



Coral Reef Long-Term Monitoring at East and West Flower Garden Banks: 2024 Annual Report



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Cover photo: Healthy colony of *Orbicella faveolata* at East Flower Garden Bank. Photo: Tiffany Crumbley/Crumbley Photography



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Report Availability

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Abstract

This report summarizes fish and benthic community observations and water quality data collected from the coral reefs of East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB) in 2024, alongside 35 years of historical long-term 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 America. The annual coral reef long-term monitoring program, initiated in 1989, is funded by FGBNMS and the Bureau of Ocean Energy Management, with support from the National Marine Sanctuary Foundation. In 2024, mean coral cover was 56%, estimated from random transect data within EFGB and WFGB one-hectare study sites. Coral cover at WFGB has increased significantly over time, while EFGB values have remained stable. The *Orbicella* spp. complex, listed as threatened under the Endangered Species Act, accounted for the majority of the coral cover within the study sites. Benthic composition across the coral reef differed significantly between the banks, largely due to differences in macroalgae and colonizable substrate cover. While mean macroalgae cover was 24% within the EFGB study site and 17% within the WFGB study site, macroalgae cover has increased significantly at both banks since 1998. The reef fish community was dominated in terms of relative abundance by the families Labridae (wrasses) and Pomacentridae (damsel-fishes). Biomass was evenly distributed between large and small individuals, and herbivores represented the greatest mean biomass at both banks, marking a notable shift from previous years when piscivores dominated total biomass. No non-native regal demoiselles were observed in 2024; however, they were observed on nearby artificial reef sites and Stetson Bank. A significant increase in reef-cap seawater temperature was detected at both banks from 1990 to 2024. In situ reef-depth temperature data indicated that EFGB approached, while WFGB exceeded, site-specific bleaching thresholds in 2024. Coral paling was first observed in July, with bleaching more pronounced later in the season. At WFGB, 10.3% of coral cover was bleached and 46.9% was pale by late September. The 2024 bleaching event followed a major bleaching event in 2023 and is the first instance of consecutive bleaching years recorded at FGBNMS. The Flower Garden Banks remains one of very few reefs in the Western Atlantic Ocean where coral reefs appear to have resisted degradation over the last several decades. Continued long-term monitoring will be essential for understanding the mechanisms underlying this apparent resistance and for supporting effective sanctuary management.

Key Words

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

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



A group of scuba divers consult maps as they search for long-term monitoring station markers on the coral reef cap at East Flower Garden Bank. Photo: Jesse Cancelmo

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 reefs and banks along the outer continental shelf in the northwestern Gulf of America (Bright et al., 1985; Figure 1.1). These reefs are located approximately 20 km apart and occupy the surface expression of salt dome formations located approximately 190 km south of the Texas and Louisiana border. While each bank is comprised of several distinct habitats ranging in depth from 16–166 m (Rezak et al., 1985; Schmahl et al., 2008; Figure 1.1), this study focuses on the shallow coral reefs from 16–39 m.

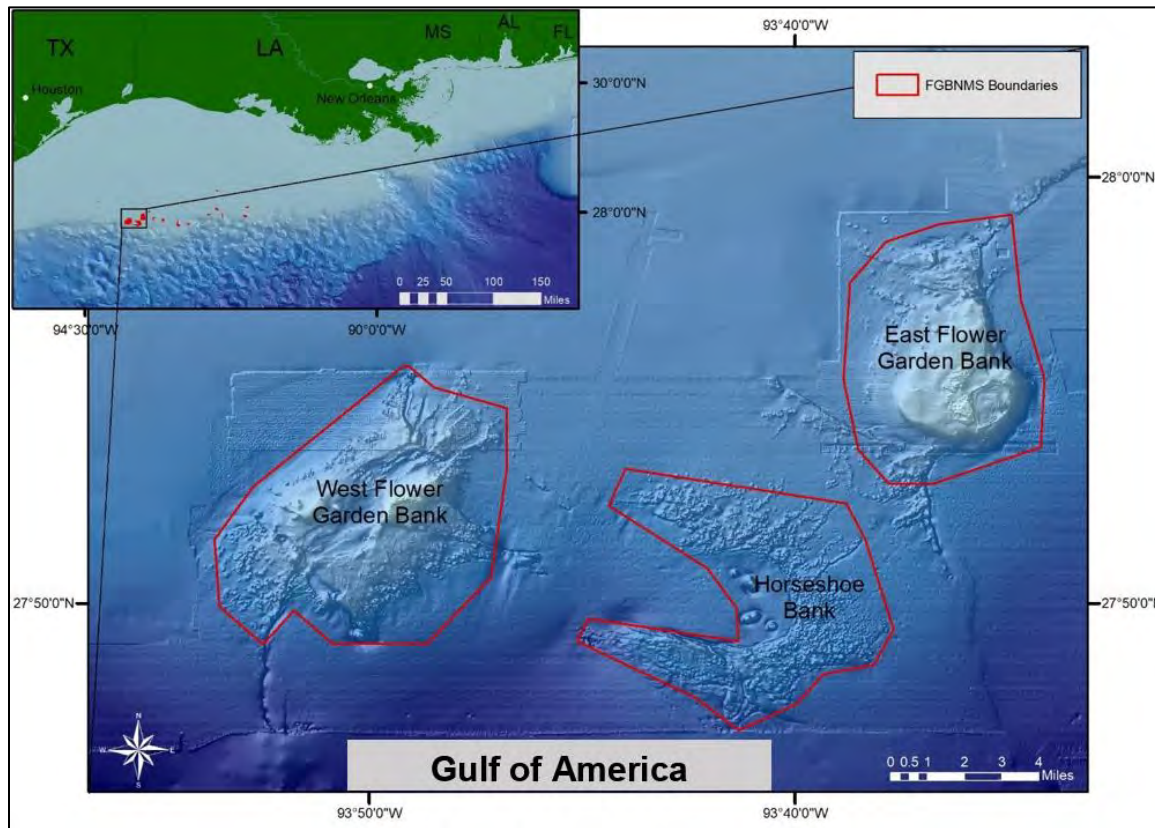


Figure 1.1. Map of EFGB, WFGB, and Horseshoe Bank with an inset of the Gulf Coast states and FGBNMS boundaries along the continental shelf of the northwestern Gulf of America. Horseshoe Bank is part of FGBNMS, but doesn't have a shallow coral reef. Image: Marissa Nuttall/NOAA

The coral caps of the banks are 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 coral 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 (Aronson et al., 2005; Bright et al., 1985; Continental Shelf Associates, 1989; Johnston et al., 2019); FGBNMS is near the northernmost

limit of coral reef development and is distanced from source populations by several hundred kilometers.

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 America (86 Fed. Reg. 4937 [Jan 19, 2021]).

Coral Reef 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 due to a joint responsibility with FGBNMS to protect these unique resources and to evaluate impacts to the reef by nearby oil and gas activities (Figure 1.2).

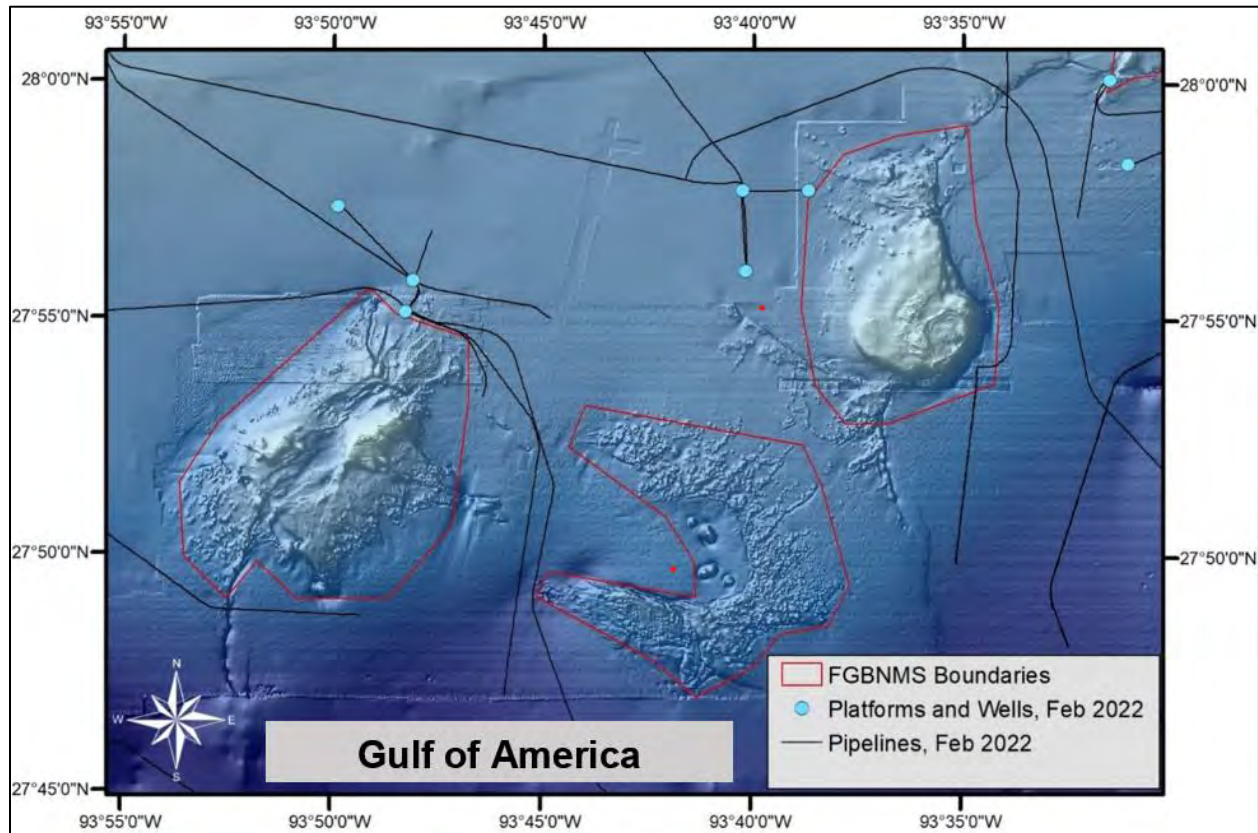


Figure 1.2. Map of oil and gas platforms, wells, and pipelines near EFGB and WFGB as of February 2022. Image: Marissa Nuttall/NOAA

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 on each bank in 1989 were permanently established, marking the start of Flower Garden Banks long-term monitoring program (Continental Shelf Associates, 1989; Gittings et al., 1992; Figure 1.3).

These study sites consist of fixed 10,000 m² reef areas (100 m × 100 m) that are revisited annually using standardized methods to collect benthic, fish, and water quality data, providing a consistent spatial framework for detecting long-term ecological change. 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.

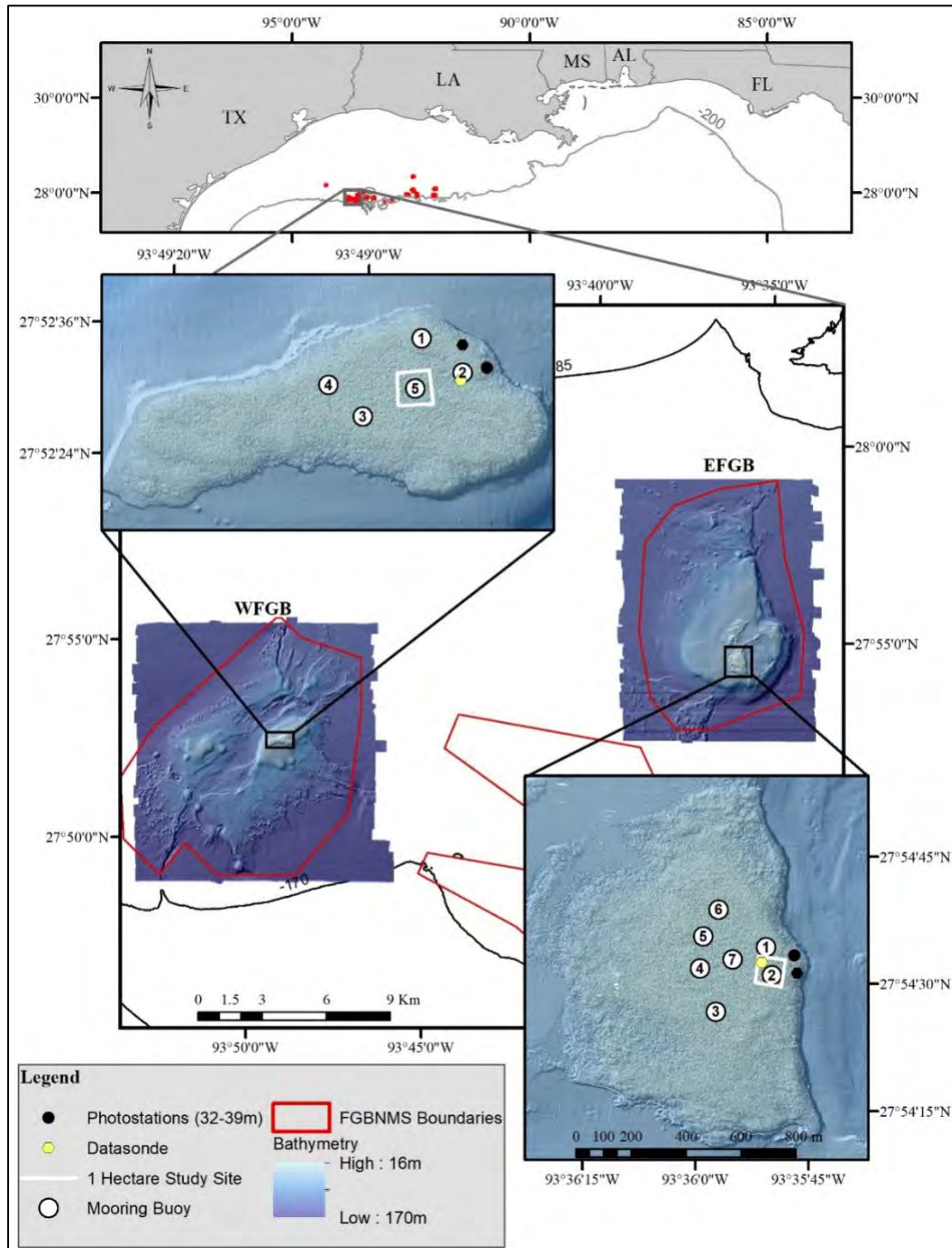


Figure 1.3. Shaded relief maps of EFGB and WFGB, with inset of the Gulf of America coastline, long-term monitoring one-hectare study sites, and datasonde and repetitive photostation locations, which range in depth from 32–39 m. Image: Marissa Nuttall/NOAA

Coral Reef Long-Term Monitoring Program Objectives

The National Marine Sanctuaries Act guides resource management at FGBNMS, including identifying threats and potential sources of impacts to the resources, such as 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.

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 shallow coral reef 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

Coral Reef Long-Term Monitoring Program Components

The shallow coral reef long-term monitoring program was designed to assess coral reef health, detect change over time, and provide baseline data in the event that natural or human-induced activities alter the integrity of EFGB and WFGB shallow reef 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 benchmark 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 to document benthic cover;
- Repetitive photostations to understand mechanism of change;
- Biennial coral demographic surveys to provide information on recruitment, coral density, and coral colony size;
- Stationary reef fish visual census surveys to assess community structure of coral reef fishes;

- Long-spined sea urchin (*Diadema antillarum*) and lobster (*Panulirus argus* and *P. guttatus*) surveys to establish current population levels and trends;
- Water quality datasondes to record salinity, temperature, and turbidity at depth; and
- Quarterly sampling of chlorophyll *a*, ammonia, nitrate, nitrite, total Kjeldahl nitrogen, and phosphorus to document water column productivity.

The long-term monitoring study area consists of several locations on the EFGB and WFGB coral caps where benthic, fish, and water quality data are collected. Data have been collected annually during summer months since 1989 at both sites. The corners and centers of each study site are marked by large eyebolts that serve 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 typically attached only during field research activities; however, due to operational constraints following the COVID-19 pandemic, buoys have remained in place in recent years (2021–2024). In addition to the study sites, 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, within the study site at 23 m depth, and WFGB, near the study site at 27 m depth (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 area at each bank.

Mooring	Lat (DDM)	Long (DDM)	Depth	Description
EFGB Mooring #2	27° 54.516' N	93° 35.831' W	19.2	Within study study, datasonde, access to deep photostations
WFGB Mooring #5	27° 52.509' N	93° 48.900' W	20.7	Within study site
WFGB Mooring #2	27° 52.526' N	93° 48.836' W	24.4	Datasonde, access to deep photostations

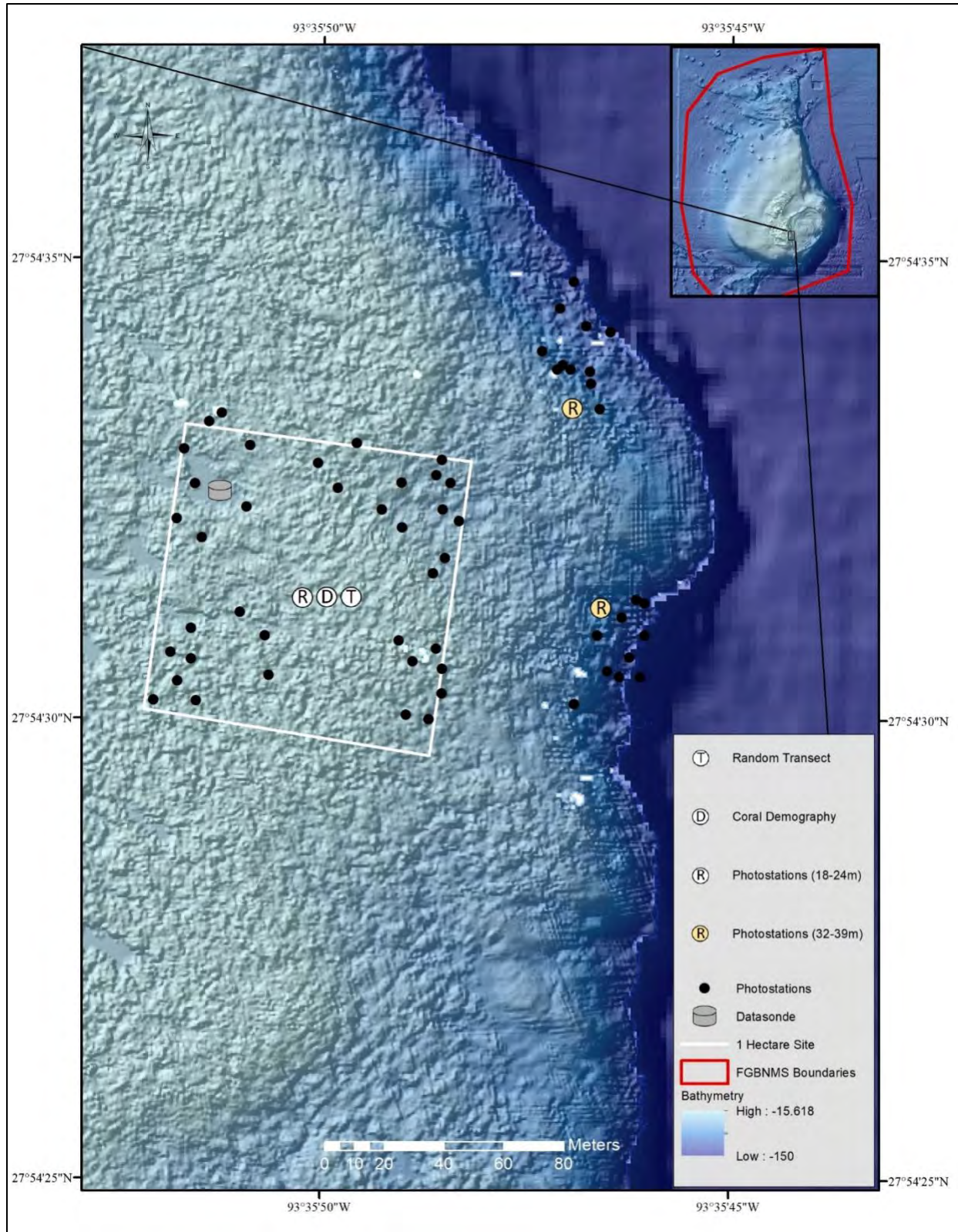


Figure 1.4. Shaded relief map of EFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also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

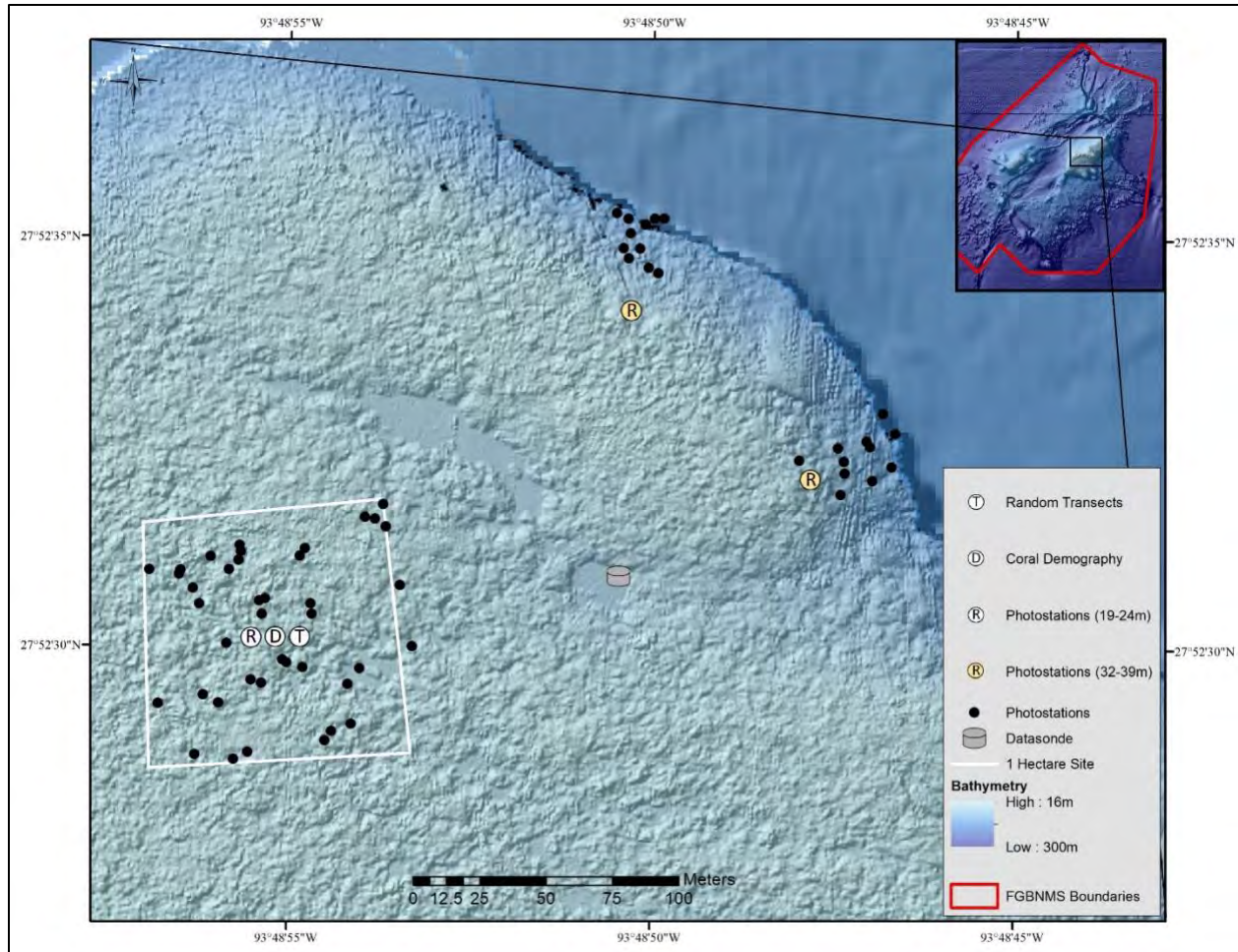


Figure 1.5. Shaded relief map of WFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also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

Coral Reef Long-Term Monitoring Field Operations and Data Collection

To date, the coral reef monitoring program has spanned 35 years, with annual collection occurring nearly continuously over this time. In 2024, long-term monitoring data were collected at both EFGB and WFGB, with scuba operations conducted from the *M/V Fling* (Table 1.2). Water quality instruments were exchanged, and data were downloaded by FGBNMS staff during field operations in July (Table 1.2). Detailed field operation methods are provided in each chapter.

Annual fieldwork at EFGB and WFGB were conducted over two cruises from July 1–4 and July 15–17, 2024 (Table 1.2), under favorable conditions, including mild currents (>0.5 kt), 20 m visibility, 29 °C water temperatures, and 2–3 foot seas. With the *R/V Manta* unavailable due to extended maintenance throughout 2024, all long-term monitoring activities were completed opportunistically aboard the *M/V Fling*. Most monitoring objectives were met; however, deep

repetitive photostations were not photographed. Datasondes at EFGB and WFGB were retrieved, downloaded, and redeployed. Temperature sensors at deep repetitive photostations were retrieved in July, 2025.

Coral paling was first observed in July during long-term monitoring cruises. Reef-wide random transect surveys were conducted at EFGB during a joint FGBNMS and National Coral Reef Monitoring Program (NCRMP) cruise from July 29–August 2, 2024. Peak thermal stress occurred in mid to late-August, when reef temperatures remained elevated for extended periods. A subsequent drilling cruise from September 22–25 provided an opportunity to photograph repetitive photostations at WFGB to assess bleaching later in the season. Time, funding, and vessel constraints limited additional data collection during the peak of thermal stress, particularly at EFGB.

The R/V *Manta* entered its annual shipyard repair period on September 30, 2023, and remained in maintenance throughout 2024. As a result, dedicated water quality cruises were not conducted during the year.

Table 1.2. Monitoring cruises completed at EFGB and WFGB in 2024.

Date	Cruise and Tasks Completed
06/30–07/05/2024	Long-term monitoring cruise on the <i>M/V Fling</i> : WFGB annual monitoring
07/14–17/2024	Long-term monitoring cruise on the <i>M/V Fling</i> : EFGB annual monitoring and water quality instrumentation exchange
07/29–08/02/2024	NCRMP and FGBNMS cruise on the <i>M/V Fling</i> : Reef-wide random transect surveys collected at EFGB and WFGB
09/22–25/2024	Drilling cruise on the <i>M/V Fling</i> : Repetitive photostations were opportunistically photographed to analyze bleaching

Chapter 2: Benthic Community



Vibrant sponges and encrusting organisms add to the coral's structural complexity at WFGB. Photo: Donavon French/CPC

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 have been conducted in a random stratified design across the coral caps to assess benthic cover outside of the EFGB and WFGB study sites in 2024. A total of 14 reef-wide transects were completed at EFGB, allowing for qualitative comparison with study site data; however, due to inclement weather, only two reef-wide transects were completed at WFGB, limiting the ability to make meaningful comparisons between methods at WFGB.

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. These stations are not intended to estimate reef-wide populations or communities and instead used to help identify mechanisms of change.

Benthic Community Methods

Random Transect Field Methods

In 2024, 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 m²). This produced a total photographed area of 8.16 m² per transect, and a minimum of 130.56 m² 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, 2021b, 2022, 2024). Consistency for photographic random transect methods was ensured by training all scientific divers in the proper operation of all the camera system from a standardized operating procedure. 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.



Figure 2.1. A diver conducts a random transect at EFGB. Photo: Ryan Hannum/CPC

In addition to random transect surveys inside the EFGB and WFGB study sites, transect surveys ($n = 14$ at EFGB, $n = 2$ at WFGB) were conducted in a random stratified design across both coral caps, in conjunction with NCRMP methods to assess benthic cover outside of the study sites and cross comparison between FGBNMS and NCRMP methods (National Centers for Coastal Ocean Science, 2018). These surveys were conducted in partnership with the NCRMP cruise aboard the M/V *Fling* in early August 2024. Limited surveys were completed at WFGB due to strong currents, and time constraints related to vessel operations and safety requirements.

Random Transect Data Processing

Prior to the adoption of Coral Point Count with Microsoft® Excel® extensions (CPCe), percent cover was originally calculated using mylar tracings and a calibrated planimeter (1989–1995; Gittings et al., 1992; Continental Shelf Associates, 1996) and later by manually generating random-dot overlays in photo-processing software (1996–2003; Dokken et al., 1999, 2003). For this analysis, all historical benthic imagery has been reprocessed using CPCe to ensure methodological consistency across the full time series, resulting in updated percent cover values that may differ from those reported in earlier publications.

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) hydrocoral; 3) sponges (including encrusting sponges); 4) CCA; 5) macroalgae (algae longer than approximately 3 mm and thick algal turfs covering underlying substrate); 6) colonizable substrate (including fine turf algae, rubble, and bare rock; Aronson & Precht, 2000; Aronson et al., 2005); 7) sand; and 8) an “other” category (biotic components such as sea urchins, ascidians, fish, serpulid worms, and unknown species). Additional features (i.e., photostation tags, tape measures, scientific equipment) and points with no data (shadows) were excluded from the analysis. Points on corals that could not be identified were labeled as “unidentified coral.” Point count analysis was applied on photos within a transect and mean percent cover for all groups was determined by averaging all transects in the study site or reef-wide surveys. Results are presented as mean percent cover \pm standard error (SE).

Incidents of coral bleaching, paling, concentrated and isolated fish biting, mortality, and disease 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 considered “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).

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 composition in EFGB and WFGB random transects were evaluated with non-parametric distance-based analyses using Primer[®] version 7.0 (Anderson et al., 2008; Clarke et al., 2014). Percent cover data were square root-transformed to reduce the influence of dominant groups. 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 benthic group to the observed dissimilarity (Clarke et al., 2014). Coral community was compared between study sites 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 surveys 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 2024), yearly mean benthic cover was visualized for each study site using principal coordinates ordination (PCO), based on Euclidean distance similarity matrices, with percent variability explained on each canonical axis. Yearly means were used to evaluate interannual trends in benthic community composition, as the monitoring design is intended to detect year-to-year change rather than within-year spatial variability. A time series trajectory with Pearson 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 contributed to observed dissimilarities (Clarke et al., 2014).

Mean percent benthic cover from random transect surveys was analyzed from 1989 to 2024. 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).

Temporal patterns in coral beta diversity from 1992–2024 were evaluated using permutational analysis of multivariate dispersions (PERMDISP) in PRIMER® version 7. Analyses were conducted separately for EFGB and WFGB using presence/absence data derived from random transect surveys and a Jaccard similarity resemblance matrix. PERMDISP was used to calculate annual multivariate dispersion values, expressed as the mean distance of transects to the centroid for each year, which quantify beta diversity as variation in coral species composition among transects. Rather than testing categorical differences among years, dispersion values were examined as a continuous time series to assess long-term temporal patterns in beta diversity.

The range of data collected has varied slightly over time, and analyses were structured accordingly. From 1989 to 1991, only mean percent coral cover data were collected. Additional major benthic groups were incorporated in 1992, and no data were collected in 1993. As a result, trend and correlation analyses involving non-coral benthic groups were limited to years with consistent data availability. In 2023, data from 1989–2002 and 2009–2010 were reanalyzed using current methods in CPCe and Primer®.

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. In 2024, all 37 photostations were located and photographed within the EFGB study site and all 41 were located and photographed within the WFGB study site. However, permanent repetitive photostations ranging from 24–39 m in depth, which are located beyond the study site boundaries, were not photographed in 2024 due to logistical constraints.

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 m².

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 through 2008, photographs at each repetitive photostation encompassed an approximately 8 m² area. Following the transition to digital imagery, the photographed area was adjusted in 2009 (5 m²) to accommodate changes in camera equipment and image quality requirements, and has remained standardized at 5 m² from 2011 onward.

The total number of repetitive photostations has varied over time as new stations were established and others were lost or could not be relocated due to missing tags or overgrown posts; however, approximately 40 photostations have been maintained within each study site since 1989. Deep photostations (24–39 m) were established in phases. At the EFGB, nine of the 23 deep photostations were installed in 2003, with two additional stations (30 m and 31 m) added in 2013. At the WFGB, 12 of the 24 deep photostations were installed in 2012. The remaining deep photostations at both banks were added in 2017.

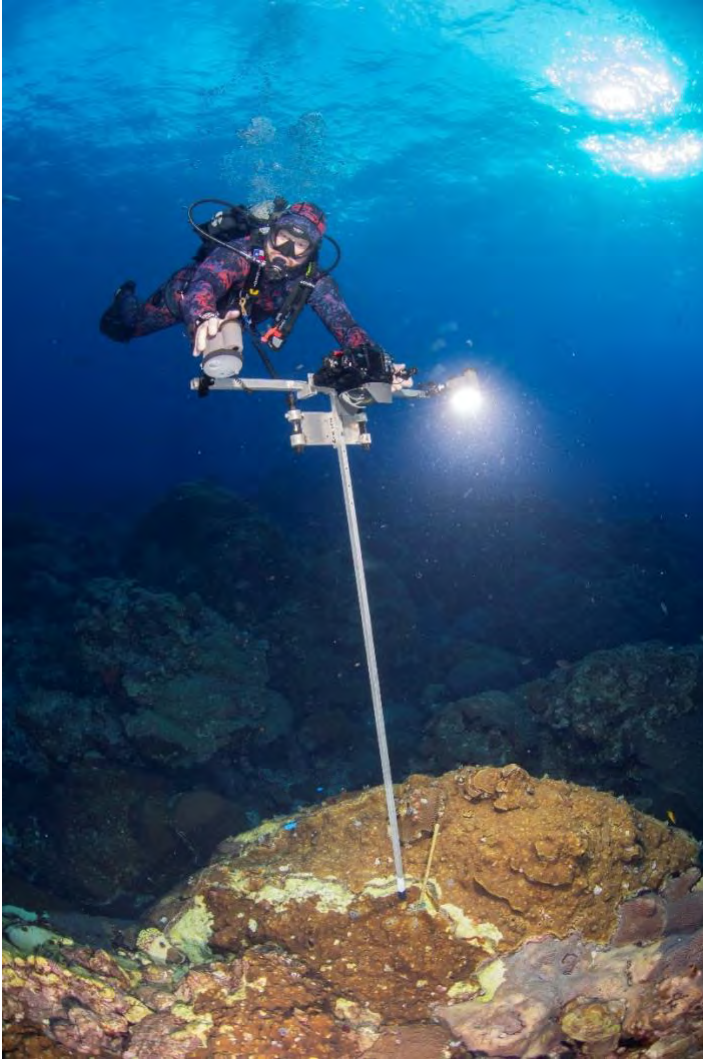


Figure 2.2. NOAA diver prepares to photograph a repetitive photostation with camera and strobes mounted to an aluminum t-frame. Photo: Donavon French/CPC

Repetitive Photostation Data Processing

Percent benthic cover from repetitive photostation images was calculated 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 \pm standard error (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 and instead are used for fate tracking and interpretation of mechanisms of change.

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. 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.

Benthic Community Results

Random Transect Mean Percent Cover

Coral, followed by macroalgae, had the highest mean percent benthic cover at EFGB and WFGB in study site surveys, and a similar pattern was observed in reef-wide surveys at EFGB in 2024 (Figure 2.3). Reef-wide survey data at WFGB in 2024 were insufficient to support statistical analysis or comparison in methods and are therefore not presented.

PERMANOVA revealed significant differences in benthic community composition between EFGB and WFGB study sites in 2024 (Table 2.1). SIMPER analysis indicated that this difference was primarily driven by higher macroalgae cover and lower coral cover at EFGB.

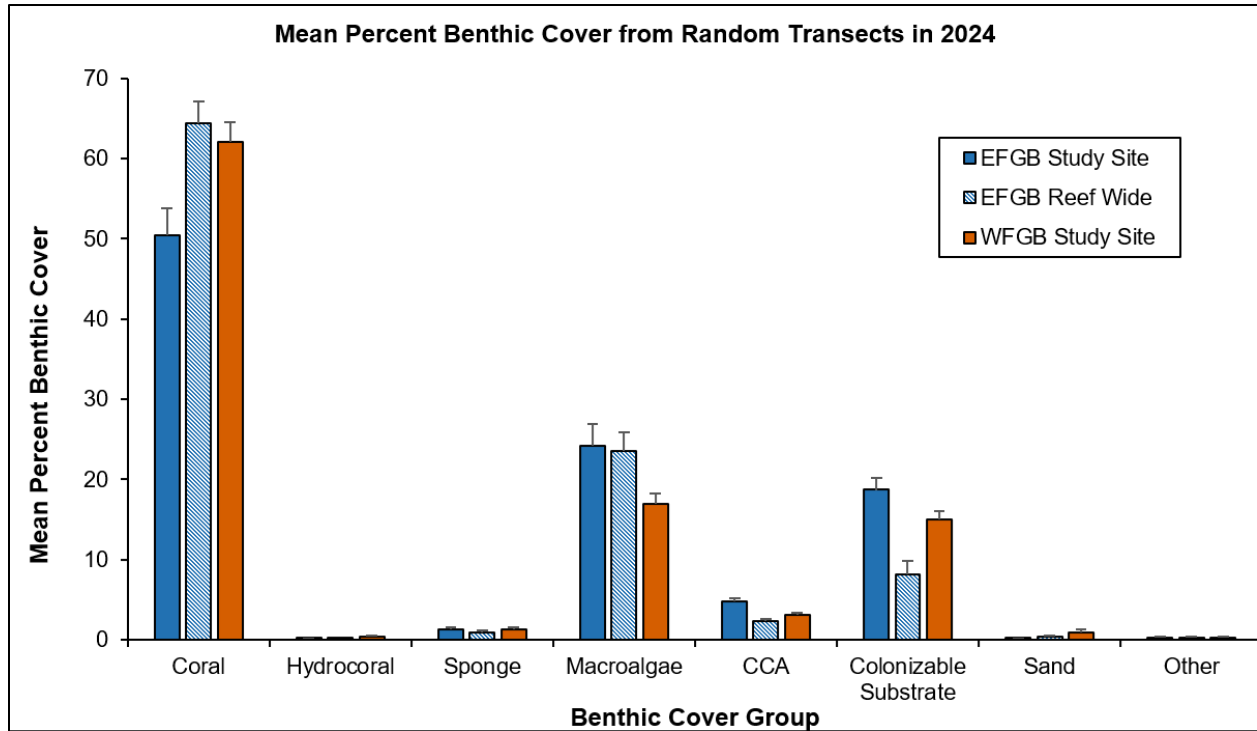


Figure 2.3. Mean percent benthic cover +SE from random transect surveys within study sites and reef-wide at EFGB and WFGB in 2024.

Table 2.1. PERMANOVA results comparing 2024 mean percent benthic cover between banks. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	14.01	1	4.54	0.01
Res	92.60	30		
Total	106.62	31		

PERMANOVA was conducted to assess differences in benthic community composition between the reef-wide and study site methods for EFGB. The analysis revealed significant differences between the two sampling methods, indicating that each method captures distinct benthic community compositions (Table 2.2). SIMPER analysis identified colonizable substrate and coral as the primary contributors to these differences. Specifically, colonizable substrate was found to be more abundant in the study site method, while coral was more prevalent in the reef-wide method.

Table 2.2. PERMANOVA results comparing EFGB 2024 mean percent benthic cover between methods. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Method (reef-wide vs. study site)	30.23	1	9.07	0.0002
Res	93.31	28		
Total	123.54	29		

Fourteen coral species were observed at each bank, although species composition differed slightly between EFGB and WFGB. *Madracis auretenra* was observed at EFGB but not at WFGB, whereas *Scolymia cubensis* was recorded at WFGB but not at EFGB. (Figure 2.4). *Orbicella franksi* was the most abundant coral species observed within EFGB ($21.20 \pm 2.82\%$) and WFGB ($31.80 \pm 3.68\%$) study site surveys. *Porites astreoides* ranked second within the EFGB study site ($7.65 \pm 0.74\%$) while *Pseudodiploria strigosa* ranked second within the WFGB study site ($8.04 \pm 1.74\%$; Figure 2.4). The *Orbicella* spp. complex, consisting of *O. franksi*, *O. faveolata*, and *O. annularis*, are listed as threatened species under the Endangered Species Act (16 U.S.C. § 1531 *et seq.*).

PERMANOVA analysis revealed no significant difference in coral species composition between banks for all surveys (Table 2.3).

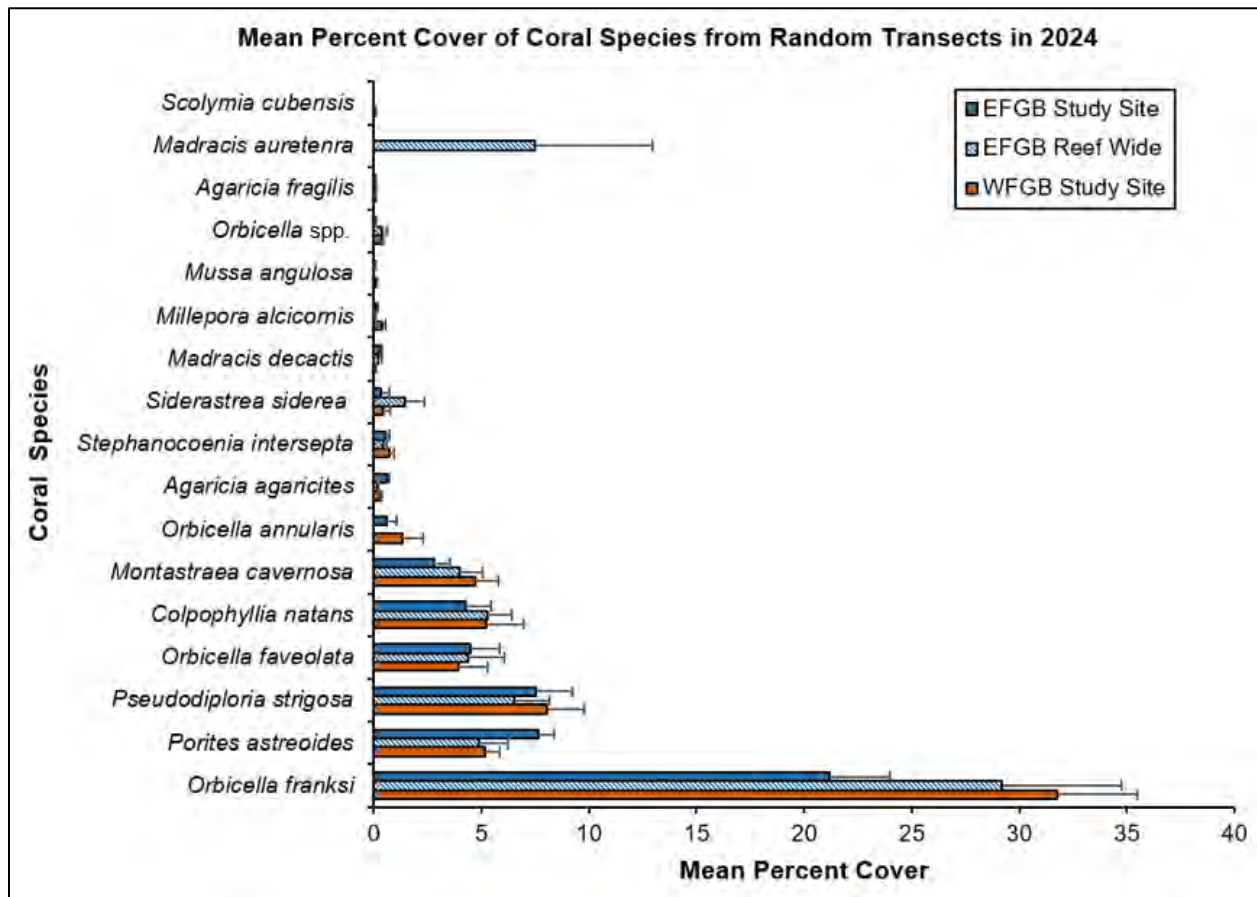


Figure 2.4. Mean percent cover + SE of observed coral species from random transect surveys within EFGB and WFGB study sites and reef-wide surveys in 2024.

Table 2.3. PERMANOVA results comparing coral species mean percent cover from EFGB and WFGB study site random transects in 2024.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	16.07	1	1.6487	0.1184
Res	292.41	30		
Total	308.48	31		

Coral species diversity metrics were calculated for each random transect within study site surveys at EFGB and WFGB in 2024 and then summarized by bank as mean \pm SE (Table 2.4). ANOSIM analysis conducted on transect-level diversity metrics detected no significant differences in coral community diversity between banks. Reef-wide diversity metrics were not analyzed due to insufficient sample size at WFGB.

Table 2.4. Mean (\pm SE) coral species diversity measures calculated from random transect surveys within EFGB and WFGB study sites in 2024.

Random Transect Coral Diversity Measures	EFGB	WFGB
Margalef's species richness (d)	2.66 \pm 0.08	2.43 \pm 0.12
Pielou's evenness (J')	0.90 \pm 0.01	0.87 \pm 0.01
Shannon diversity (H'(loge))	1.91 \pm 0.04	1.80 \pm 0.06

At EFGB, an average of $1.42 \pm 0.61\%$ of coral cover was affected by bleaching, $12.9 \pm 2.91\%$ showed signs of paling, and less than 1% was impacted by fish biting. At WFGB an average of $0.11 \pm 0.06\%$ of coral cover was affected by bleaching, $8.81 \pm 1.96\%$ showed signs of paling, and less than 1% was impacted by fish biting. Bleaching increased later in the season as detailed in 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 0.66% of coral cover in all study site random transect surveys.

Random Transect Long-Term Trends

Mean percent coral cover in study sites (17–27 m) from 1989 to 2024 ranged from 45–67% at EFGB and 44–66% at WFGB. Coral cover increased significantly over time at the WFGB study site ($\tau = 0.35$, $p < 0.007$; Figure 2.5), and remained stable at the EFGB study site. Data on sponge, CCA, macroalgae, colonizable substrate, and sand were first collected in 1992.

From 1992 to 2024, macroalgae cover ranged between 10–38% and increased significantly in both the EFGB and WFGB study sites ($\tau = 0.43$, $p < 0.001$; $\tau = 0.34$, $p < 0.009$, respectively; Figure 2.5). Over the same period, colonizable substrate declined significantly at both EFGB ($\tau = -0.36$, $p < 0.006$) and WFGB ($\tau = -0.60$, $p < 0.001$). Hydrocoral cover declined significantly at both EFGB and WFGB from 1992 to 2024 ($\tau = -0.65$, $p < 0.001$; $\tau = -0.38$, $p < 0.004$, respectively; Figure 2.5). Sponge cover declined significantly at EFGB ($\tau = -0.26$, $p = 0.046$)

Macroalgae and colonizable substrate were strongly negatively correlated at EFGB ($r = -0.82$, $p < 0.001$) and WFGB ($r = -0.73$, $p < 0.001$), suggesting that macroalgae growth was preferentially occupying previously available space and not outcompeting other biotic cover.

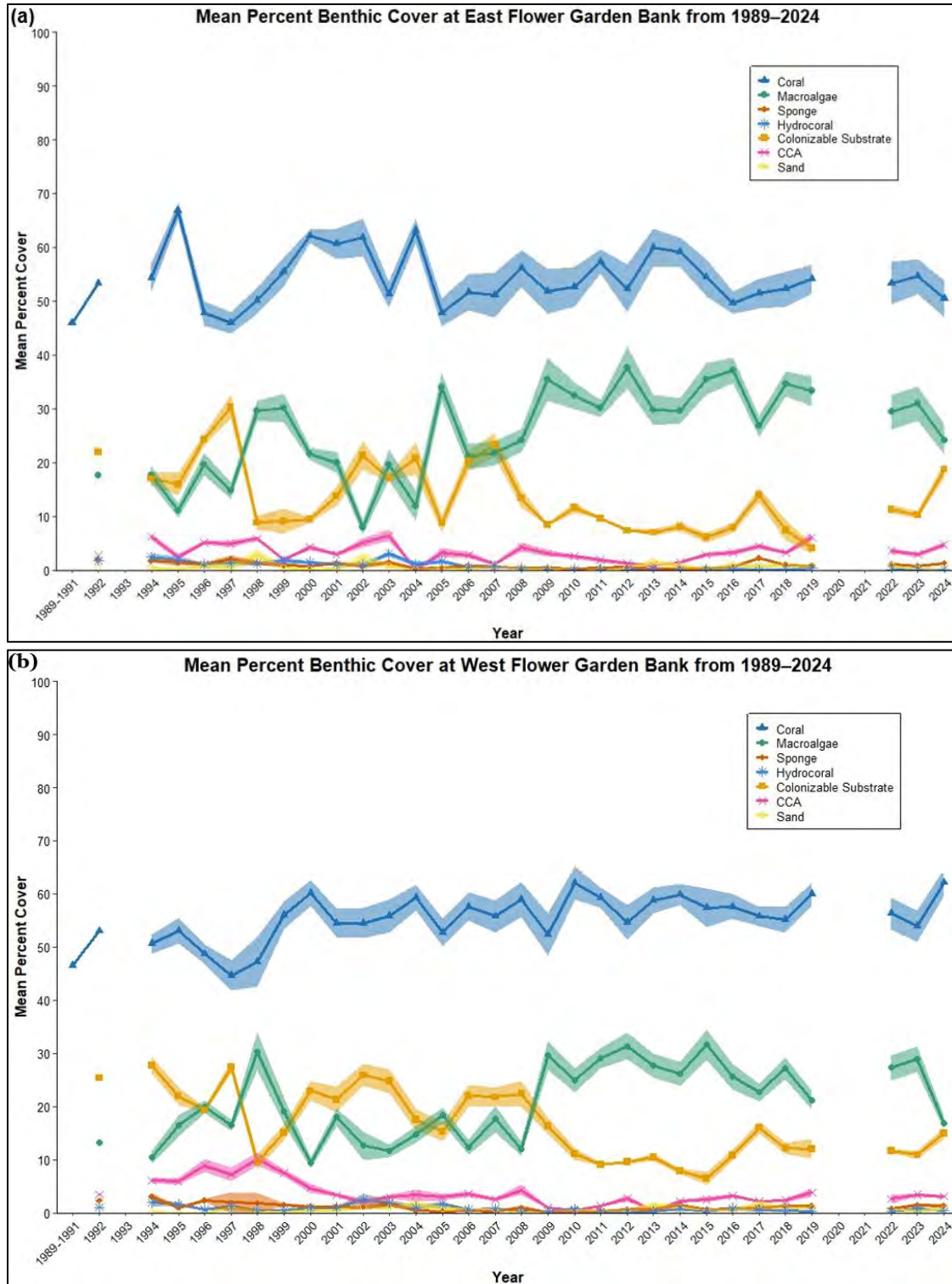


Figure 2.5. Mean percent benthic cover \pm SE bands from random transect surveys within (a) EFGB and (b) WFGB study sites from 1989 to 2024. Only coral cover data were reported from 1989–1991 and no mean percent cover data were reported in 1993, 2020, or 2021. Sources: Gittings et al., 1992 (1989 to 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); Eisenbach et al., 2025 (2023).

For available yearly mean benthic percent cover data (1992 to 2024), SIMPROF analysis detected three significant year clusters in the EFGB study site (2000–2001, 2008, 2017, 2022, and 2024; 1998–1999, 2005, 2009–2016, 2018–2019, 2023; and 1992, 1994–1997, 2002–2004, 2006–2007; Figure 2.6). Macroalgae mean percent cover contributed to the majority of the dissimilarity among all year group clusters.

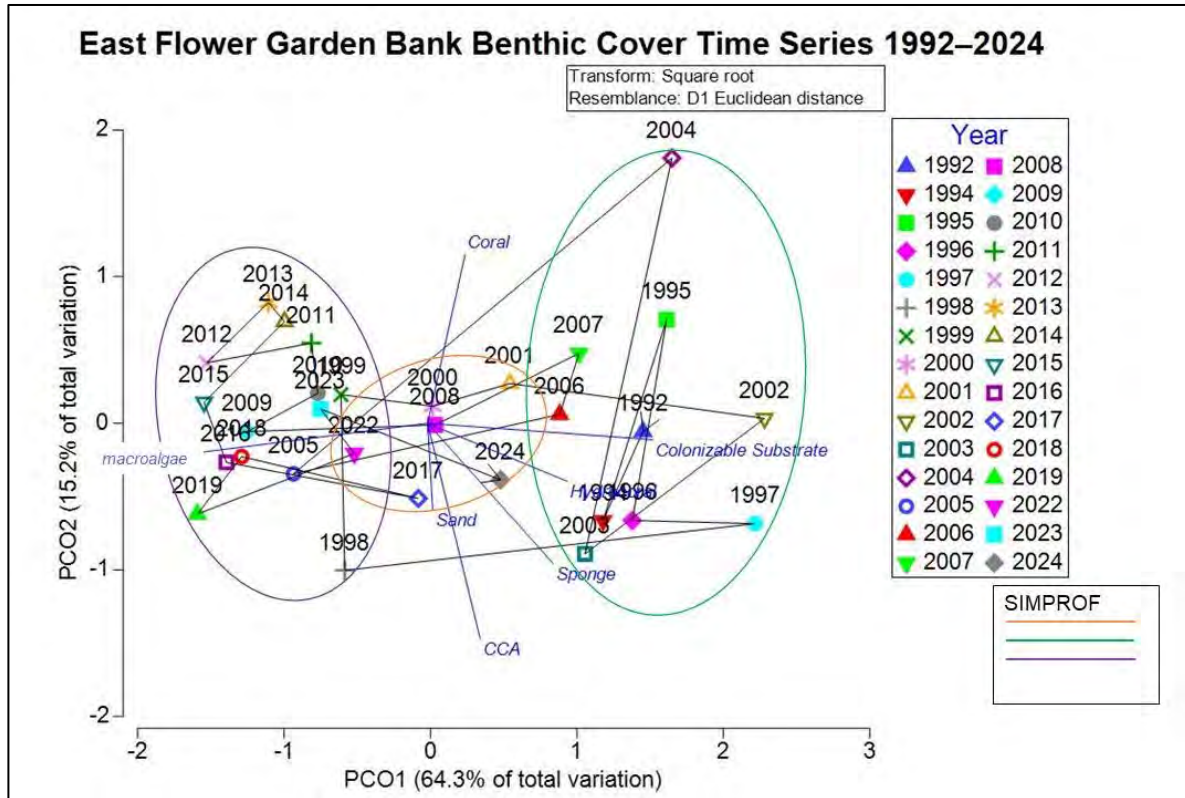


Figure 2.6. PCO plot for random transect benthic cover analysis within the EFGB study site from 1992–2024. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

Yearly mean benthic percent cover from 1992 to 2024 at the WFGB study site resulting in four significant year clusters (1992, 2000, 2002–2003, 2006, and 2008; 2001, 2004–2005, 2007, 2017, 2019, and 2024; 1994–1997, and 1999; and 1998, 2009–2016, 2018, 2022, and 2023; Figure 2.7). SIMPER analysis revealed that differences between the groups were primarily driven by macroalgae.

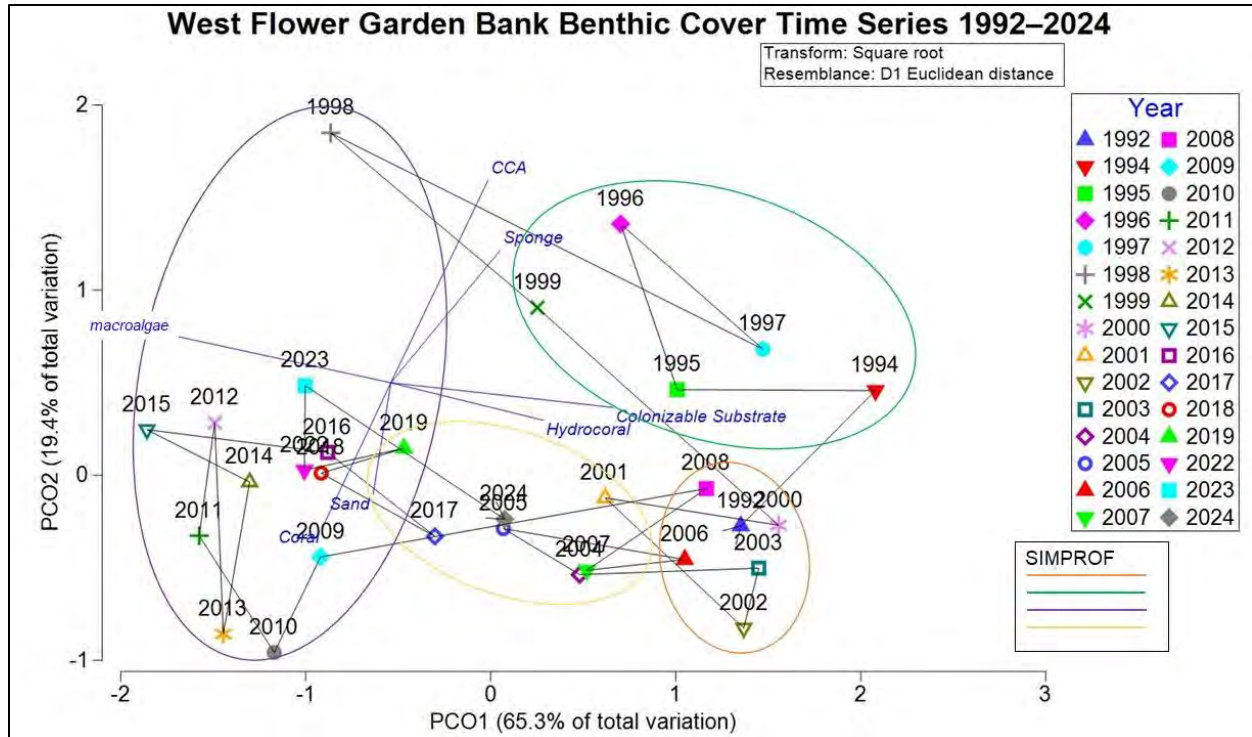


Figure 2.7. PCO plot for random transect benthic cover analysis within the WFGB study site from 1992 to 2024. The ovals are SIMPROF groups representing significant year clusters grouped by color. The blue vector lines represent the directions of the variable gradients for the plot.

PERMANOVA results revealed statistically significant differences in benthic community composition between EFGB and WFGB from 1992 to 2024 (Table 2.5; Pseudo-F = 11.212, $p = 0.0001$). Significant variation was also observed across years (Pseudo-F = 14.611, $p = 0.0001$), and a significant interaction between Bank and Year (Pseudo-F = 2.596, $p = 0.0001$) indicates that temporal patterns in benthic community composition differed between the two banks. SIMPER analysis indicated that overall differences between EFGB and WFGB were relatively modest (average dissimilarity = 16.45%) and were primarily driven by macroalgae (21.81%), colonizable substrate (19.56%), and coral cover (15.29%), with macroalgal cover generally higher at EFGB and colonizable substrate more abundant at WFGB.

Table 2.5. PERMANOVA results comparing random transect benthic percent cover from EFGB and WFGB random transects from 1992–2024.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	90.44	1	11.21	0.0001
Year	1443.80	29	14.61	0.0001
Bank x Year	256.47	29	2.60	0.0001
Res	2920.10	857		
Total	4739	916		

To further interpret this interaction, separate PERMANOVA analyses were conducted for each bank with Year as the sole factor. Benthic community composition differed significantly among

years at both EFGB (Pseudo-F = 8.32, $p = 0.0001$; Table 2.6) and WFGB (Pseudo-F = 8.92, $p = 0.0001$; Table 2.7), indicating pronounced interannual change in benthic cover at both sites.

Table 2.6. PERMANOVA results comparing random transect benthic percent cover at EFGB random transects from 1992–2024.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	873.01	29	8.32	0.0001
Res	1591.70	440		
Total	2464.71	469		

Table 2.7. PERMANOVA results comparing random transect benthic percent cover at WFGB random transects from 1992–2024.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	824.15	29	8.92	0.0001
Res	1328.50	417		
Total	2152.6	446		

Coral cover remained consistently high at both banks across the time series, while macroalgae and colonizable substrate exhibited greater interannual variability. These trends align with the results of PERMANOVA and SIMPER, and illustrate gradual changes in benthic community composition over time (Figure 2.8).



Figure 2.8. Heatmaps of mean percent benthic cover across all random transects at EFGB (top) and WFGB (bottom) from 1992 to 2024. Warmer colors indicate higher average percent cover.

Beta diversity derived from random transect surveys, quantified as multivariate dispersion (PERMDISP) based on coral species presence/absence data, exhibited a significant temporal pattern at EFGB from 1992 to 2024 ($F = 4.29$, $p = 0.0001$). Mean distances to centroid were generally higher during the 1990s and early 2000s, indicating greater variability in coral species composition among transects during that period. Dispersion values became lower and more constrained in more recent years, with consistently reduced values observed from approximately 2018 through 2024, suggesting increased similarity in coral composition among transects through time. In contrast, dispersion values at WFGB did not exhibit a significant temporal pattern over the monitoring period ($F = 1.00$, $p = 0.66$) and remained relatively consistent through time, indicating stable variability in coral species composition among transects.

Repetitive Photostation Mean Percent Cover

In 2024, coral and macroalgae were the dominant benthic cover categories in EFGB repetitive photostations, while coral and colonizable substrate dominated in WFGB repetitive photostations (Figure 2.9; Table 2.8). Analysis of repetitive photostations in July 2024 revealed early signs of thermal stress at EFGB, with both bleaching and paling observed, though paling was more prominent than bleaching ($1.89 \pm 0.78\%$ and $9.13 \pm 2.18\%$ of coral cover,

respectively). At WFGB, bleaching was minimal at $0.15 \pm 0.12\%$ and paling was also low at $2.41 \pm 0.67\%$ (Figure 3.1). As is typical for this reef, more extensive bleaching was documented later in the year (see Chapter 3).

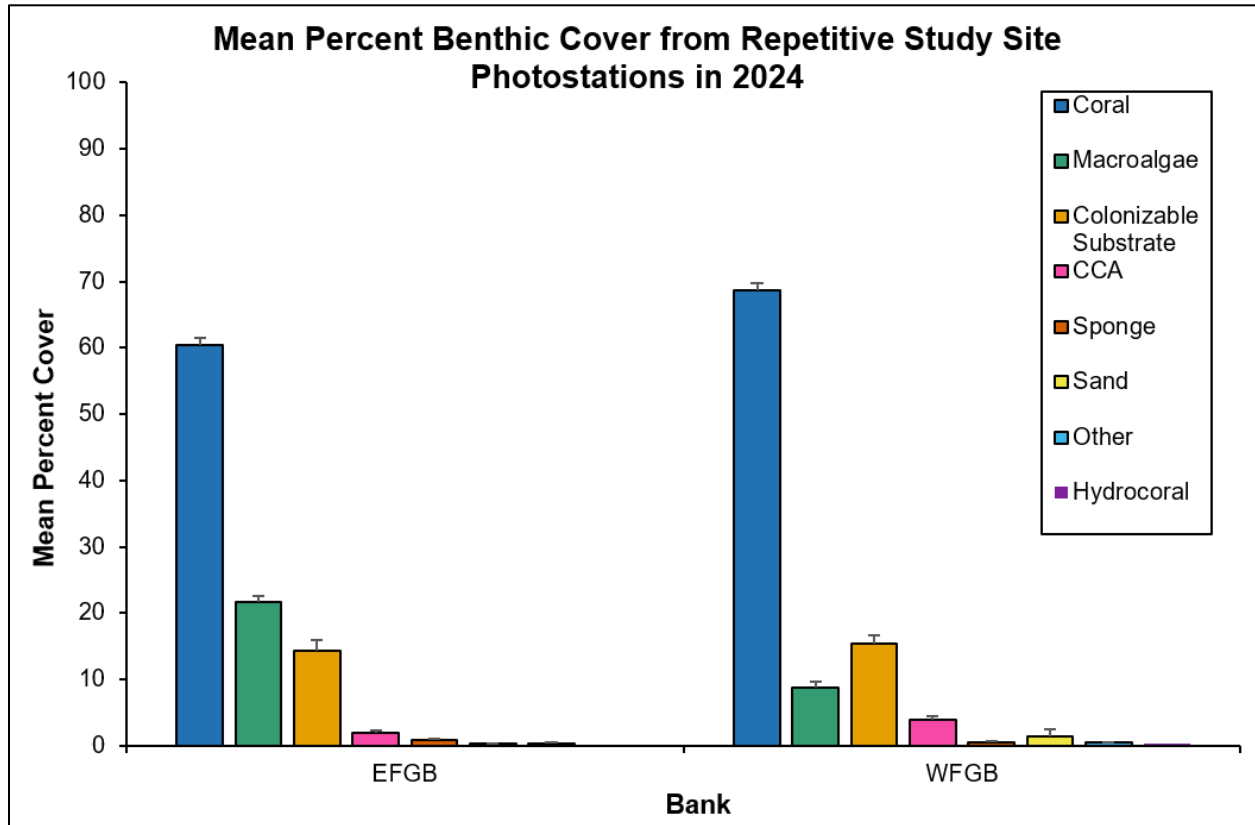


Figure 2.9. Mean percent benthic cover + SE within EFGB and WFGB repetitive photostations in 2024.

Table 2.8. Range of mean percent cover categories from EFGB and WFGB repetitive photostations, and all photostations combined in 2024.

Percent Cover Range	EFGB	WFGB	EFGB and WFGB combined
Coral	24.00–83.84%	31.91–85.71%	24.00–85.71%
Macroalgae	4.30–60.00%	0.00–24.73%	0.00–60.00%
CCA	0.00–6.19%	0.00–10.64%	0.00–10.64%
Colonizable substrate	0.00–43.16%	5.10–32.98%	0.00–43.16%
Sponge	0.00–9.47%	0.00–3.23%	0.00–9.47%
Sand	0.00–3.06%	0.00–37.76%	0.00–37.76%
Other	0.00–2.06%	0.00–2.27%	0.00–2.27%
Hydrocoral	0.00–4.26%	0.00–9.57%	0.00–9.57%

The repetitive photostations ranged in depth from 18–25 m at EFGB (averaging 21 m depth) and 20–24 m at WFGB (averaging 22 m depth). In 2024, mean percent cover of benthic categories showed substantial spatial variability among individual photostations at both banks (Table 2.8).

Fourteen coral species were recorded in EFGB repetitive photostations and 15 were recorded at WFGB. *Orbicella franksi* was the dominant species at EFGB ($30.86 \pm 2.94\%$), followed by *P.*

strigosa ($10.71 \pm 1.79\%$) and *Orbicella faveolata* ($6.33 \pm 1.79\%$; Figure 2.10). *Orbicella franksi* was the dominant coral at WFGB ($35.13 \pm 2.61\%$), followed by *P. strigosa* ($9.59 \pm 1.62\%$) and *Porites astreoides* ($6.17 \pm 0.81\%$; Figure 2.10).

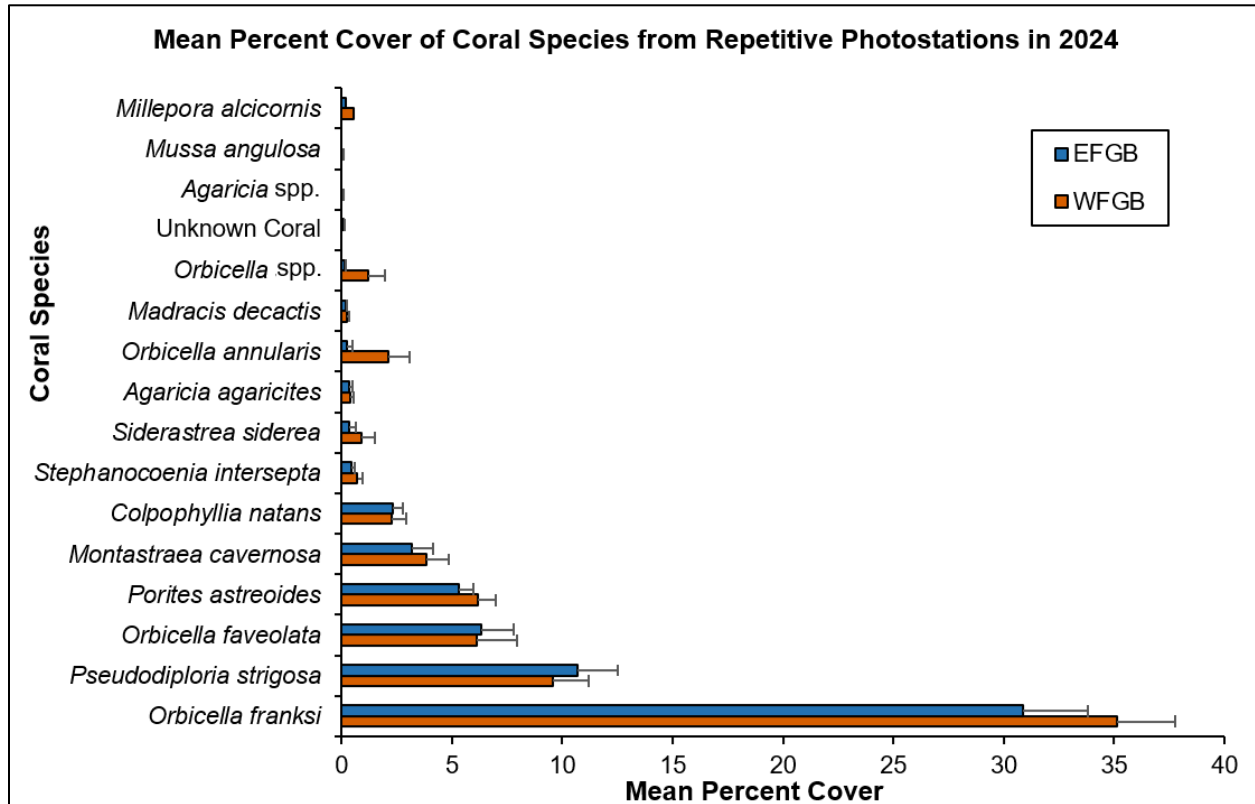


Figure 2.10. Mean percent coral cover + SE within EFGB and WFGB repetitive photostations in 2024.

In 2024, repetitive photostation surveys were limited to shallow reef depths (≤ 27 m) where coral communities are relatively similar, as deeper photostations (extending to 39 m) were not surveyed due to time constraints. Consequently, coral cover was evaluated across a narrow depth range, and no significant relationship between coral species mean percent cover and depth was detected at either EFGB (PERMANOVA, Pseudo-F = 1.05, $p = 0.387$) or WFGB (Pseudo-F = 0.96, $p = 0.581$). This lack of a depth-related pattern reflects limited depth coverage rather than the absence of depth-related differences in coral cover.

Qualitative Analysis of Repetitive Photostations and Coral Disease

In 2024, disease-like lesions were observed across multiple coral species at EFGB and during annual repetitive photostation surveys. These photostations remained critical for documenting site-specific changes in coral health and fate tracking of partial mortality, tissue loss, and recovery. Colonies exhibiting signs of disease, including marginal lesions and multifocal tissue loss, were recorded, and the total number of colonies per station was used to calculate prevalence. Affected species included *Pseudodiploria strigosa*, *Colpophyllia natans*, *Porites astreoides*, *Mussa angulosa*, and *Orbicella* spp.

Disease prevalence increased in 2024 compared to previous years, reaching 4.7% ($\pm 0.8\%$ SE) at WFGB and 3.2% ($\pm 0.6\%$ SE) at EFGB. This reflects a rise from the 2.1–2.6% prevalence documented in 2022, when *Pseudodiploria strigosa* (7–8%), *Colpophyllia natans* (11–18%), and *Orbicella* spp. (1%) were the most affected (Johnston et al., 2023). In 2023, disease observations were minimal and many colonies showed signs of recovery. However, in 2024, signs of active tissue loss and partial mortality re-emerged across multiple species, and photostation comparisons indicated increased coral degradation relative to 2023 (Figure 2.12).

Notably, several *Porites astreoides* colonies experienced near-total mortality attributed to disease, and bleaching was observed in *Montastraea cavernosa* colonies across photostations. While some species, such as *Pseudodiploria strigosa*, exhibited regrowth, station-level assessments still reflected broader declines in coral health. Importantly, these declines were not captured in percent coral cover alone, emphasizing the need for focused assessments.

To capture these nuanced changes, each photostation was assigned a qualitative status using a five-color grading system:

- Coral and/or Sponge Growth (Green): >50% of dominant corals and/or sponges showed evidence of healthy growth, or stations with prior disease or degradation showed recovery and regrowth into previously lost areas.
- Coral and Sponge Stable (Blue): >50% of dominant corals and sponges showed little to no growth and no increase in degradation, indicating overall stability.
- Coral Degradation (Yellow): Sites where ~25% or more of dominant corals showed degradation. Disease took priority over damselfish gardening in this classification. Stations were labeled degraded if $\geq 25\%$ of corals showed disease-related tissue loss, or if damselfish gardening affected ≥ 1 m² or expanded compared to previous surveys.
- Sponge Degradation (Orange): Sites where approximately $\geq 33\%$ of dominant sponges showed signs of degradation, primarily due to algal overgrowth or tissue wasting.
- Coral and Sponge Degradation (Pink): Sites where a majority (>50%) of dominant corals and sponges showed evidence of degradation and overall cover declined.

From 2023 to 2024, EFGB stations were most commonly graded yellow (coral degradation; 37.8%) or blue (unchanged; 35.1%). WFGB stations were predominantly blue (59.5%), with 35.7% showing coral degradation (yellow). Two EFGB stations (5.4%) were assigned the most severe grade (pink), indicating concurrent declines in both coral and sponge cover (Figure 2.11; Figure 2.12). Notably, no stations were assigned to the sponge-only degradation category (orange) in 2023–2024, in contrast to its prevalence in 2022–2023 at WFGB. Instead, sponge declines in 2024 were only observed in conjunction with coral declines, reflected by the emergence of the pink category (both coral and sponge degraded), which was absent in the 2022–2023 comparison. This shift may suggest that sponge degradation was less widespread in 2024 or more tightly linked to coral disease dynamics at specific sites. These qualitative assessments, supported by disease prevalence data, point to a decline in reef condition in 2024.

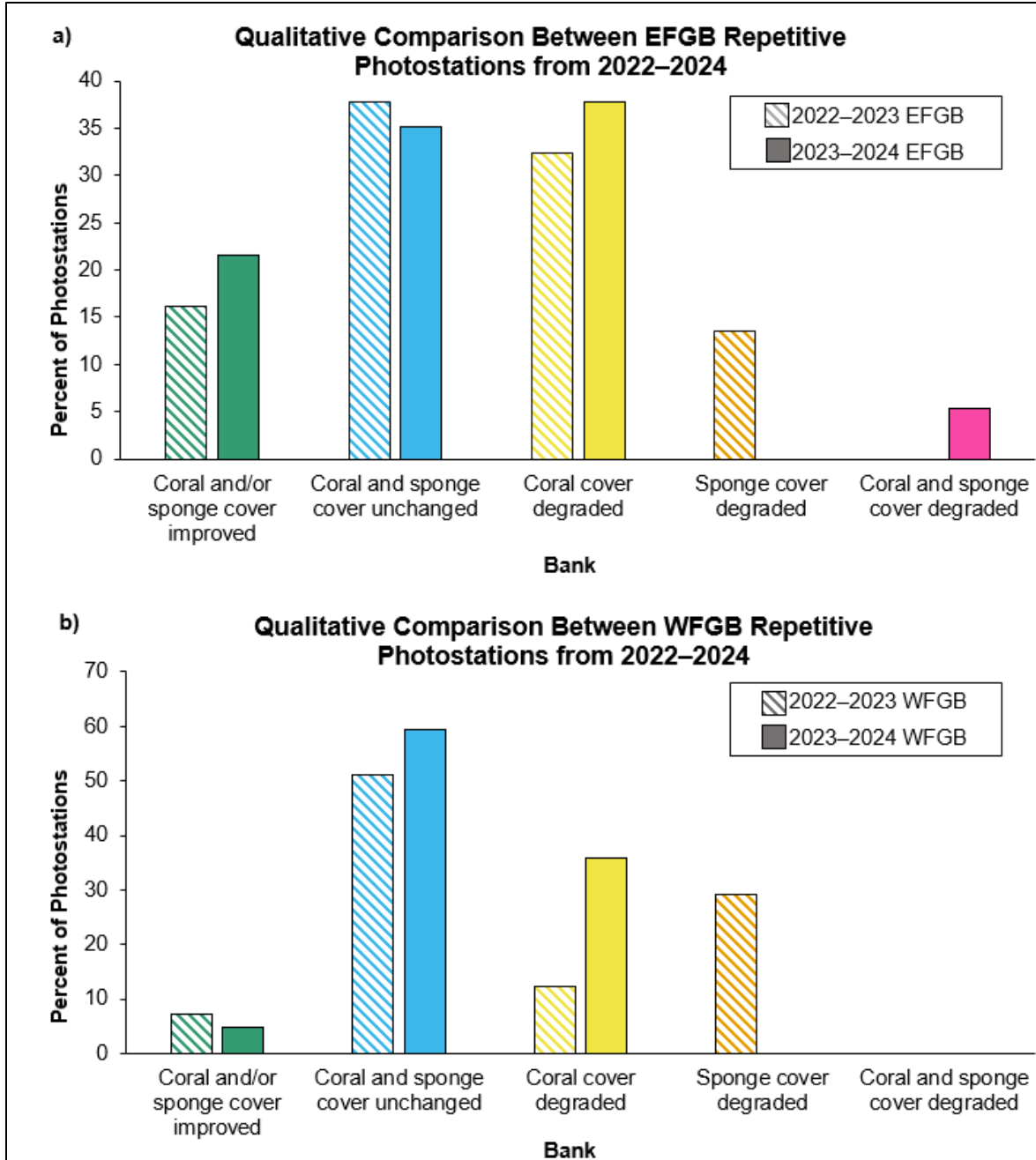


Figure 2.11. Percent of photostations by condition category, illustrating changes in coral and sponge cover at EFGB (a) and WFGB (b) from 2022–2024 based on qualitative assessments.

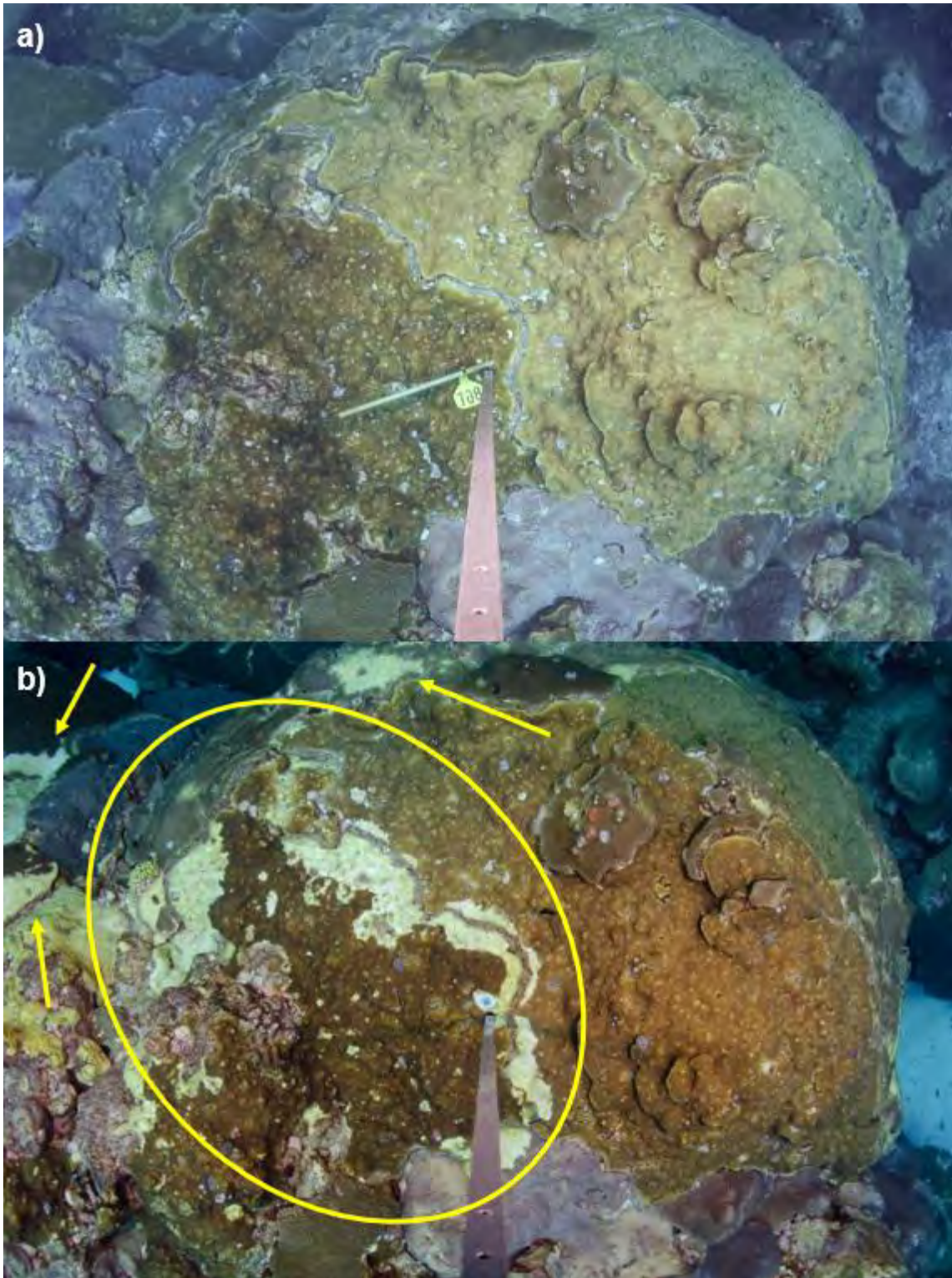


Figure 2.12. WFGB repetitive photostation 708 in (a) 2023 and (b) 2024. In 2023, the station was free of disease. By 2024, active lesions (circled in yellow) were present and widespread across multiple colonies and species. Photo: Donavon French/CPC

Repetitive Photostation Long-Term Trends

Twenty-four EFGB photostations and 27 WFGB photostations (ranging in depth from 20–24 m) have been monitored since the beginning of the program, spanning 1989 to 2024. Mean percent coral cover at EFGB photostations increased from $58.00 \pm 3.67\%$ in 1989 to $61.76 \pm 2.87\%$ in 2024. At WFGB photostations, mean percent coral cover increased from $49.95 \pm 2.99\%$ in 1989 to $68.65 \pm 2.50\%$ in 2024 (Figure 2.13).

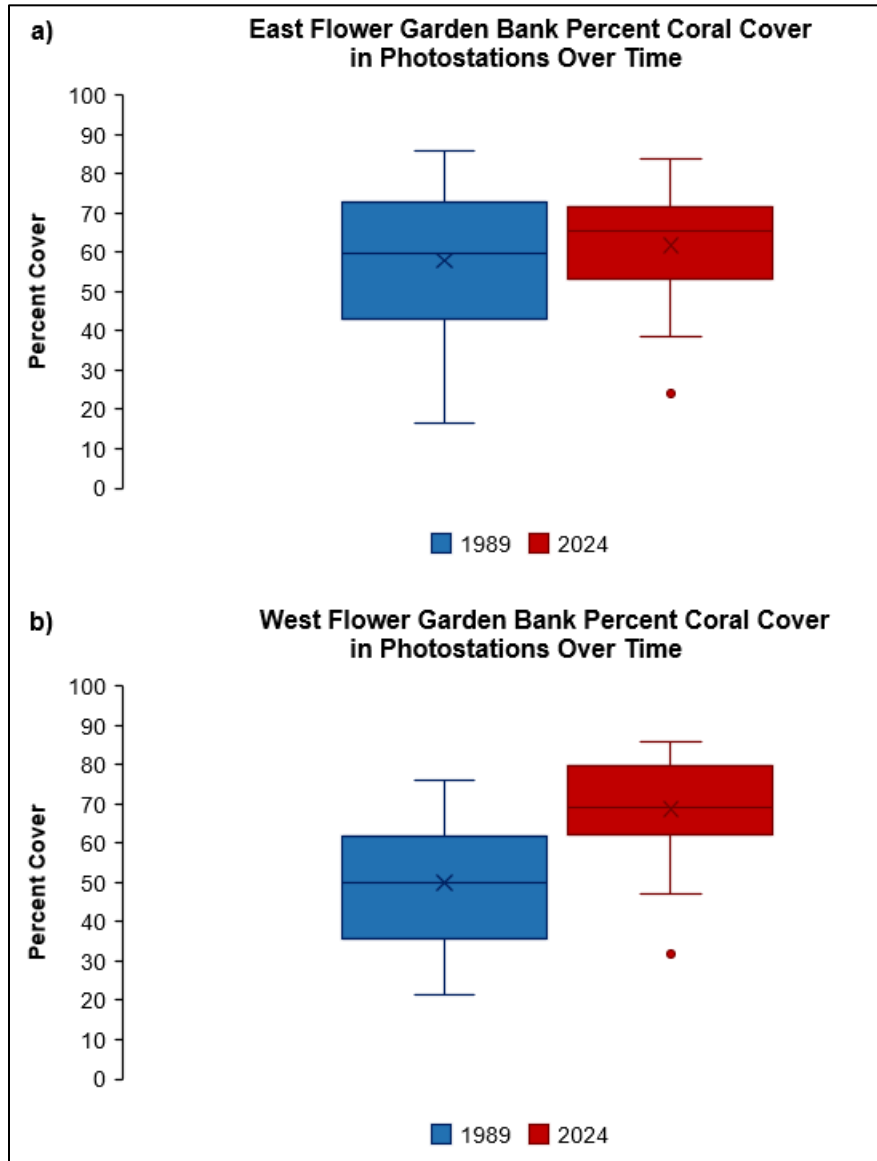


Figure 2.13. Box plot depicting percent coral cover in (a) EFGB ($n = 24$) and (b) WFGB ($n = 27$) repetitive photostations in 1989 and 2024.

Figure 2.14 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 2024 include recruitment and

growth of *P. strigosa* and *P. astreoides* in the center (Figure 2.14a; Figure 2.14b). 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 2024, and a black *Ircinia strobilina* sponge that was present in 1989 was absent in 2024 (Figure 2.14c; Figure 2.14d). In WFGB photostation #503, a large *C. natans* colony present in 1989 was absent in 2024 (Figure 2.14e; Figure 2.14f).

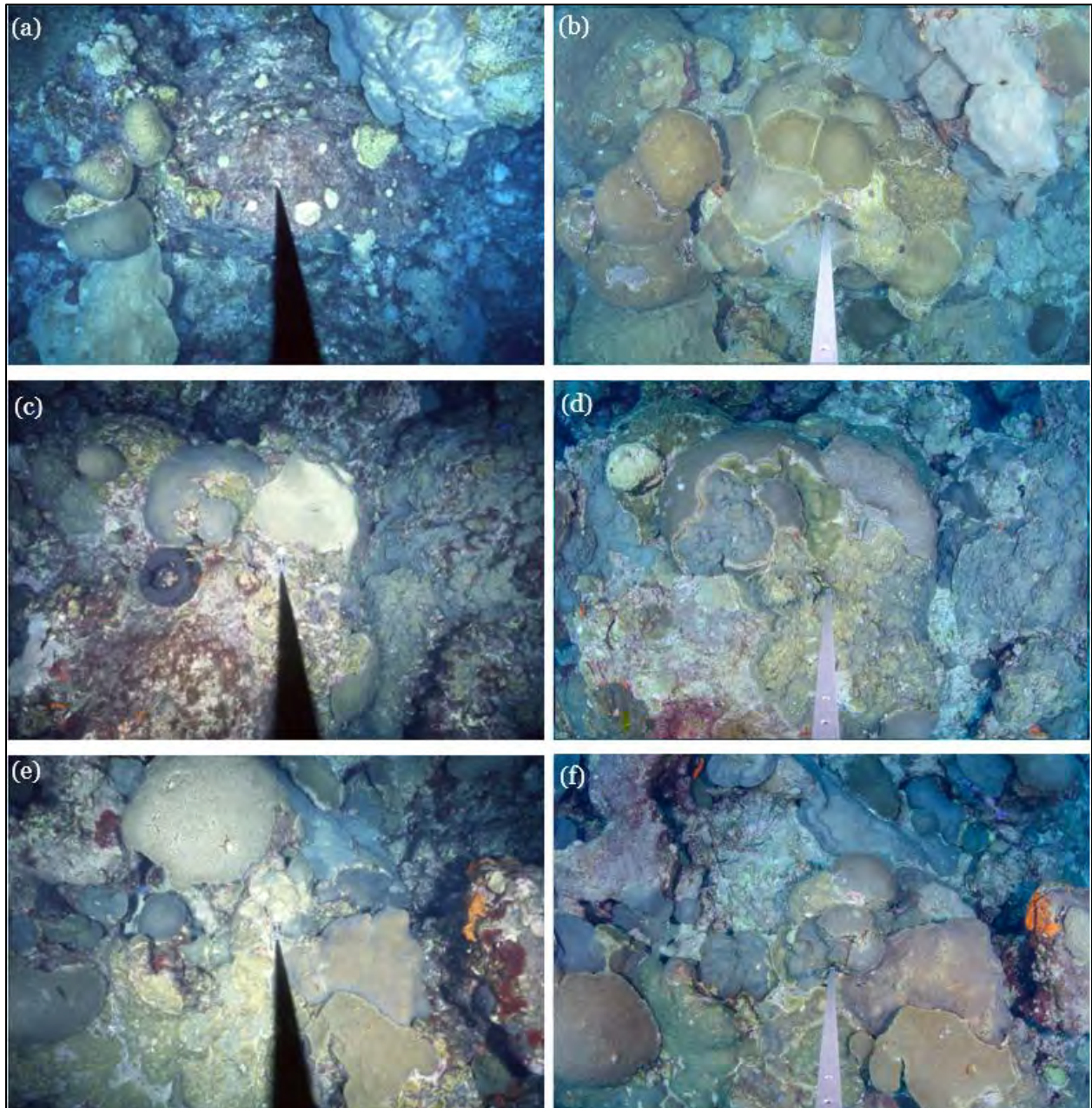


Figure 2.14. Time series of three repetitive photostations: EFGB photostation #102 (20 m) from (a) 1989 to (b) 2024; WFGB photostation #501 (20 m) from (c) 1989 to (d) 2024; and WFGB photostation #503 (20 m) from (e) 1989 to (f) 2024. Photos: (a, c, e) Minerals Management Service (Gittings et al., 1992); (b, d, f) NOAA

Discussion

Despite widespread regional coral cover declines (Wicquart et al., 2025), coral cover within EFGB and WFGB study sites remained high in 2024. Mean coral cover from random transects within study sites remained above 50% at both banks, and cover within repetitive photostations remained near or above 60%, continuing a pattern observed throughout the 35-year monitoring record. This pattern of persistently high coral cover and dominance by similar reef-building species is consistent with observations from reefs in the southern Gulf, where regional assessments have documented relatively high coral cover and dominance by stress-tolerant species such as *Orbicella*, despite broader Caribbean-wide declines (Wicquart et al., 2025). Coral cover increased significantly at WFGB over time and remained stable at EFGB. While coral cover and dominant coral species composition were broadly similar between EFGB and WFGB, analyses of overall benthic community structure over time revealed significant differences between the two banks. These differences were driven less by losses of reef-building corals and more by shifts in other benthic groups, particularly changes in macroalgae and colonizable substrate.

Although coral cover was significantly higher in repetitive photostations (60–68%, at depths ranging from 18–24 m) than in random transects within study sites (50–62%, at depths from 19–24 m) at both banks, this difference reflects the intentional placement of photostations in areas of high coral cover for tracking colony-level processes. Photostations were not designed to estimate overall reef cover or support community analysis. Both study-site transects and reef-wide transects use the same sampling design and similar analytical methods to characterize benthic cover, but differ in spatial extent. Study-site transects represent benthic composition within fixed one-hectare monitoring areas, while reef-wide transects apply the same approach across a much larger portion of each bank. In 2024, coral cover was higher in reef-wide transects than in study-site transects, indicating that study sites are not biased toward unusually high-coral areas. Instead, both transect types provide representation of benthic composition. In contrast, repetitive photostations are used to track individual coral colonies and site-specific processes such as disease, bleaching, and growth rates, while controlling for small-scale environmental heterogeneity.

As in previous years, *O. franksi* was the dominant species at both banks, with *P. strigosa*, *O. faveolata* and *P. astreoides* ranking as secondary predominant species. Patterns of change through time also differed between the two banks. At EFGB, coral communities showed a subtle but significant tendency to become more similar to one another across the study site over the monitoring record, especially in recent years. Early in the time series, coral composition varied more among transects, but since about 2018, sites have shown consistently similar coral communities. This suggests a gradual convergence in community composition rather than a major reorganization of the reef. In contrast, WFGB did not show a comparable shift. Coral communities there have maintained a similar level of variability among the site throughout the monitoring period. One possible explanation for the increasing similarity observed at EFGB is long-term stabilization of a few dominant species, potentially influenced by environmental conditions, selective mortality during past stress events, or competitive advantages of resilient taxa such as *Orbicella franksi*. However, these interpretations remain speculative. Because overall coral cover has remained high, this pattern likely reflects fine-scale reorganization within

an otherwise persistent reef community rather than a decline in reef condition. Additional data on recruitment, mortality, and coral demographics would be needed to evaluate the mechanisms underlying this subtle shift in community structure.

Macroalgae remained the second most abundant benthic group in random transects at both banks in 2024, with cover averaging around 30% over the past decade. Although macroalgae cover has fluctuated, trend analysis indicates a significant long-term increase at both EFGB and WFGB. Multivariate analysis confirmed that macroalgae was a consistent contributor to the modest but statistically significant differences in community composition between the two banks. Macroalgae and colonizable substrate were strongly negatively correlated, suggesting that macroalgae primarily expanded over available hard bottom rather than outcompeting live coral. The sustained high coral cover and lack of correlation between coral and macroalgae support previous findings that, unlike many shallow Caribbean reefs, macroalgae increases at EFGB and WFGB have not been associated with coral loss (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). This pattern instead aligns with findings from other high-latitude and subtropical reef systems, where relatively high macroalgal cover can occur under low human impact and otherwise healthy conditions (Harriott and Banks, 2002; Vroom and Braun, 2010).

Fourteen coral species were recorded in random transects at both banks in 2024, with no significant difference in species composition or diversity metrics between banks. However, reef-wide surveys were only completed at EFGB, limiting broader comparisons. Species observed in random transects but not in repetitive photostations included *Agaricia fragilis*, *Scolymia cubensis*, and *Madracis auretenra*. In a broader Caribbean context, coral species richness at EFGB and WFGB is relatively low compared to many tropical Caribbean reefs, which often support several dozen scleractinian species (Bright et al., 1985; Johnston et al., 2017a). Although EFGB and WFGB are fully developed coral reefs, their location near the northern latitudinal limit of reef development in the Gulf likely contributes to their relatively low coral species richness compared to many tropical Caribbean reefs, even as overall coral cover remains high. (Bright et al., 1985; Aronson et al., 2005). Although the coral species present on the reef caps are broadly shared with Caribbean reefs, the shallow coral reef habitats at EFGB and WFGB differ from many Caribbean systems in that octocorals (gorgonians) are absent in shallow habitats, and *Acropora* corals are exceedingly rare (Bright et al., 1985; Aronson et al., 2005; Wicquart et al., 2025).

Despite exposure to disturbances such as coral bleaching, disease outbreaks, and hurricanes, coral cover within the EFGB and WFGB study sites has remained stable or gradually increased since monitoring began in 1989. Coral cover at these sites is currently estimated to be 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 may be attributed to the banks' remote, offshore locations and the influence of surrounding deep water, which contribute to more stable environmental conditions compared to shallow, coastal reefs (Aronson et al., 2005; Johnston et al., 2015). However, relatively high coral cover has also been documented on some reefs in the southern Gulf, particularly in nearshore systems such as the Lobos–Tuxpan Reef System, where average coral cover has been reported at approximately

56%, compared to ~22% in the more impacted Veracruz Reef System (Correa et al., 2025; Wicquart et al., 2025). These reefs occur in more coastal, sediment-influenced environments than EFGB and WFGB, yet demonstrate that parts of the Gulf can retain comparatively high coral cover under a range of environmental conditions (Gil-Agudelo et al., 2020). Together, these patterns suggest that while offshore setting and deep-water influence likely contribute to the exceptional condition of the EFGB and WFGB, high coral cover in the Gulf can also occur in some nearshore systems under favorable local conditions.

Localized mortality was documented in July 2016 approximately 275 meters outside the EFGB study site, and coral bleaching was observed at both banks in the fall of 2016, 2021, and 2023. However, these events did not result in significant declines in coral cover within the study sites (Johnston et al., 2018b, 2019, 2022; Eisenbach et al., 2025). Coral paling and bleaching were again observed during summer 2024 surveys, though prevalence was lower than peak observations in September (see Chapter 3). While coral mortality associated with past stress events has been limited, the capacity of EFGB and WFGB to recover from two consecutive bleaching years remains uncertain. These patterns at FGBNMS occurred within the context of a broader regional and global disturbance event. The period from 2023 through 2025 has been characterized as a major global coral bleaching event, with bleaching-level heat stress affecting most reef regions, including the Caribbean (Wicquart et al., 2025). Regional assessments indicate that many Caribbean reefs experienced substantial bleaching, disease, and associated coral cover declines during this period, highlighting that the stressors observed at FGBNMS were not isolated events but part of a widespread response to elevated ocean temperatures (Wicquart et al., 2025). In this context, the relatively limited coral mortality observed at EFGB and WFGB to date suggests a degree of resistance, but continued monitoring will be critical to determine whether this resilience can be sustained under repeated and intensifying thermal stress.

In 2024, disease prevalence increased compared to prior years, with partial mortality observed in multiple species, including *P. strigosa*, *P. astreoides*, *M. cavernosa*, and *O. franksi*. Although disease-related impacts at EFGB and WFGB remained limited compared to many Caribbean reefs (Wicquart et al., 2025), the return of active disease following minimal observations in 2023 suggests renewed stress on coral health. Disease-related tissue loss and partial mortality were not always reflected in changes in percent coral cover, highlighting the value of qualitative photostation assessments for detecting early and localized stress responses. Some colonies showed evidence of recovery from past disease, but others experienced substantial tissue loss, indicating that disease remains an important, ongoing stressor even within an otherwise high-cover reef system. Continued disease tracking will be essential for understanding long-term trends and informing adaptive management.

Although EFGB and WFGB benefit from their depth and isolation, they are not immune to broader impacts linked to environmental change and human activity. Recurring bleaching and disease outbreaks reflect increasing regional pressures (Johnston et al., 2018b, 2019, 2023; Eisenbach et al., 2025; Wicquart et al., 2025). The 2016 localized mortality event, while severe, may have resulted from a subsurface high-salinity, low-dissolved oxygen intrusion rather than broader regional stressors (Bright and Gittings, 2023). Ongoing threats such as rising ocean temperatures, invasive species, and water quality degradation remain concerns for FGBNMS

resources (Office of National Marine Sanctuaries, 2008; Nuttall et al., 2014; Johnston, 2016b). Long-term protections in the sanctuary, including restrictions on oil and gas activities, vessel discharges, and anchoring, are intended to reduce anthropogenic stressors and support the maintenance of a healthy reef environment and ecosystem. While this study cannot directly evaluate the effectiveness of these protections, the relatively low level of local human disturbance provides important context for interpreting long-term reef condition at EFGB and WFGB. In contrast to many reefs exposed to intense coastal development, pollution, and physical damage, the sanctuary experiences comparatively limited direct human pressure, which may help sustain environmental conditions favorable to reef health. As oceanographic conditions in the Gulf continue to evolve (Karnauskas et al., 2015), sustained monitoring will remain essential to documenting ecological change and supporting adaptive conservation efforts.

Chapter 3: Coral Bleaching



A cluster of coral colonies bleaches at FGBNMS. Photo: G.P. Schmah/NOAA

Bleaching Introduction

Mild to moderate coral bleaching events have been documented at FGBNMS in previous years (e.g., 1990, 1995, 2005, 2010, 2016, 2021, 2023) 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). However, the frequency of moderate bleaching events has increased, with climate models predicting that by 2040, shallow coral reefs (<30 m) will face severe thermal stress annually (Johnston et al., 2019; Dias et al., 2023). Notably, consecutive bleaching events occurred in 2023 and 2024, raising concerns about the long-term resilience of these reefs, which have historically recovered from such stressors.

In 2024, repetitive photostations were photographed in July, when early signs of thermal stress, including some bleaching and paling, were observed in corals. Reef-wide transects were completed at EFGB during an NCRMP cruise at the end of July. Repetitive photostations at WFGB were photographed opportunistically in September to assess bleaching progression. Together, these efforts from cruises conducted June 30–July 5 (WFGB), July 14–17 (EFGB), July 29–August 2 (EFGB reef-wide), and September 22–25 (WFGB) provided the dataset used to evaluate the extent and progression of thermal stress and bleaching across the sanctuary in 2024.

Coral Bleaching Methods

Coral stress was assessed by measuring the prevalence of bleaching. Prevalence was measured in two ways: 1) as the proportion of the total percent coral cover affected by bleaching or paling and 2) 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. Bleaching percentages at WFGB in July and September 2024 were compared using a Mann-Whitney U test in R (v4.3.1).

Prevalence by number of colonies affected was calculated only for WFGB repetitive photostations in 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 WFGB study site.

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 (i.e., number of days

above high temperatures) during bleaching years, were estimated and interpolated as polynomial bleaching curves. Data from 2009 to 2023 were used because there were no gaps in seawater temperature records, and 2010, 2016, 2021, and 2023 were documented bleaching years (Johnston et al., 2019). Although 2024 also experienced widespread bleaching, threshold curves were not updated this year due to delays in instrument recovery and therefore lack of temperature data on the reef. This analysis continues to provide 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 July 2024 revealed early signs of thermal stress at EFGB, with both bleaching and paling observed, though paling was more prominent than bleaching. At EFGB, 1.89% of coral cover exhibited bleaching, while 9.13% showed signs of paling. At WFGB, bleaching was minimal at 0.15%, and paling was also low at 2.41% (Figure 3.1).

By late July, when reef-wide random transects were conducted at EFGB, paling became more widespread at EFGB, while bleaching remained at similar levels. During this survey, 1.42% of coral cover was bleached, and 12.9% was pale, suggesting continued mild coral stress. Due to time constraints, reef-wide transects were not conducted at WFGB (Figure 3.1).

By late September, repetitive photostation data indicated increased signs of coral stress at WFGB, where opportunistic surveys were conducted. Bleached coral cover rose to 10.3%, and 46.93% was pale, representing a significant increase in bleaching compared to earlier in the season ($U = 1358.5, p < .001$; Figure 3.1).

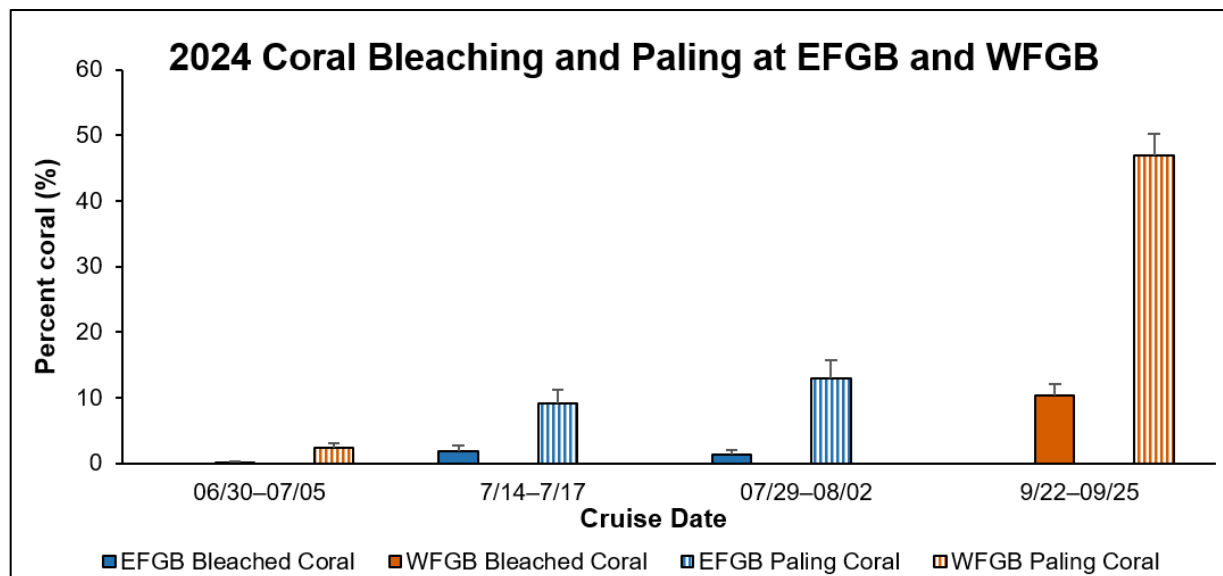


Figure 3.1. Percentage of coral cover affected by bleaching or paling at EFGB and WFGB in 2024. Data were collected from repetitive photostations on June 30–July 5 (WFGB); July 14–17 (EFGB); and September 22–25 (WFGB); and from reef-wide random transects at EFGB conducted on July 29–August 2. No repetitive photostation surveys were conducted at EFGB in September, and no reef-wide transects were conducted at WFGB; therefore, the absence of data for these banks and periods reflects lack of sampling rather than changes in bleaching condition.

Prevalence by Colony

The prevalence of coral bleaching and paling, measured as a proportion of affected colonies in repetitive photostations, was assessed at WFGB during the September cruise. On average, $12.18 \pm 1.01\%$ of coral colonies were pale and $11.56 \pm 1.81\%$ were bleached ($n=36$; Figure 3.2).

Prevalence was not calculated for July surveys due to the minimal occurrence of bleaching and paling observed during that period.

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 thresholds for each bank, the thermal stress indices calculated effectively segregate bleaching years (2010, 2016, 2021, and 2023) from non-bleaching years at EFGB and WFGB (Figure 3.2). At EFGB, 33 days or more above $30\text{ }^{\circ}\text{C}$ or 59.3 days above $29.5\text{ }^{\circ}\text{C}$ corresponded with bleaching years based on these thresholds. At WFGB, the threshold was lower; a total of 22.3 days or more above $30\text{ }^{\circ}\text{C}$ or 28.8 days above $29.9\text{ }^{\circ}\text{C}$ indicated bleaching onset (Figure 3.2).

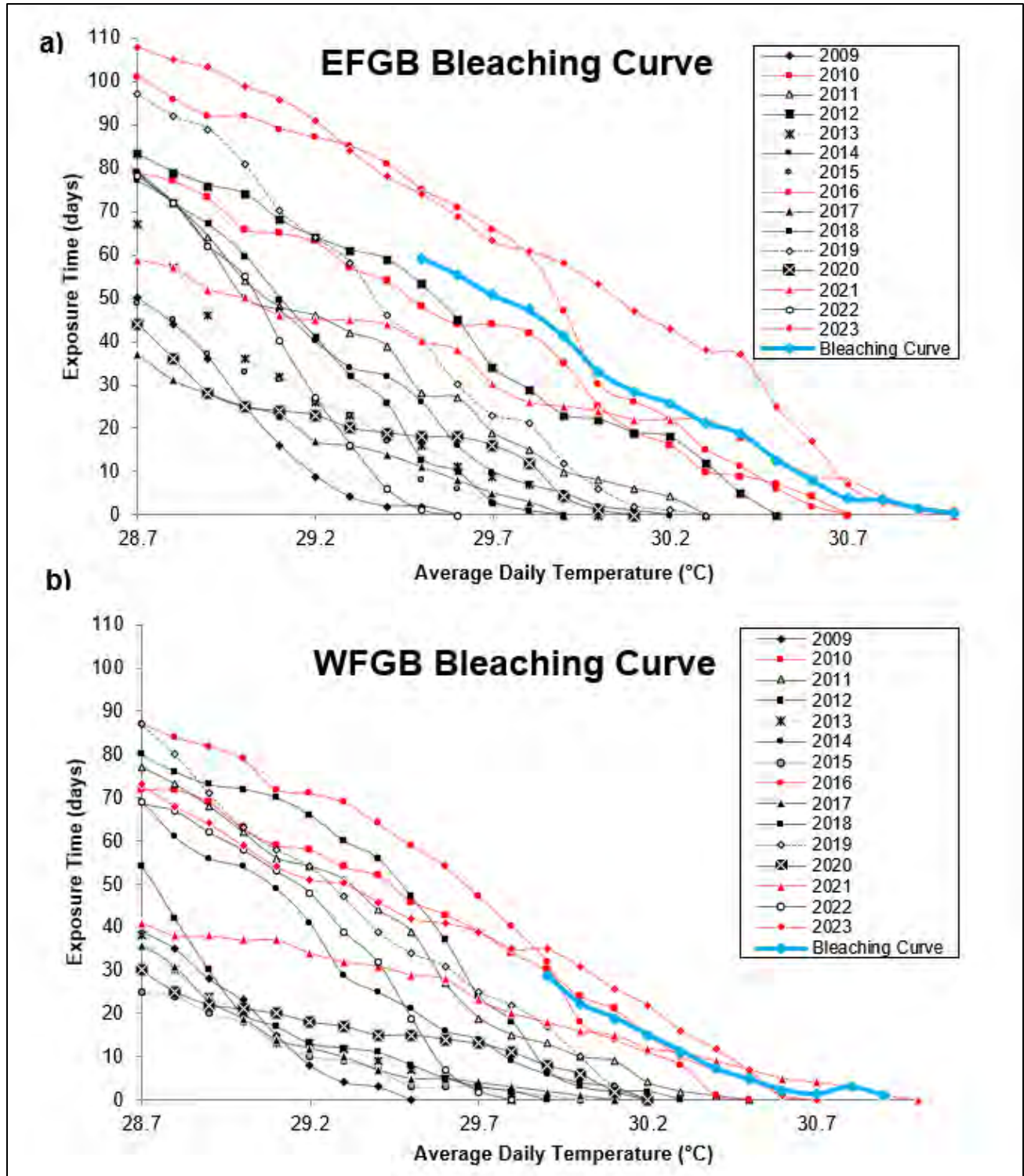


Figure 3.2. Bleaching threshold curves for (a) EFGB and (b) WFGB based on daily mean seawater temperature (°C) at depth (24 m) and exposure time (number of days). Red trend lines indicate bleaching years and black trend lines indicate non-bleaching years. Estimated bleaching curves are in blue.

At EFGB, in situ sensors at 23 m (SBE 16plus and the Sofar Ocean® buoy) recorded 71 and 82 days above 29 °C and 27 and 28 days above 30 °C in 2024, respectively. The Sofar buoy

detached during Hurricane Francine in September, but had already captured the majority of the high-temperature period. Although values remained slightly below the bleaching threshold of 33 days above 30 °C, both records exceeded the threshold of 59.3 days above 29.5 °C.

At WFGB, SBE 16plus sensors at 27 m recorded 89 days above 29.5 °C and 27 days above 30 °C, exceeding both bleaching thresholds for that site (≥ 28.8 days above 29.9 °C or ≥ 22.3 days above 30 °C). These observations verify the application of bleaching curves to indicate significant thermal stress at FGBNMS and their potential to be used as triggers for management action.

Discussion

Bleaching and paling observed at EFGB and WFGB in 2024 occurred within the context of a major global and regional thermal stress event (Reimer et al., 2024). The period from 2023 through 2024 has been identified as part of the fourth global coral bleaching event, which affected most reef regions worldwide, including the Caribbean (Reimer et al., 2024; Doherty et al., 2025; Wicquart et al., 2025). During this period, many Caribbean reefs experienced widespread bleaching and associated coral cover declines (Wicquart et al., 2025). In this broader context, bleaching observed at EFGB and WFGB in 2024 reflects stressors affecting reefs across the Caribbean and Gulf and provides an important opportunity to evaluate how a historically resilient offshore reef system responds to repeated and intensifying thermal stress.

The prevalence of coral bleaching and paling at EFGB and WFGB in 2024 indicated moderate thermal stress, with paling slightly more prevalent than full bleaching. During the 2023–2024 global bleaching event, some regions experienced rapid escalation from thermal stress to widespread bleaching and mortality, such as in Martinique where 86% of colonies bleached in 2023 and recovery was further limited by continued warming in 2024 (Bon et al., 2025; Wicquart et al., 2025). Similarly severe impacts were documented in the Cayman Islands, where peak bleaching approached 80% at Little Cayman and resulted in roughly 54% coral mortality; impacts were greatest in shallow, specialist species, while deeper-living and stress-tolerant taxa exhibited higher survival (Goodbody-Gringley & Chequer, 2025; Doherty et al., 2025). The predominance of paling at EFGB and WFGB during much of the 2024 season suggests that, although thermal stress was sufficient to trigger visible stress responses, the intensity and duration of exposure were generally lower than in some Caribbean locations that experienced more severe bleaching and widespread mortality during the same event.

Compared to prior bleaching years at FGBNMS, the 2024 event was notable less for extreme bleaching prevalence and more for its timing and repetition. Bleaching and paling have been documented at the banks in previous warm years, including 2010, 2016, 2021, and 2023, with most events followed by limited mortality and evidence of recovery (Aronson et al., 2005; Johnston et al., 2015; Johnston et al., 2018b; Johnston et al., 2019; Johnston et al., 2022; Eisenbach et al., 2025). Although bleaching can be a precursor to coral mortality, most bleached corals at FGBNMS have historically recovered following thermal stress events and were not accompanied by significant declines in coral cover (Johnston et al., 2019, 2023; Eisenbach et al., 2025). 2024 represents the first instance in which two consecutive bleaching years occurred at EFGB and WFGB. Earlier bleaching events were typically separated by cooler years that allowed for recovery, whereas the close spacing of the 2023 and 2024 events introduces the potential for

cumulative stress. Recovery from the 2024 event has not yet been documented, leaving uncertainty about longer-term consequences of repeated thermal stress.

In addition, thermal stress and visible paling developed relatively early in the 2024 season compared to some past events, and late-season surveys were not completed at both banks, limiting direct comparison of peak bleaching conditions. Together, these differences suggest that although bleaching severity in 2024 was moderate relative to some past events, its occurrence immediately following another major heat-stress year marks a meaningful shift in thermal stress experienced by EFGB and WFGB.

Differences in bleaching patterns between EFGB and WFGB in 2024 suggest that local oceanographic conditions likely influence thermal stress outcomes at the banks. At EFGB, in situ records indicated sustained exposure to elevated temperatures during the summer, though overall heat stress appeared slightly lower than in 2023 based on the number of days above critical thresholds. Early-season surveys documented only mild bleaching, but the absence of late-season assessments, when bleaching typically peaks, limited the ability to fully evaluate seasonal impacts and directly compare outcomes across years. At WFGB, late-season surveys documented increased bleaching and paling relative to earlier summer conditions, consistent with continued exposure to thermal stress into September. Differences between EFGB and WFGB have been observed during past disturbance events. For example, during the 2016 bleaching event, EFGB experienced substantially higher bleaching prevalence than WFGB due to differences in thermal exposure at depth, despite both banks being subjected to the same regional heat-stress event (Johnston et al., 2019). Additionally, the localized mortality event in 2016 occurred only at EFGB, indicating that local environmental conditions likely played an important role and that not all disturbance events affecting the banks are driven by the same regional processes (Johnston et al., 2018b; Johnston et al., 2019).

Although depth differences between photostations at EFGB and WFGB are small, temperature differences between banks are more likely driven by oceanographic processes (Johnston et al., 2019). Circulation in the northern Gulf is strongly influenced by prevailing currents and the position and persistence of loop current eddies (Jarosz et al., 2014; Muller-Karger et al., 2015; Johnston et al., 2019), which can create differences in water quality between EFGB and WFGB. Large-scale disturbances such as hurricanes can also differentially affect the banks depending on storm tracks, intensity, and local circulation patterns (Aronson et al., 2005; Johnston et al., 2019). While 2024 sampling limitations prevent a full seasonal comparison between banks, these circulation-driven differences provide a plausible explanation for why bleaching responses may vary spatially across the sanctuary, and highlight the importance of coordinated, late-season monitoring at both banks during future thermal stress events. Measuring bleaching and paling using both percent cover and colony-level prevalence provides a more complete picture of reef condition during thermal stress. While differences between these metrics may be small in non-stress years, prevalence surveys are especially valuable during bleaching events because they detect early or subtle stress responses that may not yet be apparent in changes to coral cover.

Improved understanding of bleaching dynamics at FGBNMS will require more consistent and coordinated survey coverage across both banks during bleaching years. Currently, bleaching

response efforts are conducted opportunistically, often constrained by vessel availability, staffing, and funding limitations, resulting spatial and temporal gaps in data. Dedicated response missions that include both repetitive photostations and reef-wide random transects at each bank during peak thermal stress periods would provide more comprehensive datasets and improve interpretation of bleaching patterns.

The development of site-specific bleaching threshold curves at EFGB and WFGB provides an additional tool for guiding these efforts. These thresholds successfully identified years with documented bleaching, demonstrating their value as early-warning indicators of bleaching risk. When combined with real-time temperature data, threshold exceedance could be used to trigger targeted response surveys and management actions, allowing staff to prioritize monitoring during periods of highest risk. Integrating threshold-based alerts with coordinated field surveys would strengthen the sanctuary's capacity to detect bleaching early, assess its severity, and support timely, adaptive management of sanctuary resources.

The occurrence of coral thermal stress in both 2023 and 2024 represents the first back-to-back bleaching years recorded at FGBNMS and underscores the growing frequency and intensity of coral bleaching events, as predicted with projected climate models (Heron et al., 2016; Hughes et al., 2017, 2018; Dias et al., 2023). Although the severity and spatial patterns varied between banks and years, the recurrence of thermal stress events aligns with broader regional trends and projections of increased bleaching frequency due to ocean warming (Dias et al., 2023; Wicquart et al., 2025). Continued long-term monitoring will be critical for understanding how repeated exposure affects coral community resilience and for guiding responsive management strategies within the sanctuary. In particular, fate tracking of individual coral colonies through repetitive photostations and emerging approaches such as photogrammetry will be increasingly important for documenting partial mortality, recovery, and long-term survival following repeated stress events. The persistence of such events in upcoming years further underscores the need for broader climate intervention strategies, as local protections alone cannot prevent thermal stress driven by large-scale ocean warming. Safeguarding these reefs is essential not only for the sanctuary's long-term resilience, but also because their relatively high coral cover, history of recovery following past stress events, and persistence of stress-tolerant species (Aronson et al., 2005; Johnston et al., 2019) make them an important potential source of ecological insight and genetic diversity for informing potential recovery pathways in the Gulf and Caribbean.

Chapter 4: Sea Urchin and Lobster Surveys



A Caribbean spiny lobster (*Panulirus argus*) walking over star corals (*Orbicella* spp.). Photo: G.P. Schmahl/NOAA

Sea Urchin and Lobster Introduction

The long-spined sea urchin (*Diadema antillarum*) is an important herbivore on coral reefs throughout the Caribbean, but in 1983 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 America; 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.

Sea Urchin and Lobster Methods

Field Methods

Due to the nocturnal behavior of *Diadema antillarum*, *Panulirus argus*, and *Panulirus guttatus*, visual surveys were conducted at night, a minimum of 1.5 hours after sunset. Surveys

are conducted along six 100-m transect lines at each study site (four perimeter lines and two center crosshairs), with a 2-m-wide belt transect surveyed along each line, totaling 1,200 m² per bank. Two divers swim slowly along each transect, each surveying a 1-m swath on either side of the line using flashlights to inspect reef crevices for individuals of the target species.

In 2024, limited field access reduced survey coverage. At EFGB, four perimeter transects were completed, totaling 800 m² surveyed. At WFGB, the same four perimeter transects were completed, along with one 50-m centerline, resulting in a total surveyed area of 900 m².

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

Each transect represented one sample. Sea urchins and lobsters observed on each 100-m transect line were summed and density was calculated as number of individuals per 100 m² for each species \pm SE. In 2024, sampling effort differed slightly between banks: EFGB was surveyed using four 100 m transects, while WFGB included four 100 m transects and one additional 50 m transect. The 50 m transect at WFGB was scaled and included in density calculations (individuals per 100 m²). 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 and correlation tests in R version 4.2.2 (Clarke et al., 2014).

Sea Urchin and Lobster Results

Density of *D. antillarum* was 3.25 ± 0.15 individuals/100 m² within the EFGB study site and 74.33 ± 2.11 individuals/100 m² within the WFGB study site in 2024 (Figure 4.1). Two *P. guttatus* was observed in the EFGB study site and no *P. argus* were observed at WFGB (Figure 4.2).

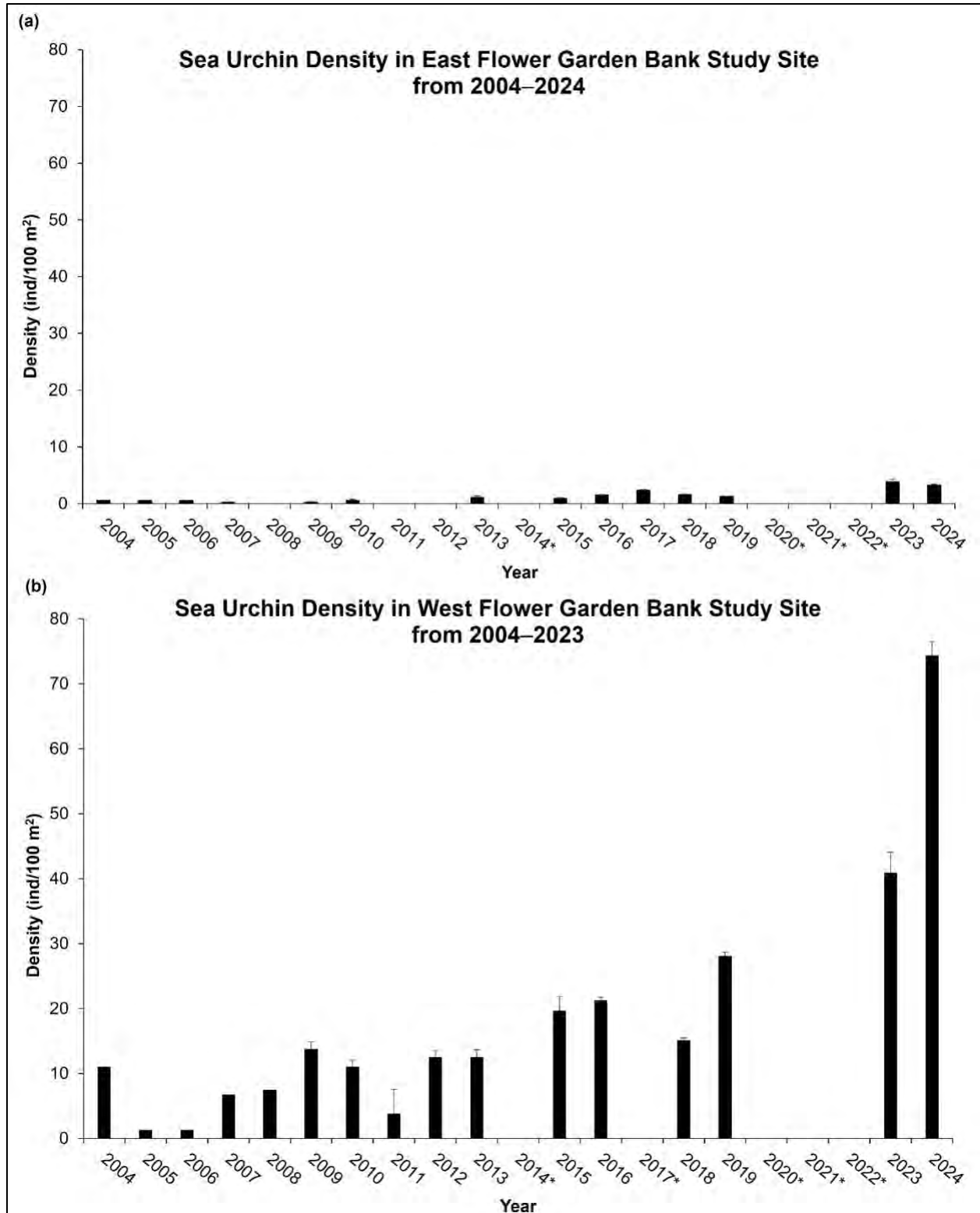


Figure 4.1. Mean sea urchin density (individuals per 100 m²) \pm SE at (a) East Flower Garden Bank and (b) West Flower Garden Bank from 2004 to 2024. Asterisks following a year indicate no data were available for that year. Standard error values were not available from 2004 to 2008. Sources: Precht et al., 2006; Zimmer et al., 2010 (2004–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009–2023); Eisenbach et al., 2025 (2024).

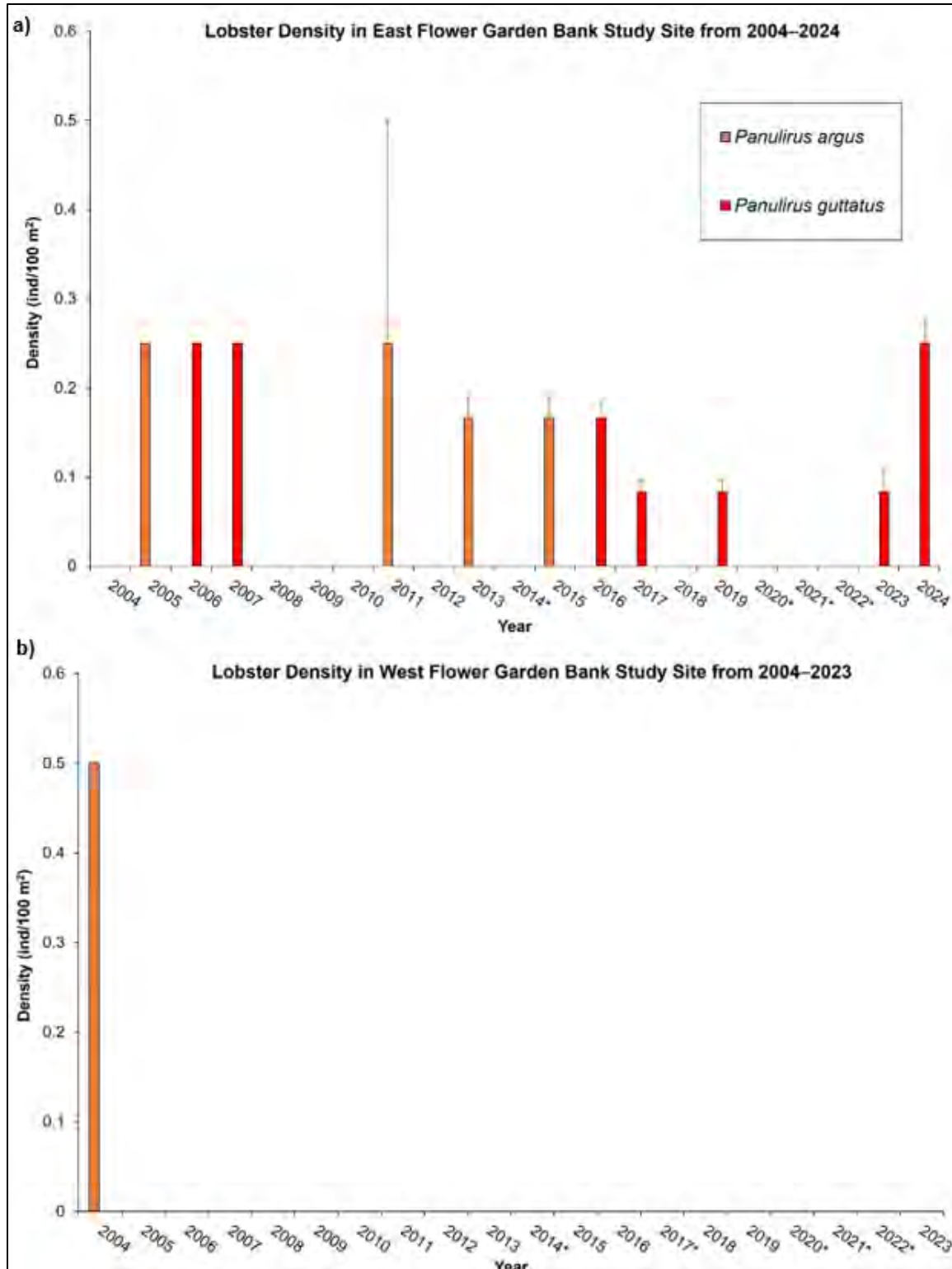


Figure 4.2. Mean lobster density (individuals per 100 m²) \pm SE at (a) East Flower Garden Bank and (b) West Flower Garden Bank from 2004 to 2024. Asterisks following a year indicate no data were available for that year. Standard error values were not available from 2004 to 2008. Sources: Precht et al., 2006; Zimmer et al., 2010 (2004–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009–2023); Eisenbach et al., 2025 (2024).

Since 2004, *D. antillarum* densities have ranged from 0–3.83 individuals/100 m² within the EFGB study site and 1.25–74.33 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- $F_{1,39} = 27.34$, $p < 0.05$). 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 2024. Coherence plots found no 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). Following the mass die-off in 1983, *D. antillarum* populations have not recovered to historic levels, where pre-die off densities were estimated at more than 140 individuals/100 m² at EFGB (Gittings & Bright, 1986, 1987; Gittings, 1998). At WFGB, pre-die off densities were lower than at EFGB, estimated at 50 individuals/100 m². For the first time since the die-off, *D. antillarum* densities at WFGB exceeded the historic levels in 2024.

After 1983, densities declined to near zero at both banks (Gittings and Bright, 1987). Recovery throughout the Caribbean has been slow and patchy since the die-off (Edmunds and Carpenter, 2001; Kramer, 2003; Carpenter and Edmunds, 2006). At nearby Stetson Bank, *D. antillarum* densities increased from 2009 to 2014 and have plateaued in recent years, averaging 118 individuals/100 m² in 2023 (Nuttall et al., 2020; O'Connell et al., 2024). No pre-die-off estimates of sea urchin abundance are available for Stetson Bank.

Diadema antillarum populations within the EFGB study site remained low during the 2024 monitoring period, consistent with densities reported in previous studies (Zimmer et al., 2010; Johnston et al., 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025). Populations have been consistently higher within the WFGB study site than the EFGB study site. However, the lack of correlation between sea urchin abundance and algal cover suggests that urchins are not currently exerting significant control on benthic algae, likely due to incomplete recovery to pre-die-off densities. Studies suggest sea urchin densities >1 per m² contribute to healthy and resilient reef systems (Mumby, et al. 2006, 2007). While neither EFGB or WFGB have urchin densities are above that threshold, WFGB urchin densities are close at 0.74 individuals per m².

Lobster densities within the EFGB and WFGB study sites have remained low throughout the monitoring program with no lobster observed in surveys at WFGB since 2004. Although occasionally observed by divers, lobsters occur in low abundance across these two banks.

In 2022, another mass die-off of *Diadema antillarum* occurred in the Caribbean resulting in high mortality rates (Hylkema et al., 2023; Butler et al., 2024). The illness, caused by a

scuticociliate (Hewson et al., 2023), expressed similar symptoms to the die-off in the 1980s and quickly spread across thousands of kilometers in the Caribbean (Hylkema et al., 2023). While this second mass-die has not impacted FGBNMS urchin populations, these annual surveys allow us to closely monitor populations for changes if a delayed impact occurs.

Chapter 5: Fish Surveys



A school of blue tangs (*Acanthurus coeruleus*) above the reef: Photo: G.P. Schmahl/NOAA

Fish Surveys Introduction

Fish are important indicators of ecosystem health and contribute to socioeconomics via fishing activity (Knowlton & Jackson, 2008; Jackson et al., 2014). Divers conducted stationary reef fish visual surveys in EFGB and WFGB study sites to examine and compare fish community composition and changes over time. The sanctuary tracks commercially important, foundation, and invasive/non-native species populations to monitor their impacts on the reef. Commercially important species such as groupers and snappers are not only important to the ecosystem health, but also the local economy. Invasive lionfish (*Pterois volitans*) were first observed in the sanctuary in 2011 and non-native regal demoiselle (*Neopomacentrus cyanomos*) were first observed in 2019.

Fish Survey Methods

Field Methods

Fishes were assessed by divers using the modified stationary reef fish visual census originally described by Bohnsack and Bannerot (1986). Twenty-six randomly located surveys were conducted within the study site at EFGB and 24 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 and fish were counted and binned into size categories (for more detailed methods, refer to Johnston et al. [2017a]; Figure 5.1).



Figure 5.1. A NOAA diver conducts a fish survey at EFGB. Photo: Jesse Cancelmo/Jesse Cancelmo Photography

All fish species observed within the first five minutes of the survey were recorded while the diver slowly rotated in place in the center of an 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 2024 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. Datasheets were retained, reviewed, and compared to data entered in the database to check for entry errors, and 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) and based on information from FishBase (Froese & Pauly, 2019).

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 \pm SE per 100 m², and calculated as the total number of individuals per sample divided by the area of the survey cylinder (176.7 m²) 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 \pm SE. Fish biomass was expressed as kilograms \pm SE per 100 m² 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, stingrays, and sharks were removed from biomass analyses only, due to their rare nature and large size.

For family level 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).

Fish Survey Results

A combined total of 23 families and 68 species (55 at EFGB and 57 at WFGB) were observed in 2024 at both study sites. Mean species richness was 19.85 ± 0.76 per survey at EFGB, 19.50 ± 0.54 per survey at WFGB, and 19.68 ± 0.47 per survey for both study sites combined. Brown chromis (*Azurina multilineata*) had the highest relative abundance of all species in EFGB surveys (40.14%), followed by creole wrasse (*Bodianus parrae*; 16.98%), Atlantic creolefish (*Paranthias furcifer*; 11.71%), bluehead (*Thalassoma bifasciatum*; 7.67%), and bicolor damselfish (*Stegastes partitus*; 3.68%). In WFGB surveys, brown chromis had the highest relative abundance (43.20%), followed by Atlantic creolefish (13.85%), creole wrasse (10.74%), bluehead (4.53%), and blue chromis (*Azurina cyanea*; 2.85%).

Sighting Frequency and Occurrence

The most frequently sighted species were brown chromis and bluehead, observed in 98% of surveys at EFGB and WFGB combined. Other frequently sighted species included blue tang

(*Acanthurus coeruleus*), sharpnose puffer (*Canthigaster callisterna*) and bicolor damselfish (Table 5.1). No manta ray (*Mobula birostris*) or 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 2024.

Fish Species	EFGB	WFGB	Combined
Bluehead (<i>Thalassoma bifasciatum</i>)	96.2%	100.0%	98.0%
Brown Chromis (<i>Azurina multilineata</i>)	100.0%	95.8%	98.0%
Blue Tang (<i>Acanthurus coeruleus</i>)	96.2%	91.7%	94.0%
Sharpnose Puffer (<i>Canthigaster callisterna</i>)	92.3%	91.7%	92.0%
Bicolor Damselfish (<i>Stegastes partitus</i>)	88.5%	79.2%	84.0%
Spanish Hogfish (<i>Bodianus rufus</i>)	84.6%	83.3%	84.0%
Creole Wrasse (<i>Bodianus parrae</i>)	80.8%	83.3%	82.0%
Blue Chromis (<i>Azurina cyanea</i>)	73.1%	87.5%	80.0%
Atlantic Creolefish (<i>Paranthias furcifer</i>)	69.2%	87.5%	78.0%
Great Barracuda (<i>Sphyraena barracuda</i>)	65.4%	87.5%	76.0%

Density

Mean fish density (individuals/100 m²) was 113.60 ± 15.34 in EFGB surveys, 102.50 ± 13.33 in WFGB surveys, and 108.27 ± 10.15 for all study site surveys combined. Density was significantly higher in EFGB surveys (Table 5.2). SIMPER analysis identified greater abundance of blue tang (7.47%) and sharpnose puffer (5.92%) at WFGB and greater abundance of bluehead (4.90%) 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 2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	6307.1	1	3.7357	0.0001
Res	81041	48		
Total	87348	49		

Table 5.3. Mean density (individuals ± SE/100 m²) of the 15 most abundant species from EFGB and WFGB study site surveys in 2024. The density of **bolded** species was significantly different between EFGB and WFGB.

Fish Species	EFGB	WFGB	Combined
Brown chromis (<i>Azurina multilineata</i>)	45.6 ± 7.3	44.3 ± 7.9	45.0 ± 5.3
Creole wrasse (<i>Bodianus parrae</i>)	19.3 ± 6.8	11.0 ± 2.6	15.3 ± 3.8
Atlantic creolefish (<i>Paranthias furcifer</i>)	13.3 ± 3.5	14.2 ± 4.3	13.7 ± 2.7
Bluehead (<i>Thalassoma bifasciatum</i>)	8.7 ± 1.5	4.6 ± 0.6	6.8 ± 0.9
Bicolor damselfish (<i>Stegastes partitus</i>)	4.2 ± 0.8	2.4 ± 0.5	3.3 ± 0.5
Chub complex (<i>Kyphosus</i> spp.)	3.3 ± 1.0	1.0 ± 0.3	2.2 ± 0.6
Blue chromis (<i>Azurina cyanea</i>)	1.2 ± 0.3	2.9 ± 1.1	2.0 ± 0.6
Blue tang (<i>Acanthurus coeruleus</i>)	1.3 ± 0.1	2.4 ± 0.3	1.8 ± 0.2

Fish Species	EFGB	WFGB	Combined
Sharpnose puffer (<i>Canthigaster callisterna</i>)	1.3 ± 0.2	2.4 ± 0.3	1.8 ± 0.2
Threespot damselfish (<i>Stegastes planifrons</i>)	1.5 ± 0.3	1.5 ± 0.4	1.5 ± 0.2
Queen Parrotfish (<i>Scarus vetula</i>)	1.6 ± 0.3	1.2 ± 0.2	1.4 ± 0.2
Cocoa damselfish (<i>Stegastes variabilis</i>)	1.5 ± 0.4	0.8 ± 0.2	1.2 ± 0.2
Spanish hogfish (<i>Bodianus rufus</i>)	1.1 ± 0.2	1.2 ± 0.2	1.2 ± 0.1
Reef butterflyfish (<i>Chaetodon sedentarius</i>)	0.9 ± 0.2	1.3 ± 0.2	1.1 ± 0.2
Great barracuda (<i>Sphyaena barracuda</i>)	0.6 ± 0.1	1.5 ± 0.5	1.0 ± 0.2
Mean Density	113.60 ± 15.34	102.50 ± 13.33	108.27 ± 10.15

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). Invertivores comprised the majority of the small size class at both study sites. Planktivores were predominant in the mid-range size classes while piscivores and herbivores predominated the large size classes (Figure 5.2)

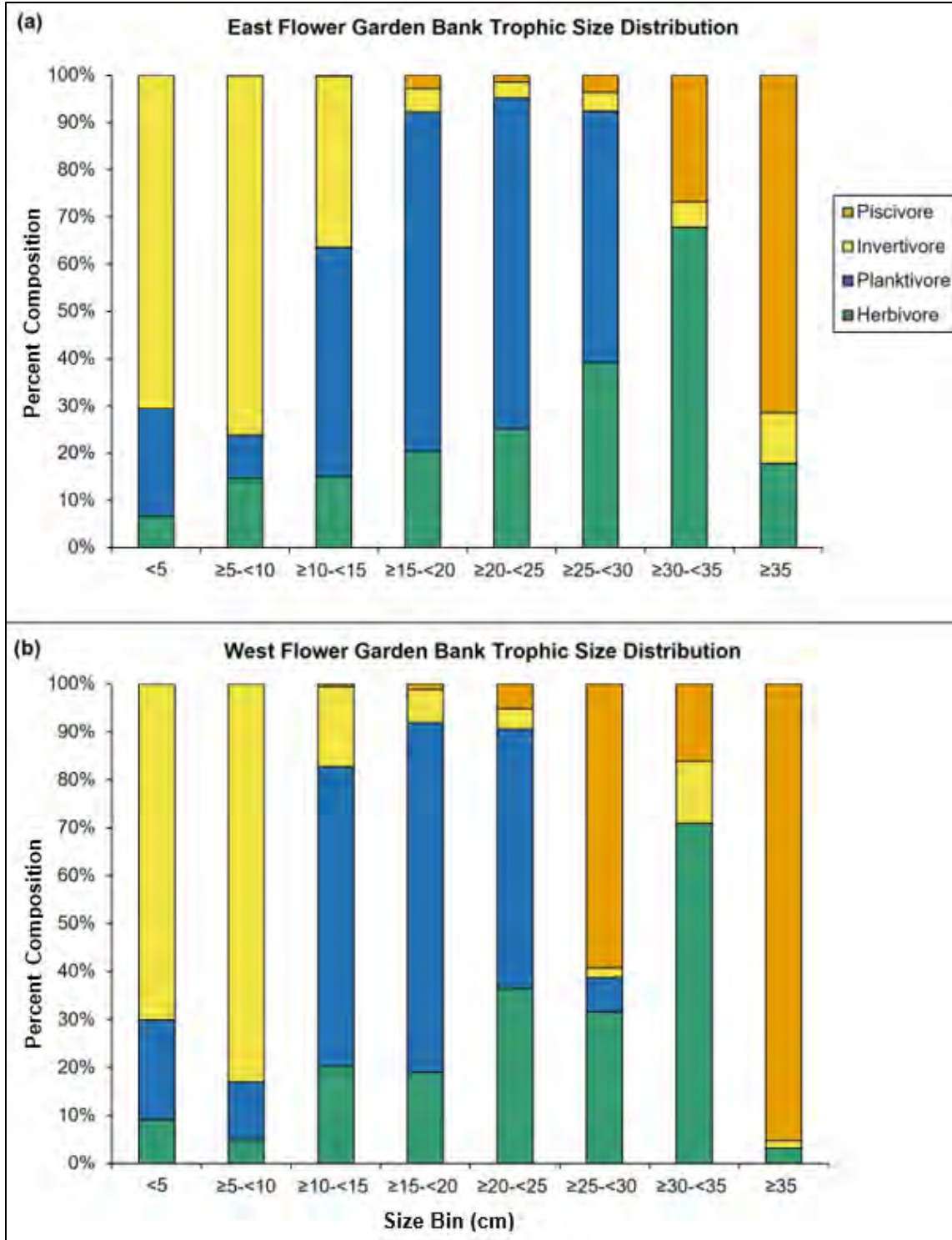


Figure 5.2. Fish size distribution by trophic guild at (a) EFGB and (b) WFGB study sites in 2024.

Biomass

Mean biomass ($\text{kg} \pm \text{SE}/100 \text{ m}_2$) in 2024 was 6.33 ± 1.20 in EFGB surveys, 5.68 ± 0.80 in WFGB surveys, and 6.02 ± 0.73 across both study site surveys combined. PERMANOVA analysis indicated that fish community based on biomass differed significantly between banks (Table 5.4). SIMPER analysis identified the main contributor to differences were a greater local biomass of queen parrotfish (*Scarus vetula*; 6.01%) and brown chromis (4.63%) at EFGB and redband parrotfish (*Sparisoma aurofrenatum*; 4.25%) at WFGB.

Table 5.4. PERMANOVA results comparing fish communities based on biomass at EFGB and WFGB study sites in 2024. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	6320.6	1	2.3644	0.0005
Res	1.2831E+05	48		
Total	1.3463E+05	49		

When classified by trophic guild, herbivores exhibited the highest mean biomass across all surveys, while invertivores had the lowest (Table 5.5). Significant differences in trophic guild were detected between study sites. Across all sites combined, herbivores accounted for approximately 35% of total biomass, followed by planktivores (32%), piscivores (24%), and invertivores (9%).

Table 5.5. Mean biomass ($\text{kg} \pm \text{SE}/100 \text{ m}_2$) for each trophic guild from EFGB and WFGB study site surveys, and surveys from both banks combined, in 2024.

Trophic Guild	EFGB	WFGB	Combined
Herbivore	2.59 ± 0.51	1.59 ± 0.32	2.11 ± 0.30
Invertivore	0.65 ± 0.13	0.44 ± 0.09	0.55 ± 0.08
Planktivore	2.51 ± 0.49	1.24 ± 0.25	1.90 ± 0.27
Piscivore	0.57 ± 0.11	2.42 ± 0.49	1.46 ± 0.21

Mean biomass for each species, grouped by trophic guild, is presented in Table 5.6. At the EFGB study site, 48% of herbivore biomass was contributed by the chub complex (*Kyphosus* spp.). For invertivores, the greatest contribution was from gray snapper (*Lutjanus griseus*; 27%). For piscivores, great barracuda contributed the greatest biomass (55%). For planktivores, the greatest contribution was from Atlantic creolefish (74%). At the WFGB study site, 18% of herbivore biomass was contributed by the chub complex. For invertivores, the greatest contribution was from brown chromis (31%). For piscivores, great barracuda contributed the greatest biomass (52%). For planktivores, the greatest contribution was from Atlantic creolefish (64%; Table 5.6).

Table 5.6 Biomass ($\text{kg} \pm \text{SE}/100 \text{ m}_2$) of each species grouped by trophic guild, from EFGB and WFGB study site surveys, and surveys from both banks combined, in 2024.

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Herbivore	Chub complex (<i>Kyphosus</i> spp.)	1.25 ± 0.56	0.29 ± 0.10	0.79 ± 0.30
Herbivore	Stoplight parrotfish (<i>Sparisoma viride</i>)	0.46 ± 0.09	0.27 ± 0.10	0.37 ± 0.07
Herbivore	Queen parrotfish (<i>Scarus vetula</i>)	0.26 ± 0.06	0.26 ± 0.05	0.26 ± 0.04
Herbivore	Black durgon (<i>Melichthys niger</i>)	0.18 ± 0.06	0.23 ± 0.05	0.21 ± 0.04

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Herbivore	Redband parrotfish (<i>Sparisoma aurofrenatum</i>)	0.08 ± 0.02	0.23 ± 0.06	0.15 ± 0.03
Herbivore	Blue tang (<i>Acanthurus coeruleus</i>)	0.14 ± 0.03	0.16 ± 0.03	0.15 ± 0.02
Herbivore	Princess parrotfish (<i>Scarus taeniopterus</i>)	0.09 ± 0.02	0.10 ± 0.04	0.10 ± 0.02
Herbivore	Bicolor damselfish (<i>Stegastes partitus</i>)	0.03 ± 0.01	0.01 ± <0.01	0.02 ± <0.01
Herbivore	Doctorfish (<i>Acanthurus chirurgus</i>)	0.03 ± 0.01	<0.01 ± <0.01	0.02 ± 0.01
Herbivore	Ocean surgeonfish (<i>Acanthurus tractus</i>)	0.02 ± 0.01	0.01 ± <0.01	0.01 ± <0.01
Herbivore	Striped parrotfish (<i>Scarus iseri</i>)	0.02 ± 0.01	0.00 ± 0.00	0.01 ± 0.01
Herbivore	Yellowtail damselfish (<i>Microspathodon chrysurus</i>)	0.01 ± <0.01	0.02 ± 0.01	0.01 ± <0.01
Herbivore	Cocoa damselfish (<i>Stegastes variabilis</i>)	0.01 ± <0.01	0.01 ± <0.01	0.01 ± <0.01
Herbivore	Bucktooth parrotfish (<i>Sparisoma radians</i>)	0.00 ± 0.00	0.01 ± <0.01	<0.01 ± <0.01
Herbivore	Dusky damselfish (<i>Stegastes adustus</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Herbivore	Redlip blenny (<i>Ophioblennius macclurei</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Herbivore	Greenblotch parrotfish (<i>Sparisoma atomarium</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Brown chromis (<i>Azurina multilineata</i>)	0.15 ± 0.03	0.14 ± 0.03	0.14 ± 0.02
Invertivore	Gray snapper (<i>Lutjanus griseus</i>)	0.18 ± 0.14	0.02 ± 0.01	0.10 ± 0.07
Invertivore	Spanish hogfish (<i>Bodianus rufus</i>)	0.06 ± 0.02	0.05 ± 0.02	0.06 ± 0.01
Invertivore	Reef butterflyfish (<i>Chaetodon sedentarius</i>)	0.03 ± 0.01	0.04 ± 0.01	0.04 ± 0.01
Invertivore	Yellow goatfish (<i>Mulloidichthys martinicus</i>)	0.05 ± 0.04	0.02 ± 0.01	0.03 ± 0.02
Invertivore	Rock beauty (<i>Holacanthus tricolor</i>)	0.03 ± 0.02	0.02 ± 0.01	0.02 ± 0.01
Invertivore	Bluehead (<i>Thalassoma bifasciatum</i>)	0.03 ± 0.02	0.01 ± <0.01	0.02 ± 0.01
Invertivore	Ocean triggerfish (<i>Canthidermis sufflamen</i>)	0.00 ± 0.00	0.04 ± 0.04	0.02 ± 0.02
Invertivore	Rock hind (<i>Epinephelus adscensionis</i>)	<0.01 ± <0.01	0.03 ± 0.03	0.02 ± 0.02
Invertivore	Threespot damselfish (<i>Stegastes planifrons</i>)	0.02 ± 0.01	0.01 ± <0.01	0.02 ± <0.01
Invertivore	Smooth trunkfish (<i>Lactophrys triqueter</i>)	0.03 ± 0.01	<0.01 ± <0.01	0.02 ± 0.01
Invertivore	Squirrelfish (<i>Holocentrus adscensionis</i>)	0.02 ± 0.01	<0.01 ± <0.01	0.01 ± 0.01
Invertivore	Queen angelfish (<i>Holacanthus ciliaris</i>)	0.02 ± 0.01	0.00 ± 0.00	0.01 ± 0.01
Invertivore	Yellowhead wrasse (<i>Halichoeres garnoti</i>)	0.01 ± <0.01	0.01 ± <0.01	0.01 ± <0.01
Invertivore	Porcupinefish (<i>Diodon hystrix</i>)	0.00 ± 0.00	0.02 ± 0.02	0.01 ± 0.01

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Invertivore	Honeycomb cowfish (<i>Acanthostracion polygonium</i>)	0.01 ± 0.01	0.00 ± 0.00	0.01 ± <0.01
Invertivore	Gray triggerfish (<i>Balistes capriscus</i>)	0.00 ± 0.00	0.01 ± 0.01	0.01 ± 0.01
Invertivore	Neon goby (<i>Elacatinus oceanops</i>)	<0.01 ± <0.01	0.01 ± 0.01	0.01 ± 0.01
Invertivore	Sharpnose puffer (<i>Canthigaster callisterna</i>)	<0.01 ± 0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Sergeant major (<i>Abudefduf saxatilis</i>)	0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Spotfin butterflyfish (<i>Chaetodon ocellatus</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Longsnout butterflyfish (<i>Prognathodes aculeatus</i>)	<0.01 ± 0.000	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Clown wrasse (<i>Halichoeres maculipinna</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Spotted trunkfish (<i>Lactophrys bicaudalis</i>)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Beaugregory (<i>Stegastes leucostictus</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Orangespotted filefish (<i>Cantherhines pullus</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Queen triggerfish (<i>Balistes vetula</i>)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Banded butterflyfish (<i>Chaetodon striatus</i>)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Puddingwife (<i>Halichoeres radiatus</i>)	<0.01 ± <0.01	0.00 ± 0.00	<0.01 ± <0.01
Invertivore	Goldentail moray (<i>Gymnothorax miliaris</i>)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Scrawled cowfish (<i>Acanthostracion quadricornis</i>)	<0.01 ± <0.01	0.00 ± 0.00	<0.01 ± <0.01
Invertivore	Bridled goby complex (<i>Coryphopterus glaucofraenum</i> complex)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Invertivore	Redspotted hawkfish (<i>Amblycirrhitus pinos</i>)	<0.01 ± <0.01	0.00 ± 0.00	<0.01 ± <0.01
Piscivore	Great barracuda (<i>Sphyraena barracuda</i>)	0.32 ± 0.08	1.25 ± 0.40	0.77 ± 0.20
Piscivore	Rainbow runner (<i>Elagatis bipinnulata</i>)	0.01 ± 0.01	0.46 ± 0.46	0.23 ± 0.22
Piscivore	Dog snapper (<i>Lutjanus jocu</i>)	0.01 ± 0.01	0.45 ± 0.23	0.22 ± 0.11
Piscivore	Horse-eye jack (<i>Caranx latus</i>)	0.12 ± 0.07	0.12 ± 0.12	0.12 ± 0.07
Piscivore	Lionfish (<i>Pterois volitans/miles</i>)	0.00 ± 0.00	0.08 ± 0.03	0.04 ± 0.02
Piscivore	Graysby (<i>Cephalopholis cruentata</i>)	0.04 ± 0.01	0.02 ± 0.01	0.03 ± 0.01
Piscivore	Yellowmouth grouper (<i>Mycteroperca interstitialis</i>)	0.05 ± 0.03	0.00 ± 0.00	0.02 ± 0.02
Piscivore	Red snapper (<i>Lutjanus campechanus</i>)	0.00 ± 0.00	0.03 ± 0.03	0.01 ± 0.01
Piscivore	Black jack (<i>Caranx lugubris</i>)	0.02 ± 0.01	0.00 ± 0.00	0.01 ± 0.01

Trophic Guild	Fish Species	EFGB	WFGB	Combined
Piscivore	Bar jack (<i>Caranx ruber</i>)	0.01 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01
Piscivore	Scamp (<i>Mycteroperca phenax</i>)	<0.01 ± <0.01	0.00 ± 0.00	<0.01 ± <0.01
Planktivore	Atlantic creolefish (<i>Paranthias furcifer</i>)	1.87 ± 0.72	0.79 ± 0.24	1.35 ± 0.40
Planktivore	Creole wrasse (<i>Bodianus parrae</i>)	0.64 ± 0.19	0.44 ± 0.14	0.54 ± 0.12
Planktivore	Blue chromis (<i>Azurina cyanea</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Planktivore	Sunshinefish (<i>Chromis insolata</i>)	<0.01 ± <0.01	<0.01 ± <0.01	<0.01 ± <0.01
Planktivore	Glassy sweeper (<i>Pempheris schomburgkii</i>)	<0.01 ± <0.01	0.00 ± 0.00	<0.01 ± <0.01
Planktivore	Purple reef fish (<i>Chromis scotti</i>)	0.00 ± 0.00	<0.01 ± <0.01	<0.01 ± <0.01

Abundance-Biomass Curves

Mean w-values for both the EFGB and WFGB study sites were 0.06 ± 0.01 . For all samples at each study site, mean w-values remained close to zero, 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	0.039794	1	2.3655	0.1309
Res	0.80751	48		
Total	0.84731	49		

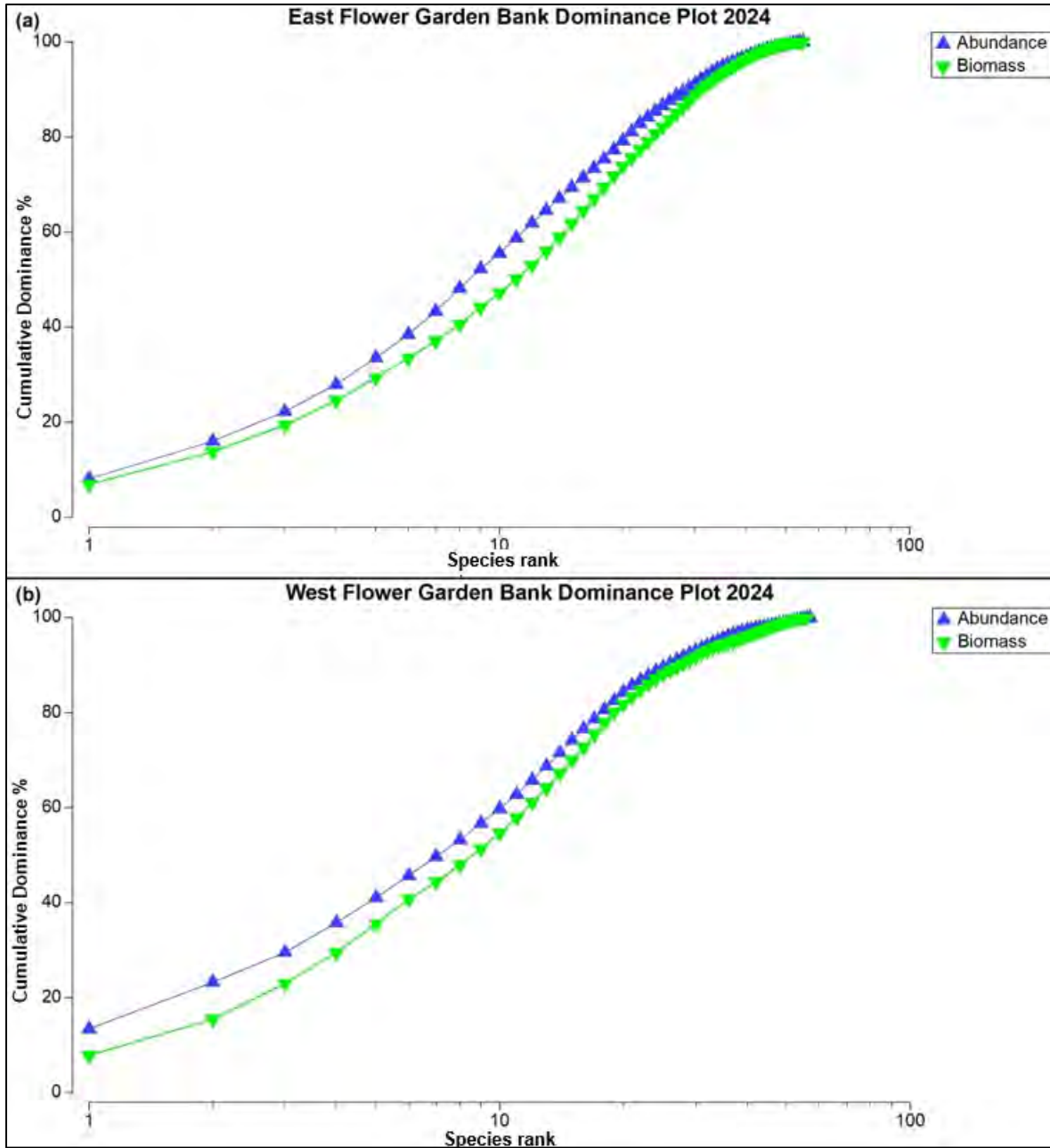


Figure 5.3. Abundance-biomass curves for (a) WFGB and (b) EFGB study sites in 2024.

Family Level Analysis

Additional analyses were conducted for grouper and snapper families due to their ecological and harvest value, as well as parrotfish and chub species due to their role as important herbivores. Analyses were also conducted for invasive lionfish (*Pterois volitans*).

In 2024, four species of grouper were observed in EFGB and WFGB surveys: graysby (*Cephalopholis cruentata*), rock hind (*Epinephelus adscensionis*), yellowmouth grouper (*Mycteroperca interstitialis*), and scamp (*Mycteroperca phenax*). The CV% for graysby and rock hind densities (20.80%) had relatively good power to detect population change while the

CV% for the density of the other two species and the biomass for all four grouper species had poor power to detect population differences (i.e., CV% > 20%). Grouper mean biomass (kg \pm SE/100 m²) was 0.074 \pm 0.033 in EFGB surveys and 0.052 \pm 0.037 in WFGB surveys. Mean biomass of small-bodied grouper (graysby and rock hind) was 0.045 \pm 0.013 in EFGB surveys and 0.052 \pm 0.037 in WFGB surveys. Mean biomass of large-bodied grouper (scamp and yellowmouth grouper) was 0.049 \pm 0.031 in EFGB surveys with no sightings in WFGB surveys. Size distributions of observed grouper in 2024 varied by species (Figure 5.4). No significant difference in the small-bodied grouper community density was detected between study sites in 2024 with a density (individuals \pm SE/100 m²) of 0.39 \pm 0.088 at EFGB and 0.17 \pm 0.072 at WFGB.

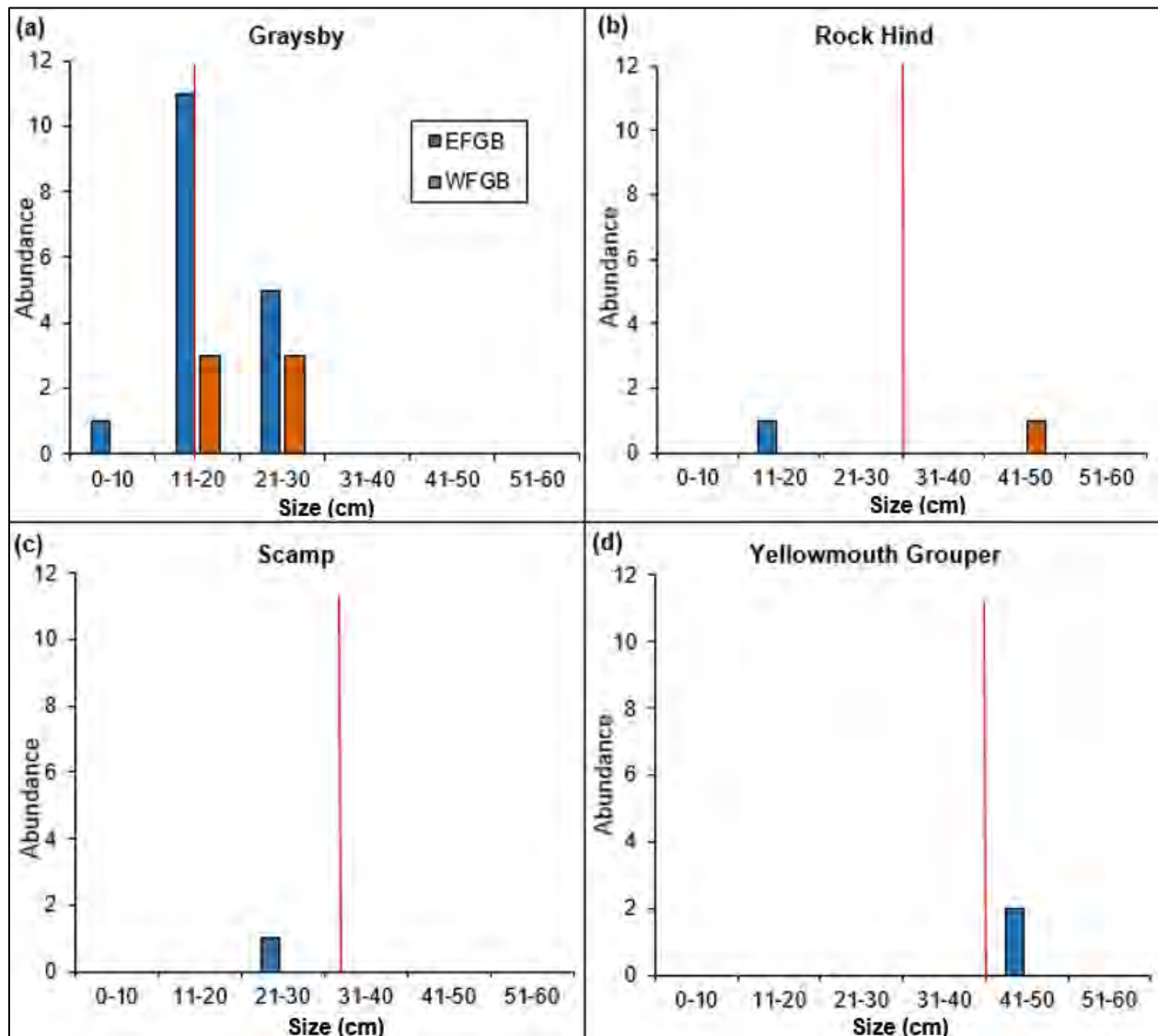


Figure 5.4. Size frequency of grouper species at EFGB and WFGB study sites in 2024: (a) graysby, (b) rock hind, (c) scamp, and (d) yellowmouth grouper. Vertical solid red lines represent estimated size of female maturity (Froese & Pauly, 2019).

Three snapper species were observed in 2024 surveys: dog snapper (*Lutjanus jocu*), gray snapper (*Lutjanus griseus*), and red snapper (*Lutjanus campechanus*; Figure 5.5). Coefficients of variation for density and biomass for all three species indicated that the data had poor power to detect population differences due to the low number of snappers observed. Mean snapper biomass ($\text{kg} \pm \text{SE}/100 \text{ m}^2$) was 0.194 ± 0.137 in EFGB surveys and 0.503 ± 0.227 in WFGB surveys. No statistical tests were run on the snapper community due to poor statistical power.

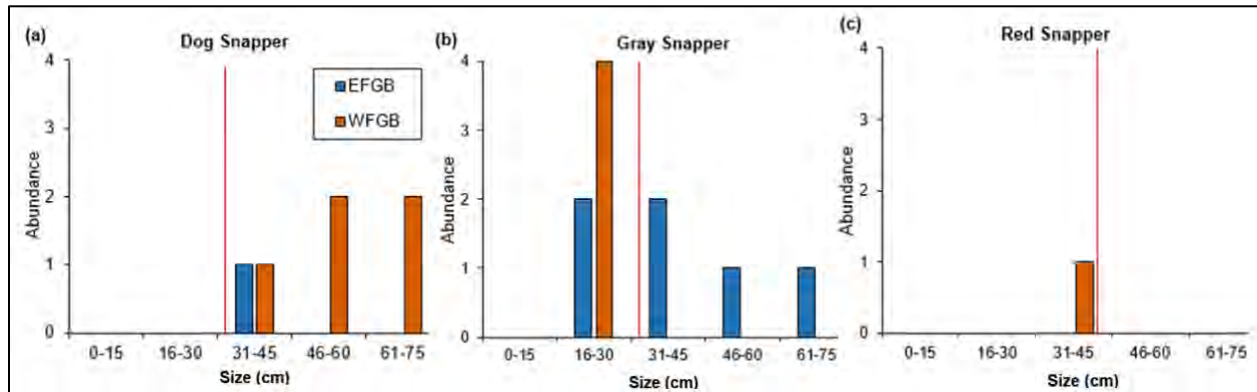


Figure 5.5. Size frequency of snapper species observed at EFGB and WFGB study sites in 2024 (a) dog snapper, (b) gray snapper, and (c) red snapper. Vertical solid red lines represent estimated size of female maturity (Froese & Pauly, 2019).

Parrotfishes are important grazers on coral reefs (Jackson et al., 2014). Seven parrotfish species were observed in EFGB and WFGB 2024 surveys: striped parrotfish (*Scarus iseri*), princess parrotfish (*Scarus taeniopterus*), queen parrotfish (*Scarus vetula*), greenblotch parrotfish (*Sparisoma atomarium*), redband parrotfish (*Sparisoma aurofrenatum*), stoplight parrotfish (*Sparisoma viride*), and bucktooth parrotfish (*Sparisoma radians*). Coefficients of variation indicated that the data had good power to detect differences in density for stoplight parrotfish (14.78%), queen parrotfish (13.82%), redband parrotfish (16.24%), and princess parrotfish (19.48%), and differences in biomass in stoplight parrotfish (18.21%) and queen parrotfish (14.29%). Mean biomass ($\text{kg} \pm \text{SE}/100 \text{ m}^2$) of parrotfishes was 0.927 ± 0.128 in EFGB surveys and 0.866 ± 0.152 in WFGB surveys. A significant difference in parrotfish densities were detected between study sites with higher densities of queen parrotfish and stoplight parrotfish at EFGB contributing to the differences. No significant differences in stoplight or queen parrotfish biomass were detected between study sites in 2024. The parrotfish population in both study sites had wide ranging size distributions (Figure 5.6).

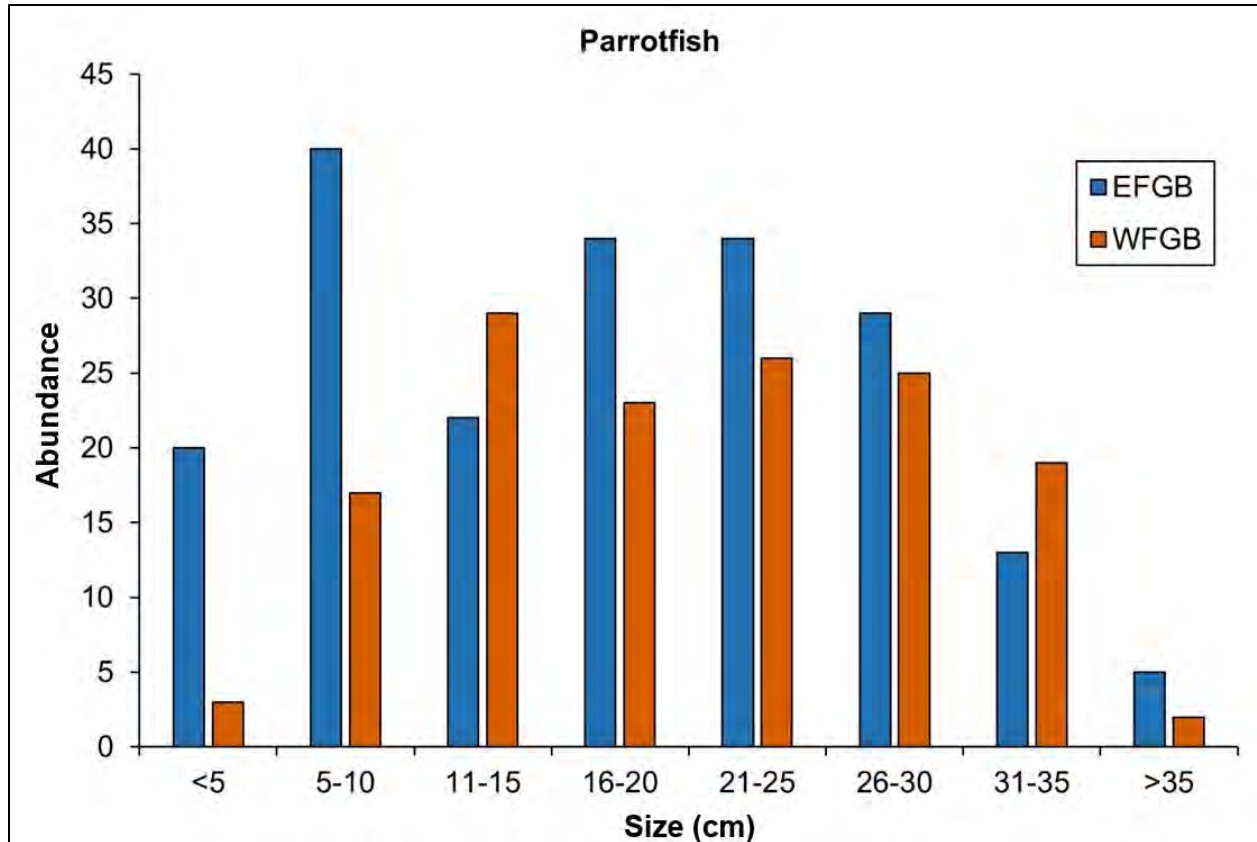


Figure 5.6. Size frequency of parrotfishes at EFGB and WFGB study sites in 2024.

Chub species are important herbivores on the reef, consuming high quantities of macroalgae per day (Dell et al., 2020). Three species of chubs are found in the sanctuary: Bermuda chub (*Kyphosus sectatrix*), brassy chub (*Kyphosus vaigiensis*), and topsail chub (*Kyphosus cinerascens*), however these three species are grouped together in a chub complex to align with past long-term monitoring efforts and reports. Coefficients of variation for density and biomass indicated that the data had poor power to detect population differences for chub species. There was a higher number of chub species observed at EFGB than at WFGB (Figure 5.7), with a mean density (individuals \pm SE/100 m²) of 3.26 ± 1.05 in EFGB surveys and 1.01 ± 0.31 in WFGB surveys. No statistical tests were run on the chub community due to poor statistical power.

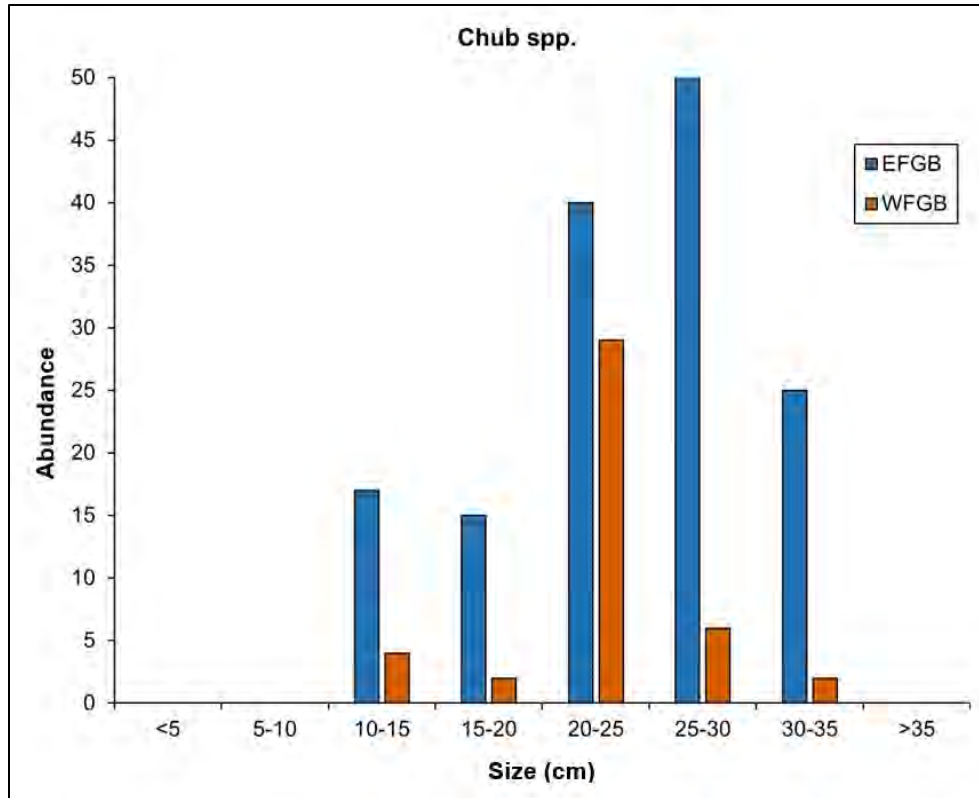


Figure 5.7. Size frequency of chub species at EFGB and WFGB study sites in 2024.

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. While lionfish were also opportunistically observed by divers during long-term monitoring field work at both banks, none were observed in fish surveys at EFGB and eight were observed at WFGB.

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 2024. While no regal demoiselles were observed at East or West Flower Garden Banks in 2024, they were observed in fish surveys at Stetson Bank during long-term monitoring field work in June 2024 (44.4% frequency in Stetson Bank fish surveys). Large abundances were also observed at nearby oil and gas rigs and the *Kraken*, a shipwreck close to Stetson Bank, during a lionfish removal cruise from August 11–15, 2024.

Fish Survey Long-Term Trends

Since 2002, mean fish density has ranged from 52.70–564.68 individuals/100 m² at the EFGB study site and 64.80–471.87 individuals/100 m² at the WFGB study site (Figure 5.8). Fish community density was compared among years and study sites when complete survey data were available (2011–2024). 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 2011–2024 (Figure 5.8). The observed dissimilarity in study site communities based on density from 2011 to 2024 was mainly attributable to variations in blue tang (5.55%) and queen parrotfish (4.20%).

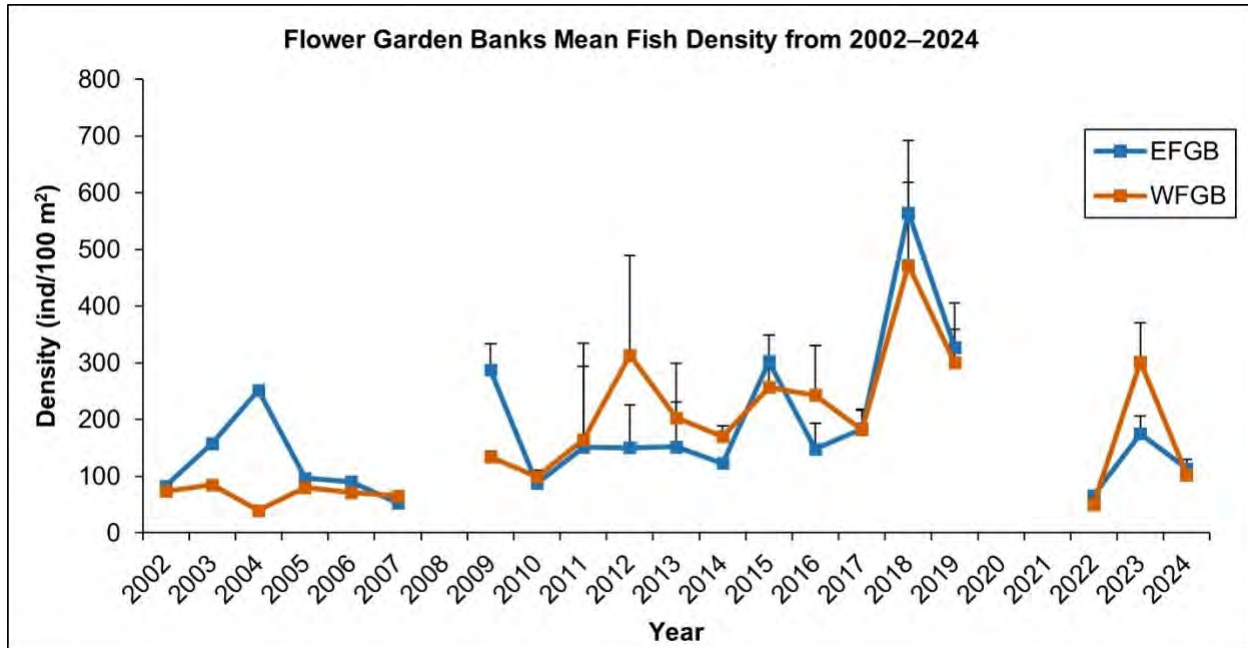


Figure 5.8. Mean fish density (individuals + SE/100 m²) at EFGB and WFGB study sites from 2002–2024. No data were collected in 2008, 2020, and 2021, and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024 (2009–2023); Eisenbach et al., 2025 (2024)

Table 5.8. PERMANOVA results comparing mean fish density between EFGB and WFGB study sites and among years from 2011 to 2024. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	14254	1	6.9547	0.0001
Year	1.4975E+05	11	6.6424	0.0001
Bank*Year	58987	11	2.6165	0.0001
Res	1.0944E+06	534		
Total	1.3186E+06	557		

Community biomass data, first collected in 2006, was highly variable in the study sites and ranged from 4.55–60.16 kg/100 m² in EFGB surveys and 2.46–27.23 kg/100 m² in WFGB surveys from 2006–2024 (Figure 5.9). 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 2011–2024 was mainly attributable to brown chromis (8.12%) and blue tang (5.96%). 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).

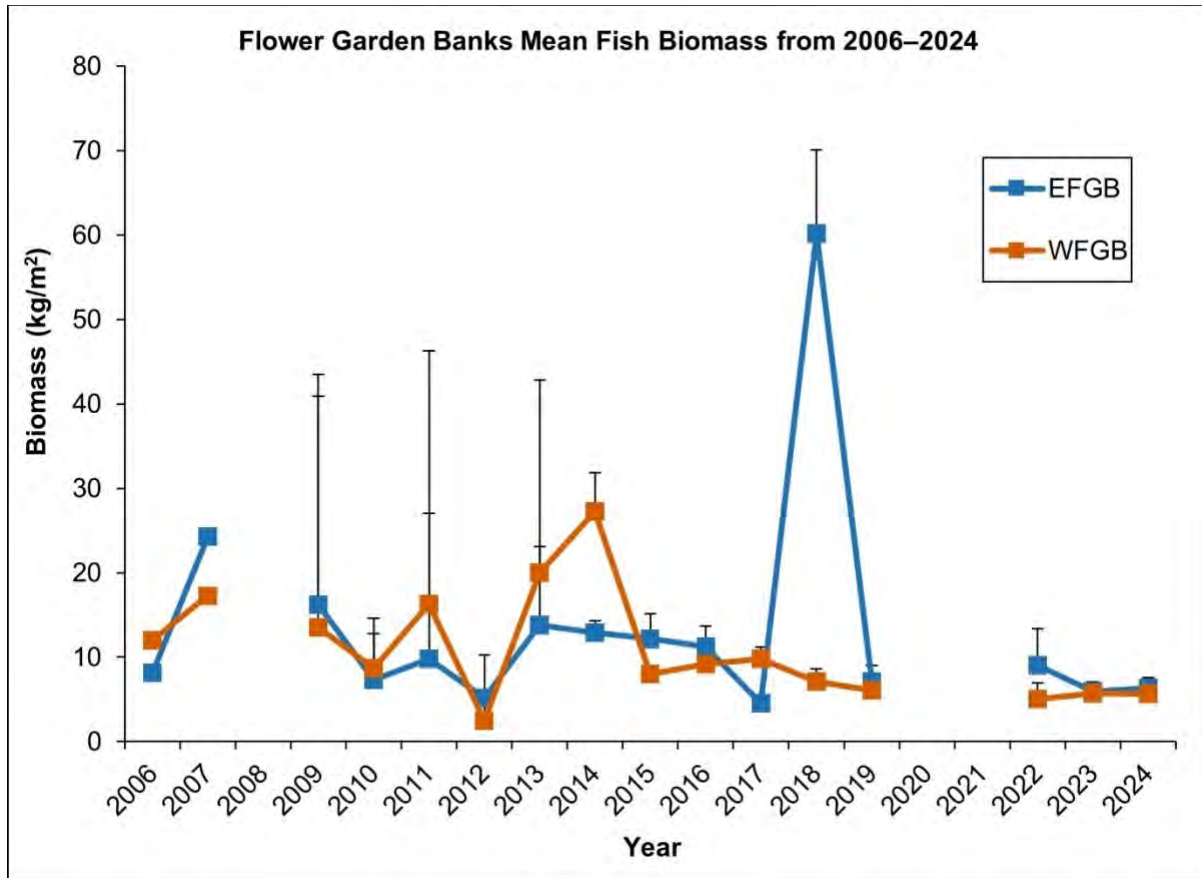


Figure 5.9. Mean fish biomass (kg/100 m²) + SE in EFGB and WFGB study sites from 2006 to 2024. No data were collected in 2008, 2020, and 2021 and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024 (2009–2023); Eisenbach et al., 2025 (2024).

Table 5.9. PERMANOVA results using mean fish biomass in EFGB and WFGB study sites and among years from 2011 to 2024. **Bold** text denotes significant values.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	8288.1	1	2.7103	0.0001
Year	4.4364E+05	11	13.188	0.0001
Bank*Year	64628	11	1.9213	0.0001
Res	1.633E+06	534		
Total	2.1511E+06	557		

Additional analyses were conducted to investigate temporal trends in grouper and snapper density from 2011 to 2024, 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 5.10; Figure 5.11).

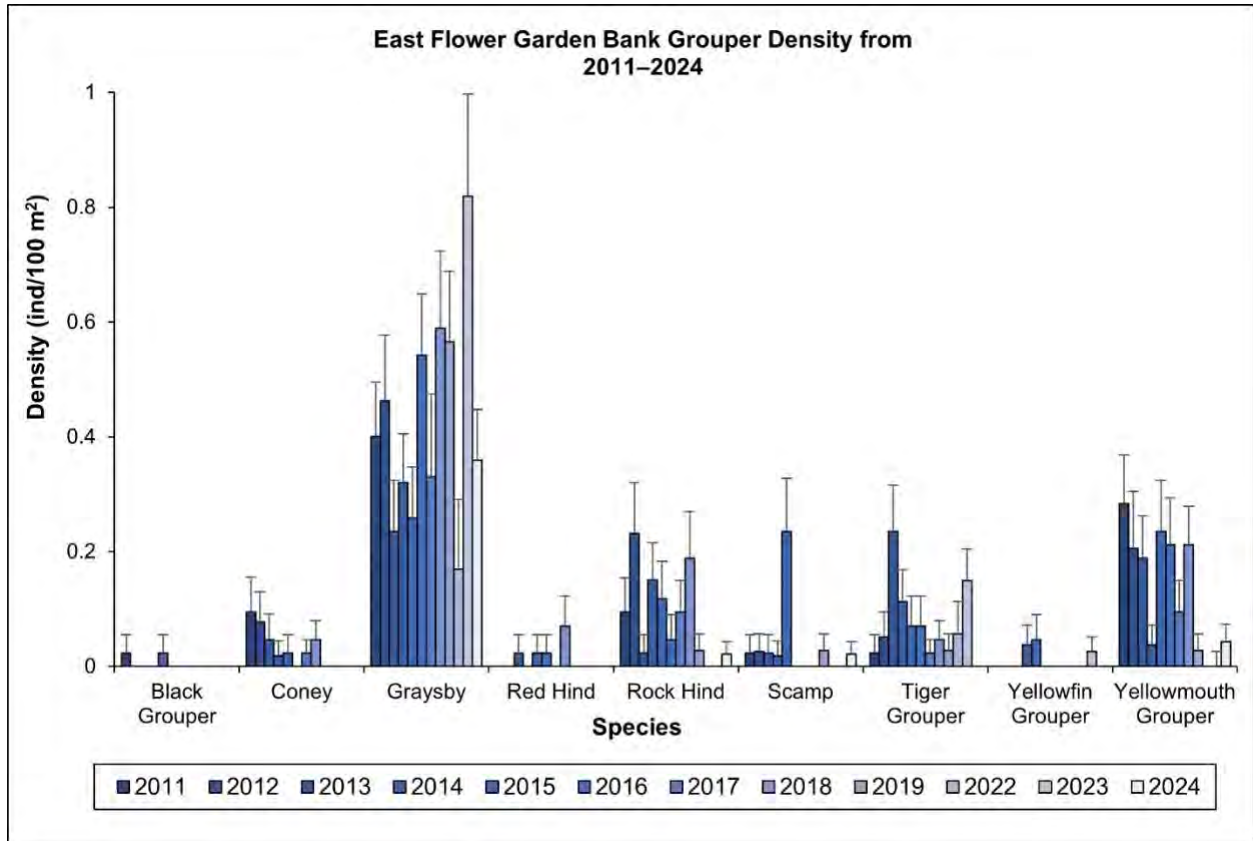


Figure 5.10. Mean density (individuals + SE/100 m²) of grouper species within EFGB study site surveys from 2011 to 2024. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025

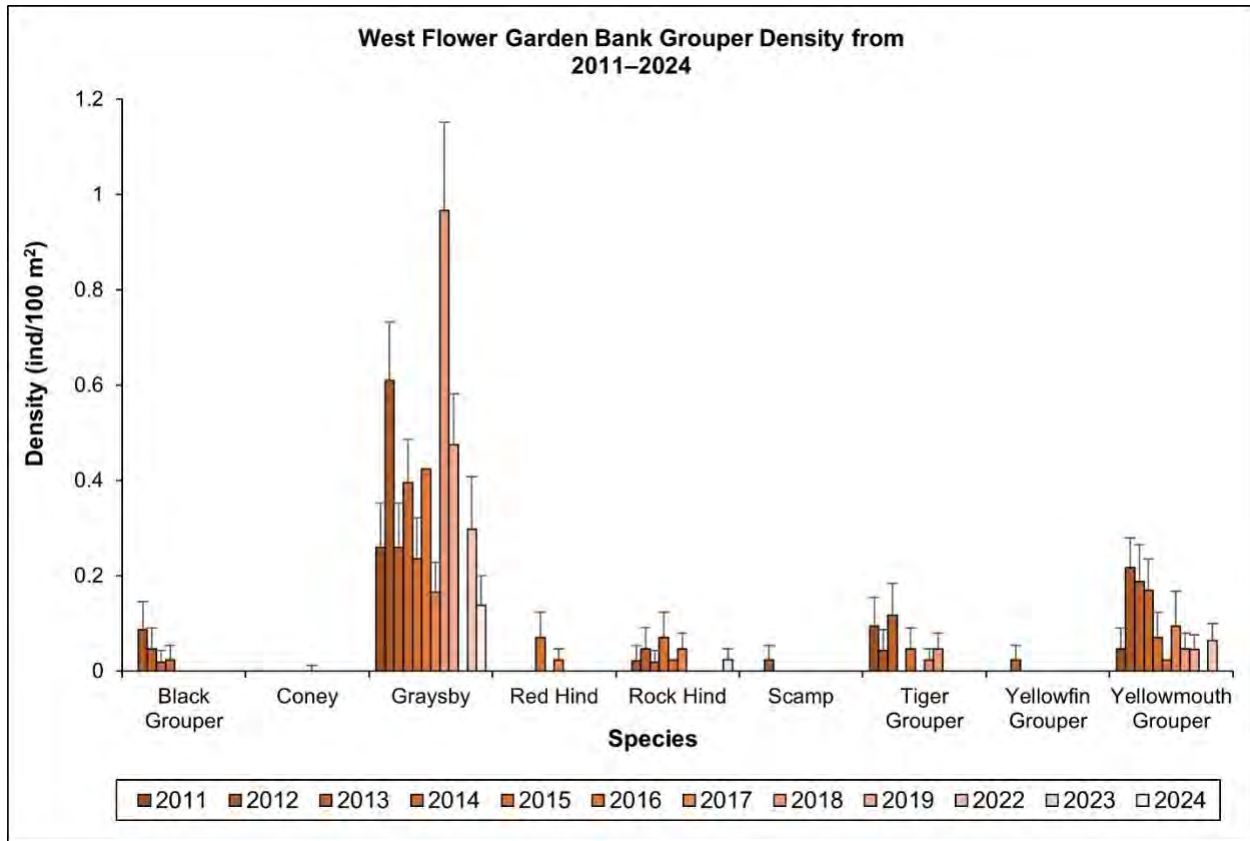


Figure 5.11. Mean density (individuals + SE/100 m²) of grouper species within WFGB study site surveys from 2011 to 2024. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025

For trend analysis, groupers were separated into large and small bodied 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 ($\tau = -0.03$, $p = 0.95$) or WFGB ($\tau = -0.18$, $p = 0.45$) from 2011–2024; however, there was a significant decline in large grouper density at both EFGB ($\tau = -0.60$, $p = 0.009$) and WFGB ($\tau = -0.68$, $p = 0.003$) from 2011–2024.

From 2011 to 2024, dog snapper were consistently denser in WFGB surveys and gray snapper were denser in most years (Figure 5.12; Figure 5.13). Mann-Kendall trend analysis revealed no significant trend in densities of either species at EFGB or WFGB from 2011–2024, suggesting stable snapper densities over the past 12 years. While a red snapper was observed in a survey in 2024, due to the rare occurrence of this species in surveys, red snapper were not included in long-term trend analysis.

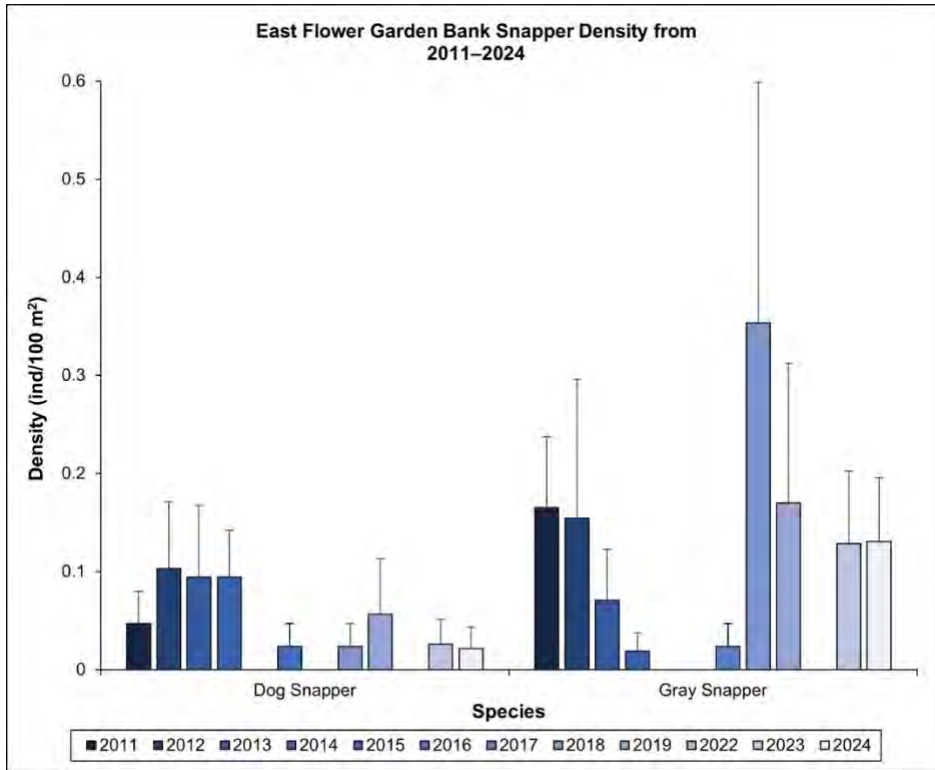


Figure 5.12. Mean density (individuals + SE/100 m²) of snapper species within EFGB study site from 2011 to 2024. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025.

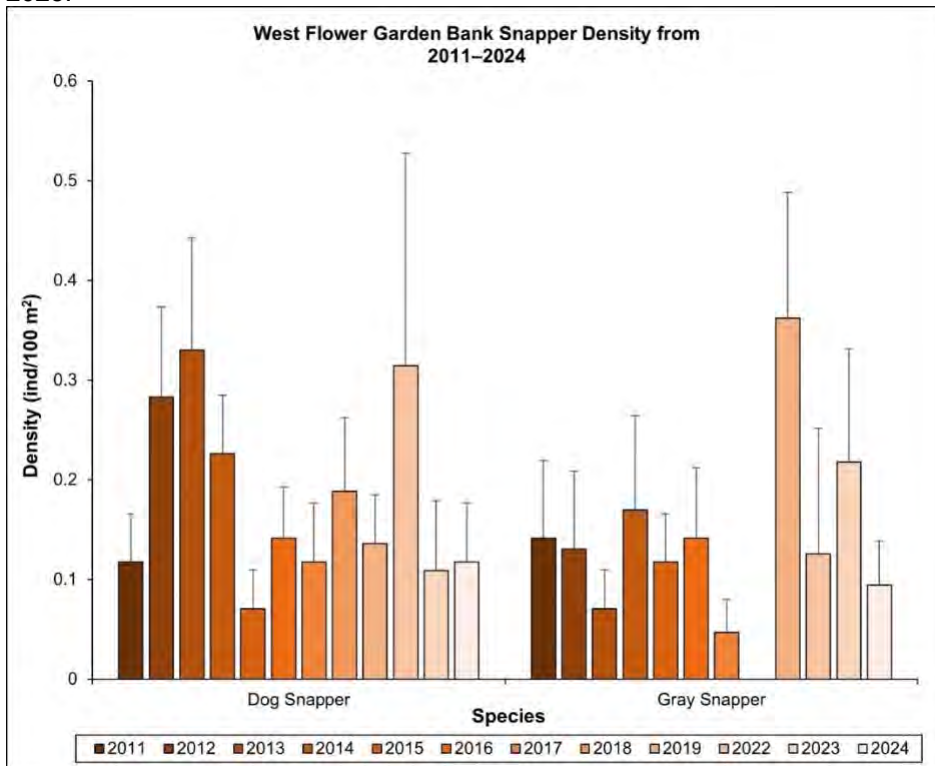


Figure 5.13. Mean density (individuals + SE/100 m²) of snapper species within WFGB study site from 2011 to 2024. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025.

Discussion

Fish communities are indicators of ecosystem health (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, 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 2024, a combined total of 68 species were observed in 50 fish surveys in the study sites on both banks. In 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 2024 study period, the herbivore guild possessed the highest mean biomass within EFGB study site surveys, contributing to 41% of the total biomass. At WFGB, piscivores possessed the highest mean biomass contributing to 43% of the total biomass within those study site surveys. This difference may be attributed to the higher macroalgae percent cover at EFGB with 24% cover in 2024 compared to the 17% macroalgae cover at WFGB in 2024. Within the herbivore guild at EFGB, 48% of the total herbivore biomass was accounted for by chub species, followed by stoplight parrotfish (18%). Historically, both banks were dominated by the piscivore trophic guild. The shift to an herbivore dominated biomass was first documented in 2023 at both EFGB and WFGB, but in 2024, WFGB returned to a piscivore trophic guild dominance. Herbivores and piscivores are both important for ecosystem health; piscivores regulate fish community, while herbivores manage macroalgae cover and promote coral growth (Burkpile & Hay, 2008; Hughes et al., 2010; Hamilton et al., 2014). This shift to higher biomass of herbivore species develops a bottom-up trophic structure causing an ecosystem shift in fish community (Hempson et al., 2018). The greater biomass of herbivores may be driven by the high macroalgae cover at EFGB.

Planktivores represented the second highest mean biomass, comprising approximately 32% of the total biomass within the study sites. Within this guild, Atlantic creolefish contributed to 71% of the total biomass, followed by creole wrasse at 28%. No giant mantas (*Mobula birostris*) were observed in surveys in 2024.

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, this situation 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.

Commercially and recreationally important grouper and snapper density was low (<1 individual/100 m²) at EFGB and WFGB study sites in 2024. The grouper species observed consisted of both juvenile and mature graysbys and rock hind, only juvenile scamp, and only mature yellowmouth grouper. A significant decrease in density of large-bodied groupers from 2011 to 2024 has been observed. This decrease may be due to increased fishing pressure, or the cryptic nature of many of these species which typically shy away from divers. The dog snappers overserved were all mature individuals, while the one red snapper was juvenile, and a mix of juvenile and mature individuals of gray snapper were seen during surveys in 2024. It should be noted that typical recruitment/nursery habitat for snappers (mangroves and seagrasses) are not present at EFGB and WFGB, and the mechanism for recruitment of this family to the area is not well understood (Mumby et al., 2004; Clark et al., 2014). Due to the biogeographic isolation of EFGB and WFGB, the fish assemblage is thought to rely on self-recruitment, as planktonic larval duration can limit larval supply and dispersal from other reefs in the southern Gulf to EFGB and WFGB; complicating the process is the dynamic nature of oceanographic conditions (i.e., variability in the Loop Current and associated eddies; Wetmore 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 at FGBNMS was 0.898 kg per 100 m² in 2024, which is considered low however, this value is not significantly different from several other Caribbean reefs, including sites in Mexico (1.592 kg/100 m²), Belize (1.500 kg/100 m²), and Honduras (2.423 kg/100 m²; AGRRA, 2024). Although low parrotfish biomass is often associated with high fishing pressure or low coral cover, neither factor has been documented at EFGB or WFGB. Given the availability of food resources and limited fishing activity at both banks, the relatively low parrotfish abundance remains unexplained.

Chub species have been considered important herbivores on reefs worldwide (Knudsen & Clements, 2016). In the Caribbean, they have been observed eating large quantities of macroalgae, specifically *Dictyota* (Dell et al., 2020). Normally seen in schools in the water column, particularly beneath the vessel, chubs are frequently seen at FGBNMS and make up the majority of the herbivore trophic guild biomass. *Dictyota* is one of the most abundant

macroalgae found in FGBNMS which may explain their high abundance, particularly at EFGB where chub biomass and macroalgae cover are higher than at WFGB.

Lionfish abundance estimates presented here are based on daytime surveys; however, lionfish are commonly most active during crepuscular periods at dawn and dusk, which may lead to underestimation of true densities. Despite this limitation, this report supports visual and quantitative assessments at EFGB and WFGB of low densities, ranging from approximately 27 to 105 individuals per hectare (Blakeway et al., 2022). These values are lower than other regions in the Western Atlantic such as North Carolina (150 individuals ha^{-1} ; Morris & Whitfield, 2009) and the Bahamas (100 to 390 individuals ha^{-1} ; Green & Côté, 2009; Darling et al., 2011). Artificial reefs in the northern Gulf have reported lionfish densities similar to those at FGBNMS (10 to 100 individuals ha^{-1} ; Dahl & Patterson, 2014). Lionfish have been consistently observed at FGBNMS since 2011. Following their arrival, rapid increases in abundance were reported through 2014, followed by a decline after 2015 (Johnston et al., 2016b), a resurgence in 2018, and a sharp drop in 2019 (Johnston et al., 2021b). Lionfish were absent in 2022 surveys, and only two individuals were observed in 2023. In 2024, eight lionfish were recorded, all at WFGB. This high variability may reflect boom-and-bust population cycles driven by episodic recruitment events, potentially limited by the banks' relative isolation, or influenced by other controls such as local removal efforts, which began in 2011. An ulcerative skin disease affecting lionfish, first reported in 2017 across Florida and the Caribbean (Harris et al., 2018), was also observed at FGBNMS and coincided with regional density declines in the northeast Gulf (Harris et al., 2020). Although the etiology of the disease remains unknown (Cody et al., 2023), it may have contributed to the reduced lionfish densities observed in 2017 and 2019 at EFGB and WFGB. Alternatively, lionfish populations may be oscillating around the banks' carrying capacities, as reported elsewhere in the northeast Gulf (Harris et al., 2023).

Culling efforts may influence lionfish populations both through direct removal and by reducing reproductive output and recruitment potential. Since 2015, permitted lionfish removal cruises have been conducted during summer months aboard the recreational dive vessel M/V *Fling* to help suppress lionfish predation on native fish communities. These cruises typically result in the removal of several hundred lionfish, but dives are limited to the upper reef crest (<40 m) and focus on areas surrounding mooring buoys at recreational dive sites (Green et al., 2014). As a result, removals do not occur across large areas of the reef and are not conducted within designated monitoring sites. In 2024, no removal cruise occurred at EFGB or WFGB before monitoring occurred, but lionfish were removed from nearby oil and gas platforms and the artificial reef *Kraken*. The higher number of lionfish observed in 2024 surveys may reflect the lack of targeted removals within the sanctuary that year. During the July 14–17, 2024 long-term monitoring cruise, 36 lionfish were opportunistically removed by partner divers at EFGB and WFGB.

The regal demoiselle, a non-native species from the Indo-Pacific, was likely introduced into the Gulf via inter-ocean transfer on oil platforms (Robertson et al., 2018). While this species is becoming more common at other locations in the sanctuary (O'Connell et al., 2024), it has not yet become abundant at EFGB and WFGB. There is concern that regal demoiselles could compete with and displace native reef fishes, such as the brown chromis (Robertson et al., 2016), although no ecological impacts have been confirmed to date. In nearby areas such as

Stetson Bank and surrounding oil and gas platforms, regal demoiselles are observed in high densities and frequently form mixed-species schools with native planktivores above artificial and natural structures, exhibiting similar behavior with no apparent effects (O’Connell et al., 2024). Sightings during the 2018 and 2019 fish surveys at EFGB and WFGB were reported to the United States Geological Survey’s Nonindigenous Aquatic Species database. FGBNMS will continue monitoring the distribution, abundance, and potential impacts of this species.

Chapter 6: Water Quality



Research staff deploy a handheld YSI® logger to collect water column profiles aboard the M/V *Fling*.
Photo: Taylor Galaviz/NOAA

Water Quality Introduction

Several water quality parameters were continuously or periodically recorded at EFGB and WFGB from January through December 2024. Salinity, turbidity, and temperature was recorded hourly by moored data loggers installed in or near the study sites at depths of approximately 24 m. At EFGB and WFGB, dissolved oxygen was also monitored continuously at a depth of 23 m and 27 m respectively through a partnership with University of Texas at Austin, Marine Science Institute. Additional temperature loggers installed at repetitive photostations at depths of 30 m and 40 m at each bank collected hourly readings; these were recovered on July 16, 2024 and July 31, 2025.

A Sofar Ocean® buoy, deployed at EFGB in June 2023, included both a sea surface temperature logger and a subsurface temperature logger at 23 m depth. However, the buoy broke free from its tether during Hurricane Francine in September 2024, and no further temperature data were recorded for the remainder of the year.

In lieu of the standard Sea-Bird® Electronics 19plus V2 CTD profiler, water column profiles were acquired in July using a YSI Pro Plus® data sonde. Water samples were not collected in 2024 due to cruise cancellations associated with vessel maintenance. This chapter presents data from moored water quality instruments and water column profiles collected in 2024 and 2025.

Water Quality Methods

Field Methods

Temperature and Salinity Loggers

The primary instrument used to record temperature, salinity, and turbidity at each bank 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, mounted on anchors positioned within sand flats at each study site (see Figure 1.4; Figure 1.5). The instruments recorded temperature, salinity, and turbidity on an hourly basis. Sensors were exchanged by divers in July 2024 and July 2025 to allow for data download and maintenance. Identical instruments were deployed during recovery dives to avoid interruptions in data collection. After recovery, data were reviewed, sensors were cleaned and tested, and battery levels were verified. Additionally, data that was collected on an hourly basis was calculated into a single daily average, which was then compared across all sensors to ensure data integrity. In July 2024, one SBE 16plus instrument deployed at EFGB was outfitted with an additional SBE 63 Optical Dissolved Oxygen Sensor.

Onset® Computer Corporation HOBO® Pro v2 U22-001 (HOBO) thermograph loggers were also used to collect hourly temperature readings. These loggers, mounted alongside SBE 16plus instruments at EFGB and WFGB, served as backup sensors. Additional HOBO loggers were deployed at 30 m and 40 m depths at each bank, attached to permanent repetitive photostation markers. The loggers were recovered in July 2025.

A Sofar Ocean Technologies® Spotter Buoy and Smart Mooring system deployed at EFGB in 2023 remained in service through September 11, 2024. The system recorded surface conditions

and subsurface temperature at 23 m depth, transmitting data hourly via satellite to a dashboard interface. The Sofar 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 data from January 1 to September 11, 2024, were available in near real time and processed into daily averages. On September 11, the buoy broke free from its tether during Hurricane Francine, and no additional data were collected for the remainder of the year while the instrument was recovered and repaired.

Water Column Profiles

In 2024, all water column profiles were collected using a handheld YSI® Pro Plus sonde, which measures temperature, dissolved oxygen, pH, and salinity. This instrument was used in place of the SBE 19plus V2 CTD, which typically provides a full suite of profile data, including turbidity and fluorescence, but was unavailable due to vessel maintenance. The YSI sonde was manually lowered from the surface to the reef cap and returned to the surface at a consistent rate, and profiles were collected during the July 2024 cruise.

Water Samples

No water samples were collected in 2024 due to a prolonged vessel maintenance schedule that limited offshore operations. Under normal conditions, water samples are collected in conjunction with CTD water column profiles, using a carousel system equipped with 12 four-liter Niskin bottles. Samples are typically collected at three depths: near the surface (~1 m), midwater (~10 m), and near the reef cap (~20 m). A blind duplicate sample is also collected during each sampling event for quality control. Collected water samples are analyzed for chlorophyll *a* (chl *a*) and nutrients, including ammonia, nitrate, nitrite, soluble reactive phosphorus (ortho phosphate), and total Kjeldahl nitrogen (TKN). Normal sampling operations are expected to resume in 2025 following the return of the R/V *Manta* from shipyard maintenance.

Water Quality Data Processing and Analysis

Temperature, salinity, dissolved oxygen, and turbidity data recorded by SBE 16plus instruments, along with backup temperature data from HOBO loggers, were downloaded and processed in July, 2024 and July, 2025. Sofar Ocean® Spotter Buoy data were processed monthly from January until the buoy detached in September. Standard QA/QC procedures were followed, including file review, instrument servicing, and battery checks in accordance with 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.

To address data gaps, satellite-derived sea surface temperature (SST) and sea surface salinity (SSS) were acquired from NOAA's Environmental Research Division Data Access Program (ERDDAP) server. SST data were obtained from the GHRSSST Level 4 MUR Global Foundation Sea Surface Temperature Analysis (v4.1), while SSS data were sourced from the MIRAS SMOS 3-Day Mean product (JPL MUR MEaSURES Project, 2025; JPL SMOS Project, 2025; NOAA Coral Reef Watch, 2025). SST data were available as daily means on a 0.01° global grid, and SSS data as 3-day means on a 0.25° grid. To reduce reliance on satellite-derived surface data and

improve in situ coverage, FGBNMS plans to install additional Sofar Ocean® buoys within the sanctuary in 2025. Daily dissolved oxygen values were calculated based on daily temperature and salinity values using oceanographic formulae (Garcia et al., 1992).

Daily mean seawater temperature, salinity, and turbidity from EFGB and WFGB were compared using paired t-tests in R (version 4.2.2). Long-term trends in seawater temperature and salinity were assessed using the Seasonal-Kendall trend test, which applies the Mann-Kendall test to monthly data series while accounting for serial correlation in recurring seasonal patterns (Hipel & McLeod, 1994; Helsel & Hirsch, 2002; Helsel et al., 2006).

Water Quality Results

Temperature

Satellite SST at EFGB ranged from 21.65 °C to 31.29 °C in 2024 and Sofar Ocean® surface sensor at EFGB ranged from 20.31 °C to 31.83 °C (January–September) in 2024. At 23 m, temperatures ranged from 19.33 °C to 30.95 °C (Figure 6.1). The 23-m backup HOBO logger and Sofar Ocean® sensor recorded similar values, while the SBE 16plus showed slight variation, with a maximum difference of 1.10 °C between the instruments (Figure 6.1). The HOBO loggers at 30-m and 40-m recorded ranges from 19.81 °C to 31.06 °C and 19.77 °C to 30.90 °C respectively.

At WFGB, surface temperatures in 2024 ranged from 21.65 °C to 31.29 °C. At 27 m, temperatures ranged from 19.60 °C to 31.02 °C (Figure 6.1). The 27-m backup HOBO logger and SBE 16plus registered similar values (Figure 6.1). The HOBO loggers at 30-m and 40-m recorded ranges from 19.64 °C to 31.00 °C and 19.57 °C to 30.56 °C respectively.

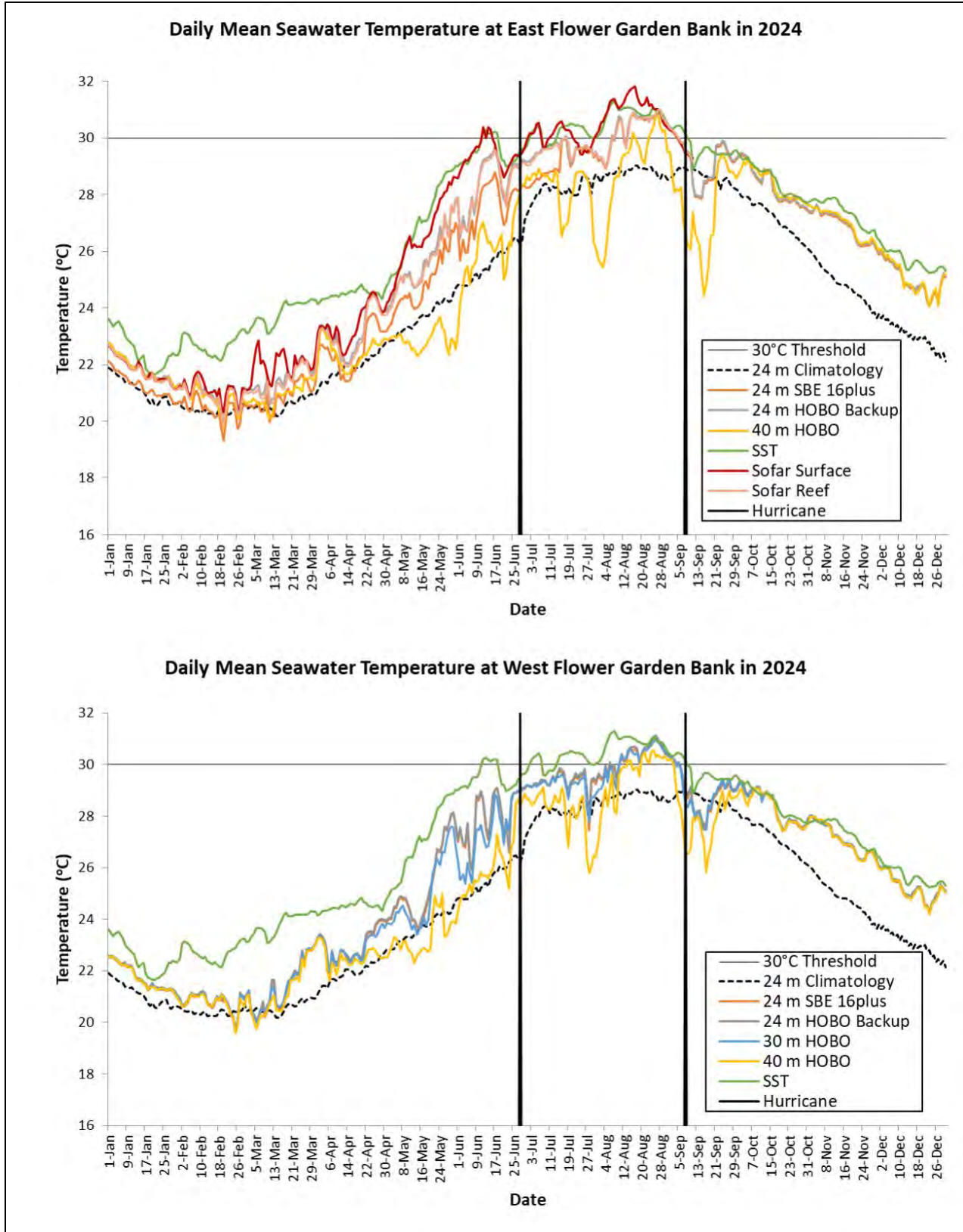


Figure 6.1. Daily mean seawater temperature (°C) at (a) EFGB and (b) WFGB from various depths in 2024, as well as the 25-year daily mean water temperature baseline. The solid black line at 30 °C is a level known to trigger coral bleaching.

Seawater temperature data obtained from loggers at EFGB (23 m) and WFGB (27 m) have been collected since 1990. While some data gaps occurred due to equipment malfunctions and changes in methods and instrumentation, long-term trends indicate increasing surface and reef cap temperatures at both banks (Figure 6.2). Seasonal-Kendall trend tests were applied to two datasets: (1) time series sea surface temperatures derived from satellite observations (OISST V2.1, AVHRR-only, daily $1/4^\circ$; NOAA NCEI, 2023), and (2) daily mean seawater temperatures recorded at reef crest depth by in situ instruments. Results revealed significantly increasing monotonic trends from 1990 to 2024 in surface waters at EFGB ($\tau = 0.35$, $z = 10.59$, $p < 0.001$) and WFGB ($\tau = 0.34$, $z = 10.17$, $p < 0.001$), as well as at depth from datasondes at EFGB (23 m; $\tau = 0.33$, $z = 7.81$, $p < 0.001$) and WFGB (27 m; $\tau = 0.32$, $z = 8.08$, $p < 0.0002$), after adjusting for seasonal correlation (Figure 6.2).

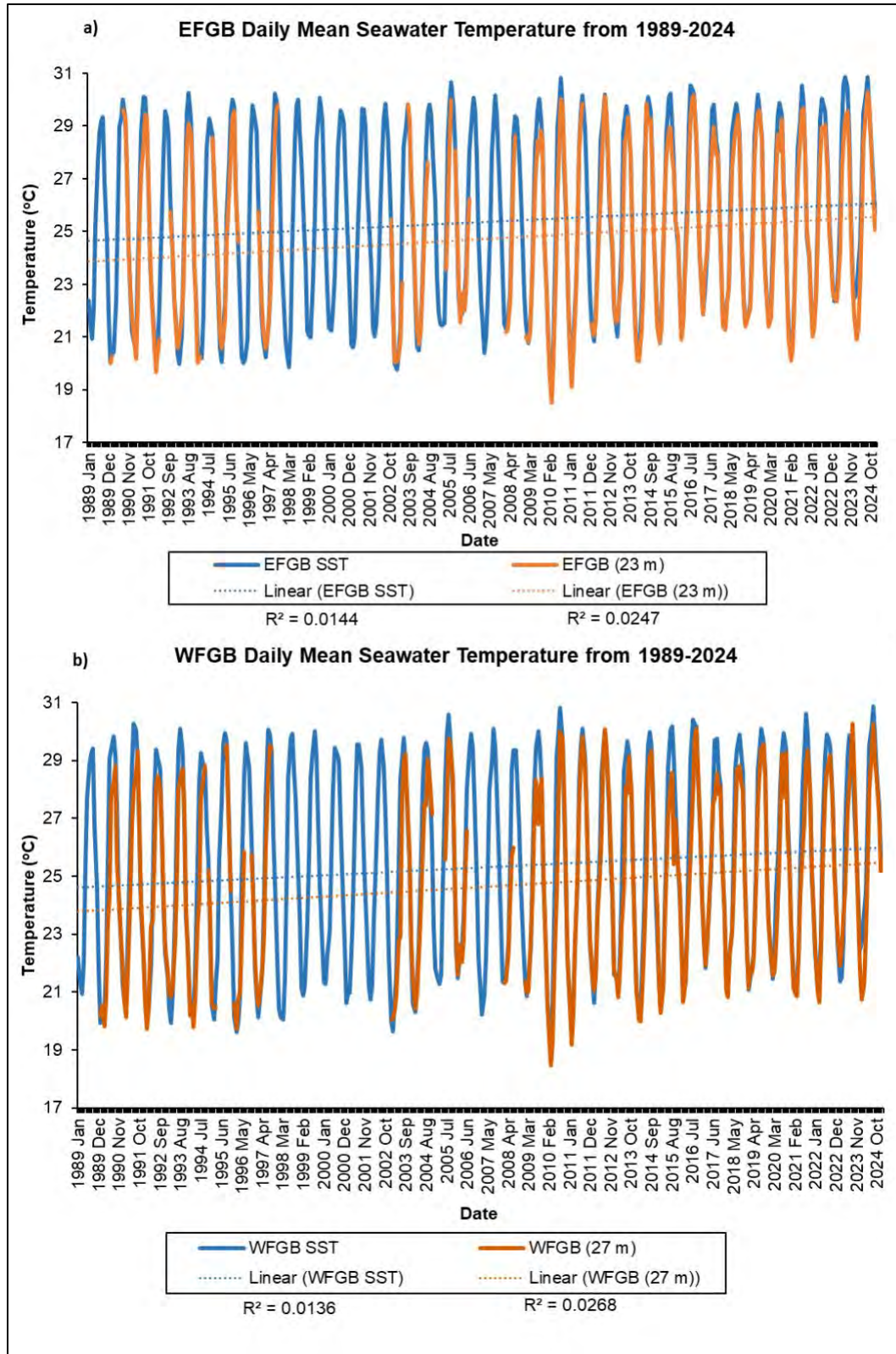


Figure 6.2. Daily mean seawater temperature (°C) demonstrates 12-month seasonal variation from various depths at (a) EFGB and (b) WFGB from 1989 to 2024, as well as a significant increase over time (trend lines).

As described in the bleaching section above, data revealed moderate thermal stress, with paling being slightly more prevalent than bleaching, indicating that the corals were responding to elevated temperatures with a combination of stress indicators (Figure 6.3). A long-term heatmap of daily temperature at 23 m (Figure 6.4) illustrates the seasonal cycle of warming and cooling from 2009 through early 2025, with sustained elevated temperatures in recent years, including 2023 and 2024. These data were adjusted to daily averages and compiled from all available in situ sensors at EFGB deployed at approximately 23 m depth, including Sea-Bird 16plus instruments, HOBO loggers, and the Sofar Ocean® Smart Mooring. The visual gap following September 2024 reflects when the figure was generated prior to incorporating later EFGB SBE 16plus data. These patterns are consistent with the increasing thermal stress observed at the sanctuary and align with statistically significant warming trends reported in this chapter.

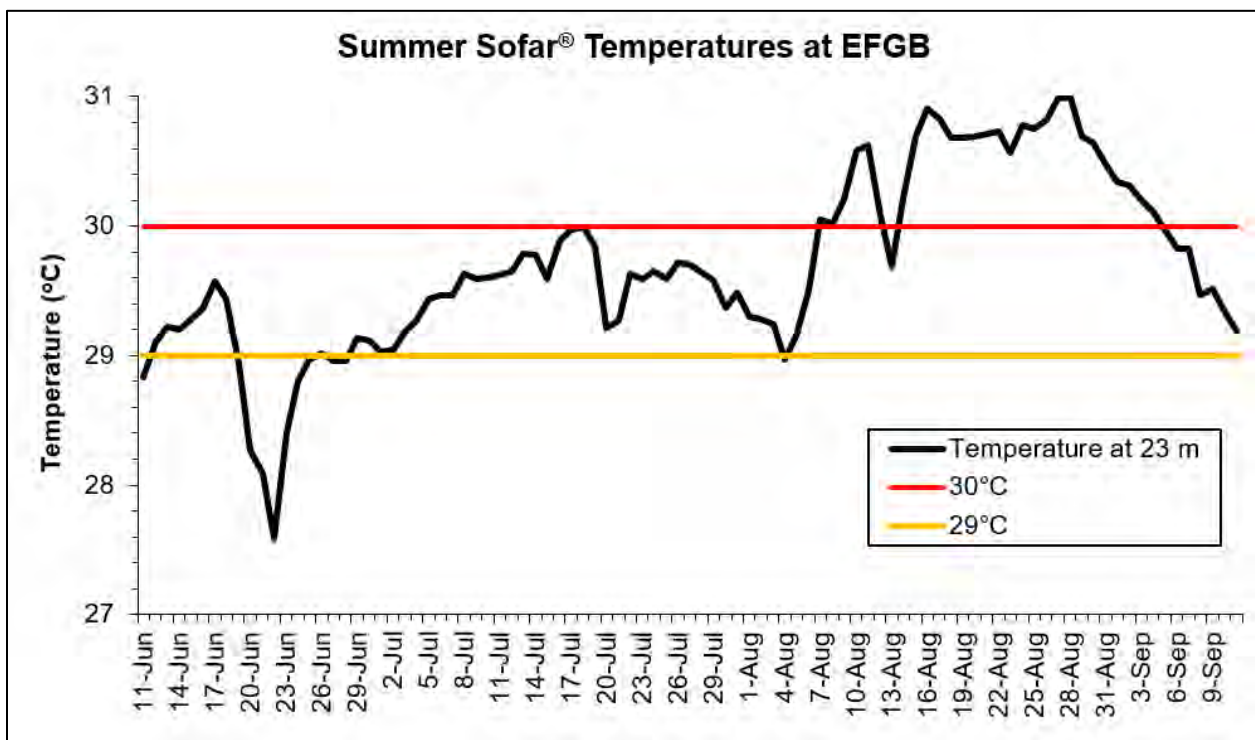


Figure 6.3. Temperature data from Sofar Ocean® Smart Mooring at EFGB at reef crest shows 82 days over 29 °C and 28 days over 30 °C.

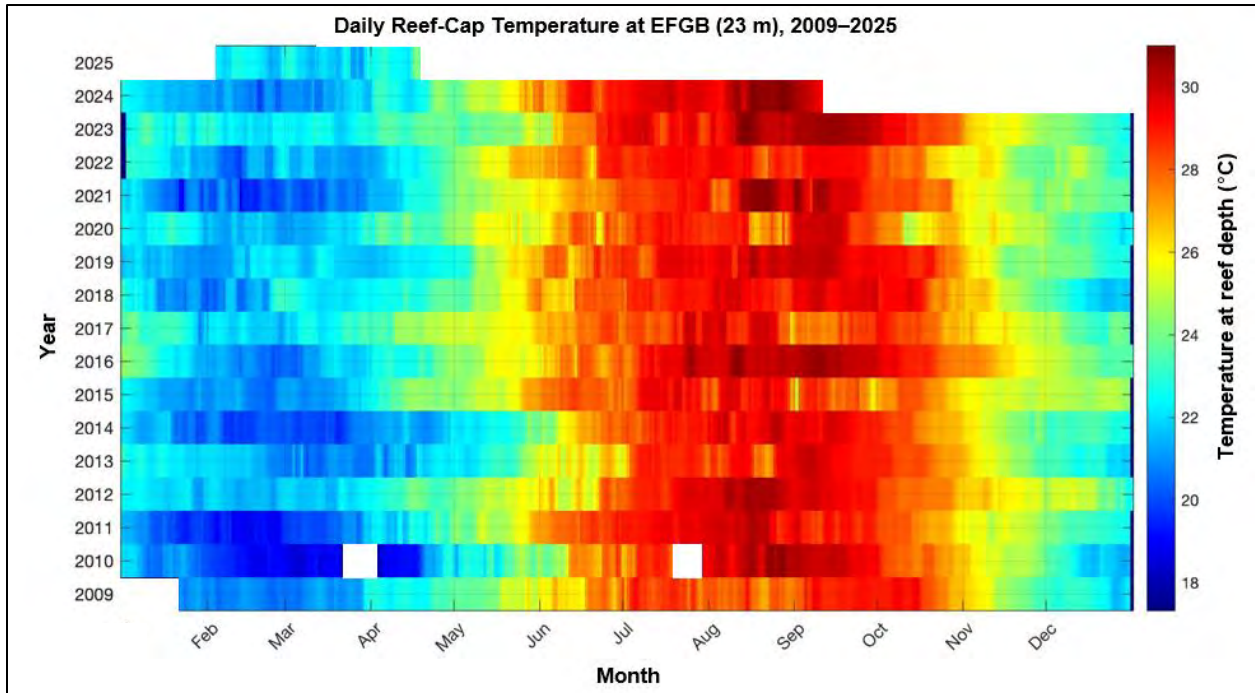


Figure 6.4. Heatmap showing daily temperature at reef depth (23 m) at EFGB from 2009 to 2025. Warmer colors indicate higher temperatures. The extended period of elevated temperatures in summer 2024 and the absence of data following the September 2024 loss of the Sofar Ocean® buoy are visible. Figure: Xingping Hu/University of Texas

Salinity

In 2024, salinity at EFGB ranged from 33.66 to 37.57 psu at the surface and 34.48 to 37.04 psu at 23 m (Figure 6.5). At WFGB, salinity ranged from 33.66 to 37.57 psu at the surface and 34.52 to 36.72 psu at 27 m (Figure 6.5). Despite no significant overall difference in 2024 daily mean salinity between EFGB 23 m and WFGB 27 m, salinity patterns show contrasting seasonal trajectories (Figure 6.5), with salinity increasing at EFGB and decreasing at WFGB from January through December 2024.

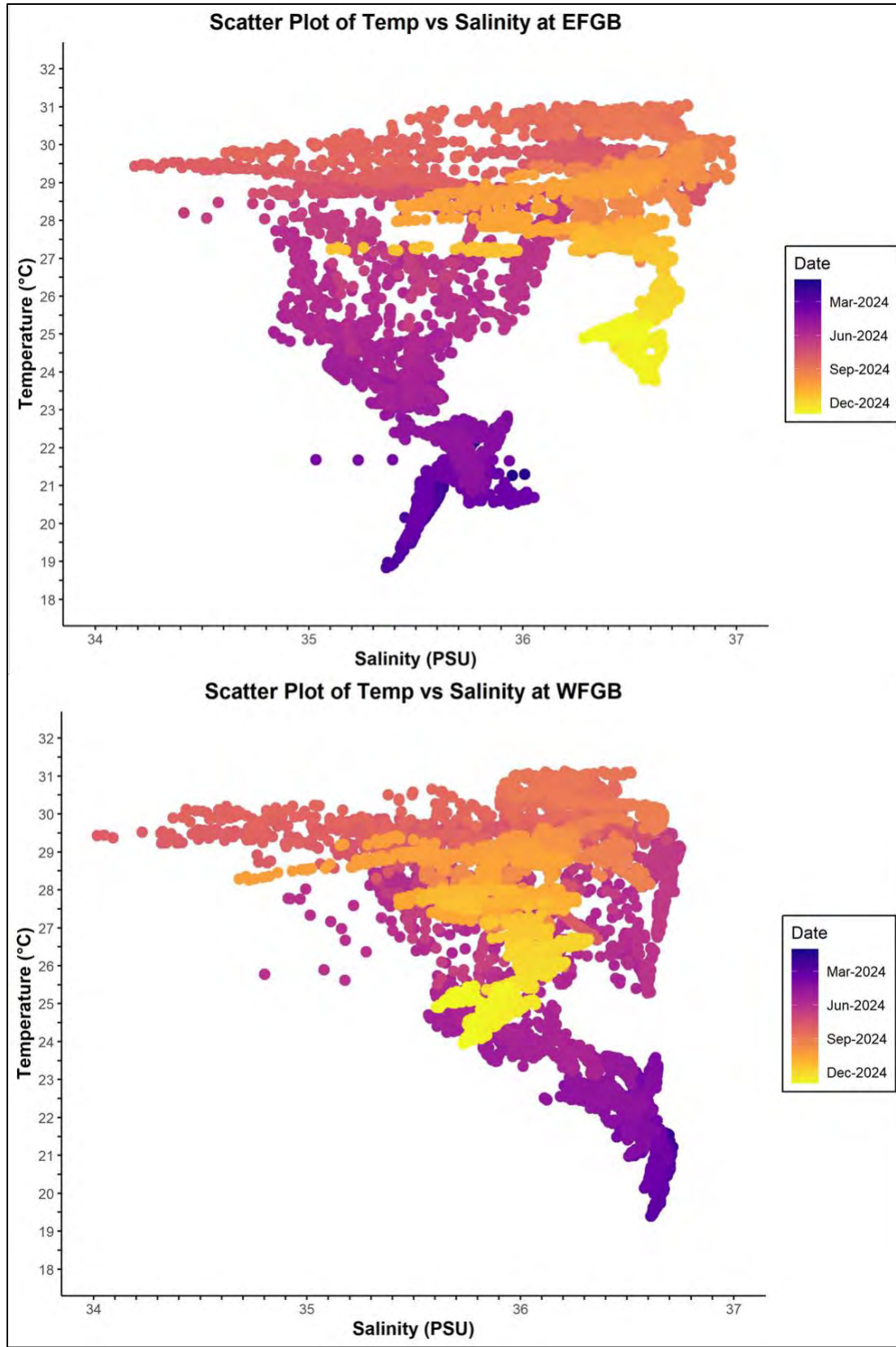


Figure 6.5. Hourly salinity (psu) against hourly temperature recorded using SBE 16plus at the reef cap station at (a) EFGB and (b) WFGB in 2024.

Salinity data obtained from loggers at EFGB (23 m) and WFGB (27 m) have been collected since 2008, with only occasional disruptions. In situ salinity at reef depth was generally more stable than satellite-derived surface salinity, which showed pronounced seasonal fluctuations, particularly in summer (Figure 6.6). These summer minima were most evident in surface records, reflecting regional freshwater riverine influence, and were muted at depth. Seasonal-Kendall trend tests applied to daily mean salinity from 2008 to 2024 indicated weak negative slopes at both EFGB ($\tau = -0.21$, $z = -3.88$, $p = 0.07$) and WFGB ($\tau = -0.18$, $z = -3.50$, $p = 0.11$), but neither trend was statistically significant after adjusting for seasonal correlation.

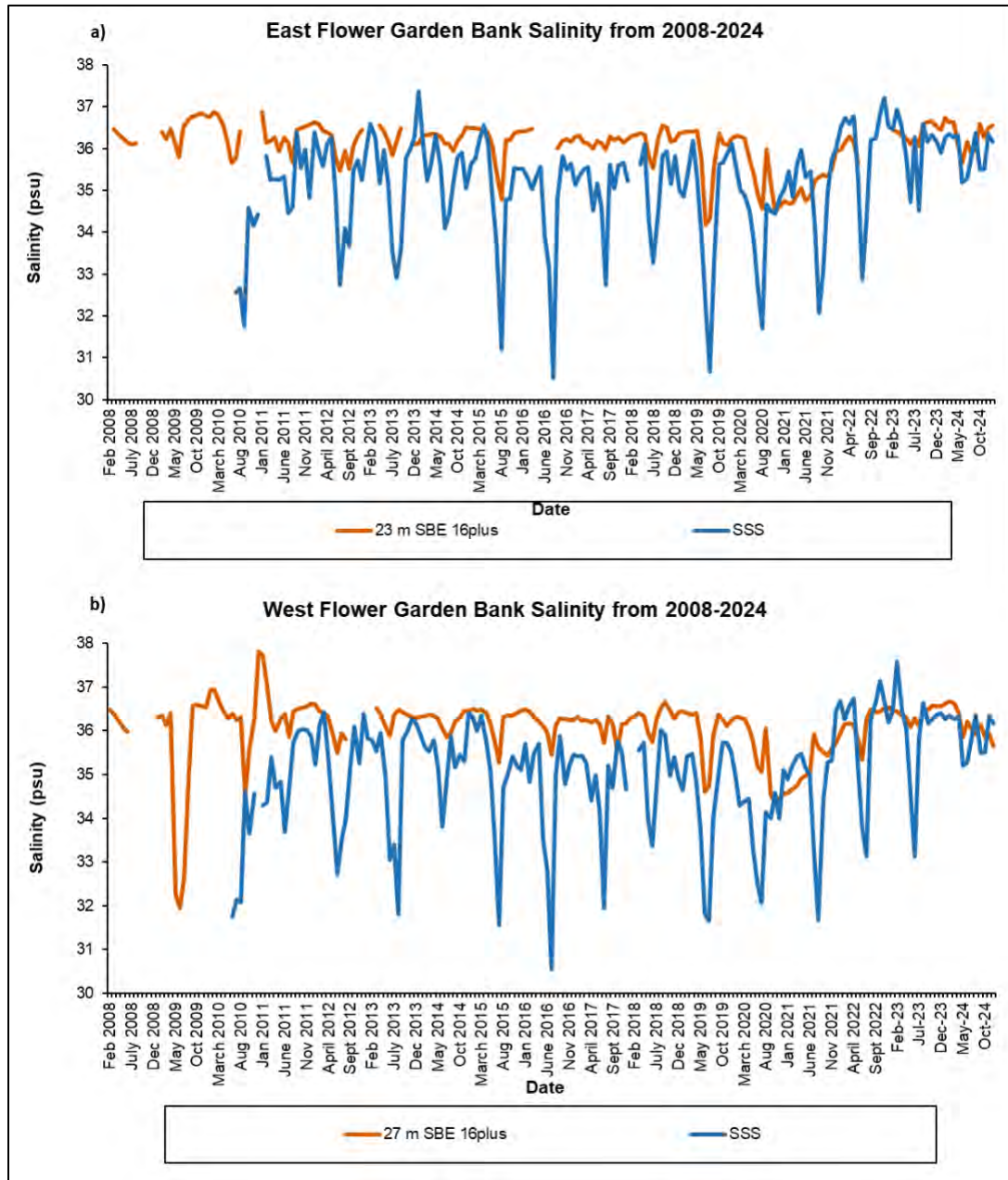


Figure 6.6. Monthly mean salinity showing seasonal variation and long-term trends at (a) EFGB (23 m) and (b) WFGB (27 m) from 2008 to 2024.

Turbidity

Turbidity sensors at EFGB and WFGB were exchanged in July 2024 and July 2025. Both banks experienced a disturbance in June 2024, coinciding with the passage of Hurricane Beryl through the Gulf. At WFGB, turbidity levels remained elevated throughout November 2024 before returning to expected conditions in December (Figure 6.7). This extended elevation coincided with increased regional storm activity in the Gulf and a potential influence from riverine sources, though the precise cause is uncertain. Turbidity typically remained low overall and within expected ranges for offshore reef environments. Values ranged from 0.00 to 7.83 ntu at EFGB and from 0.04 to 87.75 ntu at WFGB (Figure 6.7).

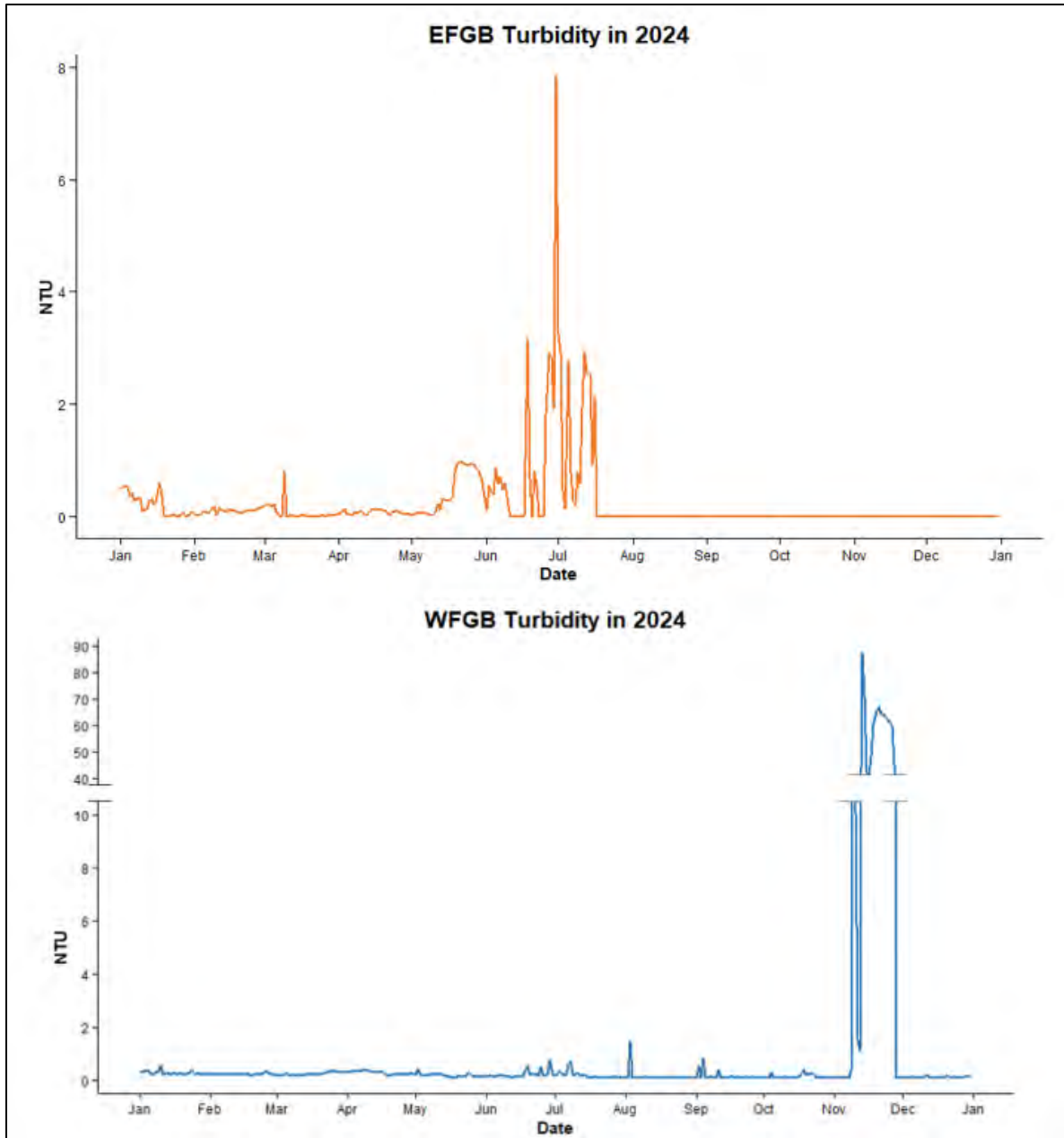


Figure 6.7. Daily mean turbidity (ntu) in 2024 at East Flower Garden Bank (23 m) and West Flower Garden Bank (27 m). EFGB turbidity is shown on a continuous y-axis, whereas the WFGB time series is displayed with a broken y-axis to accommodate an extreme turbidity event in late 2024 while preserving resolution of lower-magnitude variability earlier in the year.

Dissolved Oxygen

The dissolved oxygen sensor at EFGB was first deployed in July 2024 and exchanged in July 2025. It recorded data through the end of 2024 before its batteries failed early in 2025. Measured dissolved oxygen values at EFGB ranged from 185.66 to 207.39 $\mu\text{mol}/\text{kg}$ between July and December 2024 (Figure 6.8). Calculated values, derived from daily averaged temperature and salinity following Garcia et al. (1992), ranged from 186.11 to 225.94 $\mu\text{mol}/\text{kg}$ at EFGB and from 186.21 to 224.72 $\mu\text{mol}/\text{kg}$ at WFGB. At EFGB, the calculated dissolved oxygen values closely matched the measured values throughout the deployment period, providing confidence in both the sensor performance and calculations.

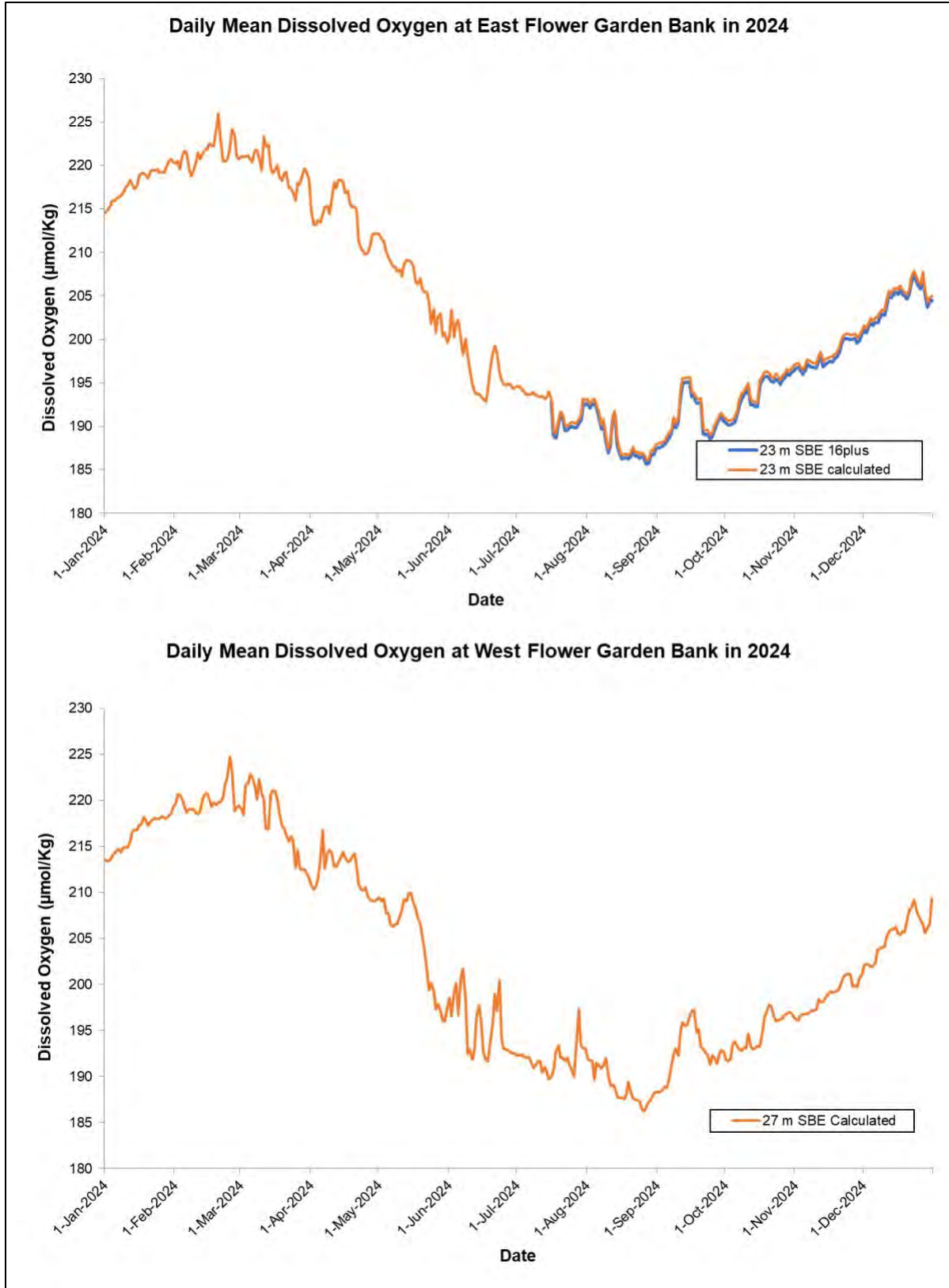


Figure 6.8. Daily mean dissolved oxygen ($\mu\text{mol/Kg}$) values in 2024 from EFGB (23 m) and WFGB (27m).

Water Column Profiles

Only one water column profile was collected at each bank in 2024 due to limited offshore access. Water column temperatures at both EFGB and WFGB varied by less than 2 °C through 20 m depth and indicated well-mixed conditions throughout the water column (Figure 6.9). Salinity values were also similar between banks, with differences averaging less than 1 psu. Dissolved oxygen levels varied by less than 1 mL/L between the two sites, consistent with mixing throughout the upper water column. pH remained stable across depths at both banks, with variation less than 0.2 units (Figure 6.9).

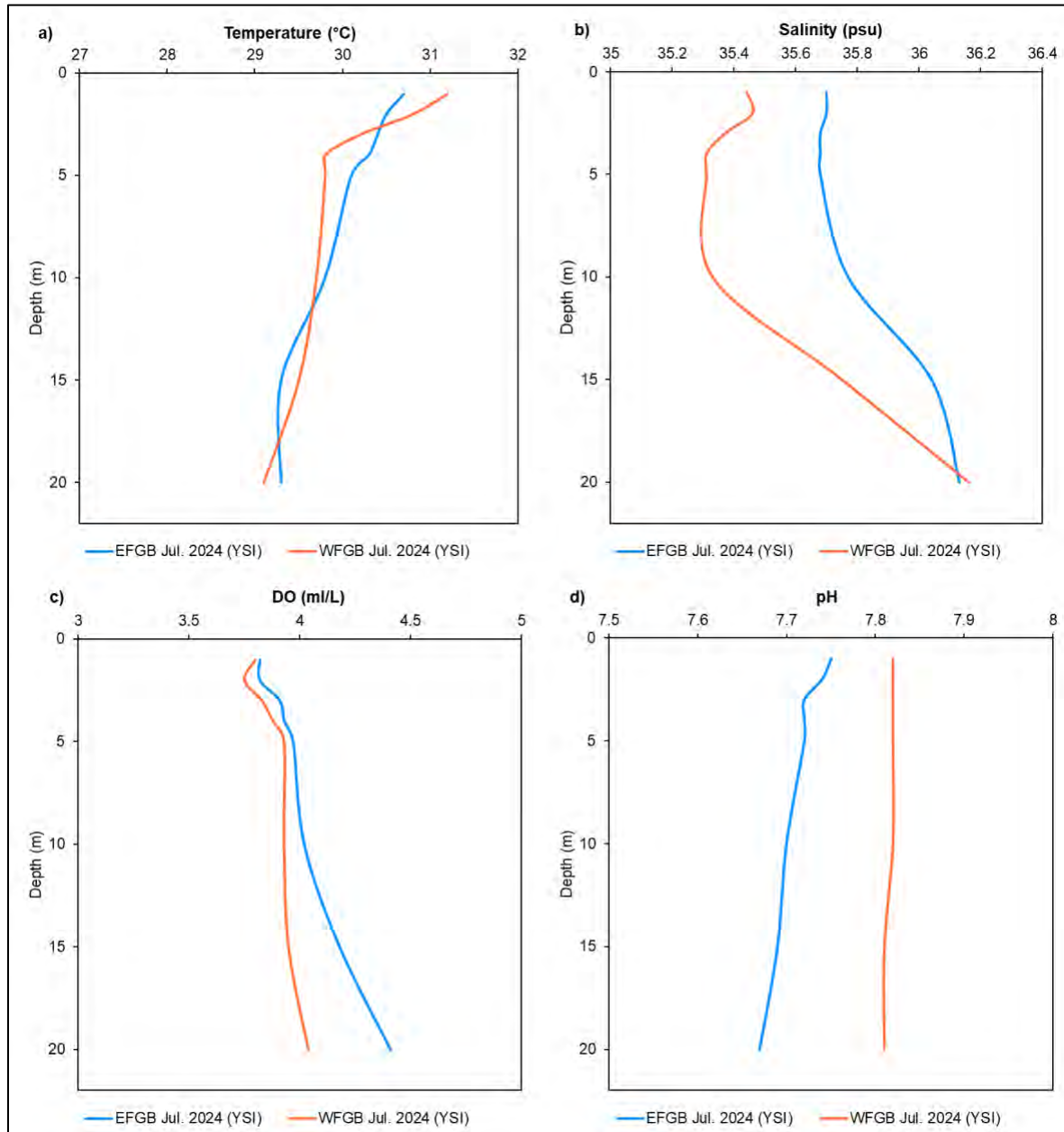


Figure 6.9. (a) Temperature, (b) salinity, (c) dissolved oxygen (DO), and (d) pH water column profile data from EFGB and WFGB in January 2024.

Water Samples

The first chl *a* and nutrient samples were collected as part of the long-term monitoring program in 2002. Since then, quarterly nutrient levels have typically been below detection limits, with occasional ammonia and TKN detections prior to 2012 (Figure 6.10; Figure 6.11). Although no samples were collected in 2024 due to vessel limitations that prevented the deployment of water sampling equipment, the dataset represents more than two decades of near-continuous monitoring at EFGB and WFGB. As a result of the missed sampling, ocean carbonate measurements typically conducted by TAMU-CC in tandem with nutrient sampling were also not performed.

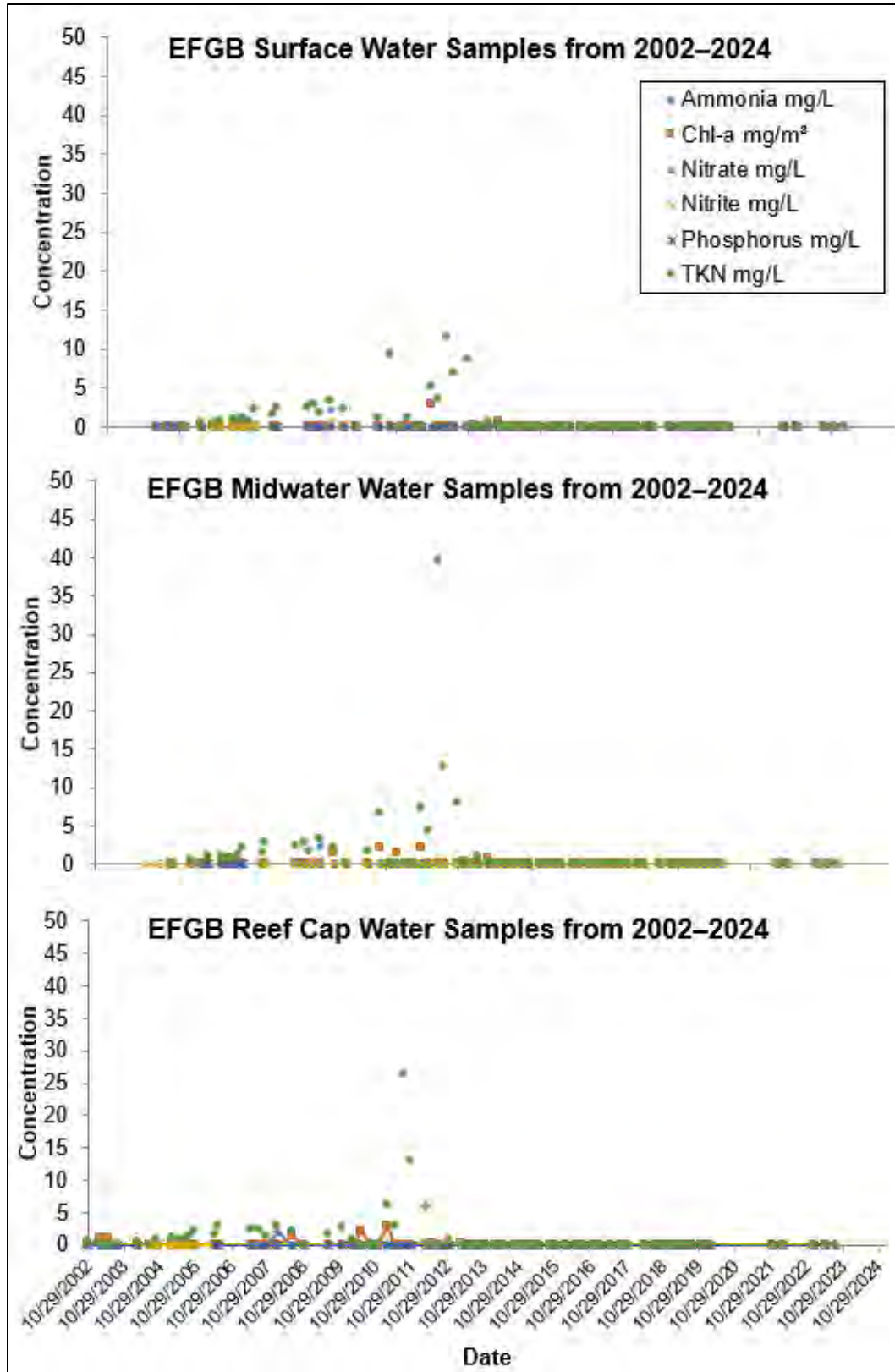


Figure 6.10. Nutrient concentrations from EFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2024. While the 2024 axis is shown to preserve the full monitoring timeline, no water samples were collected in 2024 due to vessel limitations.

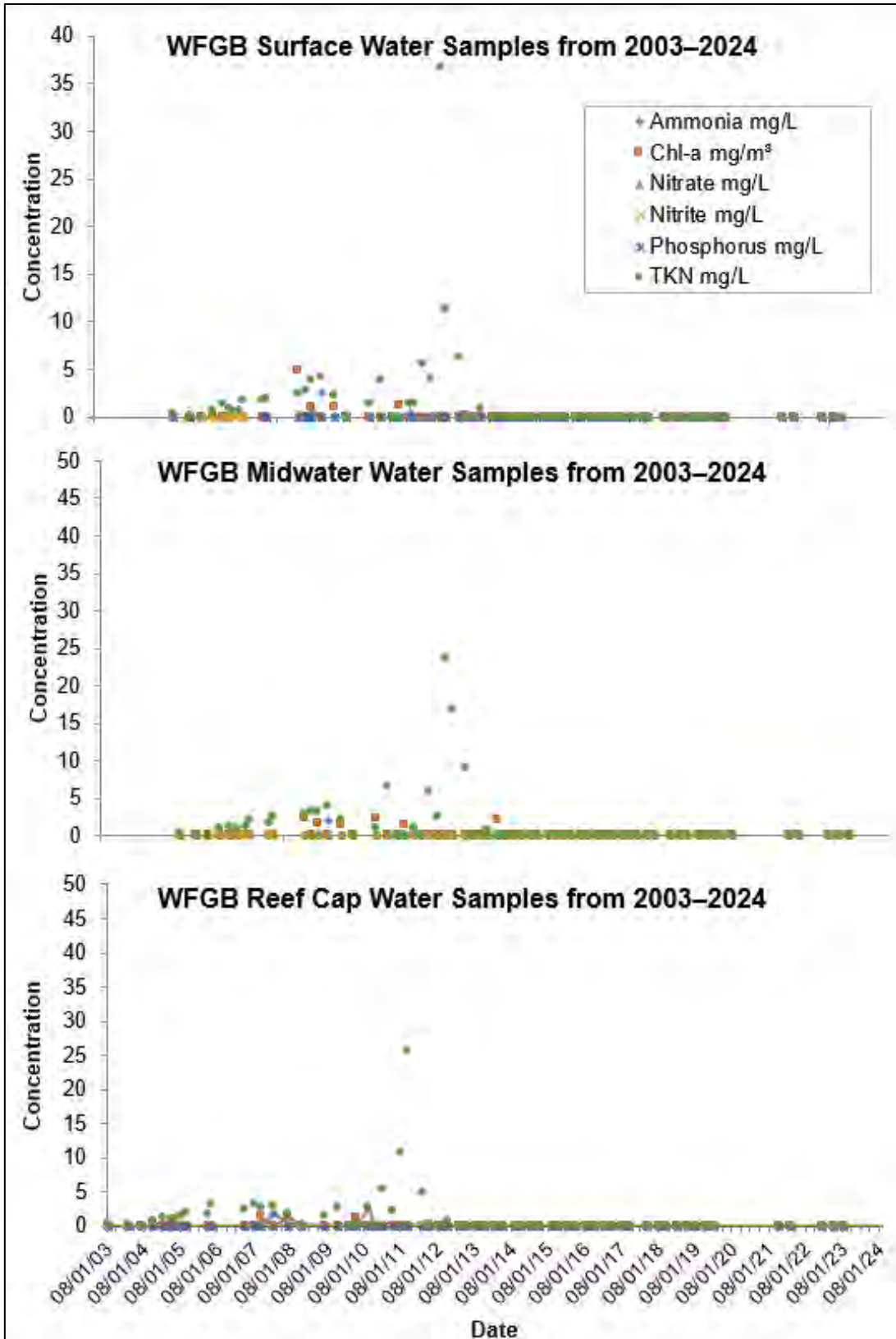


Figure 6.11. Nutrient concentrations from WFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2003 through 2024. While the 2024 axis is shown to preserve the full monitoring timeline, no water samples were collected in 2024 due to vessel limitations.

Water Quality Discussion

Water quality monitoring in 2024 was limited by the extended unavailability of the R/V *Manta*, which restricted offshore access and prevented use of the program's CTD rosette for quarterly water sampling. Despite these limitations, available in situ and satellite data captured key oceanographic conditions at EFGB and WFGB, including elevated seawater temperatures that exceeded the historical bleaching threshold in August and September, contributing to observed coral bleaching.

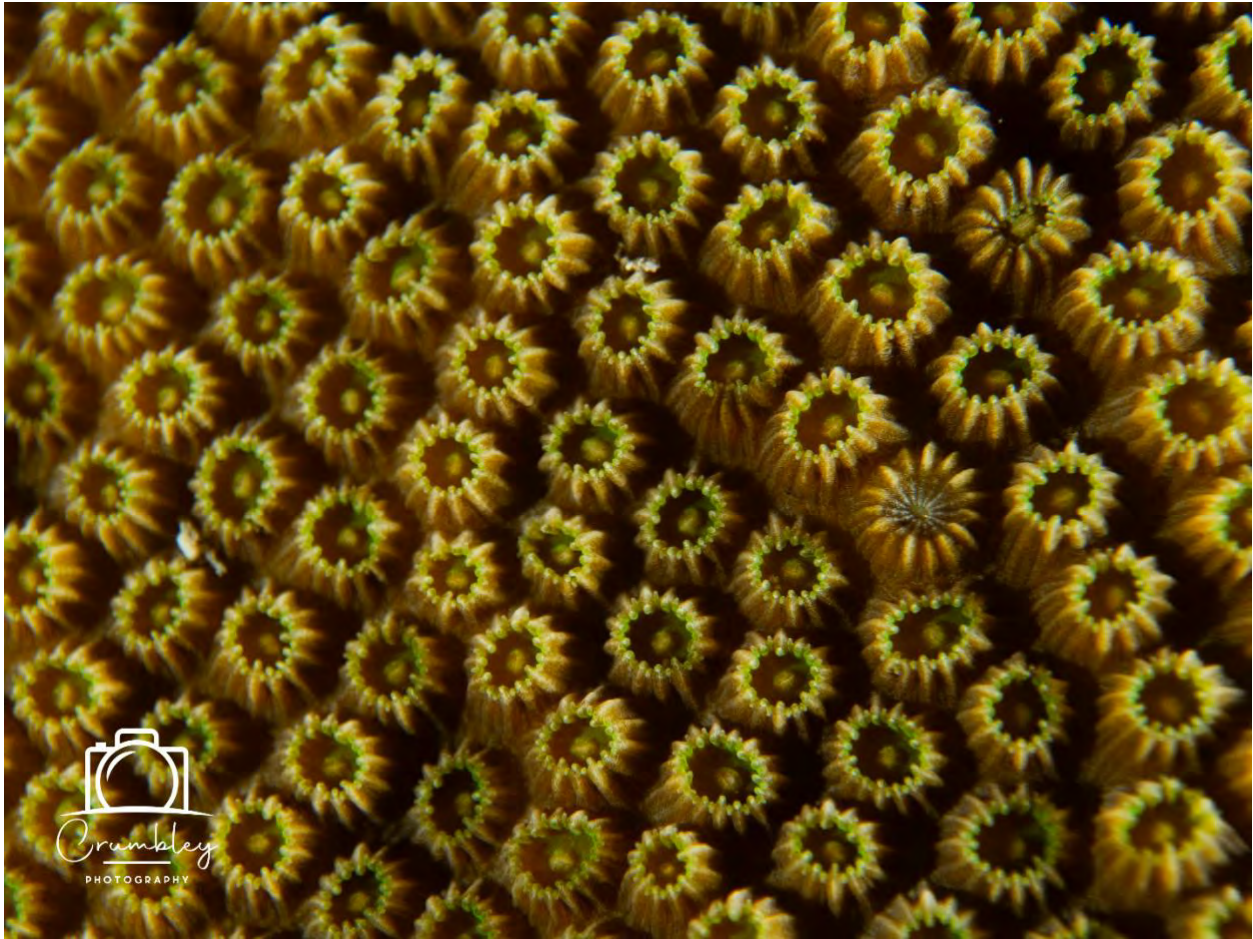
Long-term temperature trends at both banks continue to show significant monotonic increases, with reef cap and surface waters warming steadily since 1990, elevating the likelihood of future bleaching events. In 2024, two hurricanes, Hurricane Beryl (June 26) and Hurricane Francine (September 8), passed within 200 kilometers of East and West Flower Garden Banks. Both storms had acute impacts, briefly decreasing water temperatures and salinity at the banks that were quickly restored to anticipated levels over a few days. While the impacts of the storm were short lived, the sudden decrease in temperature helps to reduce bleaching stress across the banks.

Mean SSS fluctuated considerably at both banks, with reef cap salinity remaining consistent with long-term averages during the first half of the year. Despite annual variability, salinity at depth remained within the expected range for coral reefs in the western Atlantic (31 to 38 psu; Coles & Jokiel, 1992). As in previous years, the probable source of low-salinity water at the banks is the nearshore river-seawater mix originating primarily from the Mississippi and Atchafalaya river watersheds, which occasionally extends to the outer continental shelf (Kealoha et al., 2020).

No nutrient or carbonate samples were collected in 2024 due to equipment inoperability. However, nutrient levels in past years have remained below detection limits, consistent with the oligotrophic nature of offshore waters in the Gulf. Turbidity data were only partially recovered but indicated peak values in June, likely driven by Hurricane Beryl. These transient increases, combined with thermal stress and variable salinity, suggest the potential for compounding environmental stressors on reef communities.

The water column connects coral reef habitats to both aquatic and terrestrial systems. As such, water quality data remain a critical component of the sanctuary's long-term monitoring program, providing insight into the presence of land-based inputs that can affect coral reef ecosystem function. Although no quarterly water samples were collected in 2024, most surface and reef cap temperature data were captured through moored instrumentation and the Sofar Ocean® buoy at EFGB. The long battery life and durability of the SBE 16plus and HOBO instruments helped minimize data gaps, though extended deployment periods contributed to occasional sensor malfunctions. While equipment durability and satellite data helped reduce overall gaps in 2024, the loss of key instruments, including the Sofar buoy at EFGB, underscores the vulnerability of offshore monitoring infrastructure and the importance of continued investment in redundancy and rapid deployment capacity. The Sofar buoy at EFGB was reinstalled in February 2025, while the remaining instruments are scheduled to be recovered and redeployed in July 2025.

Chapter 7: Conclusions



A close-up photo of an *Orbicella faveolata* colony. Photo: Tiffany Crumbley/Crumbley Photography

Despite widespread coral reef declines globally, mean coral cover within EFGB and WFGB long-term monitoring study sites has ranged from 40–60% throughout the 35-year monitoring period. 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 at EFGB in 2024 as part of the long-term monitoring program, allowing for benthic cover calculations in a random stratified design across the reef cap, rather than just in study sites. Surveys were not completed at WFGB due to time constraints. 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 fish assemblage. The comparatively low abundance and diversity among some groups may reflect recruitment limitation associated with the biogeographic isolation of the banks and the absence of nearby nursery habitats. Herbivores accounted for the highest biomass in 2024, with chubs and stoplight parrotfish as the dominant contributors. 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 2024 bleaching event marks the second consecutive year of observed coral stress at FGBNMS, signaling a potential shift toward more frequent thermal disturbances in the region. While bleaching severity varied across banks and survey periods, the occurrence of back-to-back events is cause for concern. Recurrent thermal stress, even when moderate, can weaken coral resilience, reduce reproductive output, and increase susceptibility to disease and future disturbances (Hughes et al., 2017). These findings highlight the need for proactive management strategies that account for cumulative impacts over time. Continued long-term monitoring, coupled with responsive bleaching surveys during thermal events, will be essential for detecting changes in coral condition, informing restoration priorities, and guiding adaptive management within the sanctuary.

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 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. Although monitoring is sounding alarms concerning changes, particularly with regard to increasing ocean temperatures, 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.

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Glossary of Acronyms

ANOSIM	analysis of similarity
BOEM	Bureau of Ocean Energy Management
CCA	crustose coralline algae
chl <i>a</i>	chlorophyll <i>a</i>
CPCe	Coral Point Count® with Excel® extensions
CTD	conductivity, temperature, and depth
CV%	coefficient of variation
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
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

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