1970–2012. Global Coral Reef Monitoring Network, IUCN, Gland, Switzerland. 304 pp.

Johnston, M.A., M.F. Nuttall, R.J. Eckert, J.A. Embesi, N.C. Slowey, E.L. Hickerson, and G.P. Schmahl. 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, LA. OCS Study BOEM 2013-215. 362 pp.

Johnston, M.A., M.F. Nuttall, R.J. Eckert, J.A. Embesi, N.C. Slowey, E.L. Hickerson, and G.P. Schmahl. 2015. Long-term monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 2011–2012, volume 1: technical report. U.S. Dept. of Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2015-027. 194 pp.

Johnston, M.A., M.F. Nuttall, R.J. Eckert, J.A. Embesi, T.K. Sterne, E.L. Hickerson, G.P. Schmahl. 2016a. Persistence of Coral Assemblages in Flower Garden Banks National Marine Sanctuary, Gulf of Mexico. Coral Reefs 35: 821-826.

Johnston, M.A., M.F. Nuttall, R.J. Eckert, J.A. Embesi, T.K. Sterne, E.L. Hickerson, and G.P. Schmahl. 2016b. Rapid invasion of Indo-Pacific lionfishes *Pterois volitans* (Linnaeus, 1758) and *P. miles* (Bennett, 1828) in Flower Garden Banks National Marine Sanctuary, Gulf of Mexico, documented in multiple data sets. Bioinvasions Records 5:115–122.

Johnston, M.A., R.J. Eckert, M.F. Nuttall, T.K. Sterne, J.A. Embesi, D.P. Manzello, E.L. Hickerson, and G.P. Schmahl. 2017a. Long-term monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 2013–2015, volume 1: technical report. U.S. Dept. of Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEM 2017-058. 186 pp.

Johnston, M.A., T.K. Sterne, R.J. Eckert, M.F. Nuttall, J.A. Embesi, R. Walker, X. Hu, E.L. Hickerson, and G.P. Schmahl. 2017b. 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, M.A., T.K. Sterne, R.D. Blakeway, J. MacMillan, M.F. Nuttall, X. Hu, J.A. Embesi, E.L. Hickerson, and G.P. Schmahl. 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, M.A., M.F. Nuttall, R.J. Eckert, R.D. Blakeway, T.K. Sterne, E.L. Hickerson, G.P. Schmahl, M.T. Lee, J. MacMillan, and J.A. Embesi. 2018b. Localized coral reef mortality event at East Flower Garden Bank, Gulf of Mexico. Bulletin of Marine Science 95:239–250.

Johnston, M.A., E.L. Hickerson, M.F. Nuttall, R.D. Blakeway, T.K. Sterne, R.J. Eckert, and G.P. Schmahl. 2019. Coral bleaching and recovery from 2016 to 2017 at East and West Flower Garden Banks, Gulf of Mexico. Coral Reefs 38:787–799.

Johnston, M.A., R.D. Blakeway, K. O'Connell, J. MacMillan, M.F. Nuttall, X. Hu, J.A. Embesi, E.L. Hickerson, and G.P. Schmahl. 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.

JPL MUR MEaSUREs Project. 2015. GHRSST Level 4 MUR Global Foundation Sea Surface Temperature Analysis (v4.1). Ver. 4.1. PO.DAAC, CA, USA. Dataset. 10/18/2019. https://doi.org/10.5067/GHGMR-4FJ04.

Karnauskas, M., M.J. Schirripa, J.K. Craig, G.S. Cook, C.R. Kelble, J.J. Agar, B.A. Black, D.B. Enfield, D. Lindo-Atichati, B.A. Muhling, K.M. Purcell, P.M. Richards, and C. Wang. 2015. Evidence of climate-driven ecosystem reorganization in the Gulf of Mexico. Global Change Biology 21:2554–2568.

Knowlton, N. and J.B.C. Jackson. 2008. Shifting baselines, local impacts, and global change on coral reefs. PLoS Biology 6:e54.

Kohler, K.E. and S.M. Gill. 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 and Geosciences 32:1259–1269.

Kramer, P.A. 2003. Synthesis of coral reef health indicators for the western Atlantic: results of the AGRRA program (1997–2000). Atoll Research Bulletin 496:1–58.

Morris, J.A. Jr., and P.E. Whitfield. 2009. Biology, ecology, control and management of the invasive Indo-Pacific Lionfish: an updated integrated assessment. NOAA Technical Memorandum NOS NCCOS 99. 57 pp.

Mumby, P.J., A.J. Edwards, J.E Arias-González, K.C. Kindeman, P.G Blackwell, A. Gall, M.I. Gorczynska, A.R. Harborne, C.L. Pescod, H. Renken, C.C.C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427:533–536.

Mumby, P.J. and R.S. 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 transition. Springer, Netherlands. 509–519 pp.

NOAA CRCP (NOAA Coral Reef Conservation Program). 2020. A status report for Florida's Coral Reef (2014–2018 data). NOAA CRCP and the University of Maryland Center for Environmental Science.

Cambridge, Maryland, USA.
https://www.coris.noaa.gov/monitoring/status_report/docs/FL_508_compliant.pdf

NOAA Coral Reef Watch. 2020. NOAA Coral Reef Watch Version 3.1 Daily Global 5-km Satellite Sea Surface Temperature Product, January 1 – December 31, 2019. NOAA Coral Reef Watch, College Park, Maryland, USA: NOAA Coral Reef Watch. Accessed 3/15/2020. https://coralreefwatch.noaa.gov/satellite/hdf/index.php.

NOAA NCEI (National Centers for Environmental Information). 2020. State of the climate: national climate report for annual 2019. Dataset. 8/12/2020. https://www.ncdc.noaa.gov/sotc/national/201913.

Nowlin, W.D., A.E. Jochens, R.O. Reid, and S.F. DiMarco. 1998. Texas-Louisiana shelf circulation and transport processes study: synthesis report. Volume II: appendices. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 98-0036. 288 pp.

Nuttall, M.F., M.A. Johnston, R.J. Eckert, J.A. Embesi, E.L. Hickerson, and G.P. Schmahl. 2014. Lionfish (*Pterois volitans* [Linnaeus, 1758] and *P. miles* [Bennett, 1828]) records within mesophotic depth ranges on natural banks in the Northwestern Gulf of Mexico. Bioinvasions Records 3:111–115.

Nuttall, M.F., R.D. Blakeway, J. MacMillan, T.K. Sterne, X. Hu, J.A. Embesi, E.L. Hickerson, M.J. Johnston, G.P. Schmahl, and J. Sinclair. 2018. Stetson Bank long-term monitoring: 2017 annual report. National Marine Sanctuaries Conservation Series ONMS-19-03. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Flower Garden Banks National Marine Sanctuary, Galveston, TX. 94 pp.

Nuttall, M.F., P.J. Somerfield, T.K. Sterne, J.T. MacMillan, J.A. Embesi, E.L. Hickerson, M.A. Johnston, G.P. Schmahl, and J. Sinclair. 2020. Stetson Bank long-term monitoring: 1993–2015. National Marine Sanctuaries Conservation Series ONMS-20-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, Silver Spring, MD. 156 pp.

Ogden, J. and R. Wicklund (eds). 1988. Mass bleaching of coral reefs in the Caribbean: a research strategy. National Undersea Research Program Research Report 88-2. U.S. Department of Commerce. Washington, DC. 51 pp.

ONMS (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.

ONMS (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.

Otero, E. and K.K. Carbery. 2005. Chlorophyll *a* and turbidity patterns over coral reefs systems of La Parguera Natural Reserve, Puerto Rico. Revista Biologia Tropical 53:25–32.

Pattengill, C.V. 1998. The structure and persistence of reef fish assemblages of the Flower Garden Banks National Marine Sanctuary. Doctoral dissertation, Texas A&M University, College Station, TX. 176 pp.

Pattengiill-Semmens, C.V. and B.X. Semmens. 1998. An analysis of fish survey data generated by nonexpert volunteers in the Flower Garden Banks National Marine Sanctuary. Gulf of Mexico Science 16:196–207.

Precht, W.F., R.B. Aronson, K.J.P. Deslarzes, M.L. Robbart, A. Gelber, D. Evans, B. Gearheart, and B. Zimmer. 2006. Long-term monitoring at the East and West Flower Garden Banks, 2002–2003: final report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2004–031. 182 pp.

REEF (Reef Environmental Education Foundation). 2014. Reef Environmental Education Foundation. World Wide Web electronic publication. Accessed 4/24/2014. www.reef.org

Roberson, K., S. Viehman, and R. Clark. 2014. Development of benthic and fish monitoring protocols for the Atlantic/Caribbean Biological Team: National Coral Reef Monitoring Program. NOAA Coral Reef Conservation Program. 8 pp.

Robertson, D.R., N. Simoes, C. Gutiérrez Rodríguez, V.J. Piñeros, and H. Pérez-España. 2016. An Indo-Pacific damselfish well established in the southern Gulf of Mexico: prospects for a wider, adverse invasion. Journal of the Ocean Science Foundation 19:1–17.

Robertson, D.R., O. Dominguez-Dominguez, B. Victor, and N. Simoes. 2018. An Indo-Pacific damselfish (Neopomacentrus cyanomos) in the Gulf of Mexico: origin and mode of introduction. PeerJ 6:e4328.

Rooker, J.R., Q.R. Dokken, C.V. Pattengill, and G.J. Holt. 1997. Fish assemblages on artificial and natural reefs in the Flower Garden Banks National Marine Sanctuary, USA. Coral Reefs 16:83–92.

Ruzicka, R., K. Semon, M. Colella, V. Brinkhuis, J. Kidney, J. Morrison, K. Macaulay, J.W. Porter, M. Meyers, M. Christman, and J. Colee. 2009. Coral Reef Evaluation and Monitoring Project annual report. NOAA/NOS MOA-2001-683/7477. 111 pp.

Sale, P.F. 1991. The ecology of fishes on coral reefs. Academic Press, Inc., San Diego, California. 754 pp.

Schmahl, G.P., E.L. Hickerson, and W.F. Precht. 2008. Biology and ecology of coral reefs and coral communities in the Flower Garden Banks region, northwestern Gulf of Mexico. In: Riegl, B. and R. Dodge (eds.), Coral reefs of the USA. Springer Netherlands. 221–261 pp.

Simons, R.A. 2019. ERDDAP. https://coastwatch.pfeg.noaa.gov/erddap. Monterey, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, Environmental Research Division.

SOKI Wiki. 2014. Abundance biomass curve (ABC method) - Indicators - Confluence, SOKI, Antarctic Climate and Ecosystems Co-operative Research Centre. Accessed 12/8/ 2014. http://www.soki.aq/x/foFm

Steneck, R.S., S. Arnold, and H. DeBey. **2011**. Status and trends of Bonaire's reefs 2011 and cause for grave concerns. University of Maine and NOAA National Marine Fisheries Service, Silver Spring, MD. 137 pp.

Toth, L.T., R.V. Woesik, T.J.T. Murdocj, S.R. Smith, J.C. Ogden, W.F. Precht, and R.B. Aronson. **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.

van Hooidonk, R., J.A. Maynard, J. Tamelander, J. Gove, G. Ahmadia, L. Raymundo, G. Williams, S. Heron, and S. Planes. **2016**. Local-scale projections of coral reef futures and implications of the Paris Agreement. Nature Scientific Reports 6:39666.

Wetmore, L.S., M.A. Dance, R.L. Hill, and J. Rooker. **2020**. Community dynamics of fish assemblages on mid-shelf and outer-shelf coral reefs in the Northwestern Gulf of Mexico. Frontiers in Marine Science 7:1–15.

Zimmer, B., L. Duncan, R.B. Aronson, K.J.P. Deslarzes, D. Deis, M.L. Robbart, W.F. Precht, L. Kaufman, B. Shank, E. Weil, J. Field, D.J. Evans, and L. Whaylen. **2010**. Long-term monitoring at the East and West Flower Garden Banks, 2004–2008. Volume I: Technical report. U.S. Dept. of the Interior, Bureau of Ocean Energy Management, Regulation, and Enforcement, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study BOEMRE 2010-052. 310 pp.

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