Office of National Marine Sanctuaries
National Oceanic and Atmospheric Administration

NATIONAL MARINE SANCTUARIES CONSERVATION SERIES

Long-Term Monitoring at East and West Flower Garden Banks: 2023 Annual Report

January 2025
National Marine Sanctuaries Conservation Science Series ONMS-25-01

sanctuaries.noaa.gov

U.S. Department of Commerce
Jeremy Pelter, Acting Secretary

National Oceanic and Atmospheric Administration
Vice Admiral Nancy Hann, Deputy Under Secretary for Operations and performing the duties of Under Secretary of Commerce for Oceans and Atmosphere and NOAA Administrator

National Ocean Service
Nicole LeBoeuf, Assistant Administrator

Office of National Marine Sanctuaries
John Armor, Director

Report Authors:
Olivia J. Eisenbach[1], Michelle A. Johnston[2], Kelly O'Connell[1], Ryan Hannum[1], Donavon R. French[1], Marissa F. Nuttall[1], and Jacque Cresswell[1]

[1] CPC, Inc.

[2] Flower Garden Banks National Marine Sanctuary

Suggested citation: Eisenbach, O. J., Johnston, M. A., O'Connell, K., Hannum, R., French, D. R., Nuttall, M. F., & Cresswell, J. (2025). *Long-term monitoring at East and West Flower Garden Banks: 2023 annual report*. National Marine Sanctuaries Conservation Series ONMS-25-01. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Cover photo: A longsnout butterflyfish (*Prognathodes aculeatus*) hides in a crevice at Flower Garden Banks National Marine Sanctuary. Photo: Marissa Nuttall/CPC

About the National Marine Sanctuaries Conservation Series

The Office of National Marine Sanctuaries, part of the National Oceanic and Atmospheric Administration, serves as the trustee for a system of underwater parks encompassing more than 620,000 square miles of ocean and Great Lakes waters. The 17 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 or historical research and monitoring projects. The series facilitates integration of natural sciences, socioeconomic and social 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.

Disclaimer

Report Availability

Electronic copies of this report may be downloaded from the Office of National Marine Sanctuaries website.

Contact

Kelly O'Connell
Research Operations Specialist
CPC Inc., contracted to
NOAA Flower Garden Banks National Marine Sanctuary
NOAA Galveston Laboratory
4700 Avenue U, Bldg. 216
Galveston, TX 77551
(409) 356-0387
Kelly.oconnell@noaa.gov

Table of Contents

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 2023, along with 34 years of historical monitoring data. EFGB and WFGB are part of the National Oceanic and Atmospheric Administration's 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 FGBNMS and the Bureau of Ocean Energy Management, with support from the National Marine Sanctuary Foundation. In 2023, mean coral cover, estimated from random transect data within EFGB and WFGB one-hectare study sites, was 55%. Mean macroalgae cover was 31% within the EFGB study site and 29% within the WFGB study site. Since 1989, mean coral cover has increased significantly at WFGB and remained stable at EFGB. Mean macroalgae cover has increased significantly at both banks since 1998. This is the second year that data from reef-wide stratified random photo transects were include in the report in addition to data from the study sites on each bank. Mean benthic cover in reef-wide random transects was similar to that in study site random transects; however, more data are needed for a trend analysis. 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 study sites. The reef fish community was comprised primarily of the families Labridae (wrasses) and Pomacentridae (damselfishes). Biomass was evenly distributed between large and small individuals, and herbivores represented the greatest mean biomass at both banks. One manta ray and two lionfish were observed. No non-native regal demoiselles were observed in 2023; however, they were observed in Stetson Bank surveys and on nearby artificial reef sites. During 2023, water temperatures at reef depth exceeded 30 °C for 53 days at EFGB, surpassing the local threshold for coral bleaching (33 days). At WFGB, temperatures exceeded 30 °C for 31 days, surpassing the local threshold for coral bleaching (22.3 days). By late September, coral bleaching affected an average of 28.06% of reef-wide coral cover at EFGB and 5.61% of reef-wide coral cover at WFGB, with paling affecting an average of 50.64% and 17.09% of coral cover, respectively. A significant increase in seawater temperature was detected at both banks from 1990 to 2023. The Flower Garden Banks remains one of very few reefs in the Western Atlantic Ocean where resources have resisted degradation over the last several decades. Maintaining successful stewardship will require continuing the strong links between sanctuary management and long-term monitoring.

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, coral disease

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

A scuba diver cleans algae off a long-term monitoring station marker on the coral reef cap at East Flower Garden Bank. Photo: G.P. Schmahl/NOAA

Habitat Description

The coral reef-capped East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB), located within Flower Garden Banks National Marine Sanctuary (FGBNMS), 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 the surface expression of salt dome formations located approximately 190 km south of the Texas and Louisiana border, containing several distinct habitats ranging in depth from 16–166 m (Rezak et al., 1985; Schmahl et al., 2008; Figure 1.1).

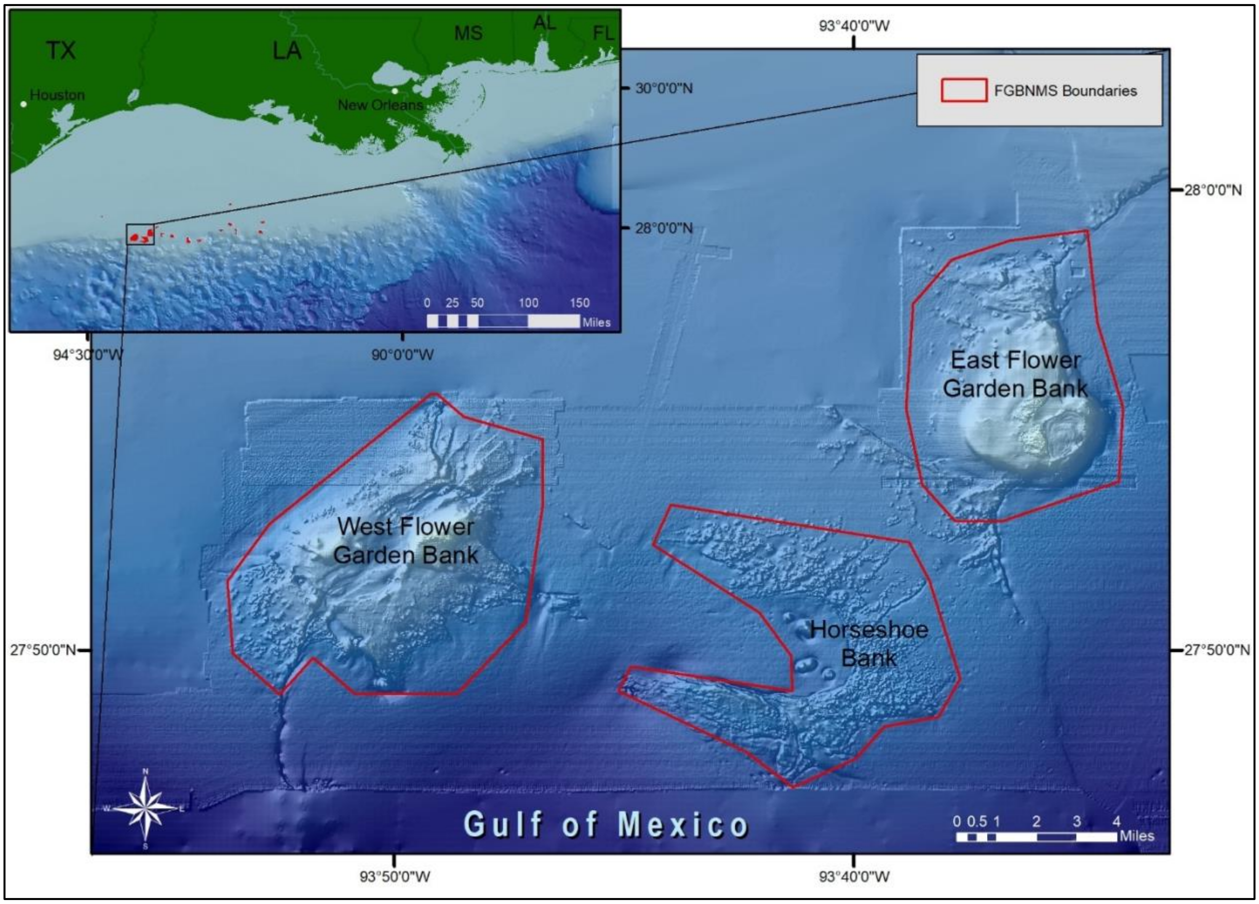


Figure 1.1. Map of EFGB, WFGB, and Horseshoe Bank with an inset of the Gulf Coast states and FGBNMS boundaries along the continental shelf of the northwestern Gulf of Mexico. Horseshoe Bank is not part of the study area, but is now part of FGBNMS. Image: Marissa Nuttall/CPC

The caps of the banks are approximately 20 km apart and within the photic zone, where conditions are ideal for colonization by shallow water species of corals, algae, invertebrates, and fish that are also found in the Caribbean region (Goreau & 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 16–50 m. Although the coral species found on the reef caps of the banks are the same as those on Caribbean reefs, octocorals are absent in shallow habitats, and scleractinian corals of the genus *Acropora* are exceedingly rare. These differences are likely due to remoteness, depth, and the latitude of the banks; FGBNMS is near the northernmost limit of coral reef development and is distanced from source populations by several hundred kilometers (Bright et al., 1985; Continental Shelf Associates, 1989).

FGBNMS was designated in 1992 (15 C.F.R. § 922.120) by the National Oceanic and Atmospheric Administration (NOAA) under the National Marine Sanctuaries Act. The sanctuary was expanded in 1996 to include Stetson Bank, and expanded once again in 2021 to include an additional 14 reefs and banks along the continental shelf of the northwestern Gulf of Mexico (86 Fed. Reg. 4937 [Jan 19, 2021]).

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, and now the Bureau of Ocean Energy Management [BOEM]) has supported monitoring at EFGB and WFGB to determine if the reefs are impacted by nearby oil and gas activities (Figure 1.2).

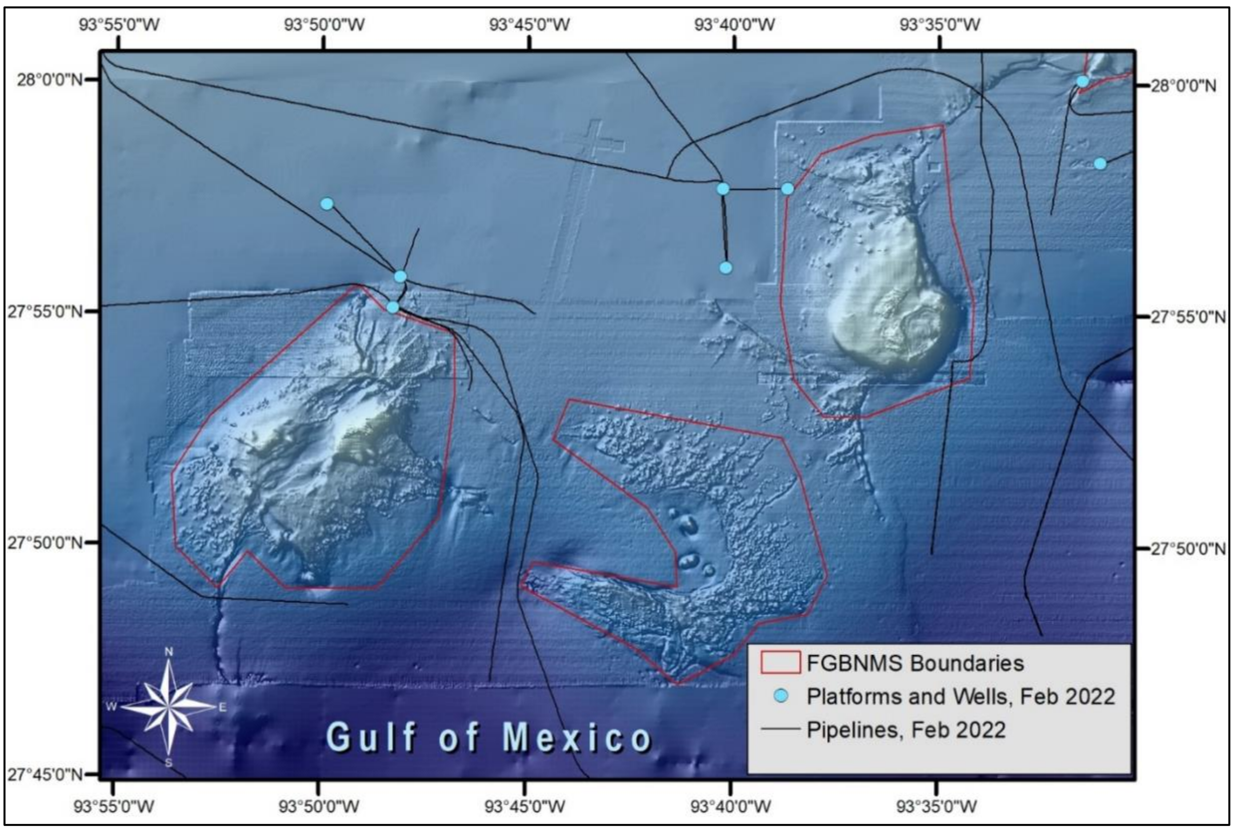


Figure 1.2. Map of oil and gas platforms, wells, and pipelines near EFGB and WFGB as of February 2022. Image: Marissa Nuttall/CPC

Initially under industry funding, then Minerals Management Service funding and a contract with Texas A&M University (TAMU), one-hectare long-term monitoring study sites were established on each bank in 1989, marking the start of the Flower Garden Banks long-term monitoring program (Continental Shelf Associates, 1989; Gittings et al., 1992; Figure 1.3). Monitoring was conducted by both TAMU and environmental consulting firms through competitive contracts until 2009, at which time BOEM and NOAA established an interagency agreement for FGBNMS to carry out the long-term monitoring program.

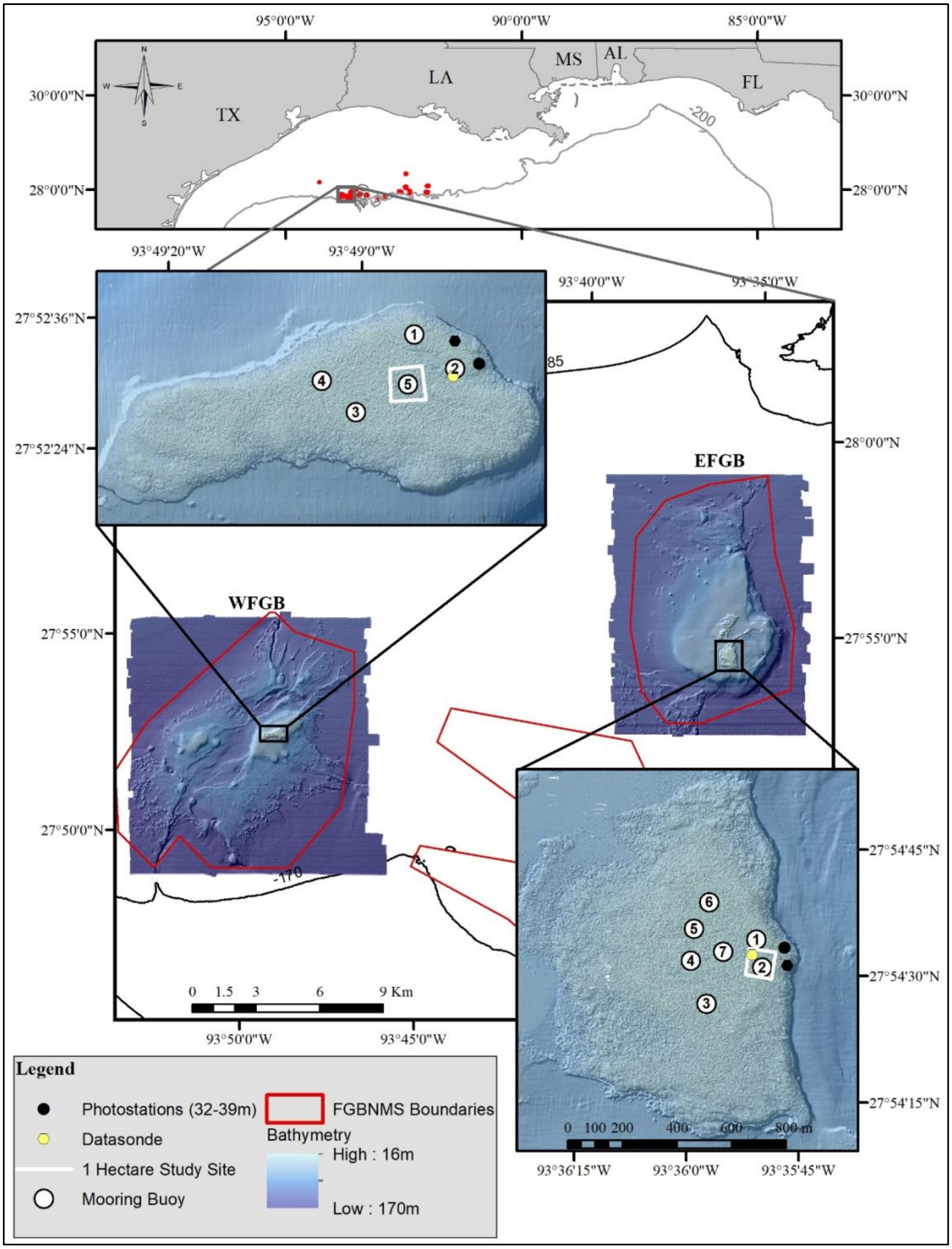


Figure 1.3. Shaded relief maps of EFGB and WFGB, with inset of the Gulf of Mexico coastline, long-term monitoring one-hectare study sites, and datasonde and repetitive photostation locations, which range in depth from 32–39 m. Image: Marissa Nuttall/CPC

Long-Term Monitoring Program Objectives

FGBNMS priorities include managing natural resources as stated in the National Marine Sanctuaries Act and identifying threats and potential sources of impacts to coral reefs, 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 long-term monitoring program is of significant interest to both NOAA and BOEM, which share responsibility to protect and monitor these important marine resources. The five objectives (and corresponding indicators) of the FGBNMS long-term monitoring program are to:

- 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 demographics
- Identify changes in coral reef health resulting from both natural and human-induced stressors to facilitate management responses
    - Indicators: Bleaching, disease, and invasive species
- Facilitate adaptive management of activities impacting reef-related resources
    - Indicators: Baseline data and image archive of damage to resources
- 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 on environmental conditions and the status of living resources
    - Indicators: Published, peer-reviewed annual reports

Long-Term Monitoring Program Components

The long-term monitoring 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 alter the integrity of EFGB and WFGB 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;
- 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 *P. guttatus*) surveys establish current population levels and trends;
- Water quality datasondes record salinity, temperature, and turbidity at depth; and
- Quarterly sampling of chlorophyll *a*, ammonia, nitrate, nitrite, total Kjeldahl nitrogen, and phosphorus documents water column productivity.

The long-term monitoring study area consists of several locations on the EFGB and WFGB coral reef caps 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 m^2 study sites (100 m x 100 m or 1 hectare; hereafter referred to as "study sites") at EFGB and WFGB. The corners and centers of the study sites are marked by large eyebolts as reference markers. Depth ranges from 17–27 m within the EFGB study site and 18–25 m within the WFGB study site (Figure 1.4; Figure 1.5). Mooring buoy anchors (#2 at EFGB and #5 at WFGB) were installed near the study site centers to facilitate field operations (Figure 1.3; Table 1.1). Mooring buoys are attached at these sites only during field research activities, thus restricting access at other times. Additionally, permanent repetitive photostations were installed at each bank beyond the study site boundaries to capture benthic cover in depth ranges of 32–39 m: 23 repetitive photostations are located east of mooring buoy #2 at EFGB and 24 repetitive photostations are located north and east of mooring buoy #2 at WFGB (Figures 1.4; Figure 1.5). Water quality datasondes are located near mooring buoy #2 at EFGB and WFGB (Figures 1.3–1.5). Additional temperature loggers are paired with repetitive photostations at 30 m and 40 m at both banks (Figures 1.4; Figure 1.5).

Table 1.1. Coordinates and depths for permanent moorings within the 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
WFGB Mooring #2	27° 52.526' N	93° 48.836' W	24.4

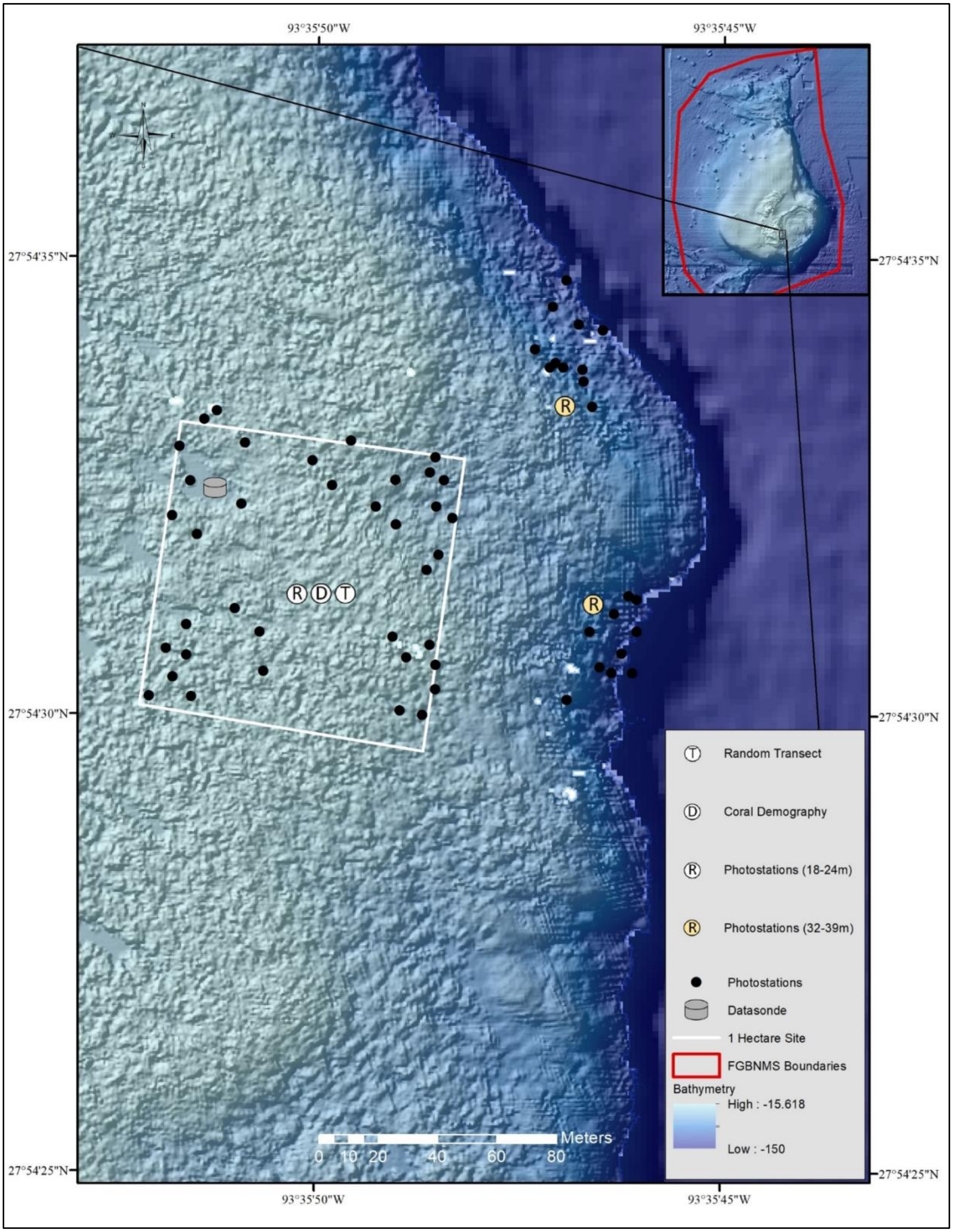


Figure 1.4. Shaded relief map of EFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also shown. Image: Marissa Nuttall/CPC

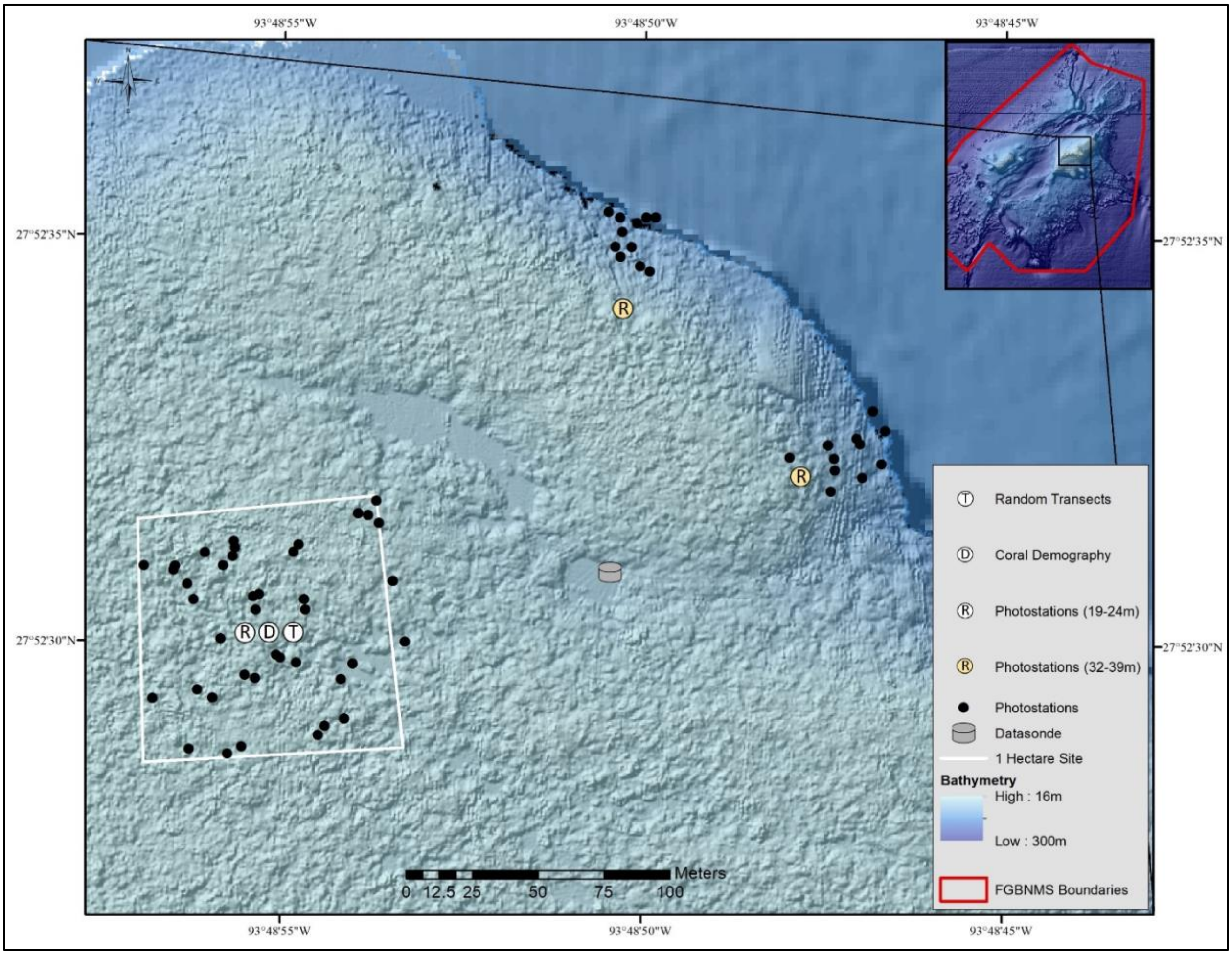


Figure 1.5. Shaded relief map of WFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also shown. Image: Marissa Nuttall/CPC

Long-Term Monitoring Field Operations and Data Collection

To date, the monitoring program has spanned 34 years, and annual data have been collected nearly continuously. Long-term monitoring data were collected at both EFGB and WFGB in 2023 and scuba operations were conducted from the NOAA R/V *Manta* and M/V *Fling* (Table 1.2). Water samples were collected, water quality instruments were exchanged, and data were downloaded by FGBNMS staff during water quality cruises in January, May, and August 2023 and July 2024 (Table 1.2). See each respective chapter for detailed field operation methods.

Annual field work at EFGB and WFGB was conducted August 14–18, 2023 (Table 1.2), with mild surface and bottom currents (>0.25 kt), clear water column visibility (25 m), 29 °C water temperature, and 2–3 foot seas. Due to emergency vessel maintenance conducted earlier in the month that impacted the cruise schedule, EFGB and WFGB field work was combined into one five-day cruise. All monitoring tasks within study sites were completed. The datasondes at EFGB and WFGB were retrieved, downloaded, and replaced.

Coral paling was reported by recreational divers in late August; therefore, a response cruise was conducted from September 1–2 to photograph repetitive photostations at EFGB and WFGB. As bleaching progressed, reef-wide random photo transect benthic surveys were collected by FGBNMS divers during a National Coral Reef Monitoring Program (NCRMP) cruise aboard the M/V *Fling* from September 18–22, 2023 (Table 1.2), as time did not allow for these surveys outside the study site during the August 2023 cruise. Three FGBNMS divers also joined a recreational cruise aboard the M/V *Fling* from November 11–12, 2023 to collect thermally tolerant coral samples.

The R/V *Manta* entered into its annual shipyard repair period on September 30, 2023. Due to these repairs, as well as unfavorable weather, a water quality cruise was not conducted in November 2023.

Table 1.2. Monitoring cruises completed at EFGB and WFGB in 2023.

Date	Cruise and Tasks Completed
01/09–10/2023	Water quality cruise on R/V *Manta*: Instrument exchange and water samples collected
05/02–04/2023	Water quality cruise on R/V *Manta*: Instrument exchange and water samples collected
08/14–18/2023	Long-term monitoring cruise on R/V *Manta*: EFGB and WFGB annual monitoring and water samples collected in study sites
09/01–02/2023	Bleaching response cruise on M/V *Fling*: Repetitive photostations photographed
09/18–22/2023	NCRMP and FGBNMS cruise on M/V *Fling*: Reef-wide random transect surveys collected at EFGB and WFGB
11/11–12/2023	Bleaching response cruise on M/V *Fling*: Coral samples collected at EFGB and WFGB
07/14–17/2024	Long-term monitoring cruise on M/V *Fling*: Water quality instrument exchange at EFGB and WFGB

Chapter 2: Benthic Community

Fish swim above corals at FGBNMS. Photo: Tiffany Crumbley/Crumbley Photography

Benthic Community 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 study sites. These surveys were used to compare habitat and document benthic reef community status and trends within and between the study sites. In addition, photo transect surveys were conducted in a random stratified design across the coral caps to assess benthic cover outside of the EFGB and WFGB study sites.

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 located; most were selectively installed in areas with high coral cover. While these stations can help identify directions and causes of change, they are not intended to estimate reef-wide populations or communities.

Benthic Community Methods

Random Transect Field Methods

In 2023, 16 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 Sony® Alpha 6600 digital camera with a Sony® E 16mm 2.8f lens in a Nauticam® NA-A6600 housing and a Nauticam® 4.33” fisheye dome port, mounted on a 0.58-m T-frame 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 perpendicular to the slope of the bottom substrate. 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 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) to digital videography using video still frame grabs (2002 to 2009) and then digital still images (2010 to 2023; Gittings et al., 1992; Continental Shelf Associates, 1996; Dokken et al. 1999, 2003; Precht et al., 2006; Zimmer et al., 2010; Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021, 2022, 2024). 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; Continental Shelf Associates, 1996). From 1996 to 2003, random-dot layers were generated manually in photo software programs (Dokken et al., 1999, 2003).

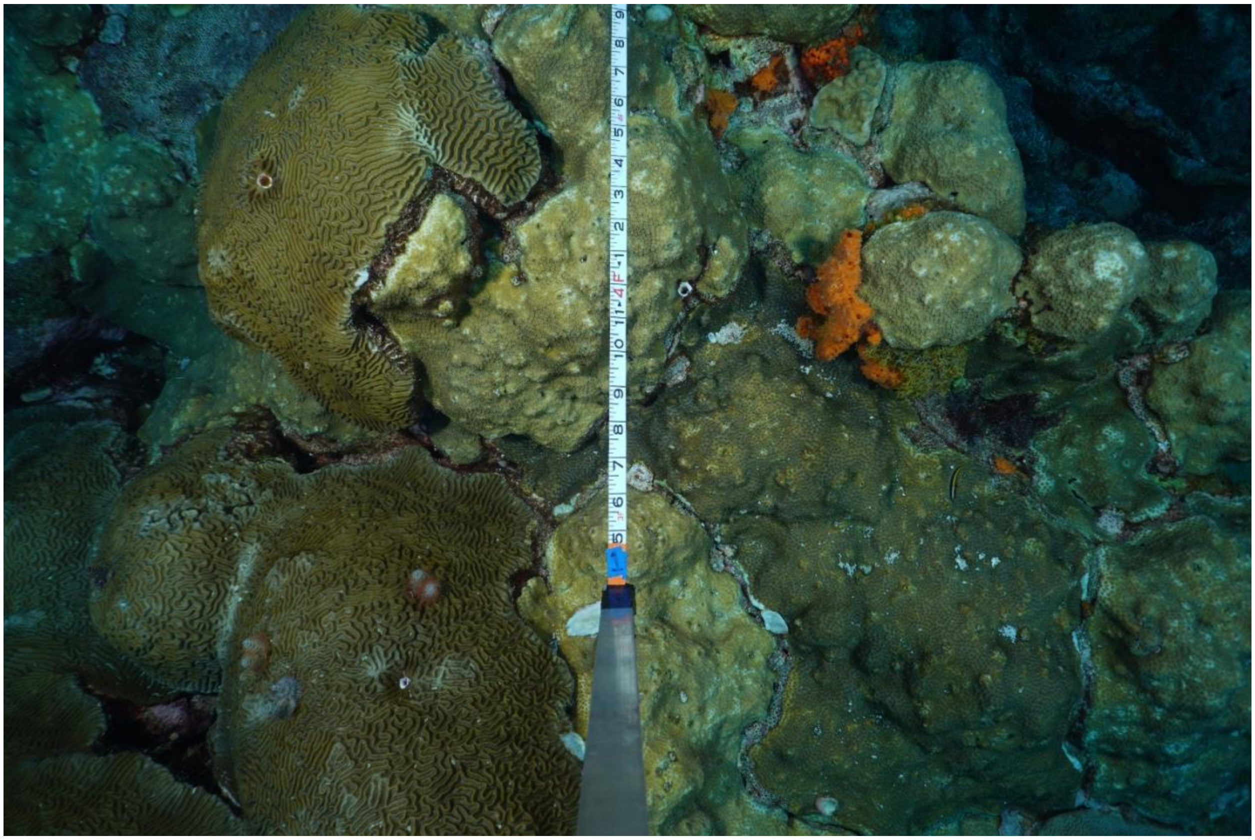


Figure 2.1. Photo taken at a marked interval along a random transect with a camera mounted to an aluminum t-frame at EFGB in 2023. Photo: Josh Harvey/CPC

In addition to random transect surveys inside the EFGB and WFGB study sites, transect surveys (n = 16 at EFGB, n = 13 at WFGB) were conducted in a random stratified designed across both coral caps to assess benthic cover outside of the study sites (National Centers for Coastal Ocean Science, 2018). These surveys were conducted in partnership with the NCRMP cruise aboard the *M/V Fling* in September 2023. Fewer surveys were completed at WFGB due to inclement weather in the field.

Random Transect Data Processing

Mean percent benthic cover from random transect images was analyzed using CPCe version 4.1 with a 510-point overlay randomly distributed among all images within a transect (30 spatially random points per image; Aronson et al., 1994; Kohler & Gill, 2006). Organisms or substrate type positioned beneath each random point were tallied, and organisms were identified to the lowest possible taxonomic level. 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 & Precht, 2000; Aronson et al., 2005), 6) sand, and 7) an "other" category (biotic components such as sea urchins, ascidians, fish, serpulid worms, 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 study site and all transects in reef-wide surveys. Results are presented as mean percent cover ± standard error (SE).

Incidents 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" (Lang et al., 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. Any point that landed on a lesion (a stark border dividing healthy coral tissue from white, denuded skeleton along colony margins) during an observed coral disease outbreak was classified as "disease" (Johnston et al., 2023).

Consistency for photographic random transect methods was ensured by training all scientific divers in the proper operation of all the camera systems. 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 study sites as well as reef-wide surveys (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 among groups (Clarke et al., 2014).

Coral species composition was compared between study sites and within reef-wide surveys 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 banks from all survey combined. 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 2023), benthic groups by year and 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 similarity profile analysis (SIMPROF) tests to identify significant ($\alpha = 0.05$) clusters within the data (Clarke et al., 2008). 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 2023. Monotonic trends in mean percent cover data were assessed using the Mann-Kendall trend test in R version 4.2.2 (Hipel & McLeod, 1994; Helsel & Hirsch, 2002). Tests for significant correlation among benthic cover groups were completed in R version 4.2.2 with Pearson's correlation (Helsel & 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. In 2023, data from 1989–2002 were reanalyzed using current methods in CPCe and Primer®. This report includes data from both methods to show differences in results; however, moving forward, all future reports will only include reanalyzed data.

Repetitive Photostation Field Methods

Repetitive photostations, marked by permanent pins with numbered tags on the reef, were located by scuba divers using underwater maps displaying compass headings and distances to each station. All 37 photostations were located and photographed within the EFGB study site and all 41 were located and photographed within the WFGB study site. In addition, permanent repetitive photostations ranging in depth from 32–39 m and located beyond the study site boundaries were also documented. Twenty-two out of 24 photostations were located and photographed at EFGB and 22 out of 24 photostations were located and photographed at WFGB, representing 92% of deep photostations at both EFGB and WFGB.

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.

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 2023) (Gittings et al., 1992; Continental Shelf Associates, 1996; Dokken et al., 1999, 2003; Precht et al., 2006; Zimmer et al., 2010; Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024). 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 or not located due to missing tags or overgrown station posts. Nevertheless, approximately 40 photostations have been maintained within each study site since 1989. Within the 32–39 m depth range, nine of the 23 EFGB deep photostations were established in 2003 and 12 of the 24 WFGB deep 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.

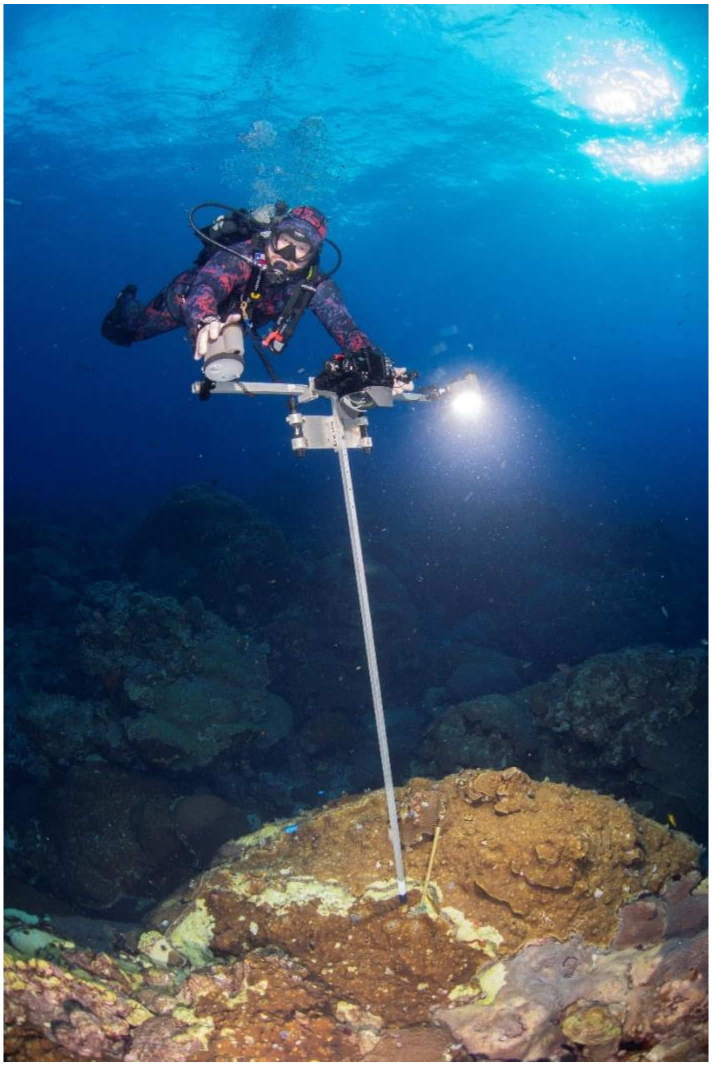


Figure 2.2. NOAA diver prepares to photograph a repetitive photostation with camera and strobes mounted to an aluminum t-frame. Photo: Donavon French/CPC

Repetitive Photostation Data Processing

Mean percent benthic cover from repetitive photostation images was analyzed using CPCe version 4.1 (Aronson et al., 1994; Kohler & 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 bank. Results are presented as mean percent cover ± SE. Repetitive photostation comparisons were only made with other repetitive photostations and not with data from random transects. Because photostations were not randomly selected, they are not intended to estimate study site or reef-wide cover or populations.

Consistency for repetitive photographic methods was ensured by using trained divers and standardized camera settings, equipment, and operating procedures. Photographs were reviewed in the field promptly after images were taken to ensure the quality was sufficient for analysis. After all benthic components were identified in CPCe files, QA/QC consisted of an independent review by a separate, trained researcher, different from the CPCe analyzer. Any mistakes were corrected before percent cover analysis was completed.

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 2022 ($n = 24$ at EFGB and $n = 27$ at WFGB) using a paired t-test in R version 4.2.2.

Benthic Community Results

Random Transect Mean Percent Cover

Coral, followed by macroalgae, had the highest mean percent benthic cover at EFGB and WFGB in both study site and reef-wide surveys in 2023 (Figure 2.3).

PERMANOVA revealed no significant differences in benthic community composition between EFGB and WFGB study sites and/or in reef-wide surveys in 2023 (Table 2.1). Given this finding, study site and reef-wide surveys at each bank were combined for some analyses.

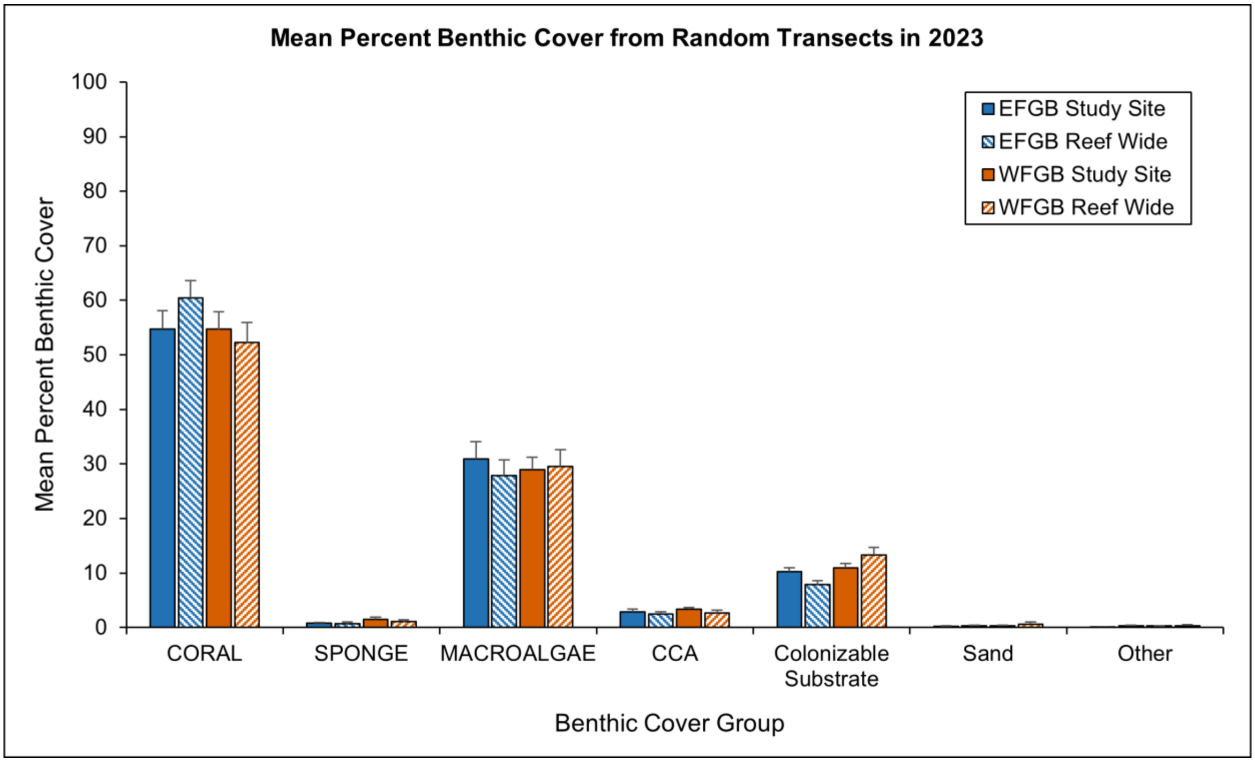


Figure 2.3. Mean percent benthic cover +SE from random transect surveys within study sites and reef-wide at EFGB and WFGB in 2023.

Table 2.1. PERMANOVA results comparing 2023 mean percent benthic cover between banks and by survey location.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	6.59	1	2.06	0.10
Location (study site vs. reef-	1.84	1	0.576	0.63
Bank x Location	4.23	1	1.32	0.232
Res	182.70	57		
Total	195.4	60		

Fifteen species of coral were observed within the EFGB study site and reef-wide surveys. Sixteen and 15 species of coral were observed in the WFGB study site and reef-wide surveys, respectively (Figure 2.4). *Orbicella franksi* was the most abundant coral species observed within EFGB (24.35 ± 3.16%) and WFGB (25.00 ± 2.98%) study site surveys. *Porites astreoides* ranked second within the EFGB study site (9.01 ± 1.31%) while *Pseudodiploria strigosa* ranked second within the WFGB study site (6.51 ± 1.32%; Figure 2.4). The species of the *Orbicella* spp. complex, consisting of *O. franksi, O. faveolata*, and *O. annularis*, are listed as threatened species under the Endangered Species Act.

PERMANOVA analysis revealed no significant difference in coral species composition between study site and reef-wide surveys, but there were significant differences between banks for all surveys (Table 2.2). SIMPER analysis indicated that this was due to significantly higher *O. franksi* cover in the EFGB surveys (contributing 60%).

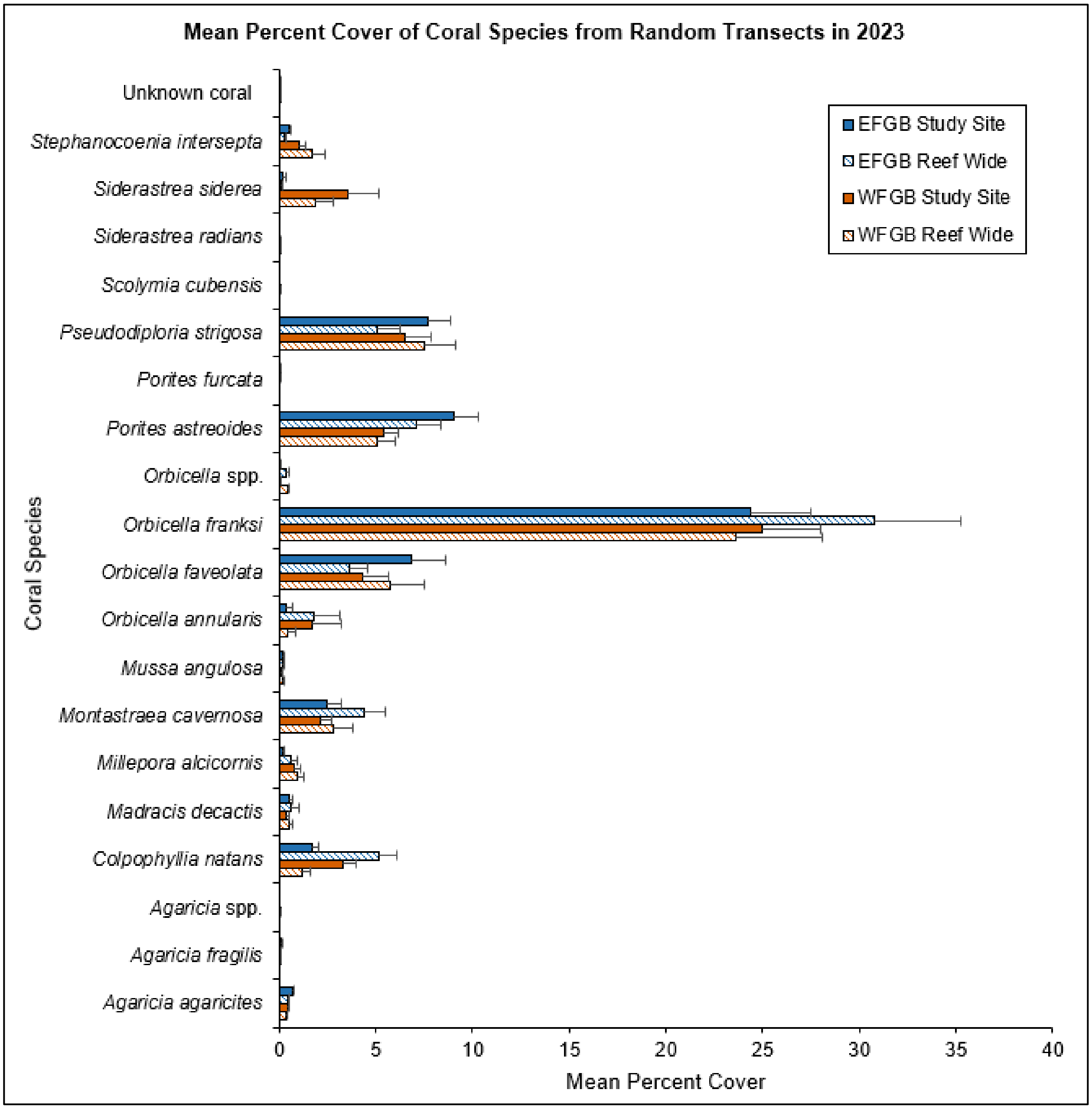


Figure 2.4. Mean percent cover + SE of observed coral species from random transect surveys within EFGB and WFGB study sites and reef-wide surveys in 2023.

Table 2.2. PERMANOVA results comparing coral species mean percent cover from EFGB and WFGB study site random transects and reef-wide surveys in 2023. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	24.927	1	2.2296	**0.0279**
Location (study site vs. reef-wide)	10.944	1	0.97894	0.4377
Bank x Location	20.174	1	1.8045	0.0741
Res	637.25	57		

Source	Sum of Squares	df	Pseudo-F	P (perm)
Total	693.99	60		

Coral species diversity measures were averaged for all survey locations by bank in 2023 (Table 2.3). ANOSIM analysis revealed no significant differences in coral community diversity metrics between banks for all surveys.

Table 2.3. Mean coral species diversity measures ± SE from all EFGB and WFGB random transect surveys in 2023.

Random Transect Coral Diversity Measures	EFGB	WFGB
Margalef's species richness (d)	2.62 ± 0.10	2.99 ± 0.07
Pielou's evenness (J')	0.88 ± 0.01	0.89 ± 0.01
Shannon diversity (H'(loge))	1.87 ± 0.05	2.00 ± 0.02

Less than 1% of coral cover was affected by coral bleaching, paling, or fish biting; however, coral bleaching did occur later in the season (see Chapter 3). 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 & Bruckner 1998; Bruckner et al., 2000). Signs of recent mortality and coral disease totaled 0.25% of coral cover in all random transect surveys.

Random Transect Long-Term Trends

Mean percent coral cover in study sites (17–27 m) from 1989 to 2023 ranged from 40–64% at EFGB and 37–66% at WFGB. It increased significantly in the WFGB site over that time period ($\tau = 0.57$, $p < 0.001$; Figure 2.6), and remained stable at the EFGB study site. Data on sponge, CCA, macroalgae, colonizable substrate, and sand data began to be collected in 1992. In the original analysis of benthic cover, macroalgae was consistently below 5% within the study sites prior to 1999 with an increase to 20% in 1999 (Figure 2.5; Johnston et al., 2016a). A 2023 reanalysis of historical data from 1992–2001, using current CPCe techniques, revealed that macroalgae cover ranged between 10–30% prior to 1999, only slightly lower than the 30% average observed over the past 13 years (Figure 2.6). Reflecting its habitat preference, macroalgae cover varied inversely with colonizable substrate; they were significantly correlated at EFGB ($\tau = -8.31$, $p < 0.001$) and WFGB ($\tau = -7.86$, $p < 0.001$). While macroalgae colonized available substrate, it did not outcompete or displace coral. From 1992 to 2022, macroalgae significantly increased in EFGB ($\tau = 0.47$, $p < 0.001$) and WFGB ($\tau = 0.40$, $p = 0.003$) study sites, while colonizable substrate significantly decreased in EFGB ($\tau = -0.43$, $p = 0.001$) and WFGB ($\tau = -0.55$, $p < 0.001$) sites.

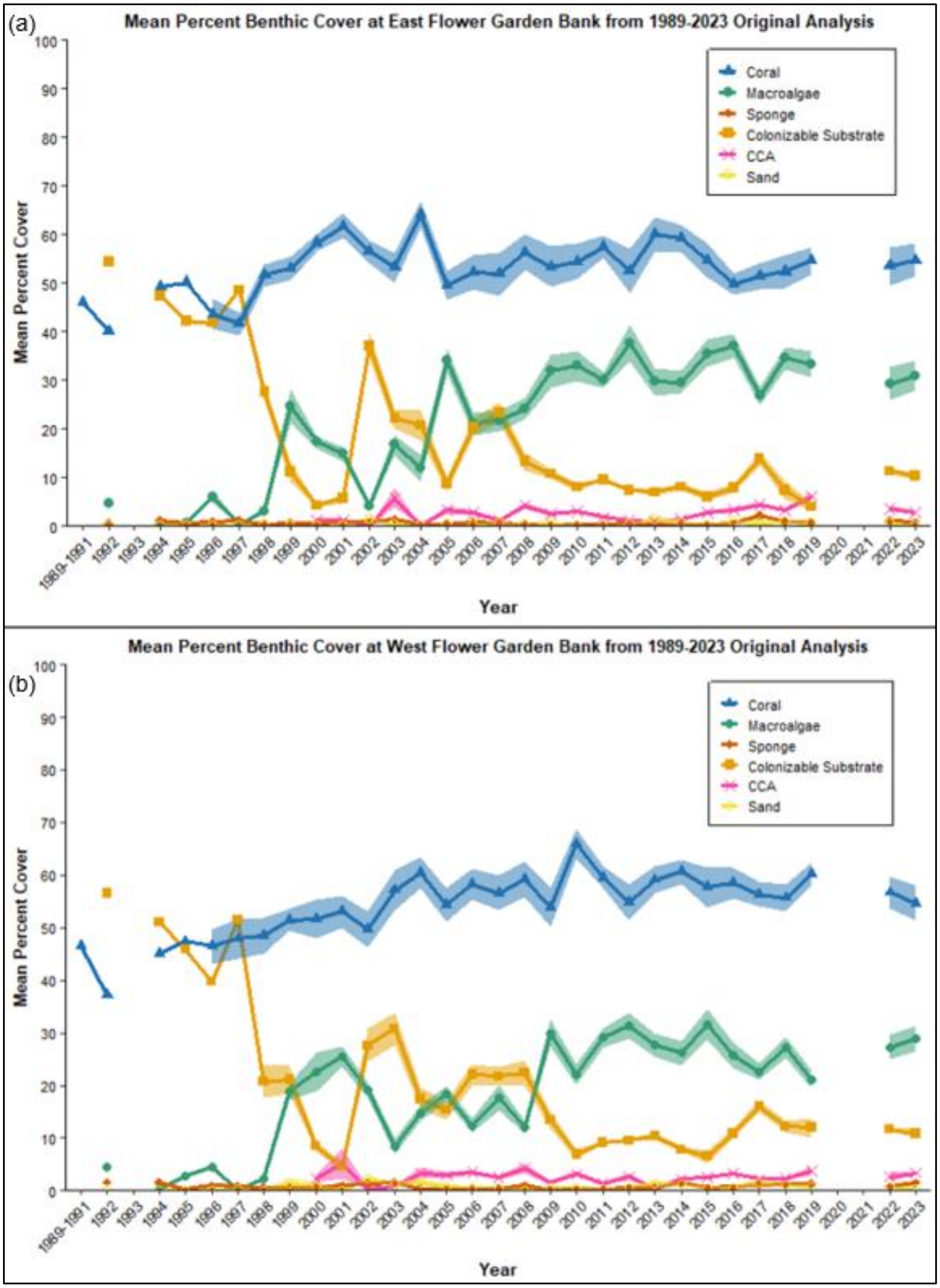


Figure 2.5. Mean percent benthic cover ± SE bands from random transect surveys within (a) EFGB and (b) WFGB study sites from 1989 to 2023 using original analysis methods. Only coral cover data were reported from 1989–1991 and no mean percent cover data was reported in 1993, 2020, or 2021. Source: Gittings et al., 1992 (1989 to 1991); Continental Shelf Associates, 1996 (1992 to 1995); Dokken et al., 2003 (1996 to 2001); Precht et al., 2006; Zimmer et al., 2010 (2002 to 2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009 to 2022)

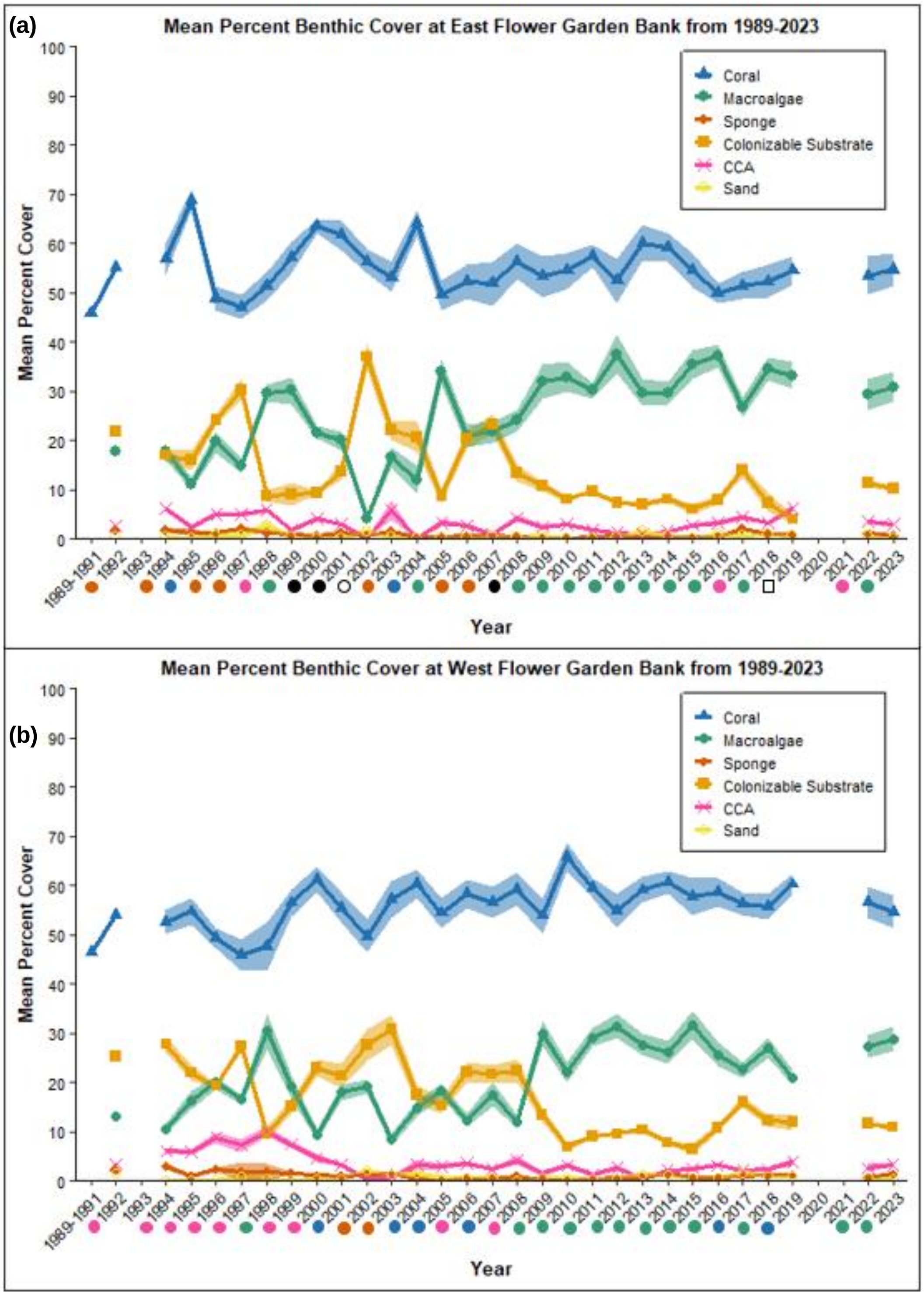


Figure 2.6. Mean percent benthic cover ± SE bands from random transect surveys within (a) EFGB and (b) WFGB study sites from 1989 to 2023 using data from the re-analysis of historical data. The colored dots below the years on the x-axis represent significant year clusters corresponding to SIMPROF groups in Figures 2.7 and 2.8. Only coral cover data were reported from 1989–1991 and no mean percent cover data were reported in 1993, 2020, or 2021. Sources: Gittings et al., 1992 (1989 to 1991); Continental Shelf Associates, 1996 (1992 to 1995); Dokken et al., 2003 (1996 to 2001); Precht et al., 2006; Zimmer et al., 2010 (2002 to 2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009 to 2022)

For available yearly mean benthic percent cover data (1992 to 2023), SIMPROF analysis detected five significant year clusters in the EFGB study site (1992, 1994, 1996–1997, 2003, and 2006–2007; 1995 and 2004; 1998, 2017, and 2022; 1999, 2005, 2009 to 2016, 2018, and 2023; 2000–2001 and 2008), while 2019 weas not grouped with any other years (Figure 2.7). Colonizable substrate and macroalgae mean percent cover contributed to the majority of the dissimilarity among all year group clusters.

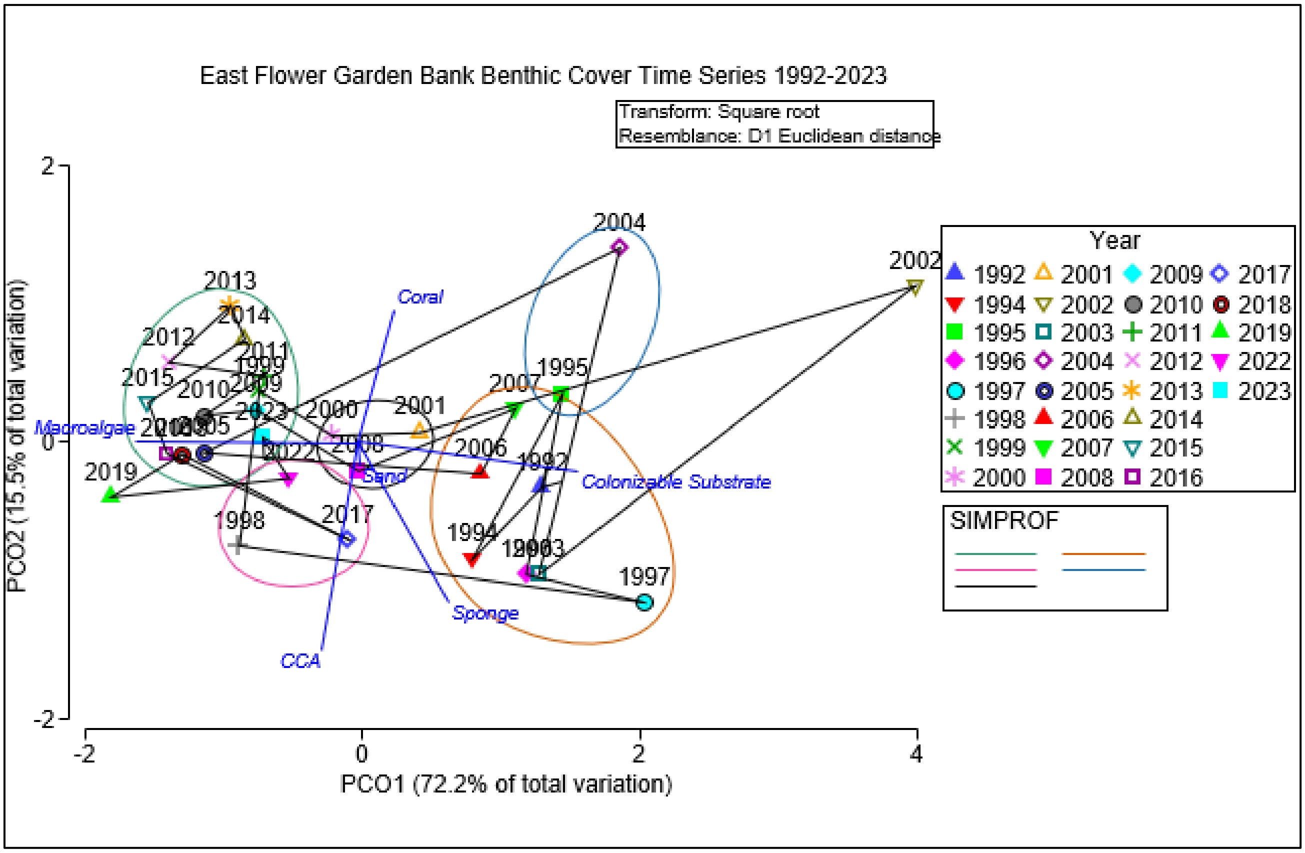


Figure 2.7. PCO plot for random transect benthic cover analysis within the EFGB study site from 1992–2023. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

Yearly mean benthic percent cover from 1992 to 2023 at the WFGB study site displayed a similar pattern to EFGB, resulting in four significant year clusters (1992–1997, 1999–2000, 2006, and 2008; 1998, 2009–2016, 2018, and 2022–2023; 2001, 2004–2005, 2007, 2017, and 2019; 2002–2003; Figure 2.8). Macroalgae contributed to the majority of the dissimilarity among year group clusters.

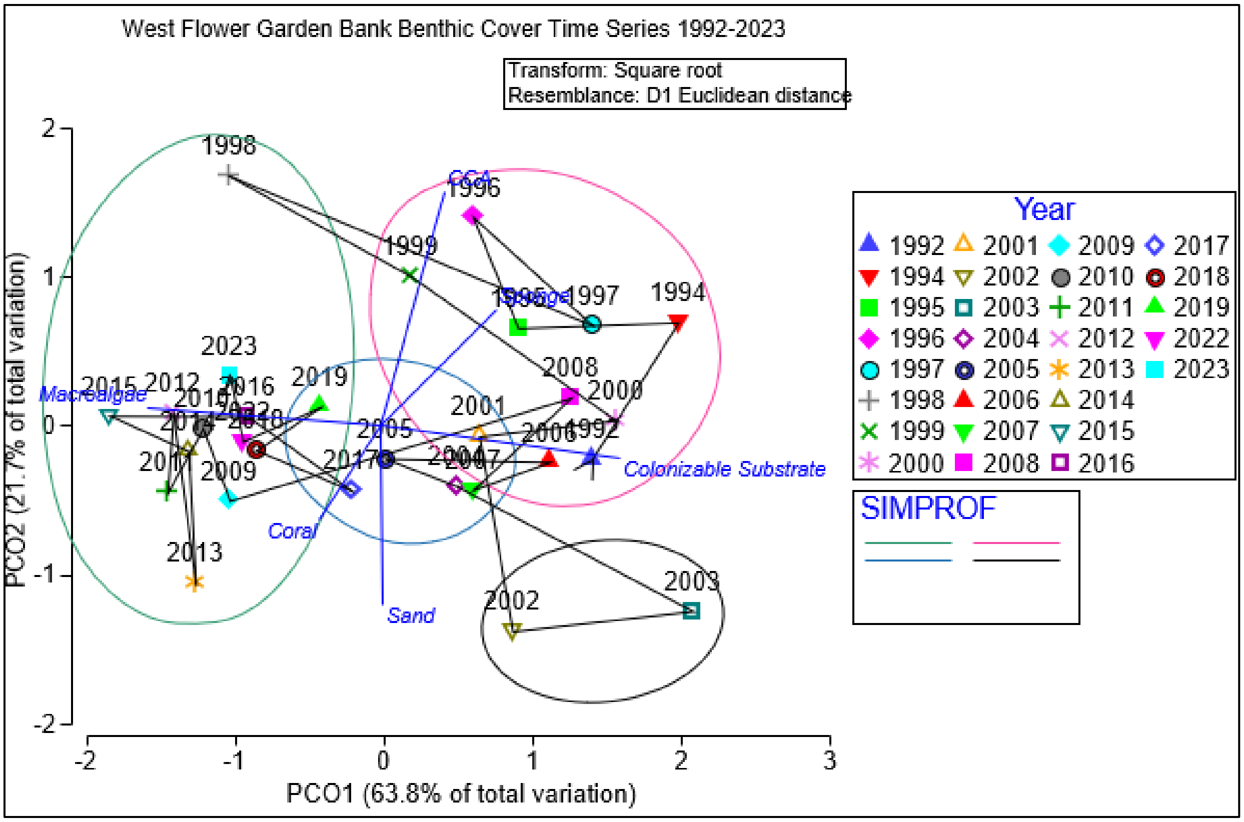


Figure 2.8. PCO plot for random transect benthic cover analysis within the WFGB study site from 1992 to 2023. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

PERMANOVA revealed no significant differences between study sites, suggesting that EFGB and WFGB study sites were similar to each other from 1992 to 2023 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 2023 (Figure 2.9; Table 2.4). Less than 1% of coral cover at EFGB and WFGB was affected by coral bleaching, paling, or fish biting in August 2023; however, coral bleaching did occur later in the year (see Chapter 3).

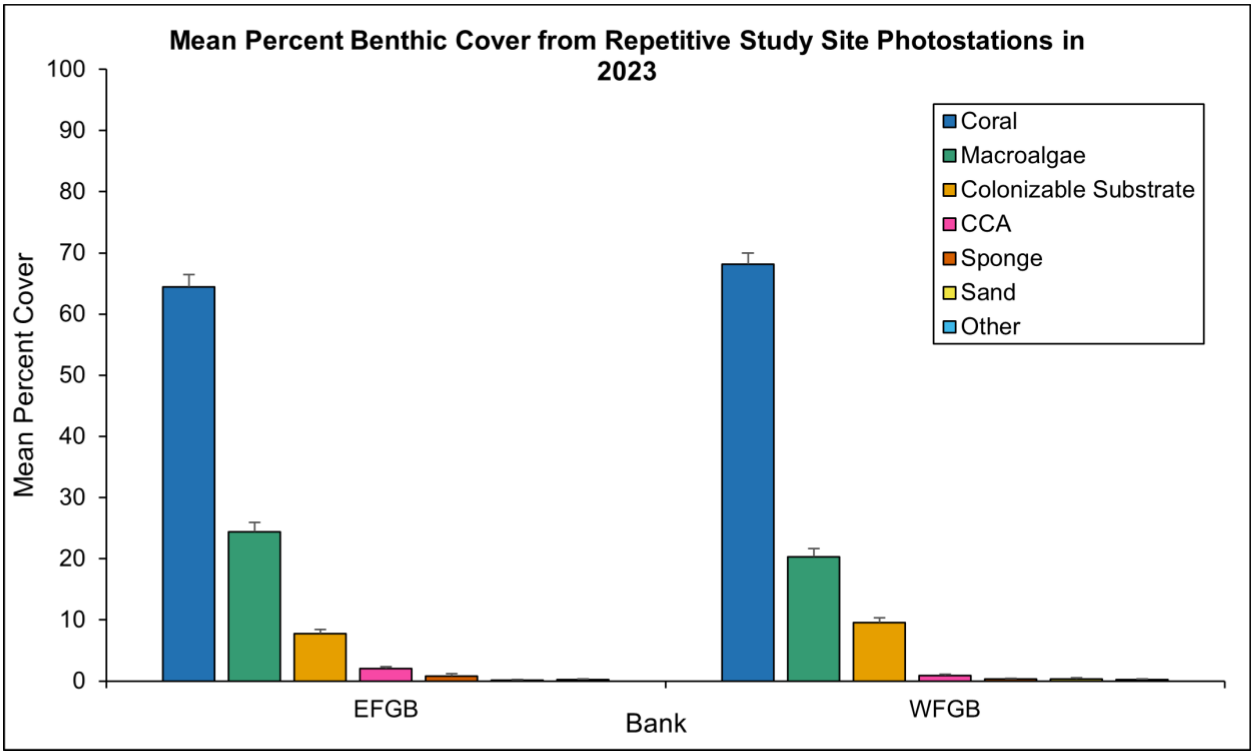


Figure 2.9. Mean percent benthic cover + SE within EFGB and WFGB repetitive photostations in 2023.

Table 2.4. Range of mean percent cover categories from EFGB and WFGB repetitive photostations, and all photostations combined in 2023.

Percent Cover Range	EFGB	WFGB	EFGB and WFGB combined
Coral	20.62–90.00%	30.30–92.13%	20.62–92.13%
Macroalgae	6.67–59.79%	4.12–46.87%	4.12–59.79%
CCA	0.00–10.31%	0.00–5.21%	0.00–10.31%
Colonizable substrate	0.00–26.80%	1.12–40.25%	1.12–40.25%
Sponge	0.00–22.22%	0.00–3.13%	0.00–22.22%
Sand	0.00–2.22%	0.00–3.06%	0.00–3.06%
Other	0.00–2.06%	0.00–2.13%	0.00–2.13%

The repetitive photostations ranged in depth from 18–39 m at EFGB (averaging 25 m depth) and 20–38 m at WFGB (averaging 26 m depth). Mean percent benthic coral cover categories ranged widely among the photostations (by about 70% at each bank; Table 2.5).

Fifteen coral species were recorded in EFGB repetitive photostations and 14 were recorded at WFGB. *Orbicella franksi* was the dominant species at EFGB (36.92 ± 2.39%), followed by *P. strigosa* (7.62 ± 1.21%) and *Montastraea cavernosa* (5.68 ± 1.19%; Figure 2.10). *Orbicella franksi* was the dominant coral at WFGB (39.16 ± 2.58%), followed by *M. cavernosa* (6.83 ± 1.27%) and *P. strigosa* (6.69 ± 1.21%; Figure 2.11).

A significant correlation between percent coral cover and depth was found in the repetitive photostations at EFGB (Table 2.6) and WFGB (Table 2.7) in 2023, suggesting that coral cover within the photostations changed with depth.

Table 2.6. PERMANOVA results comparing 2023 coral species mean percent cover by depth in EFGB repetitive photostations. **Bold** text denotes significant value.

EFGB	Sum of Squares	df	Pseudo-F	P (perm)
Depth	31.586	1	7.9266	**0.0007**
Res	95.635	24		
Total	323.22	58		

Table 2.7 PERMANOVA results comparing 2023 coral species mean percent cover by depth in WFGB repetitive photostations. **Bold** text denotes significant value.

WFGB	Sum of Squares	df	Pseudo-F	P (perm)
Depth	12.995	1	3.3691	**0.0245**
Res	123.7	32		
Total	273.52	62		

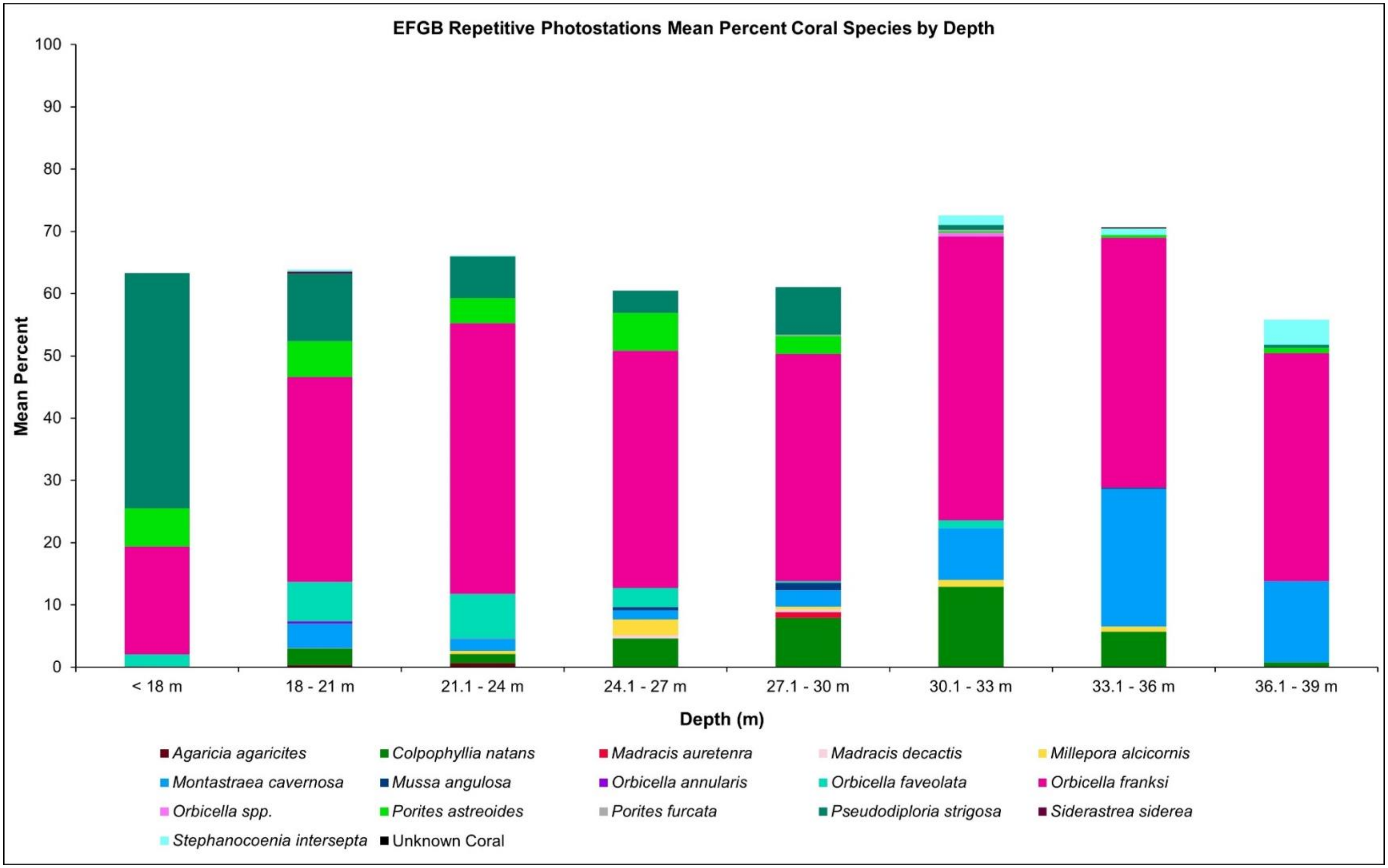


Figure 2.10. Mean percent cover of coral species at EFGB repetitive photostations by depth in 2023.

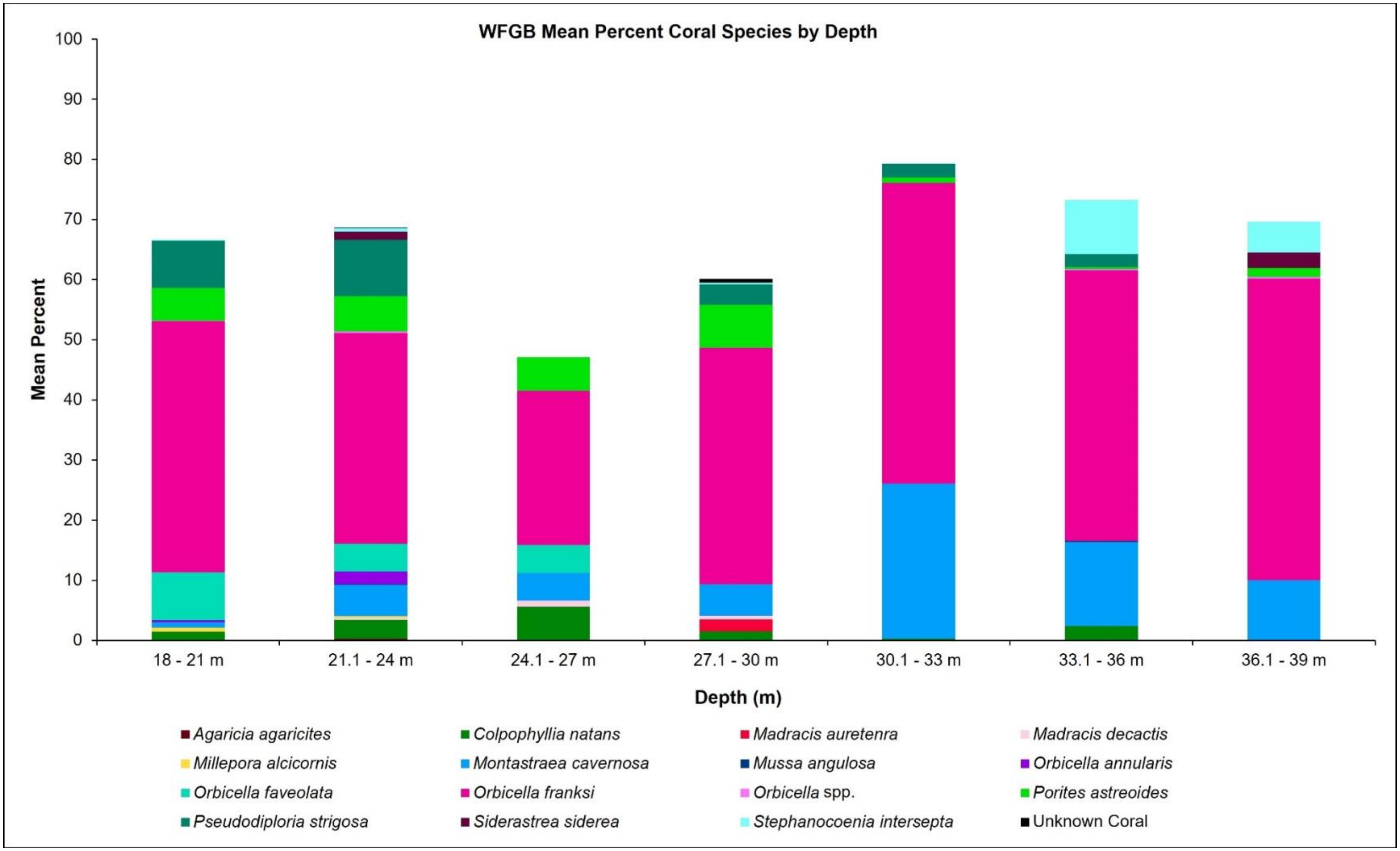


Figure 2.11. Mean percent cover of coral species at WFGB repetitive photostations by depth in 2023..

Qualitative Analysis of Repetitive Photostations and Coral Disease

Coral disease-like lesions (2.1–2.6% prevalence) were observed on seven coral species during EFGB and WFGB surveys in August and September 2022 (Johnston et al., 2023). The repetitive photostations were invaluable for documenting the disease outbreak on impacted colonies. As described in Johnston et al. (2023), the total number of colonies within each photostation were summed, and the number of colonies with lesions was recorded. Marginal and/or multi-focal lesions and tissue loss were observed, affecting the dominant coral species *Pseudodiploria strigosa* (7–8% prevalence), *Colpophyllia natans* (11–18% prevalence), and *Orbicella* spp. (1% prevalence). In 2023, few observations of the suspected disease were observed, and many corals showed signs of recovery, though some mortality was observed (Figure 2.13). A qualitative comparison of EFGB and WFGB repetitive photostations from 2022 to 2023 was made to assess changes of coral and sponge health and coverage of each photostation. Changes were categorized into five color grades:

- Green: a majority of corals and/or sponges grew and their cover increased
- Blue: little to no growth or degradation was observed
- Yellow: a majority of corals were degraded or a prolific colony lost a notable amount of cover
- Orange: a majority of sponges were degraded or a prolific sponge lost a notable amount of cover
- Red: a majority of sponges and corals were degraded and their cover decreased.

From 2022 to 2023, EFGB repetitive stations were primarily unchanged (blue, 38%) or exhibited coral degredation (yellow, 32%), while over half of WFGB repetitive stations were unchanged (blue, 51%), and over a quarter exhibited decreased sponge cover (orange, 29%; Figure 2.12). There were no photostations at EFGB or WFGB that exhibited both coral and sponge cover decreases (red).

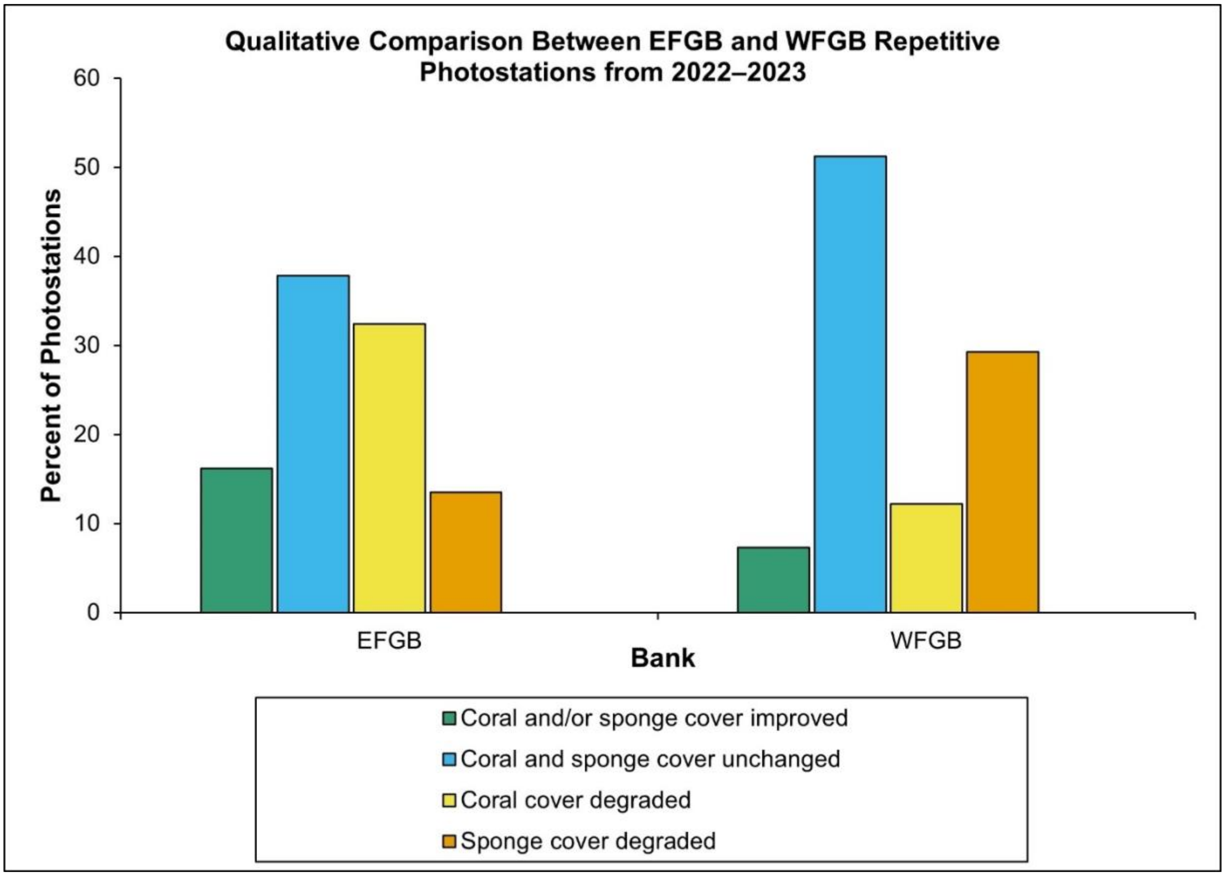


Figure 2.12. Qualitative comparison of photostation coral and sponge cover change from 2022–2023 at EFGB and WFGB.

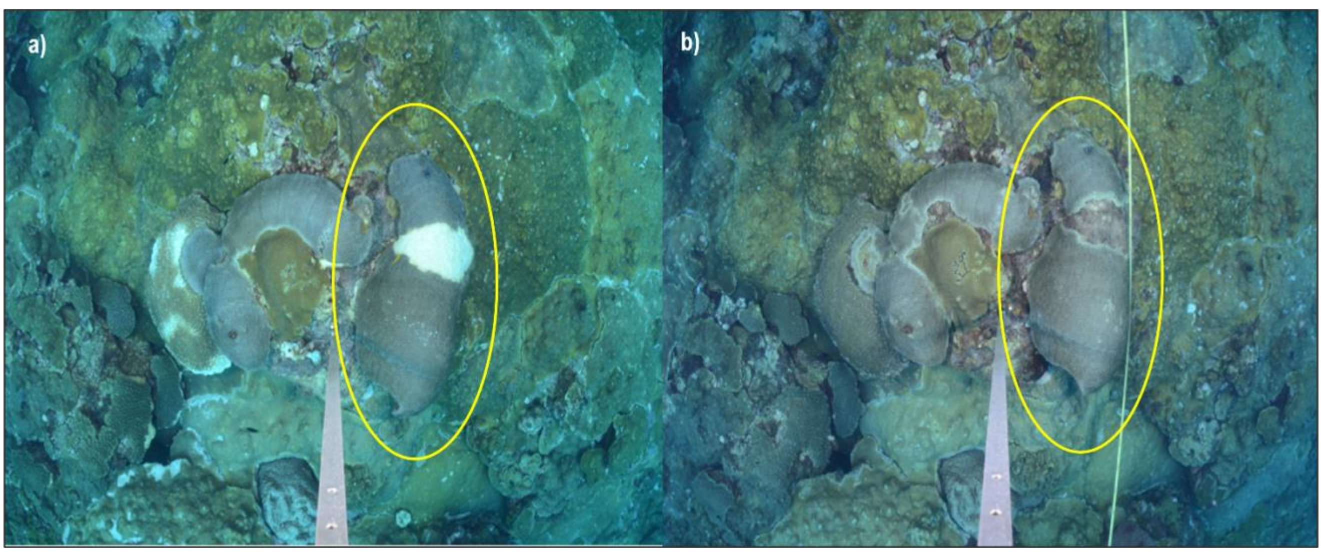


Figure 2.13 EFGB repetitive photostation 301 in (a) 2022 and (b) 2023. In 2022, a disease lesion was visible on the *P. strigosa* colony circled in yellow. By 2023, the lesion was absent, resulting in visible mortality in the affected area. Photo: Donavon French/CPC

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 1989 to 2023. Mean percent coral cover was 58.72 ± 3.80% in 1989 and 66.31 ± 2.73% in 2023 among the 24 EFGB photostations (though not a significant difference). It increased significantly at the 27 WFGB photostations (50.30 ± 3.06 % in 1989 to 69.07 ± 2.25 % in 2023 (Figure 2.14).

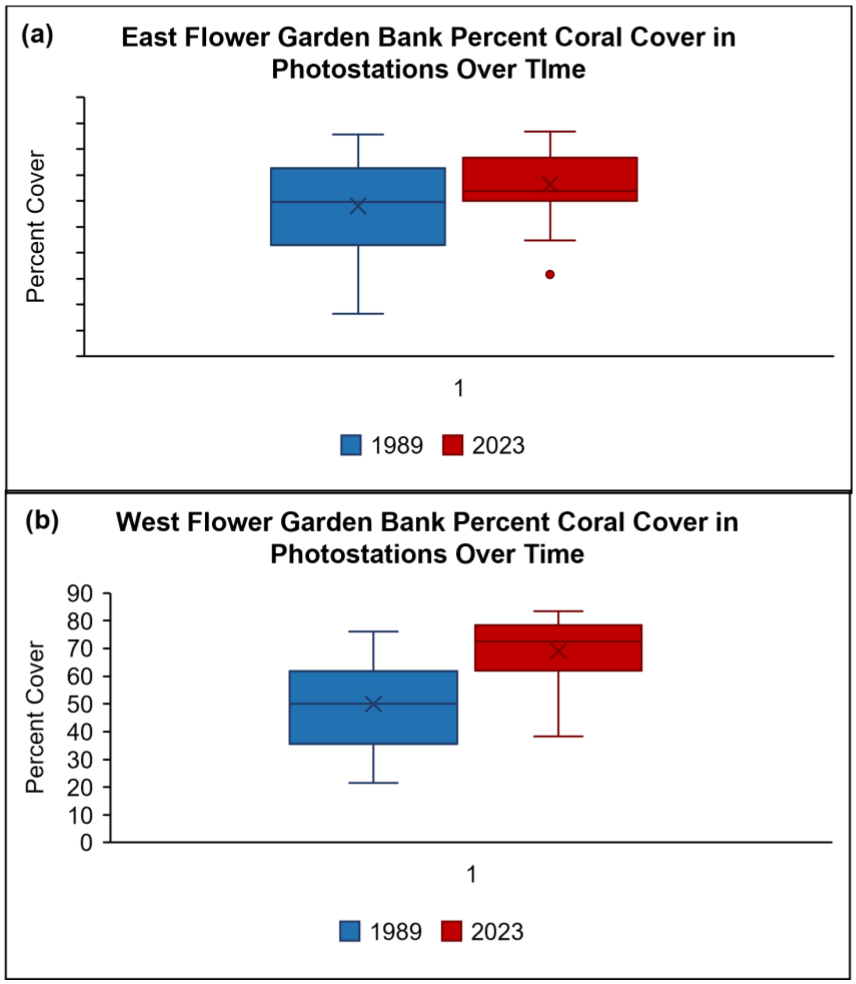


Figure 2.14. Box plot depicting percent coral cover in (a) EFGB (n = 24) and (b) WFGB (n = 27) repetitive photostations in 1989 and 2023.

As an example of the value of long-term repetitive photographs, Figure 2.15 documents changes in three 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 photostation #102, changes from 1989 to 2023 include recruitment and growth of *P. strigosa* and *P. astreoides* in the center (Figure 2.15a; Figure 2.15b). This photostation represents an extreme example of increased coral cover, but clearly demonstrates long-term changes that may be difficult to detect in the short term. In WFGB photostation #501, *O. franksi* cover increased from 1989 to 2023, and a black *Ircinia strobilina* sponge that was present in 1989 was absent in 2023 (Figure 2.15c; Figure 2.15d). In WFGB photostation #503, a large *C. natans* colony present in 1989 was absent in 2023 (Figure 2.15e; Figure 2.15f).

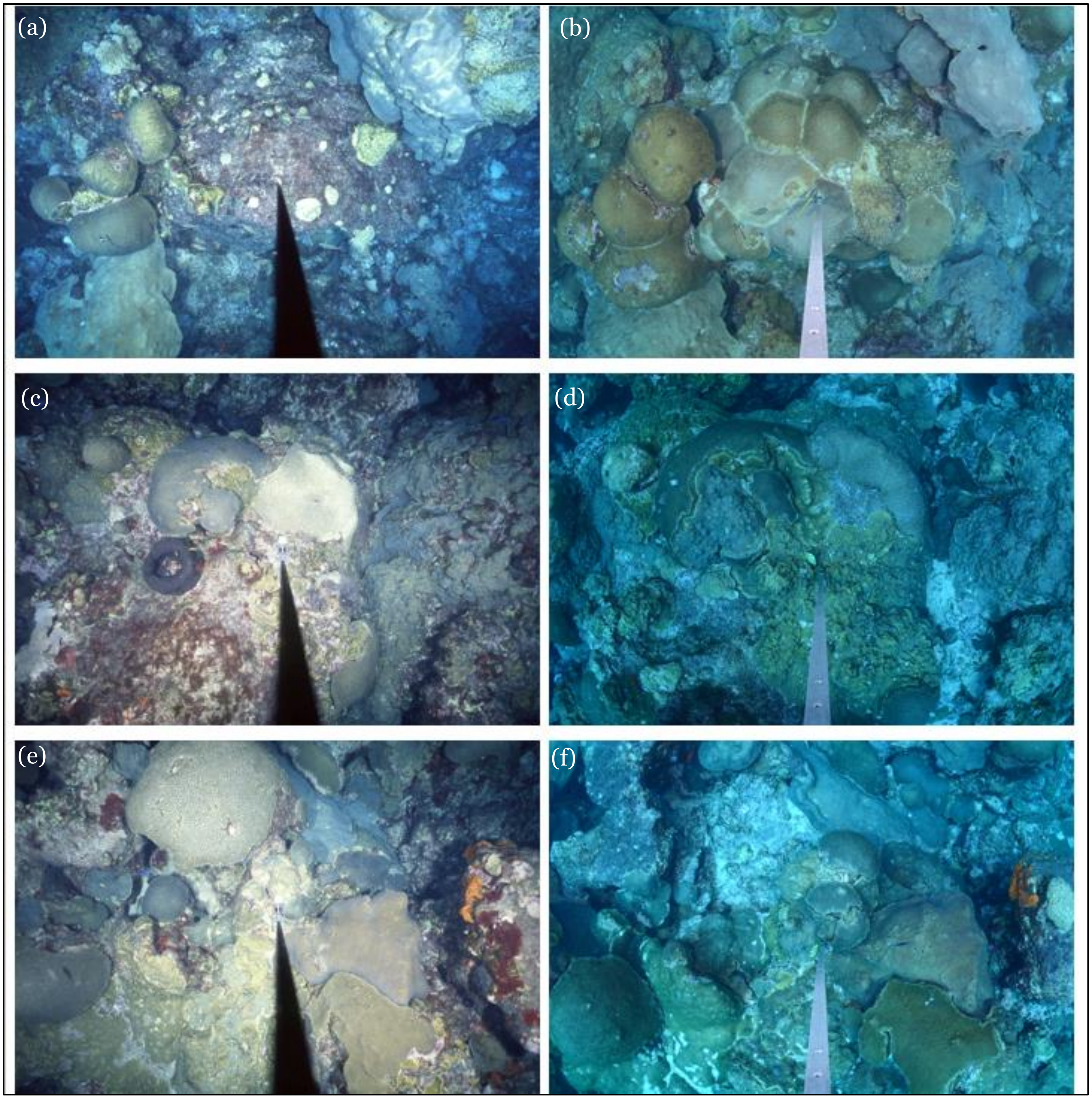


Figure 2.15. Time series of three repetitive photostations: EFGB photostation #102 (20 m) from (a) 1989 to (b) 2023; WFGB photostation #501 (20 m) from (c) 1989 to (d) 2023; and WFGB photostation #503 (20 m) from (e) 1989 to (f) 2023. Photos: (a, c, e) Minerals Management Service (Gittings et al., 1992); (b, d, f) NOAA

Discussion

Despite global coral reef declines in recent decades, coral cover within EFGB and WFGB study sites has remained consistently high, near or above 50% throughout 34 years of monitoring. Initial analysis indicated an increase in macroalgae cover from approximately 5% to 20% between 1998 and 1999. However, reanalysis using current CPCe methods revealed no change in macroalgae cover at EFGB (remaining at 30%) and a decrease from 30% to 20% at WFGB during this period. Over the past decade, macroalgae cover has returned to approximately 30%. It is noteworthy, however, that transect images from the Flower Garden Banks prior to the die-off of *Diadema antillarum* sea urchins in 1983 show very little macroalgae cover, suggesting that increases may have happened following the decline of this significant herbivore.

Despite the higher algae cover, the inverse relationship between macroalgae and colonizable substrate suggests that macroalgae has expanded over exposed hard bottom rather than outcompeting coral or sponges. Thus, unlike many shallow water reefs in the Caribbean and worldwide, this increase in macroalgae has not been associated with a decline in coral cover at EFGB and WFGB (Gardner et al., 2003; Mumby & Steneck, 2011; DeBose et al., 2012; Jackson et al., 2014; Johnston et al., 2016a, 2017a, 2017b, 2018a, 2020, 2022, 2024).

While a localized mortality event occurred in July 2016 outside the EFGB study site (approximately 275 meters away) and both banks experienced coral bleaching in the fall of 2016 and 2021, neither event resulted in significant declines in coral cover within the study sites (Johnston et al., 2018b, 2019, 2022). Bleaching was observed again in September 2023, but mortality is expected to be low, as signs of recovery were noted in November 2023.

Seventeen coral species were observed in random and reef-wide transect surveys at EFGB and WFGB in 2023, while 14 species were observed in repetitive photostations. Species observed in random transects but not in repetitive photostations included *Agaricia fragilis, Scolymia cubensis*, and *Siderastrea radians*. Conversely, *Madracis auretenra* was observed in repetitive photostations but not in random transects. Though coral cover was significantly higher in repetitive photostations (68%, at depths ranging from 18–39 m) than in random transects (55%, at depths from 19–23 m) at both banks, this is because the photostations were selectively established in high-cover areas of the reefs. They were not intended for use in estimating reef cover or to conduct community analysis. Randomly located benthic transects are used for that purpose; repetitive photostations allow tracking of individual corals and processes (competition, overgrowth, growth rates, etc.) over time while controlling for small-scale environmental heterogeneity.

Despite disturbances such as bleaching events, hurricanes, and disease outbreaks, no overall decline in coral cover has been observed at EFGB and WFGB since 1989. In fact, coral cover in the study sites has gradually increased over time. Coral cover at FGBNMS is six to 11 times higher than in selected locations across the Caribbean (Caldow et al., 2009; Clark et al., 2014; Jackson et al., 2014; Johnston et al., 2017a, 2017b). Coral resistance and resilience at the Flower Garden Banks may be attributed to the banks' offshore, remote locations and the surrounding deep water, which creates a cleaner, more stable environment compared to shallow, coastal reefs (Aronson et al., 2005; Johnston et al., 2015). Additionally, long-term protections from human stressors—such as oil and gas activities, vessel discharges, and anchoring—have contributed to maintaining favorable environmental quality at these sites.

However, despite their remoteness and depth, EFGB and WFGB are not immune to impacts associated with human activity. The localized mortality event in 2016, the increasing frequency of bleaching events, and the 2022 disease outbreak may have ties to human activities and links to climate change (Johnston et al., 2018b, 2019, 2023). Continued threats such as climate change, invasive species, and water quality degradation remain concerns for FGBNMS resources (Office of National Marine Sanctuaries, 2008; Nuttall et al., 2014; Johnston et al., 2016b). As environmental conditions in the Gulf of Mexico evolve (Karnauskas et al., 2015), ongoing monitoring will be crucial to document ecosystem changes and support timely responses.

Chapter 3:
Coral Bleaching

A bleaching coral (*Montastraea cavernosa*) at EFGB. Photo: Donavon French/CPC

Bleaching Introduction

Mild to moderate coral bleaching events have been documented at FGBNMS in previous years (e.g., 1990, 1995, 2005, 2010, 2016, 2021) when seawater temperatures exceeded 30 °C. These events, however, did not result in significant coral mortality or substantial changes in overall mean coral cover (Hagman & Gittings, 1992; Continental Shelf Associates, 1996; Precht et al., 2008; Eakin et al., 2010; Zimmer et al., 2010; Johnston et al., 2013, 2019). Recently, the frequency of moderate bleaching events has increased, and climate models predict that shallow coral reefs (<30 m) will face severe thermal stress on an annual basis by 2040 (Johnston et al., 2019; Dias et al., 2023).

In late August 2023, corals at FGBNMS began to bleach and pale due to a prolonged period in which seawater temperature exceeded 30°C. Response cruises were conducted at EFGB and WFGB to photograph corals in repetitive study site photostations and in reef-wide random transects, documenting the extent of the event. Significant bleaching caused by high water temperatures occurred in other parts of the Western Atlantic, including Florida and the Caribbean, in summer 2023. This chapter highlights a comprehensive overview of the 2023 bleaching event, drawing on multiple response cruises (September 1–2; September 18–22; November 11–12) conducted to assess its extent.

Coral Bleaching Methods

Coral stress was assessed by measuring the prevalence of bleaching. Prevalence was measured in two ways: as the proportion of the total percent coral cover affected by bleaching or paling and as the proportion of colonies affected by bleaching or paling. The proportion of percent cover affected was calculated using CPCe software (see Benthic Community Methods: Random Transect Data Processing for detailed methods), where any point landing on white coral was classified as "bleached" and points landing on pale coral were classified as "paling." Data were then averaged across EFGB and WFGB and calculated for repetitive photostations and reef-wide random transects.

Prevalence by number of colonies affected was calculated only for repetitive photostations in early September by dividing the number of colonies exhibiting bleaching or paling by the total number of colonies observed. Prevalence was calculated for each photostation and averaged across each bank. This provided a measure of the proportion of stressed colonies in the EFGB and WFGB study sites, with a focus on early season coral stress.

No formal bleaching surveys were conducted during the November cruise; however, thermally resistant coral colonies were collected and stored at the Aquarium at Moody Gardens for preservation. Divers observed signs of bleaching and paling, as well as some indications of coral recovery.

Bleaching threshold curves were updated in 2023 to analyze thermal stress, based on the number of days corals were exposed to elevated temperatures (Johnston et al., 2019). To estimate time-temperature bleaching threshold curves (Berkelmans, 2002; Manzello et al., 2007) at EFGB and WFGB, daily mean seawater temperature at depth from 2009 to 2023 was examined, and days averaging between 29 °C to 31 °C were tallied in 0.1-degree °C increments.

Total days at each temperature increment were summed for both bleaching and non-bleaching years, and bleaching thresholds for each bank based on exposure time (number of days above high temperatures) during bleaching years were estimated and interpolated as polynomial bleaching curves for each bank. Data from 2009 to 2023 were utilized because there were no gaps in seawater temperature data, and 2010, 2016, 2021, and 2023 were documented bleaching years (Johnston et al., 2019). This analysis provided insight into the relationship between temperature exposure and the onset of bleaching events.

Coral Bleaching Results

Prevalence by Percent Cover

Analysis of repetitive photostations in early September 2023 indicated a moderate proportion of percent cover was affected by bleaching or paling at EFGB, and paling was more prominent than bleaching. At WFGB, both bleaching and paling were less widespread during this period. The data showed that 7.08% of coral cover at EFGB exhibited bleaching, while 21.57% showed signs of paling. In contrast, 1.48% of coral cover was bleached and 6.92% was pale at WFGB (Figure 3.1).

By mid-to-late September, when reef-wide random transects were conducted, coral stress had worsened, particularly at EFGB. The transects, which were conducted over a larger area of the reef, suggested an increase in both bleaching and paling. In September, 28.06% of coral cover was bleached in EFGB random transects, and 50.64% was pale, indicating a major bleaching event. Bleaching and paling also worsened at WFGB (5.61% bleached and 17.09% pale), but prevalence remained lower compared to EFGB (Figure 3.1).

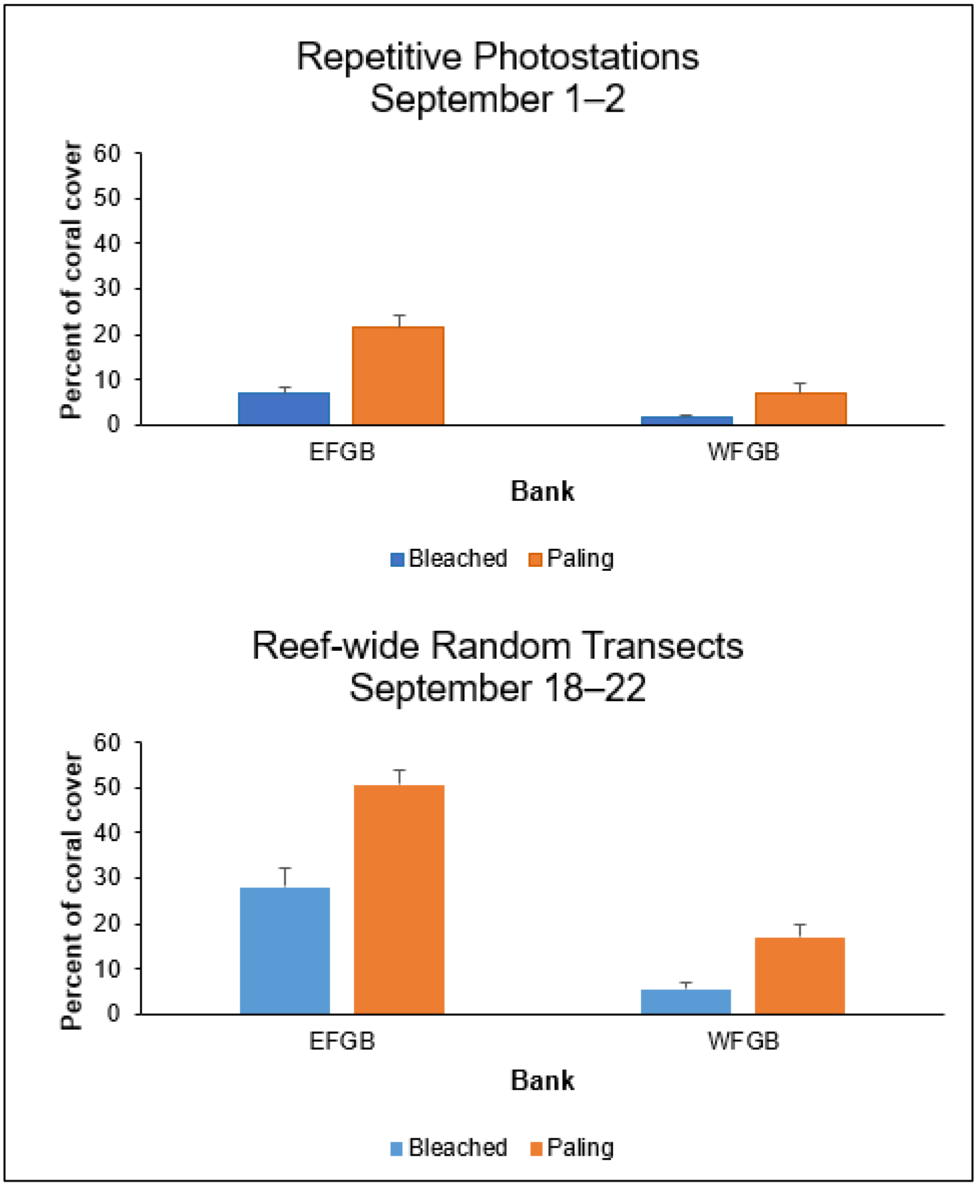


Figure 3.1. Percentage of coral cover affected by bleaching or paling at EFGB and WFGB. Data were collected from repetitive photostations on September 1–2, 2023 and from reef-wide random transects conducted on September 18–22, 2023.

Prevalence by Colony

The prevalence of coral bleaching and paling as a proportion of affected colonies in repetitive photostations was assessed at EFGB and WFGB during the early September cruise. At EFGB an average of 28.84 ± 2.57% of coral colonies were pale and 21.43 ± 2.07% were bleached (Figure 3.2). Prevalence by colony was lower at WFGB; 12.88 ± 0.95% of colonies were pale and, and the prevalence of bleaching at 2.16 ± 0.46% of colonies were bleached (Figure 3.2).

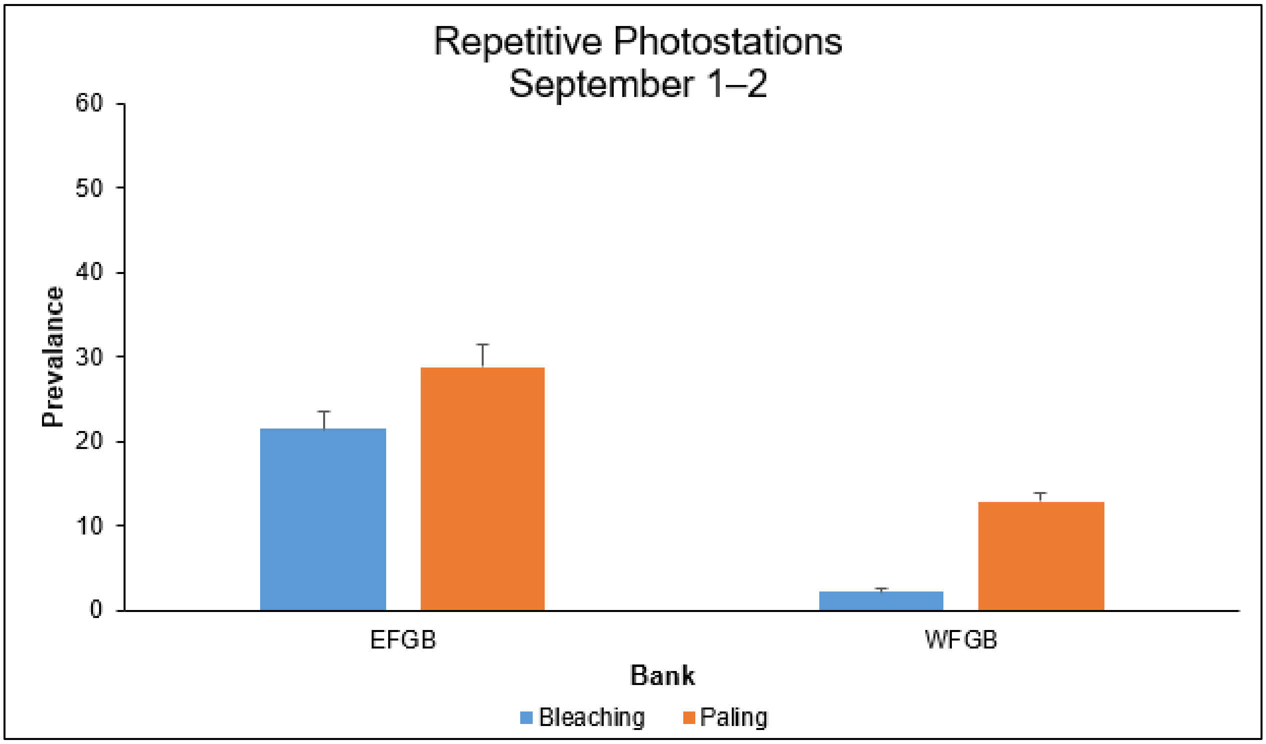


Figure 3.2. Prevalence of bleaching and paling, as a percentage of colonies affected, at EFGB and WFGB. Data were collected from repetitive photostations on September 1–2.

Bleaching Curves

Bleaching threshold curves illustrate the importance of both temperature and duration of exposure in triggering coral bleaching. Corals are sensitive not only to elevated temperatures but also to the length of time they remain above critical temperature thresholds. When examining bleaching threshold curves for each bank, the thermal stress indices calculated were able to effectively segregate bleaching years (2010, 2016, 2021, and 2023) from non-bleaching years at EFGB and WFGB (Figure 3.3). At EFGB, 33 days or more above 30 °C or 59.3 days above 29.5 °C would result in coral bleaching based on calculated thresholds. At WFGB, the threshold was slightly lower; a total of 22.3 days or more above 30 °C or 28.8 days above 29.9 °C would result in coral bleaching (Figure 3.3).

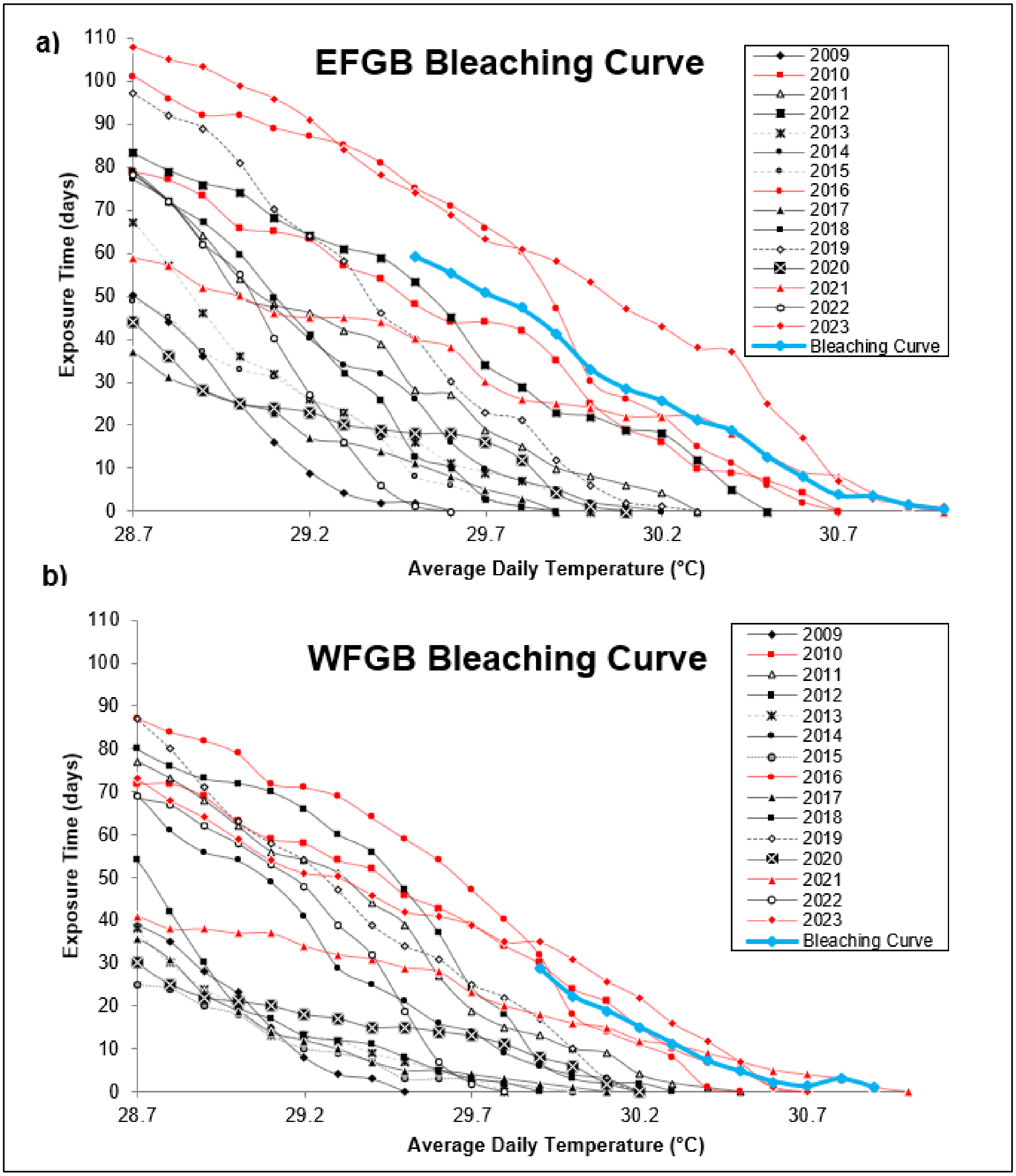


Figure 3.3. Bleaching threshold curves for (a) EFGB and (b) WFGB based on daily mean seawater temperature (°C) at depth (24 m) and exposure time (number of days). Red trend lines indicate bleaching years and black trend lines indicate non-bleaching years. Estimated bleaching curves are in blue.

By September 1, during the first bleaching reconnaissance cruise when repetitive photostations were photographed at EFGB and WFGB, EFGB had experienced 24 consecutive days at temperatures exceeding 30 °C. By September 18, when the reef-wide transects were conducted, EFGB had been exposed to temperatures above 30 °C for 41 consecutive days, surpassing the bleaching threshold of 33 days. In contrast, WFGB had experienced only five non-consecutive days above 30 °C by September 1. By September 18, this had increased to 21 non-consecutive days, approaching the bleaching threshold of 22.3 days for WFGB (Figure 3.3).

Discussion

The results indicate significant coral stress at EFGB, where the prevalence and severity of bleaching and paling were notably higher than at WFGB. Initial observations at EFGB revealed early signs of stress, which progressively worsened throughout the bleaching event. In contrast, WFGB exhibited milder conditions, with less prevalent bleaching.

Bleaching at both banks coincided with a prolonged period of elevated water temperatures. EFGB experienced 99 days with temperatures exceeding 29 °C, including 53 days above 30 °C. It wasn't until early October that temperatures at EFGB dipped below 30 °C, and they remained above 29 °C until mid-October, which likely exacerbated coral bleaching as September progressed. Similarly, WFGB experienced 59 days with temperatures exceeding 29 °C, including 31 days above 30 °C. Temperatures at WFGB did not drop below 29 °C until October 11. The differences in thermal exposure between the two banks likely contributed to the observed variation in bleaching and paling, as EFGB was subjected to more prolonged thermal stress.

These findings emphasize the significance of localized factors in influencing seawater temperatures, which determine coral bleaching outcomes. These often include depth and circulation patterns. But while the shallow photostations at WFGB are, on average, deeper than those at EFGB, the difference is slight, and temperature differences are more likely driven by oceanographic factors. Circulation patterns in the northern Gulf of Mexico, including water flow and mixing across banks, are strongly influenced by the strength and direction of prevailing currents (Jarosz et al., 2014; Muller-Karger et al., 2015; Johnston et al., 2019), with additional variation determined by the location and persistence of loop current eddies. These eddies can lead to substantial and persistent differences in water quality between the EFGB and WFGB, and can significantly influence bleaching severity.

Measuring bleaching and paling prevalence in terms of both percent cover affected and number of colonies affected helps to provide a comprehensive evaluation of coral reef health during a bleaching event. In non-stress years or during milder events, such as that recorded at WFGB, the differences between these methods are less pronounced. Incorporating prevalence assessments during bleaching years, however, allows for a more accurate representation of community-wide impacts and aids in guiding management decisions.

The 2023 bleaching event, just two years after the previous bleaching event, is consistent with climate model projections, which forecast an increase in the frequency and intensity of coral bleaching events on tropical reefs (Heron et al., 2016; Hughes et al., 2017, 2018; Dias et al., 2023). This trend is already apparent in other regions of the Western Atlantic, including Florida Keys National Marine Sanctuary and the greater Caribbean, where severe bleaching and coral mortality were also documented in 2023. The marine heatwave that devastated corals in these regions serves as a stark warning, emphasizing the urgent need to proactively protect the corals at FGBNMS. Protecting these reefs is critical, not only for the sanctuary's future but also for their potential role in supporting the recovery and restoration of neighboring reefs in the Gulf of Mexico and Caribbean.

Chapter 4: Sea Urchin and Lobster Surveys

A long-spined sea urchin (*Diadema antillarum*) atop brain coral (*Pseudodiploria strigosa*) at East Flower Garden Bank. Photo: Donavon French/CPC

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 at FGBNMS (Gittings & 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 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. One diver began on the right side of the line and the other on the left.

Divers swam slowly along the boundary line, each 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 recorded the number observed. In 2023, all lines were surveyed within the EFGB and WFGB study sites.

Consistency for the survey method was ensured by using multiple scientific divers trained to identify sea urchin and lobster species 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 data sheets 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

Sea urchins and lobsters observed on each 100-m line were summed for each study site. Each 100-m transect represented one sample. 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 was used to examine differences in density among years and one-hectare study sites with a Bray-Curtis similarity matrix and added dummy variable (value of 1). Covariance between sea urchin density and macroalgae percent cover was examined using coherence plots in PRIMER.

Results

Density of *D. antillarum* was 3.83 ± 0.44 individuals/100 m^2 within the EFGB study site and 40.92 ± 3.11 individuals/100 m^2 within the WFGB study site in 2023 (Figure 4.1). One *P. guttatus* was observed in the EFGB study site and no *P. argus* were observed (Figure 4.1).

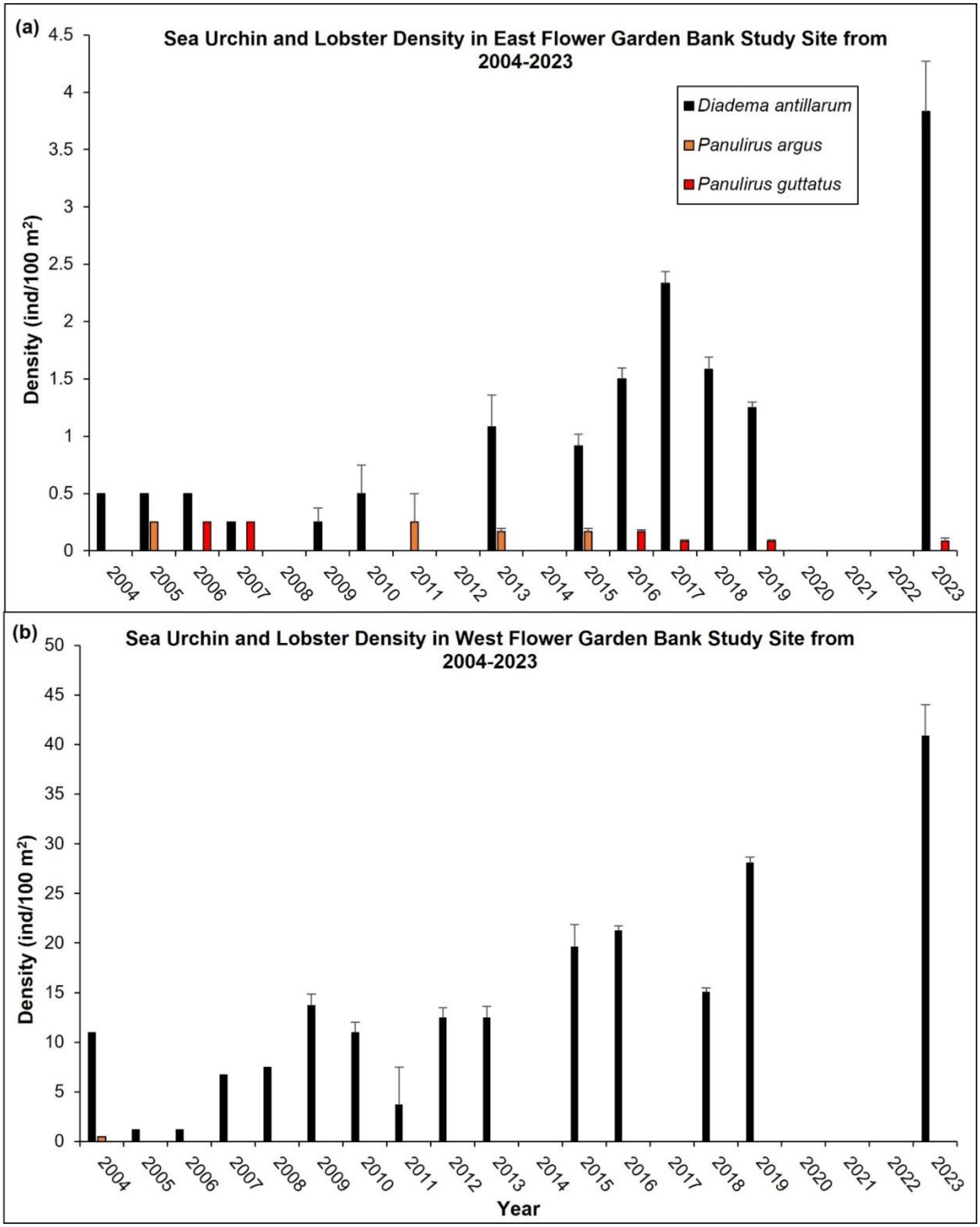


Figure 4.1. Mean sea urchin and lobster density (individuals/100 m^2) + SE within (a) EFGB and (b) WFGB study sites from 2004 to 2023 (note the difference in scale on the y-axis). No data were available for either bank in 2014, 2020, 2021, and 2022 and at WFGB in 2017. SE values were not available from 2004 to 2008. Sources: Precht et al., 2006; Zimmer et al., 2010 (2004 to 2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021, 2022, 2024 (2009 to 2018)

Since 2004, *D. antillarum* densities have ranged from 0–3.83 individuals/100 m² within the EFGB study site and 1.25–40.92 individuals/100 m² within the WFGB study site. Higher numbers of *D. antillarum* have been observed during surveys at the WFGB study site throughout the monitoring program (Figure 4.1). PERMANOVA showed that sea urchin density was significantly greater within the WFGB study site (pseudo-$F_{1,39}$ = 28.26, $p < 0.05$). Since 2004, lobster densities have ranged from 0–0.25 individuals/100 m² within the EFGB and WFGB study sites combined.

Due to the importance of *D. antillarum* as herbivores on coral reefs, sea urchin density and macroalgae percent cover for EFGB and WFGB were tested for covariance from 2004 to 2023. Coherence plots found significant covariance between urchin density and macroalgae percent cover at both EFGB and WFGB.

Discussion

Diadema antillarum is an important herbivore on coral reefs, helping to reduce macroalgae cover through grazing, which makes room for coral growth and new recruits (Edmunds & Carpenter, 2001; Carpenter & Edmunds, 2006). After the mass die-off in 1983, *D. antillarum* populations have not recovered to historic levels, which were at least 140 individuals/100 m² at EFGB and 50 individuals/100 m² at WFGB (Gittings & Bright, 1986, 1987; Gittings, 1998). Post-1983, *D. antillarum* densities dropped to near zero (Gittings & Bright, 1987). Since then, patchy but limited recovery has been documented in the Caribbean region (Edmunds & Carpenter, 2001; Kramer, 2003; Carpenter & Edmunds, 2006). *Diadema antillarum* densities at nearby Stetson Bank increased from 2009 to 2014 and have plateaued in recent years, averaging 118 individuals/100 m² in 2023 (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 study site remained low during the 2023 monitoring period and were similar to those reported in previous studies (Zimmer et al., 2010; Johnston et al., 2017a, 2017b, 2018a, 2020, 2021, 2024). Populations have been consistently higher within the WFGB study site than the EFGB study site. Regardless, the lack of correlation between sea urchin abundances and algae cover suggest that sea urchins are not currently exerting significant control over benthic algae, likely owing to a lack of full recovery to pre-die-off levels. Lobster densities within EFGB and WFGB 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 5: Fish Surveys

A school of gray snappers (*Lutjanus griseus*) and a chub (*Kyphosus sectatrix/incisor*) swim above the reef crest. Photo: Tiffany Crumbley/Crumbley Photography

Introduction

Divers conducted stationary reef fish visual surveys in EFGB and WFGB study sites to examine and compare fish community composition and changes over time.

Methods

Field Methods

Fishes were assessed by divers using the modified stationary reef fish visual census originally described by Bohnsack and Bannerot (1986). Twenty randomly located surveys were conducted within the study site at EFGB and 25 were conducted within the 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, refer to Johnston et al. [2017a]; Figure 5.1)

Figure 5.1. A NOAA diver conducts a fish survey at WFGB. Photo: Donavon French/CPC

All fish species observed within the first five minutes of the survey were recorded while the diver slowly rotated in place in the imaginary cylinder. Immediately following this 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. 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 centimeter. 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 2023 surveys. Fish surveys began in the early morning (after 0700 CDT), and were conducted throughout the day until dusk (1900 CDT).

Consistency in the survey method was ensured by using scientific divers who were trained to identify FGBNMS fish species and were experienced in the survey technique. Equipment checklists were provided in the field to ensure divers had equipment for assigned tasks, which included a pre-marked PVC measuring stick for size reference.

Data Processing

Surveyors reviewed and entered data in a Microsoft® Excel® database the same day the survey took place. Data sheets were retained, 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 & Harper, 1988; Froese & Pauly, 2019). Species were classified into four categories: herbivores (H), piscivores (P), invertivores (I), and planktivores (PL), as defined by NOAA's Center for Coastal Monitoring and Assessment BioGeography Branch fish-trophic level database (Caldow et al., 2009).

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 of a given species divided by the total of all species, multiplied by 100. Density was expressed as the number of individual fish ± SE per 100 m^2, and calculated as the total number of individuals per sample divided by the area of the survey cylinder (176.7 m^2) and multiplied by 100. Sighting frequency for each species was the percentage 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 kilograms ± SE per 100 m^2 and computed by converting length data to weight using the allometric length-weight conversion formula (Bohnsack & Harper, 1988) based on information provided by FishBase (Froese & 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:

$$CV\% = (SE/\bar{X}) \times 100$$

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 dispersion-weighted 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 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 and 2022 to 2023), 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) Monotonic trends in long-term grouper and snapper densities were assessed using the Mann-Kendall trend test in R version 4.2.2.

Dominance plots were generated based on species abundance and biomass with Primer® version 7.0 (Anderson et al., 2008; Clarke et al., 2014). For each survey, w-values (difference between the biomass and abundance curves) were calculated (Clarke, 1990). 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 study sites were assessed using ANOVA on untransformed data with Euclidean distance similarity matrices in R version 4.2.2 (Clarke et al., 2014).

Results

A combined total of 25 families and 74 species (66 at EFGB and 59 at WFGB) were observed in 2023 at both study sites. Mean species richness was 21.59 ± 0.89 per survey at EFGB, 19.92 ± 0.71 per survey at WFGB, and 20.69 ± 0.57 per survey for both study sites combined. Bonnetmouth (*Emmelichthyops atlanticus*) had the highest relative abundance of all species in EFGB surveys (34.80%), followed by brown chromis (*Azurina multilineata*; 14.92%), creole wrasse (*Bodianus parrae*; 10.05%), bluehead (*Thalassoma bifasciatum;* 7.91%), and Atlantic creolefish (*Paranthias furcifer*; 5.78%). In WFGB surveys, bonnetmouth had the highest relative abundance (65.72%), followed by brown chromis (16.40%), creole wrasse (2.53%), Atlantic creolefish (2.51%), and bluehead (2.21%).

Sighting Frequency and Occurrence

The most frequently sighted species was brown chromis, observed in 100% of surveys at EFGB and WFGB. Other frequently sighted species included Atlantic creolefish, blue tang (*Acanthurus coeruleus*), and bluehead (Table 5.1). One manta ray (*Mobula birostris*) and no sharks were observed in surveys; mantas and sharks are considered "rare," typically occurring in <20% of all surveys (Reef Environmental Education Foundation [REEF], 2014).

Table 5.1. Sighting frequencies for the 10 most frequently sighted species at EFGB and WFGB study sites in 2023.

Fish Species	EFGB	WFGB	Combined
Brown chromis (*Azurina multilineata*)	100.0%	100.0%	100.0%
Atlantic creolefish (*Paranthias furcifer*)	100.0%	100.0%	100.0%
Blue tang (*Acanthurus coeruleus*)	100.0%	88.5%	93.8%
Bluehead (*Thalassoma bifasciatum*)	100.0%	88.5%	93.8%
Bicolor damselfish (*Stegastes partitus*)	95.5%	88.5%	91.7%
Creole wrasse (*Bodianus parrae*)	95.5%	84.6%	89.6%
Black durgon (*Melichthys niger*)	90.9%	84.6%	87.5%
Stoplight parrotfish (*Sparisoma viride*)	72.7%	84.6%	79.2%
Queen parrotfish (*Scarus vetula*)	81.8%	76.9%	79.2%
Great barracuda (*Sphyraena barracuda*)	50.0%	88.5%	70.8%

Density

Mean fish density (individuals/100 m²) was 174.85 ± 31.44 in EFGB surveys, 301.66 ± 68.88 in WFGB surveys, and 243.54 ± 40.67 for all study site surveys combined. Density was significantly greater in WFGB surveys (Table 5.2). SIMPER analysis identified greater abundance of great barracuda (5.33%) at WFGB and greater abundance of blue tang (*Acanthurus coeruleus*; 5.40%) and redband parrotfish (*Sparisoma aurofrenatum*; 5.24%) at EFGB as the main contributors to the differences between banks (Table 5.3).

Table 5.2**.** 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	6453	1	3.63	**0.0002**
Res	84677	46		
Total	88130	47		

Table 5.3. Mean density (individuals ± SE/100 m^2) of the 20 most abundant species from EFGB and WFGB study site surveys in 2023. The density of **bolded** species was significantly different between EFGB and WFGB.

Fish Species	EFGB	WFGB	Combined
Bonnetmouth (*Emmelichthyops atlanticus*)	58.5 ± 28.6	197.0 ± 65.3	133.5 ± 38.7
Brown chromis (*Azurina multilineata*)	25.1 ± 3.7	49.2 ± 13.0	38.1 ± 7.4
Creole wrasse (*Bodianus parrae*)	22.6 ± 5.7	7.8 ± 2.4	14.6 ± 3.1
Bluehead (*Thalassoma bifasciatum*)	13.5 ± 2.6	7.1 ± 1.2	10.0 ± 1.4
Atlantic creolefish (*Paranthias furcifer*)	10.2 ± 3.5	7.9 ± 1.5	8.9 ± 1.8
Bicolor damselfish (*Stegastes partitus*)	7.6 ± 1.6	4.1 ± 0.7	5.7 ± 0.9
Blue chromis (*Azurina cyanea*)	4.0 ± 1.5	3.1 ± 0.7	3.5 ± 0.8
Threespot damselfish (*Stegastes planifrons*)	3.3 ± 0.9	1.8 ± 0.6	2.5 ± 0.5
Chub complex (*Kyphosus* spp.)	4.2 ± 1.3	0.8 ± 0.4	2.4 ± 0.7
Blue tang (*Acanthurus coeruleus*)	2.3 ± 0.3	1.9 ± 0.2	2.1 ± 0.2
Sharpnose puffer (*Canthigaster rostrata*)	2.8 ± 0.6	1.2 ± 0.2	1.9 ± 0.3
Queen parrotfish (*Scarus vetula*)	2.2 ± 0.5	1.5 ± 0.2	1.8 ± 0.3
Yellowhead wrasse (*Halichoeres garnoti*)	1.3 ± 0.4	1.6 ± 0.4	1.5 ± 0.3
Stoplight parrotfish (*Sparisoma viride*)	1.4 ± 0.3	1.4 ± 0.2	1.4 ± 0.2
Purple reeffish (*Chromis scotti*)	1.1 ± 0.6	1.4 ± 0.6	1.3 ± 0.4
Black durgon (*Melichthys niger*)	1.1 ± 0.2	1.3 ± 0.2	1.2 ± 0.1
Princess parrotfish (*Scarus taeniopterus*)	1.7 ± 0.4	0.7 ± 0.3	1.2 ± 0.2
Great barracuda (*Sphyraena barracuda*)	0.4 ± 0.1	1.7 ± 0.2	1.1 ± 0.2
Sunshinefish (*Chromis insolata*)	0.3 ± 0.1	1.7 ± 1.1	1.0 ± 0.6
Redband parrotfish (*Sparisoma aurofrenatum*)	1.6 ± 0.3	0.5 ± 0.1	1.0 ± 0.2
Mean density	174.85 ± 31.44	301.66 ± 68.88	243.54 ± 40.67

Trophic Guild Analysis

Size-frequency distributions using relative abundance were graphed for each of the four assigned trophic guilds (herbivores, piscivores, invertivores, and planktivores; Figure 5.2). Piscivores dominated the small size class at both study sites. Planktivores dominated the mid-range size classes. Invertivores and herbivores were more variable across mostly mid-size classes (Figure 5.2).

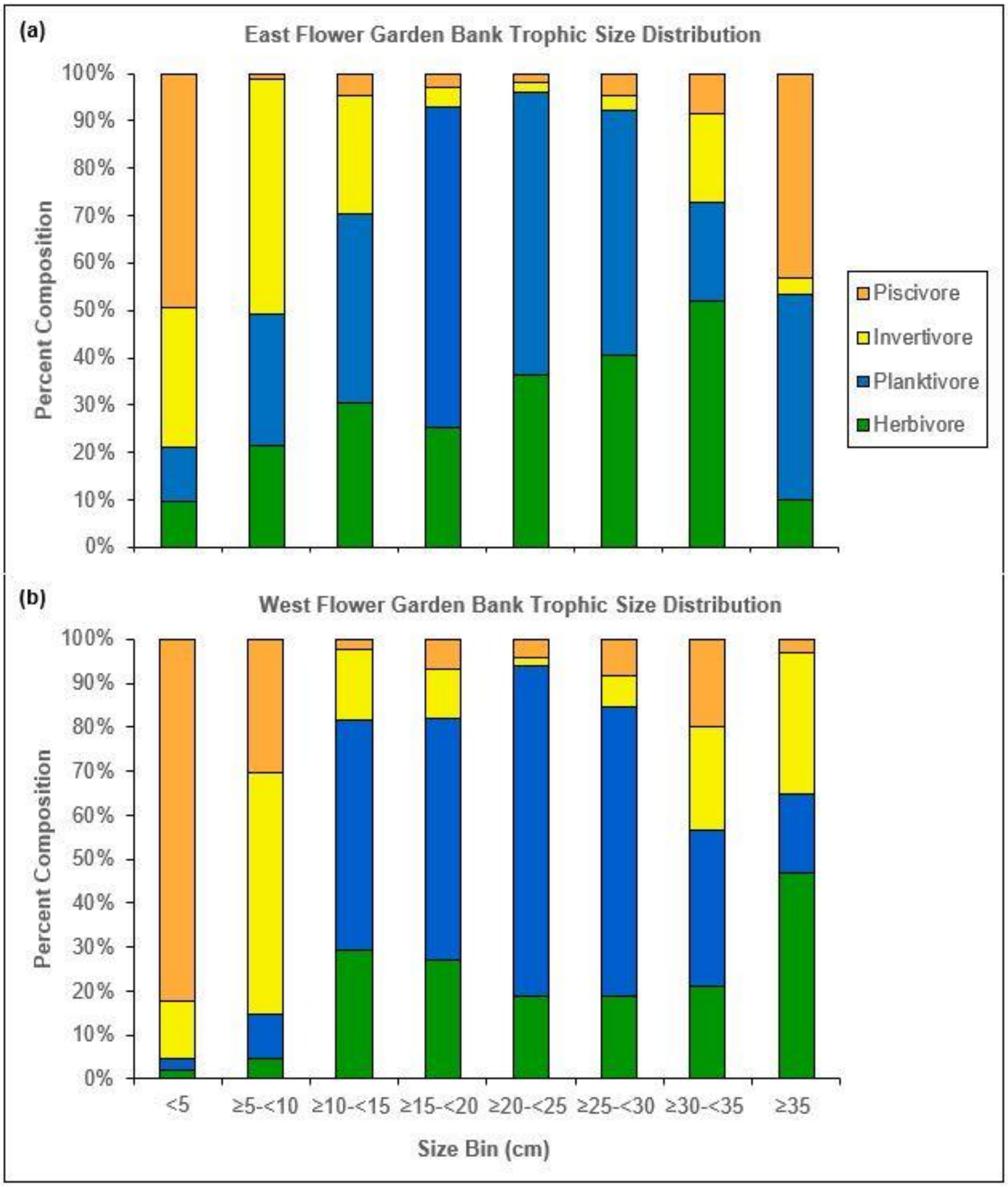


Figure 5.2. Fish size distribution by trophic guild at (a) EFGB and (b) WFGB study sites in 2023.

Biomass

Mean biomass (kg ± SE/100 m^2) was 5.93 ± 1.17 in EFGB surveys, 5.74 ± 0.84 in WFGB surveys, and 5.83 ± 0.69 for study site surveys combined in 2023. PERMANOVA analysis revealed that fish communities were significantly different between banks based on biomass distribution across species (Table 5.4). SIMPER analysis identified the main contributor to differences were a greater local biomass of blue tang (6.31%) at EFGB and great barracuda the WFGB (5.05%).

Table 5.4. PERMANOVA results comparing fish communities based on biomass at EFGB and WFGB study sites in 2023. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	5234	1	1.87	**0.0103**
Res	1.2863E+05	46		
Total	1.3386E+05	47		

When classified by trophic guild, herbivores possessed the highest mean biomass for all surveys; invertivores had the lowest mean biomass (Table 5.5). Significant differences were found among trophic guilds between study sites. Overall, herbivores represented approximately 38% of biomass, followed by planktivores (29%), piscivores (18%), and invertivores (15%) for study sites combined.

Table 5.5. Mean biomass (kg ± SE/100 m^2) for each trophic guild from EFGB and WFGB study site surveys, and surveys from both banks combined, in 2023.

Trophic Guild	EFGB	WFGB	Combined
Herbivore	2.66 ± 0.57	1.81 ± 0.35	2.20 ± 0.32
Invertivore	0.47 ± 0.10	1.25 ± 0.25	0.89 ± 0.13
Planktivore	2.30 ± 0.49	1.18 ± 0.23	1.69 ± 0.24
Piscivore	0.50 ± 0.11	1.50 ± 0.29	1.04 ± 0.15

Mean biomass for each species, grouped by trophic guild, is presented in Table 5.6. At the EFGB study site, 37% of herbivore biomass was contributed by the chub complex (*Kyphosus* spp.). For invertivores, the greatest contribution was from brown chromis (11%). For piscivores, great barracuda contributed the greatest biomass (42%). For planktivores, the greatest contribution was from Atlantic creolefish (53%; Table 5.6).

At the WFGB study site, 33% of herbivore biomass was contributed by the chub complex. For invertivores, the greatest contribution was from ocean triggerfish (*Canthidermis sufflamen*; 63%). For piscivores, great barracuda contributed the greatest biomass (67%). For planktivores, the greatest contribution was from creole wrasse (9%; Table 5.6).

Table 5.6 Biomass (kg ± SE/100 m^2) of each species grouped by trophic guild, from EFGB and WFGB study site surveys, and surveys from both banks combined, in 2023.

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Herbivore	Chub complex (*Kyphosus* spp.)	0.99 ± 0.30	0.60 ± 0.52	0.78 ± 0.31
Herbivore	Stoplight parrotfish (*Sparisoma viride*)	0.51 ± 0.13	0.39 ± 0.09	0.45 ± 0.08
Herbivore	Black durgon (*Melichthys niger*)	0.46 ± 0.11	0.32 ± 0.11	0.39 ± 0.08
Herbivore	Queen parrotfish (*Scarus vetula*)	0.17 ± 0.04	0.19 ± 0.05	0.18 ± 0.03
Herbivore	Blue tang (*Acanthurus coeruleus*)	0.13 ± 0.03	0.11 ± 0.02	0.12 ± 0.02
Herbivore	Princess parrotfish (*Scarus taeniopterus*)	0.13 ± 0.04	0.09 ± 0.03	0.11 ± 0.02
Herbivore	Redband parrotfish (*Sparisoma aurofrenatum*)	0.17 ± 0.07	0.05 ± 0.02	0.10 ± 0.03
Herbivore	Doctorfish (*Acanthurus chirurgus*)	0.05 ± 0.02	0.01 ± 0.00	0.03 ± 0.01
Herbivore	Bicolor damselfish (*Stegastes partitus*)	0.01 ± 0.004	0.01 ± 0.003	0.01 ± 0.002
Herbivore	Ocean surgeonfish (*Acanthurus tractus*)	0.01 ± 0.01	0.01 ± 0.003	0.01 ± 0.003
Herbivore	Yellowtail damselfish (*Microspathodon chrysurus*)	0.01 ± 0.005	0.01 ± 0.002	0.01 ± 0.002
Herbivore	Striped parrotfish (*Scarus iseri*)	0.01 ± 0.005	0.01 ± 0.004	0.01 ± 0.003
Herbivore	Greenblotch parrotfish (*Sparisoma atomarium*)	0.002 ± 0.002	0.01 ± 0.004	0.01 ± 0.002
Herbivore	Cocoa damselfish (*Stegastes variabilis*)	0.003 ± 0.001	0.004 ± 0.002	0.003 ± 0.001
Herbivore	Bucktooth parrotfish (*Sparisoma radians*)	0.003 ± 0.003	<0.001 ± <0.001	0.002 ± 0.001
Herbivore	Parrotfish spp. (*Scarus* spp./*Sparisoma* spp.)	0.001 ± 0.001	0.001 ± 0.001	0.001 ± 0.001
Herbivore	Dusky damselfish (*Stegastes adustus*)	0.001 ± 0.001	<0.001 ± <0.001	<0.001 ± <0.001
Herbivore	Redlip blenny (*Ophioblennius macclurei*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Invertivore	Ocean triggerfish (*Canthidermis sufflamen*)	0.04 ± 0.04	0.79 ± 0.36	0.44 ± 0.20
Invertivore	Brown chromis (*Azurina multilineata*)	0.05 ± 0.02	0.13 ± 0.04	0.09 ± 0.02
Invertivore	Gray snapper (*Lutjanus griseus*)	0.05 ± 0.03	0.08 ± 0.04	0.07 ± 0.03
Invertivore	Reef butterflyfish (*Chaetodon sedentarius*)	0.03 ± 0.01	0.05 ± 0.02	0.04 ± 0.01
Invertivore	Queen angelfish (*Holacanthus ciliaris*)	0.01 ± 0.01	0.04 ± 0.04	0.03 ± 0.02
Invertivore	Threespot damselfish (*Stegastes planifrons*)	0.03 ± 0.01	0.02 ± 0.01	0.02 ± 0.01
Invertivore	Smooth Ttunkfish (*Lactophrys triqueter*)	0.04 ± 0.03	0.01 ± 0.004	0.02 ± 0.01
Invertivore	Rock beauty (*Holacanthus tricolor*)	0.01 ± 0.01	0.02 ± 0.01	0.02 ± 0.01
Invertivore	Spanish hogfish (*Bodianus rufus*)	0.02 ± 0.01	0.01 ± 0.004	0.02 ± 0.005
Invertivore	French angelfish (*Pomacanthus paru*)	0.04 ± 0.04	0.00 ± 0.00	0.02 ± 0.02

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Invertivore	Spotted trunkfish (*Lactophrys bicaudalis*)	0.02 ± 0.02	0.01 ± 0.01	0.02 ± 0.01
Invertivore	Yellow goatfish (*Mulloidichthys martinicus*)	0.00 ± 0.00	0.03 ± 0.01	0.01 ± 0.01
Invertivore	Bluehead (*Thalassoma bifasciatum*)	0.01 ± 0.003	0.01 ± 0.003	0.01 ± 0.002
Invertivore	Porcupinefish (*Diodon hystrix*)	0.03 ± 0.03	0.00 ± 0.00	0.01 ± 0.01
Invertivore	Clown wrasse (*Halichoeres maculipinna*)	0.01 ± 0.01	0.01 ± 0.01	0.01 ± 0.01
Invertivore	Yellowhead wrasse (*Halichoeres garnoti*)	0.01 ± 0.005	0.01 ± 0.002	0.01 ± 0.002
Invertivore	Puddingwife (*Halichoeres radiatus*)	0.02 ± 0.01	0.00 ± 0.00	0.01 ± 0.01
Invertivore	Sergeant major (*Abudefduf saxatilis*)	0.01 ± 0.005	0.01 ± 0.01	0.01 ± 0.004
Invertivore	Honeycomb cowfish (*Acanthostracion polygonium*)	0.02 ± 0.02	0.00 ± 0.00	0.01 ± 0.01
Invertivore	Orangespotted filefish (*Cantherhines pullus*)	0.001 ± 0.001	0.01 ± 0.01	0.01 ± 0.004
Invertivore	Longsnout butterflyfish (*Prognathodes aculeatus*)	0.01 ± 0.004	0.005 ± 0.002	0.01 ± 0.002
Invertivore	Banded butterflyfish (*Chaetodon striatus*)	0.003 ± 0.002	0.01 ± 0.01	0.004 ± 0.003
Invertivore	Sharpnose puffer (*Canthigaster rostrata*)	0.004 ± 0.001	0.003 ± 0.001	0.004 ± 0.001
Invertivore	Blue angelfish (*Holacanthus bermudensis*)	0.01 ± 0.01	0.00 ± 0.00	0.003 ± 0.003
Invertivore	Scrawled cowfish (*Acanthostracion quadricornis*)	0.01 ± 0.01	0.00 ± 0.00	0.002 ± 0.002
Invertivore	Whitespotted filefish (*Cantherhines macrocerus*)	0.00 ± 0.00	0.002 ± 0.001	0.001 ± 0.001
Invertivore	Spotfin butterflyfish (*Chaetodon ocellatus*)	0.001 ± 0.001	0.001 ± 0.001	0.001 ± 0.001
Invertivore	Redspotted hawkfish (*Amblycirrhitus pinos*)	<0.001 ± <0.001	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Yellowtail reeffish (*Chromis enchrysurus*)	0.00 ± 0.00	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Goldentail moray (*Gymnothorax miliaris*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Invertivore	Bandtail puffer (*Sphoeroides spengleri*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Invertivore	Scrawled filefish (*Aluterus scriptus*)	0.00 ± 0.00	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Beaugregory (*Stegastes leucostictus*)	0.00 ± 0.00	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Slippery dick (*Halichoeres bivittatus*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Invertivore	Neon goby (*Elacatinus oceanops*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Invertivore	Blenny spp. *(Emblemariopsis* spp.)	0.00 ± 0.00	<0.001 ± <0.001	<0.001 ± <0.001
Piscivore	Great barracuda (*Sphyraena barracuda*)	0.21 ± 0.08	1.01 ± 0.24	0.64 ± 0.15
Piscivore	Dog snapper (*Lutjanus jocu*)	0.03 ± 0.03	0.30 ± 0.25	0.17 ± 0.14

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Piscivore	Bonnetmouth (*Emmelichthyops atlanticus*)	0.01 ± 0.01	0.15 ± 0.08	0.09 ± 0.04
Piscivore	Tiger grouper (*Mycteroperca tigris*)	0.18 ± 0.10	0.00 ± 0.00	0.08 ± 0.05
Piscivore	Graysby (*Cephalopholis cruentata*)	0.05 ± 0.02	0.02 ± 0.01	0.03 ± 0.01
Piscivore	Bar jack (*Caranx ruber*)	0.01 ± 0.004	0.01 ± 0.004	0.01 ± 0.003
Piscivore	Lionfish (*Pterois volitans/miles*)	0.01 ± 0.01	0.003 ± 0.003	0.01 ± 0.004
Piscivore	Yellowmouth grouper (*Mycteroperca interstitialis*)	0.005 ± 0.005	0.005 ± 0.004	0.005 ± 0.003
Piscivore	Black jack (*Caranx lugubris*)	0.00 ± 0.00	0.01 ± 0.01	0.004 ± 0.004
Piscivore	Yellowfin grouper (*Mycteroperca venenosa*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001
Planktivore	Atlantic creolefish (*Paranthias furcifer*)	1.21 ± 0.58	0.54 ± 0.17	0.84 ± 0.28
Planktivore	Creole wrasse (*Bodianus parrae*)	1.08 ± 0.34	0.63 ± 0.20	0.84 ± 0.19
Planktivore	Blue chromis (*Azurina cyanea*)	0.01 ± 0.003	0.01 ± 0.002	0.01 ± 0.002
Planktivore	Sunshinefish (*Chromis insolata*)	<0.001 ± <0.001	0.01 ± 0.01	0.00 ± 0.003
Planktivore	Purple reeffish (*Chromis scotti*)	0.004 ± 0.004	<0.001 ± <0.001	0.002 ± 0.002
Planktivore	Greenband wrasse (*Halichoeres bathyphilus*)	<0.001 ± <0.001	0.00 ± 0.00	<0.001 ± <0.001

Abundance-Biomass Curves

Mean w-values for both the EFGB and WFGB study sites were 0.05 ± 0.01. For all samples at each study site, mean w-values remained close to 0, indicating a balanced community where biomass was spread evenly between large and small individuals (Figure 5.3). ANOVA comparisons of w-values between study sites revealed no significant dissimilarities between the dominance plot w-values (Table 5.7).

Table 5.7. ANOVA results comparing w-values in EFGB and WFGB study sites.

Source	Sum of Squares	df	F-value	P
Bank	0.0201	1	0.676	0.415
Res	1.3672	46		

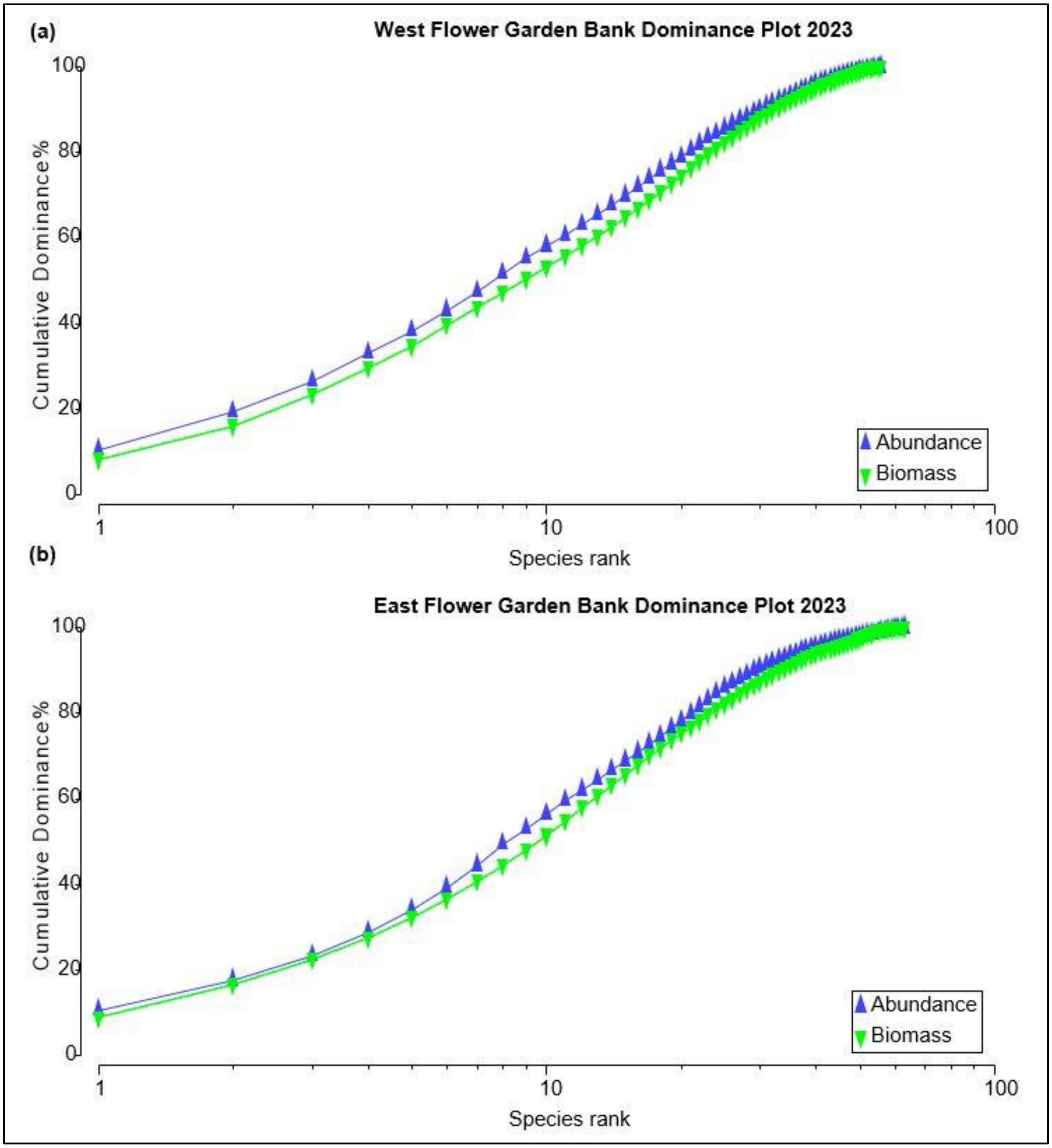


Figure 5.3. Abundance-biomass curves for (a) WFGB and (b) EFGB study sites in 2023.

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. Analyses were also conducted for invasive lionfish (*Pterois volitans*).

In 2023, four species of grouper were observed in EFGB and WFGB surveys: graysby (*Cephalopholis cruentata*), yellowfin grouper (*Mycteroperca venenosa*), tiger grouper (*Mycteroperca tigris*), and yellowmouth grouper (*Mycteroperca interstitialis*). The CV% for graysby density (19.35%) had relatively good power to detect population change while the CV% for the density of the other three species and the biomass for all four grouper species had poor power to detect population differences (i.e., CV% > 20%). Grouper mean biomass was 234.92 ± 93.37 in EFGB surveys and 24.01 ± 10.74 in WFGB surveys. Mean biomass (kg ± SE/100 m^2) of small-bodied grouper (graysby) was 0.051 ± 0.015 in EFGB surveys and 0.019 ± 0.009 in WFGB surveys. Mean biomass of large-bodied grouper (tiger, yellowfin, and yellowmouth grouper) was 0.184 ± 0.097 in EFGB surveys and 0.005 ± 0.004 in WFGB surveys. Size distributions of observed grouper in 2023 varied by species (Figure 5.4). A significant difference in the graysby community density was detected between study sites in 2023 with a higher density (individuals ± SE/100 m^2) at EFGB (0.85 ± 0.18) than at WFGB (0.30 ± 0.11).

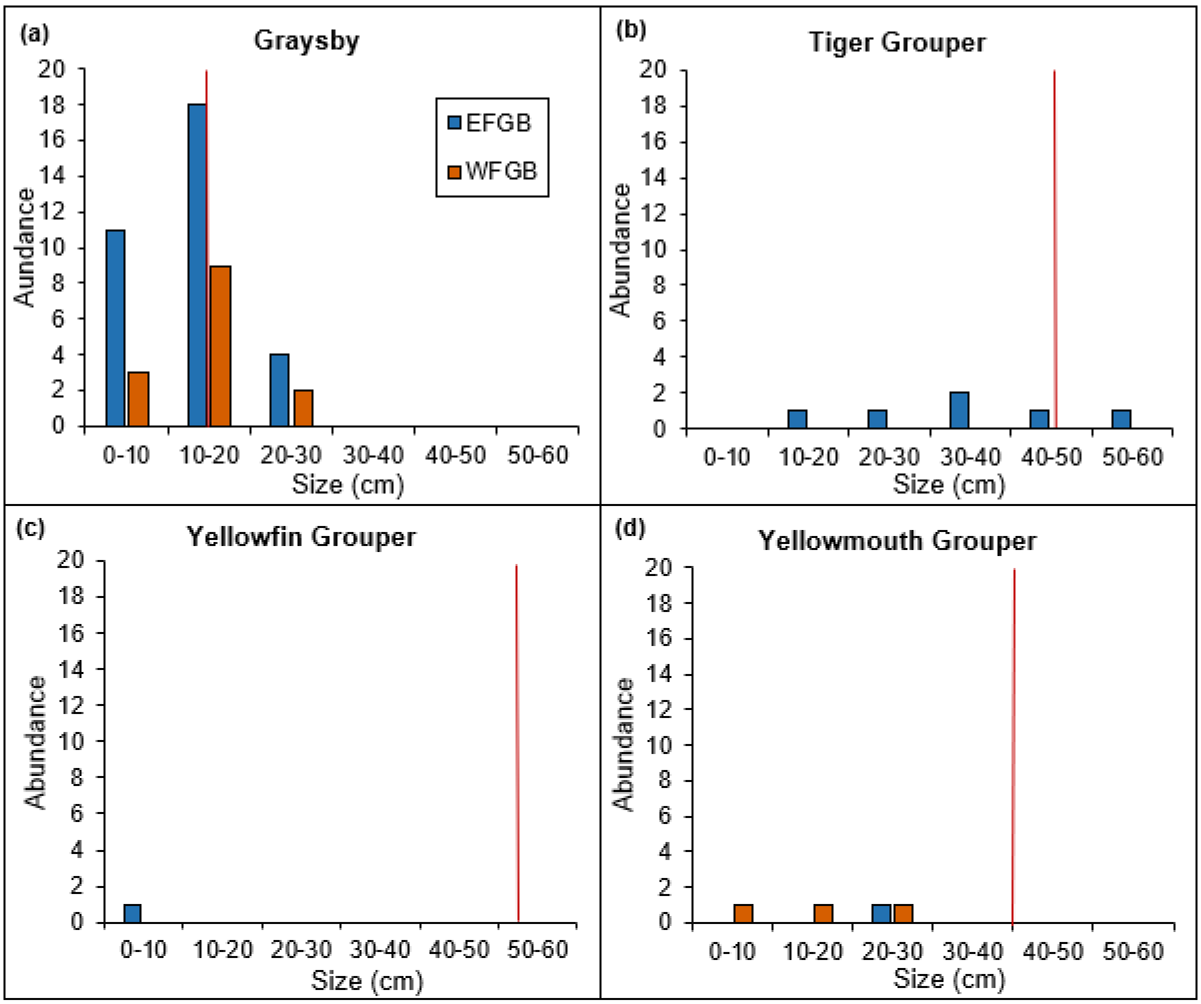


Figure 5.4. Size frequency of grouper species at EFGB and WFGB study sites in 2023: (a) graysby, (b) tiger grouper, (c) yellowfin grouper, and (d) yellowmouth grouper. Vertical solid red lines represent estimated size of female maturity (Froese & Pauly, 2019).

Two snapper species were observed in 2023 surveys: dog snapper (*Lutjanus jocu*) and gray snapper (Figure 5.5). Coefficients of variation for density and biomass for both species indicated that the data had poor power to detect population differences due to the low number of snapper observed. Mean snapper biomass (kg ± SE/100 m^2) was 0.077 ± 0.060 in EFGB surveys and 0.377 ± 0.066 in WFGB surveys. No statistical tests were run on the snapper community due to poor statistical power.

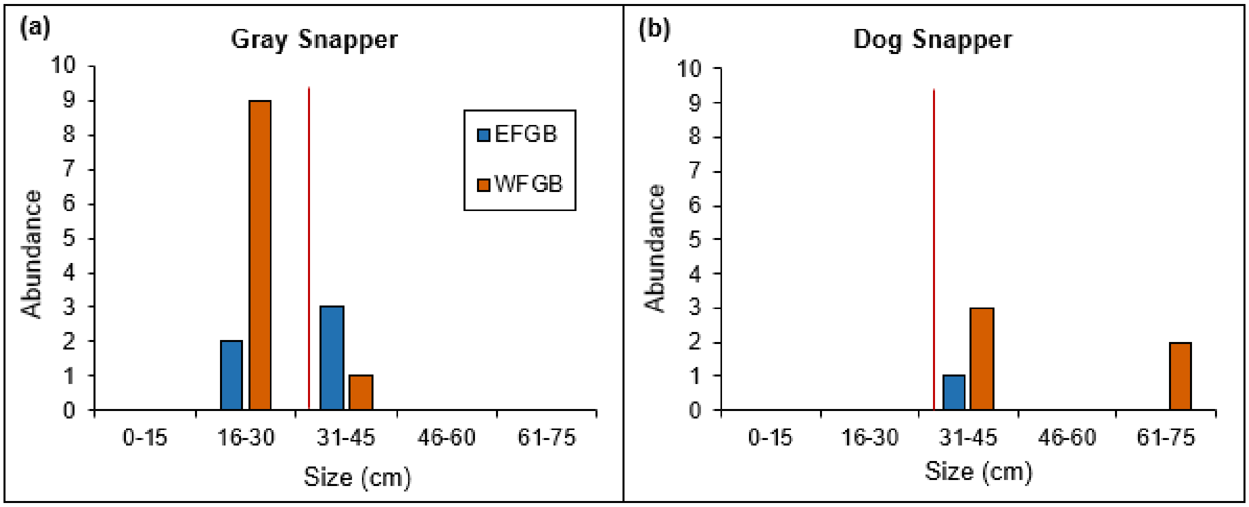


Figure 5.5. Size frequency of snapper species observed at EFGB and WFGB study sites in 2023 (a) gray snapper and (b) dog snapper. Vertical solid red lines represent estimated size of female maturity (Froese & Pauly, 2019).

Parrotfishes are important grazers on coral reefs (Jackson et al., 2014). Seven parrotfish species were observed in EFGB and WFGB 2023 surveys: striped parrotfish (*Scarus iseri*), princess parrotfish (*Scarus taeniopterus*), queen parrotfish (*Scarus vetula*), greenblotch parrotfish (*Sparisoma atomarium*), redband parrotfish, stoplight parrotfish, and bucktooth parrotfish (*Sparisoma radians*). Coefficients of variation indicated that the data had good power to detect differences in density for stoplight parrotfish (13.03%), queen parrotfish (14.91%), and redband parrotfish (18.68%), and differences in biomass in stoplight parrotfish (17.59%) and queen parrotfish (16.93%). Mean biomass (kg ± SE/100 m^2) of parrotfishes was 0.995 ± 0.174 in EFGB surveys and 0.738 ± 0.111 in WFGB surveys. No significant differences in stoplight or queen parrotfish densities were detected between study sites, however there was a significantly higher density of redband parrotfish at EFGB. No significant differences in stoplight or queen parrotfish biomass were detected between study sites in 2023. A high abundance of juvenile parrotfish was observed in surveys at EFGB in 2023 (Figure 5.6).

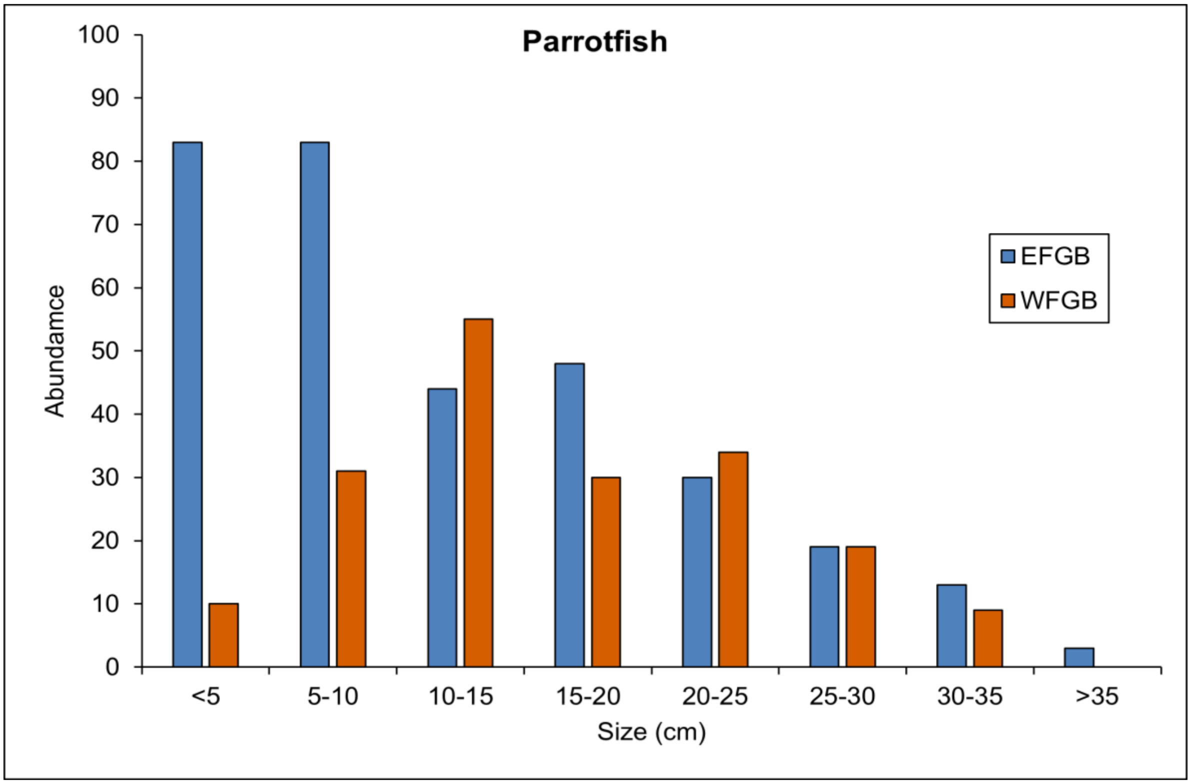


Figure 5.6. Size frequency of parrotfishes at EFGB and WFGB study sites in 2023.

Invasive and Non-native Species

Lionfish (*Pterois* spp.), an invasive species native to the Indo-Pacific, were first observed by scuba divers in FGBNMS in 2011 and in study site surveys in 2013. Two lionfish were observed in surveys in 2023: one at EFGB and one at WFGB. Lionfish were also opportunistically observed by divers during long-term monitoring field work and during lionfish removal cruises held June 11–14, 2023 and July 23–26, 2023.

Regal demoiselle (*Neopomacentrus cyanomos*), a non-native species from the Indo-Pacific region, were last observed in a survey at East and West Flower Garden Banks in 2019, but none were observed during surveys or opportunistically observed by divers during long-term monitoring efforts in 2023. While no regal demoiselles were observed at East or West Flower Garden Banks in 2023, they were observed in fish surveys at Stetson Bank during long-term monitoring field work in July 2023 (68.2% frequency in Stetson Bank fish surveys). Large abundances were also observed at *Kraken*, a shipwreck close to Stetson Bank, during a lionfish removal cruise from July 23–26, 2023.

Fish Survey Long-Term Trends

Since 2002, mean fish density has ranged from 52.70–564.68 individuals/100 m^2 at the EFGB study site and 64.80–471.87 individuals/100 m^2 at the WFGB study site (Figure 5.7). Fish community density was compared among years and study sites when complete survey data were available (2011–2023). PERMANOVA analysis revealed significant differences between study sites among years, and a significant interaction (Table 5.8), demonstrating fish community, based on density, was highly variable among years and locations from 2011–2023 (Figure 5.7). The observed dissimilarity in study site communities based on density from 2011 to 2023 was mainly attributable to variations in blue tang (5.71%) and queen parrotfish (4.43%).

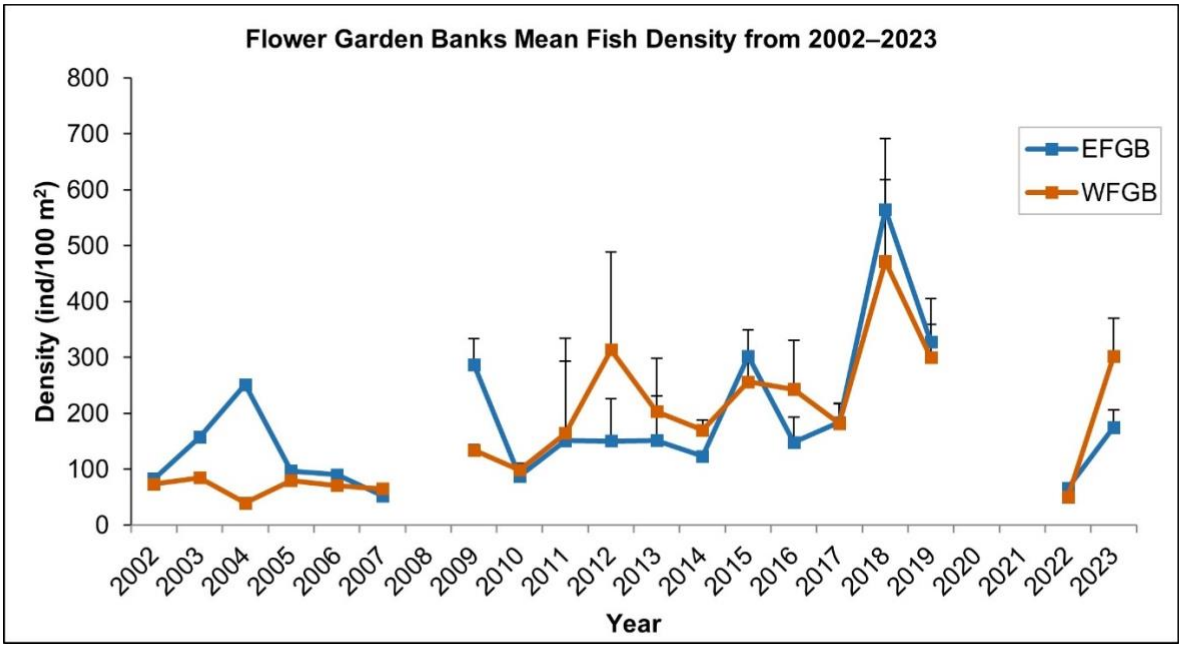


Figure 5 7. Mean fish density (individuals + SE/100 m^2) at EFGB and WFGB study sites from 2002–2023. No data were collected in 2008, 2020, and 2021, and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024 (2009–2023)

Table 5.8. PERMANOVA results comparing mean fish density between EFGB and WFGB study sites and among years from 2011 to 2023. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	12827	1	6.3135	**0.0001**
Year	1.1642E+05	10	5.73	**0.0001**
Bank*Year	53177	10	2.6173	**0.0001**
Res	9.8742E+05	486		
Total	1.1711E+06	507		

Community biomass data, first collected in 2006, was highly variable in the study sites and ranged from 4.55–60.16 kg/100 m^2 in EFGB surveys and 2.46–27.23 kg/100 m^2 in WFGB surveys from 2006–2023 (Figure 5.8). PERMANOVA analysis revealed significant differences between study sites, among years, and a significant interaction (Table 5.9). The observed dissimilarity in community based on biomass between study sites from 2011–2023 was mainly attributable to brown chromis (8.18%) and blue tang (5.94%). 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).

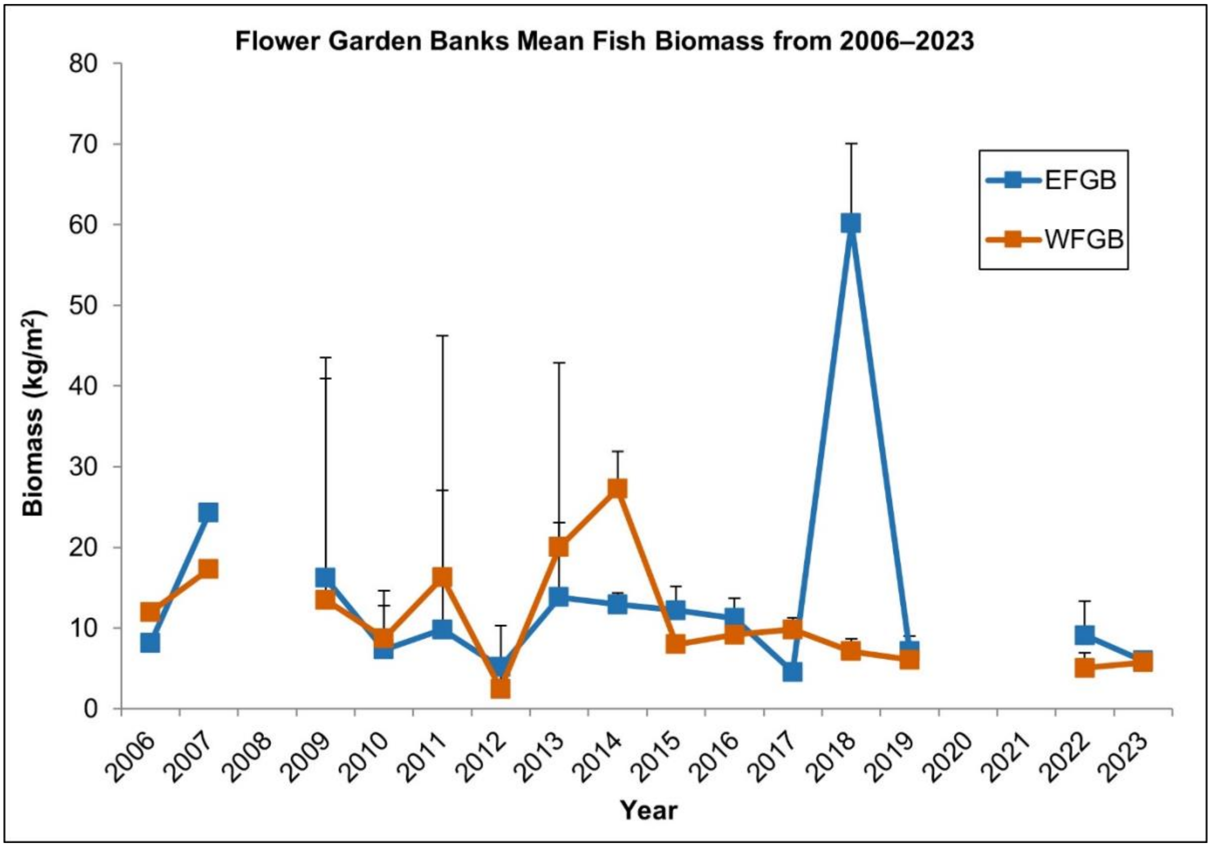


Figure 5.8. Mean fish biomass (kg/100 m^2) + SE in EFGB and WFGB study sites from 2006 to 2023. No data were collected in 2008, 2020, and 2021 and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024 (2009–2023)

Table 5.9. PERMANOVA results using mean fish biomass in EFGB and WFGB study sites and among years from 2011 to 2023. **Bold** text denotes significant values.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	8021.7	1	2.5822	**0.0001**
Year	4.1404E+05	10	13.328	**0.0001**
Bank*Year	60750	10	1.9556	**0.0001**
Res	1.5098E+06	486		
Total	1.9945E+06	507		

Additional analyses were conducted to investigate temporal trends in grouper and snapper density from 2011 to 2023, when complete survey data were available. The most common grouper species at both EFGB and WFGB study sites were graysby and yellowmouth grouper. Tiger grouper, scamp, coney, red hind, and rock hind were, on average, more commonly seen in EFGB surveys, and black grouper have not been encountered in surveys since 2015 (Figure 5.9; Figure 5.10).

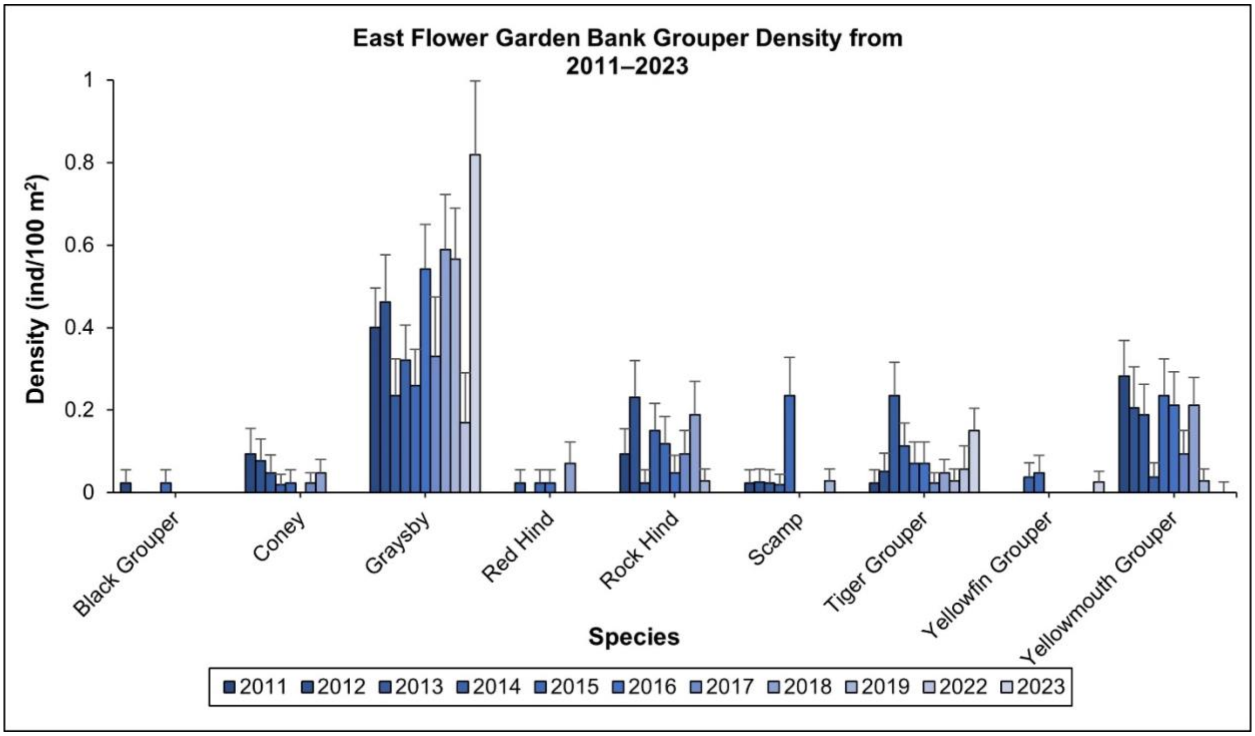


Figure 5.9. Mean density (individuals + SE/100 m^2) of grouper species within EFGB study site surveys from 2011 to 2023. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024

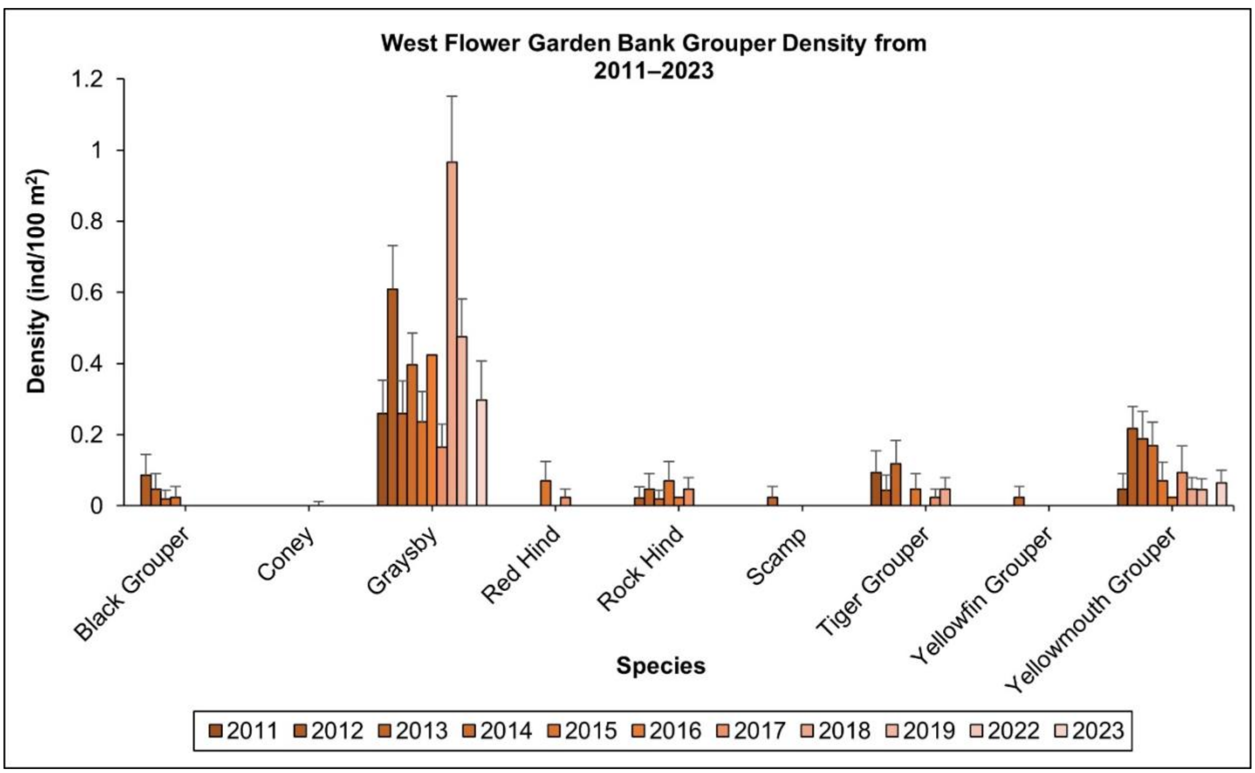


Figure 5.10. Mean density (individuals + SE/100 m^2) of grouper species within WFGB study site surveys from 2011 to 2023. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024

For trend analysis, groupers were separated into large and small grouper categories, with small groupers containing coney, graysby, red hind, and rock hind, and large groupers containing black grouper, scamp, tiger grouper, yellowfin grouper, and yellowmouth grouper. Mann-Kendall trend analysis revealed no significant trend for small grouper density at EFGB ($\tau = 0.09$, $p = 0.76$) or WFGB ($\tau = -0.05$, $p = 0.88$) from 2011 – 2023; however, there was a significant decline in large grouper density at both EFGB ($\tau = -0.55$, $p = 0.02$) and WFGB ($\tau = -0.62$, $p = 0.01$) from 2011–2023).

From 2011 to 2023, dog snapper were consistently denser in WFGB surveys and gray snapper were denser in most years (Figure 5.11; Figure 5.12). Man-Kendall trend analysis revealed no significant trend in densities of either species at EFGB or WFGB from 2011–2023, suggesting stable snapper densities over the past 12 years.

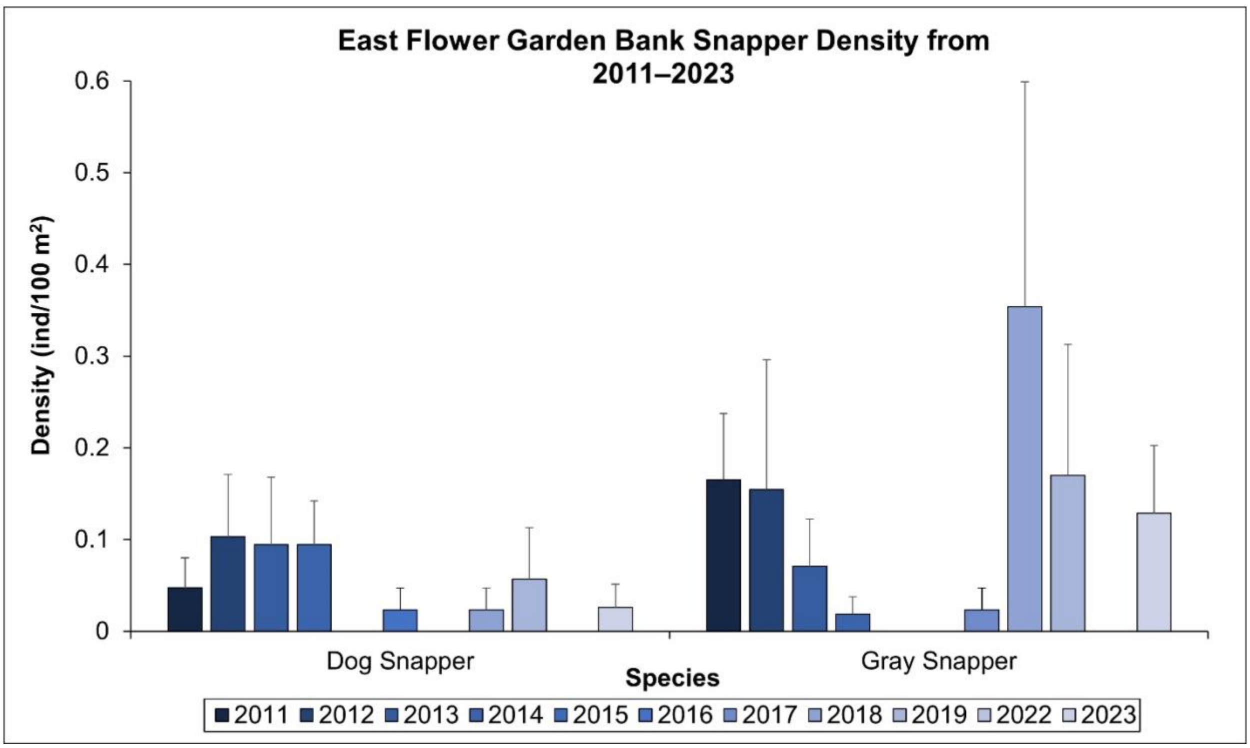


Figure 5.11. Mean density (individuals + SE/100 m^2) of snapper species within EFGB study site from 2011 to 2023. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024

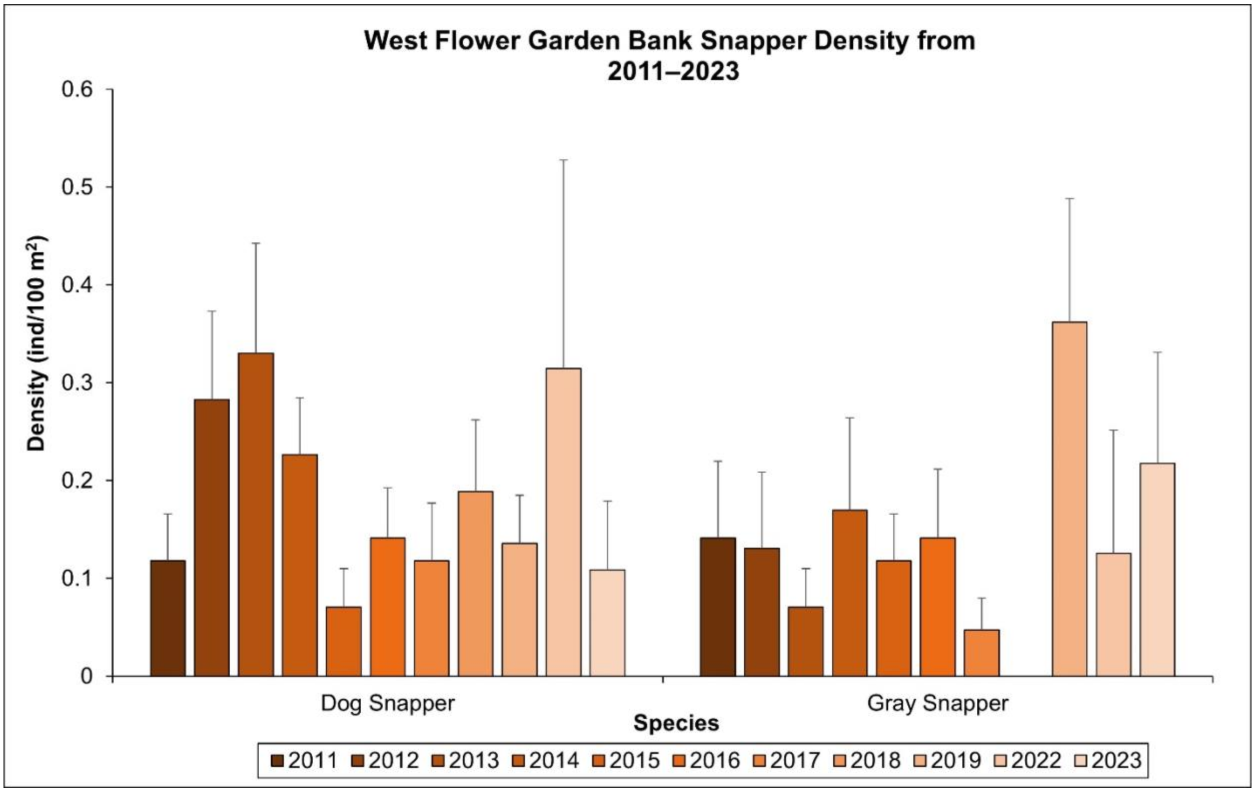


Figure 5.12. Mean density (individuals + SE/100 m^2) of snapper species within WFGB study site from 2011 to 2023. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021, 2024

Discussion

Fish communities are indicators of ecosystem health (Sale, 1991; Knowlton & Jackson, 2008; Jackson et al., 2014). With an understanding of levels of interannual natural variation, they can be an important component of long-term monitoring programs. Historically, the fish communities at EFGB and WFGB have been considered low in diversity but high in biomass (Zimmer et al., 2010). The fish assemblages of the two banks differ somewhat from Caribbean and other lower-latitude reefs, likely because they occur near the northern latitudinal limit of coral reef development in the Gulf of Mexico, are remote from other tropical reef communities, and exist in slightly different habitat types (e.g., on reefs composed almost exclusively of massive hard corals). Reef Environmental Education Foundation (REEF) databases, consisting of reports from 2,653 individual surveys, have documented over 280 fish species from the Flower Garden Banks. These include only a few snappers and grunts, which are more common in other areas of the Western Atlantic (Rooker et al., 1997; Precht et al., 2006; Johnston et al., 2017a). In 2023, a combined total of 74 species were observed in 48 fish surveys in the study sites on both banks. From July 31 to August 4, 2023, REEF conducted fish surveys at FGBNMS and on two nearby oil and gas platforms. Their team of fish experts and enthusiasts completed a total of 297 surveys, with 204 conducted at EFGB and WFGB. The expedition documented 170 species overall, with 145 species observed at EFGB and WFGB (REEF, 2023).

EFGB and WFGB also have lower abundance of herbivorous fishes than other Caribbean reefs (Dennis & Bright, 1988; Bauer et al., 2015a, 2015b, 2015c; Caldow et al., 2015; Clark et al., 2015a, 2015b). Prior to 1985, low macroalgae cover was reported at the Flower Garden Banks, but recent reanalysis indicates a gradual increase over time following a sharp rise after the mass mortality of *D. antillarum* in 1983 (see Gittings & Bright, 1987). During the 2023 study period, the herbivore guild possessed the highest mean biomass, contributing to 38% of the total biomass within study site surveys. Herbivore biomass was also greater at EFGB, where macroalgae percent cover was higher in 2023. Within the herbivore guild, 35% of the total biomass was accounted for by chub species, followed by stoplight parrotfish (20%).

Planktivores represented the second highest mean biomass, comprising approximately 29% of the total biomass within the study sites. Within this guild, Atlantic creolefish contributed to 50% of the total biomass, closely followed by creole wrasse at 49%. A giant manta (*Mobula birostris*) was observed during one survey at WFGB, though it was excluded from the analysis. If included, it would have constituted 64% of the planktivore biomass, significantly skewing the overall results.

A possible source of bias in fish counts is the presence of the research vessel over the study sites. Great barracuda, chubs, and occasionally other species congregate beneath the ship following its arrival on site. While this could inflate estimates when the vessel is directly above a survey site, it rarely occurs. In most cases, fish are more likely to be drawn away from the survey sites, as most are not directly below the vessel. The result, on average, would thus be a reduction in counts for affected species.

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; Southern Ocean Knowledge Information Wiki, 2014). At EFGB and WFGB 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.

In the 2023 surveys at EFGB, bonnetmouth had the highest relative abundance, followed by brown chromis. Similarly, bonnetmouth was the most abundant species at WFGB. These species were frequently observed in large schools ranging from 100 to 2,000 individuals within the water column. Although bonnetmouth is considered an ephemeral species, large transient schools have been consistently documented in surveys at both banks from 2016 to present (Johnston et al., 2017b, 2018a, 2020).

Combined grouper density was higher at EFGB (1.05 individuals/100 m^2) than WFGB (0.37 individuals/100 m^2) in 2023. Snapper densities at both banks were well below one individual per 100 m^2 in 2023. Observed grouper and snapper populations included both juvenile and mature individuals. Typical recruitment and nursery habitats for snappers, such as mangroves and seagrasses, are absent at these offshore banks, and the mechanisms for recruitment of this family to the area remain unclear (Mumby et al., 2004; Clark et al., 2014). The biogeographic isolation of EFGB and WFGB suggests that the fish assemblage that persists at the site relies on a high degree of self-recruitment, but sporadic recruitment events bring influxes of other species. This process is further influenced by the dynamic nature of oceanographic conditions, including the Loop Current and spin-off eddies (Wetmore et al., 2020; Limer et al., 2020).

Parrotfish are recognized as key algae grazers on coral reefs, and their abundance and biomass have been positively correlated with coral cover (Jackson et al., 2014). The mean biomass of parrotfish in FGBNMS is considered low, though not significantly different than many other Caribbean reefs (Jackson et al., 2014; Table 5.10). And while low parrotfish biomass can be associated with high fishing pressure and low coral cover, neither have been documented at EFGB or WFGB. Given the abundance of food for parrotfish at EFGB and WFGB, their low abundance is perplexing. The high abundance of juvenile parrotfish observed in EFGB surveys in 2023 may indicate a recruitment year for parrotfish species.

Table 5.10. Mean biomass (kg/100 m^2) for parrotfish at EFGB, WFGB, and other Caribbean reefs. All data, with the exception of EFGB and WFGB data, are from Atlantic and Gulf Rapid Reef Assessment (2024).

Location	Biomass (kg/100 m^2)
Mexico	1.592
Belize	1.500
EFGB and WFGB study sites	0.855
Guatemala	0.562
Honduras	2.423

Lionfish abundances reported here are from daytime surveys, but lionfish are most commonly sighted during crepuscular feeding periods at dawn and dusk. Thus, our survey data may underestimate actual densities. That said, visual and quantitative assessments at EFGB and WFGB (approximately 27–105 lionfish ha^{-1}; Blakeway et al., 2022) suggest lower densities than elsewhere in the southeast U.S. and Caribbean region, such as North Carolina (150 lionfish ha^{-1}; Morris & Whitfield, 2009) and the Bahamas (100–390 lionfish ha^{-1}; Green & Côté, 2009; Darling et al., 2011), as well as on artificial reefs in the northern Gulf of Mexico (10–100 lionfish ha^{-1}; Dahl & Patterson, 2014).

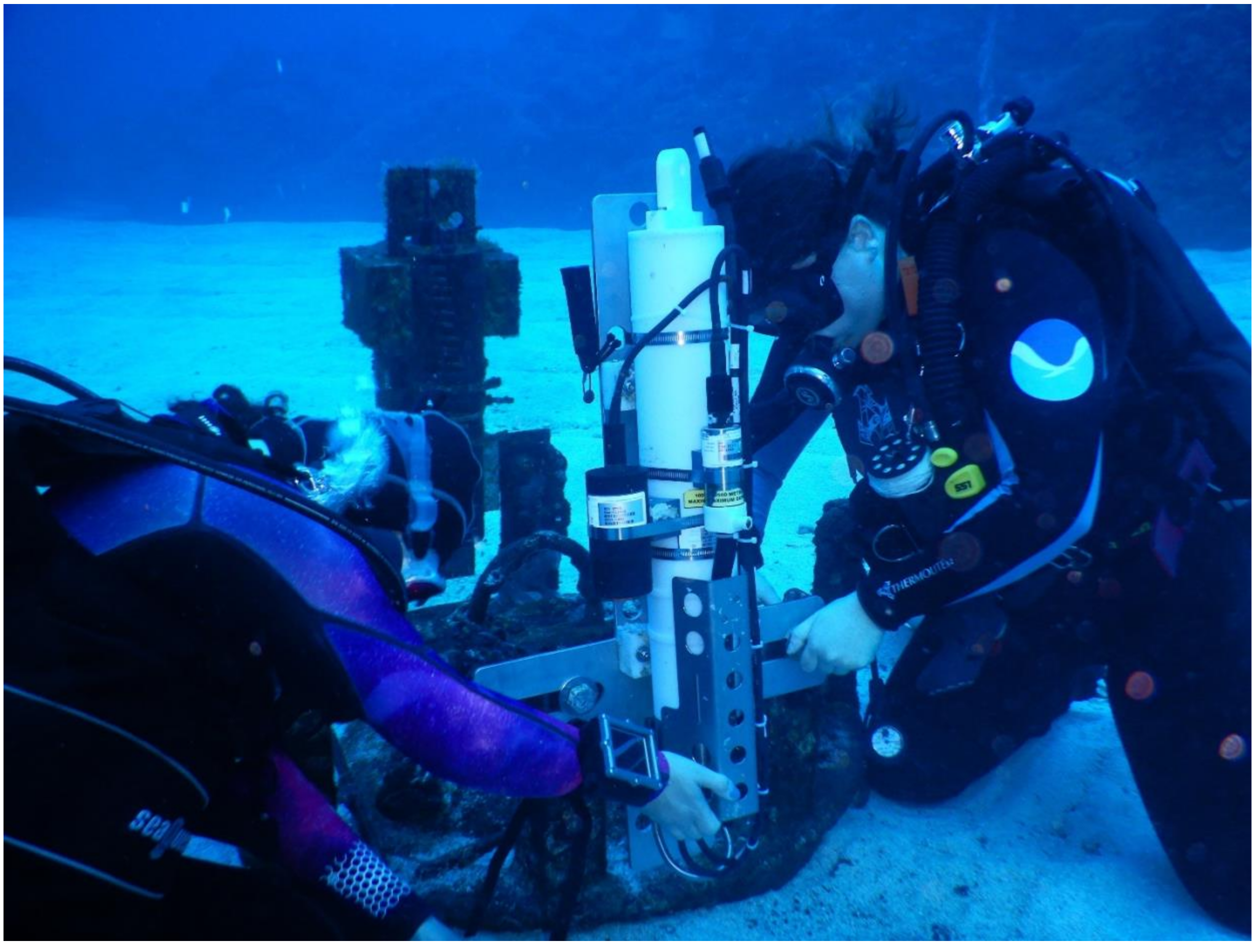
Lionfish have been consistently observed at FGBNMS since 2011. Following their arrival, divers reported rapid increases through 2014, a decline after 2015 (Johnston et al., 2016b), another rise in 2018, and then a sharp decline in 2019 (Johnston et al., 2021). Lionfish were absent in 2022 surveys of EFGB and WFGB, and in the 2023 fish surveys, only two lionfish were recorded (one at each bank).

The high variability of lionfish abundances may reflect boom and bust cycles dictated by recruitment events (which may be limited by the banks' relative isolation, as discussed above), or other controls, such as removal efforts. An ulcerative disease affecting lionfish was first reported in August 2017 across Florida and the Caribbean (Harris et al., 2018) and was also observed in FGBNMS, coinciding with density declines in the northeast Gulf of Mexico (Harris et al., 2020). Although the disease's etiology remains unknown (Cody et al., 2023), it may explain reduced lionfish densities in 2017 and 2019 at EFGB and WFGB. Alternatively, lionfish populations may be oscillating around the banks' carrying capacities, resulting in patterns reported in the northeast Gulf of Mexico (Harris et al., 2023).

Culling may also be affecting populations in multiple ways, including through direct removal and effects on reproductive output and recruitment. Since 2015, permitted lionfish removal cruises have been conducted during summer months from the recreational dive vessel M/V *Fling* to help suppress lionfish predation on native fish. Typically, several hundred lionfish are removed during these cruises, but dives are limited to the upper portion of the reef crest (<40 m) and focus around mooring buoys at recreational dive sites (Green et al., 2014). Thus, removals do not take place over large portions of the reefs, and are not conducted within the monitoring study sites. During the 2023 Lionfish Invitational removal cruises (held in June and July), 250 lionfish were removed from the banks.

The regal demoiselle, a non-native species from the Indo-Pacific, was probably introduced into the Gulf of Mexico via inter-ocean transfer of oil platforms (Robertson et al., 2018). This species is becoming more abundant in the northwestern Gulf of Mexico (O'Connell et al., 2024), but have not yet become abundant on the Flower Garden Banks. There is concern that regal demoiselles could compete with and displace native reef fish, such as the brown chromis (Robertson et al., 2016), but no impacts have yet been confirmed. In nearby locations where they are found in large numbers, regal demoiselles join large schools of planktivores above artificial and natural structures, behaving similarly, without any evident impacts. Sightings from 2018 and 2019 EFGB and WFGB fish surveys were reported to the United States Geological Survey invasive species sightings database, and FGBNMS will continue to monitor the abundance, distribution, and impacts of this species.

Chapter 6:
Water Quality



Research staff switch out a sensor at WFGB. Photo: Donavon French/CPC

Water Quality Introduction

Several water quality parameters have been continuously or periodically recorded at EFGB and WFGB from January through December 2023. Salinity, turbidity, and temperature were recorded every hour by data loggers installed in or near the study sites at depths of approximately 24 m. FGBNMS partners have been continuously monitoring dissolved oxygen at WFGB at a depth of 24 m. Temperature loggers, located at repetitive photostations at depths of 30 m and 40 m at each bank, collected hourly readings; these sensors were recovered August 15 and 17, 2023 and the remaining sensors have yet to be recovered. Additionally, a Sofar Ocean® buoy was installed at EFGB in June 2023; the installation included an additional temperature logger at 24 m depth, along with a sea surface temperature logger.

Water samples were collected in January, May, and August 2023 at three different depths within the water column and analyzed by a U.S. Environmental Protection Agency-certified laboratory for select nutrients. Water column profiles were also acquired in January in conjunction with water sample collections. Water samples are usually collected on a quarterly basis, but cruises were canceled or scaled back due to time constraints or equipment failure. This chapter presents data from moored water quality instruments, water column profiles, and water samples collected in 2023.

Water Quality Methods

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 conductivity, temperature, and depth (CTD) sensor (SBE 16plus) equipped with a WET Labs ECO NTUS turbidity meter. Instruments were located at a depth of 23 m at EFGB and 27 m at WFGB. Loggers were secured to mounting anchors and located in sand flats at each bank (see Figure 1.3; Figure 1.4). The instruments recorded temperature, salinity, and turbidity on an hourly basis. They were exchanged by divers for downloading and maintenance in January, May, and August 2023, as well as in of July 2024. 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 checked. Maintenance, as well as factory service and calibration of each instrument, was delayed in 2023 while some instruments received new sensor integrations.

Onset® Computer Corporation HOBO® Pro v2 U22-001 (HOBO) thermograph loggers were used to record temperature on an hourly basis. These loggers (attached directly to the primary SBE 16plus instrument) 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, respectively. HOBO loggers were also deployed at 30 m and 40 m stations at EFGB and WFGB (attached to permanent repetitive photostation markers). The HOBO logger at 30 m in EFGB was lost and a replacement was deployed in August, 2023. HOBO loggers at EFGB and WFGB have yet to be recovered, as time constraints prevented visitation in 2024.

On June 21, 2023, sanctuary researchers installed a Sofar Ocean® Spotter Buoy and Smart Mooring near real-time temperature sensor at EFGB. The Spotter Buoy is capable of recording wind and wave data, while the Smart Mooring records water temperature at both the surface and reef depth (23 m). Data are transmitted via satellite to a Sofar Ocean® dashboard every 30 minutes, where it can be downloaded and processed for reporting. The Sofar Ocean® system serves as a reliable temperature backup for the primary SBE 16plus logging instrument located at 23 m at EFGB and introduces an additional method of data acquisition, helping to address temperature data gaps. Sofar Ocean® data from June 21 through December 2023 were available in real time and processed into daily averages.

Water Column Profiles

Water column profiles from the surface to the reef cap were acquired in January 2023 with a Sea-Bird® Electronics 19plus V2 CTD that recorded temperature, salinity, pH, turbidity, fluorescence, and dissolved oxygen every ¼ second. The carousel package included a Sea-Bird®

55 Frame Eco water sampler equipped with 12 four-liter Niskin bottles and a Sea-Bird® Electronics 19plus V2 CTD. 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 of <1 m s^{-1}. The water column profiles were obtained on January 10, 2023. Due to damage sustained by the 19plus V2 CTD, which required servicing for most of the year, no additional complete profiles were obtained in 2023. As a backup, supplemental profiles were conducted using a handheld YSI® logger, which measured temperature, dissolved oxygen, pH, and salinity.

Water Samples

In conjunction with water column profiles using the sampling carousel described above, water samples were collected. The carousel was attached to the R/V *Manta* scientific winch cable, allowing the operator to activate the bottles for sample collection at specific depths. Four 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. A blind duplicate water sample was taken at one of the sampling depths for each sampling period.

Water samples were analyzed for chlorophyll *a* (chl *a*) and nutrients including ammonia, nitrate, nitrite, soluble reactive phosphorus (ortho phosphate), and total Kjeldahl nitrogen (TKN; Table 6.1). 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 TKN samples were collected in 1000-ml bottles with a sulfuric acid preservative. Within minutes of sampling, labeled sample containers were stored on ice at 0 °C and a chain of custody was initiated for processing at a U.S. Environmental Protection Agency-certified laboratory. The samples were transported and delivered for analysis to A&B Laboratories in Houston, Texas within 24 hours of collection.

Table 6.1. Standard U.S. Environmental Protection Agency methods used to analyze water samples collected at FGBNMS.

Parameter	Test Method	Detection Limit
Chl *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 phosphorus	SM 4500 P-E	0.02 mg/l
TKN	SM 4500NH3D	0.50 mg/l

Water samples for ocean carbonate measurements, including pH, alkalinity, CO_2 partial pressure (pCO_2), aragonite saturation state, and total dissolved CO_2, 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. 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 January 10, May 4, and August 18, 2023.

Water Quality Data Processing and Analysis

Temperature, salinity, and turbidity data recorded on SBE 16plus instruments and temperature data recorded on backup HOBO loggers were downloaded and processed in January, May, and August 2023, while Sofar Ocean® Spotter Buoy temperature data were processed monthly starting in June 2023. QA/QC procedures included a review of all files to ensure data accuracy and servicing instruments based on manufacturer recommendations. The 24-hourly readings obtained each day were averaged into a 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 database; however, these buoys were removed in late April 2019 and January 2017, respectively, due to lack of support and funding. Therefore, surface buoy readings were unavailable or absent for the 2023 analyses until June (at EFGB only). To address this gap, FGBNMS plans to install additional Sofar Ocean® buoys within the sanctuary in 2025 to enhance data collection and coverage. In lieu of in situ surface data, satellite-derived SST and SSS data for 2023 were downloaded from the NOAA Environmental Research Division Data Access Program data server for comparison to reef cap data. The SST data set used was "GHRSST Level 4 MUR Global Foundation Sea Surface Temperature Analysis (v4.1)" and the SSS data set 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; NOAA Coral Reef Watch, 2023). Satellite-derived one-day mean SST data utilized for WFGB and EFGB in 2023 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 were available as a 0.25-degree longitude/latitude level-3 gridded three-day mean dataset from MIRAS satellite observations.

Additional satellite surface parameters, including chl *a*, algal bloom index, and a suspended sediment proxy (Rrs 667), were collected by the Moderate Resolution Imaging Spectroradiometer (MODIS-Aqua) sensor with a 4 km resolution. These data were obtained from the Ocean Biology Processing Group at NASA's Goddard Space Flight Center via the FGBNMS data dashboard, managed by the University of South Florida (NASA, 2023; Otis, 2023). The algal bloom index was calculated using the method outlined by Hu and Feng (2016). River discharge data were sourced from the U.S. Geological Survey National Water Information System via the FGBNMS data dashboard (Otis, 2023; U.S. Geological Survey, 2023). Discharge values from the Mississippi River, along with Texas rivers (Colorado, Brazos, Trinity, Neches, and Sabine) were aggregated to represent discharge into the northwest Gulf of Mexico and subsequently plotted.

The 30-m and 40-m HOBO loggers were exchanged in August 2023, providing data for 8 months of the year, with the data set expected to be completed for the full year upon recovery of the remaining loggers. Results from chl *a* and nutrient analyses have been received from A&B Laboratories and compiled into an Excel table, while ocean carbonate analyses have been obtained from the CCL at TAMU-CC.

For seawater temperature, salinity, and turbidity, daily mean data from the SBE 16plus instruments at EFGB and WFGB were compared using a paired t-test in R version 4.2.2. Long-term monotonic trends for seawater temperature and salinity data were identified using the Seasonal-Kendall trend test, conducted with a Microsoft Windows® DOS executable program developed by the U.S. Geological Survey for water resource data (Hipel & McLeod, 1994; Helsel & Hirsch, 2002; Helsel et al., 2006). This test applied the Mann-Kendall trend test to each month, evaluating changes among the same months from different years, while accounting for serial correlation in recurring seasonal patterns.

Water Quality Results

Temperature

Satellite SST at EFGB ranged from 21.60 °C to 31.49 °C in 2023 and Sofar Ocean® surface sensor at EFGB ranged from 22.56 °C to 31.74 °C (June–December) in 2023. At 23 m, it ranged from 21.80 °C to 30.07 °C (Figure 6.1). The 23-m backup HOBO logger and 23-m Sofar Ocean® sensor registered similar temperatures to one another, however there were slight differences to temperatures recorded by the 23-m SBE 16plus, with a maximum difference of 1.5 °C between the SBE 16plus and Sofar Ocean® sensor (Figure 6.1). The 30-m HOBO logger at EFGB was lost and not recovered during the recovery cruise in August. At 40 m, temperatures on the HOBO logger ranged from 21.73 °C to 27.44 °C.

Surface temperature at WFGB ranged from 20.83 °C to 30.50 °C in 2023 (Figure 6.1). At 27 m, it ranged from 20.55 °C to 30.63 °C in 2023 (Figure 6.1). The 27-m backup HOBO logger registered temperatures similar to those from the 27-m SBE 16plus (Figure 6.1).

The maximum temperature at 23 m depth at EFGB was 2 °C higher than at 27 m depth at WFGB, which corresponded with a higher percentage of bleached corals observed at EFGB.

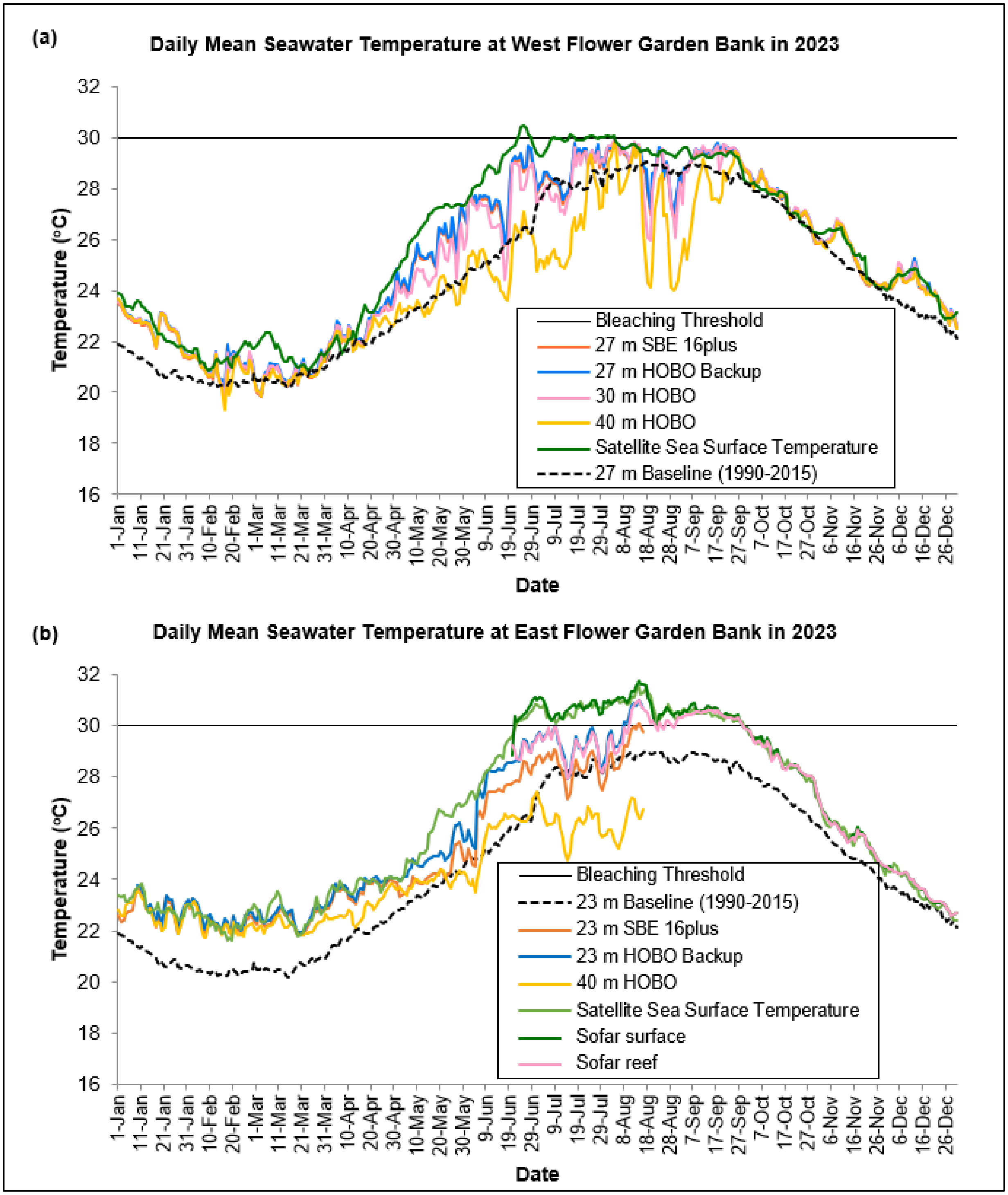


Figure 6.1. Daily mean seawater temperature (°C) at (a) EFGB and (b) WFGB from various depths in 2023, as well as the 25-year daily mean water temperature baseline. The solid black line at 30 °C is a level known to trigger coral bleaching.

Data from the 23-m SBE 16plus recorded reef cap temperatures at EFGB exceeding 30 °C for a single day in 2023; however, these readings did not align with data from the HOBO logger or the Sofar Ocean® sensor, confirming that the SBE 16plus required servicing and may not have recorded accurate temperatures. Therefore, HOBO data were used in place of the SBE 16plus. Supplementary data from the Sofar Ocean® mooring at 23 m indicated that reef cap temperatures at EFGB exceeded 30 °C for 46 days in total and surpassed 29 °C for 93 days in 2023 (Figure 6.2). In comparison, the HOBO logger at 24 m recorded temperatures exceeding 30 °C for 53 days and above 29 °C for 99 days, as detailed in Chapter 2. At WFGB, temperatures at 27 m exceeded 30 °C for 31 days in 2023. As noted in Chapter 2, coral bleaching was observed at both EFGB and WFGB in 2023.

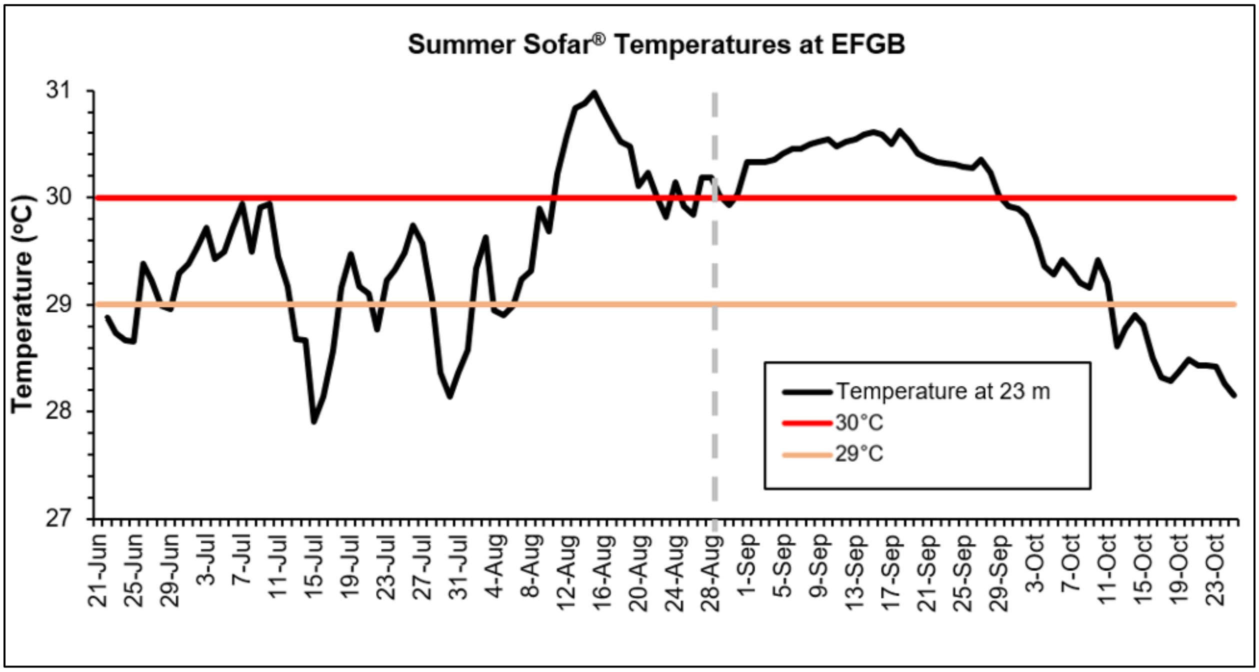


Figure 6.2. Temperature data from Sofar Ocean® Smart Mooring at EFGB at reef crest shows 93 days over 29 °C and 46 days over 30 °C. The vertical dotted grey line represents the date bleaching was first reported at FGBNMS in 2023.

Seawater temperature data obtained from loggers at EFGB (23 m) and WFGB (27 m) have been collected since 1990. Though some data gaps occurred due to equipment malfunction and changes in methods and/or instrumentation, long-term trends showed increasing surface and reef cap temperatures at EFGB and WFGB (Figure 6.3). The Seasonal-Kendall trend test on time-series satellite and daily mean seawater temperature data at depth revealed significantly increasing, monotonic trends from 1990 to 2023 in EFGB and WFGB surface waters ($\tau = 0.32$, $z = 9.43$, $p < 0.001$ and $\tau = 0.31$, $z = 8.97$, $p < 0.001$, respectively) and at EFGB (23 m) and WFGB (27 m) datasondes ($\tau = 0.29$, $z = 6.84$, $p < 0.001$ and $\tau = 0.30$, $z = 7.35$, $p < 0.0002$, respectively) after adjusting for correlation among seasons (Figure 6.3).

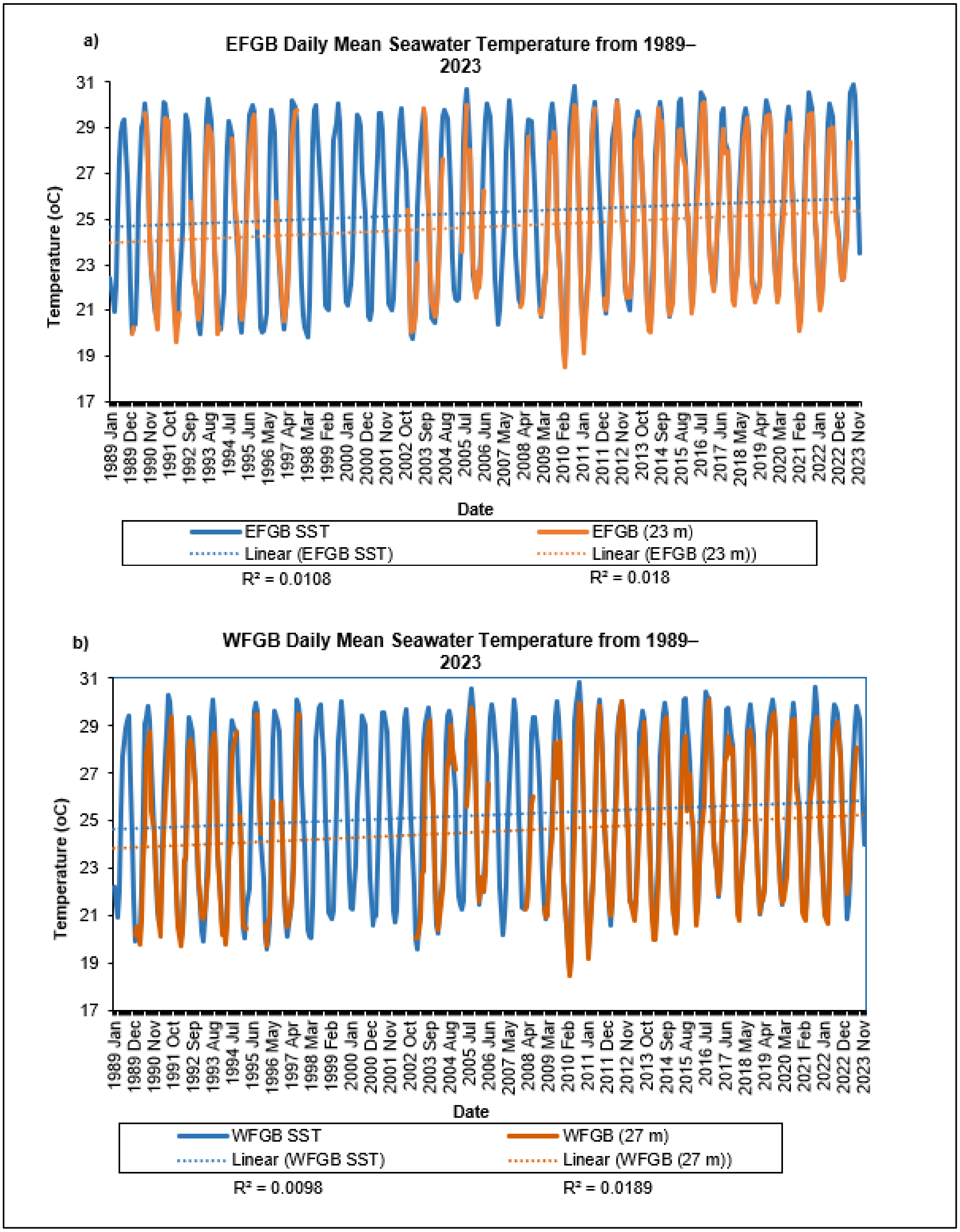


Figure 6.3. Daily mean seawater temperature (°C) demonstrates 12-month seasonal variation from various depths at (a) EFGB and (b) WFGB from 1989 to 2022, as well as a significant increase over time (trend lines).

Salinity

In 2023, salinity at EFGB ranged from 30.28 to 39.37 psu at the surface and 35.12 to 36.73 psu at 23 m (Figure 6.4). At WFGB, salinity ranged from 31.29 to 39.92 psu at the surface and 35.19 to 36.69 psu at 27 m (Figure 6.4). There was no significant difference between EFGB 23 m and WFGB 27 m SBE 16plus reef cap daily mean salinity in 2023.

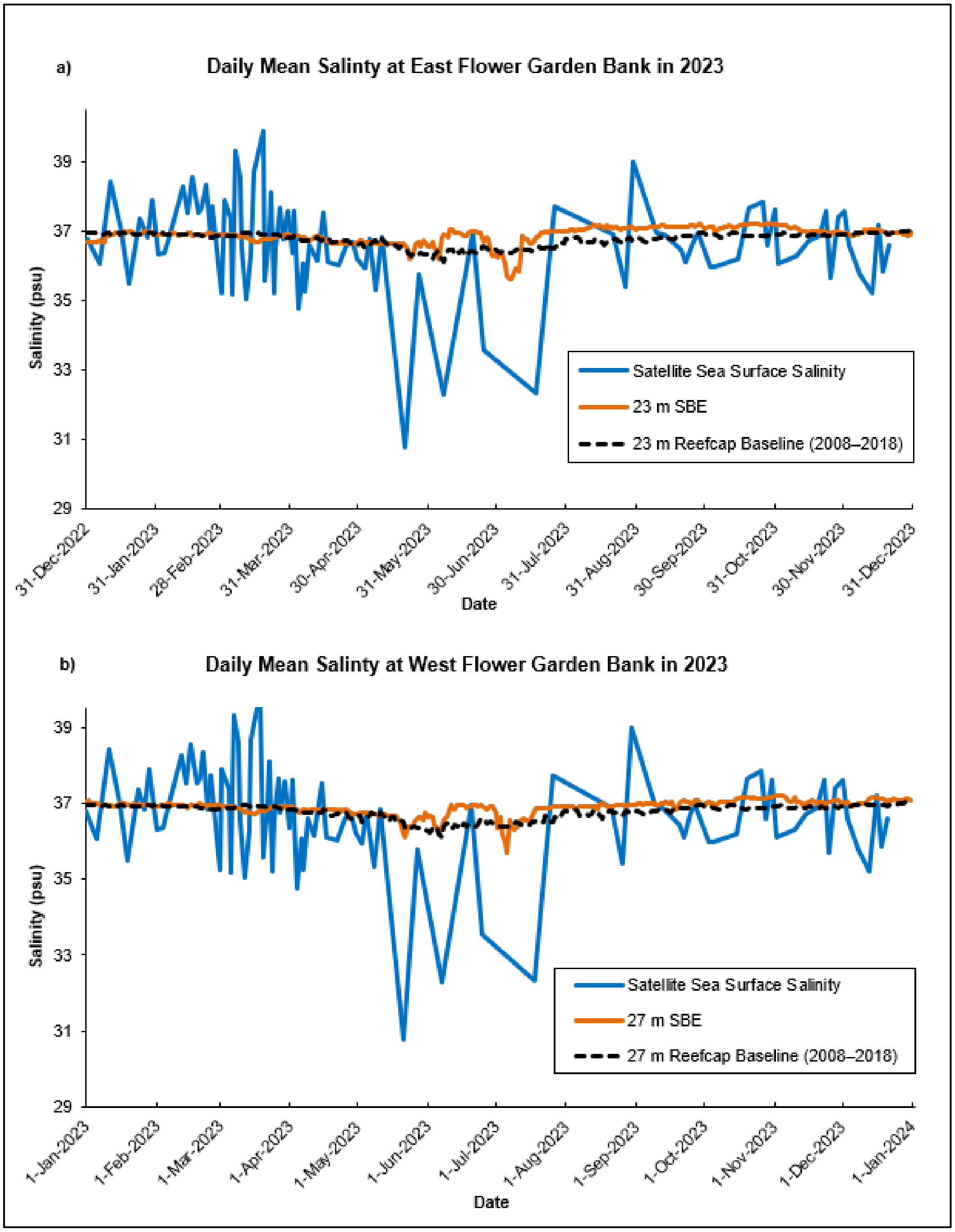


Figure 6.4. Daily mean salinity (psu) at the sea surface, SBE 16plus reef cap station, and the reef cap 10-year daily mean salinity baseline (2008–2018) at (a) EFGB and (b) WFGB in 2023.

Salinity data obtained from loggers at EFGB (23 m) and WFGB (27 m) have been collected since 2008 with only a few data acquisition disruptions. The data show consistent summer minima, often during June and particularly in surface water, long-term decreases in surface salinity at both banks, and decreasing reef cap salinity at EFGB (Figure 6.5). The Seasonal-Kendall trend test on time-series daily mean salinity data at EFGB (23 m) resulted in a significantly decreasing, monotonic trend from 2008 to 2023 and no significant trend at WFGB (27 m; $\tau = -0.29$, $z = -5.07$, $p = 0.02$ and $\tau = -0.23$, $z = -4.05$, $p = 0.07$, respectively) after adjusting for correlation among seasons.

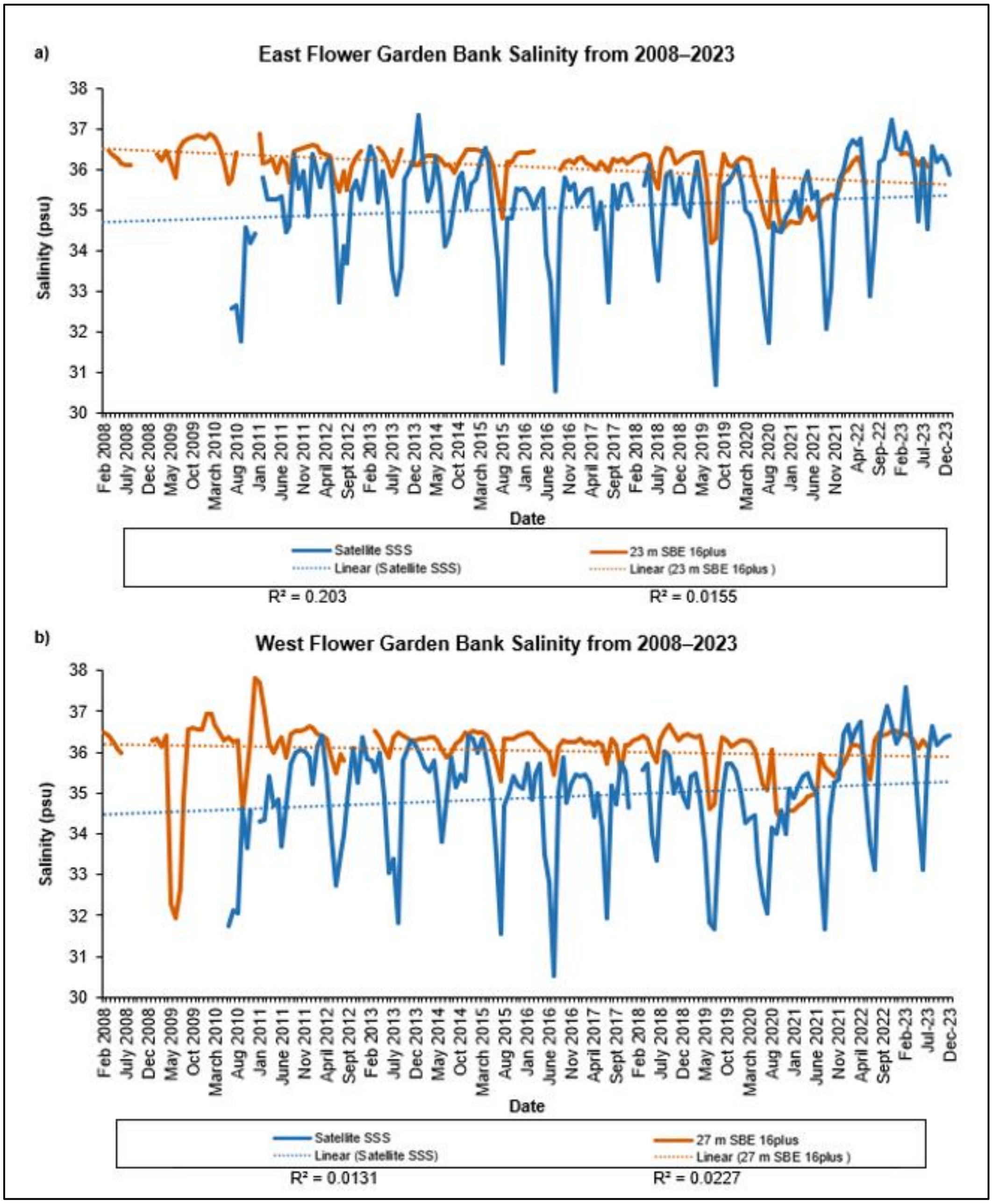


Figure 6.5. Monthly mean salinity showing seasonal variation and long-term trends at (a) EFGB (23 m) and (b) WFGB (27 m) from 2008 to 2023.

Turbidity

The turbidity sensors at EFGB experienced significant malfunctions, resulting in unreliable data throughout 2023. Some data were recovered after deploying a new instrument, but with significant corruption errors from this period were removed, leading to a data gap from May 4 to July 17. Additionally, sensors reported a substantial increase in turbidity at EFGB from July 24 to November 18. The turbidity levels recorded during this period were notably higher than historical readings, which have consistently remained below 2 ntu. These unusual readings may be attributed to sensor malfunctions or the placement of the sensor near sand on the seafloor. Due to these malfunctions, no statistical tests were performed. In 2025, the monitoring site will be restored by lifting the mounting bracket out of the sand, which will provide adequate room for the sensor to work. Returning to a routine instrument servicing rotation in 2025 will also improve the quality and consistency of recordings. For post-processing QA/QC, comparing current year data to historical accounts and adding more context to major storms will add additional clarity to the data. Turbidity values ranged from 0.00 to 14.72 ntu at EFGB and from 0.02 to 0.64 ntu at WFGB (Figure 6.6).

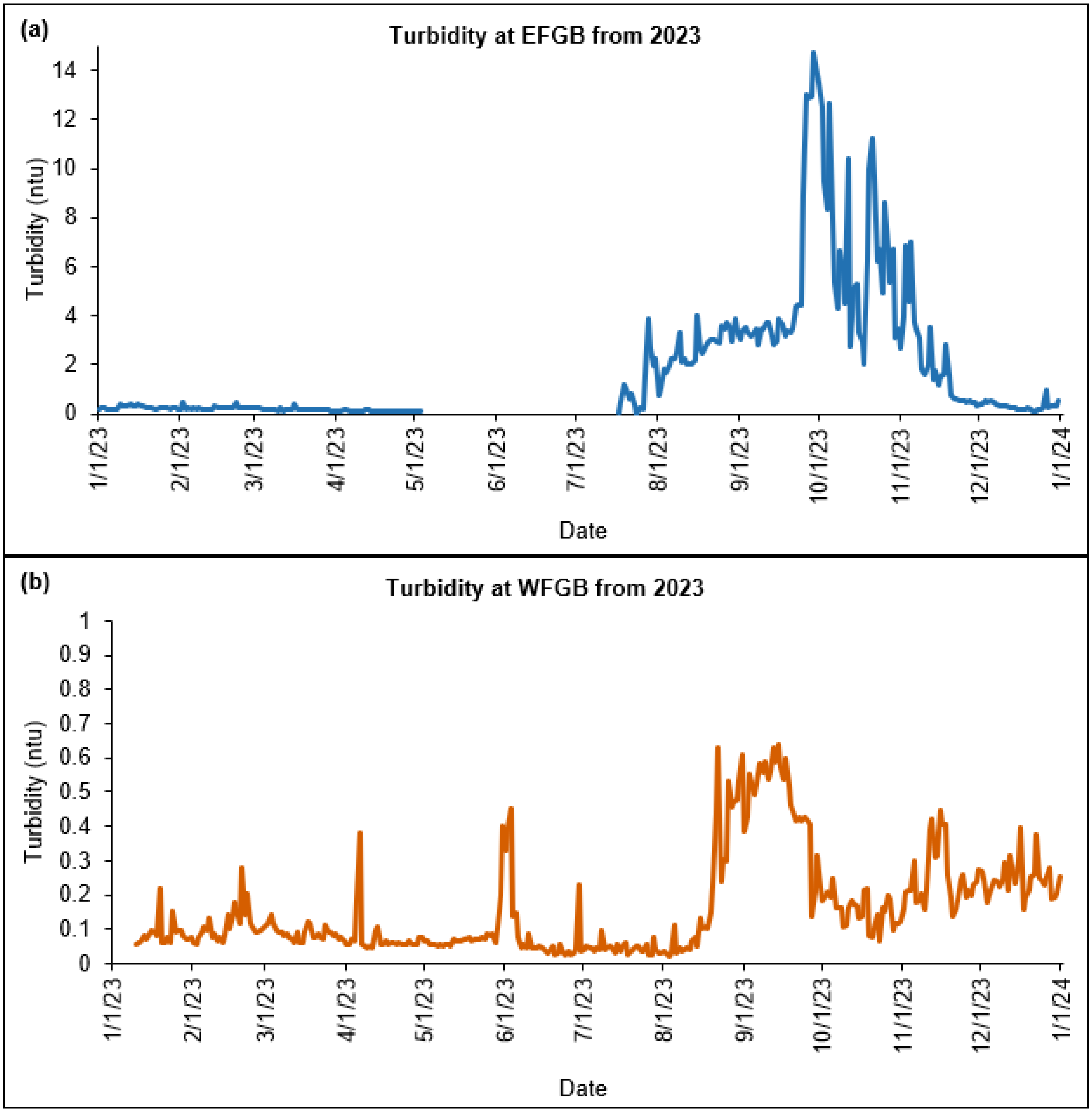


Figure 6.6. Daily mean turbidity (ntu) values in 2023 from (a) EFGB (23 m) and (b) WFGB (27m). Data corruption from malfunctioning sensors at EFGB resulted in a data gap from May 4 to July 17, 2023.

Water Column Profiles

Water column temperatures at the two banks varied by less than 2 °C for each comparative month and remained well mixed throughout the water column (Figure 6.7). Salinity values at both banks were similar, with variations averaging less than 1 psu. The conductivity sensor at WFGB experienced air bubble blockage during the shallow portions of the cast, which cleared up at a depth of 12 m, allowing for partial data to be recorded. The usable data displayed high variability and was consistently higher than other profiles collected. Dissolved oxygen values were variable at the surface but stabilized below four meters at both banks. pH levels fluctuated at the surface, likely due to oxygen mixing from the boat's engines, but became stable once the sensor descended beyond 2 m. Turbidity values were consistent between EFGB and WFGB from 2 m depth, fluctuating between 0.01 and 0.02 ntu. Fluorescence values were slightly higher at WFGB compared to EFGB (Figure 6.7).

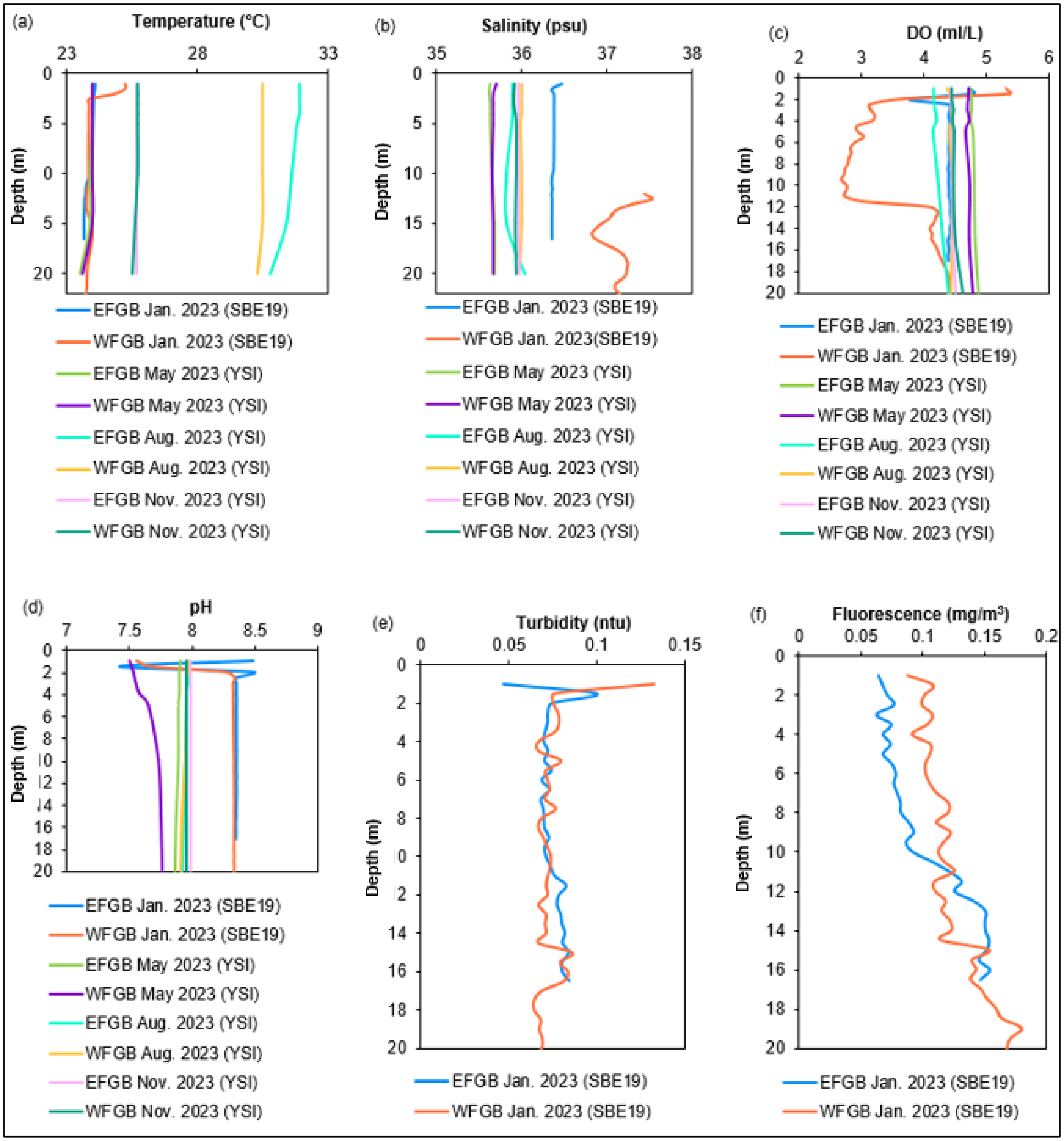


Figure 6.7. (a) Temperature, (b) salinity, (c) dissolved oxygen (DO), (d) pH, (e) turbidity, and (f) fluorescence water column profile data from EFGB and WFGB in January 2023.

Water Samples

The first chl *a* and nutrient samples were taken as part of the long-term monitoring program in 2002. Since then, quarterly nutrient levels have typically been below detection limits, with the exception of occasional ammonia and TKN detections prior to 2012 (Figure 6.8; Figure 6.9). The 2023 nutrient levels from each water column depth were below detection limits in all samples, consistent with oligotrophic oceanic conditions. Ocean carbonate measurements conducted in tandem with nutrient sampling were sent to TAMU-CC for analysis.

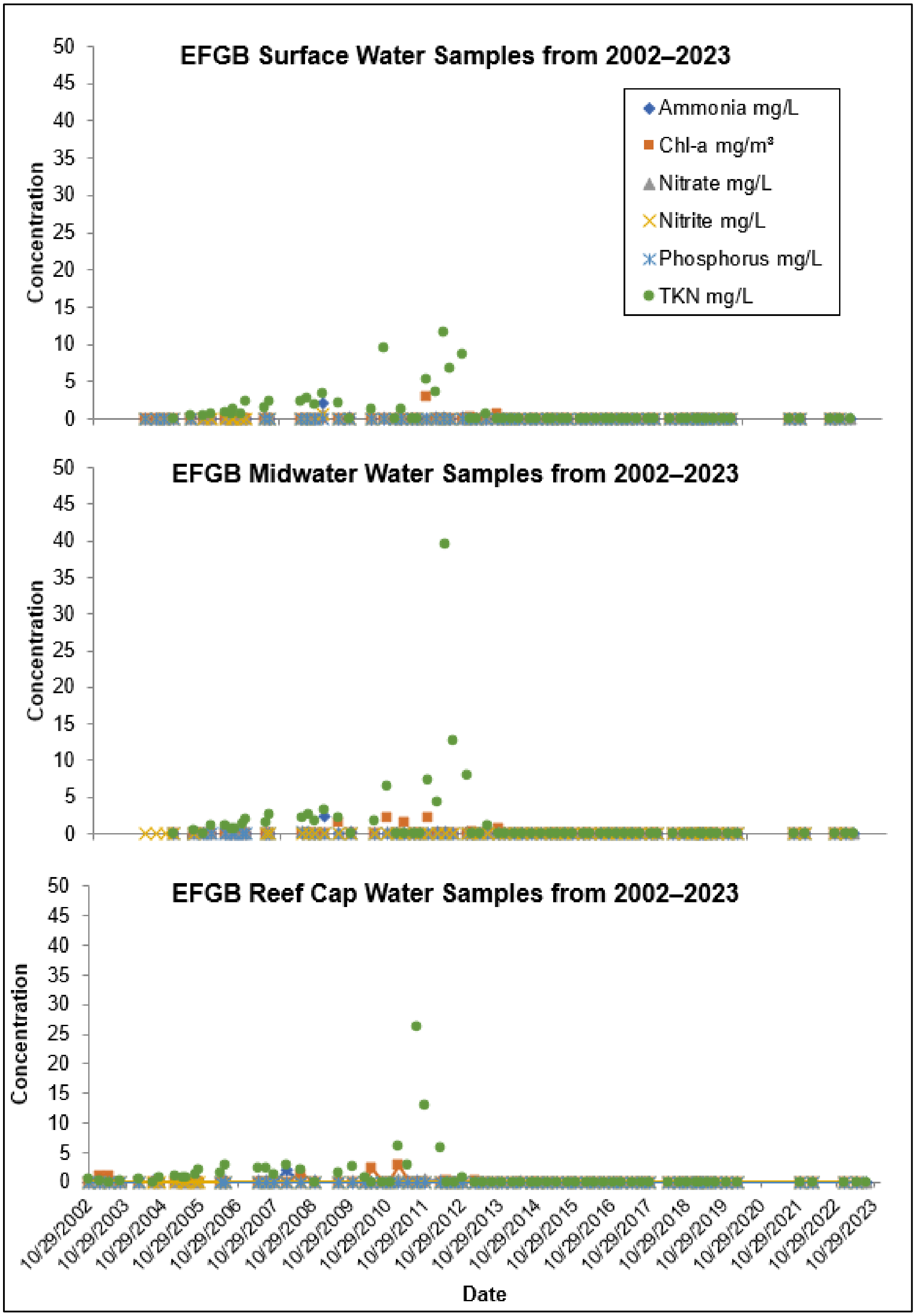


Figure 6.8. Nutrient concentrations from EFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2023.

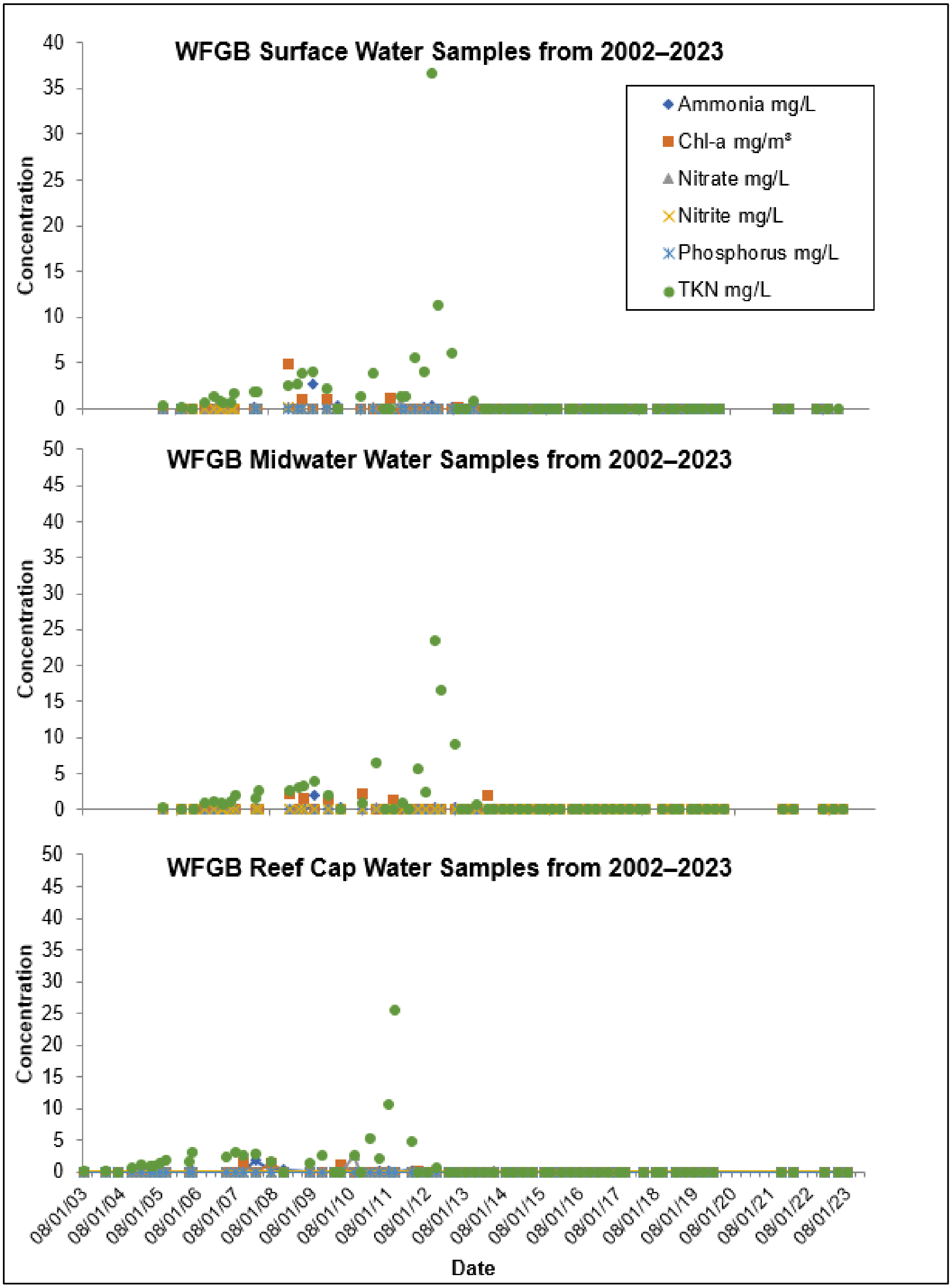


Figure 6.9. Nutrient concentrations from WFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2023.

Water samples taken on May 4 and August 17, 2023 at three distinct depth gradients (approximately 20, 10, and 1 m) were submitted to CCL research partners at TAMU-CC for analyses of multiple parameters, including pH, alkalinity, pCO_2, $\Omega_{aragonite}$, and total dissolved inorganic CO_2 (DIC; Table 6.2; Table 6.3). Salinity was within normal range for May and August, 2023 across the system at both EFGB and WFGB. Temperatures were similar to those observed since 2013. pH and $\Omega_{aragonite}$ deviations remained fairly small in 2023 and throughout the ten-year period of carbonate chemistry monitoring. Surface water pCO_2 showed more variation and a higher average in 2023 compared to 2020–2022 values. The lowest pCO_2 values, where the air-sea pCO_2 gradients were greatest, corresponded to the lowest aragonite levels and the highest DIC records in May 2023 at EFGB and WFGB.

Table 6.2. EFGB carbonate sample results for 2023 summarized at three depth gradients. Missing values were not calculated due to the lack of in situ temperature data.

Sample Date	Depth (m)	Salinity (ppt)	Temp (°C)	pH (Total)	Alkalinity (µmol/kg)	DIC (µmol/kg)	pH in situ	$\Omega_{aragonite}$	pCO_2 (µatm)
5/4/2023	20	36.10	23.90	7.988	2382.6	2025.9	8.054	n/a	n/a
5/4/2023	10	36.20	23.90	8.038	2387.0	2112.1	8.045	3.54	406.8
5/4/2023	1	36.20	24.00	8.033	2398.8	2118.2	8.058	3.53	414.5
8/17/2023	20	36.40	31.40	8.082	2398.8	2057.4	7.986	3.98	476.4
8/17/2023	10	36.30	31.60	8.089	2400.1	2055.5	7.990	4.04	471.0
8/17/2023	1	36.40	31.80	8.093	2408.3	2053.2	7.991	4.08	469.6

Table 6.3. WFGB carbonate sample results for 2023 summarized at three depth gradients. Missing values were not calculated due to the lack of *in situ* temperature.

Sample Date	Depth (m)	Salinity (ppt)	Temp (°C)	pH (Total)	Alkalinity (µmol/kg)	DIC (µmol/kg)	pH in situ	$\Omega_{aragonite}$	pCO_2 (µatm)
5/4/2023	20	36.10	23.60	8.037	2392.7	2092.9	8.058	3.5	398.6
5/4/2023	10	36.20	24.00	7.888	2407.1	2185.4	n/a	n/a	n/a
5/4/2023	1	36.20	24.00	8.037	2395.5	2105.9	8.082	3.54	408.0
8/17/2023	20	36.60	30.50	8.082	2401.2	2030.2	8.000	3.91	452.8
8/17/2023	10	36.50	30.50	8.083	2404.1	2037.3	8.001	3.93	453.9
8/17/2023	1	36.60	30.50	8.083	2395.0	2085.1	8.000	4.03	464.9

Water Quality Discussion

Limited water quality field work occurred in 2023 due to vessel mechanical malfunctions that required repairs. Seawater temperatures in 2023 were notably warm, exceeding the historical bleaching threshold in August and September, which led to coral bleaching. Significant monotonic increasing seawater temperature trends from 1990 to 2023 were observed at both banks, indicating that ocean temperatures at FGBNMS have risen over the past three decades, thereby heightening the likelihood of future bleaching events. No tropical storms or hurricanes impacted the temperature regimes over the banks in 2023.

Mean SSS fluctuated considerably at both banks, with reef cap salinity remaining consistent with long-term averages during the first half of the sampling period, but rising slightly above average in the latter half. Significant monotonic decreasing salinity trends were detected at EFGB from 2008 to 2023. Despite annual variation, salinity at depth remained within the normal range for coral reefs in the Western Atlantic (31–38 psu; Coles & Jokiel, 1992). The probable source of low-salinity water at the banks is a nearshore river-seawater mix occasionally extending to the outer continental shelf, primarily originating from the Mississippi and Atchafalaya river watersheds, exposing the banks to conditions usually restricted to waters closer to shore (Zimmer et al., 2010). Nutrient levels remained below detection limits in 2023.

The water column connects coral reef habitats, as well as aquatic and terrestrial systems. Thus, water quality data are critical components of monitoring programs, as they provide information on the incursion of land-based materials that affect critical coral reef ecosystem functions. Although not all quarterly water data were collected in 2023, most surface and reef cap temperature data were captured due to the recent installation of the Sofar Ocean® buoy at EFGB. The long battery life and robust sensors of the moored SBE 16plus and HOBO instruments helped prevent large data gaps; however, extended deployment periods led to sensor malfunctions, resulting in some data corruption. The integration of satellite data also contributed valuable surface parameters for time-series analysis.

Chapter 7: Conclusions

A golden smooth trunkfish (*Lactophrys triqueter*) swims among the reef. Photo: John Windemiller/John Windemiller Photography

Despite coral cover declines on most coral reefs of the world in recent decades, mean coral cover within EFGB and WFGB long-term monitoring study sites has ranged from 40–60% for the combined 34 years of monitoring. Even with macroalgae percent cover increasing after the mass mortality of *D. antillarum* in the 1980s, 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 study sites.

Reef-wide transect surveys (in partnership with NCRMP) were conducted in 2023 as part of the long-term monitoring program, allowing for benthic cover calculations in a random stratified design across the reef caps, not just the study sites. Although coral cover in repetitive photostations and random transect surveys is not comparable, the former is critical in enabling researchers to track individual locations over time (especially during extreme events such as the coral bleaching event). The long-term monitoring program benefits from having both random benthic surveys and repetitive monitoring stations.

The uniform biomass distribution in the reef fish community across large and small individuals suggests a minimally impacted assemblage. The comparatively low biomass and diversity among some groups of fish may reflect recruitment limitation due to the isolated nature of the banks and the lack of nearby source populations and nursery areas. Though non-native species (lionfish and regal demoiselles) are present and abundant in nearby areas, neither appears to be at levels that have affected native populations at the Flower Garden Banks. The abundance of both species, however, can increase rapidly, and thus both have the potential to pose significant problems. In the case of lionfish, continued culling is the primary recommended remediation measure. At present, no practical controls for regal demoiselles are known.

The 2023 bleaching event at FGBNMS provided critical insights into the impacts of prolonged elevated seawater temperatures on coral health and the influence of variation at a local scale on bleaching severity. Seawater temperatures on the reef caps exceeded 30 °C for prolonged periods, resulting in bleaching and paling at both EFGB and WFGB. EFGB experienced higher prevalence of bleaching and paling, reflecting greater coral stress. A significantly increasing seawater temperature trend from 1990 to 2023 was detected at both banks, raising concerns that future bleaching events are likely to occur as ocean temperatures continue to rise. The use of two approaches to evaluate bleaching and paling prevalence proved important for assessing coral stress. These findings highlight the growing thermal stress facing FGBNMS, the value of monitoring in enhancing our understanding of the threat, and the need to become more proactive if we hope to protect coral reefs from predicted climate change impacts.

The FGBNMS long-term monitoring program is one of the longest-running coral reef monitoring efforts in the world. For over three decades, it has been a critical tool for understanding the drivers of ecosystem variation at the Flower Garden Banks (Karnauskas et al., 2015). It has also helped FGBNMS and other authorities preserve the characteristics that sustain the banks' health and has alerted managers to ongoing and impending changes, enabling timely responses and actions. The monitoring program has been a guiding force for both conservation science and informed management since it began, and continues to support sanctuary education and outreach programs. And although monitoring is sounding alarms about concerning changes, particularly with regard to climate change, it also highlights the value of reducing direct human stressors to ensure a fully functioning coral reef ecosystem that is both resistant and resilient. The sanctuary's healthy reefs serve as a valuable site for the study of natural ecosystem processes, support numerous ecosystem services for visitors and residents, and could become a valuable source of genetic material to support the future restoration of degraded coral reefs in the region.

Acknowledgements

Many groups and individuals provided invaluable support to this monitoring effort, including BOEM, CPC, Inc., Texas A&M University Galveston, Moody Gardens Aquarium, Audubon Aquarium, Texas State Aquarium, the National Marine Sanctuary Foundation, NOAA Coast Watch, NCRMP, Rice University, Florida Atlantic University Harbor Branch Oceanographic Institute, and the NOAA Dive Center. In particular, we acknowledge Dr. Alicia Caporaso (BOEM) for her support and dedication to this project, Dr. Adrienne Correa (University of California Berkley) for coral disease response support, and Dr. Xinping Hu (TAMU-CC) for providing ocean carbonate data analysis. Finally, our sincere thanks are extended to the editors and reviewers who helped improve this report.

Researchers and volunteers who assisted with field operations, data collection, and/or data processing include: Amy Apprill, Aaron Bouwkamp, Marilyn Brandt, Kait Brogan, Kaitlin Buhler, Gaby Carpenter, Adrienne Correa, Jacque Cresswell, Olivia Eisenbach, Ben Farmer, Nick Farmer, Donavon French, Alex Good, Ryan Hannum, Josh Harvey, Michelle Johnston, Mikey Kent, Alyssa Lawton Chris Ledford, Daniel Lippi, Liz Maxwell, Sonora Meiling, Shelby Miskell, Marissa Nuttall, Kelly O'Connell, Terry Palmer, Jessica Pate, Jason Quetel, Ashley Rossin, Michael Studivan, Hannah Swain, Kara Titus, Bryant Turffs, Jenny Vander Pluym, and Hang Yin. The R/V *Manta* crew includes Justin Blake, Jorge Jaime, Tom McGinley, Julia Messex, and Taylor Philip. This study was partially funded through an interagency agreement between BOEM and the NOAA Office of National Marine Sanctuaries, through FGBNMS under contract number M19PG0001. Field work in 2023 was carried out under permit FGBNMS-2019-001.

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	dissolved inorganic carbon
EFGB	East Flower Garden Bank
FGBNMS	Flower Garden Banks National Marine Sanctuary
NCRMP	National Coral Reef Monitoring Program
NOAA	National Oceanic and Atmospheric Administration
PCO	principal coordinates ordination
$p\text{CO}_2$	partial pressure of CO_2
PERMANOVA	permutational multivariate analysis of variance
QA/QC	quality assurance/quality control
SE	standard error
SIMPER	similarity percentages
SIMPROF	similarity profile analysis
SSS	sea surface salinity
SST	sea surface temperature
TAMU	Texas A&M University
TAMU-CC	Texas A&M University Corpus Christi
TKN	total Kjeldahl nitrogen
WFGB	West Flower Garden Bank

Literature Cited

Anderson, M. J., Gorley, R. N., & Clarke, K. R. (2008). *PERMANOVA+ for PRIMER: Guide to software and statistical methods*. PRIMER-E Ltd.

Aronson, R. B., Edmunds, P. J., Precht, W. F., Swanson, D. W., & Levitan, D. R. (1994). Large-scale, long-term monitoring of Caribbean coral reefs: Simple, quick, inexpensive methods. *Atoll Research Bulletin*, *421*, 1–19. https://doi.org/10.5479/si.00775630.421.1

Aronson, R. B., & Precht, W. F. (2000). Herbivory and algal dynamics on the coral reef at Discovery Bay, Jamaica. *Limnology and Oceanography*, *45*, 251–255. https://doi.org/10.4319/lo.2000.45.1.0251

Aronson, R. B., Precht, W. F., Murdoch, T. J., & Robbart, M. L. (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. https://doi.org/10.18785/goms.2301.06

Atlantic and Gulf Rapid Reef Assessment. (2024). *AGRRA fish dashboard*. https://oref.maps.arcgis.com/apps/dashboards/ee6c8c8d6dbc49dcb05310392489f90a

Bauer, L., Zitello, A., Hile, S. D., & McGrath, T. (2015a). *Biogeographic characterization of fish and benthic communities, Jobos Bay, Puerto Rico 2009-06-08 to 2009-06-13 (NODC Accession 0125200)* [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Bauer, L., Hile, S. D., & McGrath, T. (2015b). *Biogeographic characterization of fish and benthic communities, Vieques, Puerto Rico 2007-05-14 to 2007-05-24 (NODC Accession 0125235)* [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Bauer, L., Hile, S. D., & McGrath, T. (2015c). *Biogeographic characterization of fish and benthic communities, St Thomas, US Virgin Islands 2012-06-12 to 2012-06-22 (NODC Accession 0125418)* [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Berkelmans, R. (2002). Time-integrated thermal bleaching thresholds of reefs and their variation on the Great Barrier Reef. *Marine Ecology Progress Weries, 229*, 73–82. https://doi.org/10.3354/meps229073

Blakeway, R. D., Fogg, A. Q., Johnston, M. A., Rooker, J. R., & Jones, G. A. (2022). Key life history attributes and removal efforts of invasive lionfish (*Pterois volitans*) in the Flower Garden Banks National Marine Sanctuary, Northwestern Gulf of Mexico. *Frontiers in Marine Science*, *9*, 774407. https://doi.org/10.3389/fmars.2022.774407

Bohnsack, J. A., & Bannerot, S. P. (1986). *A stationary visual technique for quantitatively assessing community structure of coral reef fishes*. NOAA Technical Report NMFS 41. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Bohnsack, J. A., & Harper, D. E. (1988). *Length-weight relationships of selected marine reef fishes from southeastern United States and the Caribbean*. NOAA Technical Memorandum NMFS-SEFC-215. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Bright, T. J., McGrail, D. W., Rezak, R., Boland, G. S., & Trippett, A. R. (1985). *The Flower Gardens: A compendium of information*. OCS Study MMS 85-0024. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Bruckner, A. W., & Bruckner, R. J. (1998). Destruction of coral by *Sparisoma viride*. *Coral Reefs*, 17, 350. https://doi.org/10.1007/s003380050138

Bruckner, A. W., Bruckner, R. J., & Sollins, P. (2000). Parrotfish predation on live coral: "Spot biting" and "focused biting." *Coral Reefs*, *19*, 50. https://doi.org/10.1007/s003380050225

Caldow, C., Clark, R., Edwards, K., Hile, S. D., Menza, C., Hickerson, E., & Schmahl, G. P. (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. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

Caldow, C., Roberson, K., Bauer, L., Jeffrey, C. F. G., Hile, S. D., & McGrath, T. (2015). *Biogeographic characterization of fish and benthic communities, Parguera Region, Puerto Rico 2000-08-21 to 2010-09-21 (NODC Accession 0125202)* [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Carpenter R.C., & Edmunds, P. J. (2006). Local and regional scale recovery of *Diadema* promotes recruitment of scleractinian corals. *Ecological Letters*, *9*(3), 268–277. https://doi.org/10.1111/j.1461-0248.2005.00866.x

Clark, R., Taylor, J. C., Buckel, C. A., & Kracklet, L. M. (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. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

Clark, R., Buckel, C. A., Taylor, C., Hile, S. D., & McGrath, T. (2015a). *Biogeographic characterization of fish and benthic communities, Flower Garden Banks, Texas 2010-09-10 to 2012-10-02* (NODC Accession 0118358) [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Clark, R., Hile, S. D., & McGrath, T. (2015b). *Biogeographic characterization of fish and benthic communities, St Croix, US Virgin Islands 2012-05-07 to 2012-05-18* (NODC Accession 0125237) [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Oceanographic Data Center.

Clarke, K. R. (1990). Comparisons of dominance curves. *Journal of Experimental Marine Biology and Ecology*, *138*, 143–157. https://doi.org/10.1016/0022-0981(90)90181-B

Clarke, K. R., Somerfield, P. J., & Gorley, R. N. (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. https://doi.org/10.1016/j.jembe.2008.07.009

Clarke, K . R., Gorley, R. N., Sommerfield, P. J., & Warwick, R. M. (2014). *Change in marine communities: An approach to statistical analysis and interpretation* (3rd edition). PRIMER-E.

Cody, T. T., Kiryu, Y., Bakenhaster, M. D., Subramaniam, K., Tabuchi, M., Ahasan, M. S., Harris, H. E., Landsberg, J. H., Waltzek, T. B., Fogg, A. Q., Shea, C., Pouder, D. B., Patterson III, W. F., Emory, M. E., & Yanong, R. P. (2023). Cutaneous ulcerative lesions of unknown etiology affecting lionfish *Pterois* spp. in the Gulf of Mexico. *Journal of Aquatic Animal Health, 35*(1), 20–33. https://doi.org/10.1002/aah.10174

Coles, S. L., & Jokiel, P. L. (1992). Effects of salinity on coral reefs. In D. W. Connell & D. W. Hawker (Eds.), *Pollution in tropical aquatic systems* (pp. 147–166). CRC Press.

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*. Continental Shelf Associates.

Continental Shelf Associates. (1996). *Long-term monitoring at the East and West Flower Garden Banks*. OCS Study MMS 96-0046. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Dahl, K. A., & Patterson, W. F. (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. https://doi.org/10.1371/journal.pone.0105852

Darling, E. S., Green, S. J., O'Leary, J. K, & Côté, I. M. (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. https://doi.org/10.1007/s10530-011-0020-0

DeBose, J. L., Nuttall, M. F., Hickerson, E. L., & Schmahl, G. P. (2012). A high-latitude coral community with an uncertain future: Stetson Bank, northwestern Gulf of Mexico. *Coral Reefs, 32*, 255–267. https://doi.org/10.1007/s00338-012-0971-3

DeMartini, E. E., Friedlander, A. M., Sandin, S. A., & Sala E. (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. https://doi.org/10.3354/meps07501

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

Dias, L. M., Johnston, M. A., O'Connell, K., Clift, L. W., Eisenbach, O., Hannum, R., Williams, K., French, D., Cannizzo, Z. J., & Hutto, S. (2023*). Rapid climate vulnerability assessment for Flower Garden Banks National Marine Sanctuary*. U.S. Department of Commerce, National Oceanographic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries, Flower Garden Banks National Marine Sanctuary. https://doi.org/10.25923/jdpb-zw04

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

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

Eakin, C. M., Morgan, J. A., Heron, S. F., Smith, T. B ., Liu, G., Alvarez-Filip, L., Baca, B., Bartels, E., Bastidas, C., Bouchon, C., Brandt, M., Bruckner, A. W., Bunkley-Williams, L., Cameron, A., Causey, B. D., Chiappone, M., Christensen, T. R. L., Crabbe, M. J. C., Day, O.,...& Yusuf, Y. (2010). Caribbean corals in crisis: Record thermal stress, bleaching, and mortality in 2005. *PLoS ONE*, *5*(11), e13969. https://doi.org/10.1371/journal.pone.0013969

Edmunds, P.J. & Carpenter, R. C. (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*(9), 5067–5071. https://doi.org/10.1073/pnas.071524598

Froese, R., & Pauly, D. (Eds.) (2019). *FishBase*. www.fishbase.org

Gardner, T. A., Côté, I. M, Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. *Science, 301*, 958–960. https://doi.org/10.1126/science.1086050

Gittings, S. R., & Bright, T. J. (1986). *Assessment of coral recovery following an incident of anchoring damage at the East Flower Garden Bank, Northwest Gulf of Mexico*. Contract No. NA85AA-H-CZ015. Final Report to NOAA's National Marine Sanctuaries Division.

Gittings, S. R., & Bright, T. J. (1987). Mass mortality of *Diadema antillarum* at the Flower Garden banks, northwest Gulf of Mexico: effect on algae and coral cover [Conference paper]. Benthic Ecology Meetings, Raleigh, NC.

Gittings, S. R. (1998). Reef community stability on the Flower Garden Banks, northwest Gulf of Mexico. *Gulf of Mexico Science, 16*(2), 5. https://doi.org/10.18785/goms.1602.05

Gittings, S.R ., Boland, G. S., Deslarzes, K. J. P., Hagman, D. K., & Holland, B. S. (1992). *Long-term monitoring at the East and West Flower Garden Banks*. OCS Study MMS 92-0006. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Goreau, T. F., & Wells, J. W. (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, & Côté, I. M. (2009). Record densities of Indo-Pacific lionfish on Bahamian coral reefs. *Coral Reefs*, *28*, 107. https://doi.org/10.1007/s00338-008-0446-8

Green, S. J., Dulvy, N. K., Brooks, A. L. M., Akins, J. L., Cooper, A. B., Miller, S., & Côté, I. M. (2014). Linking removal targets to the ecological effects of invaders: a predictive model and field test. *Ecological Applications*, *24*, 1311 –1322. https://doi.org/10.1890/13-0979.1

Hagman, D. K., & Gittings, S. R. (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., Fogg, A. Q., Yanong, R. P., Frasca, S. Jr., Cody, T., Waltzek, T. B., & Patterson, W. F. (2018). *First report of an emerging ulcerative skin disease in invasive lionfish*. University of Florida-IFAS Extension. https://edis.ifas.ufl.edu/publication/FA209

Harris, H. E., Fogg, A. Q., Allen, M. S., Ahrens, R. N. M., & Patterson, W.F., III. (2020). Precipitous declines in northern Gulf of Mexico invasive lionfish populations following the emergence of an ulcerative skin disease. *Scientific Reports*, *10*, 1934. https://doi.org/10.1038/s41598-020-58886-8

Harris, H. E., Patterson III, W. F., Ahrens, R. N., Allen, M. S., Chagaris, D. D., & Larkin, S. L. (2023). The bioeconomic paradox of market-based invasive species harvest: A case study of the commercial lionfish fishery. *Biological Invasions, 25*(5), 1595–1612. https://doi.org/10.1007/s10530-023-02998-5

Helsel, D. R., & Hirsch, R. M. (2002). *Statistical methods in water resources*. U.S. Department of the Interior, U.S. Geological Survey. https://doi.org/10.3133/twri04A3

Helsel, D. R., Mueller, D. K., & Slack, J. R. (2006). *Computer program for the Kendall family of trend tests*. U.S. Department of the Interior, U.S. Geological Survey. https://doi.org/10.3133/sir20055275

Heron, S. F., Johnston, L., Liu, G., Geiger, E. F., Maynard, J. A., De La Cour, J. L., Okano, R., Benavente, D., Burgess, T. F., Iguel, J., Perez, D. I., Skirving, W. J., Strong, A. E., Tirak, K., & Eakin, C. M. (2016). Validation of reef-scale thermal stress satellite products for coral bleaching monitoring. *Remote Sensing, 8*(1), 59. https://doi.org/10.3390/rs8010059

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

Hu, C., & Feng, I. (2016). Modified MODIS fluorescence line height data product to improve image interpretation for red tide monitoring in the eastern Gulf of Mexico. *Journal of Applied Remote Sensing, 11*(1), 1–11. https://doi.org/10.1117/1.JRS.11.012003

Hughes, T. P., Kerry, J. T., Álvarez-Noriega, M., Álvarez-Romero, J. G., Anderson, K. D., Baird, A. H., Babcock, R. C., Beger, M., Bellwood, D. R., Berkelmans, R., Bridge, T. C., Butler, I. R., Byrne, M., Cantin, N. E., Comeau, S., Connolly, S. R., Cumming, G. S., Dalton, S. J., Diaz-Pulido, G.,... & Wilson, S. K. (2017). Global warming and recurrent mass bleaching of corals. *Nature, 543*(7645), 373–377. https://doi.org/10.1038/nature21707

Hughes, T. P., Kerry, J. T., Baird, A. H., Connolly, S. R., Dietzel, A., Eakin, C. M., Heron, S. F., Hoey, A. S., Hoogenboom, M. O., Liu, G., McWilliam, M. J., Pears, R. J., Pratchett, M. S., Skirving, W. J., Stella, J. S., & Torda, G. (2018). Global warming transforms coral reef assemblages. *Nature*, *556*(7702), 492–496. https://doi.org/10.1038/s41586-018-0041-2

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (Eds.) (2014). *Status and trends of Caribbean coral reefs: 1970–2012*. Global Coral Reef Monitoring Network.

Jarosz, E., Wijesekera, H. W., Teague, W. J., Fribance, D. B., & Moline, M. A. (2014). Observations on stratified flow over a bank at low Froude numbers. *Journal of Geophysical Research: Oceans, 119*(9), 6403–6421. https://doi.org/10.1002/2014JC009934

Johnston, M. A., Nuttall, M. F., Eckert, R. J., Embesi, J. A., Slowey, N. C., Hickerson, E. L., & Schmahl, G. P. (2013). *Long-term monitoring at the East and West Flower Garden Banks National Marine Sanctuary, 2009–2010, volume 1: Technical report*. OCS Study BOEM 2013-215. U.S. Department of the Interior, Bureau of Ocean Energy Management, Gulf of Mexico OCS Region.

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

Johnston, M. A., Nuttall, M. F., Eckert, R. J., Embesi, J. A., Sterne, T. K., Hickerson, E. L., & Schmahl, G. P. (2016a). Persistence of coral assemblages in Flower Garden Banks National Marine Sanctuary, Gulf of Mexico. *Coral Reefs*, *35*, 821–826. https://doi.org/10.1007/s00338-016-1452-x

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*(2), 115–122. https://doi.org/10.3391/bir.2016.5.2.09

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

Johnston, M. A., Sterne, T. K., Eckert, R. J., Nuttall, M. F., Embesi, J. A., Walker, R., Hu, X., Hickerson, E. L., & Schmahl, G. P. (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, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A., Sterne, T. K., Blakeway, R. D., MacMillan, J., Nuttall, M. F., Hu, X., Embesi, J. A., Hickerson, E. L., & Schmahl, G. P. (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, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A., Nuttall, M. F., Eckert, R. J., Blakeway, R. D., Sterne, T. K., Hickerson, E. L., Schmahl, G. P., Lee, M. T., MacMillan, J., & Embesi, J. A. (2018b). Localized coral reef mortality event at East Flower Garden Bank, Gulf of Mexico. *Bulletin of Marine Science*, *95*, 239–250. https://doi.org/10.5343/bms.2018.0057

Johnston, M. A., Hickerson, E. L., Nuttall, M. F., Blakeway, R. D., Sterne, T. K., Eckert, R. J., & Schmahl, G. P. (2019). Coral bleaching and recovery from 2016 to 2017 at East and West Flower Garden Banks, Gulf of Mexico. *Coral Reefs*, *38*, 787–799. https://doi.org/10.1007/s00338-019-01788-7

Johnston, M.A., Blakeway, R. D., O'Connell, K., MacMillan, J., Nuttall, M. F., Hu, X., Embesi, J. A., Hickerson, E. L., & Schmahl, G. P. (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, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A. (2021). *Strategy for stony coral tissue loss disease prevention and response at Flower Garden Banks National Marine Sanctuary*. National Marine Sanctuaries Conservation Series ONMS-21-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A., O'Connell, K., Blakeway, R. D., MacMillan, J., Nuttall, M. F., Hu, X., Embesi, J. A., Hickerson, E. L., & Schmahl, G. P. (2021). *Long-term monitoring at East and West Flower Garden Banks: 2019 annual report*. National Marine Sanctuaries Conservation Series ONMS-21-02. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A., O'Connell, K., Blakeway, R. D., Hannum, R., Nuttall, M. F., Hickerson, E. L., & Schmahl, G. P. (2022). *Long-term monitoring at East and West Flower Garden Banks: 2020 and 2021 annual report*. National Marine Sanctuaries Conservation Series ONMS-22-01. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

Johnston, M. A ., Studivan, M. S., Enochs, I. C., Correa, A., Besemer, N., Eckert, R. J., Edwards, K., Hannum, R., Hu, X., Nuttall, M., & O'Connell, K. (2023). Coral disease outbreak at the remote Flower Garden Banks, Gulf of Mexico. *Frontiers in Marine Science*, *10*, 1111749. https://doi.org/10.3389/fmars.2023.1111749

Johnston, M. A., French, D. R., Eisenbach, O. J., O'Connell, K., Hannum, R., Nuttall, M. F., & Emmert, J. (2024). *Long-term monitoring at East and West Flower Garden Banks: 2022 annual report*. National Marine Sanctuaries Conservation Series ONMS-24-06. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

JPL MUR MEaSUREs Project. (2015). *GHRSST Level 4 MUR global foundation sea surface temperature analysis, version 4.1* [Data set]. National Aeronautics and Space Administration, Physical Oceanography Distributed Active Archive Center. https://doi.org/10.5067/GHGMR-4FJ04

Karnauskas, M., Schirripa, M. J., Craig, J. K., Cook, G. S., Kelble, C. R., Agar, J. J., Black, B. A., Enfield, D. B., Lindo-Atichati, D., Muhling, B. A., Purcell, K. M., Richards, P. M., & Wang, C. (2015). Evidence of climate-driven ecosystem reorganization in the Gulf of Mexico. *Global Change Biology*, *21*, 2554–2568. https://doi.org/10.1111/gcb.12894

Knowlton, N., & Jackson, J. B. C. (2008). Shifting baselines, local impacts, and global change on coral reefs. *PLoS Biology*, *6*, e54. https://doi.org/10.1371/journal.pbio.0060054

Kohler, K. E., & Gill, S. M. (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. https://doi.org/10.1016/j.cageo.2005.11.009

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,* 1–57.

Lang, J. C., Marks, K. W., Kramer, P. A., Kramer, P. R., & Ginsburg, R. N. (2012). *AGRRA protocols, version 5.4*. Atlantic and Gulf Rapid Reef Assessment.

Limer, B. D., Bloomberg, J., & Holstein, D. M. (2020). The influence of eddies on coral larval retention in the Flower Garden Banks. *Frontiers in Marine Science, 7, 372*. https://doi.org/10.3389/fmars.2020.00372

Manzello, D. P., Brandt, M., Smith, T. B., Lirman, D., Hendee, J. C., & Nemeth, R. S. (2007). Hurricanes benefit bleached corals. *Proceedings of the National Academy of Sciences, 104*(29), 12035–12039.

Morris, J. A., Jr., & Whitfield, P. E. (2009). *Biology, ecology, control and management of the invasive Indo-Pacific lionfish: An updated integrated assessment*. NOAA Technical Memorandum NOS NCCOS 99. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

Muller-Karger, F. E., Smith, J. P., Werner, S., Chen, R., Roffer, M., Liu, Y., Muhling, B., Lindo-Atichati, D., Lamkin, J., Cerdeira-Estrada, S., & Enfield, D. B. (2015). Natural variability of surface oceanographic conditions in the offshore Gulf of Mexico. *Progress in Oceanography, 134*, 54–76. https://doi.org/10.1016/j.pocean.2014.12.007

Mumby, P. J., Edwards, A. J., Arias-González, J. E., Kindeman, K. C., Blackwell, P. G., Gall, A., Gorczynska, M. I., Harborne, A. R., Pescod, C. L., Renken, H., Wabnitz, C. C. C., & Llewellyn, G. (2004). Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature*, *427*, 533–536. https://doi.org/10.1038/nature02286

Mumby, P. J., & Steneck, R. S. (2011). The resilience of coral reefs and its implications for reef management. In Z. Dubinsky & N. Stambler (Eds.), *Coral reefs: An ecosystem in transition* (pp. 509–519). Springer Press. https://doi.org/10.1007/978-94-007-0114-4_29

NASA. (2023). *Ocean color feature: World map of chlorophyll-a* [Data set]. National Aeronautics and Space Administration. https://oceancolor.gsfc.nasa.gov/

National Centers for Coastal Ocean Science. (2018). *National Coral Reef Monitoring Program: Assessment of coral reef benthic communities in Flower Garden Banks National Marine Sanctuary* [Data set]. https://doi.org/10.7289/v5vd6wts

NOAA Coral Reef Watch. (2023). *NOAA Coral Reef Watch version 3.1 daily global 5-km satellite sea surface temperature product, January 1, 2022–December 31, 2023* [Data set]. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Environmental Satellite, Data, and Information Service, Center for Satellite Applications and Research. https://coralreefwatch.noaa.gov/satellite/index.php

Nuttall, M. F., Johnston, M. A., Eckert, R. J., Embesi, J. A., Hickerson, E. L., & Schmahl, G. P. (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*(2), 111–115. http://dx.doi.org/10.3391/bir.2014.3.2.09

Nuttall, M.bF., Somerfield, P. J., Sterne, T. K., MacMillan, J. T., Embesi, J. A., Hickerson, E. L., Johnston, M. A., Schmahl, G. P., & Sinclair, J. (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, National Ocean Service, Office of National Marine Sanctuaries.

O'Connell, K., Hannum, R., Eisenbach, O., French, D., Nuttall, M. F., Johnston, M., Hu, X., & Taylor, T. (2024). *Stetson Bank long-term monitoring: 2023 annual report*. National Marine Sanctuaries Conservation Series ONMS-24-05. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of National Marine Sanctuaries.

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, National Ocean Service.

Otis, D. (2023). *Flower Garden Banks National Marine Sanctuary data dashboard* [Data set]. http://fgbnms-dashboard.marine.usf.edu:3000

Precht, W. F., Aronson, R. B., Deslarzes, K. J. P., Robbart, M. L., Gelber, A., Evans, D., Gearheart, B., & Zimmer, B. (2006). *Long-term monitoring at the East and West Flower Garden Banks, 2002–2003: Final report*. OCS Study MMS 2004–031. U.S. Department of the Interior, Minerals Management Service.

Precht, W. F., Aronson, R. B., Deslarzes, K. J. P., Kaufman, L. S., Robbart, M. L., Hickerson, E. L., Schmahl, G. P., & Sinclair, J. (2008). Long-term reef monitoring at the Flower Garden Banks: Status, stasis, and change. In K. B. Ritchie & B. D. Keller (Eds.), *A scientific forum on the Gulf of Mexico: The islands in the stream concept* (p. 41–44). Marine Sanctuaries Conservation Series NMSP-08-04. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean and Coastal Resource Management, National Marine Sanctuary Program. https://nmssanctuaries.blob.core.windows.net/sanctuaries-prod/media/archive/science/conservation/pdfs/gom.pdf

Reef Environmental Education Foundation. (2014). *Geographic zone report: Flower Gardens* [Data set]. https://www.reef.org/db/reports/geo?end_date=2024-12-23&format_type=chart&group_type=species&language=common&region_code=TWA&start_date=1993-01-01&zone_map=0&zones=2402

Reef Environmental Education Foundation. (2023). *Batch report—Flower Garden Banks NMS 2023 field survey trip* [Data set]. https://www.reef.org/db/reports/batch/fgbnms23.dat

Rezak, R., Bright, T. J., & McGrail, D. W. (1985). *Reefs and banks of the northwestern Gulf of Mexico: Their geological, biological, and physical dynamics*. John Wiley and Sons.

Roberson, K., Viehman, S., & Clark, R. (2014). *Development of benthic and fish monitoring protocols for the Atlantic/Caribbean Biological Team: National Coral Reef Monitoring Program*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Coral Reef Conservation Program.

Robertson, D. R., Simoes, N., Gutiérrez Rodríguez, C., Piñeros, V. J., & Pérez-España, H. (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., Dominguez-Dominguez, O., Victor, B., & Simoes, N. (2018). An Indo-Pacific damselfish (*Neopomacentrus cyanomos*) in the Gulf of Mexico: Origin and mode of introduction. *PeerJ*, *6*, e4328. https://doi.org/10.7717/peerj.4328

Rooker, J. R., Dokken, Q. R., Pattengill, C.V., & Holt G. J. (1997). Fish assemblages on artificial and natural reefs in the Flower Garden Banks National Marine Sanctuary, USA. *Coral Reefs*, *16*, 83–92. https://doi.org/10.1007/s003380050062

Sale, P. F. (1991). *The ecology of fishes on coral reefs*. Academic Press, Inc.

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

Southern Ocean Knowledge and Information Wiki. (2014). *Abundance biomass curve (ABC method) – Indicators*. Antarctic Climate and Ecosystems Co-operative Research Centre. http://www.soki.aq/x/foFm

U.S. Geological Survey. (2023). *National Water Information System: Web interface* [Data set]. https://waterdata.usgs.gov/nwis

Wetmore, L .S., Dance, M. A., Hill, R. L., & Rooker, J. (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*, 152. https://doi.org/10.3389/fmars.2020.00152

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

NATIONAL MARINE
SANCTUARIES

America's Underwater Treasures