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National Oceanic and Atmospheric Administration

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Coral Reef Long-Term Monitoring at East and West Flower Garden Banks: 2025 Annual Report

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Cover photo: Butterflyfish swim over a colony of *Pseudodiploria strigosa* at East Flower Garden Bank. Photo: Tiffany Crumbley/Crumbley Photography

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The Office of National Marine Sanctuaries, part of the National Oceanic and Atmospheric Administration, serves as the trustee for a system of underwater parks encompassing more than 629,000 square miles of ocean and Great Lakes waters. The 18 national marine sanctuaries within the National Marine Sanctuary System represent areas of America's ocean and Great Lakes environment that are of special national significance. Within their waters, giant humpback whales breed and calve their young, coral colonies flourish, and shipwrecks tell stories of our nation's maritime history. Habitats include beautiful coral reefs, lush kelp forests, whale migration corridors, spectacular deep-sea canyons, and underwater archaeological sites. These special places also provide homes to thousands of unique or endangered species and are important to America's cultural heritage. Sites range in size from less than one square mile to almost 583,000 square miles. They serve as natural classrooms and cherished recreational spots, and are home to valuable commercial industries.

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Report Availability

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

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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 2025, alongside 36 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 2025, mean coral cover, estimated from random transect data within EFGB and WFGB one-hectare study sites, was 55%. 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 33% within the EFGB study site and 13% within the WFGB study site, macroalgae cover has increased significantly at both banks since 1998. The reef fish community remained characterized by low diversity and high biomass. Trophic structure differed between banks in 2025, with piscivores comprising the greatest biomass at EFGB, while WFGB shifted to a planktivore-dominated community driven by high abundances of Atlantic creolefish. Regal demoiselles, which were not observed in 2024, were detected in low numbers in 2025, suggesting recent recruitment. A significant increase in reef-cap seawater temperature was detected at both banks from 1990 to 2025. In situ reef-depth temperature data indicated that EFGB exceeded site-specific bleaching thresholds in 2025, whereas WFGB did not meet threshold conditions associated with widespread bleaching. Coral bleaching and paling remained low in 2025, with evidence of recovery following consecutive bleaching years in 2023 and 2024, although disease persisted as a secondary stressor. 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, coral bleaching

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

Large reef-building corals rise from the seafloor, forming layered structures that provide shelter and habitat for reef fish and countless invertebrates. Schools of small fish hover above the coral. 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.

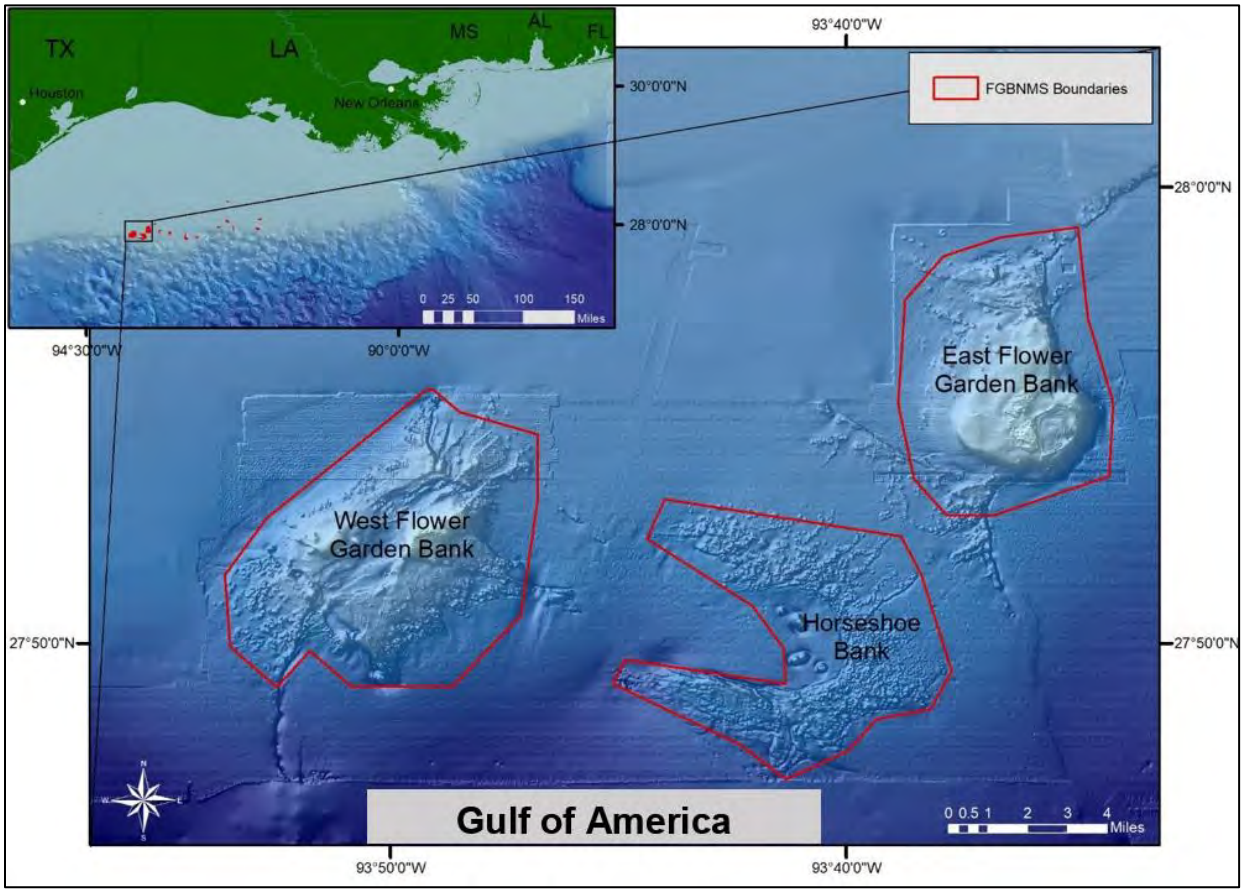


Figure 1.1. Map of EFGB, WFGB, and Horseshoe Bank with an inset of the Gulf Coast states and FGBNMS boundaries along the continental shelf of the northwestern Gulf of America. Horseshoe Bank is part of FGBNMS but doesn't have a shallow coral reef. Image: Marissa Nuttall/CPC

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 [MMS], 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).

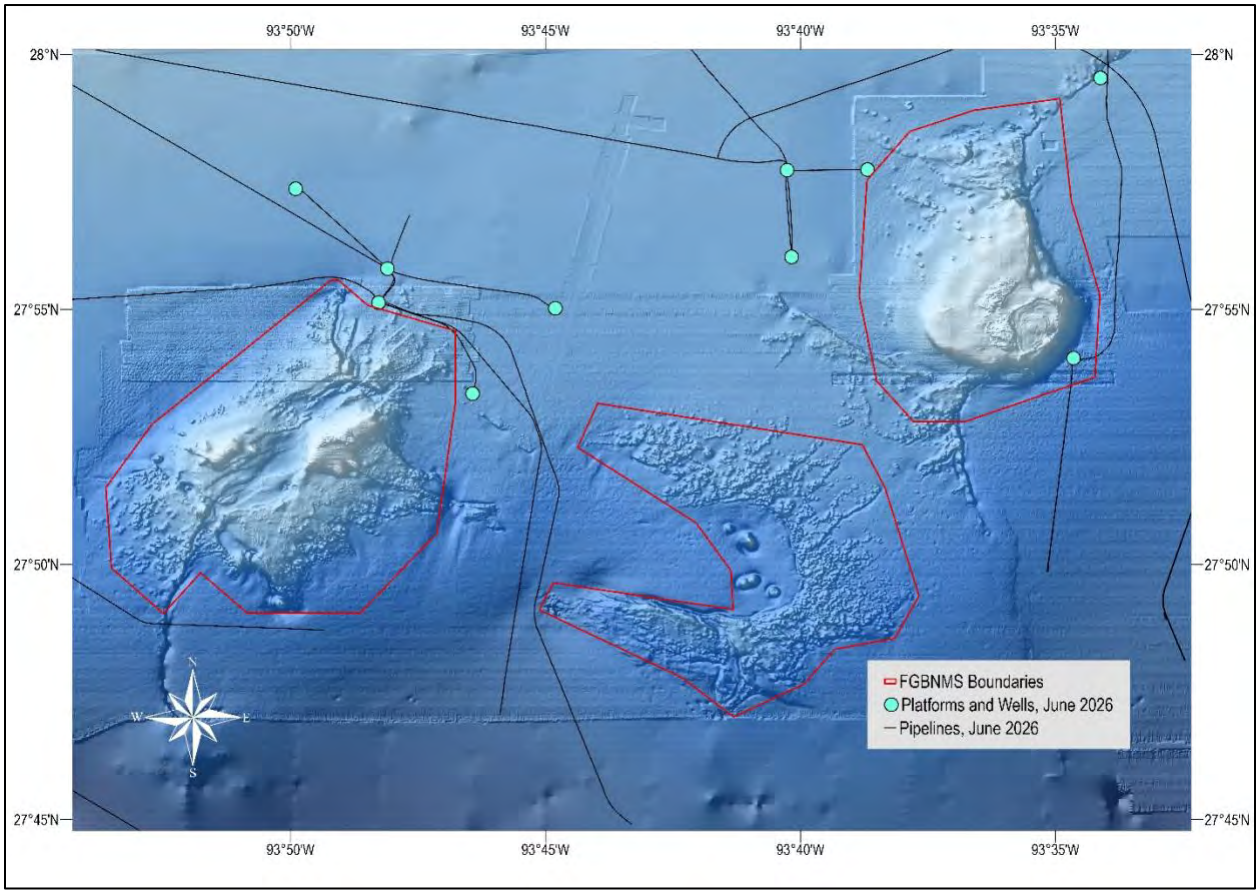


Figure 1.2. Map of oil and gas platforms, wells, and pipelines near EFGB and WFGB as of June 2026. Image: Kelly O'Connell/CPC

Initially supported by industry funding and later by the Minerals Management Service (MMS) through a contract with Texas A&M University (TAMU), the sanctuary established a long-term

monitoring program with permanent one-hectare study sites on each bank in 1989 (Continental Shelf Associates, 1989; Gittings et al., 1992; Figure 1.3). These study sites consist of fixed 10,000 m^2 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 MMS and NOAA established an interagency agreement for FGBNMS to carry out the long-term monitoring program.

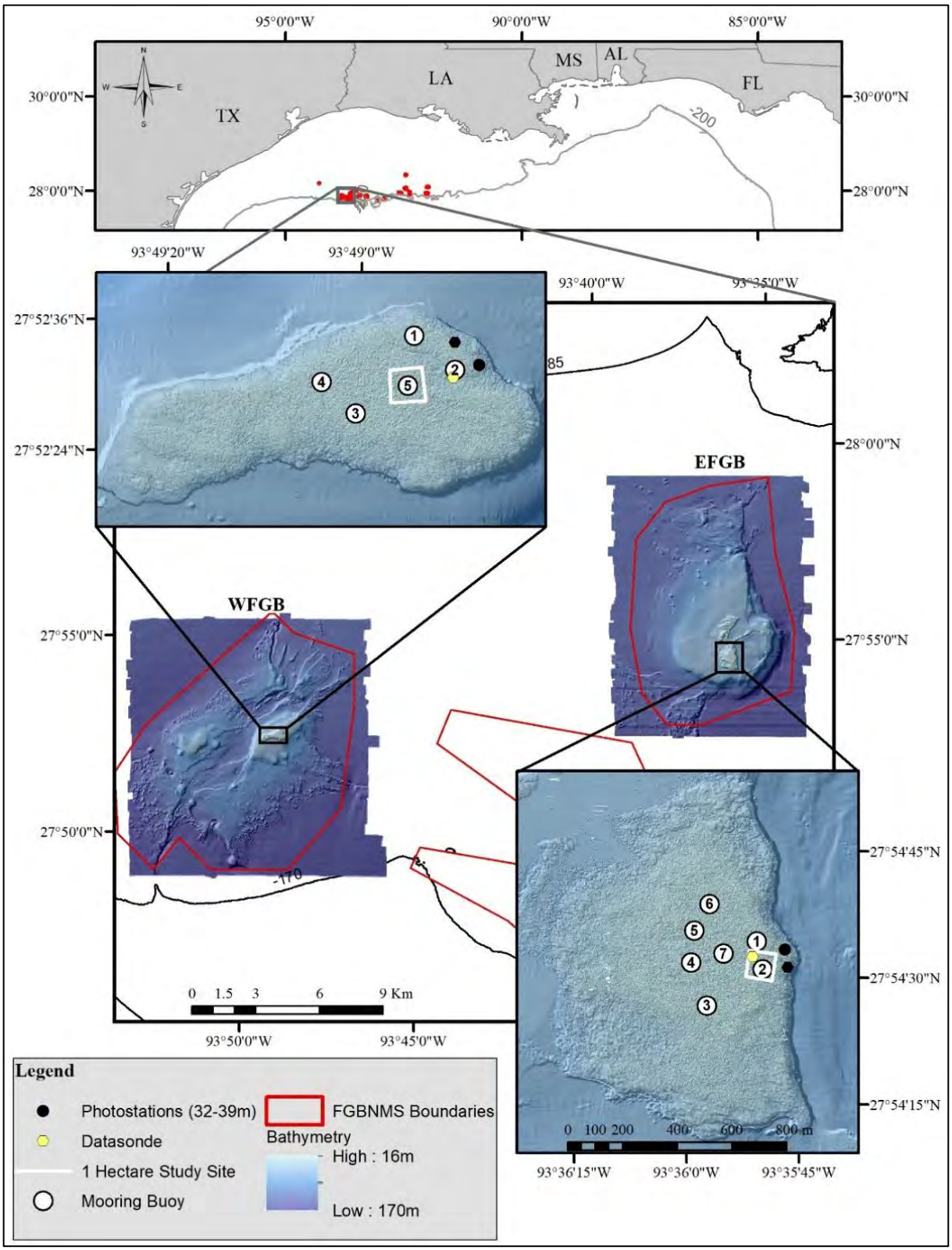


Figure 1.3. Shaded relief maps of EFGB and WFGB, with inset of the Gulf of America coastline, long-term monitoring one-hectare study sites, and datasonde and repetitive photostation locations, which range in depth from 32–39 m. Image: Marissa Nuttall/CPC

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: Long-term monitoring data, permanent photostations, and image archive supporting assessment of resource condition and damage.
- 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 (2021–2025). 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 (m)	Description
EFGB Mooring #2	27° 54.516’ N	93° 35.831’ W	19.2	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

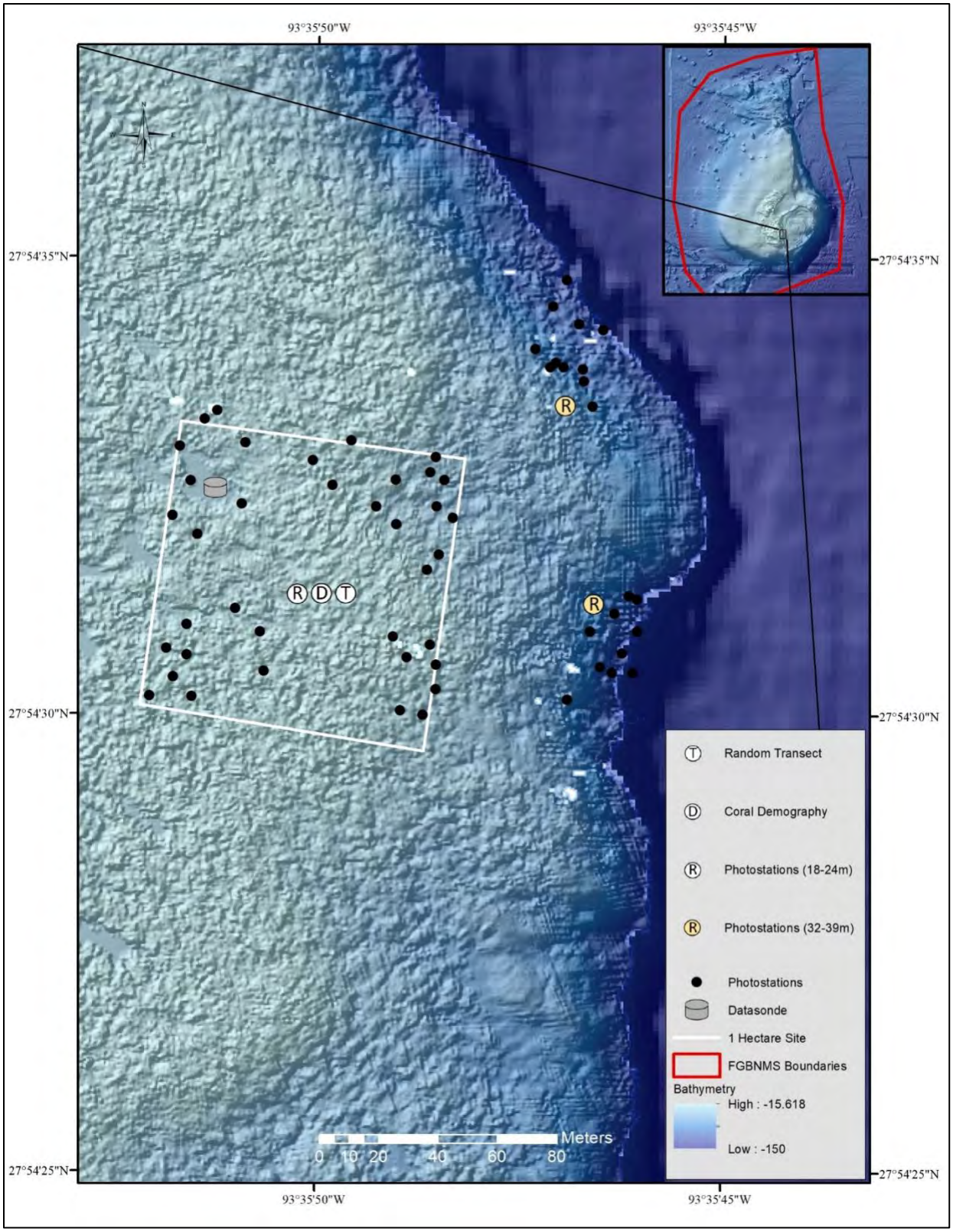


Figure 1.4. Shaded relief map of EFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also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

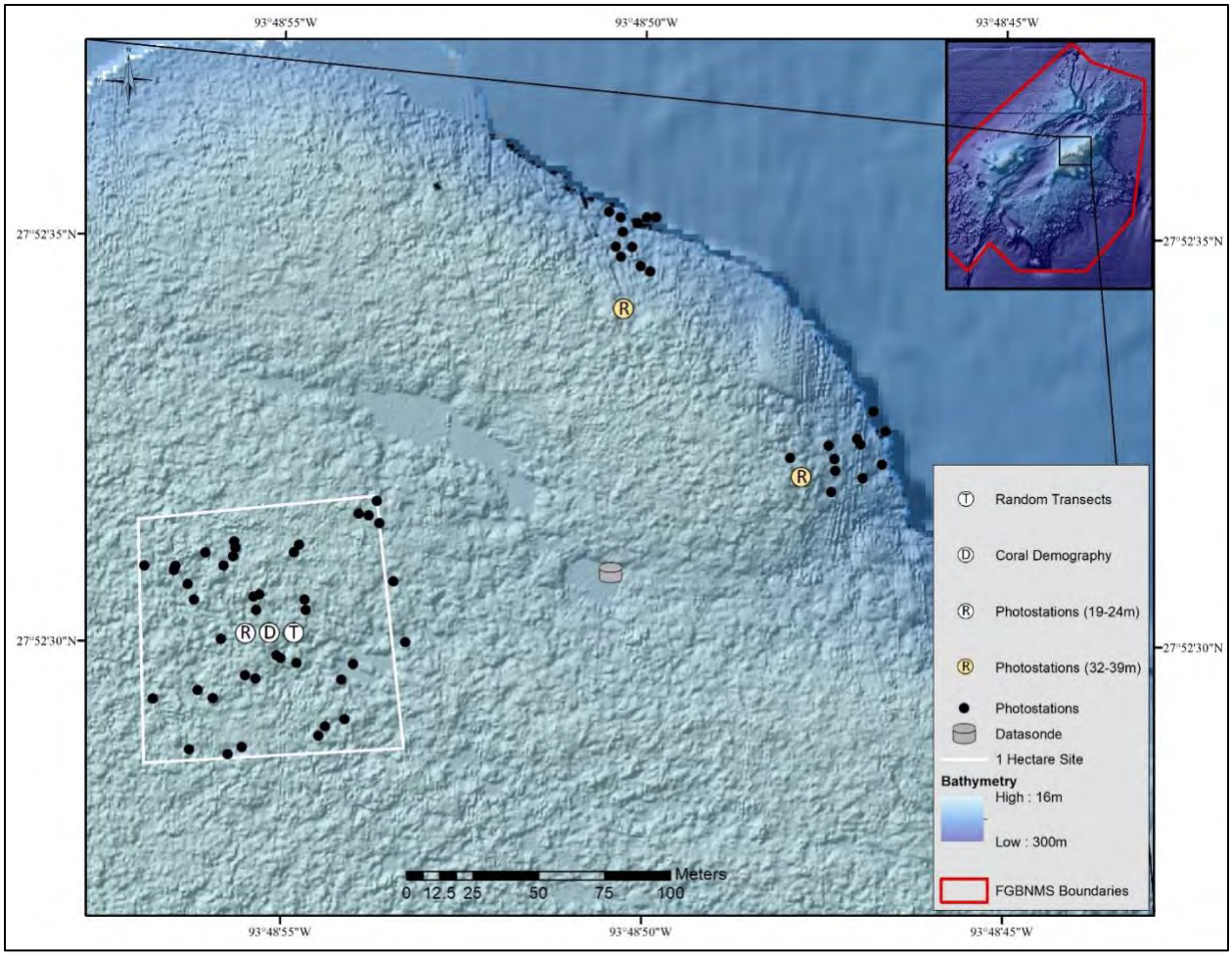


Figure 1.5. Shaded relief map of WFGB showing the location of the long-term monitoring study site, within which repetitive photostations (18–24 m), random transects, and coral demographic surveys are conducted. The locations of the water quality datasonde and repetitive photostations at 32–39 m are also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

Coral Reef Long-Term Monitoring Field Operations and Data Collection

To date, the coral reef monitoring program has spanned 36 years, with annual collection occurring nearly continuously over this time. In 2025, long-term monitoring data were collected at both EFGB and WFGB, with scuba operations conducted from the M/V *Fling* and R/V *Manta* (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 27–31 and August 4–8, 2025 (Table 1.2), under favorable conditions, including mild currents (<0.5 kt), 20 m visibility, 29°C water temperatures, and 2–4-foot seas. All monitoring objectives were met. Datasondes at EFGB and WFGB were retrieved, downloaded, and redeployed. Temperature sensors at deep repetitive photostations were retrieved in July, 2025.

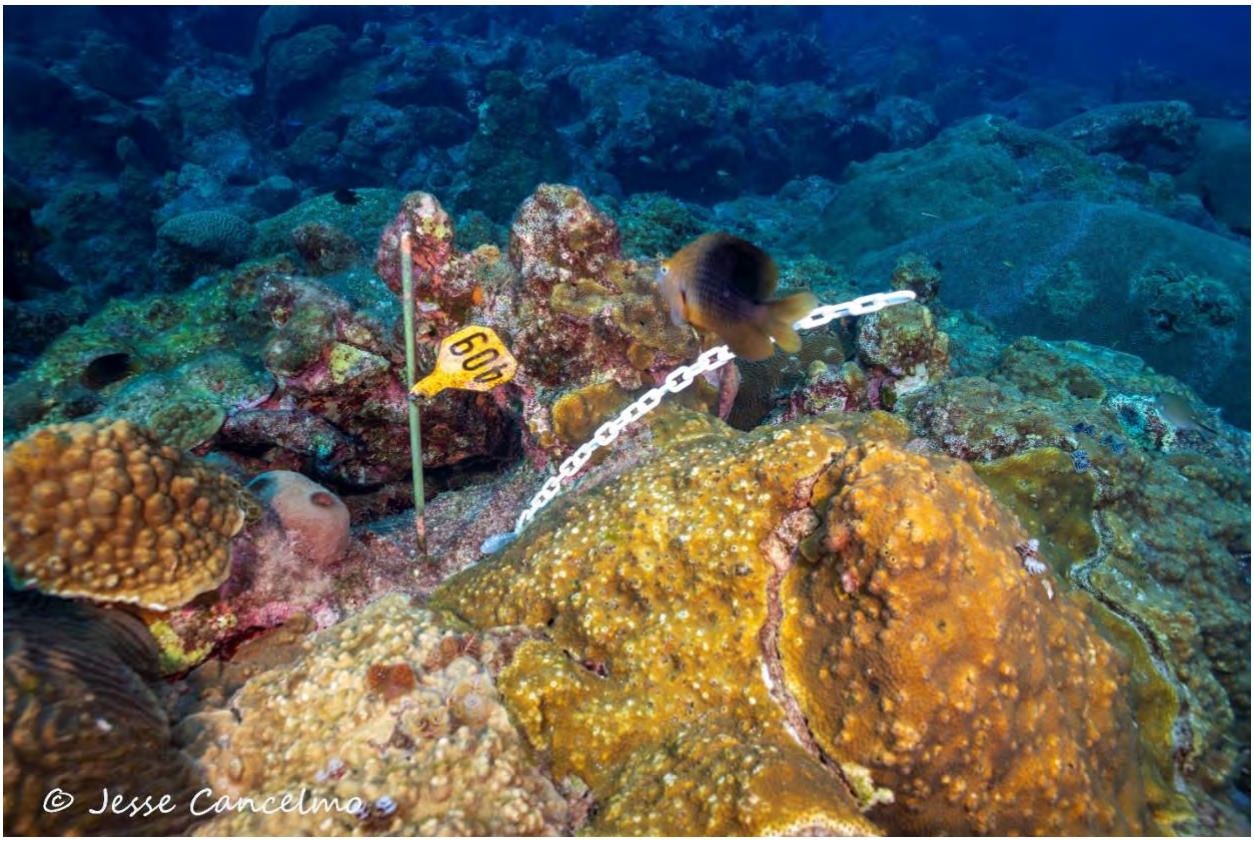
Thermal stress during the 2025 summer season was minimal compared to the elevated conditions observed in 2023–2024. Long-term monitoring activities were conducted earlier in the season, but no additional cruises occurred later in the summer to assess bleaching conditions. Based on field observations, widespread coral bleaching was not observed during the 2025 season.

The R/V *Manta* entered its annual shipyard repair period on September 30, 2023, and remained in maintenance until March 25, 2025. No dedicated water quality cruises were conducted in 2025. However, water quality sampling and related tasks were completed opportunistically alongside other scheduled cruises throughout the year to ensure continued data collection.

Table 1.2. Monitoring cruises completed at EFGB and WFGB in 2025.

Date	Cruise and Tasks Completed
02/02–02/03/2025	Mooring buoy cruise aboard chartered vessel Mas Chingon 2: Re-deployment of Sofar buoy at EFGB
07/27–07/31/2025	Long-term monitoring cruise on the M/V *Fling*: EFGB and WFGB annual monitoring within the study site; water quality instrumentation exchange
08/04–08/08/2025	Long-term monitoring cruise on the R/V *Manta*: EFGB and WFGB reef-wide random transect surveys

Chapter 2:
Benthic Community



This is one of the permanent repetitive photostations used in the FGBNMS long-term monitoring program. White chains are temporarily dropped to help make the station more visible to the photographer. Photo: Jesse Cancelmo

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. Additional photo transect surveys were conducted in a random stratified design across the coral caps to assess benthic cover outside of the study sites in 2025. A total of 21 reef-wide transects were completed at EFGB and 26 reef-wide transects were completed 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. 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 are instead used to help identify mechanisms of change.

Benthic Community Methods

Random Transect Field Methods

In 2025, 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. An additional 47 transects were conducted outside the study sites (n = 21 at EFGB, n = 26 at WFGB) in 2025. Live-boating operations allowed divers to be dropped at each randomly stratified site where transects occurred where the divers descended. A Sony® Alpha 6600 digital camera with a Sony® E 16mm 2.8f lens in a Nauticam® NA-A6600 housing and a Nauticam® 4.33” fisheye dome port, mounted on a 0.58-m T-frame and two Inon® Z240 strobes was used to capture images along the transects. The bubble level mounted to the T-frame center ensured images were perpendicular to the slope of the bottom substrate. The mounted camera was placed at pre-marked intervals 80 cm apart on a spooled 15-m measuring tape, producing 17 non-overlapping images along the transect (Figure 2.1). Each still frame image captured a 0.8 x 0.6 m area (0.48 m^2). This produced a total photographed area of 8.16 m^2 per transect, and a minimum of 130.56 m^2 photographed area per study site per year. For more detailed methods, reference Johnston et al. (2017a).

It should be noted that during the entirety of the monitoring program, a variety of underwater camera setups were used as technology advanced from 35-mm slides (1989 to 2001) to digital videography using video still frame grabs (2002 to 2009) and then digital still images (2010 to 2023; Gittings et al., 1992; Continental Shelf Associates, 1996; Dokken et al. 1999, 2003; Precht et al., 2006; Zimmer et al., 2010; Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024). Consistency for photographic random transect methods was ensured by training all scientific divers in the proper operation of the camera system from a standardized operating procedure. Equipment was 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.

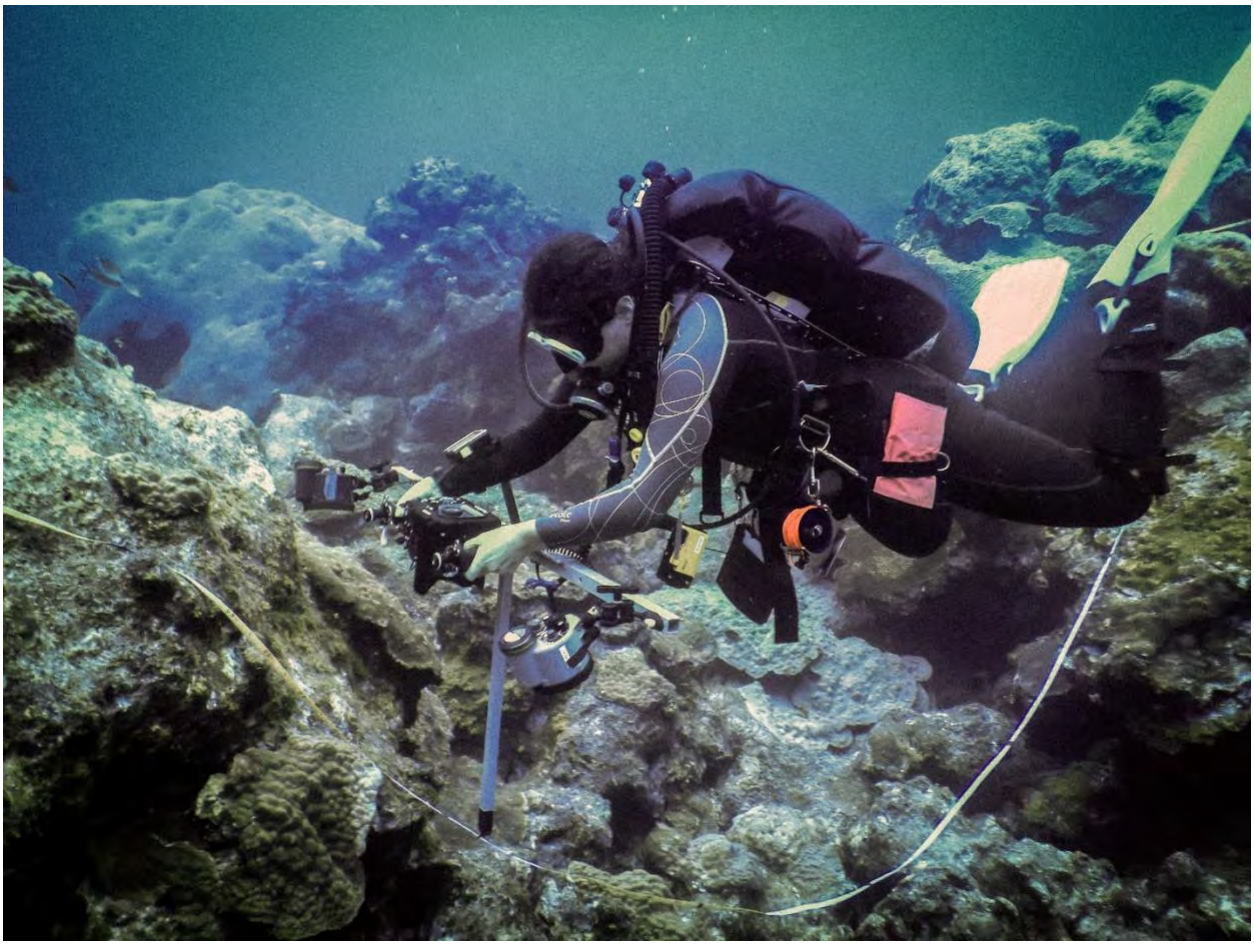


Figure 2.1. A diver conducts a random transect at EFGB. Photo: Ryan Hannum/CPC

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 publications prior to 2023. Data collected from 2004–2008 were not reanalyzed because surveys during that period used video transects rather than still imagery, and the original processed values were retained.

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 (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 ± standard error (SE).

Incidents of coral bleaching, paling, concentrated and isolated fish biting, damselfish gardening, 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. Any point that landed on a coral that was part of an algae garden from a damselfish species was characterized as "damselfish gardening." 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 was 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) based on Euclidean distance resemblance matrices was used to test for differences in benthic community composition between banks (EFGB and WFGB), survey types (study site and reef-wide surveys), and their interaction (Anderson et al., 2008). When a significant interaction was detected, additional PERMANOVAS were conducted within banks and survey types to clarify the nature of these differences. Groups contributing to observed differences were examined using similarity percentages (SIMPER) to assess the percent contribution of each benthic group to overall dissimilarity (Clarke et al., 2014).

Coral community was evaluated separately using PERMANOVA on square-root transformed coral species percent cover data with Euclidean distance resemblance matrices to test for differences between banks, survey types, and their interaction. Diversity indices for coral species, including Margalef's species richness (d), Pielou's evenness (J'), and Shannon diversity (H'), were calculated for each random transect from both study site and reef wide surveys. 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 2025), 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 principal coordinate 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 of year groups were conducted using similarity profile analysis (SIMPROF) on Euclidean distance resemblance matrices to identify significant ($\alpha = 0.05$) clusters in the data (Clarke et al., 2008). Differences in benthic community composition among years and between banks were tested using PERMANOVA. SIMPER identified groups contributing to observed dissimilarities (Clarke et al., 2014).

Mean percent benthic cover from random transect surveys was analyzed from 1989 to 2025. 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 Spearman's rank correlation (Helsel & Hirsch, 2002).

Temporal patterns in coral beta diversity from 1992–2025 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 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 formally testing categorical differences among years, dispersion values were examined as a continuous time series to evaluate long-term temporal patterns in coral community variability.

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 2025, all 37 photostations were located and photographed within the EFGB study site and 40 were located and photographed within the WFGB study site. Permanent repetitive photostations ranging in depth from 24–39 m and located beyond the study site boundaries were photographed, with 24 deep photostations located at WFGB and 22 located at EFGB.

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

It should be noted that during the entirety of the monitoring program, underwater camera setups used to capture benthic cover in the repetitive photostations changed as technology advanced from 35-mm slides and film (1989 to 2007) to digital still images (2008 to 2025) (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 m^2 area. Following the transition to digital imagery, the photographed area was adjusted in 2009 (5 m^2) to accommodate changes in camera equipment and image quality requirements, and has remained standardized at 5 m^2 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.

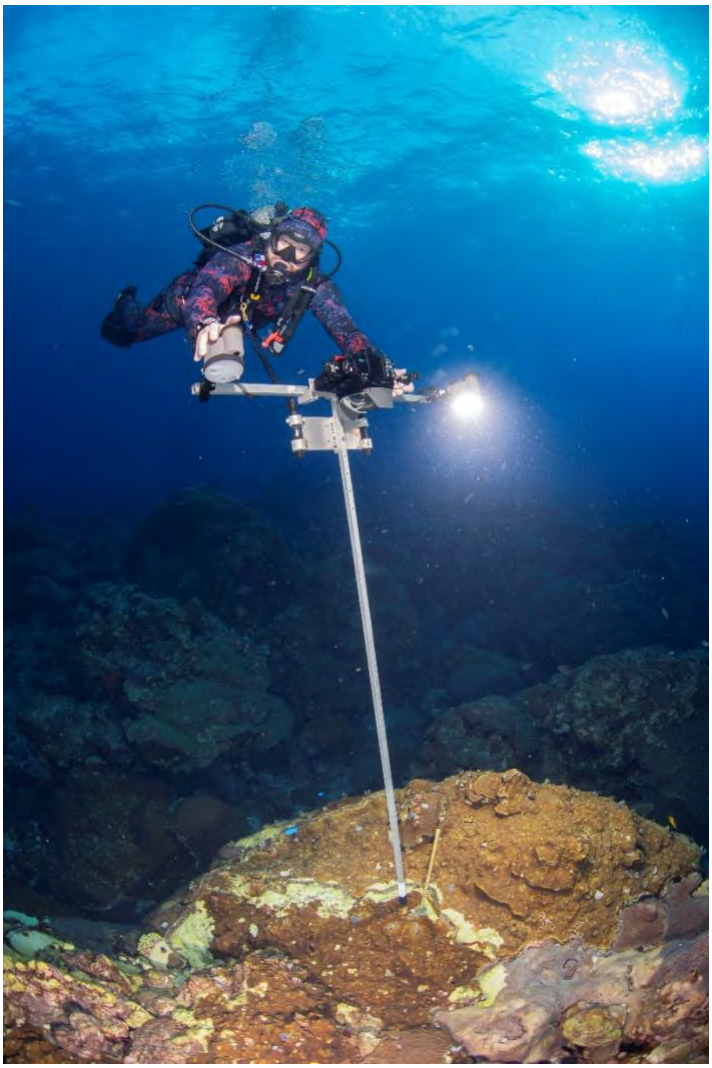


Figure 2.2. NOAA diver prepares to photograph a repetitive photostation with camera and strobes mounted to an aluminum t-frame. Photo: Donavon French/CPC

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 ± 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

Temporal trends in mean percent cover of major benthic groups at repetitive photostations within the EFGB and WFGB study sites were evaluated using Mann-Kendall trend tests applied to annual mean percent cover values. The deep photostations were established during multiple monitoring expansions between 2003 and 2017 and therefore do not represent a continuous time series across the full monitoring record; consequently, these stations were not included in the long-term trend analyses presented here.

To evaluate long-term changes in coral cover at the original repetitive photostations, a subset of stations that have been consistently monitored since the beginning of the program was analyzed separately. This subset included 24 EFGB photostations and 27 WFGB photostations located within the study sites at depths of approximately 20–24 m. Percent coral cover at these stations was compared between the first survey year (1989) and the most recent survey year (2025) using paired t-tests.

Benthic Community Results

Random Transect Mean Percent Cover

Coral had the highest mean percent benthic cover in both study site and reef-wide surveys at EFGB and WFGB in 2025 (Figure 2.3). Macroalgae was the second most abundant benthic group at the EFGB study site and in reef-wide surveys at both banks, whereas colonizable substrate was the second most abundant group at the WFGB study site. Study site surveys represent fixed long-term monitoring stations, whereas reef-wide surveys capture broader spatial variability across each bank.

PERMANOVA showed that benthic communities differed by both bank and survey type in 2025, with a significant interaction between the two variables. Therefore, the two survey types were analyzed separately within each bank to clarify these patterns.

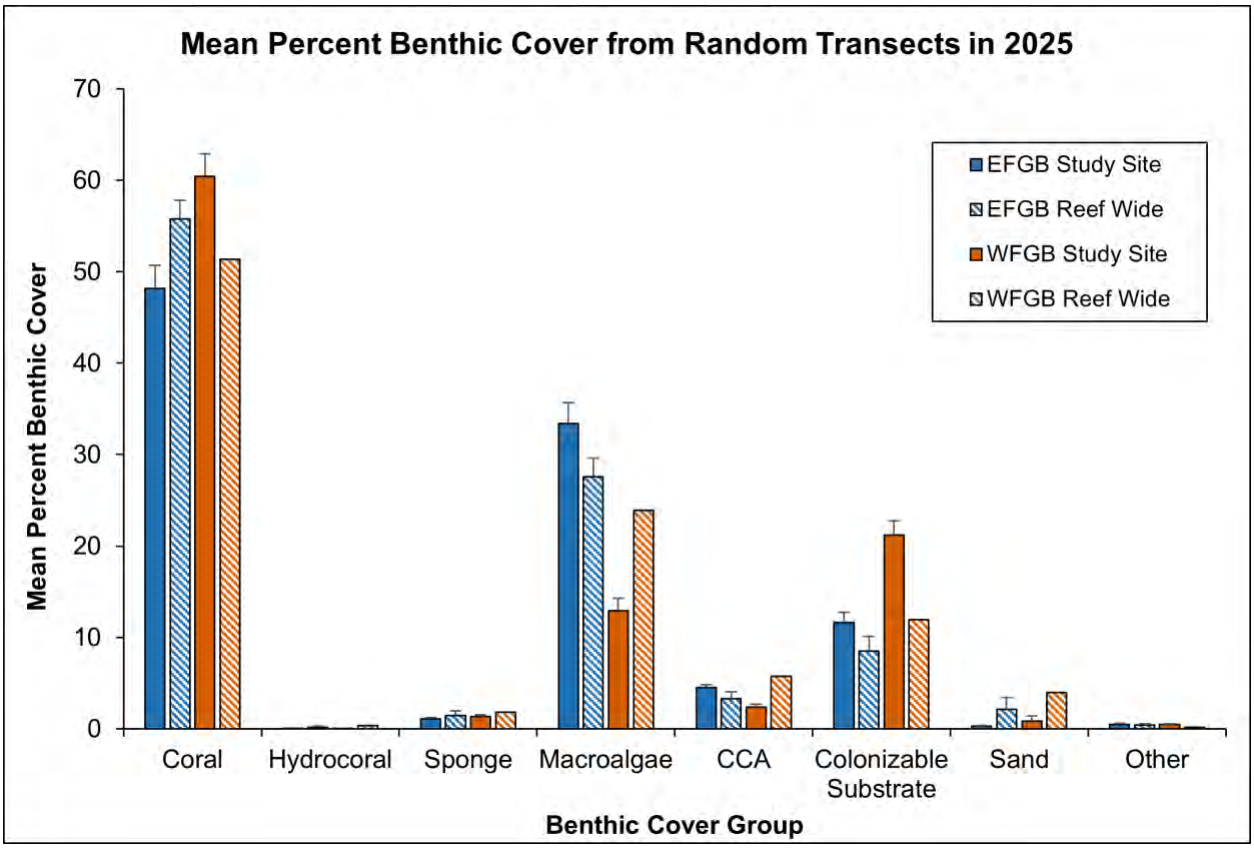


Figure 2.3. Mean percent benthic cover +SE from random transect surveys within study sites and reef-wide at EFGB and WFGB in 2025.

Table 2.1. PERMANOVA results comparing 2025 mean percent benthic cover at EFGB between survey types. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Survey Type (Study Site vs. Reef wide)	12.61	1	2.86	**0.02**
Res	154.14	35		
Total	166.75	36		

Table 2.2. PERMANOVA results comparing 2025 mean percent benthic cover at WFGB between survey types. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Survey Type (Study Site vs. Reef wide)	44.75	1	5.76	**0.001**
Res	310.60	40		
Total	355.34	41		

At EFGB, benthic community composition differed between the study site and reef-wide surveys, although the effect size was relatively small (Table 2.1). In contrast, WFGB showed a stronger difference between survey types (Table 2.2), indicating greater spatial variation across the bank. Together, these results explain the significant interaction in the full model. The distinction between study site and reef-wide surveys is more pronounced at WFGB than at EFGB and each method captures distinct benthic community compositions.

SIMPER analysis identified colonizable substrate and coral as the main contributors to the differences between the study site and reef-wide surveys at EFGB. Colonizable substrate was higher at the study sites, while coral was higher in the reef-wide surveys. SIMPER analysis indicated that macroalgae and coral were the primary contributors to the differences between study site and reef-wide surveys at WFGB. Macroalgae was more abundant in the reef-wide surveys, while coral was higher in study sites surveys.

To clarify the significant interaction between bank and survey type observed in the full model, additional PERMANOVAs were conducted separately for the study site and reef-wide datasets. At the study sites, benthic communities differed strongly between EFGB and WFGB (Table 2.3), indicating clear distinction between the two banks at the long-term monitoring stations. In contrast, bank differences in the reef wide surveys were weaker (Table 2.4), with a smaller component of variation compared to the study sites. Together, these results show that EFGB and WFGB remain distinct across both survey types, but the magnitude of the difference is far more pronounced within the study sites than in the broader reef wide sampling.

Table 2.3. PERMANOVA results comparing 2025 study site mean percent benthic cover between banks. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs WFGB)	60.65	1	25.11	**<0.001**
Res	72.46	30		
Total	133.1	31		

Table 2.4. PERMANOVA results comparing 2025 reef wide mean percent benthic cover between methods. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs WFGB)	20.90	1	2.40	**0.05**
Res	392.28	45		
Total	413.18	46		

SIMPER analysis showed that differences between EFGB and WFGB study sites were driven primarily by macroalgae and colonizable substrate. Macroalgae was more abundant at EFGB, while colonizable substrate was higher at WFGB, accounting for nearly half of the overall dissimilarity between banks. SIMPER analysis indicated that macroalgae and coral were the main contributors to differences in the reef-wide surveys. Both groups were higher at EFGB, although overall dissimilarity between banks was less than that observed at the study sites.

Sixteen coral species were recorded in the EFGB study site surveys and 15 in the EFGB reef-wide surveys, while 15 species were observed in the WFGB study sites and 16 in the WFGB reef wide surveys. A few low-abundance species differed slightly between banks or survey types, for example, *Siderastrea radians* were only recorded at EFGB, while *Tubastraea* was only found in reef-wide surveys at both banks (Figure 2.4).

Orbicella franksi was the most abundant coral species at both banks, with mean cover of 21.50 ± 2.52% (EFGB study site), 30.36 ± 3.03% (EFGB reef-wide), 34.12 ± 2.67% (WFGB study site),

and 29.39 ± 3.81% (WFGB reef-wide). At EFGB, *Porites astreoides* was the second most abundant species at the study sites (7.79 ± 0.74%) while *Pseudodiploria strigosa* ranked second in the reef-wide surveys (5.68 ± 0.63%). At WFGB, *P. strigosa* was the second most common species in both survey types, with mean cover of 8.13 ± 1.74% (study site) and 6.54 ± 0.89% (reef-wide). The *Orbicella* species complex (*O. franksi, O. faveolata,* and *O. annularis*), listed as threatened under the Endangered Species Act (16 U.S.C. § 1531 *et seq.*), also contributed substantially to overall coral cover at both banks and across both survey approaches.

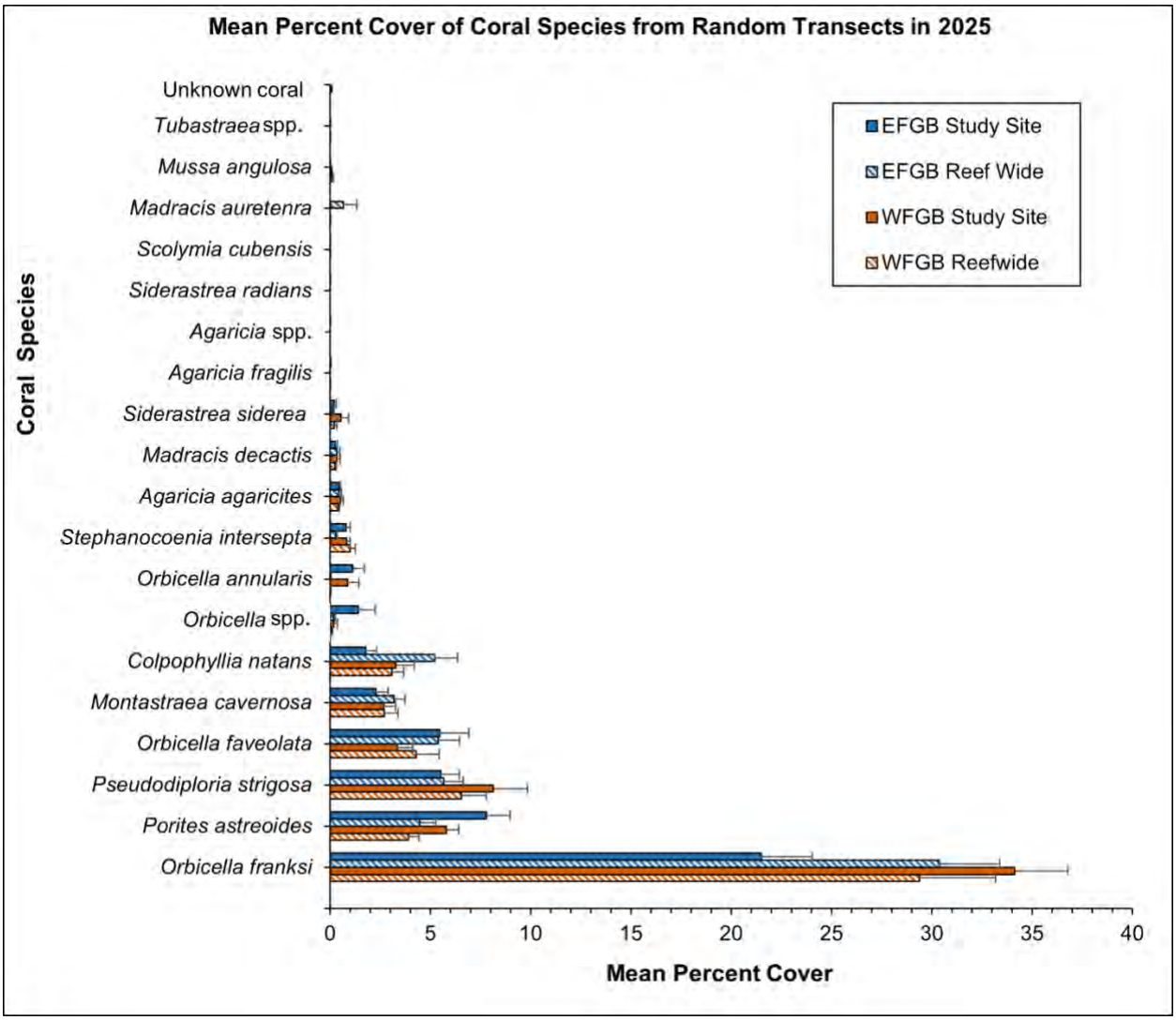


Figure 2.4. Mean percent cover + SE of observed coral species from random transect surveys within EFGB and WFGB study sites and reef-wide surveys in 2025.

A separate PERMANOVA was conducted using only coral species cover to test whether coral species composition differed by bank or survey type. No significant differences were detected for bank, survey type, or their interaction, indicating that coral species composition was broadly similar between EFGB and WFGB and between study site and reef-wide surveys (Table 2.5). These results show that differences observed in overall benthic community structure were driven primarily by non-coral benthic groups rather than shifts in coral species.

Table 2.5. PERMANOVA results comparing coral species mean percent cover from EFGB and WFGB random transects in 2025

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	14.06	1	1.35	0.22
Survey Type (Study Site vs. Reef wide)	17.24	1	1.65	0.12
Bank x Survey Type	19.04	1	1.82	0.09
Res	783.51	75		
Total	831.86	78		

Coral species diversity metrics were calculated for each sample and compared between banks within each survey method. At the study sites, ANOSIM detected a significant difference in coral diversity between EFGB and WFGB ($R = 0.274$, $p = 0.002$). In contrast, reef-wide surveys showed no detectable difference in diversity between the two banks ($R = -0.007$, $p = 0.49$; Table 2.6). These results indicate that coral diversity differs slightly between banks at the fixed long-term monitoring sites but is broadly similar across the wider reef systems. This pattern suggests that small-scale differences in coral diversity in the study sites do not extend to the broader reef.

Table 2.6. Mean coral species diversity measures ± SE from EFGB and WFGB random transect surveys in 2025.

Random Transect Coral Diversity Measures	EFGB Study Site	EFGB Reef-wide	WFGB Study Site	WFGB Reef-wide
Margalef's species richness (d)	2.04 ± 0.08	1.67 ± 0.07	1.64 ± 0.10	1.78 ± 0.08
Pielou's evenness (J')	0.67 ± 0.03	0.63 ± 0.03	0.64 ± 0.03	0.63 ± 0.04
Shannon diversity (H'(loge))	1.47 ± 0.08	1.29 ± 0.06	1.30 ± 0.07	1.23 ± 0.07

At EFGB, an average of 0.82 ± 0.43% of coral cover in the study site surveys was affected by bleaching, 3.98 ± 0.89% showed signs of paling, and less than 1% was impacted by fish biting. In the reef-wide surveys, bleaching affected 0.31 ± 0.14% of coral cover, 4.21 ± 0.90% showed paling, and fish biting remained below 1%.

At WFGB, bleaching affected an average of 0.06 ± 0.03% of coral cover in the study site surveys, with 2.61 ± 0.71% showing signs of paling and less than 1% impacted by fish biting. In the reef-wide surveys, bleaching affected 0.60± 0.46%, paling affected 3.42 ± 1.00%, and fish biting accounted for less than 1% of coral cover.

Fish biting that results in the removal of coral polyps is most commonly associated with damselfish gardening behavior or grazing by stoplight parrotfish (*Sparisoma viride*; Bruckner & Bruckner, 1998; Bruckner et al., 2000). Signs of recent mortality and coral disease are explored further in Chapter 3.

Random Transect Long-Term Trends

Mean percent coral cover in study sites (17–27 m) from 1989 to 2025 ranged from 45–67% at EFGB and 44–62% at WFGB. Coral cover increased significantly over time at the WFGB study site ($\tau = 0.42$ $p < 0.001$; 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 2025, macroalgae cover ranged between 10–38% and increased significantly in both the EFGB and WFGB study sites (τ= 0.45, $p < 0.001$; τ= 0.29, $p < 0.02$, respectively; Figure 2.5). Over the same period, colonizable substrate declined significantly at both EFGB (τ = -0.35, p= 0.007) and WFGB (τ = -0.41, p = 0.001). Hydrocoral cover declined significantly at both EFGB and WFGB from 1992 to 2025 (τ = -0.67, $p < 0.001$; τ = -0.42, $p < 0.001$, respectively; Figure 2.5). Sponge cover did not exhibit a statistically significant long-term trend at either bank, despite declines at EFGB reported in earlier analyses (O'Connell et al., 2026).

Macroalgae and colonizable substrate were strongly negatively correlated at EFGB (ρ = -0.83, $p < 0.001$) and WFGB (ρ = -0.88, $p < 0.001$), suggesting that macroalgae growth was preferentially occupying previously available space and not outcompeting other biotic cover.

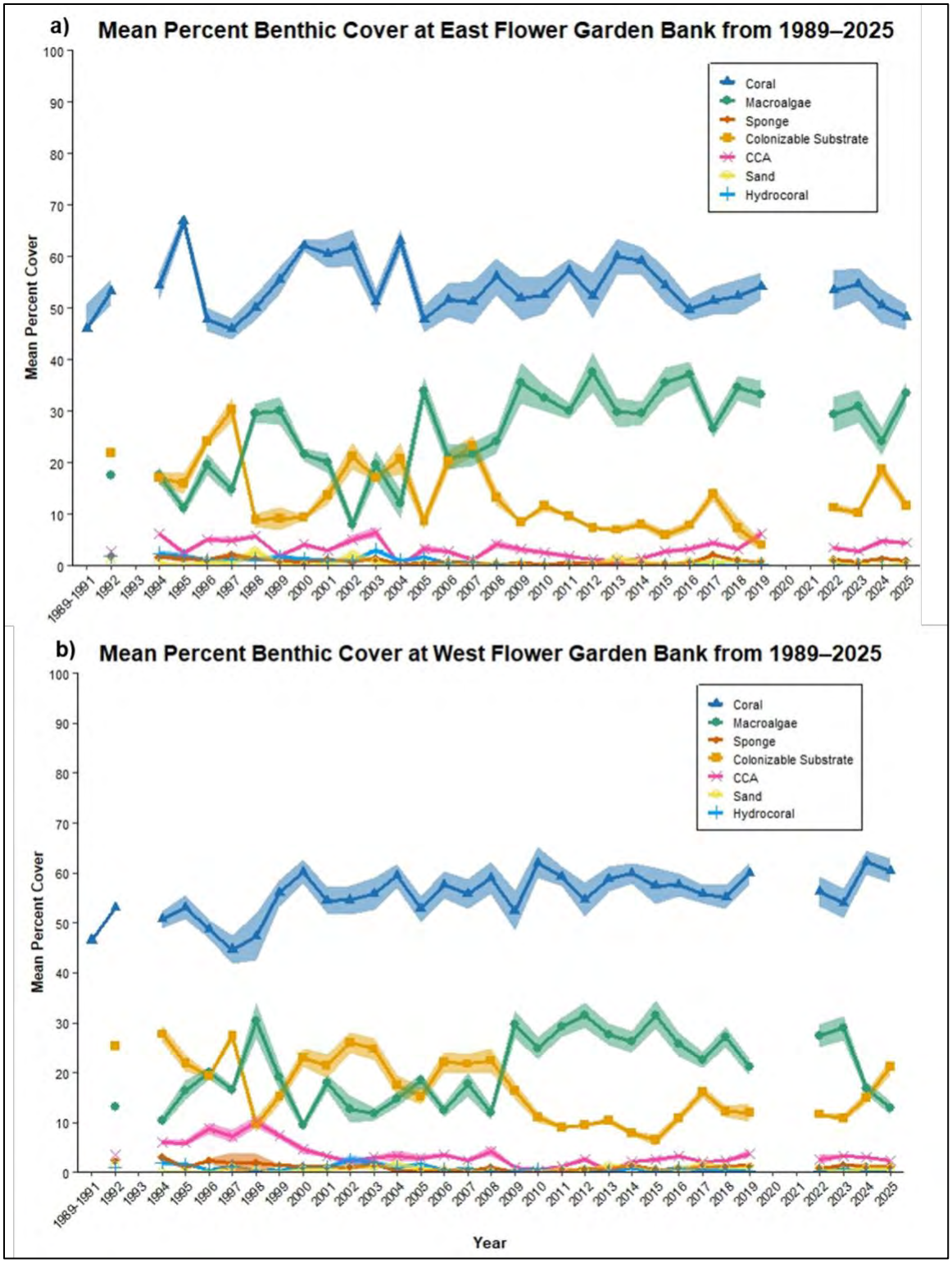


Figure 2.5. Mean percent benthic cover ± SE bands from random transect surveys within (a) EFGB and (b) WFGB study sites from 1989 to 2025. Only coral cover data were reported from 1989–1991 and no mean percent cover data were reported in 1993, 2020, or 2021. Sources: Gittings et al., 1992 (1989 to 1991); 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); O'Connell et al., 2026 (2024).

For available yearly mean benthic percent cover data (1992 to 2025), SIMPROF analysis detected four significant year clusters in the EFGB study site (1998 and 2019; 1999, 2005, 2009–2016, 2018, and 2023; 2008, 2017, 2022, and 2024–2025; 1992, 1994–1997, 2000–2004, and 2007 (Figure 2.7). SIMPER analysis identified macroalgae, colonizable substrate, and crustose coralline algae mean percent cover as consistent contributors to dissimilarity among all year group clusters.

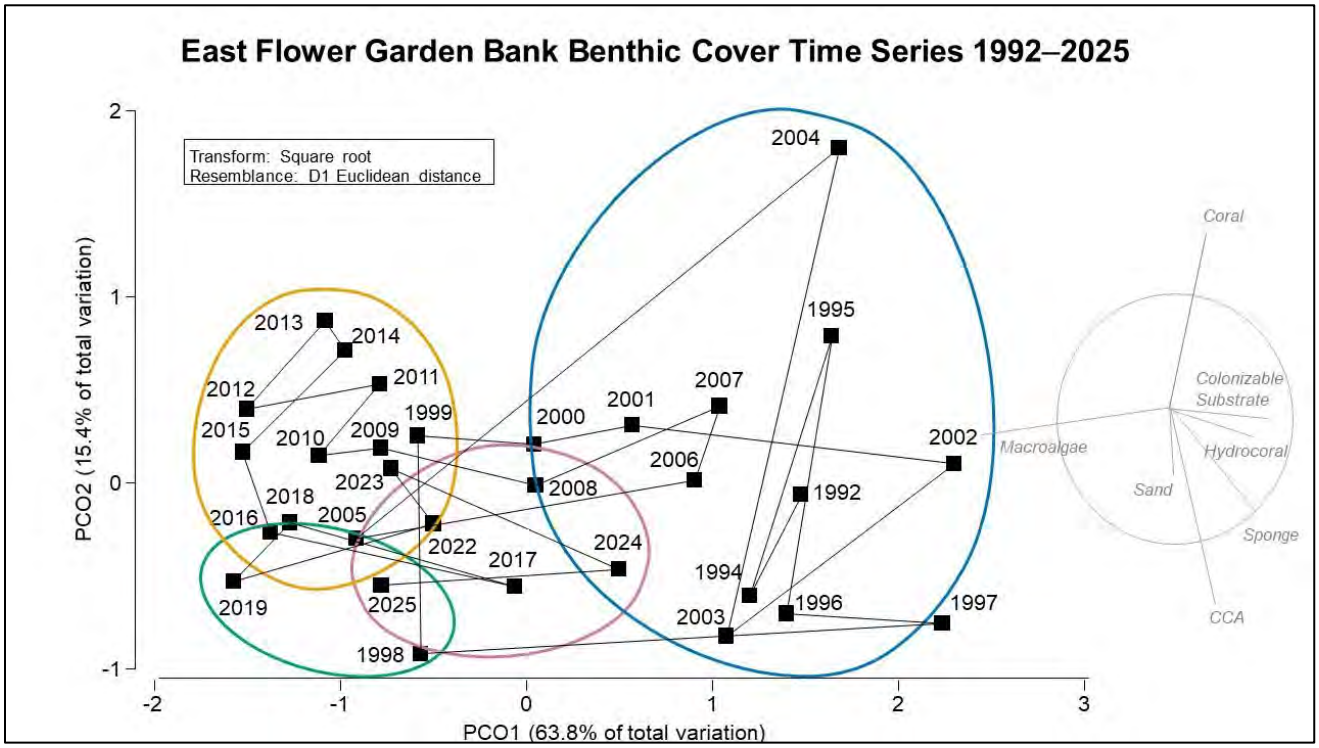


Figure 2.6. PCO plot for random transect benthic cover analysis within the EFGB study site from 1992–2025. The ovals are SIMPROF groups representing significant year clusters grouped by color. The gray vector lines represent the directions of the variable gradients for the plot.

Yearly mean benthic percent cover from 1992 to 2025 at the WFGB study site resulted in three significant year clusters (1992, 2000–2008, 2017, 2019, 2024, and 2025; 1994–1997, and 1999; and 1998, 2009–2016, 2018, 2022, and 2023; Figure 2.7). SIMPER analysis showed that macroalgae and colonizable substrate were the primary contributors to dissimilarity among year clusters at WFGB.

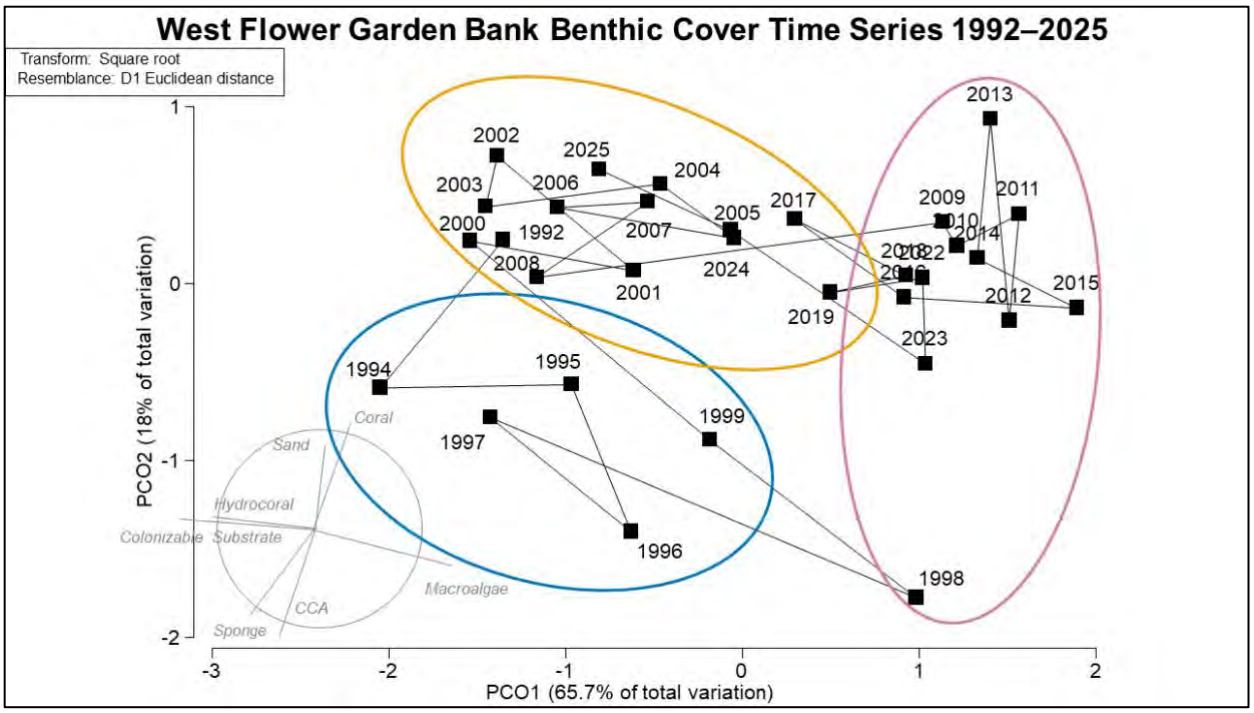


Figure 2.7. PCO plot for random transect benthic cover analysis within the WFGB study site from 1992–2025. The ovals are SIMPROF groups representing significant year clusters grouped by color. The gray vector lines represent the directions of the variable gradients for the plot.

PERMANOVA results revealed statistically significant differences in benthic community composition between EFGB and WFGB from 1992 to 2025 (Table 2.7). Significant variation was also observed across years, and a significant interaction between Bank and Year indicates that temporal patterns in benthic community composition differed between the two banks (Table 2.7). SIMPER analysis indicated that overall differences between EFGB and WFGB were relatively modest (average dissimilarity = 16.65%). Dissimilarity between banks was primarily driven by macroalgae (21.86%) and colonizable substrate (19.44%), followed by coral cover (15.43%) and crustose coralline algae (13.13%). Macroalgal cover was slightly higher at EFGB, whereas colonizable substrate and sand were more abundant at WFGB.

Table 2.7. PERMANOVA results comparing random transect benthic percent cover from EFGB and WFGB random transects from 1992–2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank (EFGB vs. WFGB)	102.90	1	12.69	**0.0001**
Year	1489.00	30	13.51	**0.0001**
Bank x Year	269.73	30	2.45	**0.0001**
Res	3432.10	934		
Total	5339.60	995		

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 (Table 2.8) and WFGB (Table 2.9), indicating pronounced interannual change in benthic cover at both sites.

Table 2.8. PERMANOVA results comparing random transect benthic percent cover at EFGB random transects from 1992–2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	903.16	30	8.18	**<0.001**
Res	1752.90	476		
Total	2656.10	506		

Table 2.9. PERMANOVA results comparing random transect benthic percent cover at WFGB random transects from 1992–2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	850.74	30	7.74	**<0.001**
Res	1679.20	458		
Total	2529.90	488		

Beta-diversity analysis, quantified as multivariate dispersion (PERMDISP) based on coral species presence/absence data, indicated significant interannual variation in coral species composition at both EFGB ($F = 4.16$, $p = 0.0001$) and WFGB ($F = 2.86$, $p = 0.0003$). However, there was no significant directional trend at either bank, indicating that while variability fluctuated among years, variation was likely a reflection of survey method.

Repetitive Photostation Mean Percent Cover

In 2025, coral was the dominant benthic cover in repetitive photostations at both EFGB and WFGB (Figure 2.8). Repetitive photostations are surveyed annually and provide a long-term record of benthic community change at fixed locations within the EFGB and WFGB study sites. Observations from 2025 were consistent with patterns documented in previous monitoring years, with coral remaining the dominant benthic group across most stations. Long-term trends in major benthic groups observed in study site repetitive photostations from 1989 through 2025 are shown in Figure 2.8.

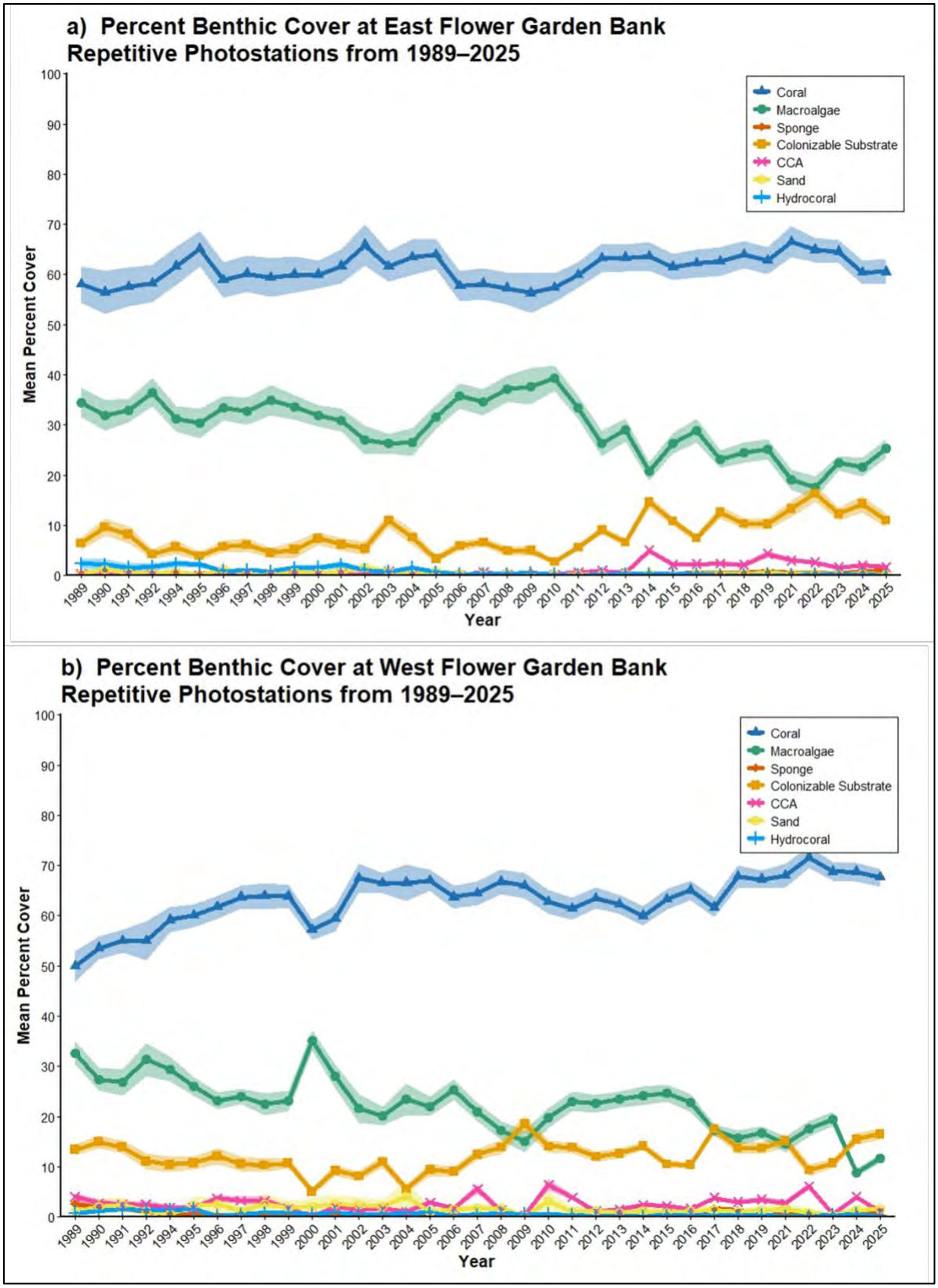


Figure 2.8. Mean percent benthic cover + SE within EFGB (a) and WFGB (b) study site repetitive photostations in from 1989–2025.

Mann–Kendall trend tests conducted for repetitive photostations within the study site at EFGB indicated significant increases in coral ($\tau = 0.31$, $p = 0.009$), crustose coralline algae (CCA; $\tau = 0.45$, $p < 0.001$), and colonizable substrate ($\tau = 0.41$, $p < 0.001$) over the monitoring period. In contrast, hydrocoral ($\tau = -0.71$, $p < 0.001$), macroalgae ($\tau = -0.45$, $p < 0.001$), and sand cover ($\tau = -0.35$, $p = 0.003$) exhibited significant declines over time. Sponge cover did not display a significant trend ($\tau = 0.14$, $p = 0.23$).

Mann–Kendall trend tests conducted for WFGB indicated a significant increase in coral cover ($\tau = 0.52$, $p < 0.001$) over the monitoring period. In contrast, hydrocoral ($\tau = -0.52$, $p < 0.001$) and macroalgae ($\tau = -0.57$, $p < 0.001$) exhibited significant declines through time. Colonizable substrate ($\tau = 0.18$, $p = 0.13$), crustose coralline algae (CCA; $\tau = -0.03$, $p = 0.81$), and sponge cover ($\tau = -0.09$, $p = 0.46$) showed no significant trends, while sand cover exhibited a marginal declining trend ($\tau = -0.23$, $p = 0.057$).

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 2025. These stations represent the subset of repetitive photostations that have been consistently surveyed throughout the monitoring record and therefore allow direct comparisons of coral cover at the same locations through time. Mean percent coral cover at EFGB photostations increased from 58.00 ± 3.67% in 1989 to 61.91 ± 2.93% in 2025; however, this change was not statistically significant (paired t-test, $p = 0.27$). At WFGB photostations, mean percent coral cover increased from 49.95 ± 2.99% in 1989 to 68.24 ± 2.31% in 2025, exhibiting a significant increase in coral cover at these stations (paired t-test, $t = 5.03$, $p < 0.001$; Figure 2.9).

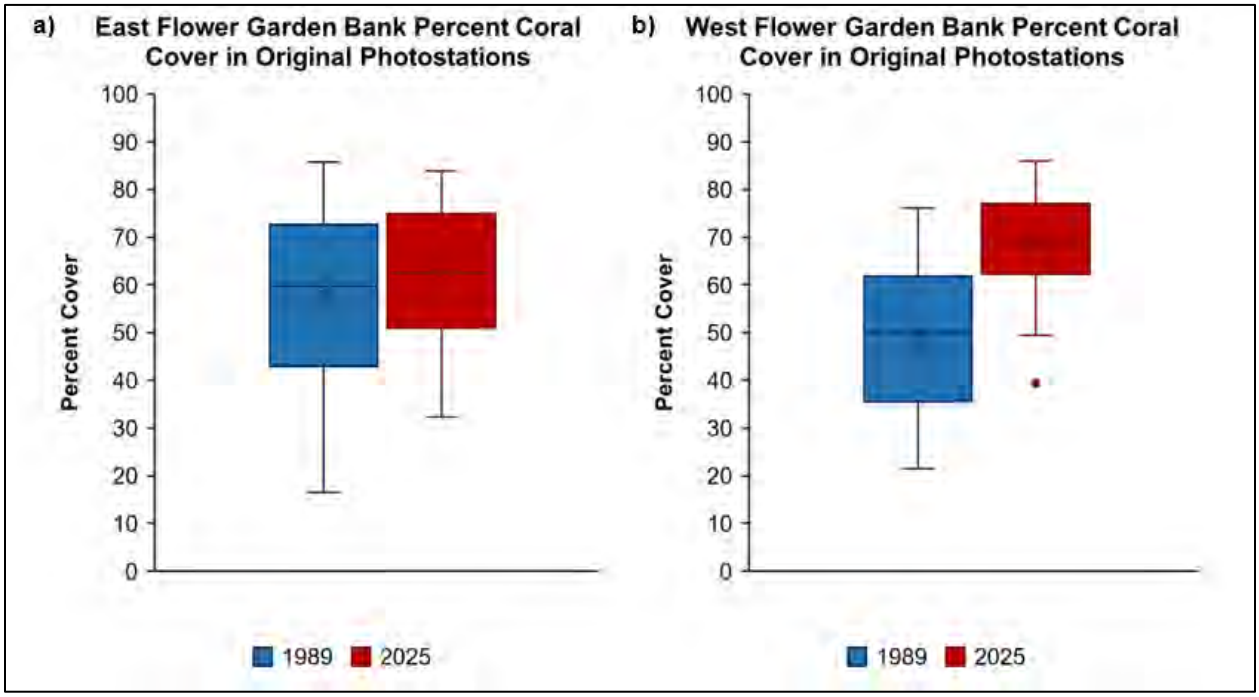


Figure 2.9. Box plot depicting percent coral cover in (a) EFGB (n = 24) and (b) WFGB (n = 27) repetitive photostations in 1989 and 2025.

Figure 2.10 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, ambient conditions, and colony health or condition. Furthermore, photo quality is affected by time of day, camera settings, and lighting. In EFGB photostation #102, changes from 1989 to 2025 include recruitment and growth of *P. strigosa* and *P. astreoides* in the center (Figure 2.10a; Figure 2.10b). 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 2025, and a black *Ircinia strobilina* sponge that was present in 1989 was absent in 2025 (Figure 2.10c; Figure 2.10d). In WFGB photostation #503, a large *C. natans* colony present in 1989 was absent in 2025 (Figure 2.10e; Figure 2.10f).

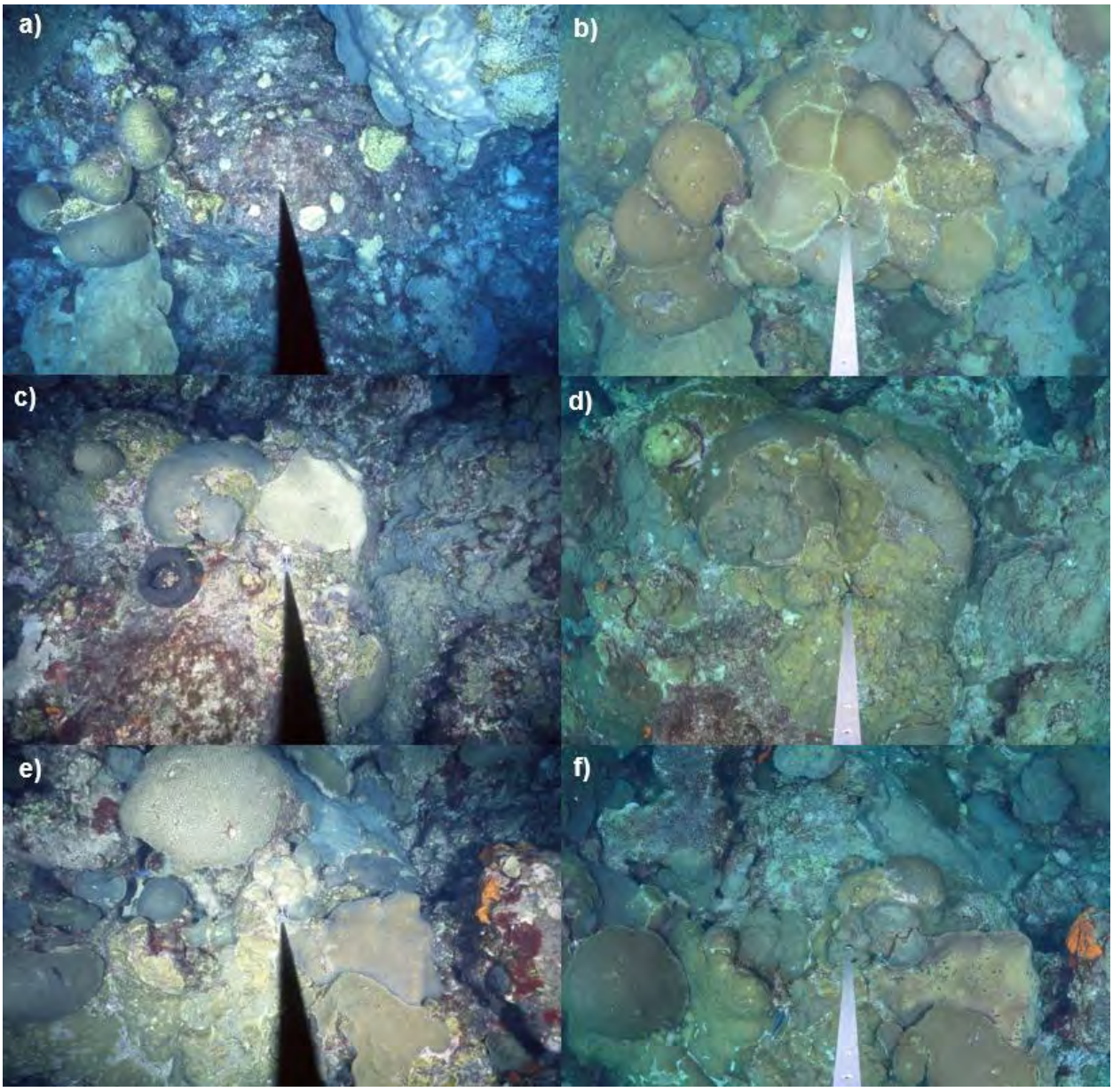


Figure 2.10. Time series of three repetitive photostations: EFGB photostation #102 (20 m) from (a) 1989 to (b) 2025; WFGB photostation #501 (20 m) from (c) 1989 to (d) 2025; and WFGB photostation #503 (20 m) from (e) 1989 to (f) 2025. Photos: (a, c, e) Minerals Management Service (Gittings et al., 1992); (b, d, f) NOAA

Additional repetitive photostations (32–39 m) located along the reef slope were established during multiple monitoring expansions and therefore do not provide a continuous time series across the full monitoring record. As a result, these deeper stations were not included in the long-term trend analyses presented above but were examined separately to evaluate differences in coral assemblages across depth zones. Analyses conducted throughout the long-term monitoring program have consistently identified significant differences in coral species mean percent cover across depth zones within the repetitive photostations at both EFGB and WFGB (Johnston et al., 2020; Johnston et al., 2021; Eisenbach et al., 2025). PERMANOVA analyses repeatedly

demonstrate a significant relationship between coral species cover and depth, indicating that coral community structure varies across the depth gradient represented by the photostations. Differences between shallow and deep photostations are driven primarily by shifts in coral species composition, with deeper photostations characterized by increased cover of *Montastraea cavernosa* and reduced cover of species such as *Pseudodiploria strigosa* and *Orbicella annularis* relative to shallower photostations (Johnston et al., 2020; Johnston et al., 2021).

Discussion

Despite widespread regional coral cover declines (Wicquart et al., 2025), coral cover within EFGB and WFGB study sites remained high in 2025. 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 36-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 the 36-year monitoring period 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 revealed significant differences between the two banks. These differences were driven by shifts in benthic groups other than coral, particularly macroalgae and colonizable substrate.

Trends observed in the repetitive photostation dataset were somewhat different than patterns documented in the random transect monitoring dataset. These differences likely reflect methodological contrasts; repetitive photostations were established in high-coral areas within fixed coral-dominated patches, to track change where coral growth may reduce space available for other benthic groups. In comparison, random transects capture broader spatial variability across the study site. Coral cover increased at both banks in the repetitive photostation dataset, whereas random transect surveys indicated increasing coral cover at WFGB but relatively stable coral cover at EFGB. Hydrocoral declined significantly at both banks in both datasets. Macroalgae and colonizable substrate exhibited opposing patterns: random transects showed increasing macroalgae and declining colonizable substrate, while repetitive photostations indicated declining macroalgae and increasing colonizable substrate at EFGB, with weaker trends at WFGB.

As in previous years, *O. franksi* was the dominant species at both banks, with *P. strigosa, O. faveolate*, and *P. astreoides* ranking as secondary predominant species. Beta-diversity analyses indicated that coral community variability fluctuated among years at both EFGB and WFGB, but no long-term directional trend in multivariate dispersion was detected at either bank. This suggests that overall, community variability across the study sites has remained relatively stable over the monitoring period. However, the short-term temporal variability in coral community composition likely reflects insufficient sampling power rather than a sustained shift in community structure. Continued monitoring will help determine whether future disturbances or environmental changes alter this long-term stability in coral community variability. The

observed variability also highlights the need to either increase sampling effort or evaluate alternative methods to improve detection of long-term trends in coral community composition.

Macroalgae remained one of the most abundant benthic groups in random transects at both banks in 2025, 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 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 or other benthic groups. 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).

Sixteen coral species were recorded in random transects at both banks in 2025. Coral diversity analyses indicated a significant difference between banks at the study sites, whereas reef-wide surveys detected no difference in diversity between EFGB and WFGB. This pattern suggests that small-scale differences in coral diversity occur within the long-term monitoring stations but do not extend across the wider reef systems. 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 support extensive, high-cover coral reef communities, their relatively low coral species richness compared to many tropical Caribbean reefs may reflect a combination of environmental factors, including their location near the northern latitudinal limit of reef development, the greater depths of reef-cap habitats and associated light limitations, and periodic disturbance from tropical storms and hurricanes (Bright et al., 1985; Aronson et al., 2005; Lugo-Fernández & Gravois, 2010). Although the coral species present on the reef caps are broadly shared with Caribbean reefs, the shallow reef-cap communities at EFGB and WFGB differ from many Caribbean reef systems in that shallow-water octocorals (gorgonians) are absent, and Acropora corals are exceedingly rare (Bright et al., 1985; Aronson et al., 2005; Wicquart et al., 2025).

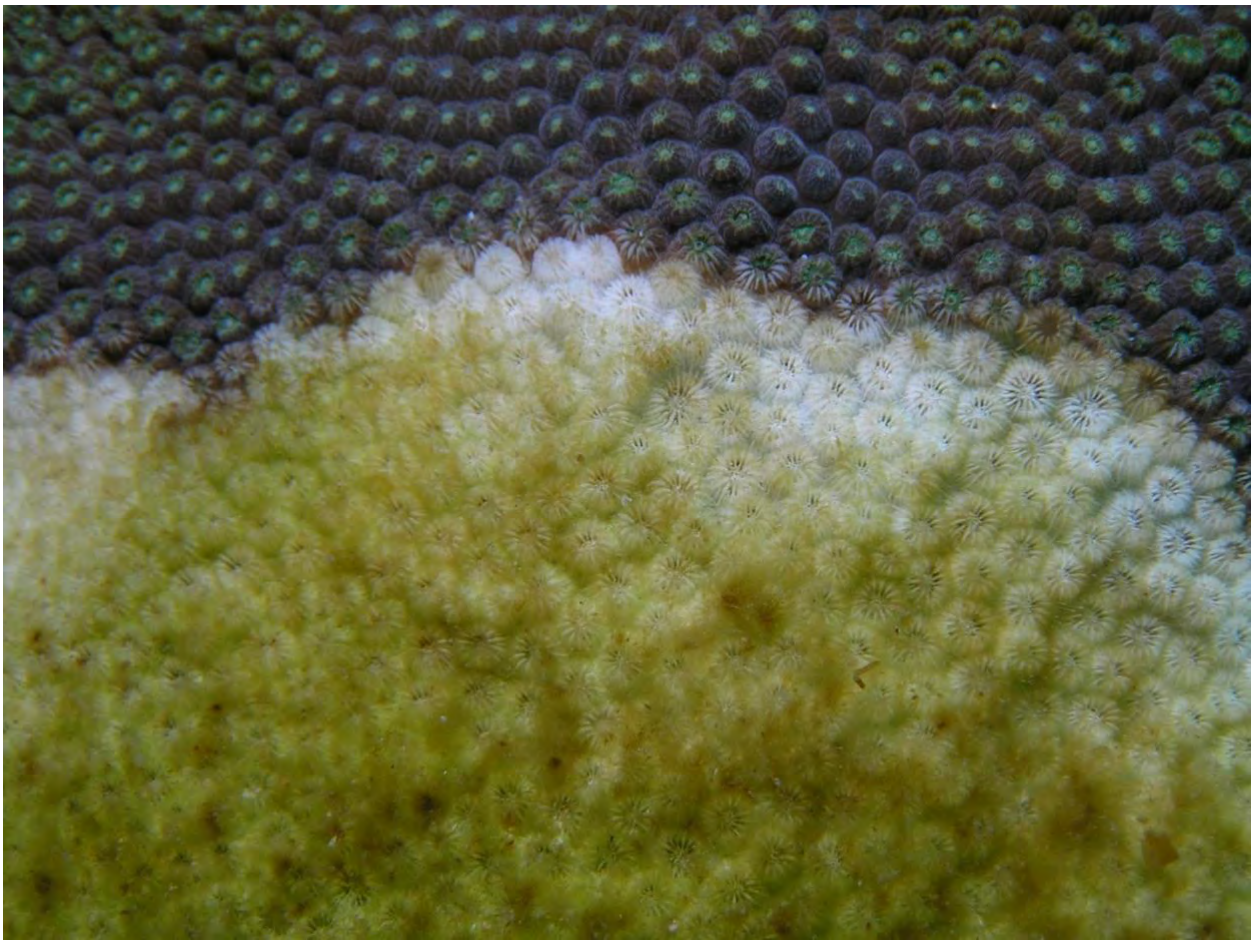
Despite exposure to disturbances such as coral bleaching, disease outbreaks, and hurricanes, coral cover within EFGB and WFGB study sites has remained stable or gradually increased since monitoring began in 1989. Coral cover at these sites is currently estimated to be six to 11 times higher than in selected locations across the Caribbean (Caldow et al., 2009; Clark et al., 2014; Jackson et al., 2014; Johnston et al., 2017a, 2017b). This high and persistent coral cover 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). 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.) While the offshore setting and deep-water influence at EFGB and WFGB likely contribute to the exceptional condition of these reefs, the presence of similarly high coral cover in some nearshore systems in the southern Gulf suggests that these factors alone do not fully explain the persistence of healthy reefs at FGBNMS.

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, 2023, and 2024. 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). 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). In this context, the relatively limited coral mortality observed at EFGB and WFGB to date suggests a degree of resilience, but continued monitoring will be critical to determine whether this resilience can be sustained under repeated and intensifying thermal stress.

Although EFGB and WFGB benefit from their depth and isolation, they are not immune to broader impacts associated with environmental change and human activities, including marine heatwaves and coral bleaching (Office of National Marine Sanctuaries, 2008; Johnston et al., 2019; Eisenbach et al., 2025), disease outbreaks (Johnston et al., 2018b; Johnston et al., 2023; O'Connell et al., 2026), invasive species (Nuttall et al., 2014), increased river discharge and coastal runoff following major storm events, and other water-quality stressors (Office of National Marine Sanctuaries, 2024). 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 and Disease



An active disease lesion causes tissue loss on an *Orbicella* colony. Photo: Emma Hickerson/NOAA

Coral Bleaching and Disease Introduction

Mild to moderate coral bleaching events have been documented at FGBNMS in previous years (e.g., 1990, 1995, 2005, 2010, 2016, 2021, 2023, and 2024) 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; Eisenbach et al., 2025; O'Connell et al., 2026). 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.

Historically, coral disease has been infrequently observed at EFGB and WFGB, and the reefs have exhibited relatively low levels of disease compared to many Caribbean reef systems (Borneman and Wellington, 2005; Johnston et al., 2023). However, as disease outbreaks have become increasingly common throughout the wider Caribbean (Wicquart et al., 2025), an unusual disease event affecting multiple coral species was documented at both banks in 2022 (Johnston et al., 2023). Disease lesions have continued to be observed during subsequent annual monitoring efforts (Eisenbach et al., 2025; O'Connell et al., 2026). As a result, documenting the prevalence of disease lesions remains an important component of annual reef assessments.

In 2025, disease-like lesions were observed across multiple coral species at East and West Flower Garden Banks (EFGB and WFGB) during annual surveys. Repetitive photostations remained critical for documenting site-specific changes in coral health and fate tracking of partial mortality, tissue loss, and recovery. Repetitive photostations were photographed in July, when early signs of thermal stress, including some bleaching, paling, and disease lesions were observed in corals. These observations provide context for evaluating patterns in coral condition across sites. Repetitive photostations and study site random transects were conducted July 27–31, 2025, followed by reef-wide transects from August 4–8, 2025. No dedicated bleaching surveys were conducted in 2025; therefore, assessments of coral condition are based on these surveys and may not capture peak bleaching conditions.

Coral Bleaching and Disease Methods

Coral stress was assessed by measuring the prevalence of bleaching, paling, and active disease lesions. 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, paling, or disease. 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 any point landing on pale coral was classified as "paling." Data were averaged across EFGB and WFGB and calculated for repetitive photostations and random transects.

Prevalence by number of colonies affected was calculated for repetitive photostations at EFGB and WFGB by dividing the number of colonies exhibiting paling, bleaching, or active disease

lesions by the total number of colonies observed. Disease was defined as visible tissue loss consistent with marginal lesions or multifocal tissue loss. Prevalence was calculated for each photostation as the percentage of coral colonies exhibiting signs of disease. To evaluate relationships with environmental gradients, disease prevalence was analyzed using a linear model with depth treated as a continuous covariate and bank as a fixed factor. This approach allowed for assessment of depth-related trends while accounting for potential differences between banks. To determine prevalence change in depth, corals were binned by depth. Depth bins represent photostations ≤22 m, 22–25 m, 25–31 m, and >31 m.

Qualitative condition of repetitive photostations was assessed using a five-category classification system to characterize overall changes in coral and sponge condition. Each photostation was assigned a condition category based on visual assessment of dominant benthic organisms using the following criteria:

- 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 ≥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 ≥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.

This classification was used to assess site-level condition and is not intended to provide quantitative estimates of prevalence.

Bleaching thresholds developed for EFGB and WFGB (Johnston et al., 2019; Eisenbach et al., 2025; O'Connell et al., 2026) were used to evaluate thermal stress in 2025. These thresholds are based on the relationship between temperature and duration of exposure and define the number of days above site-specific temperature limits associated with past bleaching years. The number of days above these thresholds was calculated for each bank and compared to previously defined bleaching conditions.

Coral Bleaching and Disease Results

Prevalence by Percent Cover

Analysis of percent coral cover exhibiting bleaching and paling across survey types in 2025 indicated generally low levels of bleaching and paling at both EFGB and WFGB (Figure 3.1). At EFGB, bleaching remained low across all survey methods, ranging from 0.31 ± 0.14% at reef-wide transects to 0.82 ± 0.42% at study site transects. In contrast, paling was more prevalent, with values of 3.98 ± 0.89% at study site transects, 4.21 ± 0.90% at reef-wide transects, and 3.46

± 0.45% at repetitive photostations. At WFGB, bleaching was similarly low across all survey types (<1%), ranging from 0.02 ± 0.02% at repetitive photostations to 0.60 ± 0.46% at reef-wide transects. Paling was variable, with 2.61 ± 0.71% at study site transects, 3.42 ± 1.00% at reef-wide transects, and 3.48 ± 0.81% at repetitive photostations.

Relative to 2024, percent coral cover affected by bleaching and paling was substantially lower across both banks in 2025, particularly at WFGB where bleaching and paling reached peak values of 10.3% and 46.9%, respectively, during September surveys. However, September surveys, when bleaching typically peaks, were not conducted in 2025, limiting direct comparison.

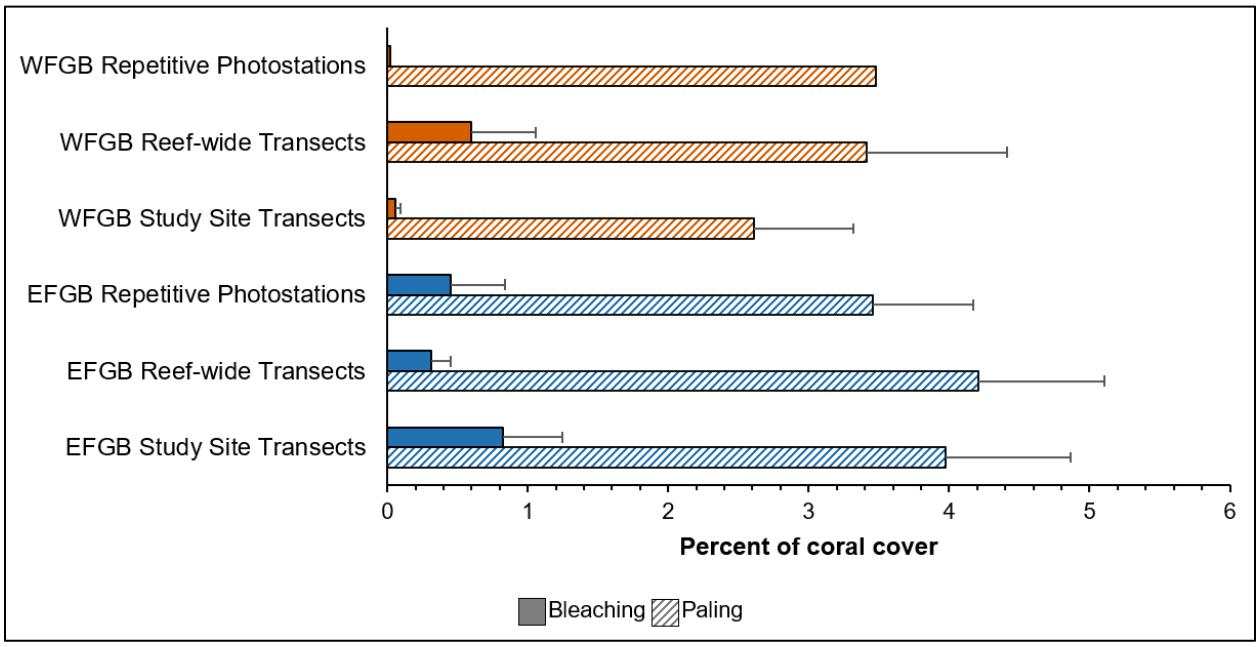


Figure 3.1. Combined percent of coral cover affected by bleaching or paling at EFGB and WFGB in 2025 from repetitive photostations, study site random transects, and reef-wide random transects.

Prevalence by Colony

The prevalence of coral paling, bleaching, and with active disease lesions was quantified as the proportion of affected colonies within repetitive photostations at EFGB and WFGB (Figure 3.2). Across both banks, bleaching prevalence remained consistently low (<1%) across all depths, while variation in condition was primarily driven by differences in paling and disease.

Disease prevalence increased significantly with depth across photostations ($p < 0.001$), with no significant differences detected between banks. This pattern was reflected across depth bins, with disease prevalence generally increasing from the shallowest (≤22 m) to deeper bins (25–31 m and >31 m) at both banks. In contrast, neither bleaching nor paling exhibited significant relationships with depth or bank ($p > 0.05$), and patterns across depth were variable and did not exhibit a consistent directional trend.

Relative to 2024, bleaching and paling prevalence at WFGB were lower in 2025, declining from approximately 12% during September surveys in 2024 to <4% for paling and <1% for bleaching

in 2025. Disease prevalence at photostations in 2025 was also lower at WFGB (decreasing from 4.7% in 2024 to 4.04%) and similar at EFGB (3.2% ± 0.6% in 2024 vs. 3.14% ± 0.72% in 2025).

Direct interannual comparisons are limited, as peak bleaching surveys (typically conducted in September) were not completed in 2025, and deeper photostations extending to ~40 m were not surveyed in 2024. As a result, elevated disease prevalence observed in deeper bins represents a newly documented pattern rather than a direct year-to-year comparison.

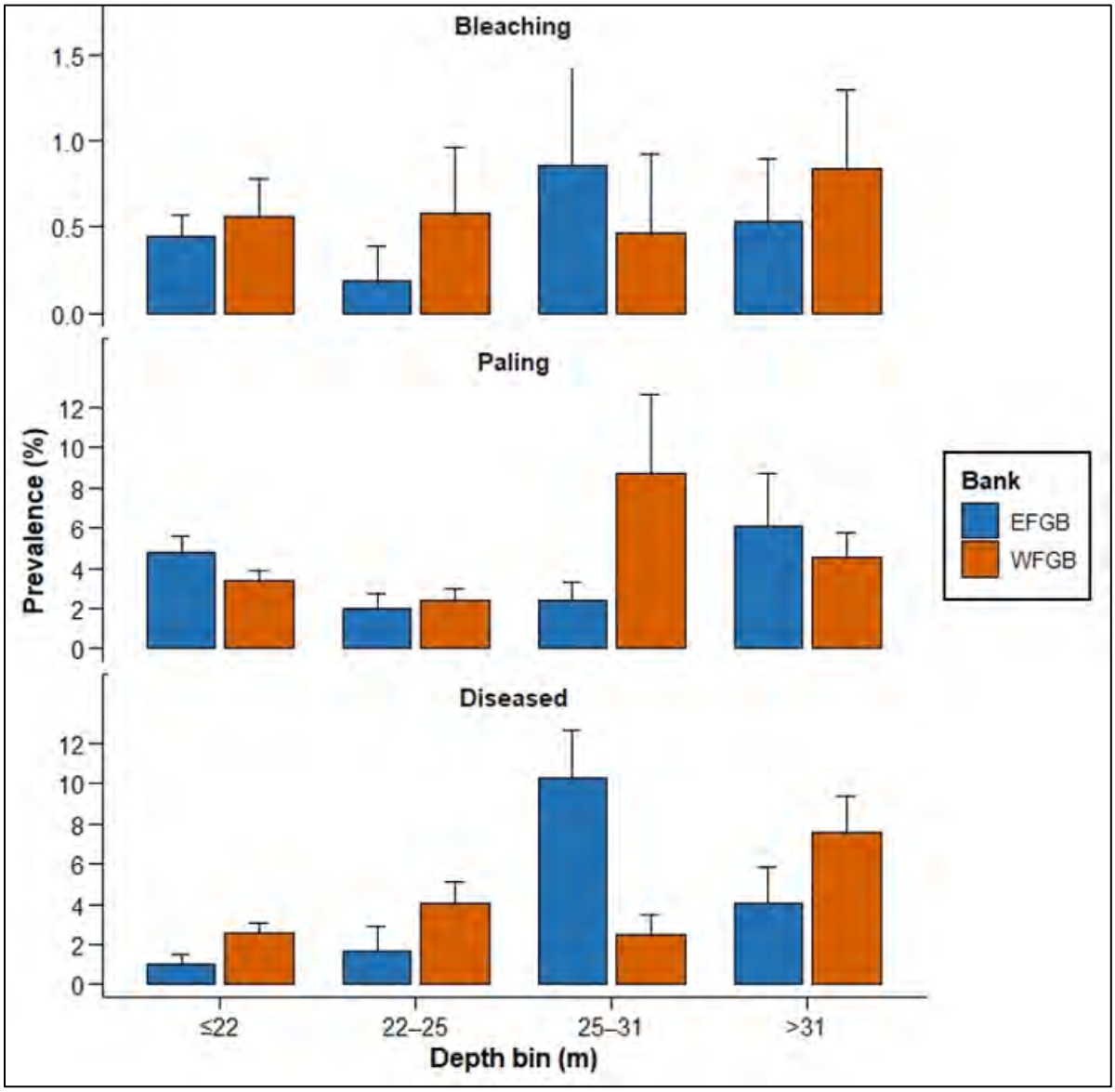


Figure 3.2. Mean (± SE) prevalence of coral paling, bleaching, and with active disease lesions across depth bins at East Flower Garden Bank (EFGB) and West Flower Garden Bank (WFGB) in 2025. Prevalence is expressed as the percent of colonies exhibiting each condition.

Qualitative Analysis of Repetitive Photostations and Coral Disease

Qualitative photostation assessments indicated variable site conditions across both banks, with stations classified as stable, improved, or degraded based on observed changes in coral and

sponge condition. Many stations exhibited a combination of growth and localized degradation rather than uniform changes in condition. Stations classified as stable or improved generally showed increases in colony size or regrowth into previously affected areas, including recovery from prior disease lesions or fish-biting damage. In contrast, degraded stations were associated with active disease lesions, recent partial mortality, or expansion of damselfish gardening, often resulting in localized tissue loss.

The most frequently affected species included *Orbicella* spp., *Porites astreoides*, and *Montastraea cavernosa*, with some impacts observed in *Pseudodiploria strigosa* and *Colpophyllia natans*. Disease expression was typically characterized by marginal tissue loss or small multifocal lesions, although several colonies exhibited larger areas of partial mortality affecting dominant species. Disease occurrence was patchy across photostations. Many stations exhibited small lesions with limited progression, while others showed more extensive tissue loss affecting individual colonies. In several cases, previously documented lesions showed little additional progression or evidence of recovery, and some colonies exhibited regrowth into previously affected areas.

Damselfish gardening was frequently observed and was commonly associated with localized tissue loss. In some cases, expansion of gardening areas resulted in substantial mortality within individual colonies or spread to adjacent colonies. Gardening activity was observed both independently and in areas previously affected by disease. Despite these stressors, many stations showed evidence of continued coral growth, with colonies expanding into available space and persistence of smaller colonies observed across sites.

From 2024 to 2025, qualitative conditions at both banks shifted away from coral degradation and towards stations classified as improved or unchanged (Figure 3.3). At EFGB, the proportion of stations classified as improved increased from 21.6% to 32.4%, while coral degradation declined from 37.8% to 24.3%. Stations classified as unchanged decreased from 35.1% to 27.0%, and the proportion of stations exhibiting both coral and sponge degradation declined from 5.4% to 2.7%. In contrast, sponge-only degradation, which was not observed in 2023–2024, was present in 2024–2025 (13.5%).

At WFGB, similar patterns were observed from 2024 to 2025 (Figure 3.3). Stations classified as improved increased from 4.8% to 32.5%, while coral degradation declined from 35.7% to 25.0%. The proportion of unchanged stations decreased from 59.5% to 32.5%. Sponge-only degradation, absent in 2023–2024, was observed in 2024–2025 (7.5%), and concurrent coral and sponge degradation was present at low levels in 2025 (2.5%). Overall, these shifts indicate a reduction in coral degradation and an increase in stations exhibiting stability or improvement relative to 2024 (Figure 3.4). However, sponge-only degradation emerged at both banks in 2025, which was not observed in 2024.

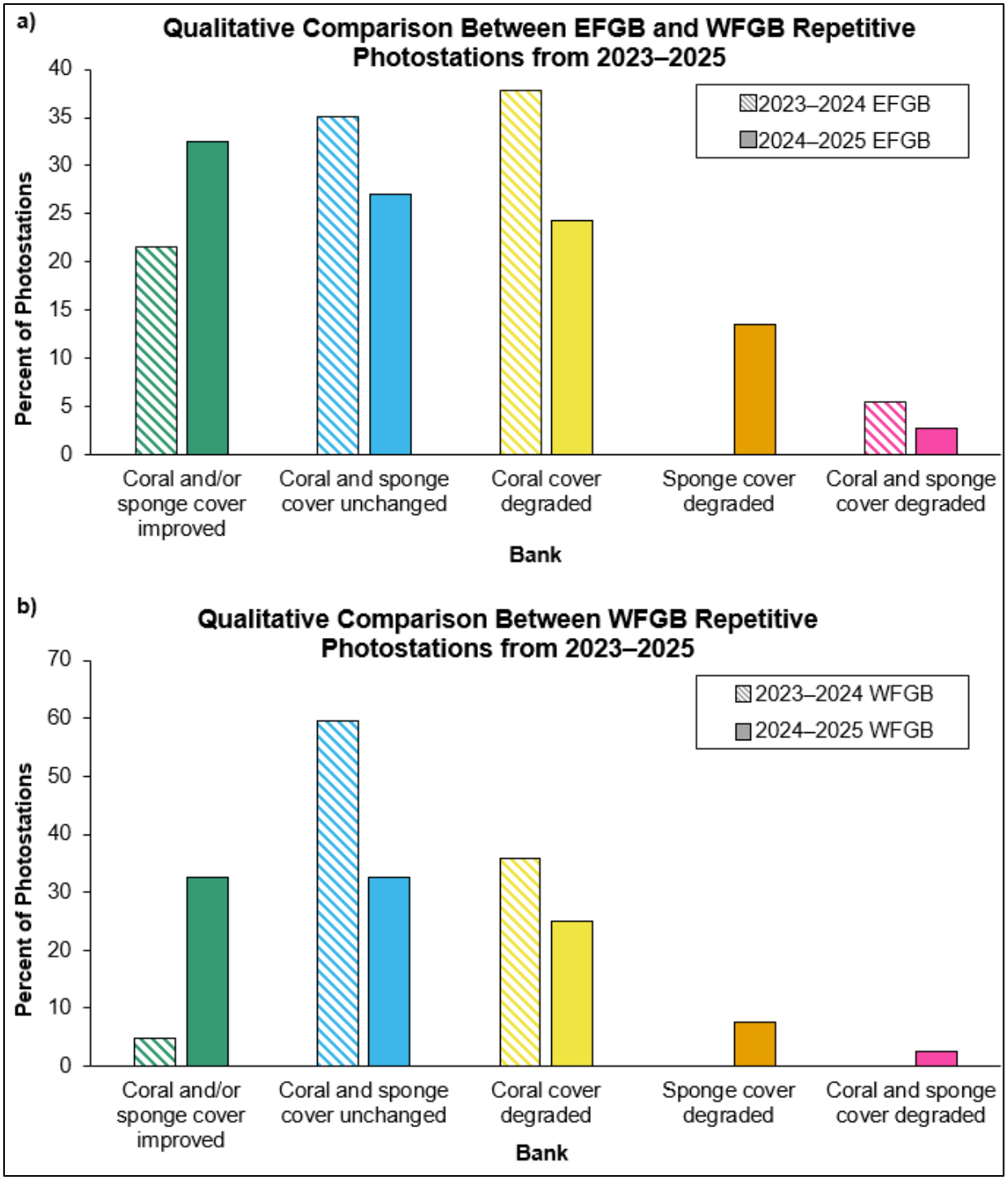


Figure 3.3. Percent of photostations by condition category, illustrating changes in coral and sponge cover at EFGB (a) and WFGB (b) from 2023–2025 based on qualitative assessments.

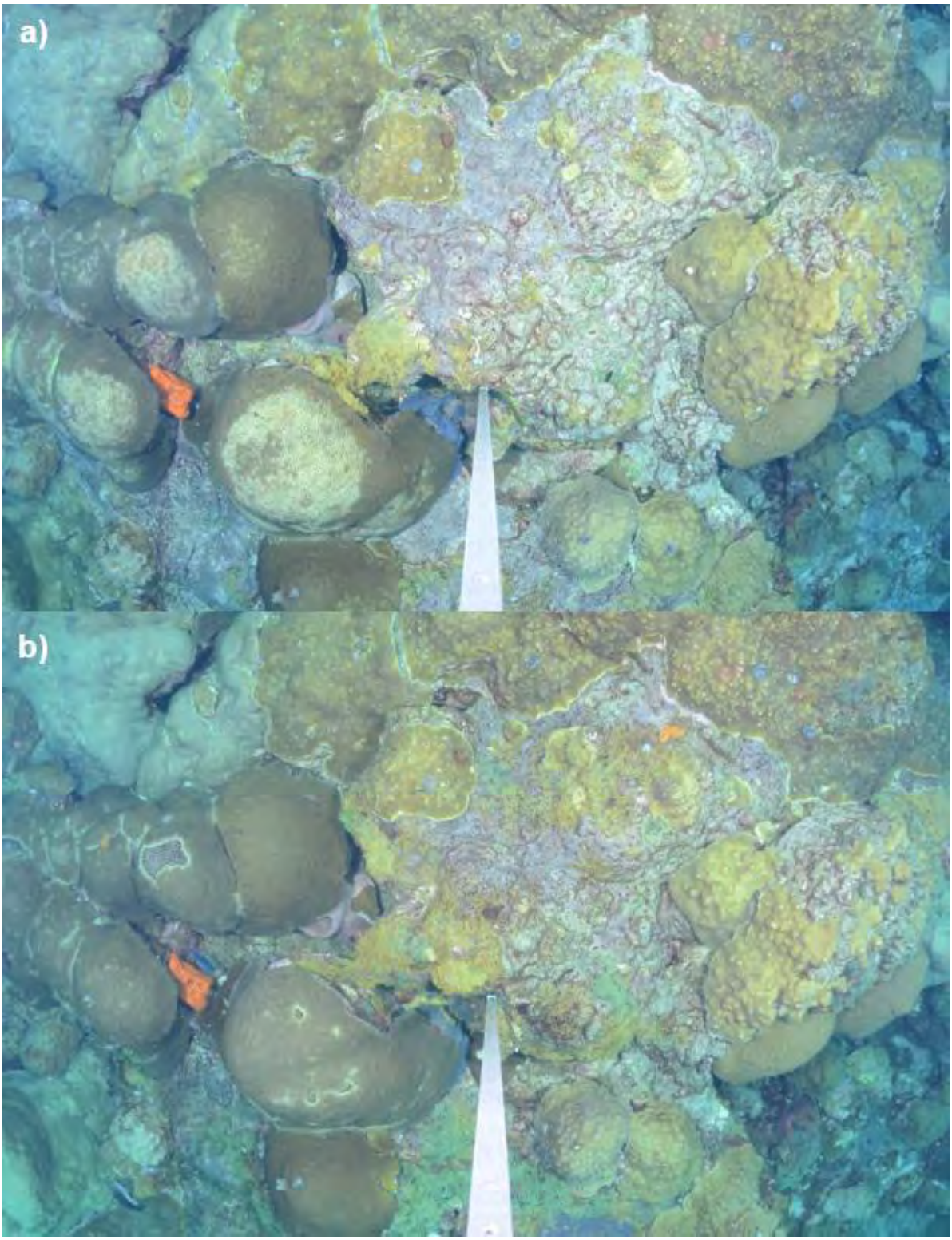


Figure 3.4. EFGB repetitive photostation 103 in (a) 2024 and (b) 2025. The *Pseudodiploria strigosa* colonies exhibited bleaching paling in 2024 (top) and returned to normal pigmentation by 2025 (bottom), indicating recovery from thermal stress. Photos: Donavon French/CPC

Bleaching Thresholds

Bleaching threshold curves developed for FGBNMS (Johnston et al., 2019; Eisenbach et al., 2025; O'Connell et al., 2026) illustrate the combined importance of temperature and duration of exposure in triggering coral bleaching. When applied to long-term temperature records, these thermal stress indices effectively segregate bleaching years (2010, 2016, 2021, and 2023) from non-bleaching years at EFGB and WFGB (O'Connell et al., 2026). At EFGB, bleaching has historically been associated with ≥33 days above 30 °C or ≥59.3 days above 29.5 °C, while at WFGB, lower thresholds have been observed (≥22.3 days above 30 °C or ≥28.8 days above 29.9 °C).

In 2025, in situ sensors at EFGB (23 m; HOBO temperature logger and Sofar buoy) recorded 84 non-consecutive days above 29.5 °C and 36 days above 30 °C. The Sofar buoy detached during Hurricane Francine in September 2024 and was not reinstalled until February 2025; however, it was deployed throughout the high-temperature period and captured most of the elevated temperature conditions. These observations indicate that corals at EFGB experienced prolonged thermal stress in 2025, despite relatively low observed bleaching during survey periods. However, the full extent of bleaching remains uncertain, as dedicated surveys were not conducted during peak thermal stress in September.

At WFGB, Sea-Bird® Electronics 16plus (SBE 16plus) sensors and HOBO loggers at 27 m recorded 20 and 21 days above 29.5 °C, respectively, and 20 and 19 days above 30 °C. These values fall below established bleaching thresholds for WFGB, indicating that thermal stress was likely insufficient to trigger widespread bleaching in 2025.

Discussion

Thermal stress conditions associated with the 2023–2024 global bleaching event (Reimer et al., 2024; Doherty et al., 2025; Wicquart et al., 2025) provide important context for interpreting coral condition at EFGB and WFGB in 2025. While many Caribbean reefs experienced widespread bleaching and coral cover declines during this period (Wicquart et al., 2025), surveys conducted at FGBNMS in 2025 documented many colonies exhibiting normal pigmentation and evidence of recovery from bleaching. These observations provide continued support of the resilience of FGBNMS following consecutive years of elevated thermal stress, which contrasts with more severe responses observed elsewhere in the Caribbean where thermal stress often resulted in widespread bleaching and mortality. For example, reefs in Martinique experienced bleaching affecting up to 86% of colonies in 2023, with limited recovery under continued warming in 2024 (Bon et al., 2025; Wicquart et al., 2025), while in the Cayman Islands, peak bleaching approached 80% and resulted in substantial coral mortality (Goodbody-Gringley & Chequer, 2025; Doherty et al., 2025). At FGBNMS, thermal conditions met or exceeded established bleaching thresholds during both 2023 and 2024 with bleaching prevalence reaching 21% at EFGB and 2% at WFGB in 2023 and 12% at WFGB in 2024. By 2025, little evidence of mortality associated with these bleaching events was observed, and many previously affected colonies exhibited signs of recovery.

Despite the low bleaching prevalence compared to other Caribbean reefs, the 2023–2024 event at FGBNMS was notable for its increased frequency, being the first documentation of

consecutive bleaching years at these reefs. Bleaching and paling have been documented at the banks in previous warm years, including 2010, 2016, 2021, 2023, and 2024, 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; O'Connell et al., 2026). Although bleaching can be a precursor to coral mortality, most affected corals at FGBNMS have historically recovered following thermal stress and have not been associated with substantial declines in overall coral cover (Johnston et al., 2019, 2023; Eisenbach et al., 2025; O'Connell et al., 2026). The 2023–2024 period represents the first instance of consecutive bleaching years at EFGB and WFGB, in contrast to earlier events that were typically separated by cooler conditions that facilitated recovery. Observations from 2025 indicate that many colonies exhibited normal pigmentation following the 2023–2024 event, suggesting recovery despite the potential for cumulative stress associated with back-to-back thermal exposure.

In 2025, bleaching was observed at low levels in August but not quantitatively characterized at its peak. However, during this time-period, differences in thermal exposure between EFGB and WFGB in 2025 were noted, suggesting that local oceanographic conditions are influencing thermal stress at the banks. At EFGB, in situ temperature records indicated prolonged exposure to elevated temperatures during the summer, with both ≥29.5 °C and ≥30 °C thresholds exceeded, suggesting sustained thermal stress conditions. In contrast, thermal exposure at WFGB remained below established bleaching thresholds, indicating lower heat stress at that site. Despite these differences in thermal exposures, bleaching prevalence was similar between the banks, indicating that bleaching responses were not proportional to differences in thermal exposure. Similar divergence between environmental conditions and biological response has been observed during past disturbance events at FGBNMS. For example, during the 2016 localized mortality and bleaching events, EFGB experienced both a localized mortality event and substantially higher bleaching prevalence while WFGB experienced no localized mortality and lower bleaching prevalence, demonstrating that though both banks are subjected to the same regional conditions, local oceanographic conditions and bank-level population differences may each play an important role in reef conditions (Johnston et al., 2019).

Differences in environmental conditions between banks are driven by local oceanographic processes, such as prevailing currents and the position and persistence of loop current eddies (Jarosz et al., 2014; Muller-Karger et al., 2015; Johnston et al., 2019), hurricanes (Aronson et al., 2005; Johnston et al., 2019), and coastal run-off (Kealoha et al. 2020a). While these circulation-driven differences provide an explanation for spatial variability in environmental conditions observed across the sanctuary, the disparate responses of the communities to environmental conditions suggest that environmental exposure alone does not fully determine reef condition.

Although disease-related impacts at EFGB and WFGB remained limited compared to many Caribbean reefs (Wicquart et al., 2025), disease remains an important and ongoing concern within this system. Observations of disease lesions consistent with the white-plague-like outbreak first documented in 2022 (Rossin, 2024) continued in 2025, particularly at photostations deeper than 25 meters where prevalence was higher than at shallow photostations. While both prevalence and qualitative assessments indicated reduced bleaching

and improving coral condition relative to 2024, coral disease and frequent damselfish gardening contributed to localized tissue loss across multiple species in 2025. Furthermore, sponge degradation was observed at both banks in 2025. While not documented previously on the coral reefs of EFGB and WFGB previously, declining sponge cover has been observed at nearby shoreward reefs in the sanctuary (Nuttall et al., 2025). Together, these patterns suggest that reef condition in 2025 was shaped less by acute thermal stress and more by a combination of chronic stressors that warrant continued monitoring and evaluation.

The increasing frequency and persistence of coral bleaching, disease outbreaks, and other stressors at EFGB and WFGB underscore the need for more comprehensive and targeted monitoring and research efforts to better understand, track, and manage these events. While patterns in coral bleaching were broadly consistent between colony-level prevalence and percent cover metrics, disease is less effectively captured using percent cover metrics, as marginal lesions and localized tissue loss are not consistently detected in point-based assessments. Furthermore, FGBNMS needs more consistent and coordinated sampling during stress events or bleaching years. In 2025, the absence of surveys during peak thermal stress limited the ability to fully evaluate bleaching extent, track coral fate, and compare responses between EFGB and WFGB. Because bleaching response efforts are conducted opportunistically and are often constrained by vessel availability, staffing, and funding limitations, survey coverage is inconsistent, resulting in spatial and temporal gaps in the data. More consistent survey coverage, including both repetitive photostations and reef-wide random transects at each bank during peak thermal stress or disease periods, would improve characterization of patterns and strengthen interpretation of the impacts of events.

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 of bleaching events projected under continued ocean warming (Heron et al., 2016; Hughes et al., 2017, 2018; Dias et al., 2023). Observations from 2025 indicate that, despite this repeated exposure, many colonies exhibited normal pigmentation and evidence of recovery following the 2023–2024 event. However, disease and other stressors are starting to play a larger role than previously documented. Continued monitoring will be critical for understanding how repeated exposure affects coral community resilience over time. In particular, fate tracking of individual coral colonies through repetitive photostations, along with emerging approaches such as photogrammetry, will be important for documenting partial mortality, recovery, and long-term survival following repeated stress events. While observations from 2025 suggest recovery, 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*) peers out from the reef. Photo: Tiffany Crumbley/Crumbley Photography

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 2025 at EFGB, four perimeter transects were completed, but only one center crosshair was done, totaling 1,000 m² surveyed. At WFGB, the same four perimeter transects were completed along with both centerlines, resulting in a completed surveyed area of 1,200 m².

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 ± SE. In 2025, EFGB and WFGB had different numbers of transects completed with five 100 m transects surveyed at EFGB and six 100 m transects at WFGB. The additional 100 m transect at WFGB was 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 4.5.2 (Clarke et al., 2014).

Sea Urchin and Lobster Survey Results

Density of *D. antillarum* was 4.25 ± 0.15 individuals/100 m² within the EFGB study site and 43.22 ± 0.53 individuals/100 m² within the WFGB study site in 2025 (Figure 4.1). One *P. guttatus* was observed in the EFGB study site and no *P. argus* were observed at either bank in 2025 (Figure 4.2).

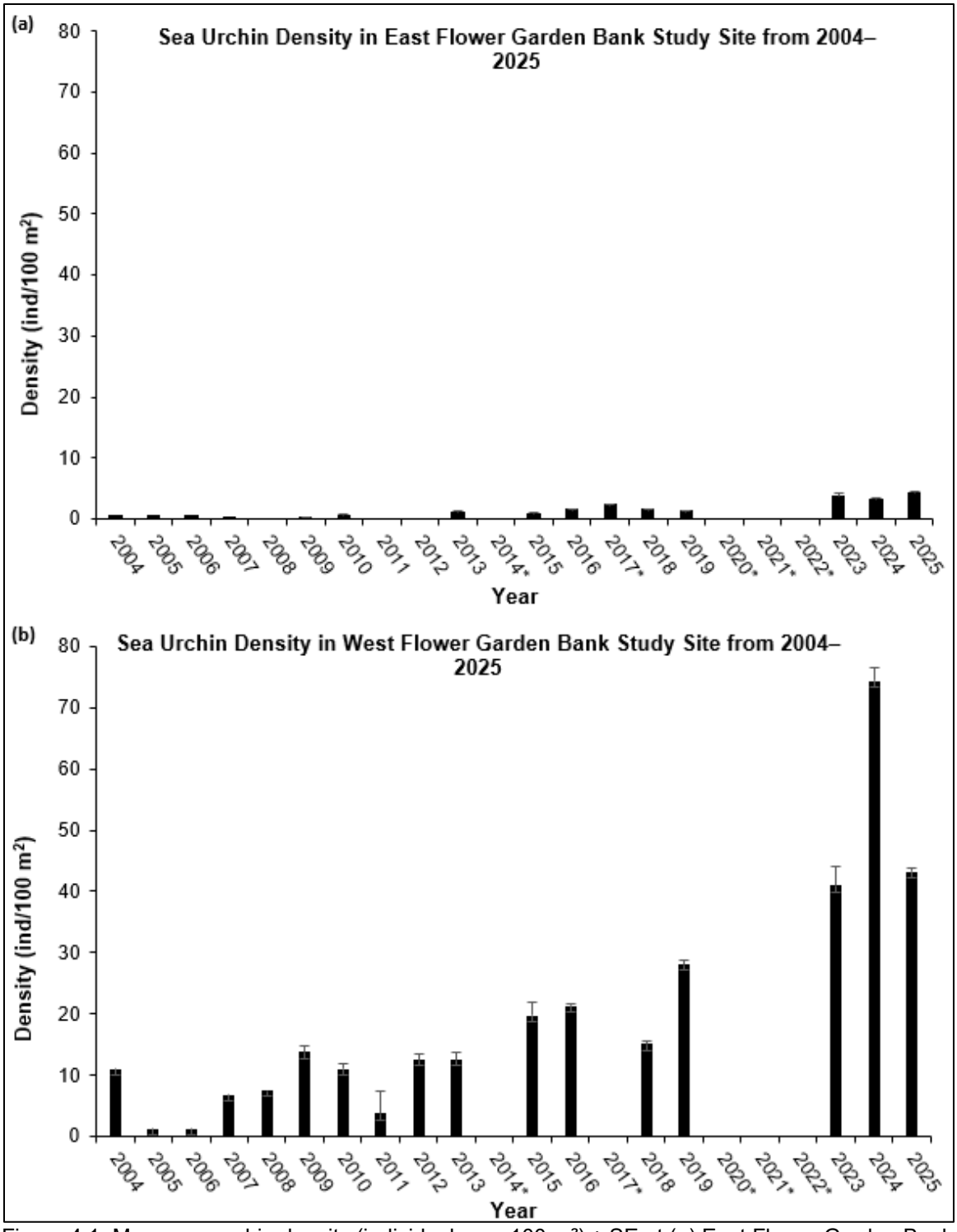


Figure 4.1. Mean sea urchin density (individuals per 100 m²) ± SE at (a) East Flower Garden Bank (EFGB) and (b) West Flower Garden Bank (WFGB) from 2004 to 2025. Asterisks following a year indicate no data were available for that year. Standard error values were not available from 2004 to 2008. Sources: Precht et al., 2006; Zimmer et al., 2010 (2004–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009–2022); Eisenbach et al., 2025 (2023); O'Connell et al., 2026 (2024).

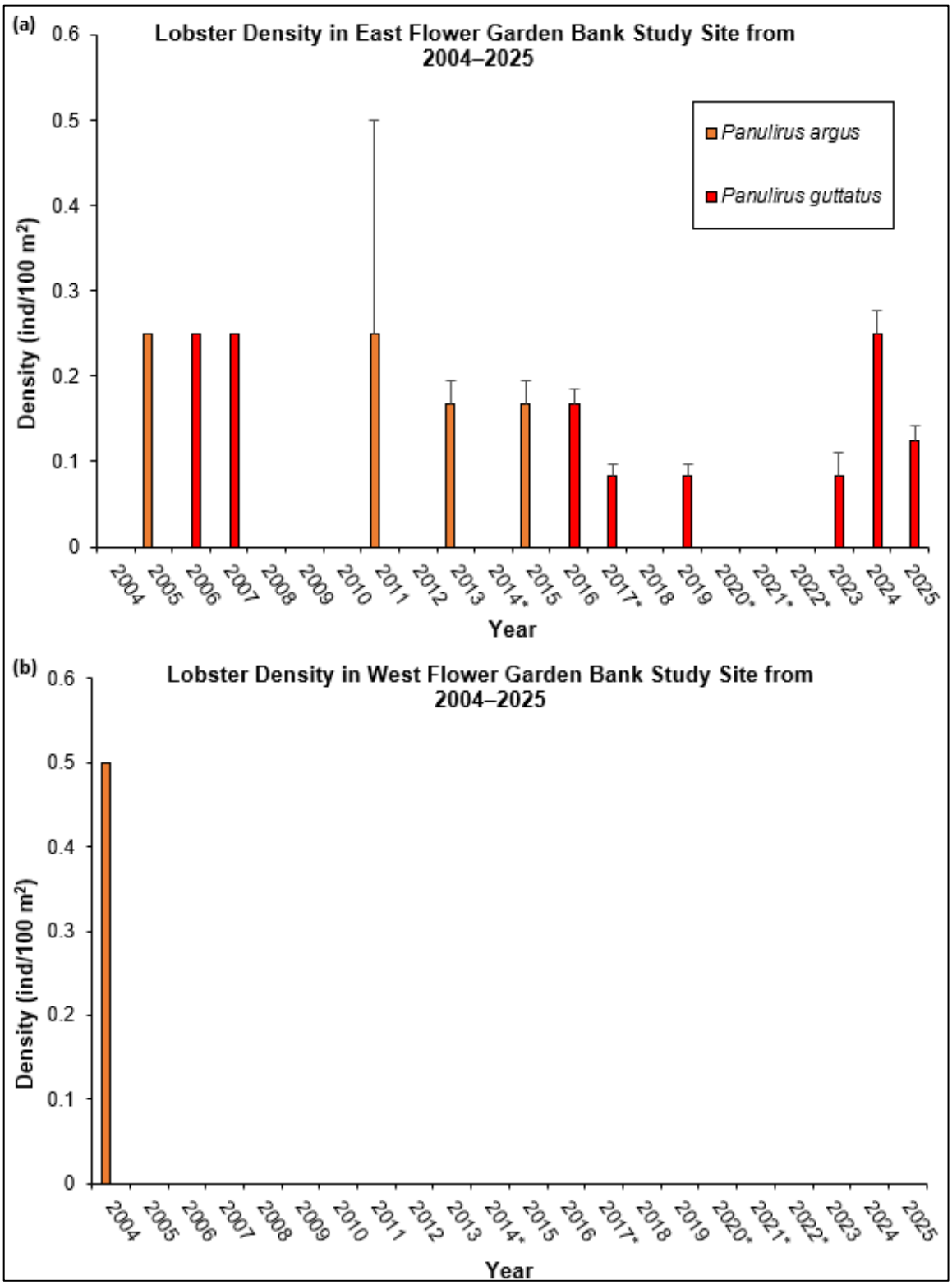


Figure 4.2. Mean lobster density (individuals per 100 m²) ± SE at (a) East Flower Garden Bank (EFGB) and (b) West Flower Garden Bank (WFGB) from 2004 to 2025. Asterisks following a year indicate no data were available for that year. Standard error values were not available from 2004 to 2008. Sources: Precht et al., 2006; Zimmer et al., 2010 (2004–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2022, 2024 (2009–2022); Eisenbach et al., 2025 (2023); O'Connell et al., 2026 (2024).

Since 2004, *D. antillarum* densities have ranged from 0–4.25 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-$F_{1,35} = 41.54$, $p < 0.05$). 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 2025. 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 with densities over 70 individuals per 100 m², however in 2025, they decreased to just below historic levels at 43 urchins per 100m². The cause of decrease in *D. antillarum* densities at WFGB is unknown at this time, but continued monitoring will assess urchin population health and determine if the WFGB urchin populations remain in decline. Though EFGB densities are much lower, they continue to slowly increase over time.

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). Other *Diadema* species experienced a similar die-off in 2022 and 2023 in the Eastern Atlantic, Mediterranean Sea, Red Sea, and Indian Ocean resulting in a global mass mortality event (Cano et al., 2025). 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. While WFGB densities decreased in 2025 from 2024, no signs of disease were observed during monitoring suggesting the decrease was unlikely due to this most recent die-off illness.

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 2025 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; O'Connell et al., 2026). 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 m^2 contribute to healthy and resilient reef systems (Mumby, et al. 2006, 2007). While neither EFGB nor WFGB have urchin densities that are above that threshold, WFGB urchin densities were close at 0.74 individuals per m^2 in 2024 but have decreased in 2025.

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.

Chapter 5:
Fish Surveys

A diver takes a photo of a school of horse-eye jacks (*Caranx latus*) swimming above the reef: Photo: Kelly O'Connell/CPC

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 at both fixed long-term monitoring study sites and random stratified reef-wide locations to examine and compare fish community composition and changes over time. Together, these complementary survey approaches allow the sanctuary to evaluate long-term trends at established monitoring stations while also capturing spatial variability in fish assemblages across each bank.

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 to 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-eight randomly located surveys were conducted within the study site at EFGB and 24 were conducted within the study site at WFGB. Twenty-one randomly located reef-wide surveys were conducted at EFGB, and 26 reef-wide surveys were conducted 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. 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 WFGB. Photo: Donavon French/CPC

All fish species observed within the first five minutes of the survey were recorded while the diver slowly rotated in place in the 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 2025 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 any mistakes were corrected prior to data processing. For each entry, fish family, trophic guild, and biomass were automatically recorded in the database (Bohnsack & Harper, 1988; Froese & Pauly, 2019). Species were classified into four categories: herbivores (H), piscivores (P), invertivores (I), and planktivores (PL), as defined by NOAA's Center for Coastal Monitoring and Assessment BioGeography Branch fish-trophic level database (Caldow et al., 2009) 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 ± SE per 100 m^2 and calculated as the total number of individuals per sample divided by the area of the survey cylinder (176.7 m^2) and multiplied by 100. Sighting frequency for each species was the percentage of samples in which the species was recorded. Mean species richness was the average number of species represented per sample ± SE. Fish biomass was expressed as kilograms ± SE per 100 m^2 and computed by converting length data to weight using the allometric length-weight conversion formula (Bohnsack & Harper, 1988) based on information provided by FishBase (Froese & Pauly, 2019). As sizes less than 35 cm were binned, the median size in each size bin was used to calculate biomass (for example, fish in the ≥5 to <10 cm size bin were assigned the total length of 7.5 cm). Observations of manta rays, 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). When a significant interaction between Year and Bank was detected, separate PERMANOVA analyses were conducted for each bank with year as the sole factor to further evaluate temporal patterns within each bank. 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 (2011–2018 and 2022–2024), 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 29 families and 86 species (77 at EFGB and 72 at WFGB) were observed in 2025 at both study site and reef-wide surveys. Mean species richness was 21.89 ± 0.52 per survey at EFGB, 21.10 ± 0.40 per survey at WFGB, and 21.47 ± 0.34 per survey for both banks at study sites and reef wide combined. Brown chromis (*Azurina multilineata*) had the highest relative abundance of all species in EFGB surveys (50.08%), followed by Atlantic creolefish (*Paranthias furcifer*; 41.47%), creole wrasse (*Bodianus parrae*; 25.24%), bluehead (*Thalassoma bifasciatum;* 20.82%), and regal demoiselle (*Neopomacentrus cyanomos*; 5.74%). In WFGB surveys, brown chromis had the highest relative abundance (33.86%), followed by Atlantic creolefish (16.11%), bluehead (11.13%), creole wrasse (9.50%), and blue chromis (*Azurina cyanea*; 3.30%).

Sighting Frequency and Occurrence

The most frequently sighted species were bluehead and brown chromis, occurring in 96% and 94% of EFGB surveys, respectively, and in 98% and 86% respectively of WFGB surveys. Other frequently sighted species included Atlantic creolefish, Spanish hogfish (*Bodianus rufus*) and sharpnose puffer (*Canthigaster callisterna*; Table 5.1). No manta rays (*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). While no manta rays were observed during fish surveys, multiple manta rays were opportunistically observed during dives at both banks.

Table 5.1. Sighting frequencies for the 10 most frequently sighted species in EFGB and WFGB surveys in both study site and reef-wide surveys in 2025.

Fish Species	EFGB	WFGB
Bluehead (*Thalassoma bifasciatum*)	95.9%	98.0%
Brown chromis (*Azurina multilineata*)	93.9%	86.0%
Atlantic creolefish (*Paranthias furcifer*)	85.7%	94.0%
Spanish hogfish (*Bodianus rufus*)	93.9%	82.0%
Sharpnose puffer (*Canthigaster callisterna*)	83.7%	90.0%
Great barracuda (*Sphyraena barracuda*)	81.6%	90.0%
Blue tang (*Acanthurus coeruleus*)	83.7%	88.0%
Stoplight parrotfish (*Sparisoma viride*)	85.7%	80.0%
Cocoa damselfish (*Stegastes variabilis*)	87.8%	74.0%
Blue chromis (*Azurina cyanea*)	73.5%	86.0%

Density

Mean fish density (individuals/100 m²) was 197.33 ± 33.29 in EFGB surveys and 108.39 ± 9.52 in WFGB surveys, with significantly higher fish densities at EFGB (Table 5.2). Fish density was also higher in reef-wide surveys compared to study site surveys at both banks (Table 5.2). SIMPER analysis identified that differences in fish assemblages between banks were driven primarily by greater abundance of sharpnose puffer at EFGB (6.99%), and greater abundance of blue tang (*Acanthurus coeruleus*; 5.33%) and bluehead wrasse (5.02%) at WFGB (Table 5.3). SIMPER analysis identified greater abundance of Spanish hogfish (6.64%) and bluehead (4.85%) in reef-wide surveys and greater abundance of blue tang (5.35%) in study site surveys as the main contributors to the difference between survey type.

Table 5.2. PERMANOVA results comparing mean fish density between EFGB and WFGB surveys and reef-wide and study site surveys (station) in 2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	7520	1	4.3947	**<0.001**
Station	5538.6	1	3.2368	**<0.001**
Bank x Station	2156.6	1	1.2603	0.19
Res	1.6256E+05	95		
Total	1.7788E+05	98		

Table 5.3. Mean density (individuals ± SE/100 m²) of the 15 most abundant species from EFGB and WFGB surveys in 2025. The density of bolded species was significantly different between EFGB and WFGB and/or reef-wide and study site surveys

Fish Species	EFGB	WFGB
Brown chromis (*Azurina multilineata*)	50.08 ± 9.82	33.86 ± 7.53
Atlantic creolefish (*Paranthias furcifer*)	41.48 ± 12.68	16.11 ± 6.26
Creole wrasse (*Bodianus parrae*)	25.25 ± 9.58	9.50 ± 2.04
Bluehead (*Thalassoma bifasciatum*)	20.82 ± 2.52	11.13 ± 1.94
Blue chromis (*Azurina cyanea*)	3.82 ± 1.64	3.30 ± 0.94
Regal demoiselle (*Neopomacentrus cyanomos*)	5.74 ± 1.96	0.28 ± 0.17
Horse-eye Jack (*Caranx latus*)	4.51 ± 2.25	0.83 ± 0.60
Great barracuda (*Sphyraena barracuda*)	3.09 ± 1.83	2.17 ± 0.44
Bicolor damselfish (*Stegastes partitus*)	3.17 ± 0.65	1.67 ± 0.38
Threespot damselfish (*Stegastes planifrons*)	2.83 ± 0.43	1.42 ± 0.33
Cocoa damselfish (*Stegastes variabilis*)	2.65 ± 0.38	1.16 ± 0.21
Sharpnose puffer (*Canthigaster callisterna*)	1.46 ± 0.25	2.45 ± 0.47
Blue tang (*Acanthurus coeruleus*)	1.34 ± 0.18	2.22 ± 0.31
Stoplight parrotfish (*Sparisoma viride*)	1.78 ± 0.23	1.09 ± 0.16
Spanish hogfish (*Bodianus rufus*)	1.54 ± 0.18	1.13 ± 0.18
Mean Density	197.33 ± 33.29	108.34 ± 9.52

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 dominated the small size class at both study sites. Planktivores dominated the mid-range size classes, while piscivores dominated the large size classes (Figure 5.2).

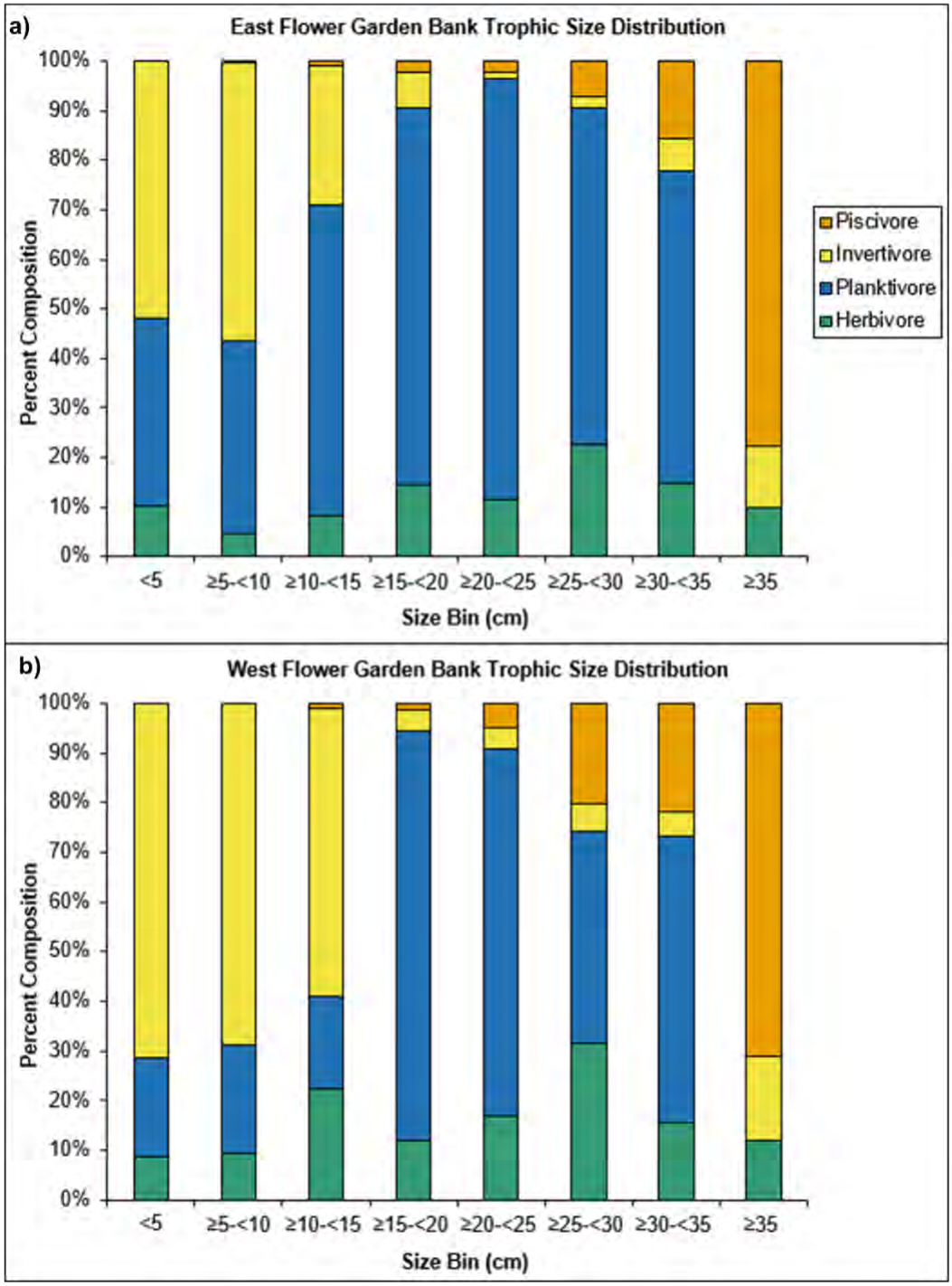


Figure 5.2. Fish size distribution by trophic guild at (a) EFGB and (b) WFGB study sites in 2025.

Biomass

Mean biomass (kg ± SE/100 m²) in 2025 was 21.80 ± 4.42 in EFGB surveys and 10.27 ± 1.13 in WFGB surveys. PERMANOVA analysis indicated that fish community, based on biomass, differed significantly between banks and between survey types (Table 5.4). SIMPER analysis showed that the differences in biomass between banks were driven by greater local biomass of

bluehead (5.22%) and great barracuda (*Sphyraena barracuda*; 5.13%) at EFGB. Differences between survey types were driven by higher biomass of bluehead (4.34%) in reef-wide surveys and great barracuda (5.08%) in study stie surveys.

Table 5.4. PERMANOVA results comparing fish communities based on biomass at EFGB and WFGB and reef wide and study site surveys in 2025. **Bold** text denotes significant value.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Bank	5962.8	1	2.1975	**<0.001**
Station	8662	1	3.1922	**<0.001**
Bank x Station	3400.7	1	1.2533	0.16
Res	2.5778E+05	95		
Total	2.7582E+05	98		

When classified by trophic guild, piscivores exhibited the highest mean biomass across all EFGB surveys, whereas planktivores had the highest mean biomass across all WFGB surveys (Table 5.5). PERMANOVA detected significant differences in trophic guild biomass between reef-wide and study site surveys, but not between banks. The high mean biomass of planktivores at WFGB is likely caused by the large schools of Atlantic creolefish that were observed in reef-wide surveys at WFGB in 2025 which contributed to 41% of total mean biomass of WFGB reef-wide surveys. This is the first observation of the planktivore trophic guild dominating mean biomass in any surveys at EFGB or WFGB.

SIMPER analysis indicated that reef-wide surveys were characterized by higher biomass of invertivores and herbivores at both banks. At EFGB, reef-wide surveys contained higher biomass of invertivores (27%) and herbivores (27%), whereas study site surveys contained higher biomass of plankivores (26%). At WFGB, reef-wide surveys contained higher biomass of herbivores (38%) and invertivores (32%).

Table 5.5. Mean biomass (kg ± SE/100 m^2) for each trophic guild from EFGB and WFGB, study site (SS), reef-wide (RW), and total surveys in 2025

Site	Herbivore	Invertivore	Planktivore	Piscivore
EFGB SS	1.84 ± 0.21	0.66 ± 0.10	8.02 ± 2.68	13.22 ± 6.09
WFGB SS	1.78 ± 0.22	0.74 ± 0.16	2.23 ± 0.77	3.16 ± 0.44
EFGB RW	3.76 ± 1.10	1.96 ± 0.44	4.41 ± 2.04	9.09 ± 4.78
WFGB RW	3.29 ± 0.10	2.06 ± 0.56	5.44 ± 1.66	1.67 ± 0.29
EFGB Total	2.66 ± 0.50	1.22 ± 0.22	6.47 ± 1.77	11.45 ± 4.01
WFGB Total	2.57 ± 0.54	1.42 ± 0.33	3.90 ± 0.96	2.34 ± 0.28

Mean biomass for each species, grouped by trophic guild, is presented in Table 5.6. At the combined study site and reef-wide EFGB surveys, 37% of herbivore biomass was contributed by black durgon (*Melichthys niger*). For invertivores, the greatest contribution was from ocean triggerfish (*Canthidermis sufflamen*; 22%). For piscivores, horse-eye jack contributed the greatest biomass (*Caranx latus*; 81%). For planktivores, the greatest contribution was from Atlantic creolefish (90%). At the WFGB combined surveys, 48% of herbivore biomass was contributed by black durgon. For invertivores, the greatest contribution was from ocean triggerfish (55%). For piscivores, great barracuda contributed the greatest biomass (60%). For planktivores, the greatest contribution was from Atlantic creolefish (88%; Table 5.6).

Table 5.6. Biomass (kg ± SE/100 m^2) of each species grouped by trophic guild, from all EFGB and WFGB surveys in 2025.

Trophic Guild	Fish Species	EFGB	WGB
Herbivore	Black durgon (*Melichthys niger*)	0.995 ± 0.458	1.227 ± 0.480
Herbivore	Stoplight parrotfish (*Sparisoma viride*)	0.648 ± 0.109	0.572 ± 0.099
Herbivore	Queen parrotfish (*Scarus vetula*)	0.239 ± 0.034	0.277 ± 0.055
Herbivore	Chub complex (*Kyphosus sectatrix/incisor* complex)	0.358 ± 0.109	0.062 ± 0.026
Herbivore	Blue tang (*Acanthurus coeruleus*)	0.172 ± 0.033	0.148 ± 0.020
Herbivore	Redband parrotfish (*Sparisoma aurofrenatum*)	0.066 ± 0.018	0.050 ± 0.016
Herbivore	Princess parrotfish (*Scarus taeniopterus*)	0.061 ± 0.016	0.047 ± 0.017
Herbivore	Striped parrotfish (*Scarus iseri*)	0.037 ± 0.014	0.026 ± 0.013
Herbivore	Ocean surgeonfish (*Acanthurus tractus*)	0.018 ± 0.007	0.042 ± 0.016
Herbivore	Midnight parrotfish (*Scarus coelestinus*)	0.000 ± 0.000	0.057 ± 0.043
Herbivore	Doctorfish (*Acanthurus chirurgus*)	0.027 ± 0.009	0.015 ± 0.007
Herbivore	Bicolor damselfish (*Stegastes partitus*)	0.015 ± 0.004	0.013 ± 0.004
Herbivore	Cocoa damselfish (*Stegastes variabilis*)	0.013 ± 0.006	0.007 ± 0.002
Herbivore	Yellowtail damselfish (*Microspathodon chrysurus*)	0.006 ± 0.003	0.008 ± 0.002
Herbivore	Greenblotch parrotfish (*Sparisoma atomarium*)	0.005 ± 0.002	0.002 ± 0.001
Herbivore	Longfin damselfish (*Stegastes diencaeus*)	0.001 ± 0.001	0.005 ± 0.002
Herbivore	Surgeonfish spp. (*Acanthurus* spp.)	0.000 ± 0.000	0.003 ± 0.003
Herbivore	Dusky damselfish (*Stegastes adustus*)	0.001 ± 0.001	0.002 ± 0.001
Herbivore	Yellowtail parrotfish (*Sparisoma rubripinne*)	0.000 ± 0.000	0.001 ± 0.001
Herbivore	Redlip blenny (*Ophioblennius macclurei*)	<0.001 ± <0.001	<0.001 ± <0.001
Herbivore	Bucktooth parrotfish (*Sparisoma radians*)	<0.001 ± <0.001	0.000 ± 0.000
Invertivore	Ocean triggerfish (*Canthidermis sufflamen*)	0.269 ± 0.136	0.782 ± 0.299
Invertivore	Gray snapper (*Lutjanus griseus*)	0.217 ± 0.086	0.192 ± 0.067
Invertivore	Brown chromis (*Azurina multilineata*)	0.199 ± 0.050	0.128 ± 0.033
Invertivore	Queen triggerfish (*Balistes vetula*)	0.150 ± 0.107	0.000 ± 0.000
Invertivore	Spanish hogfish (*Bodianus rufus*)	0.067 ± 0.011	0.046 ± 0.009
Invertivore	Reef butterflyfish (*Chaetodon sedentarius*)	0.047 ± 0.012	0.040 ± 0.008
Invertivore	Rock beauty (*Holacanthus tricolor*)	0.019 ± 0.008	0.049 ± 0.014
Invertivore	Bluehead (*Thalassoma bifasciatum*)	0.038 ± 0.006	0.025 ± 0.004
Invertivore	Smooth trunkfish (*Lactophrys triqueter*)	0.032 ± 0.013	0.018 ± 0.006
Invertivore	Queen angelfish (*Holacanthus ciliaris*)	0.019 ± 0.008	0.017 ± 0.010
Invertivore	Yellowhead wrasse (*Halichoeres garnoti*)	0.015 ± 0.005	0.018 ± 0.003
Invertivore	Yellow goatfish (*Mulloidichthys martinicus*)	0.018 ± 0.008	0.010 ± 0.006
Invertivore	Spotfin butterflyfish (*Chaetodon ocellatus*)	0.021 ± 0.007	0.007 ± 0.003
Invertivore	Honeycomb cowfish (*Acanthostracion polygonium*)	0.004 ± 0.004	0.022 ± 0.014
Invertivore	Squirrelfish (*Holocentrus adscensionis*)	0.012 ± 0.008	0.015 ± 0.009
Invertivore	Porgy spp. (*Sparidae* spp.)	0.020 ± 0.021	0.000 ± 0.000
Invertivore	Threespot damselfish (*Stegastes planifrons*)	0.014 ± 0.002	0.005 ± 0.001

Trophic Guild	Fish Species	EFGB	WGB
Invertivore	Puddingwife (*Halichoeres radiatus*)	0.011 ± 0.011	0.007 ± 0.006
Invertivore	French angelfish (*Pomacanthus paru*)	0.016 ± 0.013	0.000 ± 0.000
Invertivore	Orangespotted filefish (*Cantherhines pullus*)	0.006 ± 0.002	0.006 ± 0.002
Invertivore	Scrawled filefish (*Aluterus scriptus*)	0.001 ± 0.001	0.008 ± 0.006
Invertivore	Longspine squirrelfish (*Holocentrus rufus*)	0.004 ± 0.002	0.006 ± 0.003
Invertivore	Sharpnose puffer (*Canthigaster callisterna*)	0.004 ± 0.001	0.004 ± 0.001
Invertivore	Cottonwick (*Haemulon melanurum*)	0.000 ± 0.000	0.008 ± 0.005
Invertivore	Rock hind (*Epinephelus adscensionis*)	0.001 ± 0.001	0.004 ± 0.004
Invertivore	Scrawled cowfish (*Acanthostracion quadricornis*)	0.005 ± 0.005	<0.001 ± <0.001
Invertivore	Banded butterflyfish (*Chaetodon striatus*)	0.002 ± 0.002	0.002 ± 0.001
Invertivore	Longsnout butterflyfish (*Prognathodes aculeatus*)	0.001 ± <0.001	0.002 ± 0.001
Invertivore	Clown wrasse (*Halichoeres maculipinna*)	0.001 ± 0.001	<0.001 ± <0.001
Invertivore	Sergeant major (*Abudefduf saxatilis*)	<0.001 ± <0.001	0.001 ± 0.001
Invertivore	Spotfin hogfish (*Bodianus pulchellus*)	0.000 ± 0.000	0.001 ± 0.001
Invertivore	Redspotted hawkfish (*Amblycirrhitus pinos*)	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Beaugregory (*Stegastes leucostictus*)	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Goldentail moray (*Gymnothorax miliaris*)	<0.001 ± <0.001	0.000 ± 0.000
Invertivore	Barred Hamlet (*Hypoplectrus puella*)	<0.001 ± <0.001	0.000 ± 0.000
Invertivore	Bridled goby complex (*Coryphopterus glaucofraenum* complex)	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Neon goby (*Elacatinus oceanops*)	<0.001 ± <0.001	<0.001 ± <0.001
Invertivore	Goldface toby (*Canthigaster jamestyleri*)	<0.001 ± <0.001	0.000 ± 0.000
Invertivore	Seaweed blenny (*Parablennius marmoreus*)	<0.001 ± <0.001	0.000 ± 0.000
Invertivore	Broadstripe goby (*Elacatinus prochilos*)	0.000 ± 0.000	<0.001 ± <0.001
Invertivore	Spotted drum (*Eques punctatus*)	<0.001 ± <0.001	0.000 ± 0.000
Piscivore	Horse-eye jack (*Caranx latus*)	9.326 ± 4.148	0.356 ± 0.199
Piscivore	Great barracuda (*Sphyraena barracuda*)	1.786 ± 0.346	1.443 ± 0.190
Piscivore	Dog snapper (*Lutjanus jocu*)	0.100 ± 0.098	0.286 ± 0.118
Piscivore	Lionfish (*Pterois volitans/miles*)	0.020 ± 0.014	0.101 ± 0.046
Piscivore	Yellowmouth grouper (*Mycteroperca interstitialis*)	0.064 ± 0.038	0.025 ± 0.013
Piscivore	Graysby (*Cephalopholis cruentata*)	0.033 ± 0.009	0.041 ± 0.008
Piscivore	Bar jack (*Caranx ruber*)	0.023 ± 0.008	0.033 ± 0.020
Piscivore	Tiger grouper (*Mycteroperca tigris*)	0.027 ± 0.023	0.025 ± 0.016
Piscivore	Rainbow runner (*Elagatis bipinnulata*)	0.000 ± 0.000	0.044 ± 0.044
Piscivore	Scamp (*Mycteroperca phenax*)	0.030 ± 0.031	0.002 ± 0.002
Piscivore	Yellowfin grouper (*Mycteroperca venenosa*)	0.029 ± 0.030	0.000 ± 0.000
Piscivore	Black grouper (*Mycteroperca bonaci*)	0.000 ± 0.000	0.021 ± 0.021
Piscivore	Black jack (*Caranx lugubris*)	0.002 ± 0.002	0.009 ± 0.006
Piscivore	Spotted moray (*Gymnothorax moringa*)	0.010 ± 0.009	0.000 ± 0.000
Planktivore	Atlantic creolefish (*Paranthias furcifer*)	5.853 ± 1.718	3.439 ± 0.934
Planktivore	Creole wrasse (*Bodianus parrae*)	0.593 ± 0.286	0.445 ± 0.146

Trophic Guild	Fish Species	EFGB	WGB
Planktivore	Blue chromis (*Azurina cyanea*)	0.021 ± 0.015	0.008 ± 0.002
Planktivore	Regal demoiselle (*Neopomacentrus cyanomos*)	0.003 ± 0.001	0.002 ± 0.001
Planktivore	Sunshinefish (*Chromis insolata*)	0.002 ± 0.001	0.003 ± 0.001
Planktivore	Mackerel scad (*Decapterus macarellus*)	0.001 ± 0.001	0.002 ± 0.002
Planktivore	Round scad (*Decapterus punctatus*)	0.001 ± 0.001	0.000 ± 0.000
Planktivore	Yellowhead jawfish (*Opistognathus aurifrons*)	<0.001 ± <0.001	0.000 ± 0.000
Planktivore	Remora (*Remora remora*)	0.000 ± 0.000	<0.001 ± <0.001
Planktivore	Purple reeffish (*Chromis scotti*)	<0.001 ± <0.001	<0.001 ± <0.001

Abundance-Biomass Curves

Mean w-values for both the EFGB and WFGB surveys were 0.13 ± 0.01. For all samples at each bank, mean w-values remained close to 0, indicating a balanced community where biomass was spread evenly between large and small individuals (Figure 5.3). ANOVA comparisons of w-values between banks and survey type 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 surveys

Source	Sum of Squares	df	F-value	P
Bank	0.0246	1	2.053	0.16
Res	1.1617	97		
Total	1.1863	98		

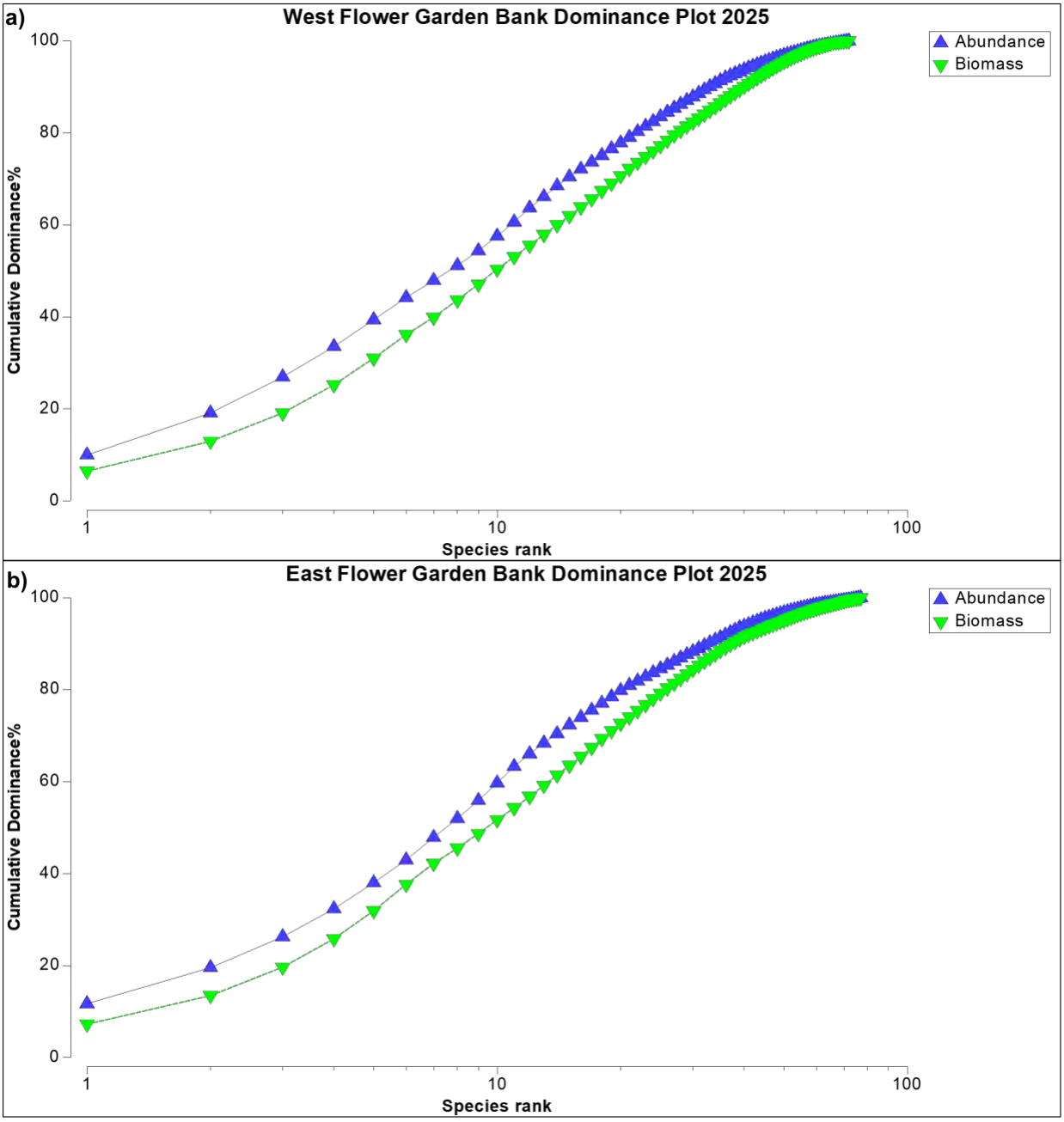


Figure 5.3. Abundance-biomass curves for (a) WFGB and (b) EFGB study sites in 2025.

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*) and non-native regal demoiselle.

In 2025, seven species of grouper were observed in EFGB and WFGB surveys: graysby (*Cephalopholis cruentata*), rock hind (*Epinephelus adscensionis*), yellowmouth grouper (*Mycteroperca interstitialis*), tiger grouper (*Mycteroperca tigris*), black grouper (*Mycteroperca*

bonaci), yellowfin grouper (*Mycteroperca venenose*), and scamp (*Mycteroperca phenax*). The CV% for the density of small bodied groupers (graysby and rock hind) had relatively good power to detect population change while the CV% for the density of the other five species (large-bodied groupers) and the biomass for all seven grouper species had poor power to detect population differences (i.e., CV% > 20%). Grouper mean biomass (kg ± SE/100 m²) was 0.184 ± 0.070 in EFGB surveys and 0.119 ± 0.033 in WFGB surveys. Mean biomass of small-bodied grouper was 0.034 ± 0.009 in EFGB surveys and 0.045 ± 0.009 in WFGB surveys. Mean biomass of large-bodied grouper (yellowmouth grouper, tiger grouper, black grouper, yellowfin grouper, and scamp) was 0.150 ± 0.070 in EFGB surveys and 0.074 ± 0.031 in WFGB surveys. Size distributions of observed grouper in 2025 varied by species (Figure 5.4). Mean biomass and density of large-bodied groupers was higher in reef-wide surveys at both EFGB and WFGB than in study site surveys. For small-bodied groupers, mean density and biomass were higher in reef-wide surveys than study site surveys at EFGB but were the same at reef wide and study site surveys at WFGB. Observations of large-bodied groupers were predominately made in reef-wide surveys, with scamp, black grouper, and yellowfin grouper only seen in reef-wide surveys (Figure 5.4). For small bodied groupers, there was no significant difference in density or biomass between EFGB and WFGB ($p>0.05$). Similarly, there was no significant difference in small bodied grouper biomass or density between reef-wide and study site surveys.

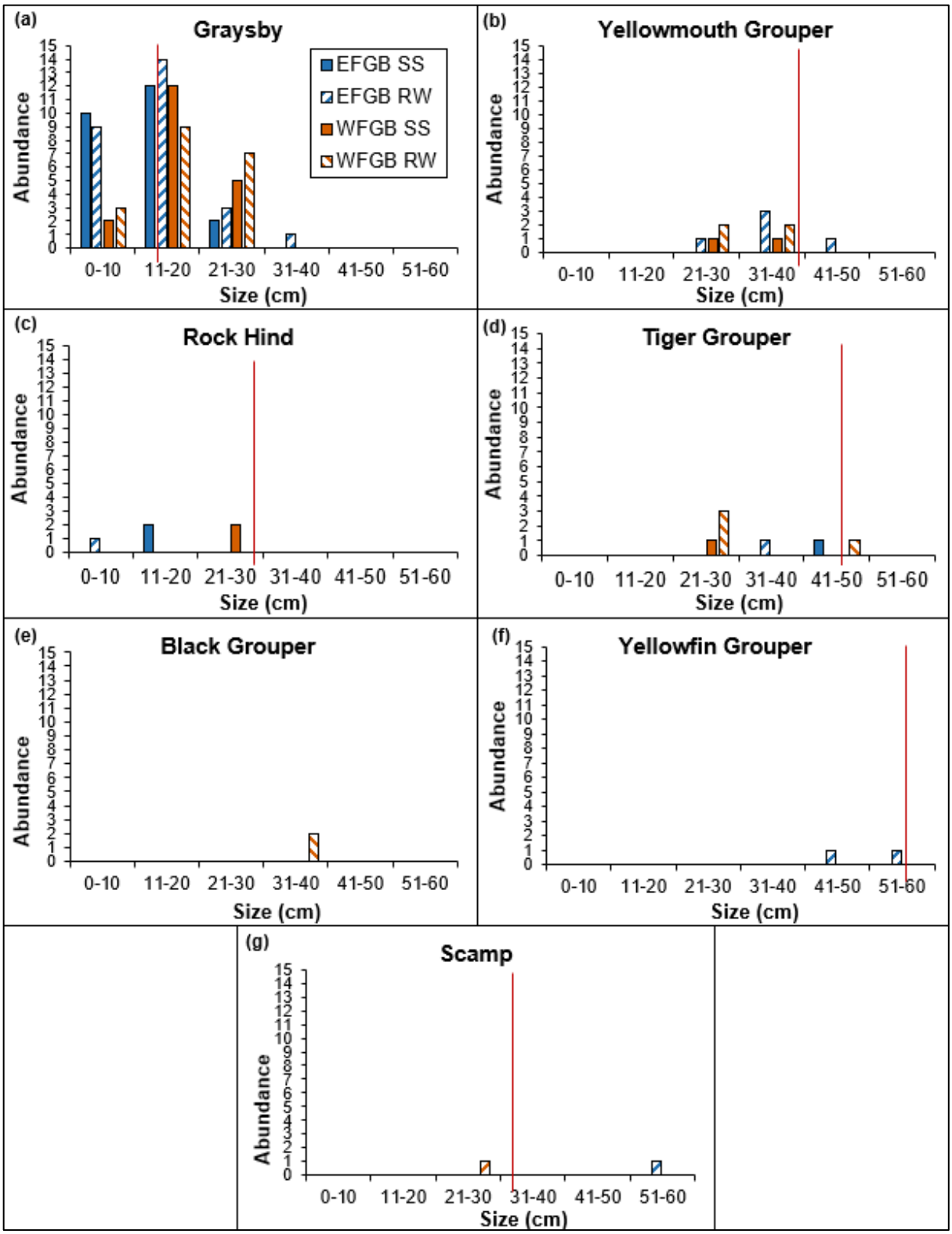


Figure 5.4. Size frequency of grouper species at EFGB and WFGB study site (SS) and reef wide (RW) surveys in 2025: (a) graysby, (b) yellowmouth grouper, (c) rock hind, (d) tiger grouper, (e) black grouper, (f) yellowfin grouper, and (g) scamp. Vertical solid red lines represent estimated size of female maturity (Froese & Pauly, 2019). The estimated female size at maturity for black grouper is 67.7 cm but is not shown.

Two snapper species were observed in 2025 surveys: dog snapper (*Lutjanus jocu*) and gray snapper (*Lutjanus griseus*; Figure 5.5). Coefficients of variation for biomass for both species indicated that the data had poor power to detect population differences, however density for both species at both banks had sufficient power to detect density differences. There was no significant difference in density between EFGB and WFGB ($p>0.05$). Mean snapper biomass (kg ± SE/100 m²) was 0.08 and 0.64 at EFGB study site and reef-wide surveys, respectively, and 0.21 and 0.73 at WFGB study site and reef wide surveys. No statistical test was run on snapper biomass due to poor statistical power.

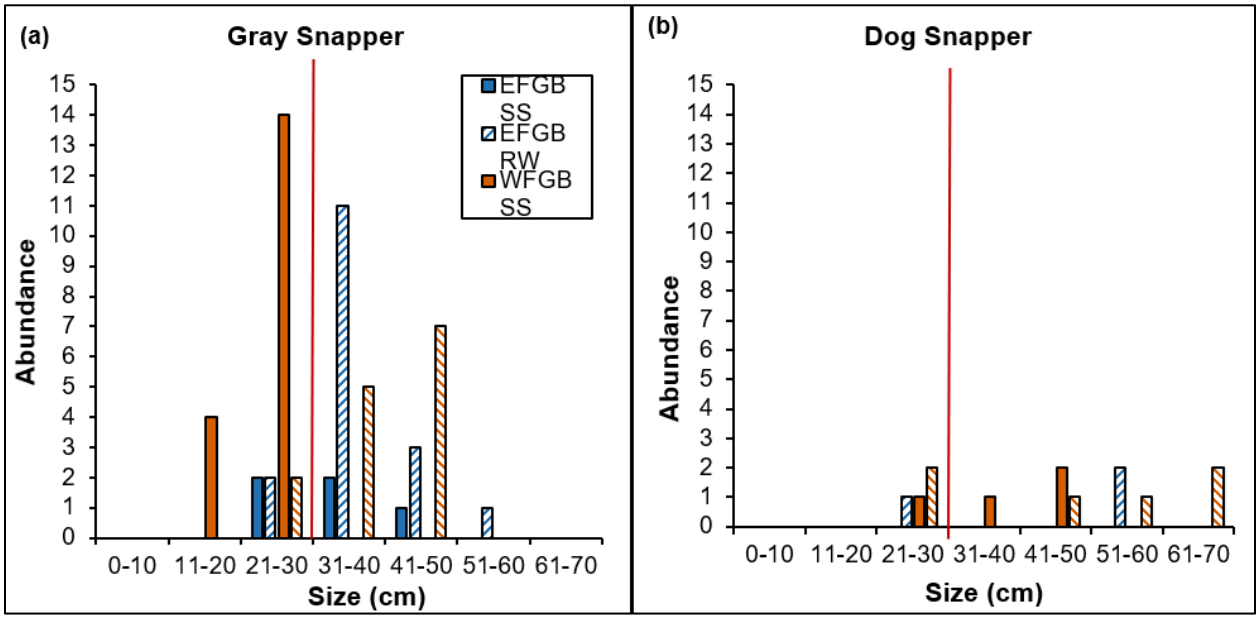


Figure 5.5. Size frequency of snapper species observed at EFGB and WFGB study site and reef wide surveys in 2025: (a) gray snapper and (b) dog snapper. Vertical solid red line represents estimated size of female maturity (Froese & Pauly, 2019).

Parrotfishes are important grazers on coral reefs (Jackson et al., 2014). Nine species of parrotfish were observed in EFGB and WFGB fish surveys in 2025: princess parrotfish (*Scarus taeniopterus*), redband parrotfish (*Sparisoma aurofrenatum*), stoplight parrotfish (*Sparisoma viride*), queen parrotfish (*Scarus vetula*) yellowtail parrotfish (*Sparisoma rubripinne*), striped parrotfish (*Scarus iseri*), greenblotch parrotfish (*Sparisoma atomarium*), midnight parrotfish (*Scarus coelestinus*), and bucktooth parrotfish (*Sparisoma radians*). Coefficients of variation indicated that the data had good power to detect differences in density between EFGB and WFGB for all nine parrotfish species. There was no significant difference in parrotfish biomass between EFGB and WFGB, however there was a significant difference between banks in parrotfish density ($p<0.05$). SIMPER analysis revealed that these differences were attributed to greater density of stoplight parrotfish, queen parrotfish, redband parrotfish, and striped parrotfish at EFGB. Parrotfish populations at both banks had wide ranging size distributions, but EFGB had high number of small juvenile parrotfish in both study site and reef-wide surveys suggesting a potential recruitment event at that bank (Figure 5.6).

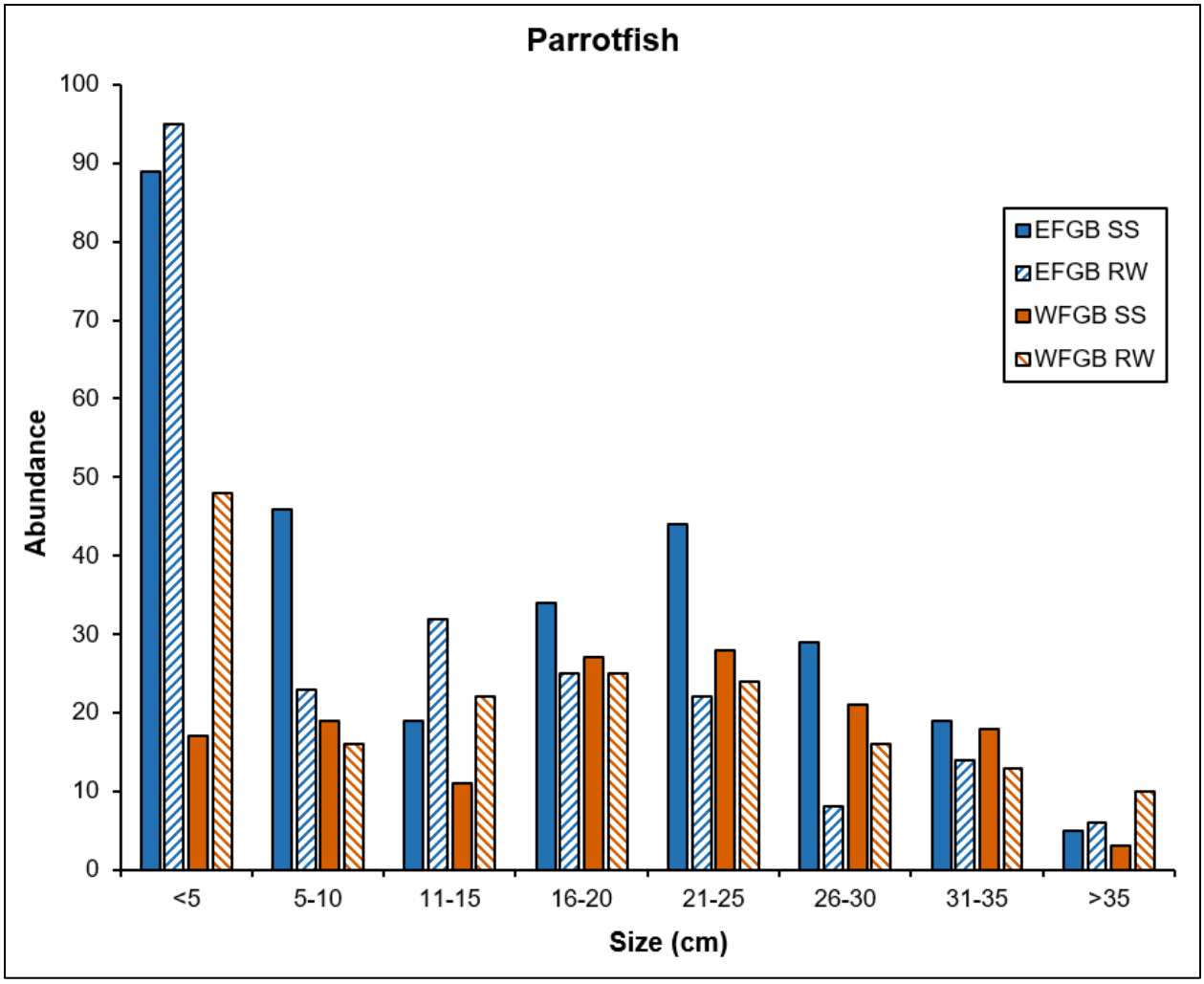


Figure 5.6. Size frequency of parrotfishes at EFGB and WFGB study site and reef-wide surveys in 2025.

Invasive and Non-native Species

Lionfish (*Pterois volitans),* 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. Fifteen lionfish were observed in surveys in 2025, two at EFGB in reef-wide surveys and 13 at WFGB in both study site (8) and reef-wide (5) surveys. Lionfish were also opportunistically observed by divers during long-term monitoring field work.

Regal demoiselle (*Neopomacentrus cyanomos*), a non-native species from the Indo-Pacific region, were observed in 2025 surveys at East and West Flower Garden Banks for the first time since 2019. In addition to observations at East and West Flower Garden Banks in 2025, regal demoiselle were also observed in fish surveys at Stetson Bank during long-term monitoring field work in June 2024 (50% frequency in Stetson Bank fish surveys). Large abundances were also observed at nearby oil and gas rigs and at *Kraken*, a shipwreck close to Stetson Bank, during lionfish removal cruises in June, July, August, and September 2025.

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 5.7). Fish community density was compared among years and study sites when complete survey data were available (2011–2025). For 2025 data, only study site surveys were used in analysis to compare to previous years. PERMANOVA analysis revealed significant effects of Year and Bank, as well as a significant Year x Bank interaction (Table 5.8), indicating that temporal patterns in fish community density differed between EFGB and WFGB. The observed dissimilarity in study site communities was mainly attributable to higher densities of blue tang (5.61%) and reef butterflyfish (4.30%) at WFGB.

To examine temporal patterns within each bank, separate PERMANOVA analyses were conducted for EFGB and WFGB with year as the sole factor. Density differed significantly among years at both EFGB (Pseudo-F = 4.0897, $p < 0.001$) and WFGB (Pseudo-F = 5.4687, $p < 0.001$), indicating pronounced interannual change in fish community based on density at both sites.

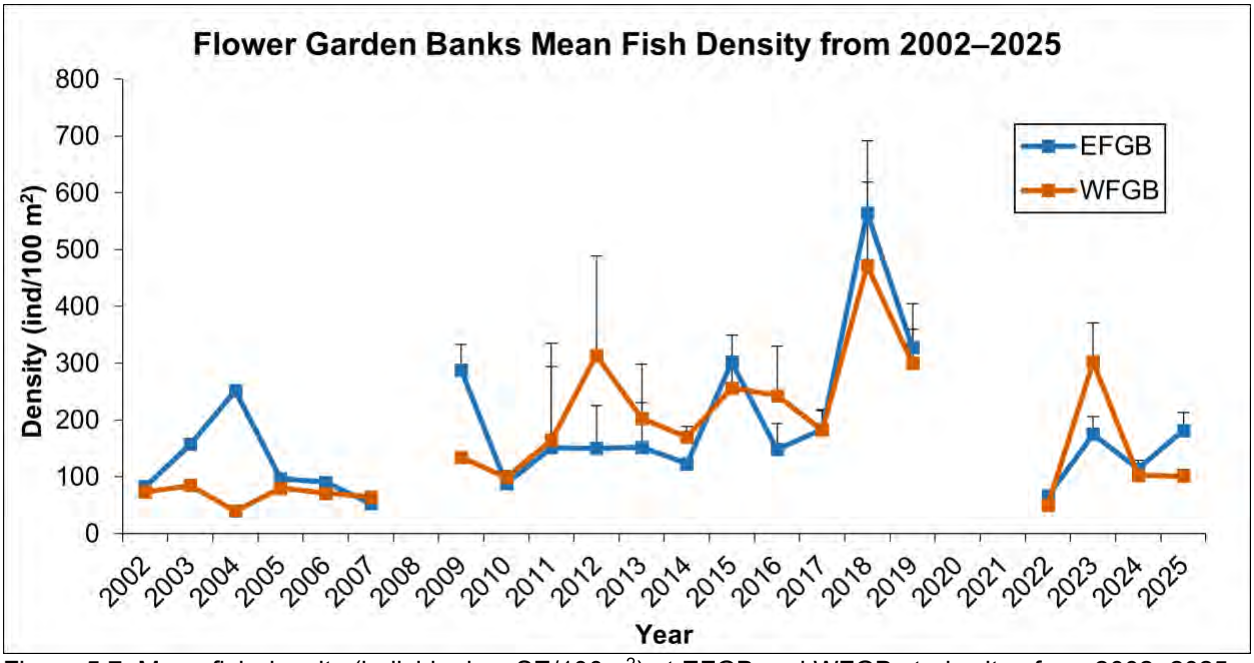


Figure 5 7. Mean fish density (individuals + SE/100m^2) at EFGB and WFGB study sites from 2002–2025. No data were collected in 2008, 2020, and 2021, and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024 (2009–2022); Eisenbach et al., 2025 (2023); O'Connell et al, 2026 (2024).

Table 5.8. PERMANOVA results comparing mean fish density between EFGB and WFGB study sites and among years from 2011 to 2025. **Bold text denotes significant value.**

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	1.6849E+05	12	6.9148	**<0.001**
Bank	15724	1	7.7436	**<0.001**
Year*Bank	63778	12	2.6175	**<0.001**
Res	1.1858E+06	584		
Total	1.4351E+06	609		

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 2006–2025 (Figure 5.8). PERMANOVA analysis indicated significant differences between banks, among years, and in the Year × Bank interaction (Table 5.9), indicating that temporal patterns in fish community biomass differed between EFGB and WFGB. The observed dissimilarity in community based on biomass between study sites from 2011–2025 was mainly attributed to brown chromis (6.79%) and blue tang (6.15%). 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).

To examine temporal patterns within each bank, separate PERMANOVA analyses were conducted for EFGB and WFGB using Year as the sole factor. Fish biomass differed significantly among years at both EFGB (Pseudo-F = 6.9447, $p < 0.001$) and WFGB (Pseudo-F = 7.5877, $p < 0.001$), indicating pronounced interannual change in fish biomass at both sites.

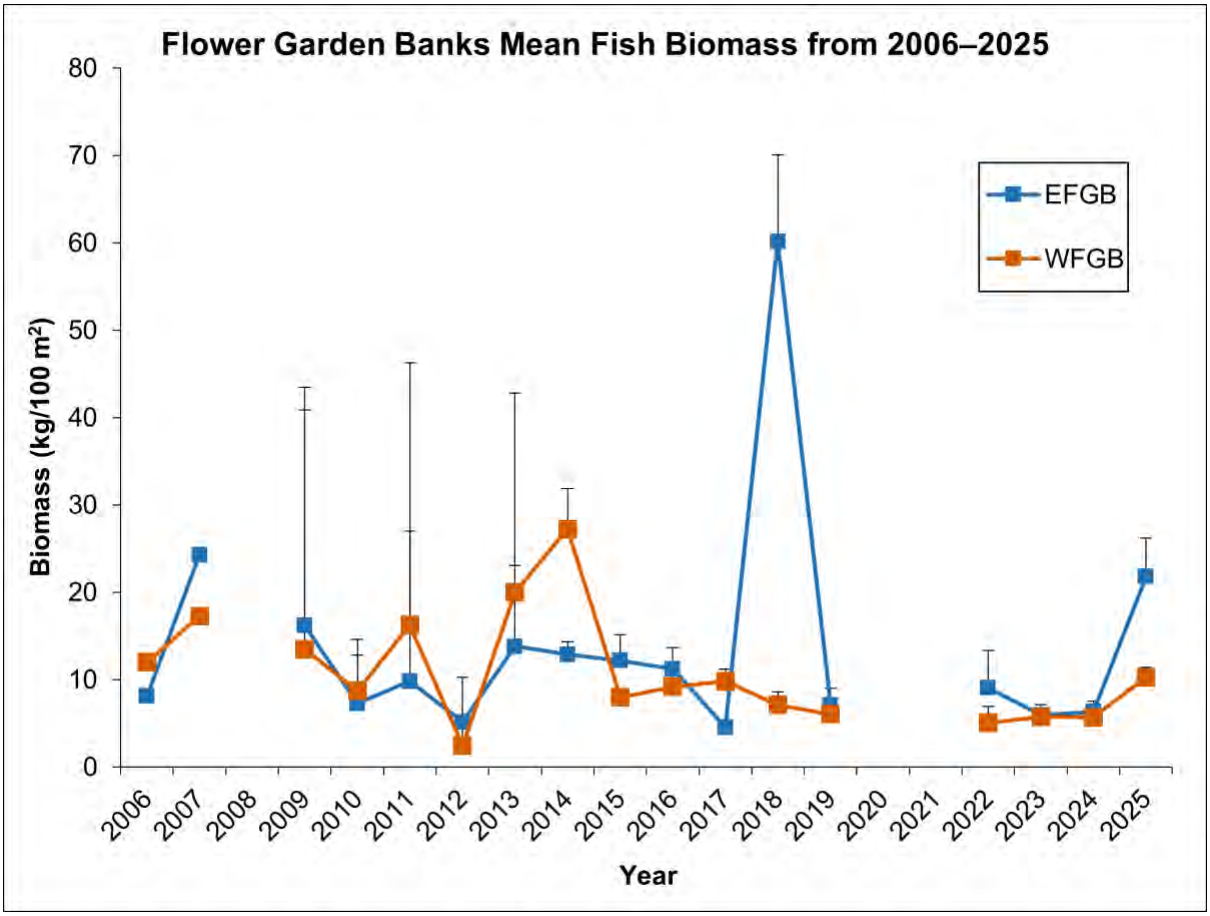


Figure 5.8. Mean fish biomass (kg/100 m2) + SE in EFGB and WFGB study sites from 2006 to 2025. No data were collected in 2008, 2020, and 2021, and SE was not available before 2009. Source: Precht et al., 2006; Zimmer et al., 2010 (2002–2008); Johnston et al., 2013, 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024 (2009–2022); Eisenbach et al., 2025 (2023); O'Connell et al., 2026 (2024).

Table 5.9. PERMANOVA results using mean fish biomass in EFGB and WFGB study sites and among years from 2011 to 2025. Bold text denotes significant values.

Source	Sum of Squares	df	Pseudo-F	P (perm)
Year	4.6117E+05	12	12.597	**<0.001**
Bank	9055.5	1	2.9682	**<0.001**
Year*Bank	70006	12	1.9122	**<0.001**
Residuals	1.7817E+06	584		
Total	2.3236E+06	609		

Additional analyses were conducted to investigate temporal trends in grouper and snapper density from 2011 to 2025, 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 study site surveys since 2015; however, black grouper were observed in reef-wide surveys in 2025 (Figure 5.9; Figure 5.10).

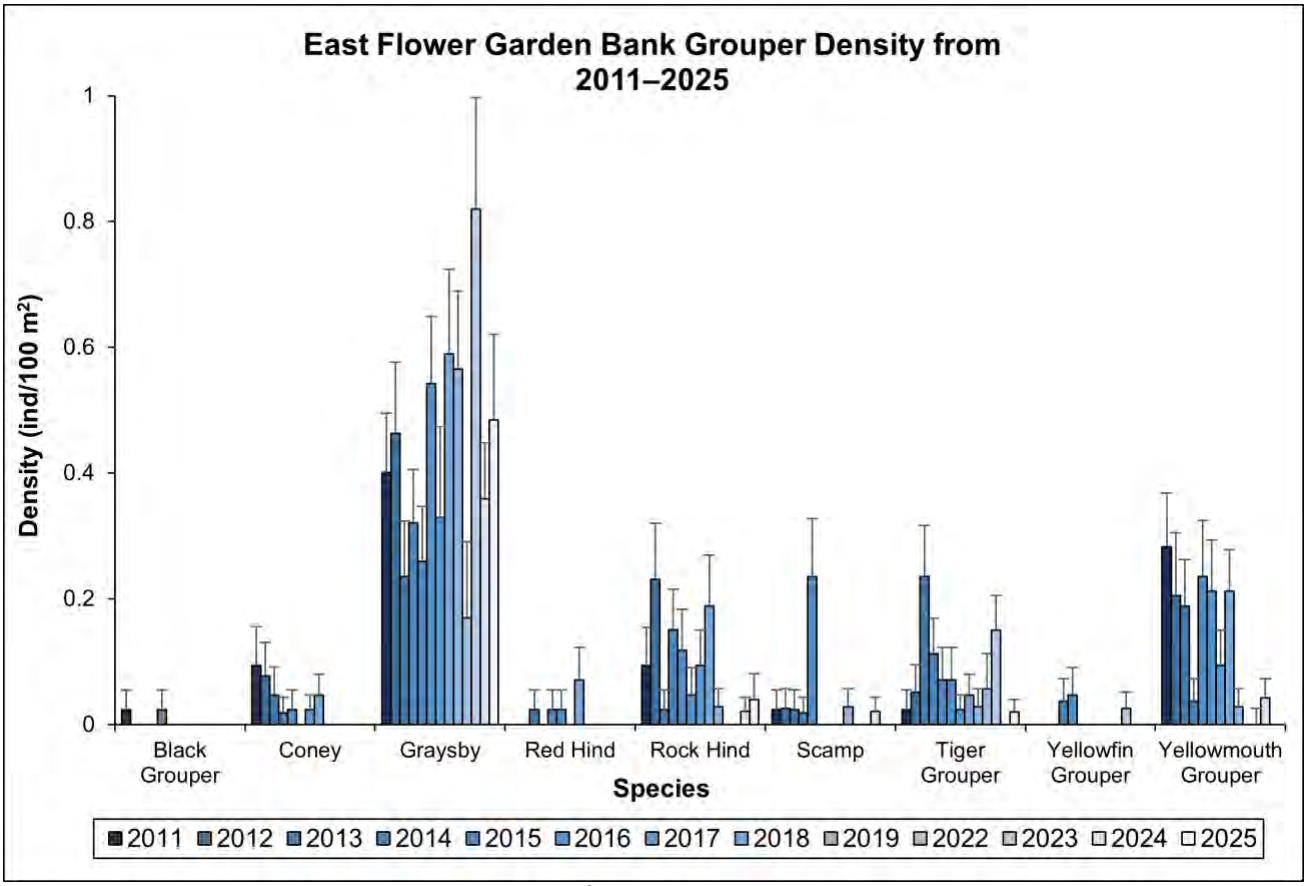


Figure 5.9. Mean density (individuals + SE/100 m^2) of grouper species within EFGB study site surveys from 2011 to 2025. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025; O'Connell et al., 2026.

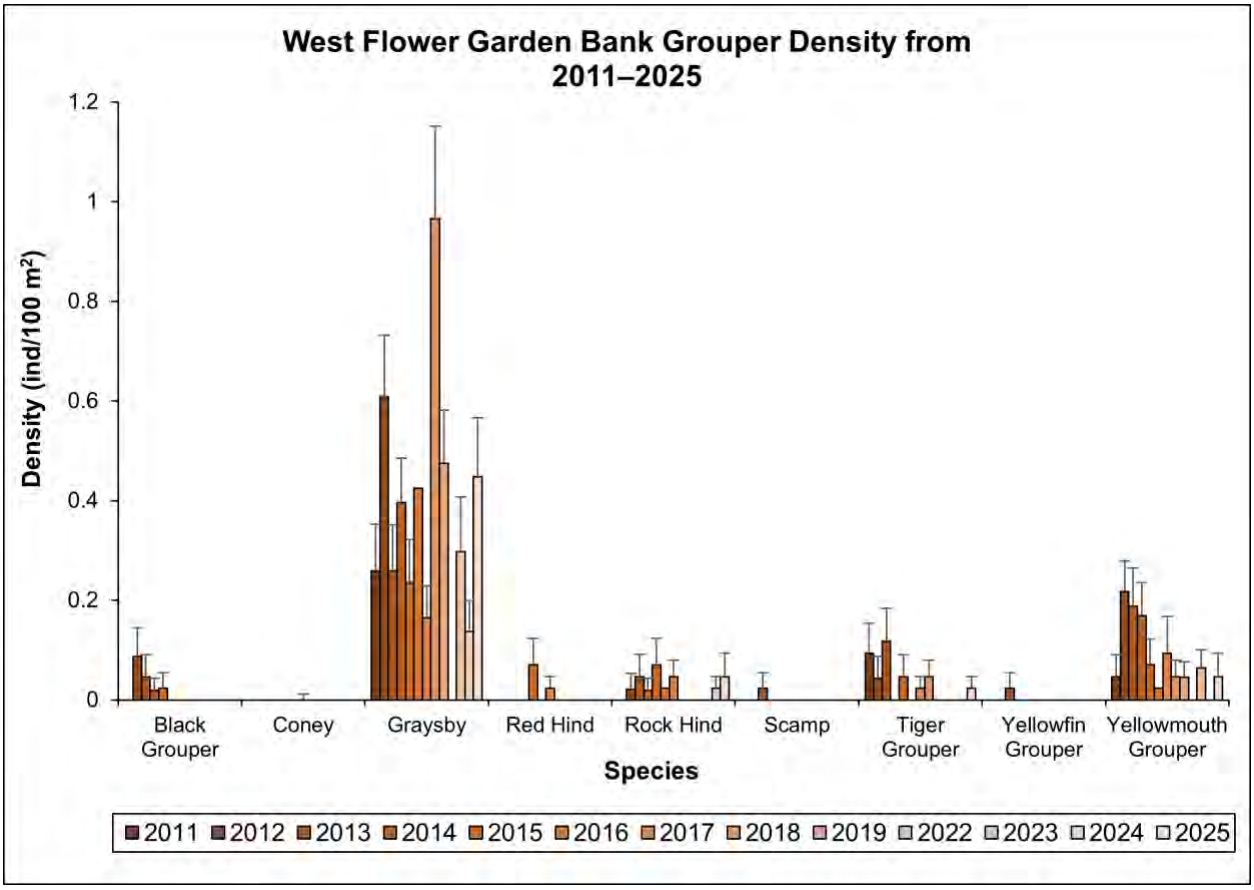


Figure 5.10. Mean density (individuals + SE/100 m2) of grouper species within WFGB study site surveys from 2011 to 2025. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025; O'Connell et al., 2026.

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 (τ = -0.03, p = 0.95) or WFGB (τ = -0.05, p = 0.85) from 2011–2025; however, there was a significant decline in large grouper density at both EFGB (τ = -0.64, p = 0.003) and WFGB (τ = -0.60, p = 0.006) from 2011–2025.

From 2011 to 2025, dog snapper were consistently denser in WFGB surveys and gray snapper were denser in most years (Figure 5.11; Figure 5.12). Mann-Kendall trend analysis revealed no significant trend in densities of either species at EFGB or WFGB from 2011–2025, suggesting stable snapper densities over the past 14 years.

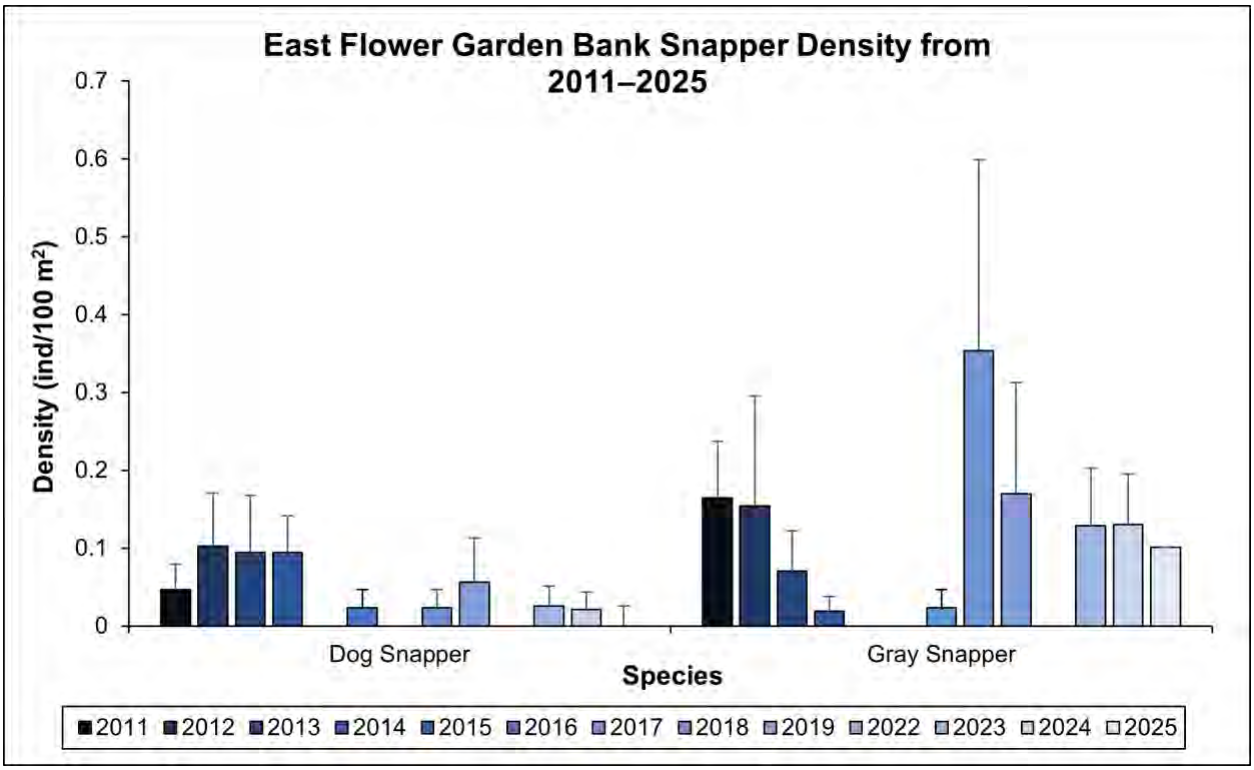


Figure 5.11. Mean density (individuals + SE/100 m²) of snapper species within EFGB study site from 2011 to 2025. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025; O'Connell et al., 2026.

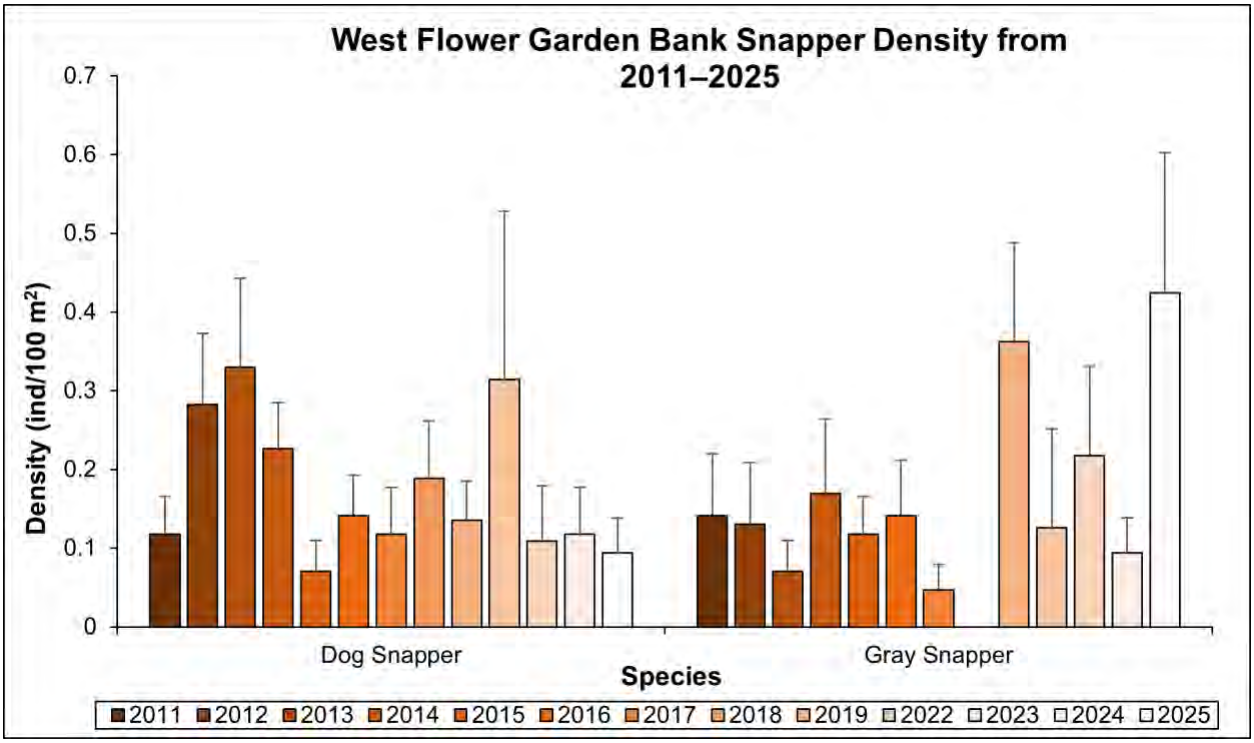


Figure 5.12. Mean density (individuals + SE/100 m²) of snapper species within WFGB study site from 2011 to 2025. Source: Johnston et al., 2015, 2017a, 2017b, 2018a, 2020, 2021b, 2024; Eisenbach et al., 2025; O'Connell et al., 2026.

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 2025, a combined total of 86 species were observed in 99 fish surveys in study site and reef-wide surveys 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 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 2025 study period, the piscivore guild possessed the highest mean biomass within both study site and reef-wide surveys at EFGB, contributing to 56% and 47% of the total biomass respectively. Similarly, piscivores possessed the highest mean biomass at WFGB in study site surveys contributing to 40% of the total biomass, while planktivores had the highest mean biomass in reef-wide surveys contributing to 44% total biomass reef wide. The difference at WFGB surveys can be contributed to the high biomass of great barracuda in study site surveys and Atlantic creolefish which had the highest biomass in reef-wide surveys. Historically, both banks were dominated by the piscivore guild, however, a shift to an herbivore dominated biomass was observed at EFGB and WFGB in 2023 and at EFGB in 2024. While WFGB has returned to a piscivore dominated biomass, WFGB has again shifted, but towards a planktivore dominated trophic guild biomass in 2025. Piscivore dominated reefs provide a top-down trophic structure supporting healthy ecosystems in fish community (Hempson, et al, 2018). This shift to higher biomass of planktivore species at WFGB, specifically in reef-wide surveys, alters the fish community trophic-structure. The greater biomass of planktivores was likely due to the high abundance of Atlantic creolefish which were observed in high numbers (greater than 100 fish) in multiple surveys in 2025.

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 m^2) at EFGB and WFGB study sites in 2025. The grouper species observed consisted of both juvenile and mature graysby, yellowmouth grouper, tiger grouper, yellowfin grouper, and scamp, and only juvenile rock hind and black grouper. A significant decrease in density of large-bodied groupers from 2011 to 2025 has been observed. This decrease may be due to increased fishing pressure and low recruitment. In 2025, more large grouper species were observed than in previous years, including black grouper which was last observed in surveys in 2015. This reappearance of multiple species is likely due to the surveys completed across the reef where divers are less present. Reef-wide surveys have not historically been performed, and therefore those surveys were not included in long-term analyses. A mix of juvenile and mature individuals of gray snapper and dog snapper were seen during surveys in 2025. 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 study sites was 1.006 kg per 100 m^2 in 2025, which is considered low. However, this value is not significantly different from several other Caribbean reefs, including sites in Mexico (1.592 kg/100 m^2), Belize (1.500 kg/100 m^2), and Honduras (2.423 kg/100 m^2; AGRRA, 2024). Although low parrotfish biomass is often associated with high fishing pressure or low coral cover, neither factor is present at EFGB or WFGB. Given the availability of food resources and limited fishing activity at both banks, the relatively low parrotfish abundance remains unexplained. However, parrotfish biomass at EFGB and WFGB has slowly increased since 2019 when parrotfish biomass was 0.740 kg/100m^2. The increasing of parrotfish biomass is likely due to successful recruitment during those years such as the possible recruitment in 2023 where a high abundance of juveniles was observed (Eisenbach et al., 2024) as well as the increasing macroalgae at EFGB and WFGB.

Chub species have also 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 are one of the top contributors 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. In 2025, chub species were not seen as frequently in surveys as in previous years, only observed in 39% of surveys at EFGB and 18% of surveys at WFGB compared to 58% frequency in surveys at EFGB and 50% at WFGB in 2024.

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 densities in 2019 (Johnston et al., 2021b). Lionfish were absent in 2022 surveys and have increased since then with two individuals in 2023 and eight observed in 2024 surveys. In 2025, a total of 15 lionfish were observed in both study site and reef-wide surveys at WFGB and only in reef-wide surveys at EFGB. 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, citizen science permitted lionfish removal cruises have been conducted during summer months 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 including the study sites at both EFGB and WFGB (Green et al., 2014). As a result, removals do not occur across large areas swaths of the reef where reef-wide surveys occurred. In 2025, two removal cruises occurred at EFGB and WFGB before monitoring occurred, and an additional two removal cruises occurred after monitoring; however, there were a low number of lionfish removed from EFGB and WFGB sites in the two cruises prior to monitoring. The higher number of lionfish observed in 2025 surveys may reflect the low number of targeted removals that occurred within the sanctuary during the first two removal cruises as well as surveys occurring reef-wide in areas where removal cruises

do not target. During the June 23–25, 2025 removal cruise, only 12 lionfish were removed at EFGB and WFGB and 41 during the July 21–23, 2025 removal cruise. In comparison, 20 lionfish were removed from EFGB and WFGB during the August 19–21, 2025 cruise and 73 during the September 22–24, 2025 removal cruise.

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. While there had been no sightings of regal demoiselles at EFGB or WFGB since 2019, there were sightings in 2025 surveys suggesting a reintroduction to these sites. Many regal demoiselles observed in 2025 at EFGB and WFGB were small juvenile individuals suggesting a recent recruitment and potential cyclical pattern of abundance on coral reefs. 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 divers prepare to redeploy the Sofar buoy at EFGB after it was detached during hurricane Francine. Photo: Captain Eric Walker/Up Above Adventures

Water Quality Introduction

Water quality parameters were continuously or periodically recorded at EFGB and WFGB from January through December 2025. Salinity, turbidity, and temperature were recorded hourly by moored data loggers installed in or near the study sites at depths of approximately 23 m (EFGB) and 27 m (WFGB). Dissolved oxygen was also monitored continuously at both banks through a partnership with University of Texas at Austin, Marine Science Institute. Additional temperature loggers deployed at repetitive photostations (approximately 30 m and 40 m depth at each bank) recorded hourly measurements. These loggers were recovered on July 31, 2025, and January 29, 2026.

Two Sofar Ocean® buoys provided supplemental temperature data. A buoy deployed at EFGB in June 2023 and redeployed in February 2025 (following loss during Hurricane Francine in 2024) recorded sea surface and reef crest temperatures. A second buoy deployed at WFGB in September 2025 recorded temperature at the surface and reef crest and included dissolved oxygen sensors at both depths. The EFGB buoy detached from its mooring in December 2025, after which only surface temperature data were available for the remainder of the year.

Water column profiles were collected in August 2025 using an SBE 19plus V2 CTD data carousel. Discrete water samples were collected for carbonate chemistry and ocean acidification analyses and processed by the University of Texas at Austin, Marine Science Institute. No samples were collected for organic analyses in 2025. This chapter presents data from moored instruments and water column profiles collected during 2025, with select recoveries extending into early 2026.

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 deployed at depths of 23 m at EFGB and 27 m at WFGB and mounted on anchors within sand flats at each study site (Figure 1.3). All sensors recorded measurements on an hourly basis.

Instruments were exchanged by divers in July 2024, July 2025, and January 2026 to facilitate data download and routine maintenance. Replacement instruments were deployed during each exchange to ensure continuous data collection. Following recovery, data were reviewed, sensors were cleaned and tested, and battery levels were verified. Hourly measurements were aggregated to daily averages for comparison across instruments and quality control.

In July 2024, one SBE 16plus deployed at EFGB was equipped with an SBE 63 Optical Dissolved Oxygen Sensor. This sensor recorded data through March 2025 before battery depletion. To supplement dissolved oxygen measurements, concentrations were estimated using temperature and salinity following established methods (Garcia et al., 1992), supported by continuous

monitoring conducted in partnership with the University of Texas at Austin, Marine Science Institute.

Onset® Computer Corporation HOBO® Pro v2 U22-001 (HOBO) thermographs were deployed alongside SBE 16plus instruments at each bank as backup temperature sensors. Additional HOBO loggers were installed at approximately 30 m and 40 m depths at each bank on permanent repetitive photostation markers. These loggers recorded temperature hourly and were recovered in July 2025.

Sofar Ocean Technologies® Spotter buoys with Smart Mooring systems provided supplemental temperature and dissolved oxygen data. The EFGB buoy, originally deployed in June 2023, was redeployed in February 2025 after loss during Hurricane Francine (September 2024) and remained operational until December 28, 2025, when the mooring failed again. A second buoy was deployed at WFGB on September 22, 2025. Both systems recorded sea surface conditions and subsurface temperature (23 m at EFGB; 27 m at WFGB) and transmitted data hourly via satellite. The WFGB buoy also recorded dissolved oxygen at both the surface and reef crest.

Sofar data were available in near real time and were processed into daily averages for analysis. These systems provided supplemental coverage and redundancy for temperature measurements and improved dissolved oxygen data availability.

Water Column Profiles

Water column profiles from the surface to the reef cap were acquired in August 2025 with a Sea-Bird® Electronics 19plus V2 CTD that recorded temperature, salinity, pH, turbidity, fluorescence, and dissolved oxygen every ¼ second. The carousel package included a Sea-Bird® 55 Frame Eco water sampler equipped with 12 four-liter Niskin bottles and a Sea-Bird® Electronics 19plus V2 CTD. Data were recorded following an initial three-minute soaking period after deployment, and the resulting profile data were processed to include only downcast data. The CTD was lowered and returned to the surface at a rate of <1 m s^{-1}.

Water Samples

Discrete water samples for carbonate chemistry were collected in August 2025 using a Sea-Bird® carousel system equipped with twelve 4-L Niskin bottles during CTD profiling operations. Samples were collected at three depths, including near the surface (~1 m), midwater (~10 m), and near the reef cap (~20 m), with a blind duplicate collected for quality control.

Routine quarterly sampling for chlorophyll *a* and nutrients, conducted as part of the long-term monitoring program since 2002, was not completed in 2025 due to vessel and equipment limitations. As a result, nutrient and chlorophyll *a* data are not available for this reporting year. Carbonate chemistry samples collected in August 2025 were processed by the University of Texas at Austin, Marine Science Institute.

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, 2025 and January, 2026. Sofar Ocean® Spotter Buoy data from EFGB was processed

monthly from January until the buoy detached in December while the Sofar Ocean® Spotter Buoy data from WFGB was processed at the time of its deployment in September through the end of 2025. 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 Sea Surface Temperature, NOAA Coral Reef Watch Daily Global 5km Satellite SST (CoralTempt), 1985-present, Daily, while SSS data were sourced from the Sea Surface Salinity, SMAP, Near Real-Time Global 0.25°, 2015-present, Daily. SST data were available as daily means on a 0.01° global grid, and SSS data as daily mean 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 2026. Daily dissolved oxygen values were taken in situ at WFGB using the Sofar Ocean® Spotter Buoy and also 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 20.06 °C to 31.11 °C in 2025, while the Sofar Ocean® surface sensor at recorded values from 21.26 °C to 31.56 °C (January–December 28). At 23 m, temperatures ranged from 21.10 °C to 30.60 °C (Figure 6.1). Measurements from the co-located HOBO logger and Sofar Ocean® subsurface sensor were consistent with the SBE 16plus records (Figure 6.1). HOBO loggers recorded temperature ranges of 21.05–30.28 °C at 30 m and 20.99–29.25 °C at 40 m (January–July).

At WFGB, satellite SST ranged from 20.06 °C to 31.11 °C in 2025. At 27 m, temperatures recorded across instruments ranged from 20.52 °C to 30.70 °C (Figure 6.1), with HOBO and Sofar measurements (September–December) consistent with SBE 16plus data. At 30 m and 40 m, HOBO loggers recorded temperature ranges of 20.40–29.90 °C and 20.45–28.59 °C, respectively (January–July).

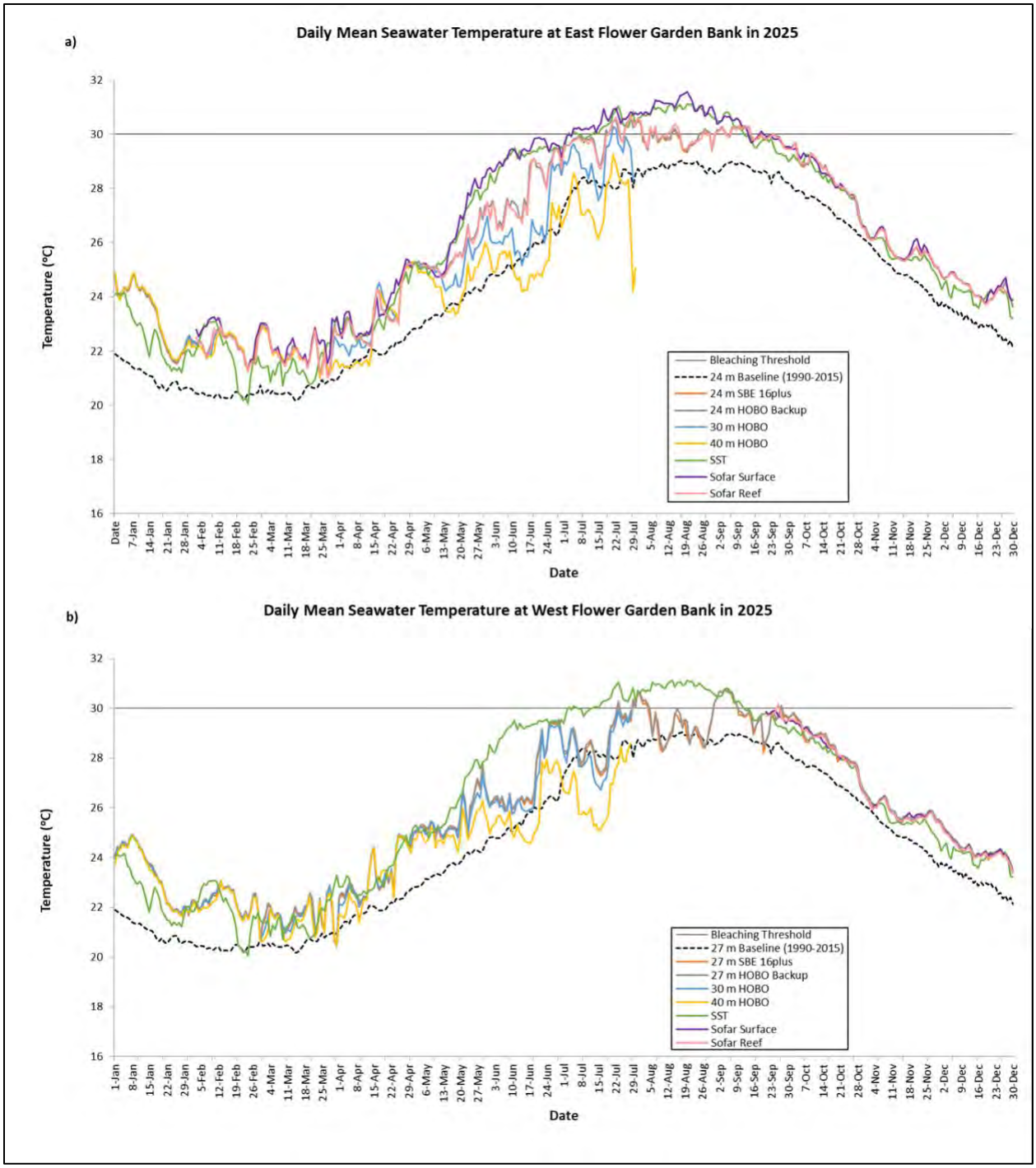


Figure 6.1. Daily mean seawater temperature (°C) at (a) EFGB and (b) WFGB from various depths in 2025, as well as the 25-year daily mean water temperature baseline. The solid black line at 30 °C is a level known to trigger coral bleaching.

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 records indicate increasing temperatures at both the surface and reef crest at each bank, with rates of warming at depth exceeding those observed at the surface (Figure 6.2).

Seasonal-Kendall trend tests were applied to (1) time series sea surface temperatures derived from satellite observations (OISST V2.1, AVHRR-only, daily 1/4°; 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 2025 in surface waters at EFGB ($\tau = 0.38$, $z = 11.36$, $p < 0.001$) and WFGB ($\tau = 0.36$, $z = 11.02$, $p < 0.001$), as well as at depth from datasondes at EFGB (23 m; $\tau = 0.37$, $z = 8.95$, $p < 0.001$) and WFGB (27 m; $\tau = 0.35$, $z = 9.01$, $p < 0.001$), after adjusting for seasonal correlation (Figure 6.2).

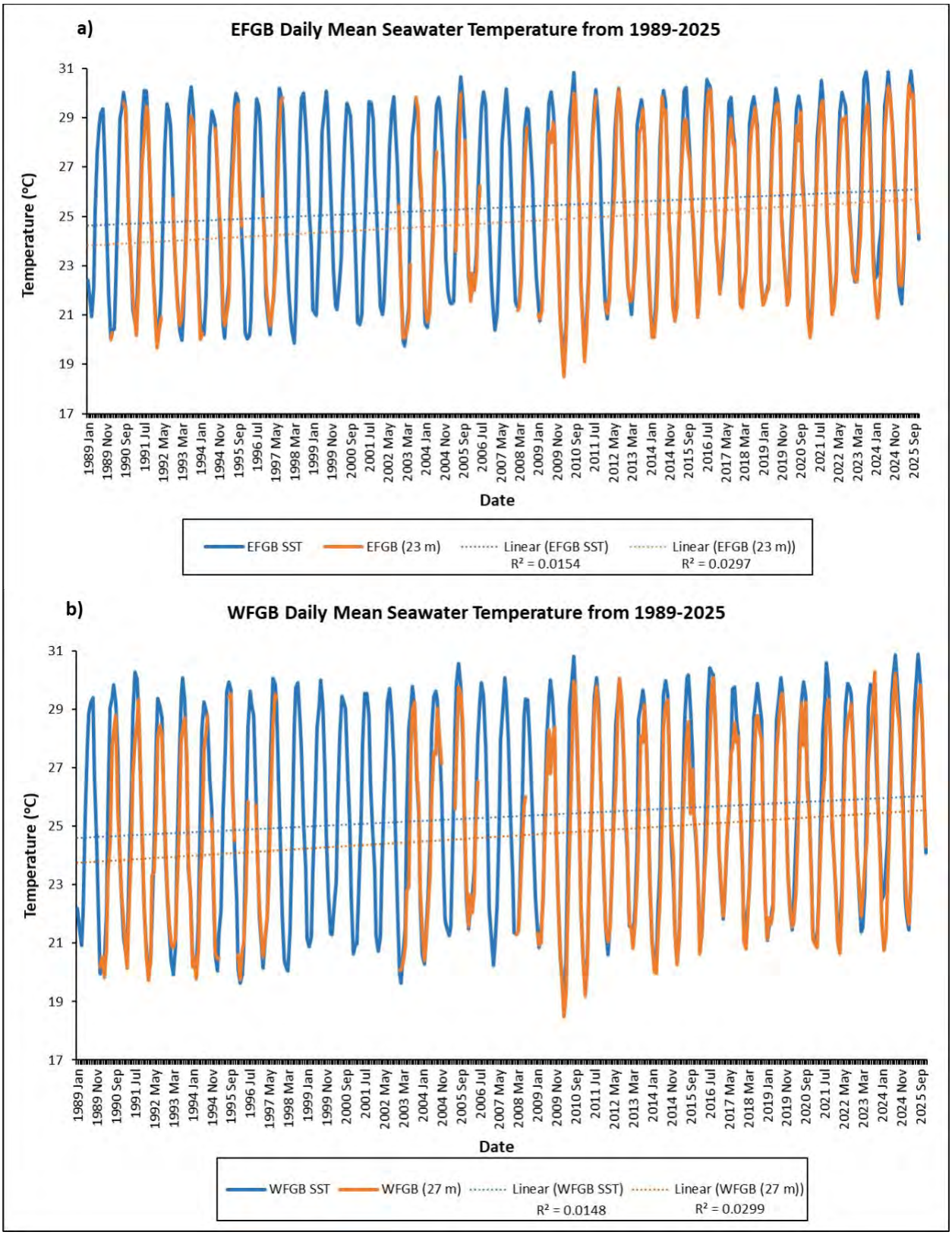


Figure 6.2. Daily mean seawater temperature (°C) demonstrates 12-month seasonal variation from various depths at (a) EFGB and (b) WFGB from 1989 to 2025, as well as a significant increase over time (trend lines).

Although bleaching surveys were not conducted during peak thermal stress (see Chapter 3), observed bleaching and paling prevalence in 2025 was lower relative to 2023–2024. Temperature patterns during summer 2025 (Figure 6.3) show increased variability compared to prior years, with frequent fluctuations around the 29 °C and 30 °C thresholds rather than sustained exceedances. Periods above these thresholds were intermittent and shorter in duration, particularly at WFGB. Long-term heatmaps of daily temperature at EFGB 23 m and WFGB 27 m (Figure 6.4) illustrate the seasonal cycle of warming and cooling from 2009 through 2025, with prolonged periods of elevated temperatures in recent years. These data were adjusted to daily averages and compiled from all available in situ sensors at EFGB and WFGB deployed at approximately 23 m and 27 m depth respectively, including Sea-Bird 16plus instruments, HOBO loggers, and the Sofar Ocean® Smart Mooring. These patterns are consistent with the increasing thermal stress observed at the sanctuary and align with statistically significant warming trends reported in this chapter.

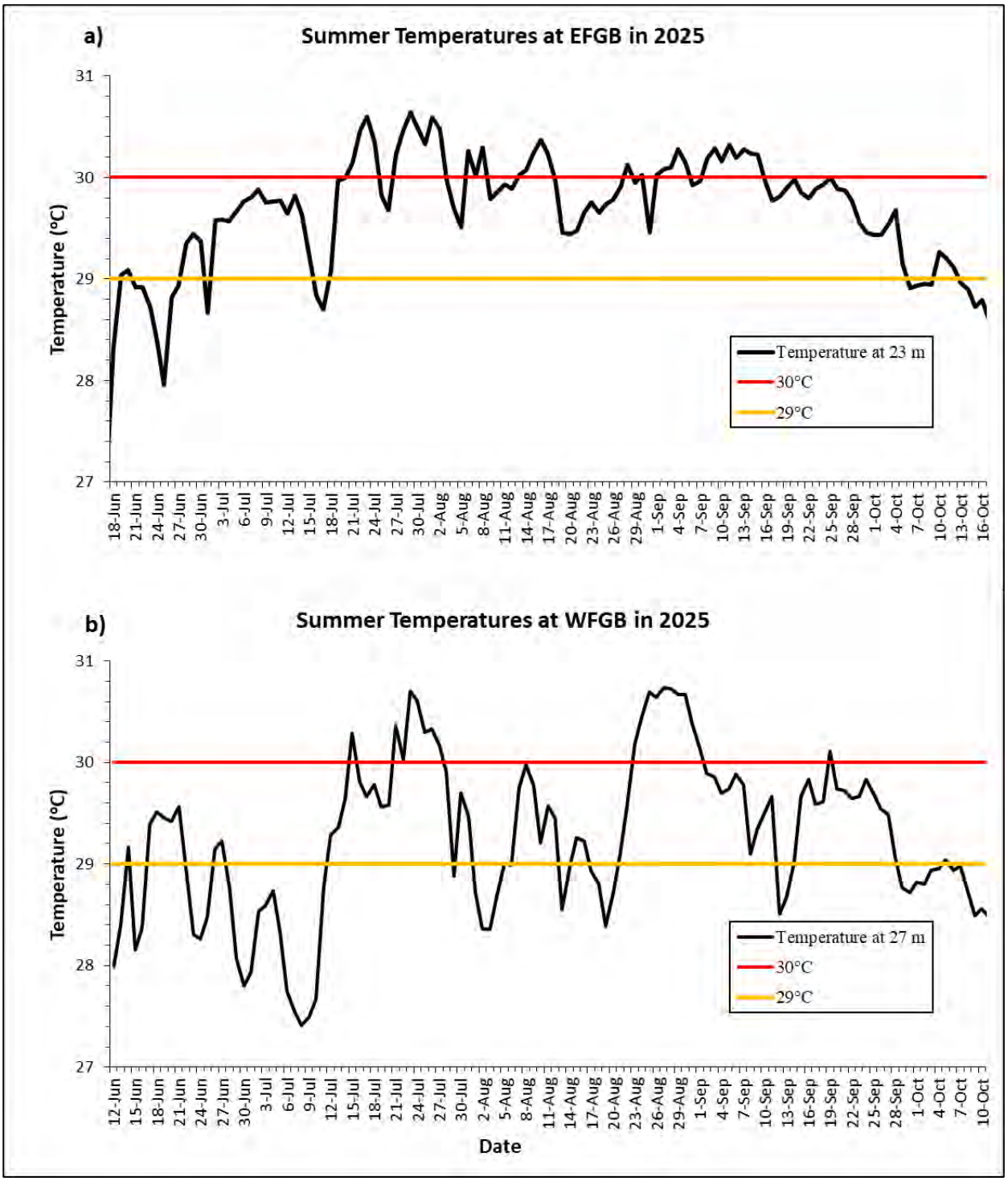


Figure 6.3. Temperature data at (a) EFGB at reef crest shows 102 days over 29 °C and 35 days over 30 °C and at (b) WFGB at reef crest shows 74 days over 29 °C and 19 days over 30 °C.

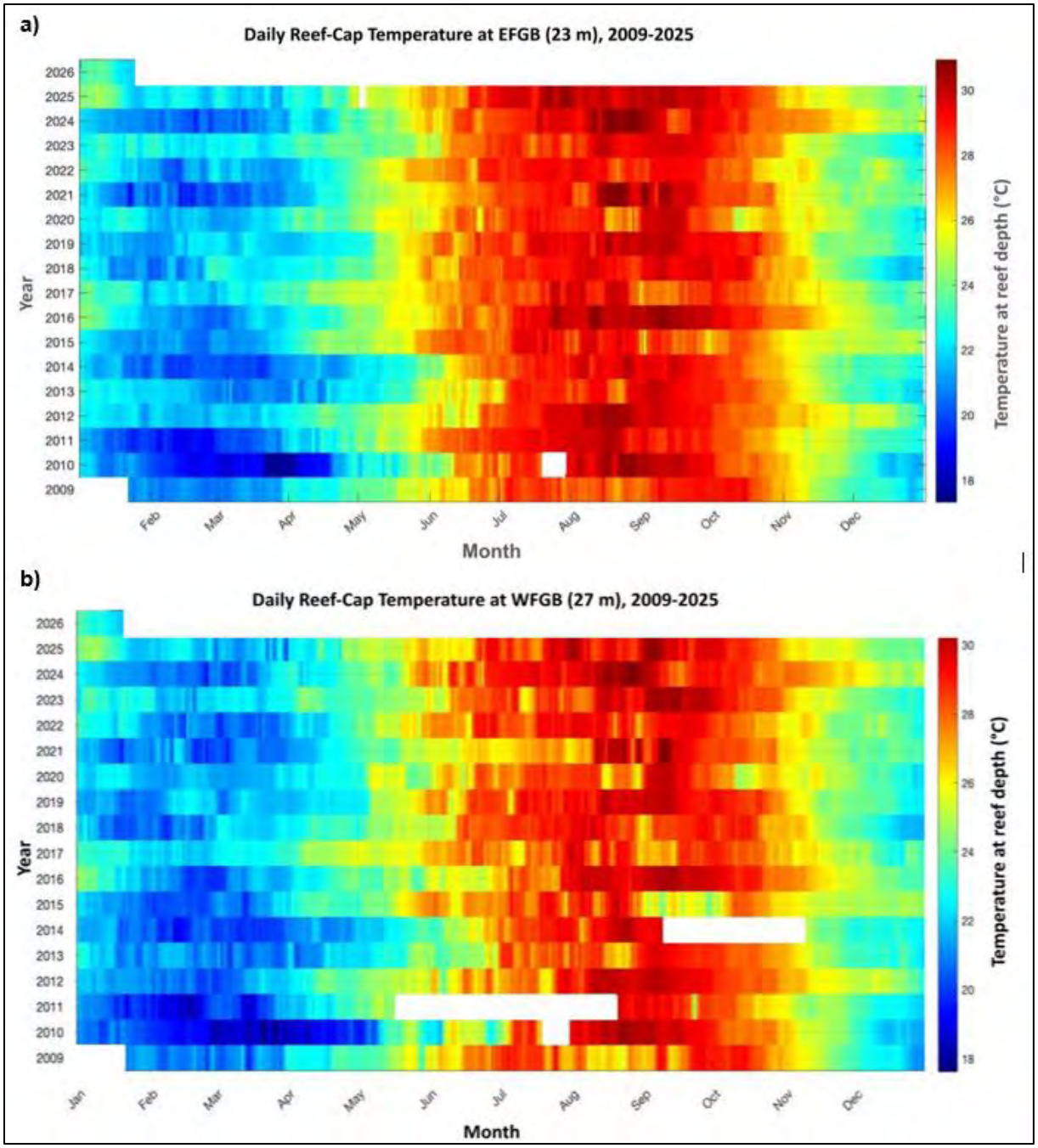


Figure 6.4. Heatmap showing daily temperature at reef depth (23 m) at EFGB (a) and WFGB (b) from 2009 to 2025. Warmer colors indicate higher temperatures. Figure: Xingping Hu/University of Texas

Salinity

In 2025, salinity at EFGB ranged from 34.12 to 38.36 psu at the surface and 34.13 to 36.81 psu at 23 m (Figure 6.5). At WFGB, salinity ranged from 34.12 to 38.36 psu at the surface and 35.33 to 36.81 psu at 27 m (Figure 6.5). The SBE 16plus at EFGB lost power on March 9, 2025 due to battery depletion and was replaced on July 31, 2025. During this period, temperature-salinity data were supplemented using satellite-derived sea surface salinity and Sofar Spotter buoy

temperature. Although the supplemented salinity data exhibit variability, they are consistent with the range of salinity values recorded by the SBE 16plus prior to instrument failure and do not appear to account for the observed differences in spring salinity patterns between EFGB and WFGB. Temperature-salinity relationships (Figure 6.5) show seasonal structuring at both banks, with lower temperatures and a broader range of salinity values during early months and warmer, more constrained salinity ranges during summer. EFGB exhibits a wider spread in salinity across temperatures, while WFGB shows a more defined seasonal transition from cooler, lower-salinity conditions to warmer, higher-salinity conditions.

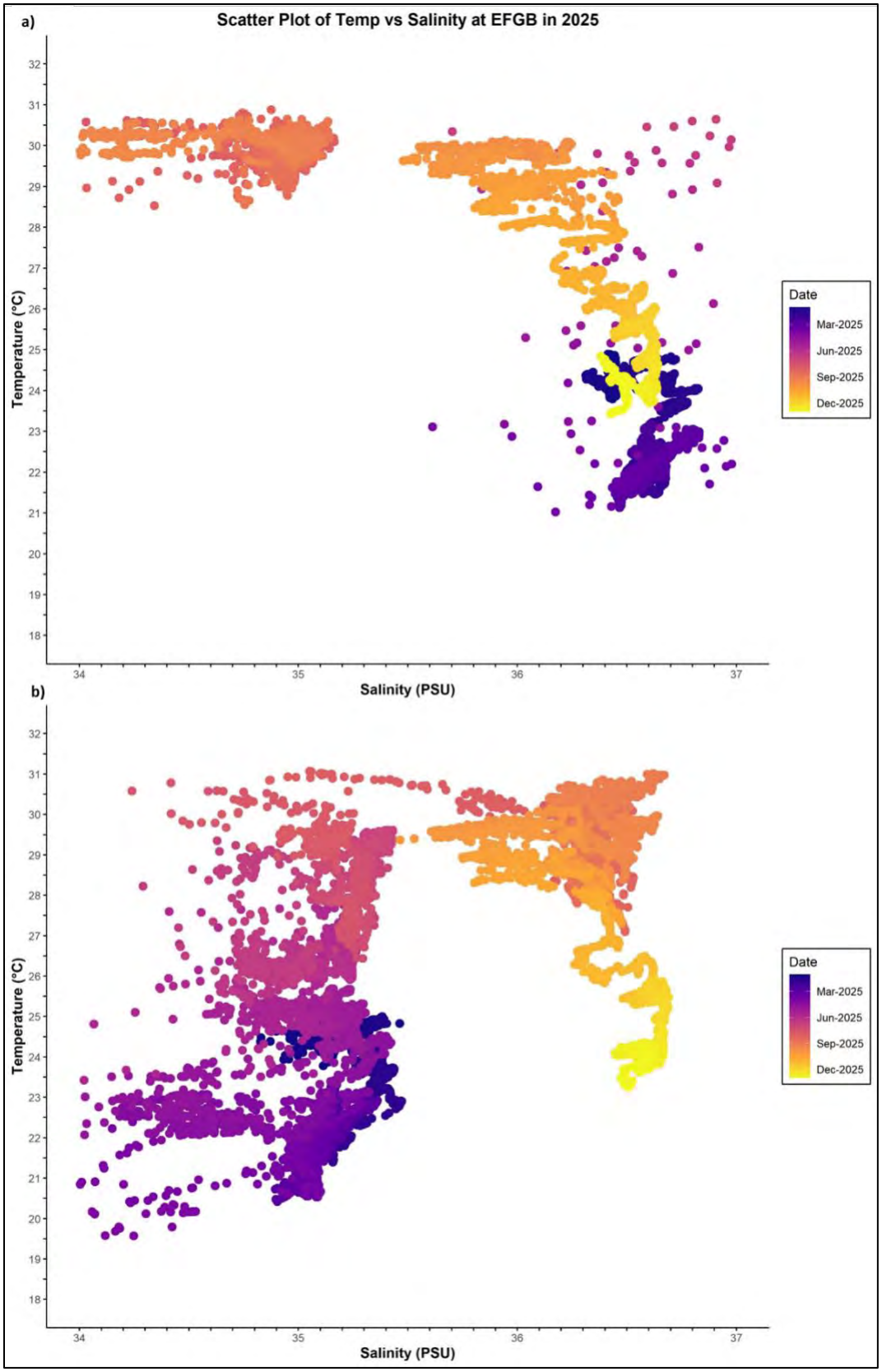


Figure 6.5. Hourly salinity (psu) against hourly temperature recorded using SBE 16plus at the reef cap station at (a) EFGB and (b) WFGB in 2025.

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 2025 indicated weak negative trends at both EFGB (τ = -0.15, z = -2.89, p = 0.16) and WFGB (τ = -0.12, z = -2.28, p = 0.28); however, neither trend was statistically significant.

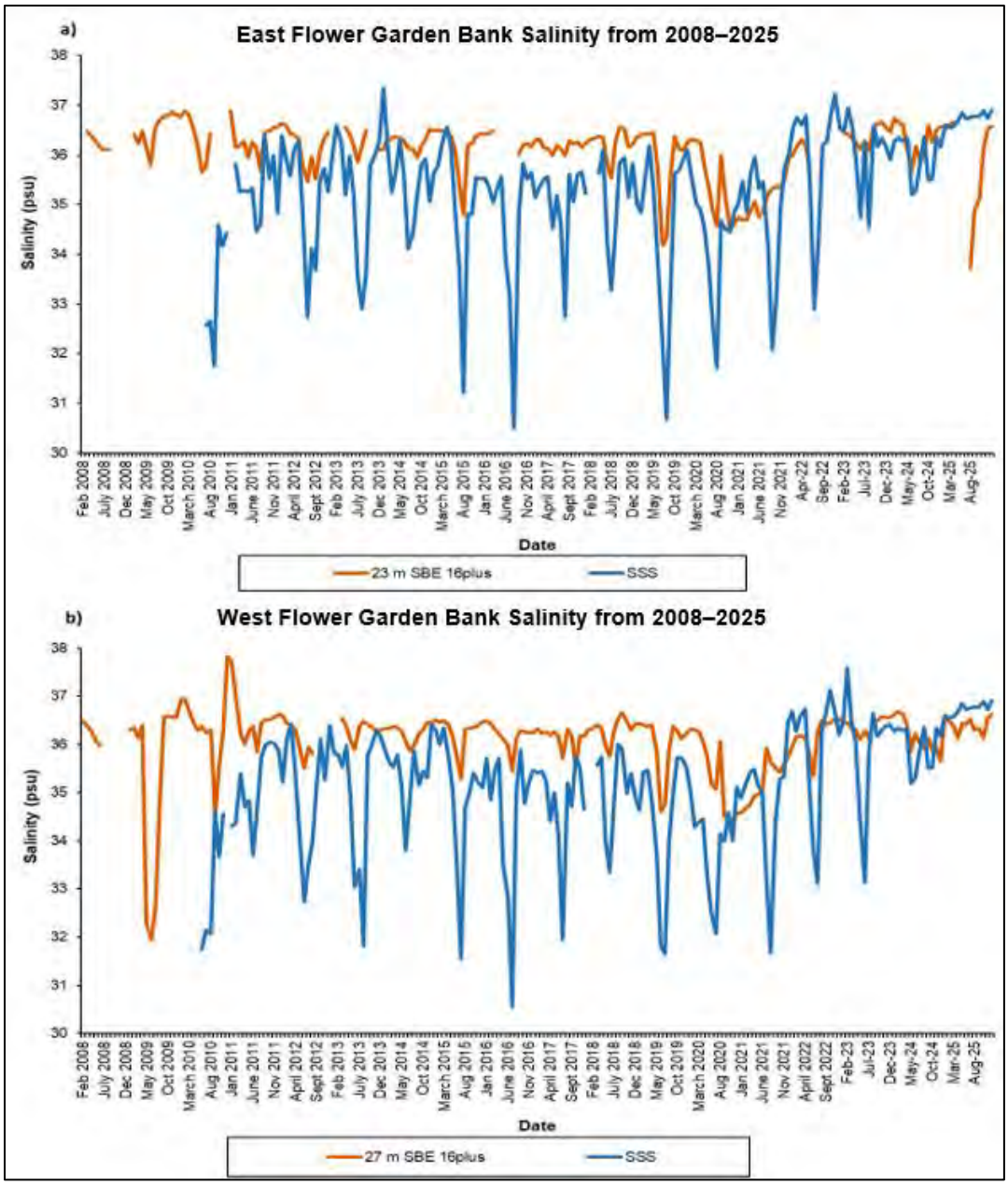


Figure 6.6. Monthly mean salinity showing seasonal variation and long-term trends at (a) EFGB (23 m) and (b) WFGB (27 m) from 2008 to 2025.

Turbidity

Turbidity sensors at EFGB and WFGB were exchanged in July 2024 and July 2025. The sensor at EFGB ceased recording in March 2025 due to battery depletion. At WFGB, turbidity was elevated from May through July before returning to lower levels by September (Figure 6.7). Overall, turbidity remained low and within expected ranges for offshore reef environments. Values ranged from 0.00 to 1.69 NTU at EFGB and from 0.05 to 6.12 NTU at WFGB (Figure 6.7).

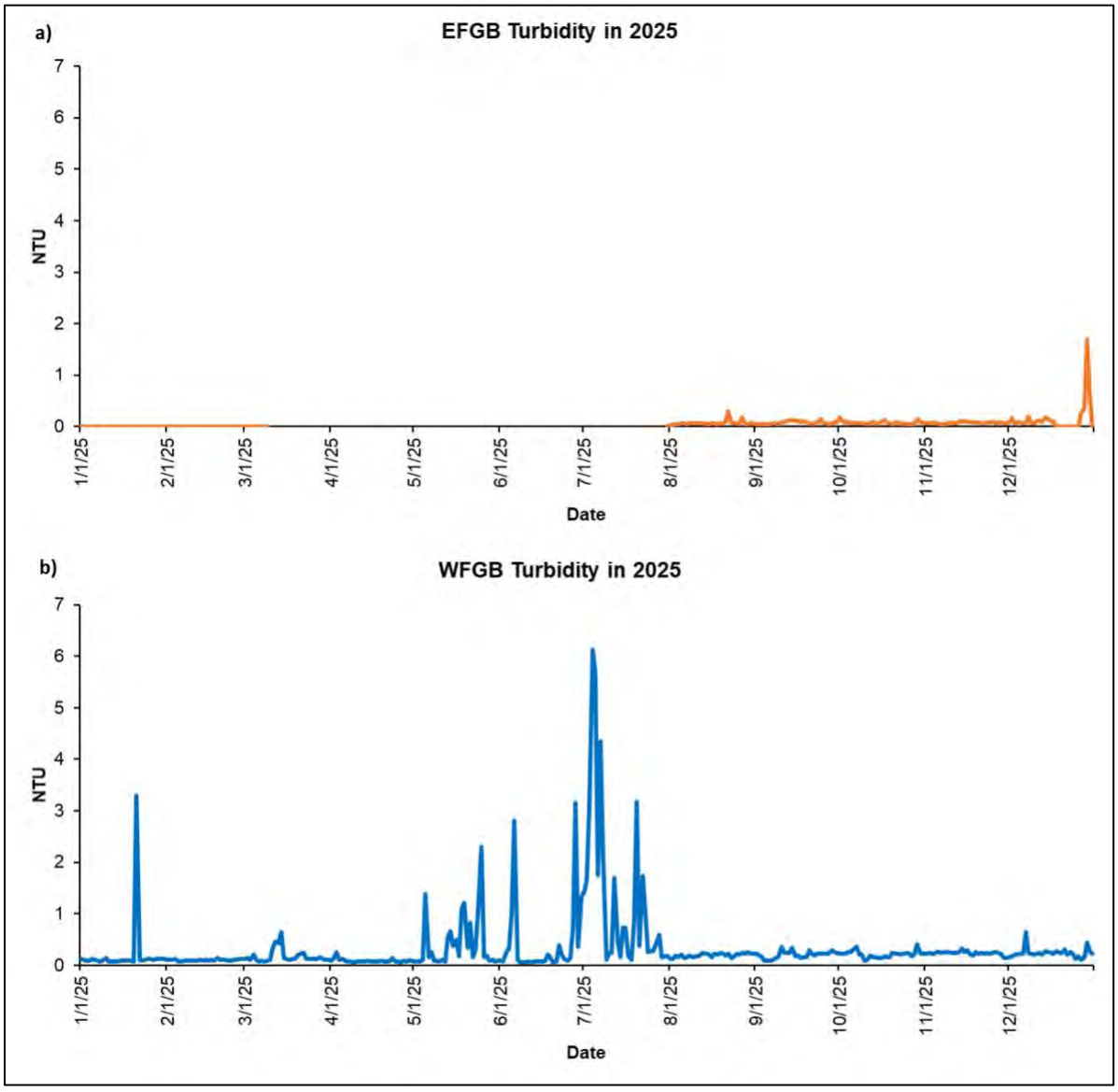


Figure 6.7. Daily mean turbidity (NTU) in 2025 at East Flower Garden Bank (a; 23 m) and West Flower Garden Bank (b; 27 m). Turbidity data at EFGB are unavailable from March through July due to instrument failure and subsequent replacement. At WFGB, turbidity remained low for most of the year, with elevated values observed from May through July.

Dissolved Oxygen

A dissolved oxygen sensor was deployed at EFGB in July 2024 and exchanged in July 2025. The sensor recorded data through March 2025 before battery depletion, resulting in a partial dataset for measured dissolved oxygen. To provide full-year coverage, dissolved oxygen concentrations were calculated from daily averaged temperature and salinity following Garcia et al. (1992). Calculated values ranged from 186.55 to 219.41 μmol kg^{-1} at EFGB and from 186.89 to 222.58 μmol kg^{-1} at WFGB in 2025 (Figure 6.8a).

Dissolved oxygen data were calculated from in situ sensors and temperature-salinity relationships at EFGB (23m) and WFGB (27m) extending back to 2015, with occasional data gaps due to equipment malfunctions (Figure 6.9). Dissolved oxygen exhibited a pronounced seasonal cycle, with higher concentrations during cooler months and lower concentrations during periods of elevated temperature (Figure 6.8). This pattern reflects a consistent inverse relationship between temperature and dissolved oxygen at both banks. Interannual variability is limited, and linear trend analyses indicate no substantial long-term change in dissolved oxygen at either bank over the period of record.

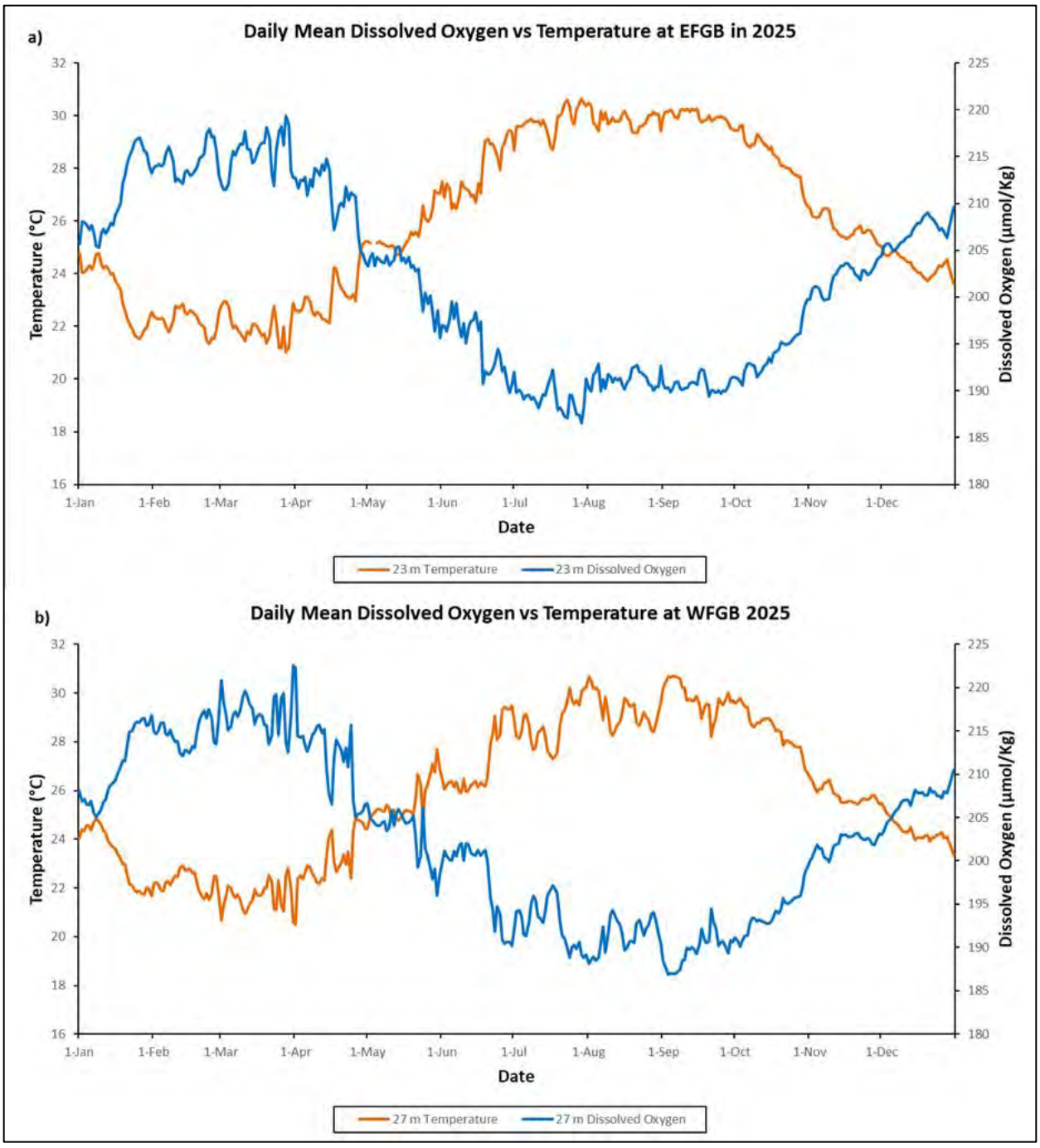


Figure 6.8. Daily mean dissolved oxygen (µmol/Kg) values from EFGB (23 m) and WFGB (27m) in 2025.

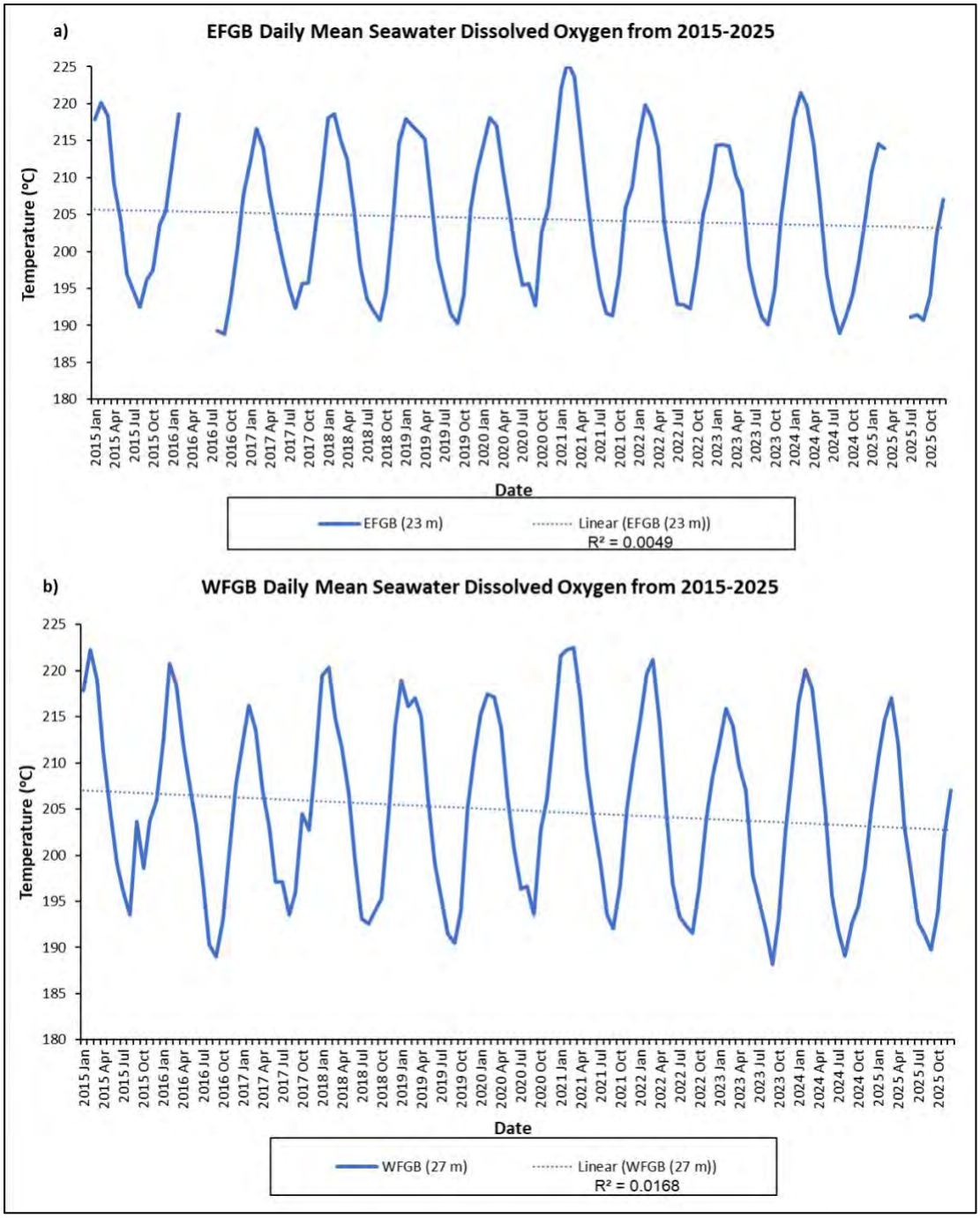


Figure 6.9. Monthly mean dissolved oxygen showing seasonal variation and long-term trends at (a) EFGB (23 m) and (b) WFGB (27 m) from 2015 to 2025.

Water Column Profiles

Only one water column profile was collected at each bank in 2025 due to limited offshore access and instrument operability. Water column temperatures at both EFGB and WFGB varied by less than 2 °C through 22 m depth, indicating well-mixed conditions (Figure 6.10). Salinity values were broadly similar between banks but exhibited depth-dependent stratification. The greatest differences occurred at mid-depth (~8–10 m), while values converged near the surface and reef cap. Dissolved oxygen concentrations were similar between banks and showed minimal variation with depth (<0.5 mL L^{-1}), consistent with well-mixed conditions. pH remained stable across depths at both banks, with variation less than 0.2 units (Figure 6.10).

Turbidity remained low overall at both banks, with the highest values occurring in the upper water column and decreasing with depth (Figure 6.10). A pronounced shallow turbidity peak was evident at WFGB, whereas EFGB showed lower and more gradual variation with depth. Elevated values in the upper water column may reflect short-term variability or measurement artifacts (e.g., air bubbles during CTD deployment). Fluorescence was relatively uniform with depth at EFGB, while at WFGB values were elevated near the surface and decreased with depth (Figure 6.10).

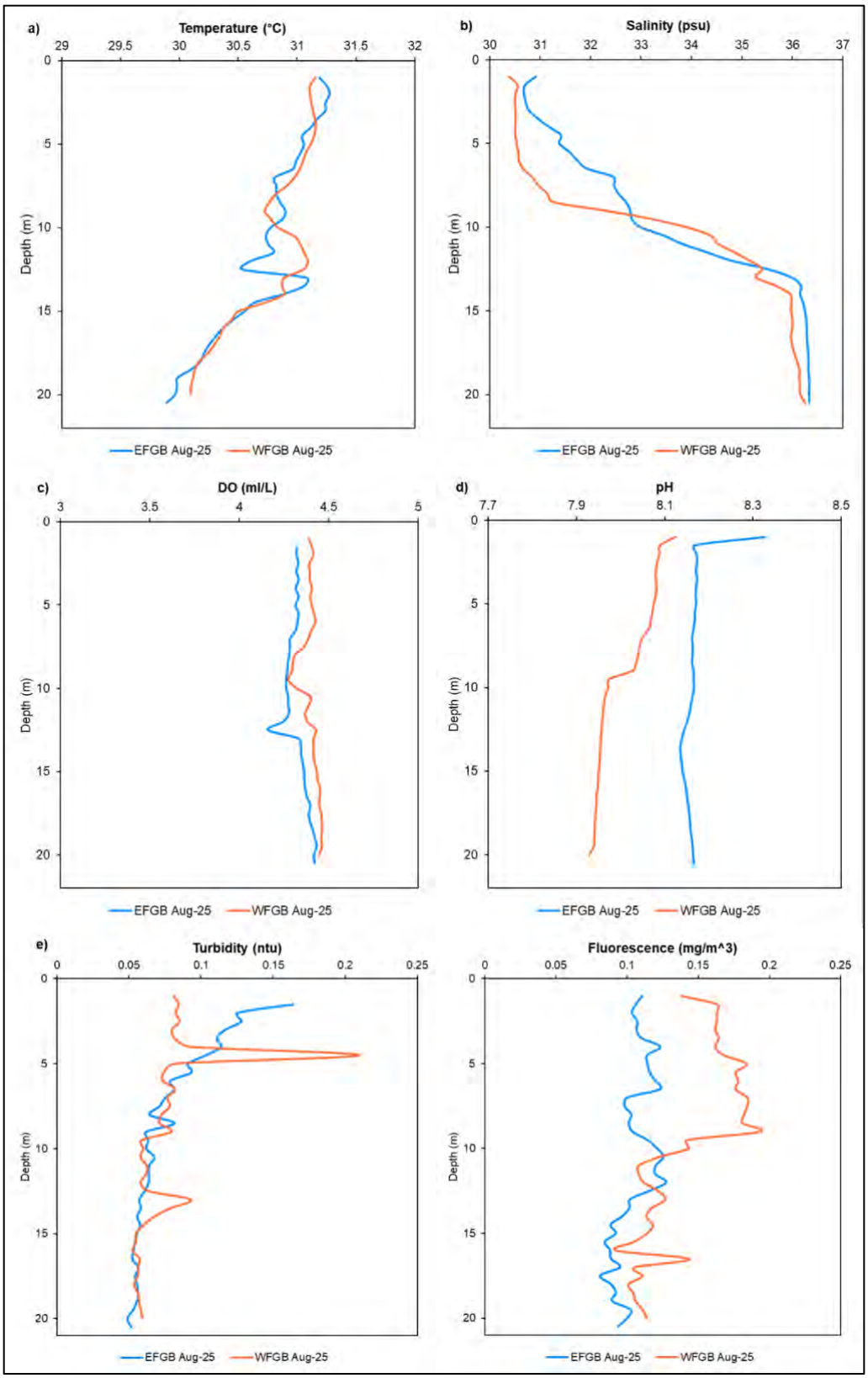


Figure 6.10. (a) Temperature, (b) salinity, (c) dissolved oxygen (DO), (d) pH, (e) turbidity, and (f) fluorescence water column profile data from EFGB and WFGB in August 2025.

Water Samples

Discrete chlorophyll *a* and nutrient sampling has been conducted as part of the long-term monitoring program since 2002. Nutrient concentrations have generally remained below detection limits, with occasional detections of ammonia and total Kjeldahl nitrogen prior to 2012 (Figure 6.11; Figure 6.12). No chlorophyll *a* or nutrient samples were collected in 2025 due to vessel and equipment limitations that restricted deployment of water sampling systems. Carbonate chemistry sampling, typically conducted in conjunction with nutrient collections, was performed once in August 2025. Despite the absence of sampling in 2025, the dataset represents more than two decades of near-continuous monitoring at EFGB and WFGB.

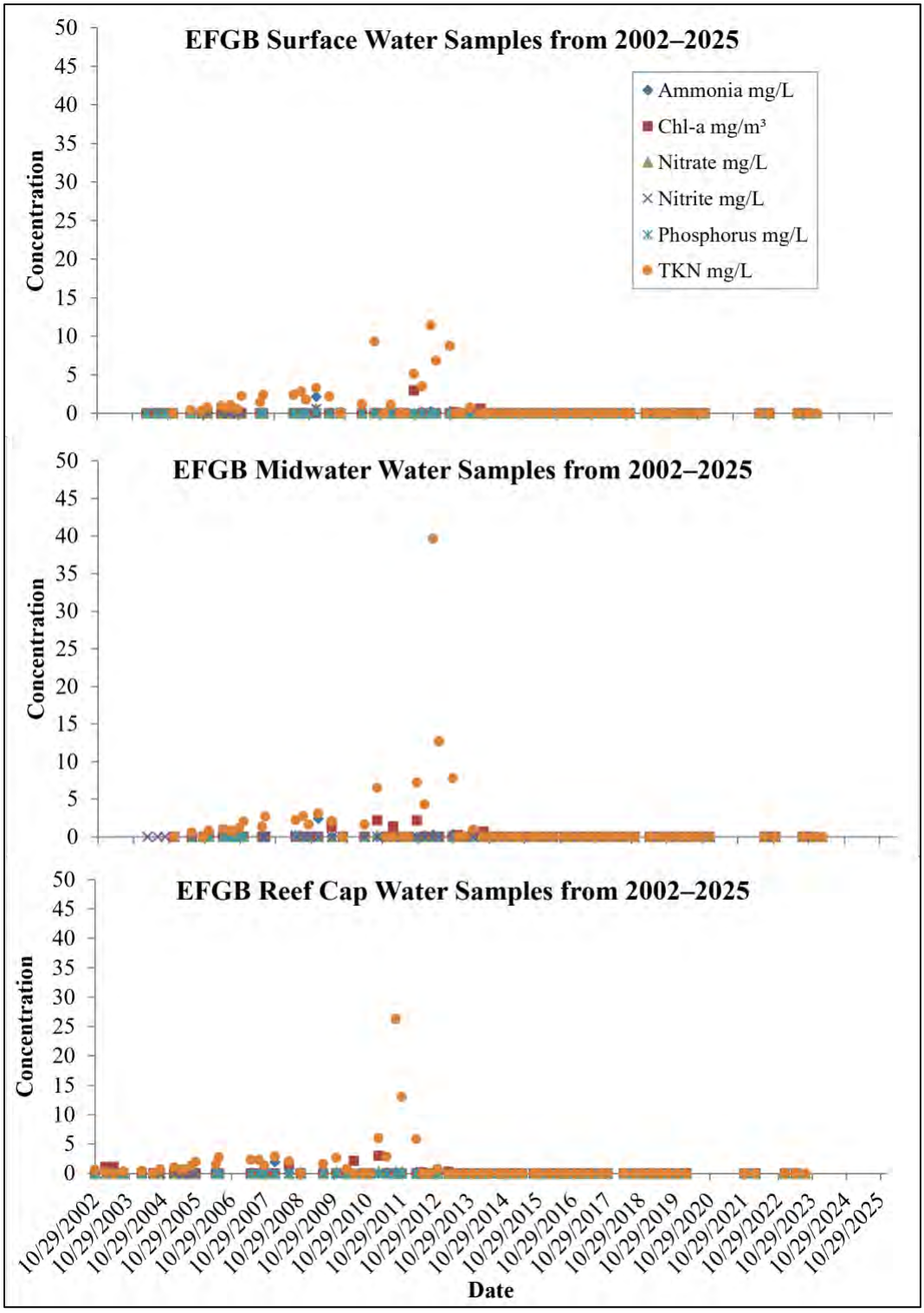


Figure 6.11. Nutrient concentrations from EFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2002 through 2025. While the 2025 axis is shown to preserve the full monitoring timeline, no water samples were collected in 2025 due to vessel and sensor limitations.

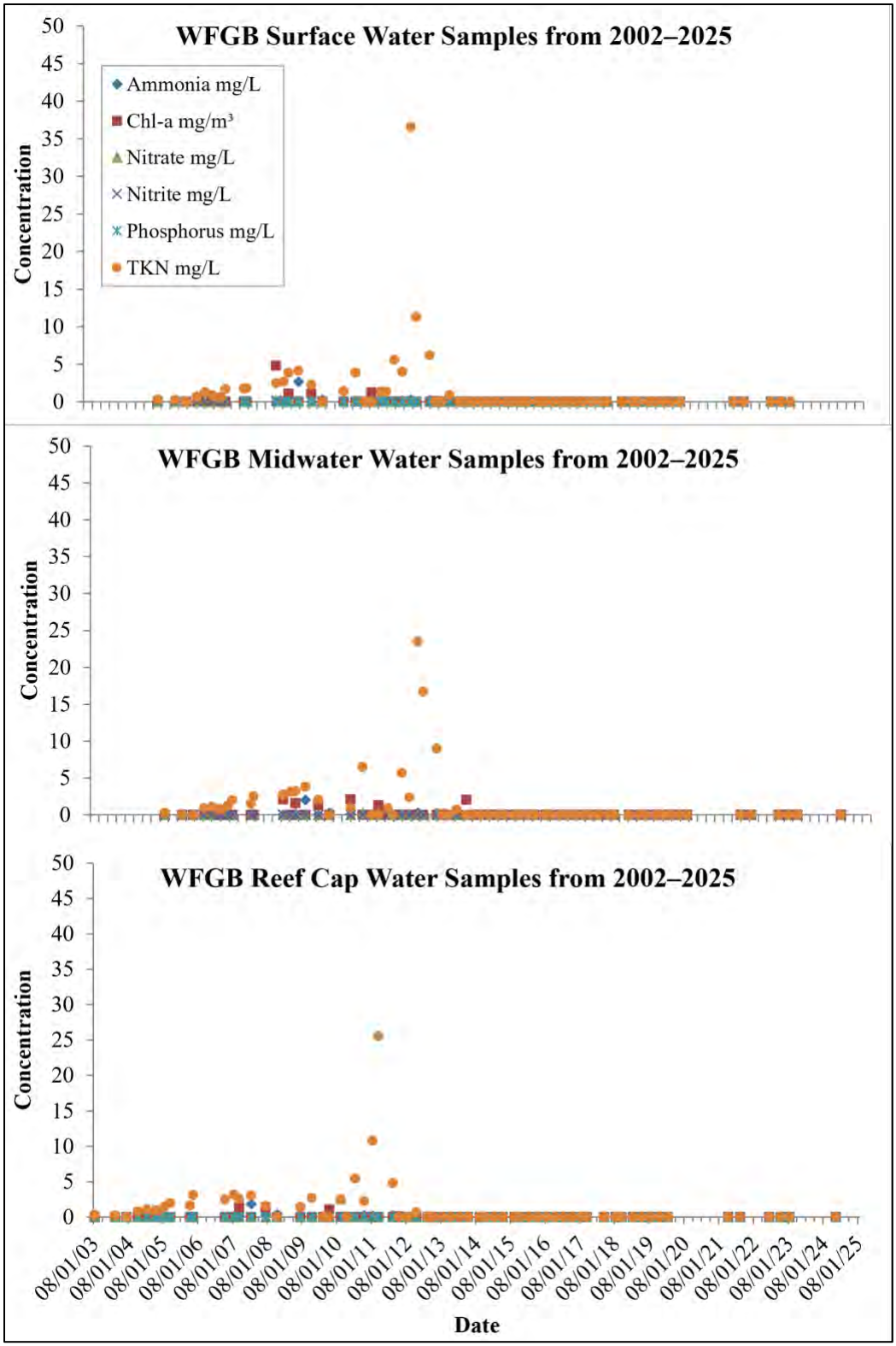


Figure 6.12. Nutrient concentrations from WFGB water samples taken at the surface (~1 m), midwater (~10 m), and reef cap (~20 m) from 2003 through 2025. While the 2025 axis is shown to preserve the full monitoring timeline, no water samples were collected in 2025 due to vessel and sensor limitations.

Water Quality Discussion

Water quality monitoring in 2025 was limited by the extended unavailability of the R/V *Manta* during the offseason, 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 during July through September. Long-term temperature trends at both banks continue to show significant monotonic increases, with reef cap and surface waters warming steadily since 1990.

Mean SSS fluctuated at both banks, while salinity at reef depth remained relatively consistent with long-term averages during the first half of the year. Despite seasonal variability, salinity at depth remained within the expected range for coral reef ecosystems in the western Atlantic (31 to 38 psu; Coles & Jokiel, 1992). As in previous years, lower-salinity surface waters likely reflect periodic influence from nearshore riverine inputs originating from the Mississippi and Atchafalaya river systems, which can extend to the outer continental shelf (Kealoha et al., 2020b).

No nutrient samples and only a single carbonate chemistry sample were collected in 2025 due to equipment and vessel constraints. However, historical data indicate nutrient concentrations are typically below detection limits, consistent with oligotrophic offshore conditions. Turbidity data were partially recovered and indicated greater variability at WFGB, with several short-term pulses during late spring and summer. Because these elevations were brief and not sustained, they likely reflect episodic oceanographic variability or sporadic storm activity rather than persistent reductions in water clarity.

Water quality data remain a critical component of long-term monitoring at FGBNMS, linking reef conditions to broader oceanographic and terrestrial influences. Although quarterly water sampling was not conducted in 2025, most surface and reef-cap temperature data were captured through moored instrumentation and the Sofar Ocean® buoy systems. The durability and extended deployment capacity of SBE 16plus and HOBO instruments minimized data gaps; however, a prolonged outage occurred at EFGB due to battery depletion. The loss of key infrastructure, including repeated mooring failures of the EFGB Sofar buoy, highlights the vulnerability of offshore monitoring systems and the importance of maintaining redundancy and rapid redeployment capacity.

Chapter 7:
Conclusions

A moray eel peers from its refuge within the reef, where crevices provide shelter and access to passing prey. Photo: Tiffany Crumbley/Crumbley Photography

Long-term monitoring at EFGB and WFGB continues to demonstrate that these reefs remain among the most stable coral reef systems in the wider Caribbean region. Coral cover has remained high over time, with increasing trends at WFGB and stable conditions at EFGB, reinforcing the importance of these sites as long-term ecological benchmarks.

The balanced 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. Fish community trophic structure has exhibited interannual variability in recent years, shifting from historical piscivore dominance to herbivore-dominated biomass in 2023–2024 and to planktivore dominance at West Flower Garden Bank in 2025. 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 Flower Garden Banks. However, continued monitoring is warranted given their potential for rapid population increases.

Thermal stress in corals in 2025 further underscore the complexity of coral response at the banks. Although EFGB exceeded established bleaching thresholds, bleaching prevalence documented during annual monitoring remained low, and many colonies exhibited evidence of recovery following consecutive bleaching years in 2023 and 2024. Because peak bleaching conditions were not surveyed in 2025, the full extent of bleaching response is unknown; however, many colonies exhibited evidence of recovery following consecutive bleaching years in 2023 and 2024. At the same time, coral health in 2025 reflected a shift away from acute thermal impacts and toward the influence of more localized and chronic stressors. Disease persisted, particularly at deeper photostations, and other benthic changes indicate that cumulative pressures continue to shape reef condition. These findings highlight the importance of considering multiple stressors when evaluating reef health, even in systems that appear stable based on coral cover alone. 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 it 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
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
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
WFGB	West Flower Garden Bank

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