Fish and Benthic Communities of the Flower Garden Banks National Marine Sanctuary: Science to Support Sanctuary Management

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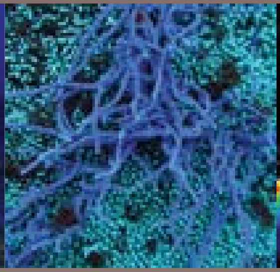

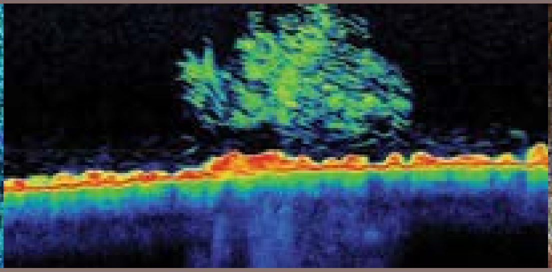

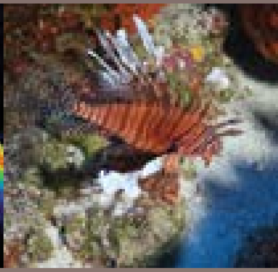

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Science to Support Sanctuary Management

Prepared for the Flower Garden Banks National Marine Sanctuary by:
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Executive Summary

Photo courtesy of M. Winfield (UNCW)

Executive Summary

This report is the culmination of three years of fish and seafloor (benthic) invertebrate community observations on the East and West Flower Garden Banks. It provides baseline information on key biological communities, and can be utilized to address resource management priorities in Flower Garden Banks National Marine Sanctuary (FGBNMS). The benthic and fish community surveys were designed and implemented by a team of multi-disciplinary scientists using three complementary techniques:

1) diver surveys, using recreational and technical scuba techniques, quantified benthic and fish communities on the coral reef at depths between 18-45 m / 59-150 ft;
2) remotely operated vehicles (ROV) conducted video surveys at depths greater than 46 m / 150 ft; and
3) fishery acoustics (sonar) surveyed fish in the water column across all habitat types and depths.

FGBNMS is one of the least impacted, thriving coral reef ecosystems in the western Atlantic and Caribbean region. It does, however, face numerous pressures that should be recognized and responded to through informed management actions. In April of 2012, NOAA published an updated management plan for the sanctuary, representing over five years of data analysis and public participation to ensure a sound strategy for conserving and protecting sanctuary resources for the future. During the management plan review process, input on potential resource protection and management issues was collected and summarized. This process identified direct and indirect impacts of fishing activities as a priority issue for management attention. Hook and line fishing (both commercial and recreational) has always been allowed within the sanctuary. However, to better understand this and other management issues, enhanced biogeographic data are needed to determine the most appropriate management actions needed to fulfill the sanctuary goals and objectives. The sanctuary Management Plan proposes a research strategy that includes characterizing FGBNMS to obtain comprehensive baseline information on fish and benthic communities prior to any management action. A second component of the strategy includes utilizing a fully-protected research area to compare to areas where fishing and other activities occur. The process of designing the research area will build upon prior successful efforts within other sanctuaries, such as Tortugas Ecological Reserve in Florida Keys National Marine Sanctuary and Gray's Reef National Marine Sanctuary, as well as the information presented in this report.

KEY FINDINGS FROM THE REPORT:

Chapter 2: Benthic Habitat Maps

- Refined benthic habitat maps of East and West Flower Garden Banks have been produced that provide more accurate estimates of benthic habitats throughout the sanctuary.
- Benthic habitat maps have been integrated into the Coastal and Marine Ecological Classification Standard and now contribute to a national framework for classifying the environment into biogeographic and aquatic settings.

Chapter 3: Benthic Communities of the Coral Reef and Upper Mesophotic Reefs

- Benthic communities are dominated by coral, with 52% mean cover per site. Three species of *Orbicella* (formerly *Montastraea*) coral, Endangered Species Act candidate species, account for a third of the total coral cover. Benthic communities are stable and healthy (observations of bleaching and disease were rare), and possibly the least impacted coral reef ecosystem in the tropical western Atlantic.
- This study included a comprehensive assessment of benthic communities on the upper mesophotic coral reef from 33-45 m. Coral species composition changed slightly with depth. Some species, such as *Montastraea cavernosa* and *Stephanocoenia intersepta,* were more abundant in deeper waters.

Executive Summary

- Benthic microalgae of the genus *Gambierdiscus*, the causative organism for ciguatera fish poisoning (CFP), were found in all strata to depths of 45 m. Six species were reported in all, three of which are known to be toxic.
- Expanding existing shallow reef monitoring activities to include the deeper reef to 50 m is recommended to fully understand the linkages between anthropogenic and natural events and natural resources.

Chapter 4: Fish Communities on the Coral Reef and Upper Mesophotic Reefs

- This is the first comprehensive assessment of fish communities on the upper mesophotic reef from 33-45 m using *in-situ* diver surveys. The families Pomacentridae (damselfish), Labridae (wrasses) and Serranidae (groupers) comprised 81% of total fish density and were recorded at all depths and habitat types with similar communities found at East and West Banks.
- The most important factors structuring fish communities on the coral reef are depth and habitat relief. Economically valuable species and apex predators such as groupers, snappers and sharks were larger and more abundant at depths >33 m.
- Apex predator biomass in the upper mesophotic strata was dominated by groupers, of which, many species are known to exhibit high site fidelity. Apex predators are important in terms of trophic flow in coral reef ecosystems; therefore, the conservation of these species and their habitats should be a management focus.
- Non-native lionfish densities were low but increased during the study period. The combination of large apex predators and lionfish provides an opportunity to examine natural predation as a biological control.

Chapter 5: Benthic and Fish Communities in the Mid to Lower Mesophotic Zone

- This research expands on prior deep-water characterizations of the sanctuary by providing a comprehensive survey design using a remotely operated vehicle (ROV) to quantify fish and benthic communities in deep (>46 m) sanctuary habitats.
- Deep coral species, such as *Stichopathes* and *Nicella,* were common in the sanctuary, with densities ranging from 0.4 to 1.2 individuals per m^2 (3.3 ft^2) on deep reef habitats.
- Deep reefs with relief ranging from 20-100+ cm yielded high fish density, biomass and species richness. *Mycteroperca phenax* (scamp), *Lutjanus campechanus* (red snapper), and *Seriola dumerili* (greater amberjack) were the dominant apex predators on deep water reefs.

Chapter 6: Mapping Fish Density Using Fishery Acoustics

- Large fish (>29 cm) density was significantly greater on East Bank in deep water (>46 m) habitats; in contrast, large fish were more abundant on the coral reefs at depths less than 46 m on West Bank.
- On the coral reefs (18-45 m), hotspots of large fish densities were consistently observed in similar locations on each bank.
- Large fish density on the West Bank coral reef was 3-10 times greater than other coral reef ecosystems (St. John, U.S. Virgin Islands; Tortugas Ecological Reserve; Vieques, Puerto Rico). Of these systems, West Bank is situated the farthest from any port, suggesting that its remote location plays a role in its condition.

Chapter 7: Conclusions/Recommendations

- Structured by depth and habitat complexity, four distinct benthic and fish communities were found in the sanctuary: coral reefs, algal nodules, coralline algal/deep reefs and softbottom.

Executive Summary

- Groupers and snappers are diverse and abundant on both banks. Species composition, biomass, and density changed with depth and habitat complexity.
- The FGBNMS Sanctuary Advisory Council should develop an approach to evaluate the required size and location of a research area. NCCOS is available to provide continuity as additional research is considered.
- These data represent the first holistic quantification of fish and benthic communities in the sanctuary. It is recommended that existing annual monitoring be expanded to incorporate techniques used in this project. Biannual spatial monitoring of the coral reefs (to 30.5 m) is being implemented by the National Coral Reef Monitoring Plan (NCRMP) to complement existing long-term monitoring.
- In order to address impacts from fishing, lionfish, and recreational diving, it is recommended that fishing and diving activities be quantified in the sanctuary.

Chapter 1
Introduction

Randy Clark, *NOAA NOS/NCCOS/CCMA*

Photo courtesy of M. Winfield (UNCW)

1.1. GOALS OF THE PROJECT

*Figure 1.1. Atlantic creolefish (*Paranthias furcifer*), sergeant majors (*Abudefduf saxatilis*) and blackbar soldierfish (*Myripristis jacobi*) in Flower Garden Banks National Marine Sanctuary (FGBNMS). Photo: NOAA NOS/ONMS/FGBNMS*

The Flower Garden Banks National Marine Sanctuary (FGBNMS) was designated in 1992 to provide protection for a unique coral reef and hardbottom ecosystem (Figure 1.1). A sanctuary management plan is a site-specific planning and management tool that describes the sanctuary's goals, objectives, regulations and boundaries; guides future activities; and sets priorities and performance measures for resource protection, research and education programs. Management plans are revised as necessary to ensure that the sanctuary continues to best conserve, protect, and enhance our nationally significant living and cultural resources. As part of the periodic sanctuary management plan review process, public scoping meetings were held in Texas and Louisiana in 2009, and six priority management issues were identified. These included visitor use, education and outreach, enforcement, fishing impacts, regional non-managed habitat protection, and pollutant discharge (USDOC/NOAA/ONMS, 2012). The Sanctuary Advisory Council recommended the implementation of a research area, prior to which a baseline assessment would be required. The Council also recommended that the research area be in place for no less than eight years and routinely monitored. The research area would dissolve after eight years unless additional action was taken to keep the area intact. The reserve would be no-take, no-use with the exception of a scientific permit. Though there have been several long-term monitoring efforts and biogeographic characterizations of the Flower Garden Banks, the data lacked the spatial and depth coverage necessary to designate and assess the performance of a proposed research area.

To address this critical data gap, NOAA's Coral Reef Conservation Program (CRCP) proposed a collaborative project that would collect fish and benthic community information throughout the East and West Flower Garden Banks to serve as baseline information to address current and future management decisions. The National Centers for Coastal Ocean Science (NCCOS), FGBNMS, National Marine Fisheries Service (NMFS), the Cooperative Institute for Ocean Exploration, Research and Technology (CIOERT), [including Florida Atlantic University and the University of North Carolina-Wilmington], Texas A&M University at Galveston (TAMUG), and Oregon State University (OSU) developed a collaborative three-year project to provide baseline data to guide the potential management action and refine methodologies that could be used to assess the efficacy of an experimental reserve. This wide-ranging collaboration allowed the project to weave together multiple observation techniques - including diver-based, remotely operated, and acoustic surveys - into an integrated characterization of sanctuary resources across depths.

Specifically, the project identified four goals:

1. Update the Sanctuary benthic habitat map;
2. Establish fish and benthic community baseline data on the coral caps, at shallow depths approximately 18-45 m using SCUBA techniques;

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3. Establish fish and benthic community baseline data for the deeper habitats, at depths approximately 46-152 m using a remotely operated vehicle (ROV); and
4. Survey and assess nocturnal fish densities over all habitat types throughout the sanctuary using split-beam echo-sounding systems.

Goal 1 was addressed by ground-truthing and interpreting previous bathymetry and backscatter data collected by FGBNMS. Goals 2-4 build directly on a recent biogeographic characterization of the shallow (18-33.5 m) banks by NCCOS and FGBNMS staff (Caldow et al., 2009); the methods and survey design developed during that characterization were improved or expanded to conduct this project. The project plan was to collect data for goals 2-4 during all three field seasons, but the Deepwater Horizon oil spill occurred during the first year of the project, reducing the overall amount of data that could be collected.

1.2. SANCTUARY DESCRIPTION

The sanctuary's geologic, geographic and resource characteristics are documented in the Final Management Plan (USDOC/NOAA/ONMS, 2012). East Flower Garden Bank (EB), West Flower Garden Bank (WB) and Stetson Bank are only three among dozens of reefs and banks scattered along the edge of the continental shelf of the northwestern Gulf of Mexico (Figure 1.2). EB and WB are located approximately 180 km south of Galveston, TX. Stetson Bank is located 48 km northwest of WB and has significantly different geologic structure, benthic and fish communities compared to the shelf edge banks. For the purpose of this report, we targeted the shallow scleractinian coral communities and associated deep water communities of EB and WB and further mention of "the sanctuary" in this report strictly refers to only those banks.

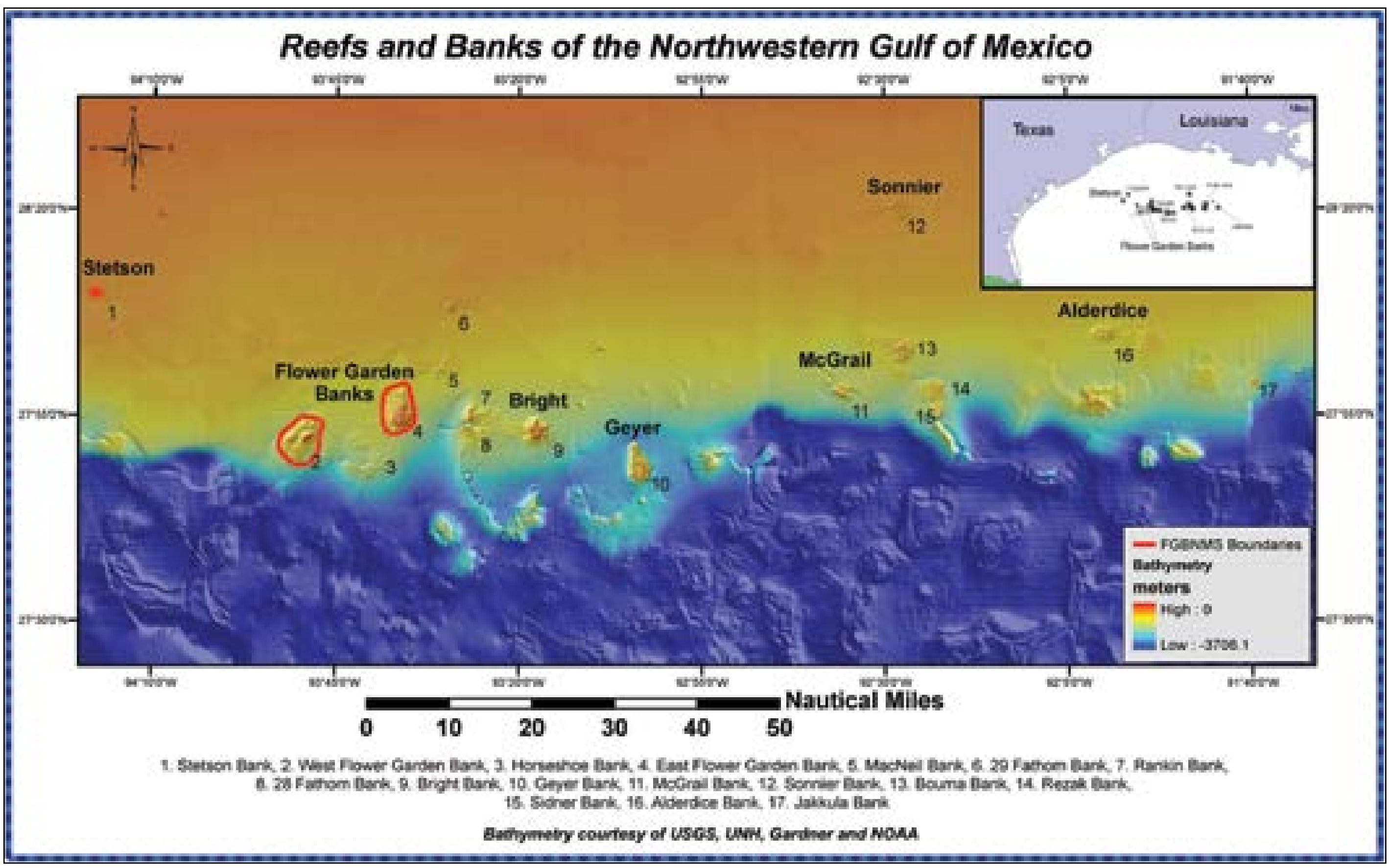

Figure 1.2. Reefs and Banks of the Northwestern Gulf of Mexico. Map Source: NOAA NOS/ONMS/FGBNMS

The banks are part of a regional ecosystem heavily influenced by circulation patterns within the Gulf. The predominant currents in the Gulf of Mexico include the Yucatan Current, which streams warm Caribbean water into the Gulf, and the Gulf Loop Current, which spawns off the Yucatan in a clockwise direction that flows east and south along Florida's Gulf coast and exits through the Straits of Florida. The location of the Loop Current is variable and spin-off eddies can have significant impacts to the sanctuary by importing animal larvae from the Caribbean.

The sanctuary is home to a wide array of marine life, including numerous species of rays and sharks, and sea turtles. Over 170 species of fish and approximately 300 species of reef invertebrates inhabit the banks. These include at least 27 species of sponges, 20 species of polychaetes, 62 species of molluscs and 36 species of echinoderms (Bright and Rezak, 1976). The salt domes underlying these banks are responsible for the framework of coral reef development, as well as providing a reservoir for oil and gas resources. Within a four-mile radius of the Flower Garden Banks, there are currently 13 production platforms (BOEM, 2013). The sanctuary contains the northernmost coral reefs in the continental U.S. and is far removed from neighboring systems. Approximately 643 km (400 miles) to the south, the nearest tropical reef system occurs in the Bay of Campeche, off the Yucatan of Mexico, while the nearest US coral reefs are 1,207 km (750 mi) southeast in the Florida Keys.

East Flower Garden Bank is a 44.3 km^2 pear-shaped dome, capped by 1 km^2 of coral reef that rises to within 17 m of the surface. West Flower Garden Bank is an 88 km^2 oblong-shaped dome, which includes 0.4 km^2 of coral reef area starting 18 m below the surface. Since 1988, Sanctuary managers have monitored fish and benthic community structure using two 100 x 100 m study sites (one on each of EB and WB). Benthic invertebrate and fish communities have been stable during this time (Johnston et al., 2013). Brain and star corals dominate the reefs, with a few coral heads exceeding 6 m in diameter. There are at least 21 species of coral, covering over 50% of the bottom to depths of 30 m, exceeding 70% cover in places to at least 40 m (Schmahl et al., 2008; and references therein). Interestingly, the reefs do not contain some species commonly found elsewhere in the Caribbean, such as many of the branching corals, sea whips or sea fans. A recent observation of note is the discovery of two live *Acropora palmata* colonies, one each at EB and WB (Zimmer et al., 2006). These colonies are some of the deepest records for this species. Despite the extensive loss of coral cover seen in other coral ecosystems in this hemisphere, live coral cover in the sanctuary has not changed significantly in the last 30 years (Rezak, 1977; Gittings et al., 1992; Gittings, 1998; Precht et al., 2006).

The sanctuary is comprised of several different biological zones, with distinct communities primarily segregated by depth (Table 1.1). For this report we attempt to use terminology that is familiar to photic and mesophotic coral reef literature. The coral reef in the sanctuary is not like the reefs of the Caribbean. There are

Table 1.1. Depth and significant characteristics of the benthic habitat classification scheme for the sanctuary.

Biological Zone	Classes	Typical depths	Characteristic	Major Habitat
Coral Reef	Shallow	18-33 m	Large boulder scleractinians, sand patches	High relief stony coral
	Upper Mesophotic	33-52 m	Plating scleractinians, knolls with macroalgae, reef/sand interface	Low relief stony coral
Coralline Algal	Mid Mesophotic	50-82 m (EB) 50-88 m (WB)	Extensive algal plains	Algal nodules
	Lower Mesophotic	46 - 82 m (EB) 46 - 88 m (WB)	Complex honeycombs to eroded reefs	Coralline algal reef
Soft Bottom		70-150 m	Predominantly mud/sand, some soft corals	Sparse cover/mud
Deep Coral		85-150 m	No hermatypic corals, few coralline algal species, fine sediment	Rock outcrops with sponge and octocorals

no surrounding seagrass meadows or reef crests that may support mangrove communities. The coral reef (Figure 1.3) is divided into a shallow portion (18-33 m) dominated by *Orbicella* spp., *Pseudodiploria* spp., *Montastraea cavernosa* and *Porites* spp.; and a deeper community (to 52 m), hereafter termed upper mesophotic coral reef, that has similar coral composition to the shallow portion, but morphology transitions from boulder/mounding to plating. Low relief sloping portions of the banks start at approximately 30 m and are areas dominated by live and rubble remnants of *Madracis auretenra* (formerly *M. mirabilis*; Locke et al., 2007).

Figure 1.3. Coral assemblage on the shallow coral reef of East Bank. Photo: NOAA NOS/NCCOS/CCMA

The reef extends to 52 m, at which point different habitats emerge with increasing depth to maximum depths of 152 m. Habitats occurring at depths greater than 40 m account for 98% of the sanctuary area. Among the deep habitats are softbottom habitats, coralline algal zones (including reefs, and algal nodules or rhodoliths), and deep reefs. These habitats are vastly different and support an amazingly distinct assemblage of sea life.

The coralline algal zone (Figure 1.4) ranges in depth from 46-88 m and is dominated by crustose coralline algae, forming individual algal nodules or rhodoliths, or forming large plates and ridges that develop into massive reef structures. A variety of sponge species are abundant in this zone, along with numerous antipatharians and octocorals. Few reef-building corals occur at these depths, and are mostly limited to small isolated colonies.

Figure 1.4. Coralline algae communities in the sanctuary and algal nodules (left) and coralline algal reefs (right). Photos: NOAA NOS/ONMS/FGBNMS and UNCW

The deep coral zone (>85 m) occurs below water depths that support active photosynthesis. The deep coral zone (Figure 1.5) is characterized by a diverse assemblage of antipatharian and gorgonian corals, crinoids, bryozoans, sponges, azooxanthellate branching corals, and small, solitary hard corals. Rock surfaces are often highly eroded and lack coralline algal growth. Reef outcrops may be covered with a thin layer of silt. Fish communities are not as diverse as those observed on the shallow reef, but do contain many species of commercial and recreationally important species.

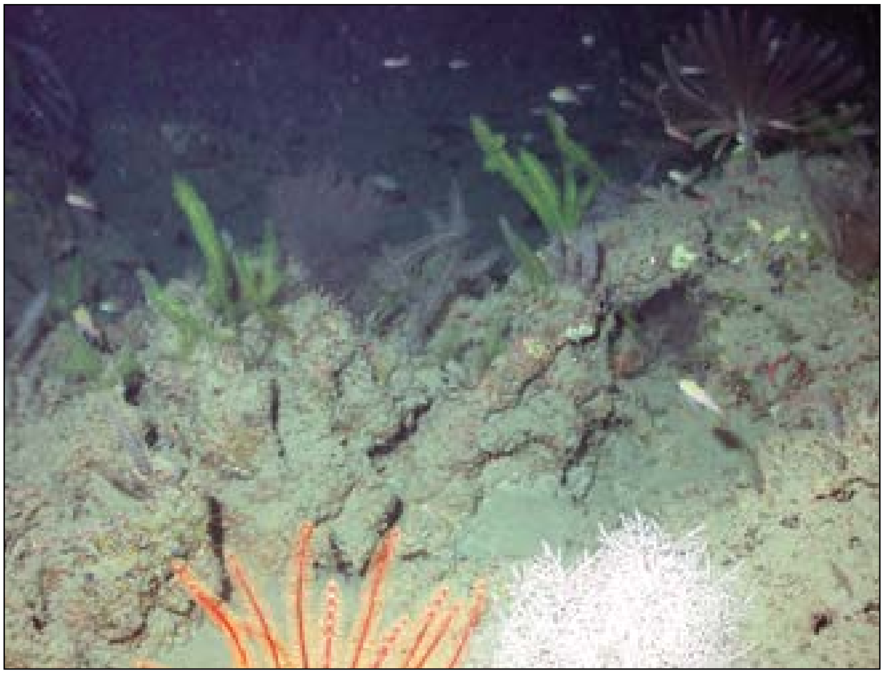

Figure 1.5. Fish and invertebrate communities on a deep reef habitat. Photos: NOAA NOS/ONMS/FGBNMS and UNCW.

The deepest areas of the sanctuary (70-150 m) are characterized by soft unconsolidated sediments composed of both terrigenous sediments originating from coastal rivers and carbonate sediments resulting from erosion of rocky outcrops and coral reef communities. Few conspicuous fishes and invertebrates occur on soft bottom communities when compared to coral reef or rocky zones. Soft bottom communities are often characterized by sand waves, burrows and mounds. Transitional zones between soft bottom communities and hard bottom features are characterized by exposed rubble, isolated patch reefs or exposed hardbottom. Areas with buried or exposed rubble are often colonized by antipatharians, octocorals or solitary hard corals (Figure 1.6; Schmahl et al., 2008).

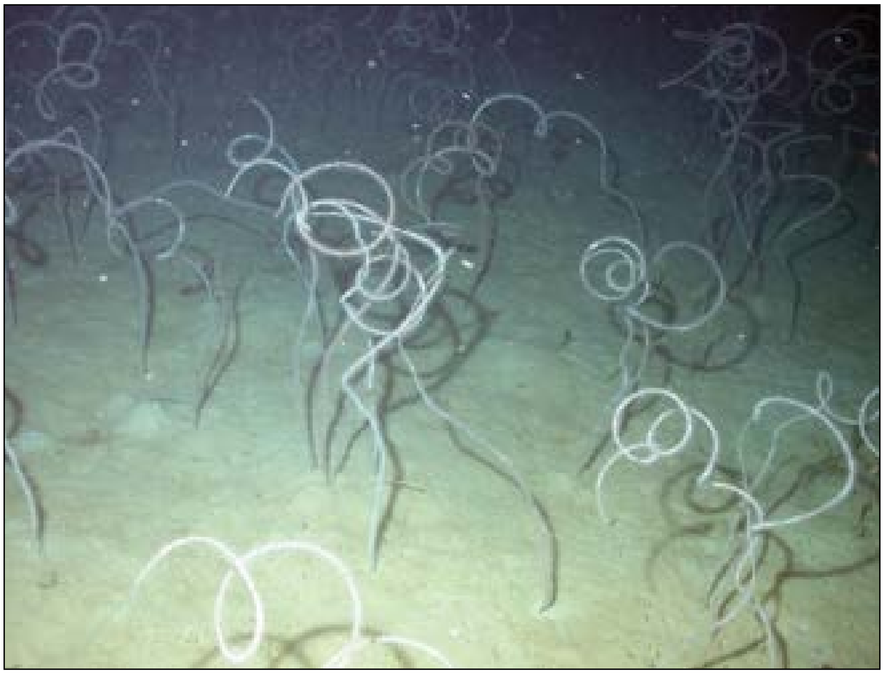

Figure 1.6. Soft corals of the genus Stichopathes *on a soft bottom habitat. Photos: NOAA NOS/ONMS/FGBNMS and UNCW*

1.3. REPORT ORGANIZATION

Subsequent chapters of this report provide information regarding the mapping update and the various sampling and analytical methods used to establish baseline data in the sanctuary. We also provide guidance and recommendations related to monitoring frequency and specific fish and benthic metrics to monitor in the future.

Chapter 2 examines the methods and accuracy assessment of developing an updated benthic habitat map for the sanctuary.

Chapter 3 examines the benthic communities of the shallow and upper mesophotic coral reefs. These habitats were surveyed with SCUBA techniques.

Chapter 4 quantifies fish communities of the shallow and upper mesophotic coral reefs. These habitats were surveyed with SCUBA techniques.

Chapter 5 quantifies fish and benthic communities on all habitats within mid and lower mesophotic depths. These habitats were surveyed with ROVs.

Chapter 6 provides an examination of fish density estimated with split-beam acoustics throughout the entire sanctuary.

Chapter 7 is dedicated to the overall conclusions of the report including recommendations for future research and monitoring.

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Chapter 2
Benthic Habitat Maps

Randy Clark, *NOAA NOS/NCCOS/CCMA*

Photo courtesy of M. Winfield (UNCW)

2.1. INTRODUCTION

The boundaries of the Flower Garden Banks National Marine Sanctuary (FGBNMS) contain three of the dozens of reefs and banks scattered along the edge of the continental shelf in the northern Gulf of Mexico (See Figure 1.2; Figure 2.1.). The banks were first described in the 1970s and 80s by scientists from Texas A&M University and the Department of Interior's Bureau of Land Management (now known as Bureau of Ocean Energy Management [BOEM]; Rezak et al., 1983). These initial characterizations provided the first benthic classification of the northern Gulf of Mexico continental shelf.

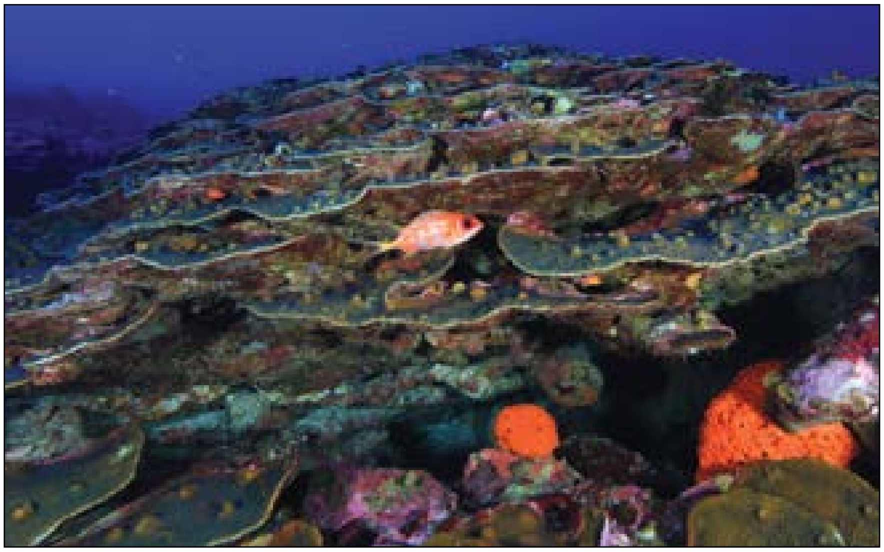


Figure 2.1. Photo of longspine squirrelfish (Holocanthus rufus) *in Flower Garden Banks National Marine Sanctuary (FGBNMS). Photo: M. Winfield (UNCW)*

In 1997, the U.S. Geological Survey (USGS) conducted the first high quality multibeam bathymetry and backscatter surveys in the northern Gulf and focused on the Flower Garden and Stetson Banks (Gardner et al., 1998). During the course of this study, this imagery was re-processed to assist the development of accurate maps of benthic features throughout most of the banks (35-150 m depths at a resolution of 5 m), as well as the East Bank (EB) and West Bank (WB) coral reefs (18-50 m depths at a resolution of 0.5 m). Additionally, the map was further revised as a result of groundtruthing surveys utilizing a remotely operated vehicle (ROV).

Mapping of benthic habitats is an integral component of effective ecosystem-based management (Cogan et al., 2009). A key requirement of the three year baseline study was an accurate benthic habitat map that could be used to assess future sanctuary management decisions. In addition, understanding the spatial distribution of habitats influenced the success of the ROV surveys, as deeper habitats are more likely to be misclassified. Knowing the spatial distribution and amount of habitat area is also crucial in quantifying the fish and benthic communities and the association with habitat type.

2.2. GENERAL MAPPING APPROACH

The approach used to refine the sanctuary habitat maps (East and West Banks only) was to: process existing benthic imagery (Gardner et al., 1998), develop derivative products, delineate habitat boundaries, conduct map accuracy assessment, conduct internal and external review, and produce final maps. We also gathered new backscatter imagery during 2011 at specific areas within the sanctuary that contained problematic imagery from 1997 (e.g., striping or motion artifacts).

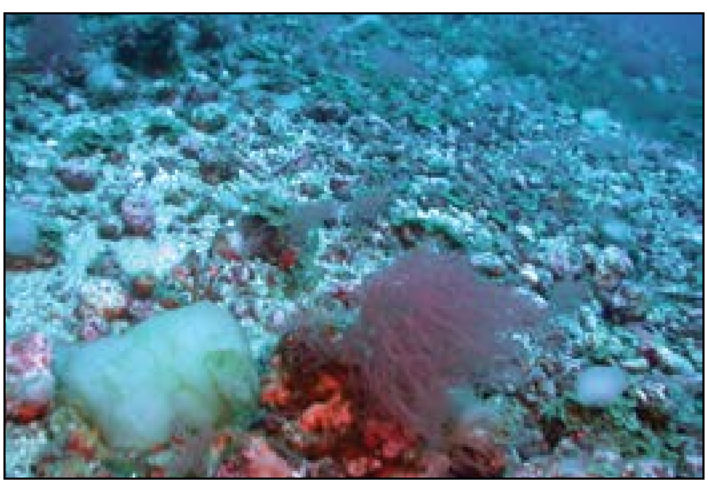
2.2.1. Benthic Habitat Scheme

The Flower Garden Banks habitat classification scheme (Schmahl et al., 2008) defines benthic communities on the basis of two ecosystem attributes: 1) biological zone and 2) major habitat (See Table 1.1). The high relief Coral Reef Zone includes the rugose boulder or reef building coral species, while the low relief 'Coral Reef Zone' typically refers to the deeper (generally between 30-52 m), less rugose and non-reef building species. A spatial assessment of fish and benthic communities was conducted on the shallow coral reef in 2006-2007 (Caldow et al., 2009). These two zones were a component of the sampling strata and were classified based on their benthic relief, e.g. high or low (Figure 2.2; see Appendix C in Caldow et al., 2009).

Figure 2.2. Photos of Coral Reef Zone, high relief coral reef habitat (left); and Coral Reef Zone, low relief coral reef habitat (right). Photos: NOAA NOS/ONMS/FGBNMS and NOAA NOS/NCCOS/CCMA/Biogeography Branch

The Coral Reef Zone is the shallowest zone occurring at depths between 18-52 m. This zone is also further partitioned into two depth classifications based on the current mesophotic reef literature (García-Sais et al., 2011; Blyth-Skyrme, 2013). The shallow reef from 18-33 m is referred to as shallow coral reef, while the reef from 33-52 m is considered upper mesophotic coral reef. These habitats are characterized by high cover of boulder and plating scleractinian species, such as *Orbicella* spp., *Montastraea cavernosa, Pseudodiploria strigosa, Porites astreoides*, and *Colpophyllia natans*. Within this zone are two distinct benthic communities and respective fish communities. The high relief coral habitat has higher species richness and cover that tends to decrease with depth. Coral form also changes with depth. The large boulder forms of *Orbicella* and *Montastraea* seen on the shallower reef tend to be more plate-like at depths beyond 30 m. The low relief coral habitat is typically found at 33-52 m. The habitat is basically large knolls (Bright et al., 1985) that extend into the depths and transition to algal nodule habitats. The knolls are colonized by the small branching coral *Madracis auretenra* (previously known as *Madracis mirabilis*) or covered by lush macroalgae, typically members of the genera *Stypopodium*, *Caulerpa*, *Dictyota* and *Lobophora*.

The Coralline Algal Zone is a large zone composed of two dominant habitat types: algal nodules and coralline algae reef (Figure 2.3). Algal nodules form extensive plains in all directions extending from the coral reef. It is a unique habitat and supports a unique infaunal and epifaunal community (Bright et al., 1985). Algal nodule dominance gives way to reefs comprised of crustose coralline algae (CCA). These reefs occur at depths below



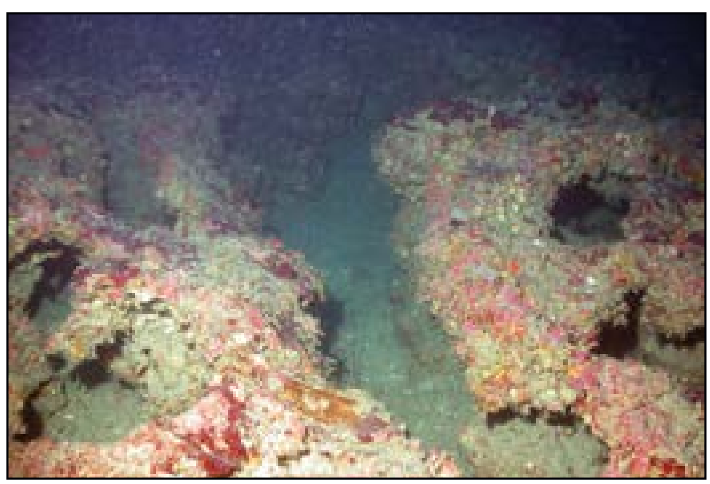


Figure 2.3. Photos of Coralline Algal Zone, Algal Nodule habitat (left); and Coralline Algal Zone, coralline algal reef habitat (right). Photos: NOAA NOS/ONMS/FGBNMS and UNCW

which most hermatypic corals grow but instead favor CCA. This zone typically occurs between 46-82 m on EB and 46-88 m at WB (Bright et al., 1985). In some places, these reefs are highly complex, at times exceeding 1 m in height, with many honeycomb-like holes and crevasses. In other areas, these reefs are highly eroded and may only extend 10-20 cm in height.

The Deep Coral Zone typically occurs below 85 m (Bright et al., 1985) and does not support hermatypic corals, and few CCA species exist (Figure 2.4). In this zone, the water is generally turbid and dark and the reef is covered with fine sediment.

Soft bottom habitats (Figure 2.4) occur predominantly at depths greater than 80 m, but may be seen as shallow as 70 m. Soft bottom areas were often colonized by black or soft coral communities, such as antipatharians, and also contain mud volcanoes and brine seeps. The imagery available was not adequate to delineate these soft bottom communities.

Figure 2.4. Photos of Deep Coral Zone, rock outcrop and sand interface (left); and Softbottom Zone, starfish on soft unconsolidated sediments (right). Photos: NOAA NOS/ONMS/FGBNMS and UNCW

2.2.2. Mapping Process

Coral reef (18-33.5 m) maps developed in 2006-07 (Caldow et al., 2009) were developed primarily to conduct a fish and benthic spatial assessment. These surveys were restricted to depths of less than 33.5 m and did not include the entirety of the coral reefs on either bank. The coral communities on both East and West Banks extend into the mesophotic zone to about 50 m. The approach used by Caldow et al. (2009) was used here to digitize distinct spatial patterns of benthic relief and develop a comprehensive map of the coral reef environment on each bank.

A suite of bathymetric derivatives (slope, slope of slope, terrain ruggedness) and backscatter imagery was used to map the deeper habitats in the sanctuary. Slope and slope of slope are GIS layers developed in ArcMap Spatial Analyst Extension. Benthic Terrain Modeler (a GIS tool developed by NOAA's Coastal Services Center) was used to generate terrain ruggedness, a rugosity-like measure, to assist in the classification of benthic features (Figures 2.5 and 2.6). Terrain ruggedness is defined as the variation in three-dimensional orientation of grid cells, using vector analysis to calculate the dispersion of vectors normal (orthogonal) within a specified neighborhood. This method effectively captures variability in slope and aspect into a single measure. Ruggedness values in the output raster can range from 0 (no terrain variation) to 1 (complete terrain variation). Typical values for natural terrains range between 0 and about 0.4.

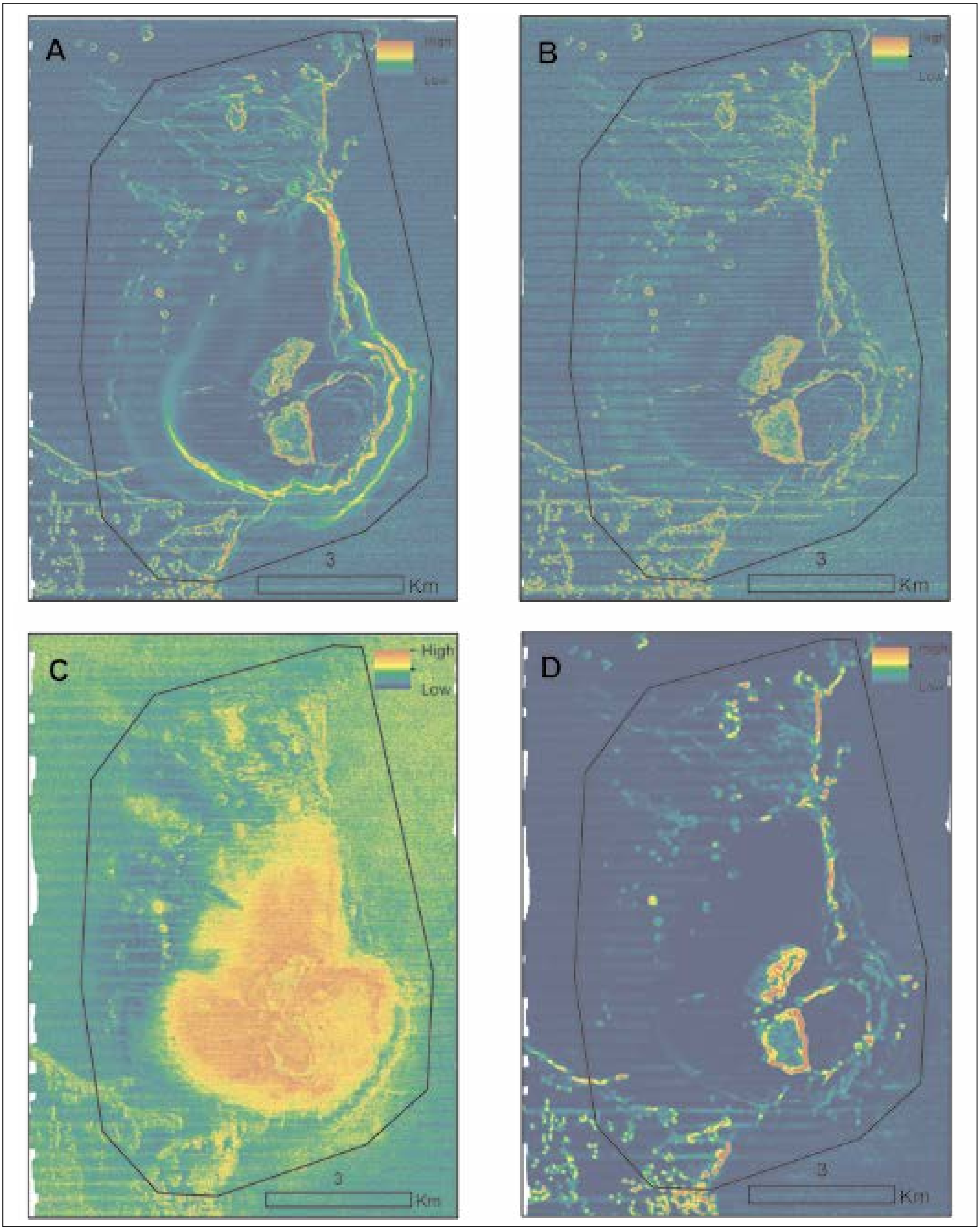


Figure 2.5. East Bank bathymetric derivative data layers used to assist with benthic habitat mapping. A) Slope, B) slope of slope, C) backscatter, and D) terrain ruggedness.

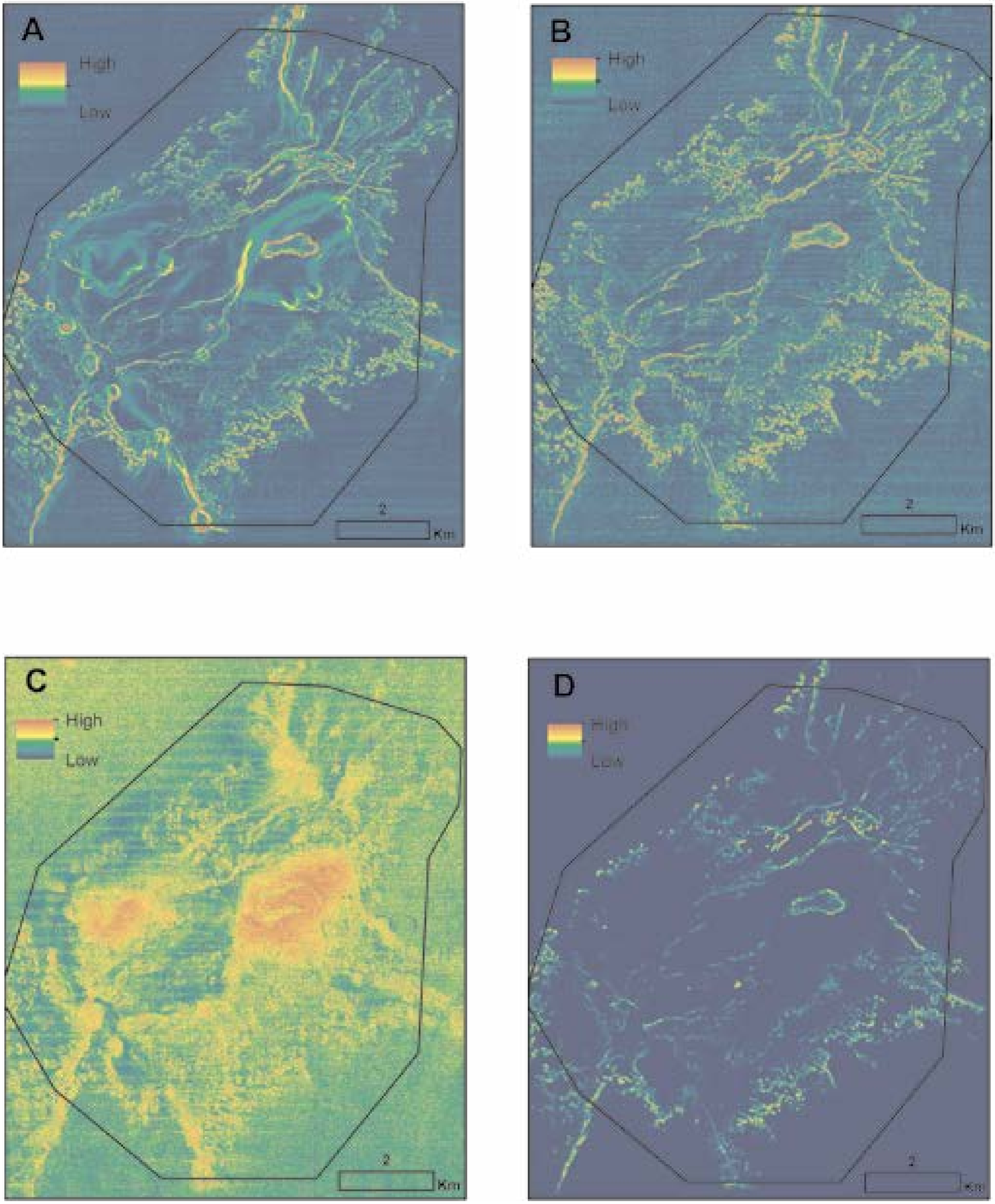


Figure 2.6. West Bank bathymetric derivative data layers used to assist with benthic habitat mapping. A) Slope, B) slope of slope, C) backscatter, and D) terrain ruggedness.

Mapping

The derived data layers were used as a backdrop for digitizing habitat polygons using the Editor tool in ArcMap 10.0. The resulting map delineated contiguous areas of similar benthic types. On the coral reef the minimum mapping unit (MMU) was approximately 2,500 m^2 (50 x 50 m grid) as this was the spatial scale of SCUBA transects on the reefs (See Appendix C and also Caldow et al., 2009). MMU was larger (31,400 m^2) for deeper habitats given the planned 100 m spatial scale of the ROV transects.

2.2.3. Accuracy Assessment

Accuracy of the FGBNMS benthic habitat maps was assessed in several ways. Error matrices were computed for biological zone and major habitat for the East and West Banks combined (Zitello et al., 2009). Accuracy assessment survey size was not sufficient to assess by bank. Overall accuracy and user's accuracy were computed directly from the error matrices (Story and Congalton, 1986). Error matrices (Tables 2.1 and 2.2) were constructed as a square array of values arranged in rows (map classification) and columns (groundtruthed classification). The overall accuracy (P_0) was calculated as the sum of the correct classifications divided by the total number of accuracy assessment samples. User's accuracy was calculated to characterize the classification accuracy of individual map categories by measuring how often map polygons of a certain habitat type were classified correctly.

Table 2.1. Error matrix for biological zone. Numbers indicate correct biological zone classified. CA=Coralline algae, CR=Coral Reef, N=total number of accuracy surveys, UA=User's Accuracy.

	CA	CR	N	UA (%)
Coralline Algal	42	-	45	93.3
Coral Reef	-	33	33	100
				P_0= 92.5%

Table 2.2. Error matrix for major habitat. Numbers indicate correct habitat classified. AN=algal nodule, CA=coralline algae, HCR=high relief coral reef, LCR=low relief coral reef, N=total number of accuracy surveys, UA=User's Accuracy.

	AN	CA	HCR	LCR	N	UA (%)
Algal Nodule	26	-	-	-	28	92.9
Coralline Algal Reef	-	13	-	-	14	92.9
High Coral Reef	-	-	24	-	24	100
Low Coral Reef	-	-	-	7	10	70
Total					***76****	
						P_0= 90%

* One AN and one CA observation was on softbottom.

During the 2011 technical diving mission on the NOAA Ship *Nancy Foster*, we implemented an independent accuracy assessment data collection on each bank (Figure 2.7). A video drop camera was used to document benthic composition at 100 randomly distributed points on each bank (25 per habitat type). Depth limitations of the camera restricted our data collection to 60 m. This limitation excluded most coralline algal habitats and all deep reef and soft bottom habitats. As such, 60 sites were assessed on EB. In addition, strong currents and poor weather conditions limited WB assessment to 18 sites. Therefore, the accuracy assessment was conducted by pooling the sites to examine accuracy by habitat type and not stratifying by bank.

Figure 2.7. NOAA Ship Nancy Foster*. Photo: NOAA NOS/NCCOS/CCMA*

The biological ROV surveys provided an excellent opportunity to obtain additional information at depths greater than 60 m where we could not use the drop camera. We used habitat type data collected from 291 *in situ* scuba surveys (Figure 2.8; See Chapters 3 and 4) and 212 ROV surveys (Chapter 5) to help inform the development of the benthic habitat map.

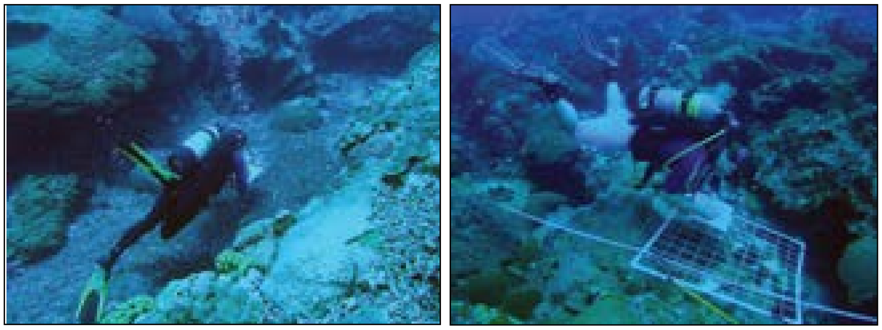


Figure 2.8. Divers conducting fish census (left) and benthic habitat composition census (right) in the shallow coral cap communities of FGBNMS. Photos: NOAA NOS/NCCOS/CCMA and J. Voss (NOAA NMFS/SEFSC)

2.2.4. Additional Classification

The Coastal and Marine Ecological Classification Standard (CMECS) provides a national framework for organizing information about coasts and oceans and their living systems (FGDC, 2012). The six components of the classification standard represent the different aspects of the seascape (water column, geoform, substrate, biotic communities, biogeographic setting, and aquatic setting), starting with the broadest systems (marine, estuarine, and lacustrine) and narrowing to the most detailed physical and biological elements associated with a specific habitat type (biotic community).

CMECS provides a structure for developing and synthesizing data so that ecosystems can be identified, characterized, and mapped in a standard way across regional and national boundaries. CMECS also supports status and trend monitoring activities, policy development, restoration planning, and fisheries management. The standard complements existing wetland and upland classification systems. As such, we related the sanctuary classification to CMECS where appropriate and included the information as attributes in the GIS products (Table 2.3). For more information on CMECS: http://www.csc.noaa.gov/digitalcoast/publications/cmecs

Table 2.3. Translation of sanctuary habitat classification scheme to Coastal and Marine Ecological Classification Standard (CMECS).

Flower Garden Habitat Unit	Biogeographic Setting	Aquatic Setting	Geoform Component	Substrate Component	Biotic Component
Algal Nodule	Northern Gulf of Mexico	Marine Offshore Subtidal	Bank	Algal Substrate	Coralline/ Crustose Algae Colonized Shallow/ Mesophotic Reef
Coral Cap	Northern Gulf of Mexico	Marine Offshore Subtidal	Bank	Coral Substrate	Branching Coral Shallow/ Mesophotic Reef
Coralline Algae	Northern Gulf of Mexico	Marine Offshore Subtidal	Bank	Algal Substrate	Coralline/ Crustose Algae Colonized Shallow/ Mesophotic Reef
Deep Reef	Northern Gulf of Mexico	Marine Offshore Subtidal	Bank	Coral Substrate	Deepwater/Coldwater Stony Coral Reef
Soft Bottom	Northern Gulf of Mexico	Marine Offshore Subtidal	Bank	Unconsolidated Mineral Substrate	NA

2.3. RESULTS AND DISCUSSION

2.3.1. Biological Zone

Limited data were available to assess map accuracy. Therefore it is suggested that additional effort be expended to adequately assess the accuracy of the maps and differences between the two banks. The only habitats that were accessible to our cameras were the coral reef, algal nodules and shallow (approximately 60 m) coralline algal reefs. No accuracy assessment data were gathered for soft bottom or deep reef communities. Error matrices for biological zone (Table 2.1) indicate that the classified maps had strong overall accuracy (92.5%).

2.3.2. Major Habitat Type

At the habitat level, coralline algal communities were divided into two habitats: algal nodule and coralline algae reefs. Both are distinct structurally and errors in accuracy were attributed to not being able to distinguish the transition of these two habitats from the imagery. Errors observed on the coral reefs were few, most of them observed in the deeper (>30 m) areas. While few errors were observed on the coral reefs, areas that have had few *in-situ* observations from SCUBA divers (e.g, depths greater than 33.5 m), yielded the most discrepancies. The most significant errors on the coral reefs were areas on WB classified as low relief. The reef on WB has a dramatic slope at approximately 33.5 m and rapidly descends to approximately 45 m, making it challenging to interpret the imagery correctly. Even with a resolution of 0.5 m, it was difficult to discern exact patterns of low relief from the adjacent high relief areas.

Using the *in-situ* surveys to assess benthic classification (Table 2.4) we observed high accuracy for most habitat types. In general, the shallower habitats yielded higher accuracy and may be a reflection of the original imagery collected. Low relief coral reef and deep reef yielded the lowest accuracy and are an indication of the poor quality of the original backscatter imagery in deep habitats. Deep reefs were commonly misclassified and were actually soft bottom. Some of the error can be attributed to the reliance of bathymetric features (high slope in combination with high rugosity measures) that may have erroneously indicated hardbottom. Low relief coral reef habitats were misclassified as a result of the changing morphology of the major scleractinian corals present. At depths greater than approximately 35 m, colonies were mostly plate-like and appear less complex than those on the shallow portions of the reefs. This appearance led to the assumption that these were the low relief Madracis-dominated reefs.

Table 2.4. Percent positive habitat classification based on in-situ *surveys with SCUBA or remotely operated vehicle (ROV).*

	Sites	Positive classification	%
High relief coral reef	240	234	97.50
Low relief coral reef	51	39	76.47
Algal nodule	38	34	89.47
Coralline algal reef	75	57	76.0
Deep reef	80	55	68.75
Softbottom	8	8	100.00

2.4. CONCLUSIONS

Overall the mapped habitats comprise 133 km^2 on WB and 113 km^2 on EB, of which 77.4 and 65.4 km^2 (respectively) fall within the sanctuary. The mapped areas that fall outside the sanctuary boundary were primarily soft bottom and deep coral habitats.

The update of the benthic habitat map yielded some significant changes (Figure 2.9). Most notably on EB were high relief coral reef and deep reefs increased by nearly 50% (Table 2.5). The area of low relief coral reef and algal nodules shows a slight decrease compared to the previous habitat map. On WB, coralline algae reef habitat increased by 177%, occurring primarily in the southern portion of the bank, with some added in the northeastern portion of the bank (Figure 2.9).

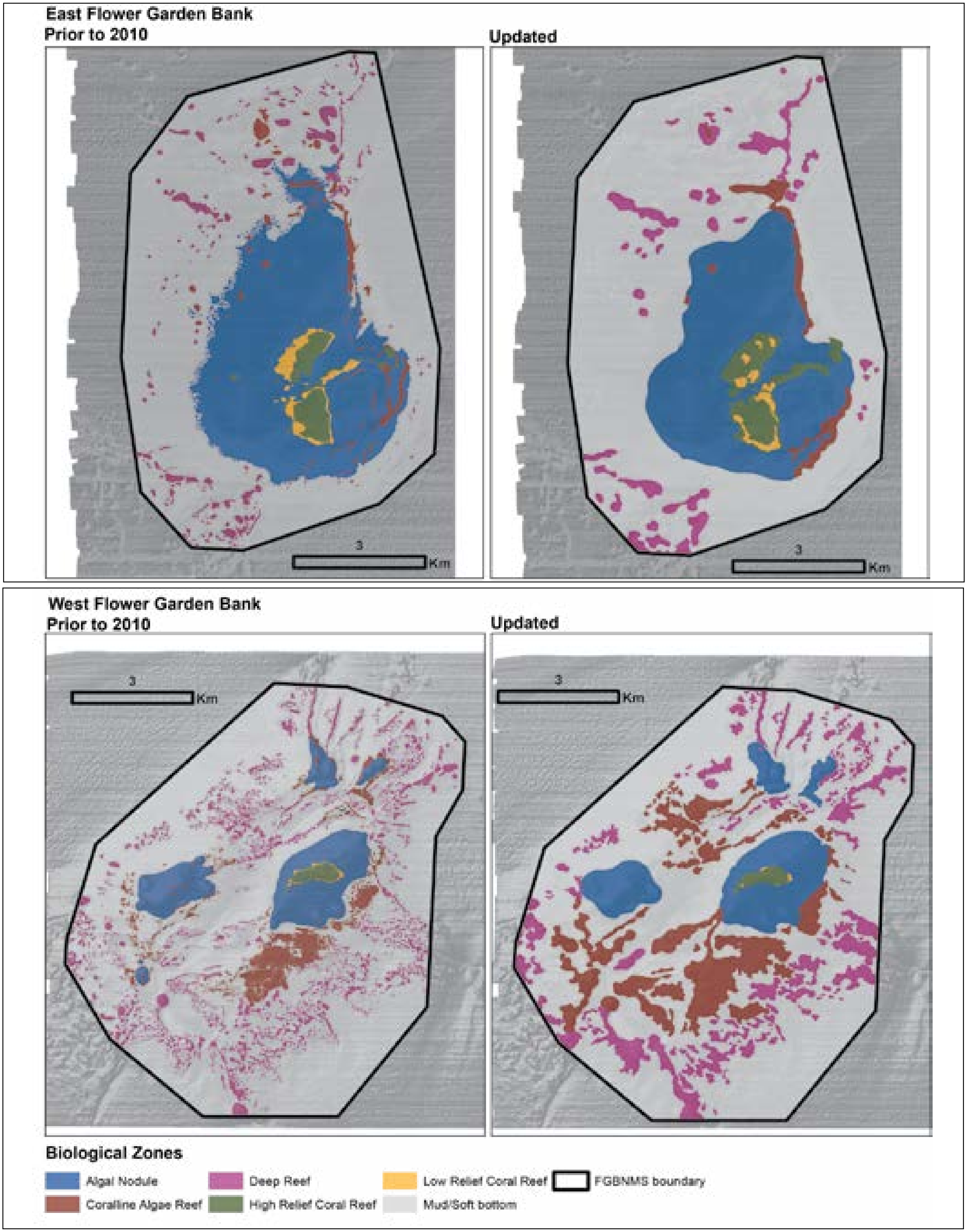


Figure 2.9. Sanctuary benthic habitat map prior to 2010 and after 2010 for East (top) and West (bottom) Banks.

Table 2.5. Area of mapped habitats inside and outside sanctuary boundaries for East and West Banks.

	West Bank	WB Sanctuary		East Bank	EB Sanctuary	
	Total Area (km)	New Area (km)	Prior Area (km)	Total Area (km)	New Area (km)	Prior Area (km)
High relief coral reef	0.45	0.45	0.42	2.15	2.15	1.49
Low relief coral reef	0.03	0.03	0.08	0.48	0.48	0.76
Softbottom	103.13	50.28	59.5	88.44	43.03	41.04
Algal nodule	7.45	7.45	6.38	15.21	15.21	18.69
Coralline algal reef	9.69	9.69	3.49	1.41	1.41	1.39
Deep reef	12.62	9.57	7.6	5.99	3.53	2.44
Total	**133.37**	**77.47**	**77.47**	**113.68**	**65.81**	**65.81**

While East and West Banks have similar benthic features, the proportion of the habitats varies substantially. EB has nearly twice the area of coral reef than WB (Table 2.5) and twice the amount of algal nodule habitat. While WB has less area for the shallow habitats (coral reef and algal nodule), there is considerably more hardbottom habitat in the deeper waters. The area of coralline algae reef habitat is about nine times greater and deep coral habitats are about three times greater on WB. EB and WB are similar in that they are both comprised of approximately 65% soft mud.

These maps formed the foundation of the stratified random sampling design for surveys of fish and benthic communities over the past three years (2010-2012). The maps were integrated into 50 m^2 grids for the coral cap surveys and 200 m^2 grids for the ROV surveys. As such, the map is a vital resource for future monitoring. It is recommended that the map be reassessed whenever possible to further supplement the accuracy assessment performed here. There is ample data on the coral reefs, and we believe they are well classified. However, this is a small portion of the overall sanctuary area. There are many habitats within the sanctuary that need to be sampled further to be properly classified.

The accuracy values obtained at FGBNMS are well within accepted accuracy measures from other areas (although classification schemes are not directly comparable; Battista et al., 2007a and b; Walker and Foster, 2009; Zitello et al., 2009). However, the accuracy assessment sample size is very low. It is recommended that a focused accuracy assessment be conducted to evaluate the classification of the current map. These maps also extend beyond the sanctuary boundary, and may serve as a framework to compare fish and benthic communities inside and outside the sanctuary.

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Chapter 3
Benthic Communities of the Coral Reef

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Photo courtesy of C.A. Buckel (NOAA NOS/NCCOS/CCFHR)

Benthic Communities

3.1. INTRODUCTION

Figure 3.1. An example of the high coral cover and rich fish communities found at Flower Garden Banks National Marine Sanctuary (FGBNMS). Photo: G. McFall (NOAA NOS/ONMS/GRNMS)

The benthic community of the Flower Garden Banks National Marine Sanctuary (FGBNMS) coral reef has a high coral and comparatively low algal cover (Gittings et al., 1992; Continental Shelf Assoc., Inc., 1996; Dokken et al., 2003; Pattengill-Semmens and Gittings, 2003; Aronson et al., 2005; Precht et al., 2006; Zimmer et al., 2010; Johnston et al., 2013). Coral species richness is lower than most Caribbean reefs (31 species; Schmahl et al., 2008); however, the community has maintained high cover (exceeding 70% cover in some areas; Figure 3.1) despite hurricanes, global bleaching and disease epidemics that have severely impacted Caribbean reefs (Gardner et al., 2003). The stable coral cover at FGBNMS has been attributed to relatively deep depths that buffer reefs from storms and high water temperature; additionally, its remote offshore location limits user access and exposure to eutrophic coastal runoff, and its sanctuary designation protects it from deleterious effects associated with the oil and gas industry. (Aronson et al., 2005; Deslarzes and Lugo-Fernández, 2007; Lugo-Fernández and Gravois, 2010).

A consistent benthic community long term monitoring (LTM) program on East and West banks (hereafter EB and WB, respectively) began in 1988 by the Bureau of Ocean Energy Management (BOEM; previously Minerals Management Service) due to concerns about impacts from offshore oil and gas development in the area (Gittings et al., 1992). Since 1993, the annual LTM surveys have been conducted by BOEM and FGBNMS. In two 100 x 100 m LTM study sites on EB and WB (one on each bank) located at depths of 17-26 m (Aronson et al., 2005), no significant long term changes in benthic community cover were detected between the designation of the sanctuary in 1992 and the most recent survey, 2011 (Johnston et al., 2013). The LTM study provides a detailed timeline of the benthic community, but the spatial extent of the study is limited. The first quantitative spatial characterization of FGBNMS coral reef communities (to 33.5 m) was conducted from 2006-2007 (Caldow et al., 2009). Benthic community results were consistent with findings from the LTM study; specifically, high coral and low algal cover. The Caldow et al. (2009) study suggested community changes with depth and recommended standardized benthic surveys be conducted for the whole coral reef, which extends to depths of approximately 52 m (see Table 1.1 for habitat descriptions). Based on these recommendations, the present three year study examines the FGBNMS communities to 150 m depth and provides a baseline for future proposed management activities. This chapter summarizes benthic community data collected using SCUBA from the coral reef (to 45 m) to address the following objectives:

- Establish a benthic community baseline,
- Quantify benthic cover of specific biotic and abiotic groups, including coral species,
- Assess relationships between depth and benthic complexity,
- Assess coral disease and bleaching,
- Assess prevalence of marine debris, and
- Determine the presence or absence of the ciguatera fish poisoning causative organism, *Gambierdiscus* spp., and describe its spatial distribution by depth and habitat type.

Data and metrics derived from this study will provide baseline information to 1) develop a spatial monitoring plan, either as an independent monitoring program or to supplement the two LTM sites, and 2) provide a foundation for assessing impacts associated with natural and/or anthropogenic events in the future.

3.2. METHODS

3.2.1. Field

Sites were selected to a depth of 45 m using a stratified random sampling approach within a pre-defined sample area, as described in Caldow et al. (2009). The sample area consisted of an array of 50 x 50 m uniformly placed grid cells (Figure 3.2). Each grid cell was an exclusive sampling unit with the site located at the centroid of each sampled grid cell (see Menza et al., 2008). Site selection was proportional to the area of each stratum. As a result of the randomized site selection, a small number of sites were surveyed across multiple years (n = 36). Given the nature of the study design, sites could be resampled in subsequent years and still represent exclusive sampling units. Strata included bank (East or West), reef complexity (high or low relief), and depth (shallow or upper mesophotic [UM]; Table 3.1).

Table 3.1. Number of diver surveyed sites by year and strata. Depth strata were defined as shallow (<33.5 m) and upper mesophotic (33.5-45 m).

Bank Relief	Depth	N Sites by Year			N Sites	N sites By Bank	N sites by Depth
		2010	2011	2012			
EAST HIGH	Shallow	39	67	38	114	135	225
EAST LOW	Shallow	6	8	7	21		
WEST HIGH	Shallow	30	28	28	86	90	
WEST LOW	Shallow	0	2	2	4		
EAST HIGH	Upper mesophotic		16	15	31	44	66
EAST LOW	Upper mesophotic		9	4	13		
WEST HIGH	Upper mesophotic		8	7	15	22	
WEST LOW	Upper mesophotic		2	5	7		
Total		**75**	**110**	**106**	**291**	**291**	**291**

The depth component of the sampling strata was defined as shallow (18-33.5 m) and UM (33.5-45 m; Table 1.1). The shallow depth stratum was surveyed during each year of the study (2010-2012). Due to vessel availability limitations resulting from the Deepwater Horizon spill in 2010, the UM depth stratum was only surveyed in 2011 and 2012 (Table 3.1). For the reef complexity component of the sampling strata, complexity was derived from a benthic habitat map of the coral reef where two distinct habitats were identified (see Chapter 2). Low-relief coral habitat mostly included a mixture of live and dead *Madracis auretenra* (formerly known as *Madracis mirabilis*), and high-relief coral habitat was composed of boulder and plating corals, including *Montastraea*, *Orbicella* and *Pseudodiploria* (previously known as *Diploria*; See Caldow et al., 2009 for a more detailed description of the relief strata).

Surveying depths beyond 33.5 m required an alternative logistical approach. Technical (or decompression) diving techniques were used to extend bottom time at depths greater than 33.5 m. Data collection methodology remained consistent for all depths (18-45 m). Surveys of the shallow and UM strata were conducted during separate sampling events but within the same seasons (end of summer/early fall) due to logistical constraints of the two diving techniques. Analysis by depth strata (shallow and UM) was chosen to highlight the additional information gained by employing technical diving techniques, which allowed for the first comprehensive *in situ* surveys of the mesophotic coral zones at FGBNMS.

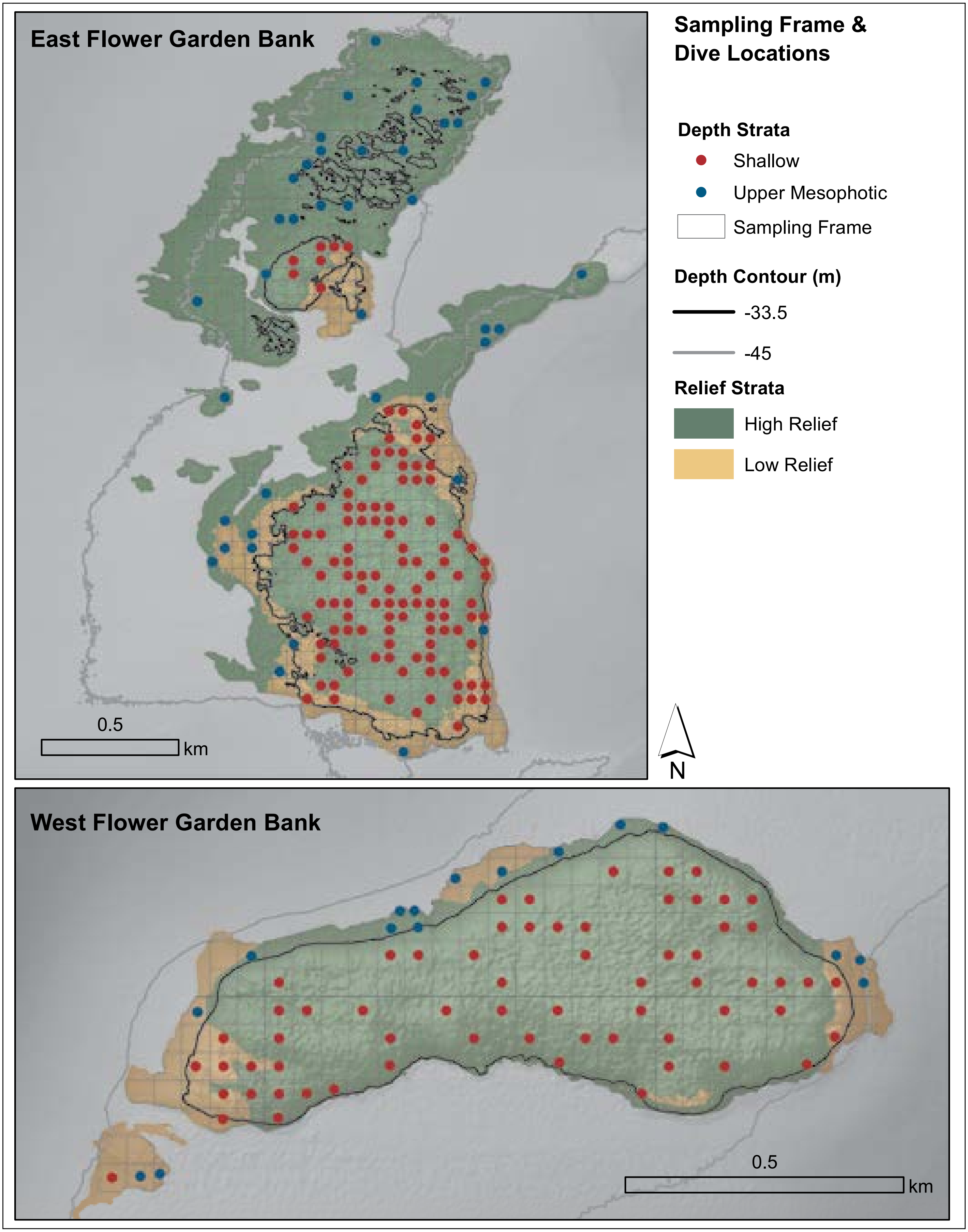

Figure 3.2. Sampling frame for the coral reef on East and West Banks with locations of dive survey locations by relief strata from 2010 – 2012.

Benthic Communities

Figure 3.3. Diver with quadrat collecting benthic data during an upper mesophotic dive survey. Photo: M. Winfield (UNCW)

An overview of benthic surveys is provided here; refer to Appendix A for further methodology details. Benthic composition data were collected in four 1 m^2 quadrats. One quadrat was located within every 6 m interval along the 25 m transect and its exact location along the transect was randomly pre-selected prior to the diver entering the water (Figure 3.3.).

Within each quadrat, both abiotic and biotic percent cover data (planar) were recorded to the nearest 0.1% (Table 3.2). Maximum height (cm) of hard substrate (i.e., scleractinian coral, rock) within the quadrat was also measured. Percent cover values reported here were the mean value of all quadrats sampled at a site. Site means were used as independent replicates in analyses. All corals were identified to the lowest possible taxon. Coral colony condition was also assessed. Where coral bleaching or disease was present on any portion of a colony, the entire colony was considered affected, with cover noted as diseased or bleached. Marine debris encountered along the transect and its colonizing organisms were described, the area of the debris, and the benthos area affected (directly or indirectly) by the debris were estimated (cm^2).

Table 3.2. Variables measured on coral reef benthic surveys, 2010-2012.

Type	Measurements: Percent Cover	Measurements: Abundance (#)
Abiotic		
Hardbottom	X	
Sand	X	
Rubble	X	
Biotic		
Coral (by species)	X	
Algae		
Macroalgae	X	
Turf	X	
Crustose coralline	X	
Sponges		
Barrel, tube, vase morphology	X	
Encrusting morphology	X	
Other macrofauna		
Anemones and hydroids	X	
Tunicates and zooanthids	X	
Macroinvertebrates		
Diadema antillarum (Long-spine Urchin)		X
Panulirus argus (Spiny Lobster)		X
Marine debris	Type, area of debris, area affected, colonizing organisms	

3.2.2. Statistical Analyses

A total of 291 sites were sampled (Figure 3.2). Analyses for both total biotic cover and all algae comparisons were based on 275 sites due to field identification irregularities at 16 sites in the shallow strata. All sites were used in analyses of abiotic, coral, hydrocoral, and sponge cover.

To examine spatial and structural trends, each site was assigned a stratum designation based on bank, relief, and depth (Table 3.1). Comparative analyses were conducted using eight possible strata. Sample size varied between strata based on areal allocation and logistic constraints due to depth. Comparisons of percent cover between banks were completed using a t-test for normal data, using a square root transformation to normalize cover data when necessary, or a Wilcoxon test for non-normal data.

Two measures of relief were used in this report: a binary relief stratum (high and low as described previously), and a continuous ruggedness measure. Benthic terrain ruggedness (described in Chapter 2; Figure 2.4) was used as a continuous measurement of rugosity and is hereafter referred to as rugosity. There was good agreement between the two measures with a significant difference between high and low relief strata ($t = 10.269$, $p < 0.0001$) as quantified by rugosity.

Univariate Analyses

Correlations were examined between actual site depth (m), rugosity, and percent cover of benthic groups and coral species. Pearson's correlation (r) was used for percent cover data that could be normalized using a square root transformation. For non-parametric data, Spearman's ρ (rho) or Kendalls' τ (tau) rank correlations were used to examine the relationships between percent cover and other community metrics. Logistic regressions (generalized linear models with binomial distributions) were used to examine occurrence of individual coral species. Inter-annual differences in species or species group cover were examined for dominant coral species using a Wilcoxon test for UM strata (for two years of data collection) and a Kruskal-Wallis test for the shallow strata (for three years of data collection). Any significant between year differences within the shallow stratum were examined further using a sequential Bonferroni correction to control the group wide Type 1 error rates (Rice, 1989).

Multivariate Analyses

Multivariate analyses examining differences in benthic communities among sites were conducted in PRIMERv6 (Clarke, 1993; Clarke & Warwick, 2001) and R statistical environment (R Development Core Team, 2013) with package "vegan" (Oksanen et al., 2013). Percent cover data for coral species, benthic biotic cover and benthic abiotic cover were 4th root transformed to increase the weight of less dominant species or groups and downweight dominant groups. The data were then converted into resemblance matrices using Bray-Curtis similarity indices for biotic data or Euclidean distance matrices for environmental data. Multi-level patterns were investigated with permutational multivariate analysis of variance (PERMANOVA). To account for the unbalanced sampling design, year (2010-2012) and bank (East and West) were treated as fixed effects, while random effects of depth strata (shallow, UM) nested within year and relief (high and low) nested within bank, were treated as random effects. Effects of depth and rugosity on coral species beta diversity were examined using PERMANOVA (adonis within package vegan). Analysis of similarities (two way ANOSIM) was used to identify broad benthic community differences between year and depth strata, as well as coral community differences between depth strata. The similarity percentages routine (SIMPER) was used to identify species (or species groups) that contributed to the separation of groups, or similarity between groups.

3.3. RESULTS AND DISCUSSION

Distinct differences between depth strata were found when examining broad benthic community characteristics, such as substratum height or percent cover of abiotic and biotic group components (ANOSIM R = 0.327, p = 0.001). Three factors comprised nearly 63% of the difference between the shallow and UM depth strata: maximum height (cm) of hard structure (i.e., scleractinian coral or rock) within the surveyed quadrats, and percent cover of hard coral and algae (Table 3.3). Shallow sites were characterized by higher relief, a greater percent cover of hard coral, and lower percent cover of algae, rubble, and sand, in contrast to UM sites. lower relief within the UM was likely due to transitions from boulder to plating coral morphology with depth. Differences between depth strata are not due to temporal differences, as benthic communities of FGBNMS were temporally stable during this study period (p = 0.191). Additional species and species group differences across all surveyed strata are explored in more detail throughout the remainder of this chapter.

Table 3.3. Benthic groups accounting for 95% of the dissimilarity between shallow and upper mesophotic (UM) sites listed in descending order of contribution. Values presented are 4th root transformed.

Benthic Groups	Shallow Ave. abund.	UM Ave. abund.	Contribution %	Cumulative %
Maximum height of coral or rock	10.50	8.54	29.71	29.71
Hard coral cover	7.33	5.72	17.59	47.30
Algal cover	5.75	6.90	15.36	62.66
Rubble cover	0.63	1.42	10.81	73.47
Sand cover	0.51	1.44	10.15	83.61
Hydrocoral cover	0.56	0.74	5.57	89.18
Sponge cover	0.72	0.81	5.35	94.54

3.3.1. Abiotic Cover

Table 3.4. Mean percent cover (± SE) of abiotic categories by bank and relief strata from diver surveys (2010-2012).

Bank	Relief	N	Hardbottom	Rubble	Sand
East	Low	34	87.8 (±3.1)	11.4 (±3.1)	0.04 (±0.04)
	High	145	96.0 (±0.8)	2.1 (±0.5)	1.5 (±0.4)
West	Low	11	89.9 (±3.4)	4.7 (±2.3)	5.4 (±2.8)
	High	101	95.0 (±1.0)	1.3 (±0.5)	3.2 (±0.8)
Coral Reef		291	94.5 (±0.7)	3.0 (±0.5)	2.1 (±0.4)

Hard substrate, or hardbottom, was the dominant abiotic cover within all surveyed strata (site ranges: 23-100%; Table 3.4) and was consistent between depth strata. Total hardbottom cover for the coral reef (overall mean 94.5 ± 0.7%; shallow 96.5 ± 0.6%; UM 87.6 ± 1.9%) was higher than previous studies (85%, Rezak et al., 1985; 89%, Caldow et al., 2009). Between banks, hardbottom cover was similar (Table 3.4; Z = -0.49397, p = 0.6213) and, as expected, positively related to rugosity (ρ = 0.305, p < 0.0001), but negatively correlated with depth (ρ = -0.3788, p < 0.0001). Rugosity and depth were negatively correlated (ρ = -0.6091, p < 0.0001).

Rubble was patchy, occurring at 98 of 291 sites, and contributed 3% (± 0.5) of the total abiotic cover (Table 3.4; Figure 3.4). Rubble was largely comprised of dead *Madracis auretenra*, likely resulting from Hurricane Rita damage in 2005 (Hickerson et al., 2008; Precht et al., 2008; Robbart et al., 2008) and/or Hurricane Ike in 2008 (Hickerson, pers. com.). Rubble cover was positively related to depth (ρ = 0.3799, p < 0.0001) and negatively related to rugosity (ρ = -0.3277, p < 0.0001). There was no significant difference in rubble cover by bank (Z = 0.15142, p = 0.8796; Table 3.4). Rubble cover within the shallow stratum (15%) was considerably less than a previous study (46%, Caldow et al., 2009). This difference between studies may be attributed to this study's extended sample size, area, a function of the randomly stratified sampling design, or to a longer temporal interval since the last disturbance. Higher (>15%) *M. auretenra* rubble cover primarily occurred at or below 30 m depth,

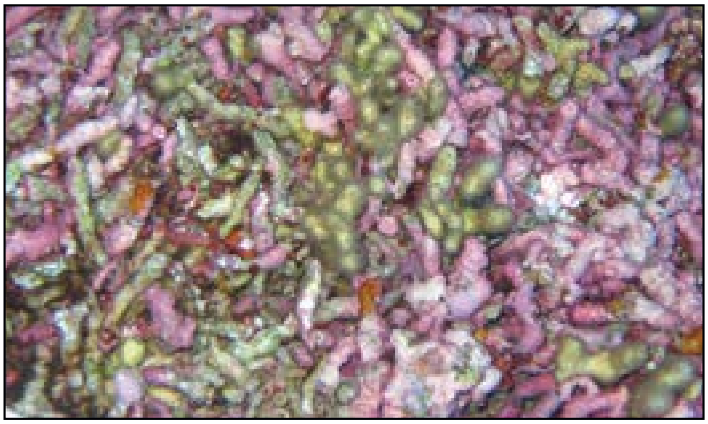
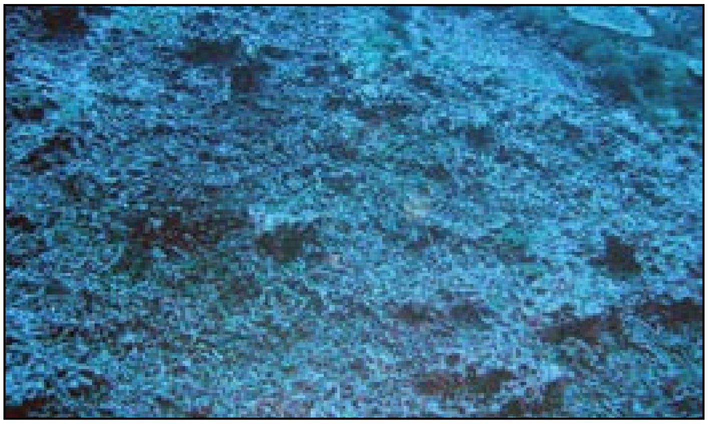

Figure 3.4. Example of rubble dominated low relief site on East bank. Photos: NOAA NOS/NCCOS/CCMA

near transitions between high and low relief strata, and contributed 10.8% to the dissimilarity between depth strata (Table 3.3). This is consistent with Bright et al. (1984) that observed greater rubble cover on the steep margins of the coral reef.

Overall, sand was a small percentage of the abiotic habitat (2.1% ± 0.4) and was observed at 101 of 291 sites. The low relief stratum had the greatest variability in mean sand cover with the highest cover occurring in the WB low relief strata (5.4% ± 2.8) and lowest cover on EB low relief (0.04%; occurring at 2 sites; Table 3.4). Sand was recorded at 57% of the UM survey sites, but only 28% of the shallow coral reef sites and contributed 10.15% to the dissimilarity between benthic communities in these strata (Table 3.3). Sand cover within the shallow strata (1%) was similar to Caldow et al. (3%; 2009). Sand cover was significantly greater ($Z = 4.60$, $p < 0.0001$) at UM sites (5.5%) than at shallow sites (1.1%) and positively correlated with depth ($\rho = 0.169$, $p = 0.0038$). The difference in sand cover between depth strata was in part due to the habitat configuration within the UM zone. Many of the high relief UM surveys were along reef-sand interfaces. While sand patches were found at all depths, extensive sand expanses were restricted to deeper habitats along the sloping edges of the coral reef within UM stratum (>33.5 m; Figure 3.5).

Multivariate analyses of abiotic substrate composition (comprised of hard, rubble, and sand substrate percent cover at each site) identified significant differences in abiotic substrate between shallow and upper mesophotic depth zones by year ($p = 0.047$).


Figure 3.5. Sand patch within shallow area of the coral reef (left) and a reef-sand interface found below 40 m (right). Photos: E. Hickerson (NOAA NOS/ONMS/FGBNMS) and C.A. Buckel (NOAA NOS/NCCOS/CCFHR)

3.3.2. Biotic Cover

Total biotic cover (defined as the sum of all biotic cover groups; Table 3.2) was very high, exceeding 90% at a majority of the sites (214 of 275 sites). Algae and hard corals contributed to the majority of the biotic cover in the sanctuary (Table 3.5), consistent with previous studies (Aronson et al., 2005; Caldow et al., 2009; Johnston et al., 2013). High biotic cover occurred within all strata (Table 3.5). While some of the lower biotic cover sites were deeper (>30 m; Figure 3.6), there was no significant relationship between total biotic cover and depth. Multivariate community analyses of biotic data identified differences between shallow and upper mesophotic, relief, and bank (p = 0.035).

*Table 3.5. Mean percent cover (± SE) for biotic categories. * Mean percent cover estimates were calculated using 275 sites.*

Bank	Relief	N	Algae*	Hard Corals	Hydrocorals	Sponges
East	Low	34	62.5 (±4.0)	31.5 (±4.1)	1.1 (±0.3)	1.9 (±0.3)
	High	145	35.0 (±1.6)	58.9 (±1.6)	0.7 (±0.1)	0.6 (±0.1)
West	Low	11	76.1 (±4.8)	18.6 (±6.8)	0.8 (±0.3)	3.2 (±1.9)
	High	101	35.7 (±1.9)	53.5 (±2.1)	0.9 (±0.2)	1.2 (±0.2)
Coral Reef		291	39.9 (±1.2)	52.3 (±1.4)	0.8 (±0.1)	1.1 (±0.1)

Spatial trends and statistical differences in specific biotic cover groups (scleractinian corals, hydrocorals, algae, sponge, and invertebrates) and individual coral species are explored in more detail in the following sections. Very few benthic cover differences were identified between survey years within the shallow stratum and no significant differences for the UM depth stratum. The few temporal trends that could be detected in this study are also noted in the following sections.

3.3.3. Coral Cover

A total of 31 different scleractinian species within 13 genera were identified during this study. Additionally, two hydrocoral species within the genus *Millepora* were also recorded. The maximum number of scleractinian species or species groups occurring at an individual site was 12 (of 31 species/species groups recorded). Twenty coral species or species complexes (e.g., *Pseudodiploria* spp., *Orbicella annularis* species complex) were recorded in the UM strata, while 31 were documented within the shallow strata. This is a greater species richness than a recent FGBNMS diver-based assessment within the same shallow depth range, that identified 15 species and 10 genera (Caldow et al., 2009), likely due to the greater spatial coverage and/or sample size of this study. Species reported in these surveys but not the 2009 study (Caldow et al., 2009) included: *Agaricia humilis*, *Pseudodiploria labyrinthiformis*, *Helioceris cucullata*, *Madracis pharensis*, *Meandrina meandrites*, *Porites divaricata* and *Siderastrea radians*. *Helioceris cucullata* was reported in a previous study (Bright et al., 1984). These species were generally rare, occurring at few sites (range 1-9 of 291) and contributed only a small amount to total coral cover (range: 0.0016-0.081%). The majority of the coral species recorded in this study have been verified by specimen or photo collections. Some of the species recorded require verification (specimen or photo collection) for inclusion on the official FGBNMS coral species list (http://flowergarden.noaa.gov/document_library/aboutdocs/fgbnmscoralcapspecies.pdf); these include: *Agaricia grahamae*, *A. humilis*, *Agaricia lamarcki*, *P. labyrinthiformis*, *M. pharensis*, *M. meandrites*, *P. divaricata* and *Porites porites*.

The dominant coral cover by genera were *Orbicella*, *Pseudodiploria*, *Montastraea*, *Colpophyllia* and *Porites*, which together contributed 88.6% of the total coral cover. Occurrence of some species and genera did change by depth strata (Figure 3.7). The dominant genera recorded here are similar to previous surveys at a smaller spatial scale and depth range (Aronson et al., 2005; Caldow et al., 2009; Johnston et al., 2013).

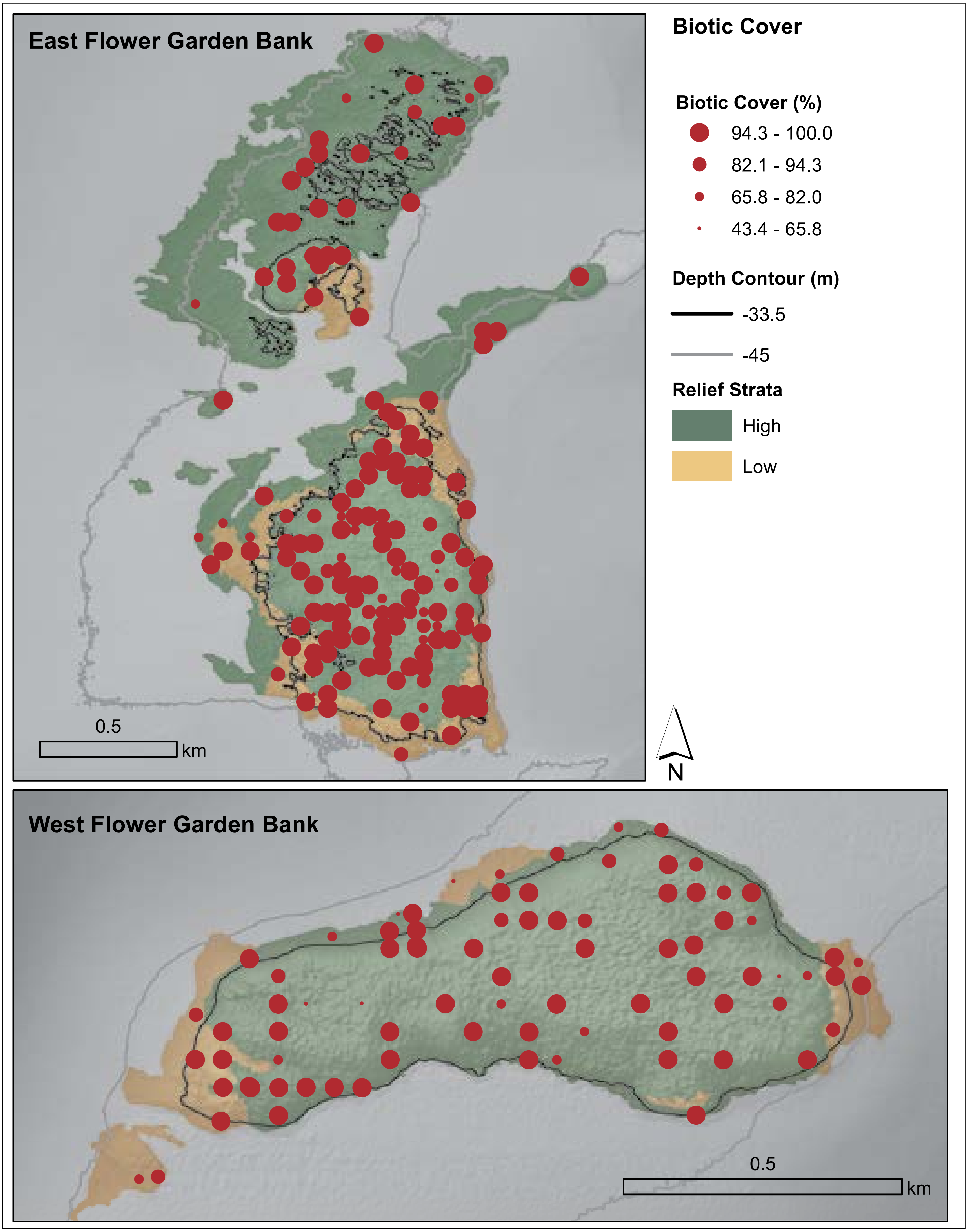

Figure 3.6. Observed mean biotic percent cover recorded during dive surveys from 2010 – 2012 (N = 275).

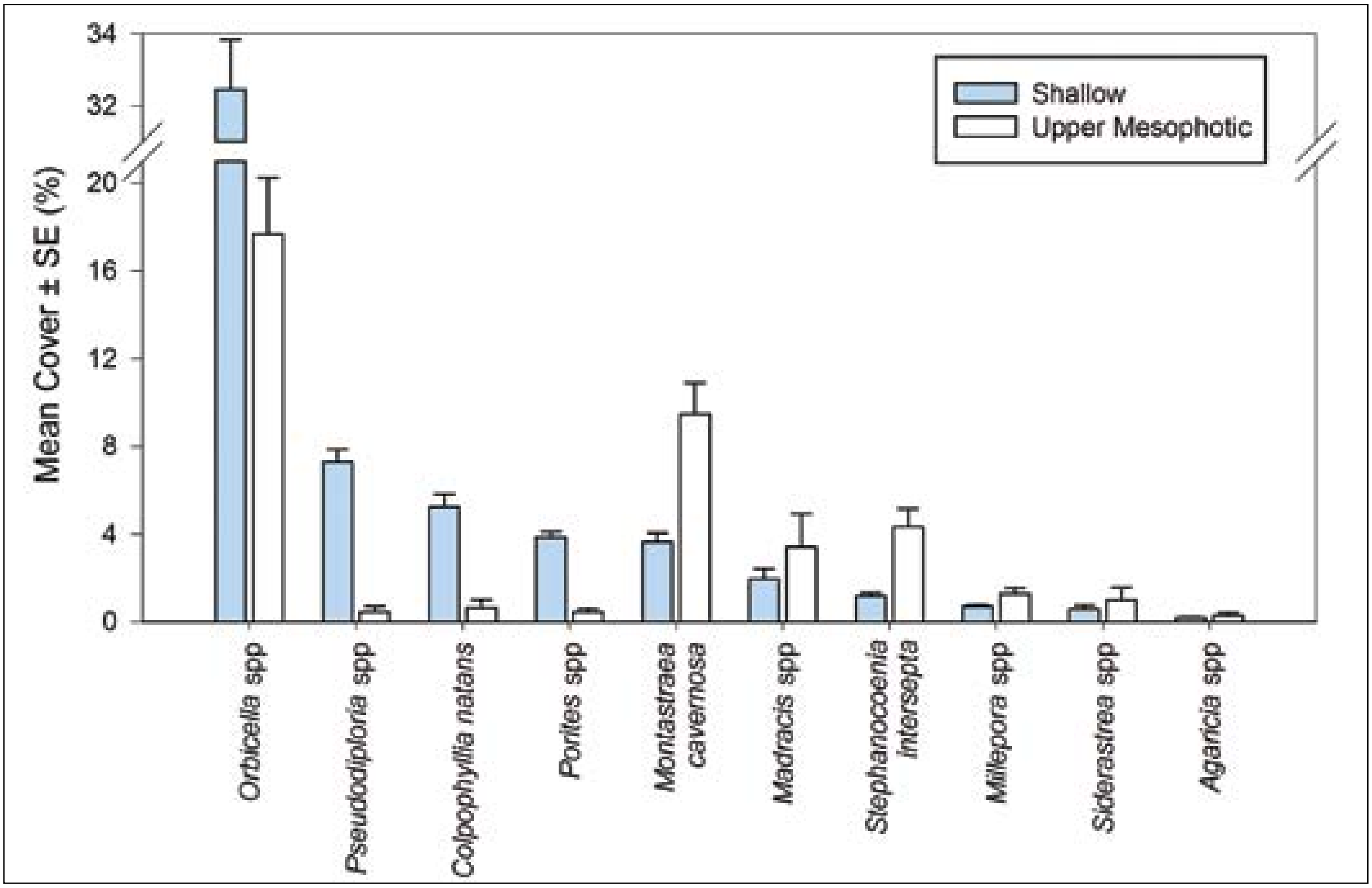

Figure 3.7. Mean coral percent cover for the top 10 coral genera, including Millepora *spp., by depth strata from diver surveys (2010 – 2012). Four species or species groups are not shown to highlight more abundant species. Not shown are:* Mussa angulosa*,* Helioceris cucullata*,* Scolymia *sp.,* Meandrina meandrites *and otherwise unidentifiable scleractinian species.*

Coral species richness declined with depth (Figure 3.8; $r = -0.614$, $p < 0.0001$) and increased with rugosity ($r = 0.504$, $p < 0.0001$). Sites with the highest species richness (10-12 scleractinian species) occurred between depths of 20 and 30 m. Within the shallow strata, species richness was significantly higher on WB than on EB (7.5 versus 6.9 species, $t = 2.278$, $p = 0.0237$). The pattern was the opposite at upper mesophotic sites, with significantly greater species richness ($t = -3.25$, $p = 0.0021$) on EB (4.4 species/site) than WB (2.8 species/site).

Coral species richness reported here (maximum 12 hermatypic corals/site) is high compared to those found on hard substrate (primarily platforms) elsewhere in the Gulf of Mexico (nine hermatypic corals reported from 42 platforms at depths ≤ 37 m; Sammarco et al., 2012). The dominant hermatypic coral (*Madracis decactis*) reported in Sammarco et al. (2012) occurred in generally low cover within this study (mean 0.4 ± 0.1%; site maximum 22%). Sammarco et al. (2012) also observed three ahermatypic corals (*Tubastraea coccinea*, *Oculina diffusa*, and *Phyllangia americana*) that were not observed in the present study. *T. coccinea* is an invasive species that can dominate platforms in the Gulf of Mexico (Sammarco et al., 2012) and has been recorded on the gas platform in FGBNMS (Hickerson et al., 2008). The species was removed in isolated clusters on WB reefs (Hickerson et al., 2008), but was not observed on either bank in this study.

Seventy-four of 291 surveyed sites had scleractinian cover ≥70%, with a maximum site cover of 97% (Figure 3.9). The mean coral cover for the study area (shallow: 56.5 ± 1.4%; UM: 38.9 ± 2.9%; Table 3.4) was higher than that reported for Caribbean and western Atlantic locations (10% cover; Gardner et al., 2003) and similar to previous estimates of scleractinian cover within the shallow regions (<33.5 m) of the coral reefs at FGBNMS (50%, Gittings, 1998; approximately 56%, Aronson et al., 2005; 48%, Caldow et al., 2009; approximately 54%, Johnston et al.,

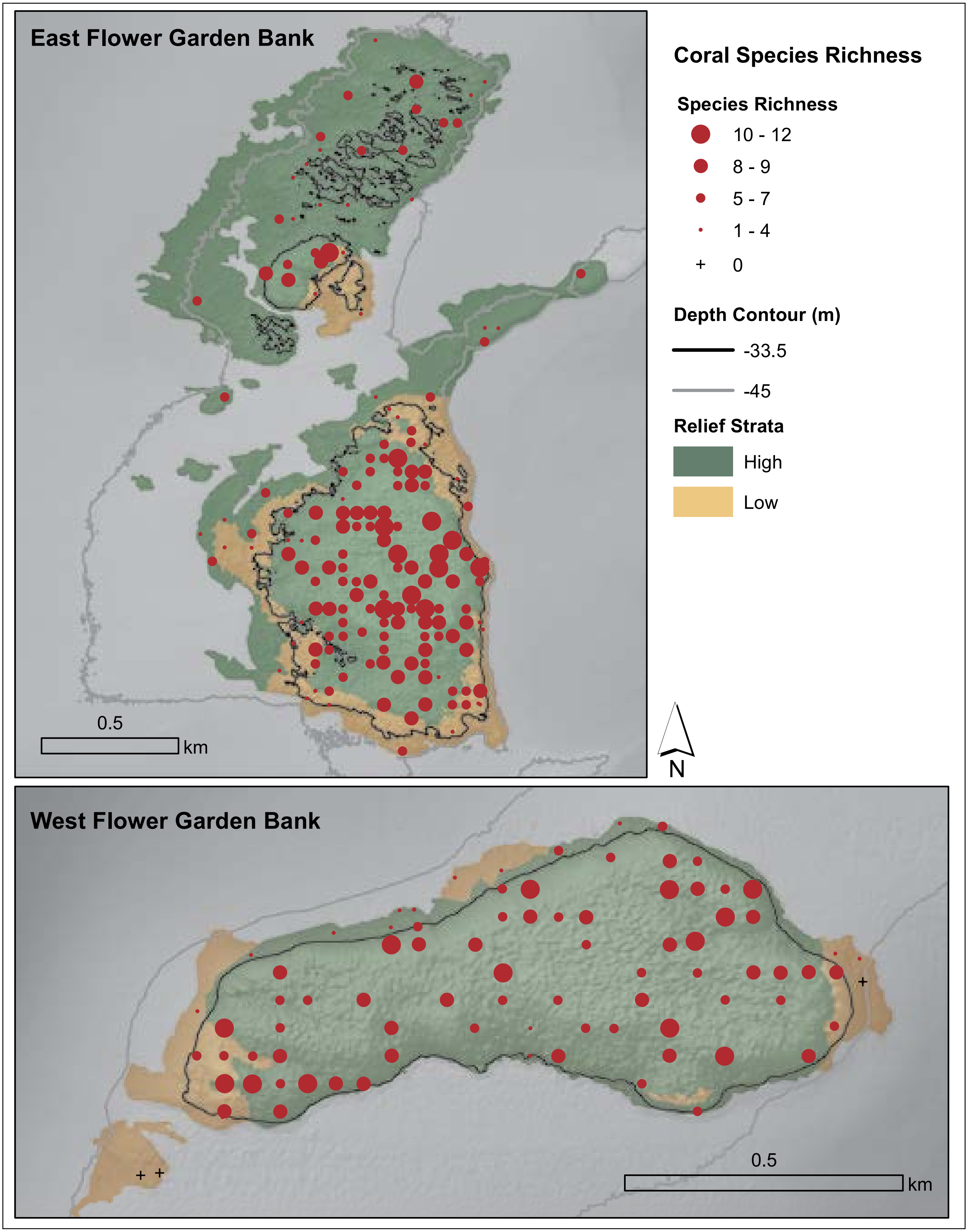

Figure 3.8. Observed scleractinian species richness recorded during dive surveys from 2010 – 2012.

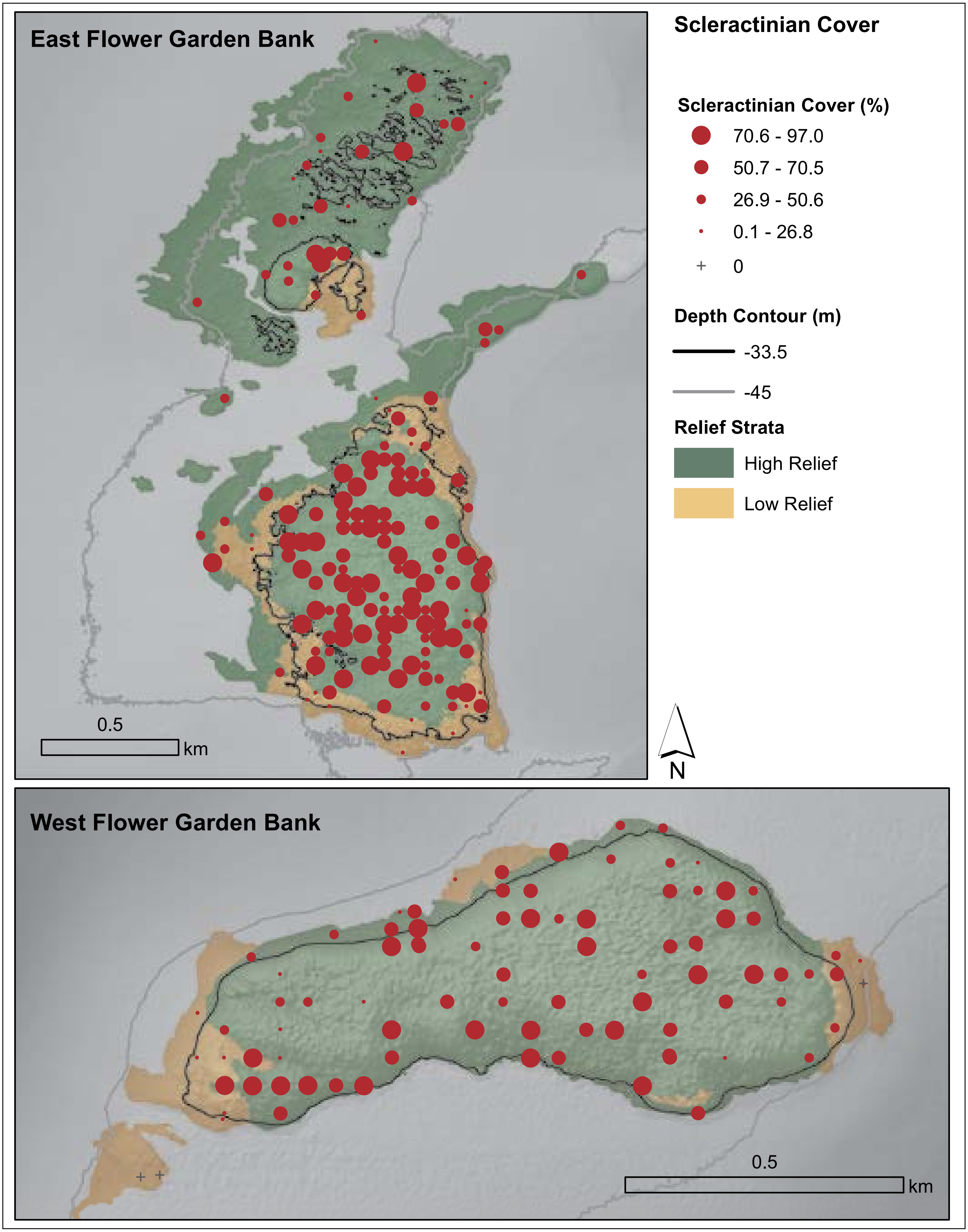

Figure 3.9. Observed mean scleractinian percent cover recorded during dive surveys from 2010 – 2012.

2013). The consistency of these estimates, spanning more than 20 years, highlights the stability of the FGBNMS coral cover; a stark contrast to coral declines elsewhere in the tropical Atlantic (Gardner et al., 2003, 2005). As expected given that scleractinians are primary reef builders, scleractinian cover was positively correlated with rugosity ($r = 0.3479$, $p < 0.001$). A previous study reported higher coral cover at approximately 26 m (edge of the coral cap) compared to the top of the cap (Caldow et al., 2009). With the extended depth and spatial coverage of this study, it was found that cover was negatively correlated with depth ($r = -0.4494$, $p < 0.0001$) and did not increase near the cap edge (26-30 m; Figure 3.9).

Although the reefs of FGBNMS are coral dominated, there were three sites where quadrats did not contain any scleractinians (Figure 3.9); however, biotic cover was still high. All were UM sites within the low relief stratum of the WB and had extensive hard substrate cover (range: 67.5-93%). Two sites were dominated by algae (46 and 90.3%) and one site had above average sponge cover (21.5%). Two additional UM sites had less than 1% scleractinian cover (one each: EB High relief and EB Low relief) and were dominated by algae (93%, 99% respectively).

The extended depth range of this study allowed us to capture the morphological transition of coral species from boulder to plating forms (Figure 3.10), as represented by the positive correlation between rugosity and coral cover and negative correlation between rugosity and depth. Many coral species transition from high relief mounds to low relief plating morphology at depth to optimize light capture (Dustan, 1975; Jaubert, 1977; Anthony et al., 2005), thus the negative correlation between rugosity and depth was consistent with our expectations. However, it is important to reiterate that the rugosity estimate was calculated for the 50 x 50 m study grid cell and was not scaled to the level of a coral colony, thus conclusions of species or functional group level relationships should be considered suggestive.

Figure 3.10. Examples of mounding morphology in the shallow zone (left) and coral plating morphology in the upper mesophotic zone (right). Photos: NOAA NOS/NCCOS/CCMA and C.A. Buckel (NOAA NOS/NCCOS/CCFHR)

Coral species in the reef community were different between depth strata (shallow versus UM) and relief between banks ($p = 0.001$). Between depth strata, average dissimilarity was 52.9%, and nine coral species accounted for 62.7% of this dissimilarity. *Orbicella franksi*, *Pseudodiploria strigosa*, *Montastraea cavernosa* and *Orbicella faveolata* accounted for 44.29% of the dissimilarity. *O. franski*, *P. strigosa*, *P. astreoides* had greater cover at shallow sites, and *M. cavernosa* had greater cover at UM sites (Table 3.6; Figure 3.11). There were no significant differences within the coral community by year or bank, similar to findings of Aronson et al. (2005). Coral species percent cover and beta diversity were each significantly different by both depth and rugosity, although additional factors were clearly important as well, as both have limited explanatory power ($p < 0.001$ for each; beta diversity $R^2 = 0.13$ for depth, 0.02 for rugosity; percent cover $R^2 = 0.089$ for depth, 0.061 for rugosity).

Figure 3.11. Montastraea cavernosa *at FGBNMS. Photo: M. Winfield (UNCW)*

Table 3.6. Coral species and other benthic groups accounting for 91% of the dissimilarity between shallow and upper mesophotic sites listed in descending order of contribution. CCA = crustose coralline algae, B/T/V = barrell/tube/vase sponge. Values presented are 4th root transformed.

Species	Shallow Ave. abund.	Upper mesophotic Ave. abund.	Contribution %	Cumulative %
Orbicella franksi	4.40	2.74	13.88	13.88
Macroalgae	4.26	6.02	10.88	24.70
Pseudodiploria strigosa	2.12	0.18	8.30	33.01
Montastraea cavernosa	1.35	2.35	8.30	41.30
Turf algae	2.86	1.77	8.28	49.58
Orbicella faveolata	1.73	0.68	7.42	57.01
Colpophyllia natans	1.54	0.20	6.39	63.40
Porites astreoides	1.74	0.37	6.02	69.42
Stephanocoenia intersepta	0.73	1.51	5.73	75.16
CCA	1.68	2.12	4.20	79.35
Madracis auretenra	0.31	0.54	3.41	82.76
Millepora alcicornis	0.44	0.74	3.24	86.00
Encrusting sponge	0.44	0.52	2.49	88.49
B/T/V sponge	0.42	0.40	2.33	90.81

Protected Coral Species

Currently there are two Caribbean/Western Atlantic coral species (*Acropora palmata* and *Acropora cervicornis*) protected by the U.S. Endangered Species Act (ESA; Federal Register 71 FR 26852 [NOAA, 2006]). Both of these species are currently listed as threatened but are candidate species for reclassification to endangered status, as are five additional species (*Orbicella faveolata*, *O. annularis*, *O. franksi, Dendrogyra cylindrus*, and *Mycetophyllia ferox*; Federal Register 77 FR 73219 [NOAA, 2012]). Two additional species are candidate species for listing under threatened status (*A. lamarcki* and *Dichocoenia stokesii*; Federal Register 77 FR 73219 [NOAA, 2012]).

Two *A. palmata* colonies were found within FGBNMS, one at each bank, during previous studies (Aronson et al., 2005; Zimmer et al., 2006); however, no *A. palmata* were encountered in this study. *A. lamarcki* was reported at four sites in our study, between depths of 20-29 m. All occurrences were low percent cover (range 0.05-0.175%) with three reports on EB and one on WB. *Orbicella annularis*, *O. faveolata*, and *O. franksi* were recorded at many sites and often in high cover (exceeding 50% combined cover at 51 of 291 sites). The distribution and cover of these species are described in more detail in the Star Coral section below. No other current or candidate ESA listed coral species were recorded in this study.

Star Corals (*Orbicella* species, *Montastraea cavernosa*)

Montastraea cavernosa, *O. annularis*, *O. faveolata*, *O. franksi* and *Stephanocoenia intersepta* belong to a larger group of stony corals commonly called 'star corals' and are considered mounding or boulder corals (Figure 3.12). Within the shallow strata, these species typically form large mounds, occasionally exceeding 2 m in height. However, at deeper depths (approximately >32 m; *O. franksi* plate at 35 m according to Dustan, 1975), the plating morphotype of these species became more prevalent (Figure 3.10). The *Orbicella* species were previously classified within the *Montastraea* genus; however, recent genetic analyses identified these species as belonging to separate genera (Budd et al., 2012). On rare occasions (7 sites, 0.18% total coral cover), a colony within the *Orbicella* genus could not be identified to species level, so *O. annularis* species complex was recorded. Since rare, this designation will not be examined further, though some species specific analyses may be confounded due to the presence of these points.

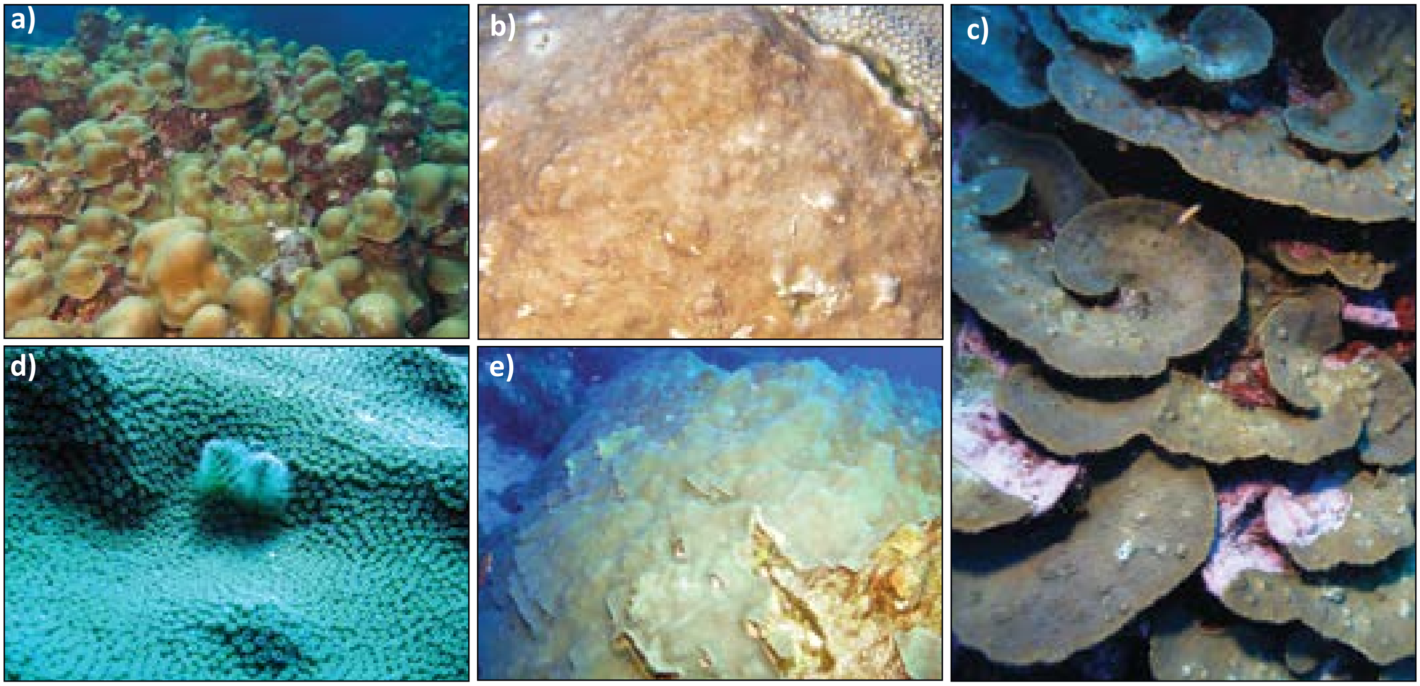

Figure 3.12. Photographs of star coral species: a) Orbicella annularis*, b)* Stephanocoenia intersepta, *c)* Oribicella franksi*, d)* Montastraea cavernosa, *and e)* Orbicella faveolata*. Photos: a) R. Eckert (NOAA NOS/ONMS/FGBNMS), b-c) C.A. Buckel (NOAA NOS/NCCOS/CCFHR), and d-e) NOAA NOS/NCCOS/CCMA*

Benthic Communities

When combined, the star corals account for 65% of the total coral coverage for this study (291 sites, 2010-2012). All but *O. annularis* were among the most dominant species, in terms of cover. *O. franksi*, *O. faveolata* and *M. cavernosa* were among the top four species in total coral cover (percent of total coral cover: 43.5%, 10.8%, 9.5%, respectively). *O. annularis* ranked 9th overall at 1.2% of the total coral cover. These findings support the conclusions of previous studies identifying the FGBNMS coral communities as dominated by *O. annularis* species complex (Rezak et al., 1985; Caldow et al., 2009; Zimmer et al., 2010; Johnston et al., 2013). Within the UM strata, cover for each star coral species was consistent between surveyed years (2011-2012). Inter-annual differences within the shallow strata were noted within the individual species sections, where they occurred.

Orbicella franksi (boulder star coral)

O. franksi was the dominant coral species (Figure 3.13), especially in the high relief strata for both UM and shallow surveys (mean cover 15 and 25% respectively, 23% overall; Figure 3.14). It was found at 240 of 291 sites. *O. franksi* occurrence was significantly related to both rugosity ($Z = 7.037$, $p < 0.001$) and depth ($Z = -6.566$, $p < 0.001$). Sites without *O. franksi* were typically at depths >30 m (41 of 51 sites; Figure 3.15), consistent with the finding of a significant negative relationship between *O. franksi* and depth ($\rho = -0.23$, $p < 0.0001$). *O. franksi* was the coral species responsible for the greatest differences recorded between the two depth strata, contributing 13.8% (Table 3.6). Cover in the shallow stratum was uniform between banks. However, within the UM stratum, EB cover was significantly greater than WB (mean cover: 20%, 6% respectively; $Z = -3.46$, $p = 0.0005$). *O. franksi* was not recorded within the WB low relief stratum of the UM surveys; this stratum had the lowest total coral cover of all strata (11.7 ± 5.7%). Cover was significantly different by relief strata ($Z = -6.305$, $p < 0.0001$). It was less common in the low relief strata (mean 8% for both shallow and UM surveys) than in the high relief strata where cover was very high at some sites (maximum % cover = 87.5%). Within the shallow strata, *O. franksi* cover was significantly different between 2010 and 2012; however, the difference was due to higher cover at a few sites in 2012 (2010 maximum: 31.3%, 2012: 87.5%). Cover in 2011 was not different from 2010 and 2012, suggesting differences were due to spatial variability and not actual increases in *O. franksi* cover.

Figure 3.13. Image of Orbicella franksi *(top) and* Obicella faveolata *(bottom) in FGBNMS. Photos: NOAA NOS/NCCOS/CCMA*

Orbicella faveolata (mountainous star coral)
The third most dominant coral species overall was *O. faveolata* with a mean cover of 5.6 ± 0.6% (Figure 3.12 and 3.13). This species occurred at 134 of 291 sites, between depths of 19-44 m, with cover ranging from 0.05-54.8% (Figure 3.16). Occurrence was significantly greater at shallower depths ($Z = -5.391$, $p < 0.001$) and higher rugosity ($Z = -5.379$, $p < 0.001$). Similarly, cover was significantly correlated to depth ($\rho = -0.368$, $p < 0.001$) and rugosity ($\rho = 0.355$, $p < 0.001$). Occurrences were rare at low relief sites (10 of 45 sites) and were generally in low cover in these habitats (0.4 ± 0.2% shallow; 1.3 ± 0.9% UM). Percent cover was consistent across years within each depth strata but was significantly greater in shallow (6.6%) than UM (2.3%) surveyed sites ($Z = -4.01$, $p < 0.0001$; Figure 3.14). Although *O. faveolata* cover was greater on the EB than WB (6.3 ± 0.8% and 4.5 ± 0.8%, respectively), the difference was not statistically significant for all sites combined and when examined within depth strata.

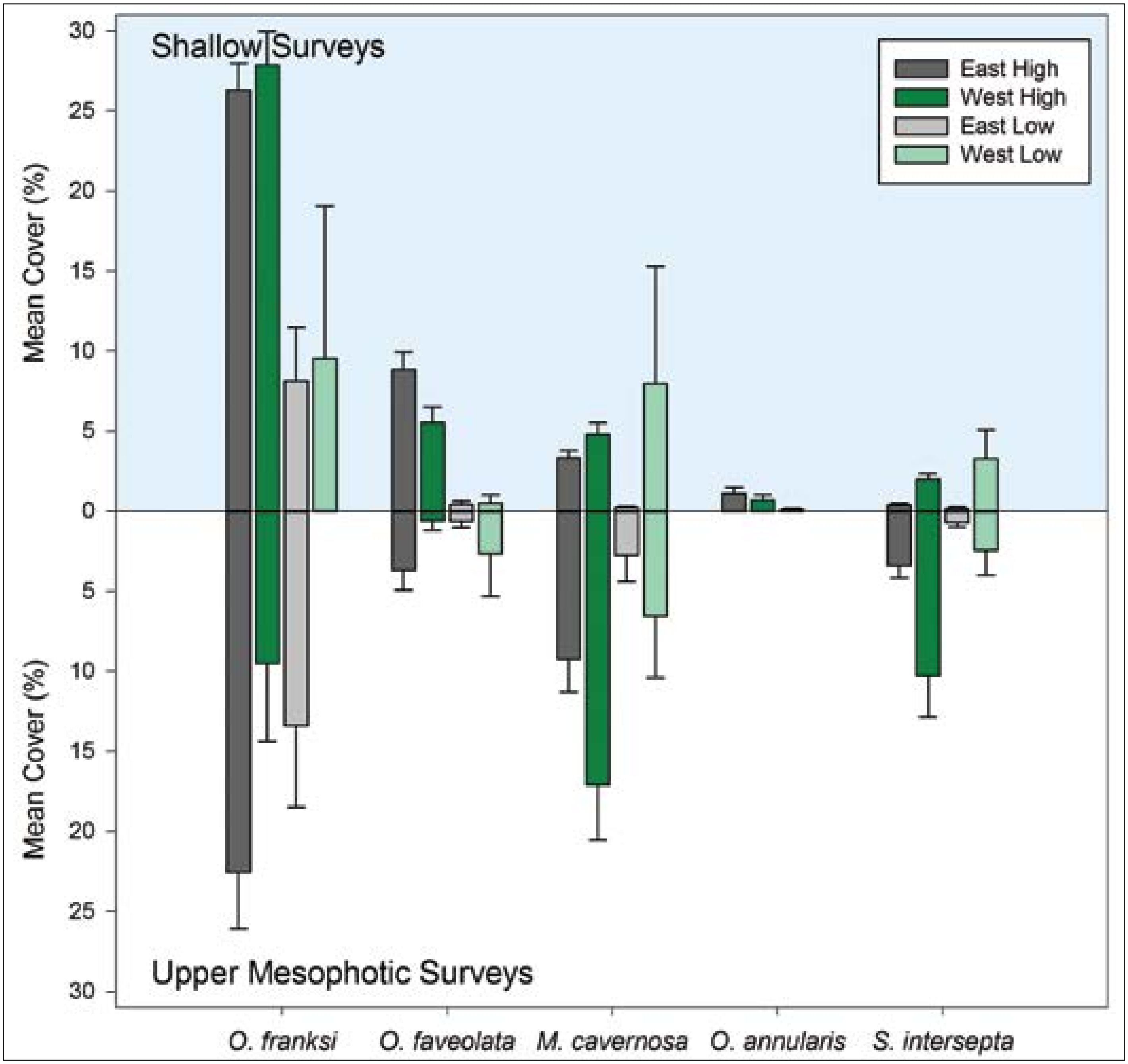

Figure 3.14. Mean percent cover (SE) for Orbicella *species,* Montastraea cavernosa*, and* Stephocoenia intersepta *by strata from diving surveys (2010 – 2012).*

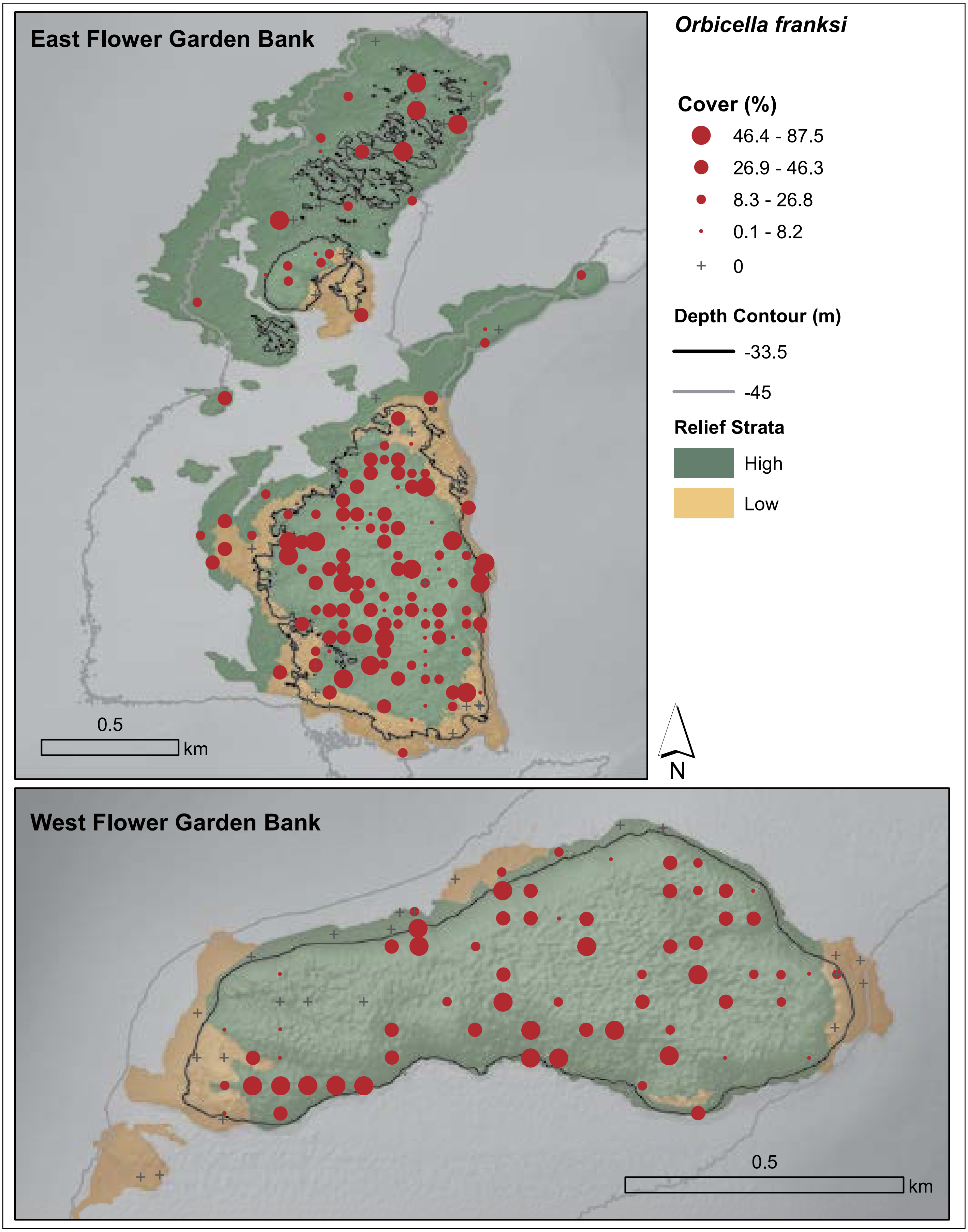

Figure 3.15. Observed mean Orbicella franksi *(boulder star coral) percent cover recorded during dive surveys from 2010 – 2012.*

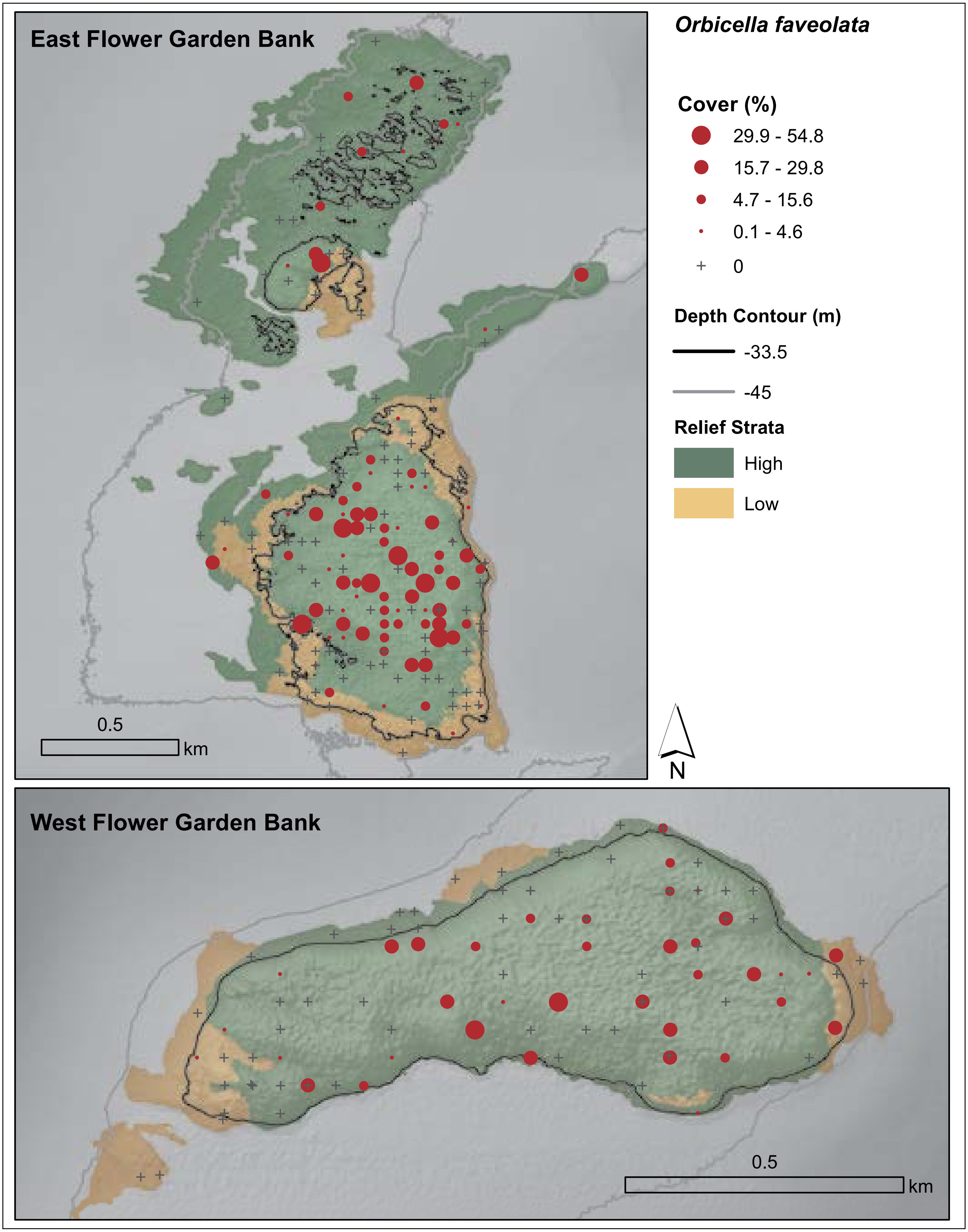

Figure 3.16. Observed mean Orbicella faveolata *(mountainous star coral) percent cover recorded during dive surveys from 2010 – 2012.*

Montastraea cavernosa (great star coral)

M. cavernosa (Figure 3.17) was found at 212 of 291 sites with cover ranging from 0.1-47% (Figure 3.18). *M. cavernosa* occurrence was not significantly different between depth strata or rugosity. However, *M. cavernosa* cover increased in deeper ($Z = 3.014$, $p = 0.0026$) and higher ($Z = -4.636$, $p < 0.0001$) relief habitats (Figure 3.14); consistent with multivariate analyses (Table 3.6). When cover was examined within depth strata, higher relief habitats remained significantly different from low relief for both depth strata. No cover differences were identified across years within each depth strata. Cover was significantly greater ($Z = 2.86$, $p = 0.0042$) on the WB than EB (6.7 ± 0.9% versus 3.9 ± 0.5%), possibly due to the greater overall depth of the WB. While *M. cavernosa* cover was greater in high relief habitats, the lack of correlation with rugosity was not unexpected as rugosity decreased with increasing depth. This lack of relationship with rugosity was possibly due to how it was measured (e.g., remotely and over a large scale [see Chapter 2]) and/or to the observed transition from mounding coral forms to plating forms at depth. There appears to be a species trade-off with depth: as cover of *M. cavernosa* increases, *O. franksi* and *O. faveolata* cover decreases.

Figure 3.17. Image of Montastraea cavernosa *in FGBNMS. Photo: J. Voss (HBOI-FAU and NOAA/CIOERT)*

Orbicella annularis (lobed star coral)

Among the *Orbicella* species, *O. annularis* cover was the lowest (Figure 3.19; Figure 3.14), ranging from 0.18-24.3% at surveyed sites. It was present at 31 of 291 sites (Figure 3.20). Similar to *O. faveolata* and *O. franksi*, *O. annularis* was primarily found at shallow sites, with the maximum depth of occurrence recorded at 32.9 m. There was a significant negative relationship between *O. annularis* occurrence and depth ($Z = -3.071$, $p < 0.001$), but not rugosity. While sites with the highest *O. annularis* cover were on the EB (Figure 3.20), there was no statistical difference in cover between banks. When analyzed within the shallow stratum where the majority of *O. annularis* was reported, there was still no cover difference by bank or relief strata. Annual differences in occurrence were recorded for *O. annularis* only for the shallow strata. This species was recorded in 2010 and 2011, with low cover (median cover 3.25%), but not in 2012 surveys. No known bleaching or disease events occurred between 2011 and 2012 surveys to affect *O. annularis* cover; therefore it is likely that the absence of *O. annularis* in 2012 is due to a combination of low occurrence and low cover rather than actual declines.

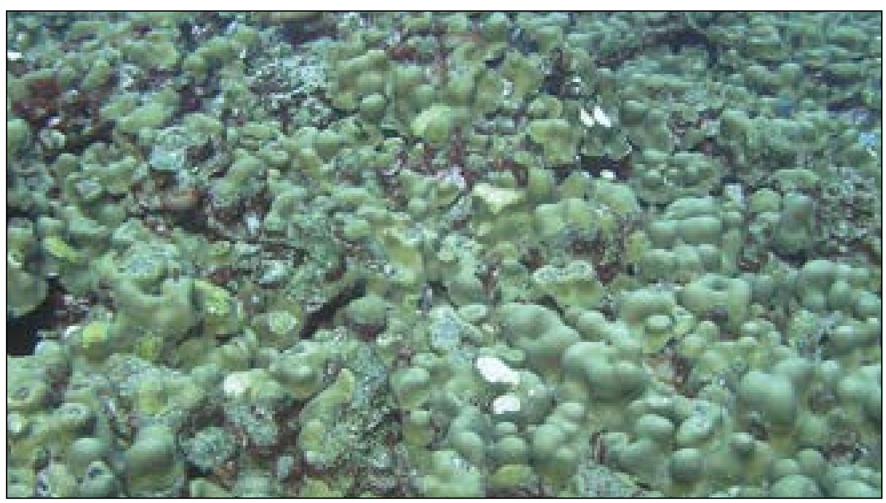

Figure 3.19. Image of Orbicella annularis *in FGBNMS. Photo: E. Hickerson (NOAA NOS/ONMS/FGBNMS)*

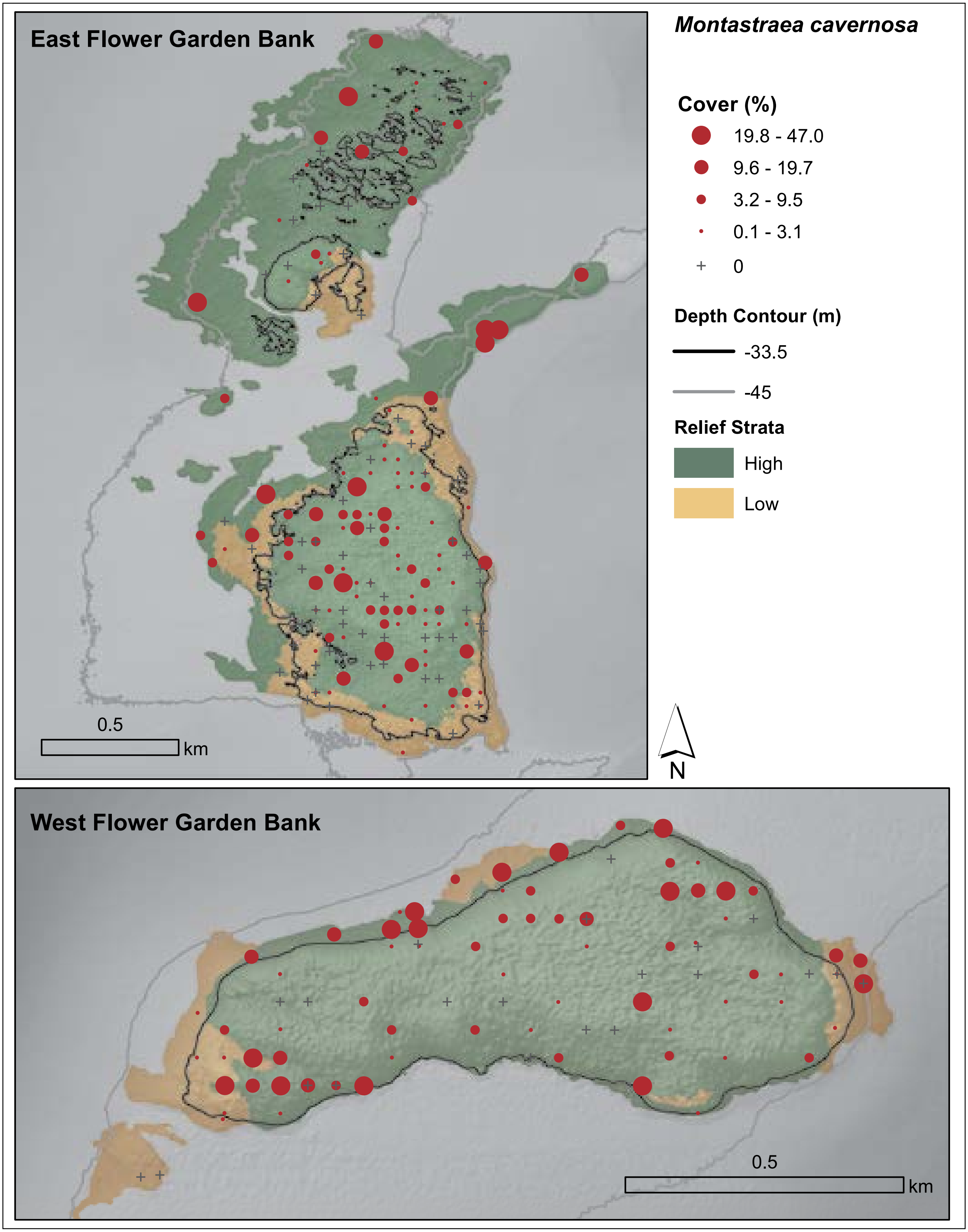

Figure 3.18. Observed mean Montastraea cavernosa *(great star coral) percent cover recorded during dive surveys from 2010 – 2012.*

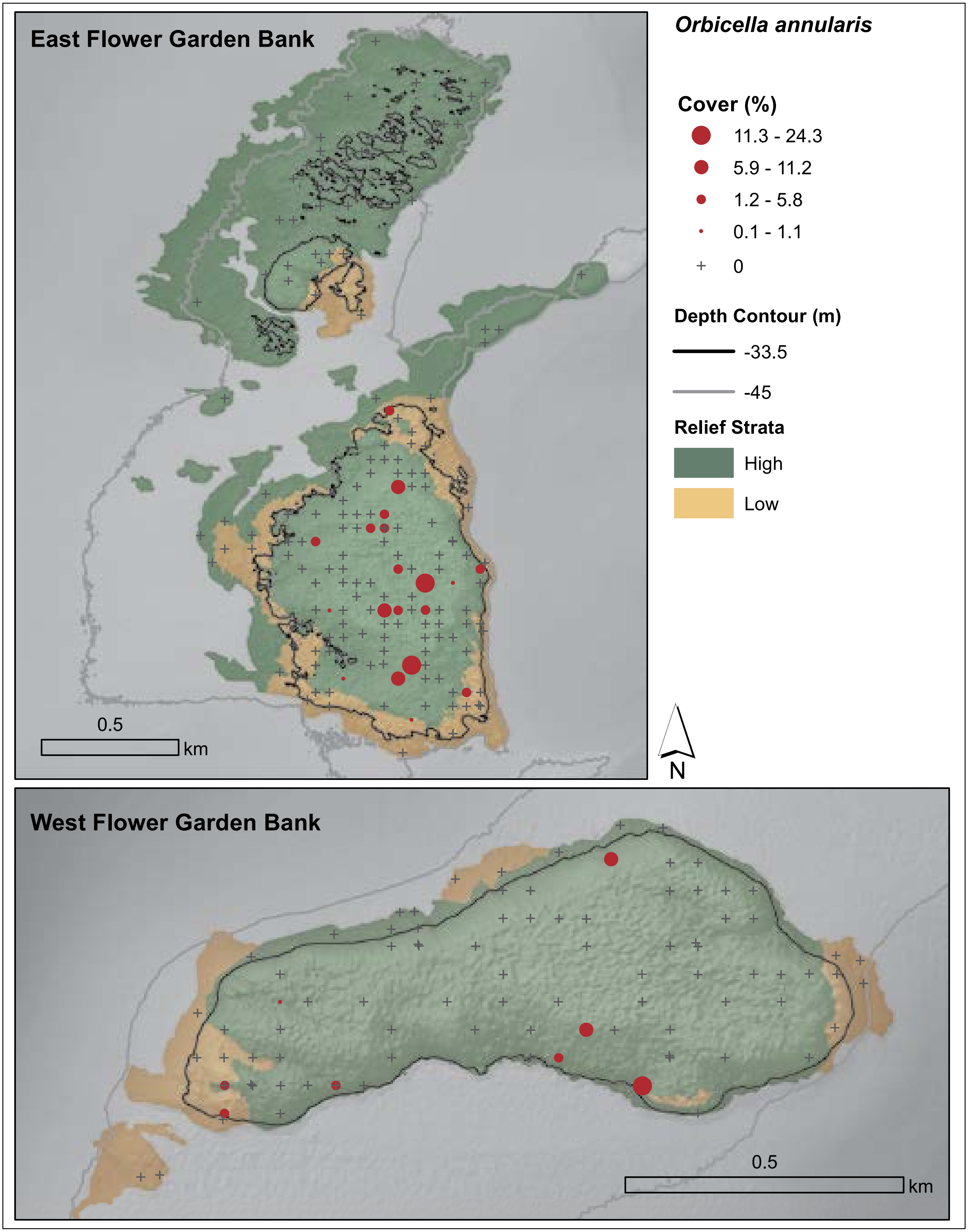

Figure 3.20. Observed mean Orbicella annularis *(lobed star coral) percent cover recorded during dive surveys from 2010 – 2012.*

Stephanocoenia intersepta (blushing star coral)

Figure 3.21. Image of Stephanocoenia intersepta *in FGBNMS. Photo: C.A. Buckel (NOAA NOS/NCCOS/CCFHR)*

Stephanocoenia intersepta (Figure 3.21) distribution was similar to *M. cavernosa*, with increasing cover at depth (ρ = 0.1227, p = 0.0078; Figure 3.22). *S. intersepta* was found at 171 sites but was recorded at more UM sites (67%) than shallow (43%). UM cover was significantly higher (mean: 4.3%) than in the shallow strata (1.1%; Z = 3.89, p < 0.0001) and contributed 6% to the dissimilarity between these two depth strata (Table 3.6). Cover of *S. intersepta* was greater on WB (3.1%) than EB (1.1%; Z = 3.91, p < 0.0001; Figure 3.13). While there was no correlation with rugosity, cover was significantly different between relief strata (Z = -2.587, p = 0.0097), with *S. intersepta* reported more frequently and in higher cover on high relief sites (Figure 3.14).

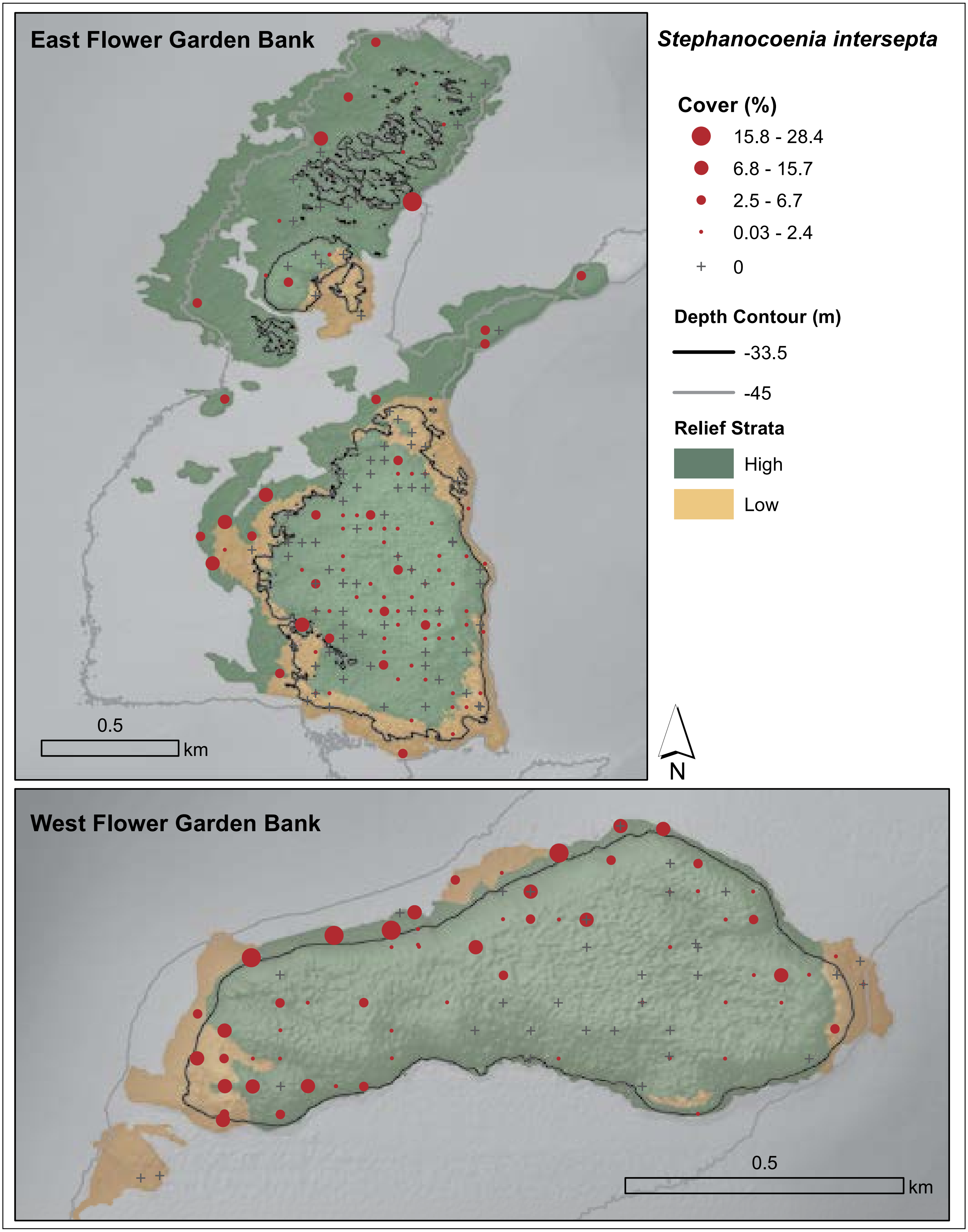

Figure 3.22. Observed mean Stephanocoenia intersepta *(blushing star coral) cover recorded during diver surveys from 2010 – 2012.*

Brain Corals (*Pseudodiploria* Species, *Colpophyllia natans*)

Corals of the *Pseudodiploria* genus and *Colpophyllia natans* are within a group of stony corals commonly called 'Brain Corals' (Figure 3.23). They have a boulder-shaped morphology at shallow depths and a flattened or "plating" morphology that becomes more common with depth. Overall, the *Pseudodiploria* genus, previously known as *Diploria*, ranked second in the percent of total coral cover (11%), and *C. natans* ranked fourth (8%; Figure 3.7). Two *Pseudodiploria* species were recorded in this study: *P. strigosa* and *P. labyrinthiformes*. One instance of *P. labyrinthiformes* was reported at a UM site, but photographic support was lacking for it to be officially recognized as a new species reported within FGBNMS.

Figure 3.23. Brain coral colonies represented in the FGBNMS: Pseudodiploria strigosa *(left) and* Colpophyllia natans *(right). Photos: J. Voss (HBOI-FAU and NOAA/CIOERT) and C.A. Buckel (NOAA NOS/NCCOS/CCFHR)*

Pseudodiploria strigosa (symmetrical brain coral)

Pseudodiploria strigosa was recorded at 196 of 291 surveyed sites and was the second most dominant coral species, in terms of coral cover (Figure 3.24). *P. strigosa* cover was higher within the high relief strata (4.6%) compared to low (1.8%, $Z = -4.104$, $p < 0.0001$), and there was no difference in cover between banks (EB 5.7%, WB 5.6%). No inter-annual differences within depth strata were identified for *P. strigosa* (Figure 3.26). Although there was less *P. strigosa* at deeper depths (Figure 3.27), moderate cover did occur at some sites (max cover: 16% UM).

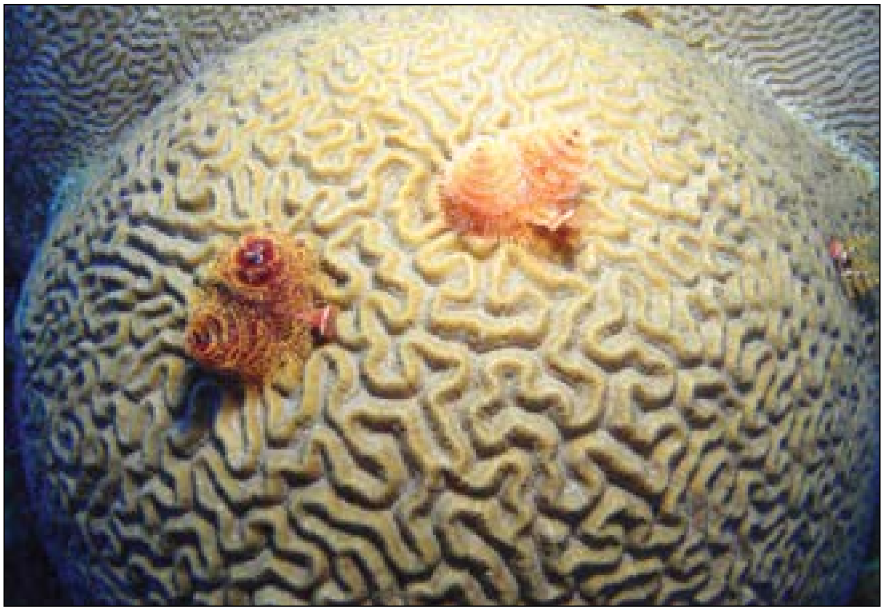

Figure 3.24. Image of Pseudodiploria strigosa *in FGBNMS. Photo: J. Voss (HBOI-FAU and NOAA/CIOERT)*

Colpophyllia natans (boulder brain coral)

Recorded at 145 of 291 sites, *C. natans* (Figure 3.25) predominantly occurred within the shallow depth strata (n sites = 140 shallow, 5 UM; Figure 3.28). *Colpophyllia natans* contributed 6.4% to the differences between depth strata (Table 3.6) and was negatively correlated with depth ($\rho = -0.3633$, $p < 0.0001$). Cover was significantly greater on EB (5.6%) than WB (1.9%; $Z = -3.546$, $p = 0.0004$) with all UM occurrences on EB within the high relief stratum (Figure 3.26). Rugosity was positively correlated with *C. natans* cover, indicating higher cover at higher relief sites ($\rho = 0.2924$, $p < 0.0001$). No inter-annual differences were identified by depth strata.

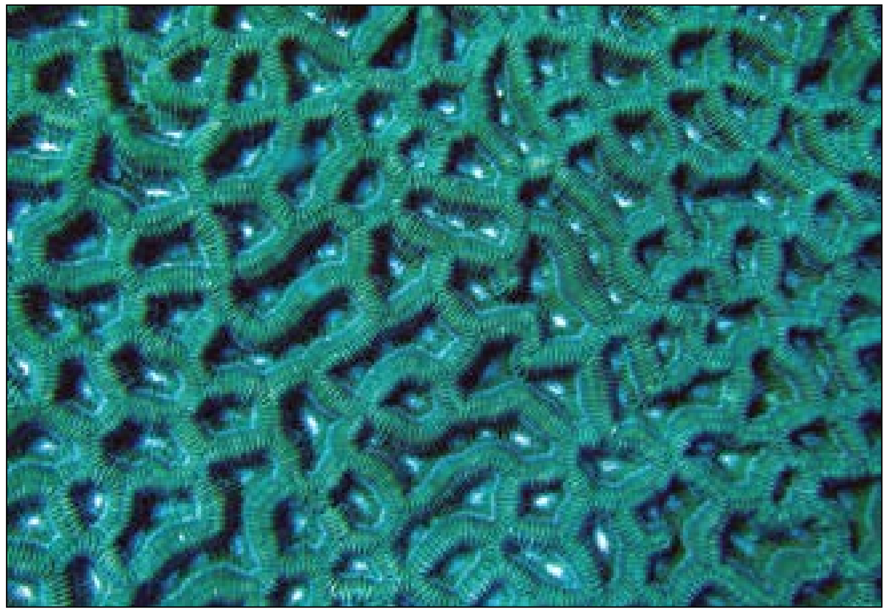

Figure 3.25. Image of Colpophyllia natans *in FGBNMS. Photo: NOAA NOS/NCCOS/CCMA*

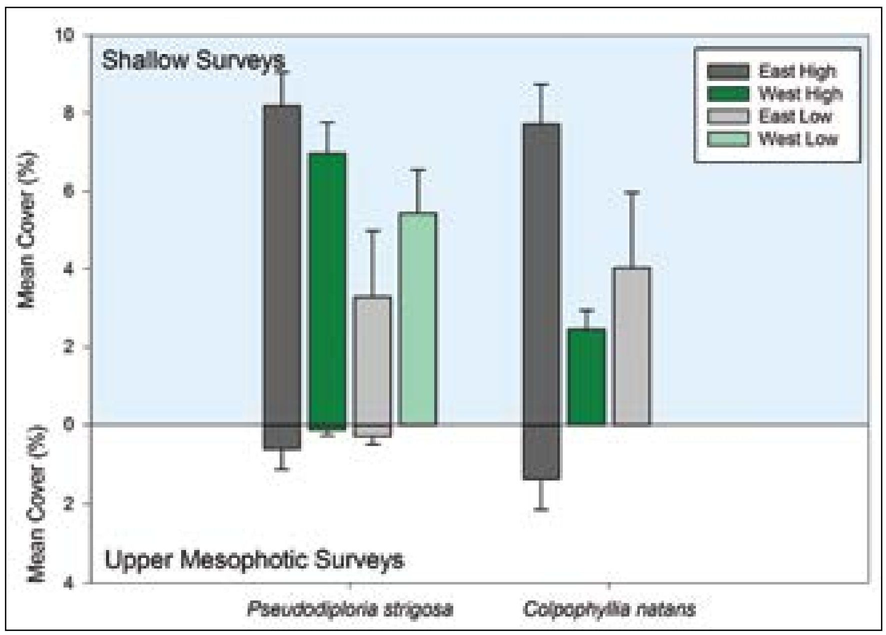

Figure 3.26. Mean percent cover (SE) of Diploria strigosa *and* Colpophyllia natans *by strata from diving surveys (2010 – 2012).*

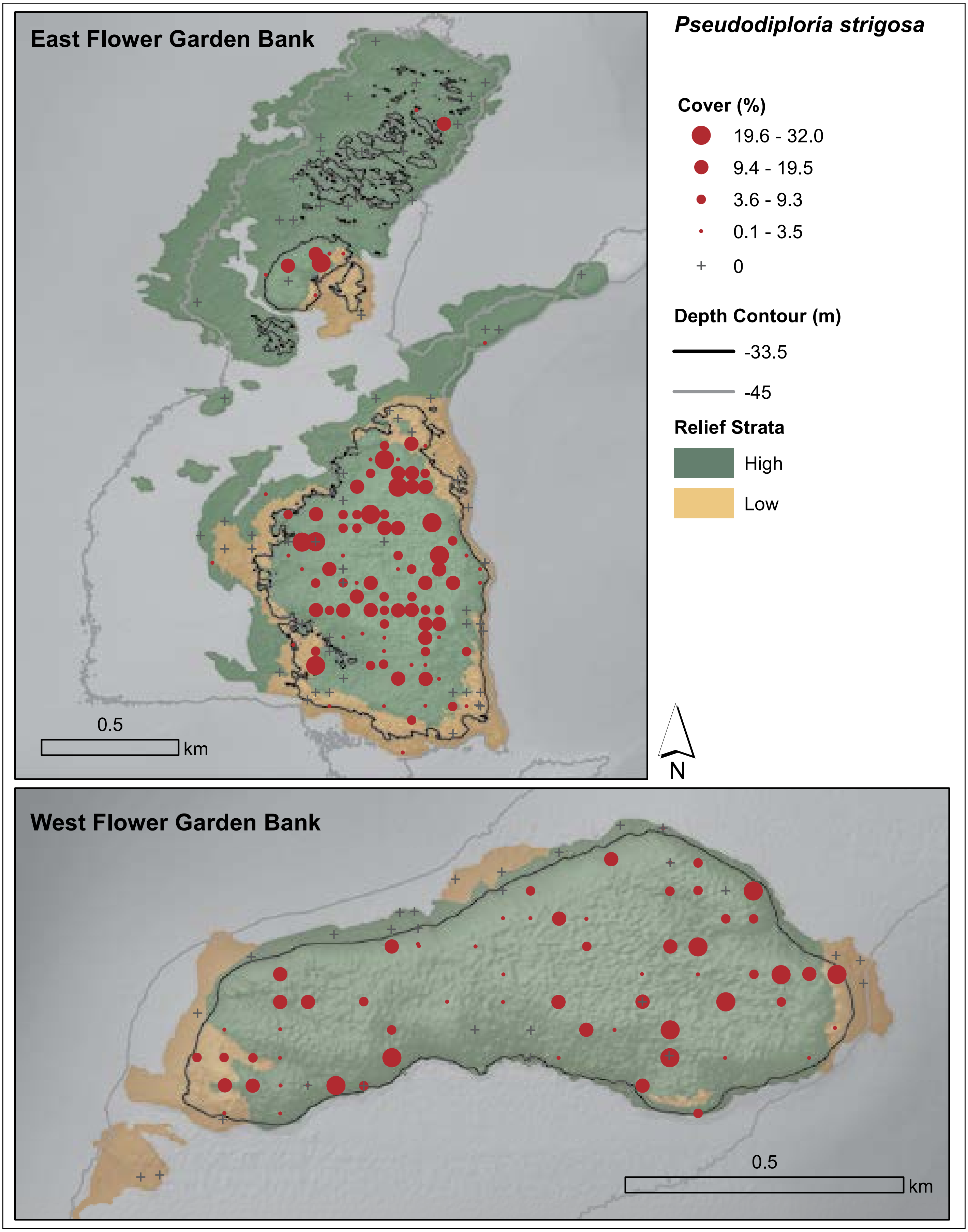

Figure 3.27. Observed mean Pseudodiploria strigosa *(symmetrical brain coral) percent cover recorded during dive surveys from 2010 – 2012.*

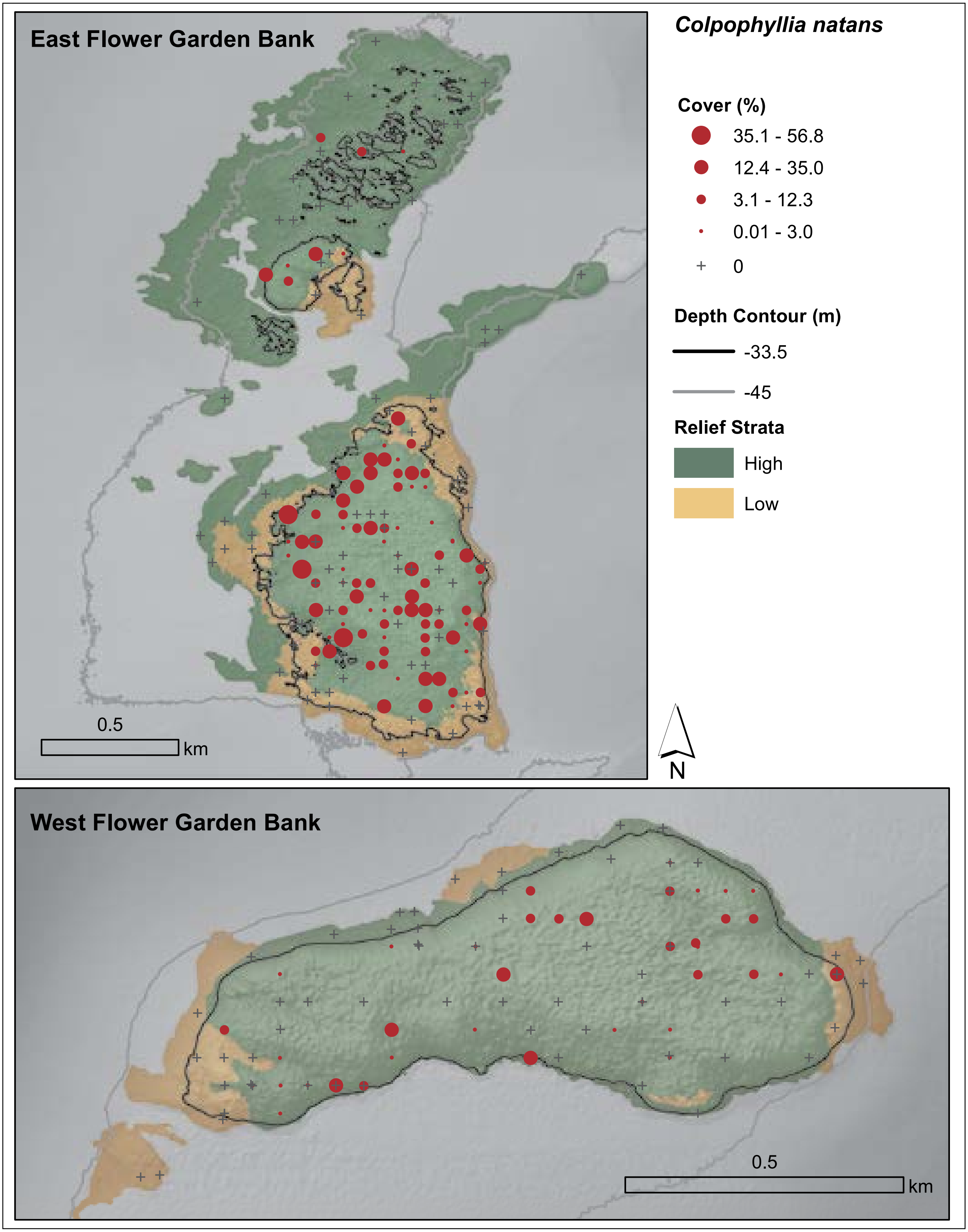

Figure 3.28. Observed mean Colpophyllia natans *(boulder brain coral) percent cover recorded during diver surveys from 2010 – 2012.*

<u>*Madracis* Species</u>

Three *Madracis* species were recorded during these surveys: *M. auretenra*, *M. decactis* and *M. pharensis* (Figure 3.29). *Madracis pharensis* was reported at one site but due to the lack of photographic documentation it was not included as an official coral species of FGBNMS and was analyzed here as *Madracis* spp.; it was not reported in Caldow et al. (2009). At two shallow sites, some *Madracis* colonies present could not be identified to species and were classified as *Madracis* spp. Mean cover of *Madracis* spp. was low (0.05%) and similar to previous reports (0.04%; Caldow et al., 2009). *Madracis decactis* (ten-ray star coral) was reported on both banks and at high and low relief sites within the shallow strata; it was only found at high relief sites within the UM strata (Figure 3.30). Cover of *M. decactis* reported here (0.4%) was similar to that found by Caldow et al. (0.3%, 2009), with higher cover within shallow strata (0.5%) than UM (0.1%; Z = -4.767, p < 0.0001).

Figure 3.29. Examples of two Madracis *species found in the FGBNMS:* Madracis auretentra *(left) and* Madracis dectatis *(right). Photos: Doug Kessling (UNCW) and C.A. Buckel (NOAA NOS/NCCOS/CCFHR).*

Madracis auretenra (Yellow pencil coral)

Madracis auretenra (Figure 3.29), previously identified as *Madracis mirabilis* (Locke et al., 2007), was recorded at 27 sites, primarily on EB, with only four occurrences on WB; cover was significantly greater on EB than WB (Figure 3.30 ; Z = -3.59, p = 0.0003). Although *M. auretenra* cover was patchy (mean cover 1.8 ± 0.5%), where it did occur it could be the dominant species at a site (max cover of 64%), which is consistent with previous observations of this species (max cover 67%; Caldow et al., 2009; Figure 3.31). *M. auretenera* cover contributed 3.4% to the significant differences recorded between depth strata with multivariate analyses, which showed higher percent cover in the UM (Table 3.6). Consistent with multivariate analyses, univariate analyses identified cover was positively correlated with depth (ρ = 0.1443, p = 0.0138) and increased in lower rugosity habitats (ρ = -0.246, p = 0.0001; Figure 3.32). *M. auretenra* cover was highest in the low relief stratum (7.4 ± 2.3%, 0.8 ± 0.4% high relief) and was the second most dominant coral species within the low relief stratum, preceded by *O. franksi* (8.5 ± 2.3%). No inter-annual differences in *M. auretenra* cover were identified for both depth strata.

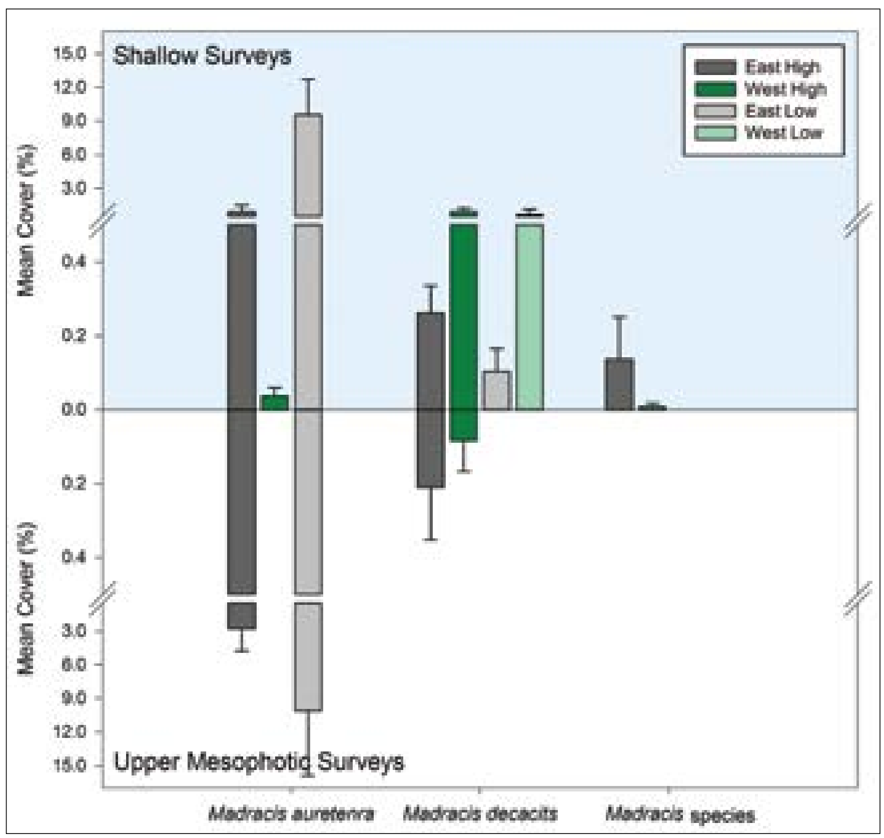

Figure 3.30. Mean percent cover (SE) for Madracis *species from diver surveys (2010 – 2012).*

Figure 3.31. Extensive M. auretenra *cover at a low relief site. Photo: J. Emmert (TAMUG)*

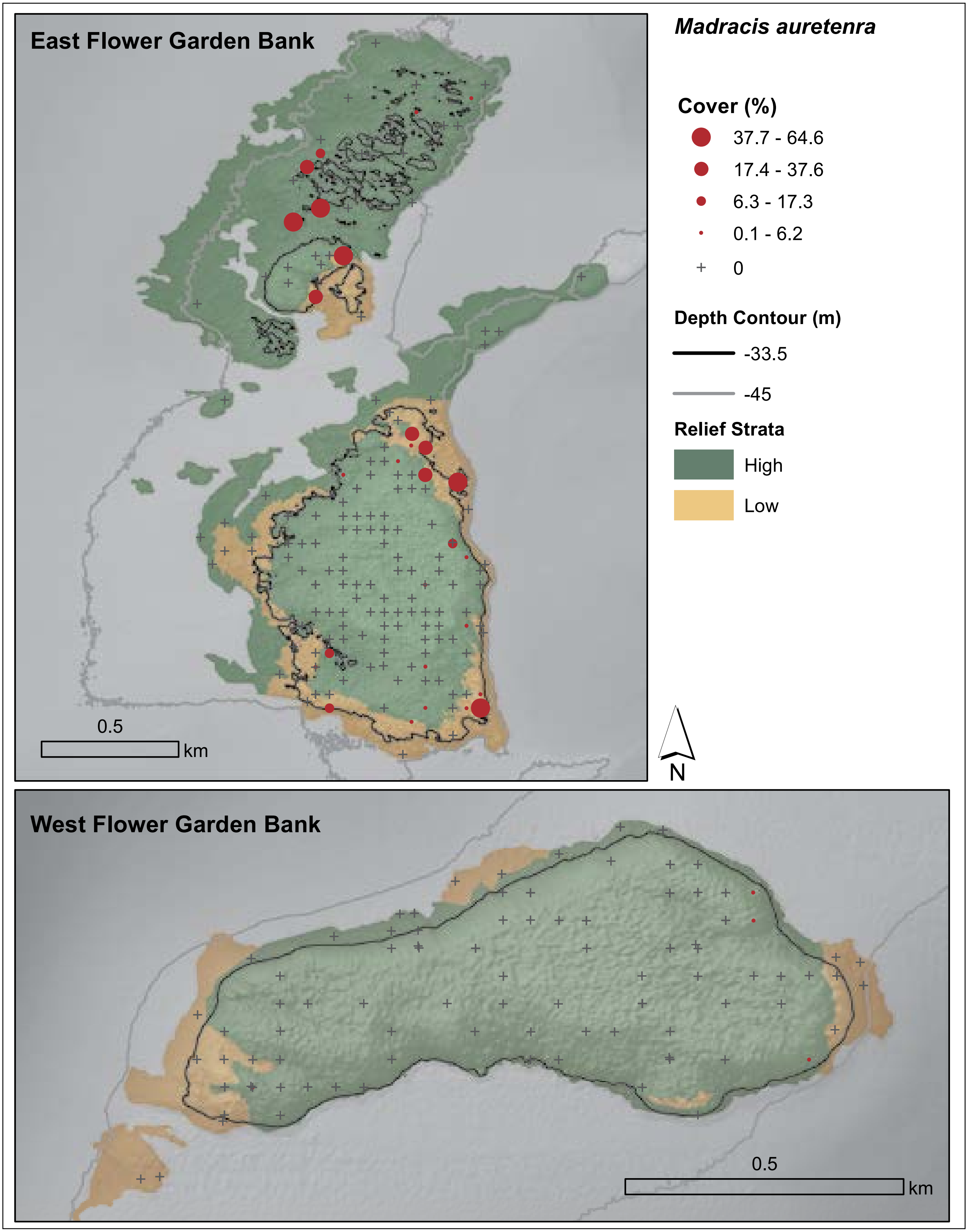

Figure 3.32. Observed mean Madracis auretenra *(yellow pencil coral) percent cover recorded during dive surveys from 2010 – 2012.*

Hydrocorals (*Millepora* Species)

Hydrocorals are commonly mistaken for stony corals (class Anthozoa) because they have a hard calcareous skeleton, but belong to an entirely different class of organisms (Hydrozoa). Two hydrocoral species were recorded during diver surveys: *Millepora complanata* and *Millepora alcicornis* (Figure 3.33). Hydrocorals were reported at 155 of 291 sites (Figure 3.35). *M. alcicornis* was the dominant hydrocoral (Figure 3.34); cover increased with depth (ρ = 0.2384, $p < 0.0001$, consistent with multivariate analyses, Table 3.6) and decreased with rugosity (ρ = -0.1779, $p = 0.0023$). There were no detectable differences in *M. alcicornis* cover between banks and across surveyed years within each depth strata. Some *Millepora* colonies at 16 shallow sites could not be identified beyond genus and were recorded as *Millepora* species (percent cover range 0.1-5.7%; overall mean ± SE: 0.09 ± 0.03%). *M. complanata* was reported only within shallow survey depths at nine sites, with a maximum site coverage of 5%. This species is not on the FGBNMS coral species list and photo documentation to confirm its presence was lacking; thus these data were included in the *Millepora* species group for analyses.

Figure 3.33. Image of hydrocoral Millepora alcicornis *(branching fire coral) in FGBNMS. Photo: C.A. Buckel (NOAA NOS/NCCOS/CCFHR)*

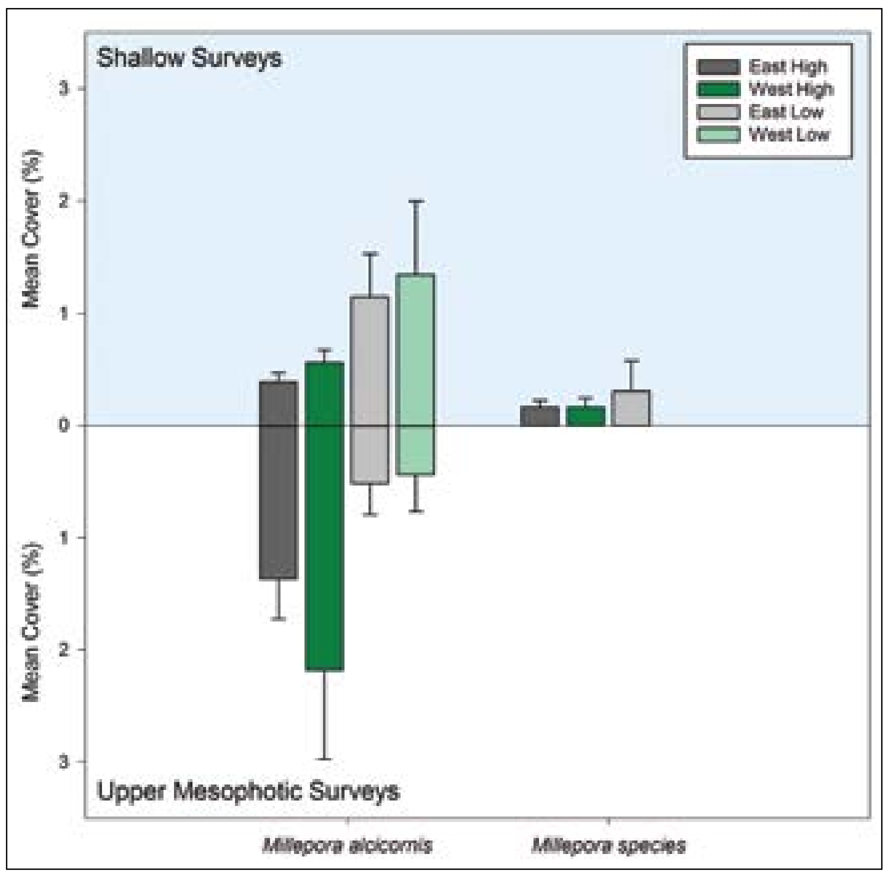

Figure 3.34. Mean percent cover (SE) for Millepora alcicornis *and* Millepora *species by strata from diver surveys (2010 – 2012).*

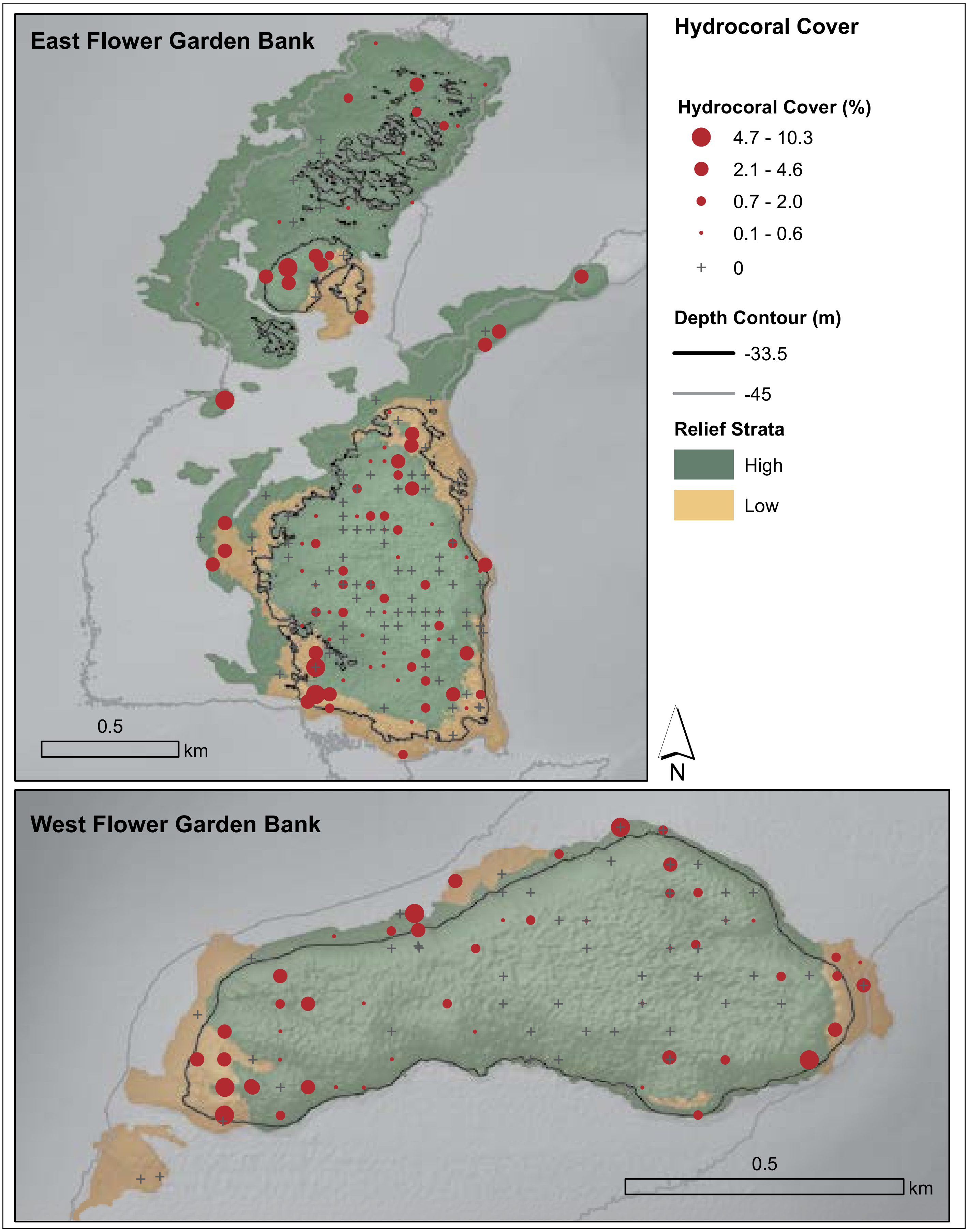

*Figure 3.35. Observed mean hydrocoral (*Millepora *spp.; firecoral) cover (%) recorded during dive surveys from 2010 – 2012.*

Coral Bleaching and Disease

Coral reefs of the FGBNMS showed resiliency to bleaching mortality following a September 2005 mass bleaching event, which impacted 45% of the coral community (Precht et al., 2008; ONMS, 2008; Robbart et al., 2008). By March 2006, bleached coral (Figure 3.36) cover had diminished to 4% (Hickerson et al., 2008), although a reduction in mean annual coral growth rates was noted (Zimmer et al., 2010). An Office of National Marine Sanctuary (ONMS) 2008 report hypothesized the deeper depths of the Flower Garden Banks provide refuge from bleaching, as the impact diminished beyond 29 m depth (ONMS, 2008). However, with increasing water temperature due to climate change, these presently deeper, offshore reefs may experience higher rates of bleaching.

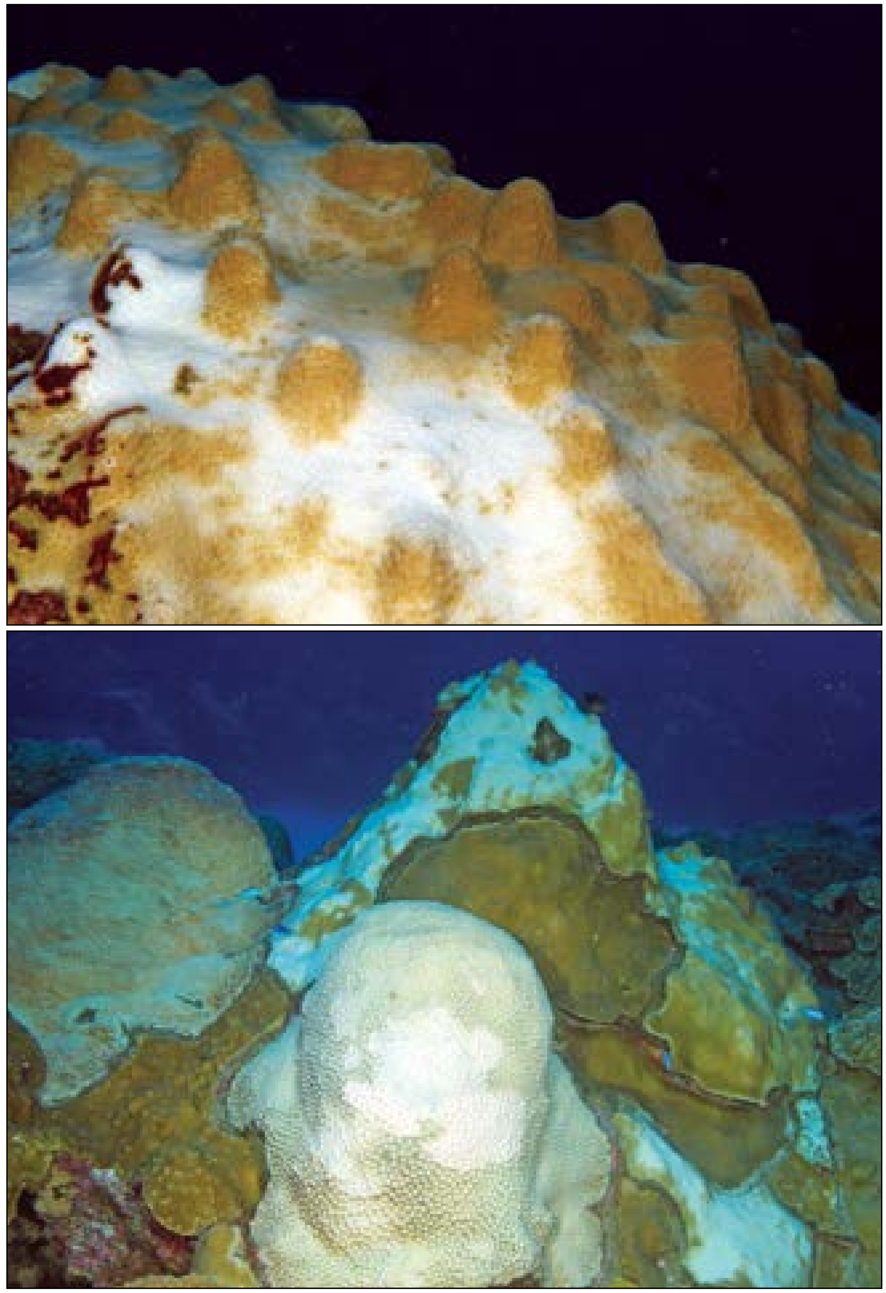

Figure 3.36. Examples of bleached coral colonies: O. faveolata *(top) and* M. cavernosa, O. faveolata *and* O. franksi *(bottom). Photos: Joyce and Frank Burek*

Bleaching at some level was reported at 131 of 291 sites surveyed in this study (Figure 3.37), with a site maximum of 38.5% bleached coral occurring at a high relief shallow site. Maximum bleaching impact was estimated in this study; where any portion of a coral colony was bleached, the entire colony was recorded as bleached. Overall, bleaching across the coral reef was low (Table 3.7; mean: 1.62 ±0.2%) and not significantly different between banks (WB: 1.6 ± 0.4%; EB 1.7 ± 0.4%), relief strata (high: 1.8 ± 0.3%, low: 1.0 ± 0.4%), and depth strata (shallow: 1.7 ± 0.3%, UM: 1.5 ± 0.5%).

Table 3.7. Mean percent bleached and diseased cover (±SE) among strata.

Bank	Relief	N sites	N sites with bleaching	N sites with disease	Diseased cover	Bleached cover
East Bank	Low	34	1	3	0.05 (±0.03)	0.8 (±0.3)
	High	145	17	27	0.2 (±0.07)	1.9 (±0.4)
West Bank	Low	11	0	0	0	1.5 (±1.3)
	High	101	18	32	0.5 (±0.2)	1.6 (±0.4)
Coral Reef		291	36	62	0.3 (±0.09)	1.6 (±0.3)

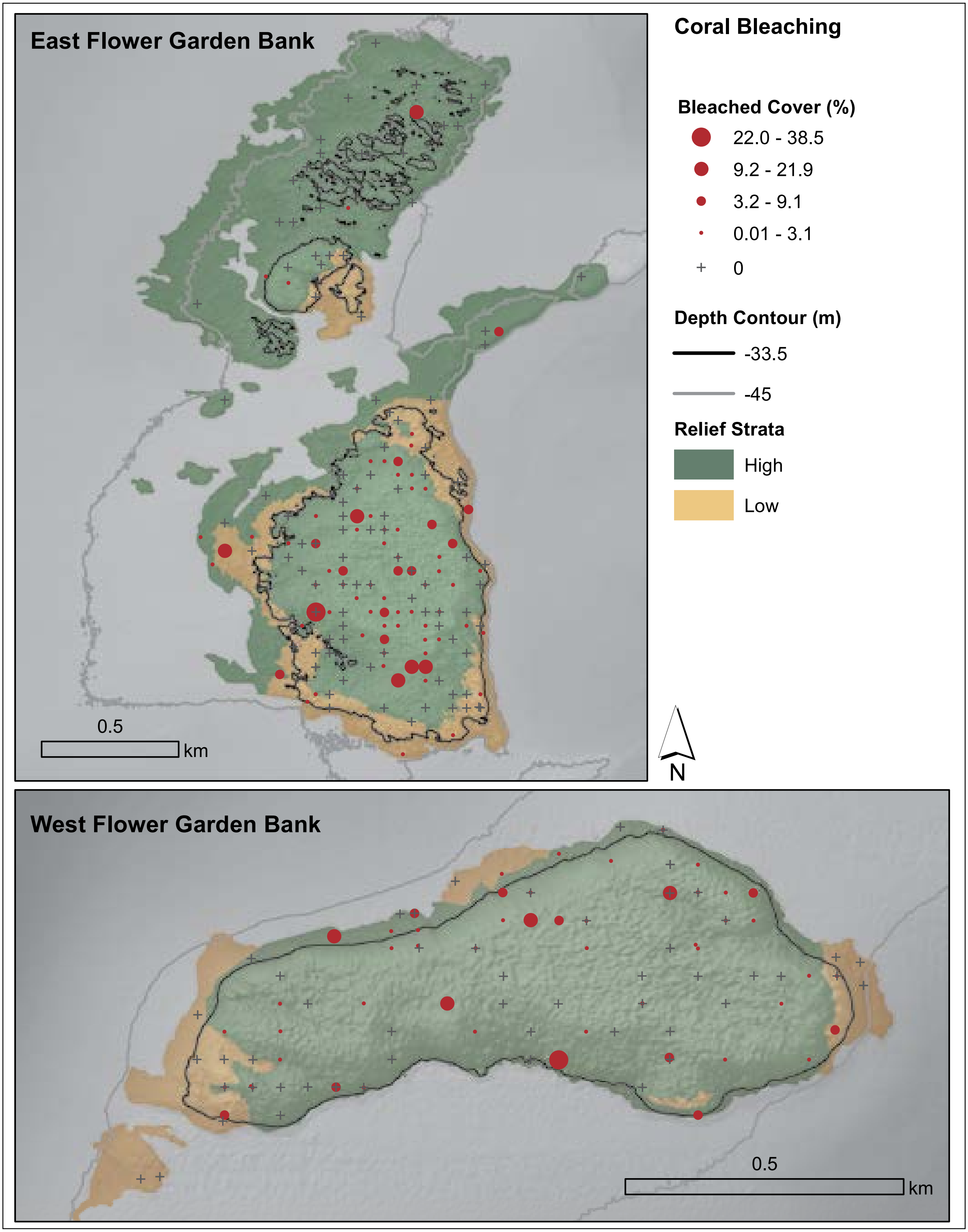

Figure 3.37. Observed mean bleached coral (scleractinian corals and Millepora *spp.) percent cover recorded during dive surveys from 2010 – 2012.*

Bleaching was not uniform among species. *O. annularis* showed the highest bleached cover (Figure 3.38), despite its relative low percent cover (mean site cover: 0.6%). *O. franksi* was the second most frequently bleached coral, followed by *O. faveolata*. It has been identified that the ribotype of phototrophic dinoflagellates (*Symbiodinium* species) can influence a colony's susceptibility to bleaching (Rowan et al., 1997; Douglas, 2003); whether these ribotypes occur at FGBNMS is presently unknown. *Millepora*, all species combined, also ranked high in percent bleached cover, despite the low mean cover of this genus across the banks (0.8%), supporting the evidence of thermal susceptibility in this genus at the banks (Hagman and Gittings, 1992).

Surveys were typically conducted in the late summer to early fall of 2010-2012 (August-October), coinciding with higher annual sea surface temperatures (Tester et al. 2013). Yet, bleaching incidences were lower than historical reports of average annual bleaching (Hagman and Gittings, 1992; Hickerson et al., 2008; Caldow et al., 2009), suggesting low impact or high recovery of resident corals. Bleached cover reported by Caldow et al. (2009) was higher (18%) because correction factors were used to standardize by total coral cover. Data presented here do not use a correction factor and represent a maximum percent bleaching impact based on survey methodology (see Methods).

Overall, mean diseased coral cover was low on the banks (0.3 ± 0.09%). While slightly higher on WB, this difference was not significant (Table 3.7). Diseased coral percent cover was not significantly correlated with depth or rugosity, likely due to low occurrences. Targeted monitoring for coral bleaching and disease is needed, especially across the full depth range of the coral reef (up to 45 m).

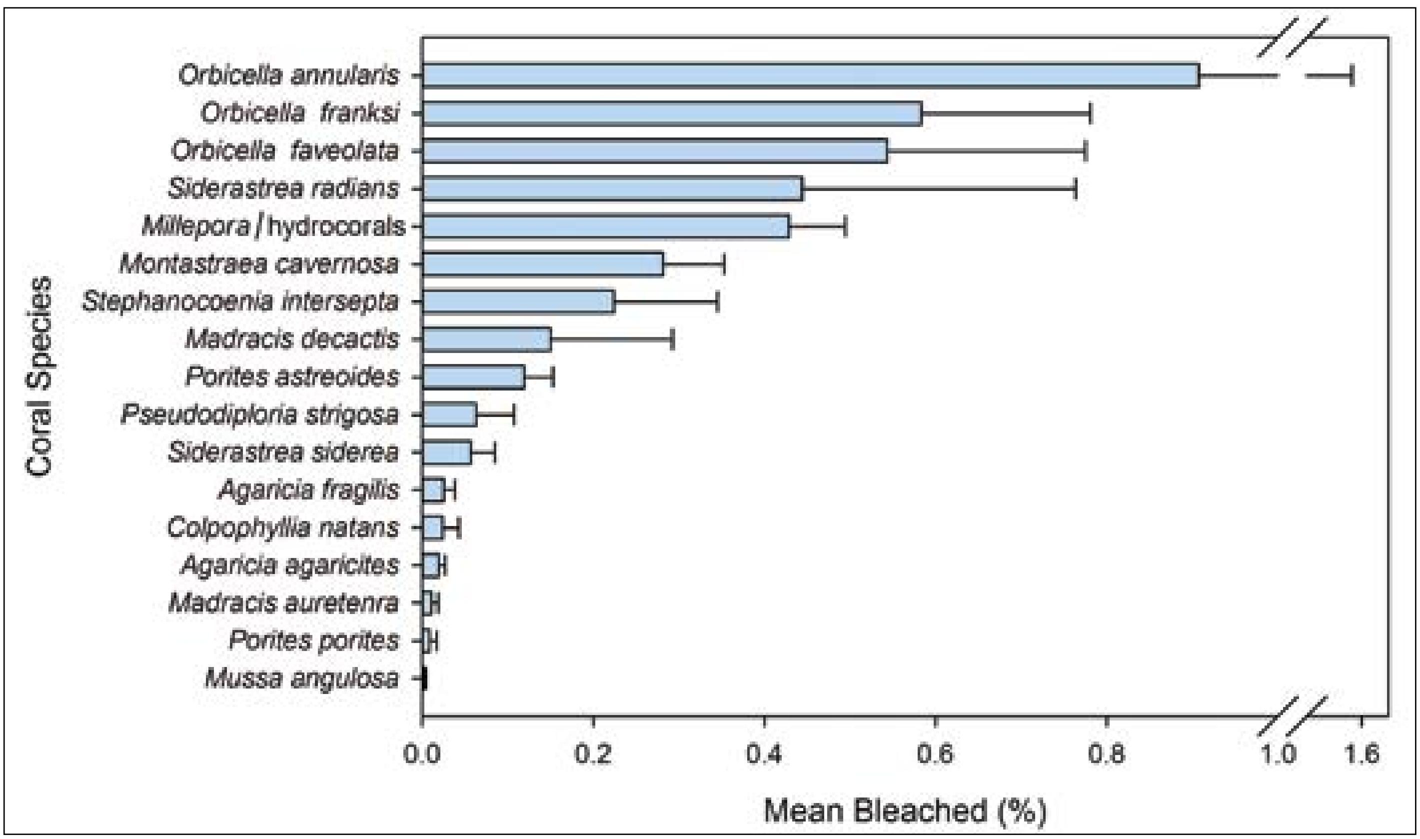

Figure 3.38. Mean (SE) bleached coral cover (%) of all impacted species from diver surveys (2010-2012).

3.3.4. Algal Cover

The algal group is comprised of three classes: macroalgae, turf algae, and crustose coralline algae (CCA). Algal cover was ubiquitous across the banks (Figure 3.39), ranging from 2.2-99.8% cover among sites (Figure 3.40; n = 275) and accounted for nearly 40% of the biotic cover of the coral reef (Table 3.8). There was no significant difference in algal cover between banks, but it was significantly higher in the low relief stratum compared to the high ($t = -23.228$, $p < 0.0001$; Table 3.8), and negatively correlated with rugosity ($r = -0.3589$, $p < 0.0001$). Algal cover contributed 15% to the differences identified between depth strata (Table 3.3), with significantly higher algal cover occurring at UM sites compared to shallow ($Z = 21.3275$, $p < 0.0001$), which was also reflected in the positive correlation with depth ($r = 0.3265$, $p < 0.0001$). It was hypothesized that significant differences over time might be found for relatively fast-growing benthic algal species; however, no significant inter-annual differences were recorded for algal cover by depth strata.

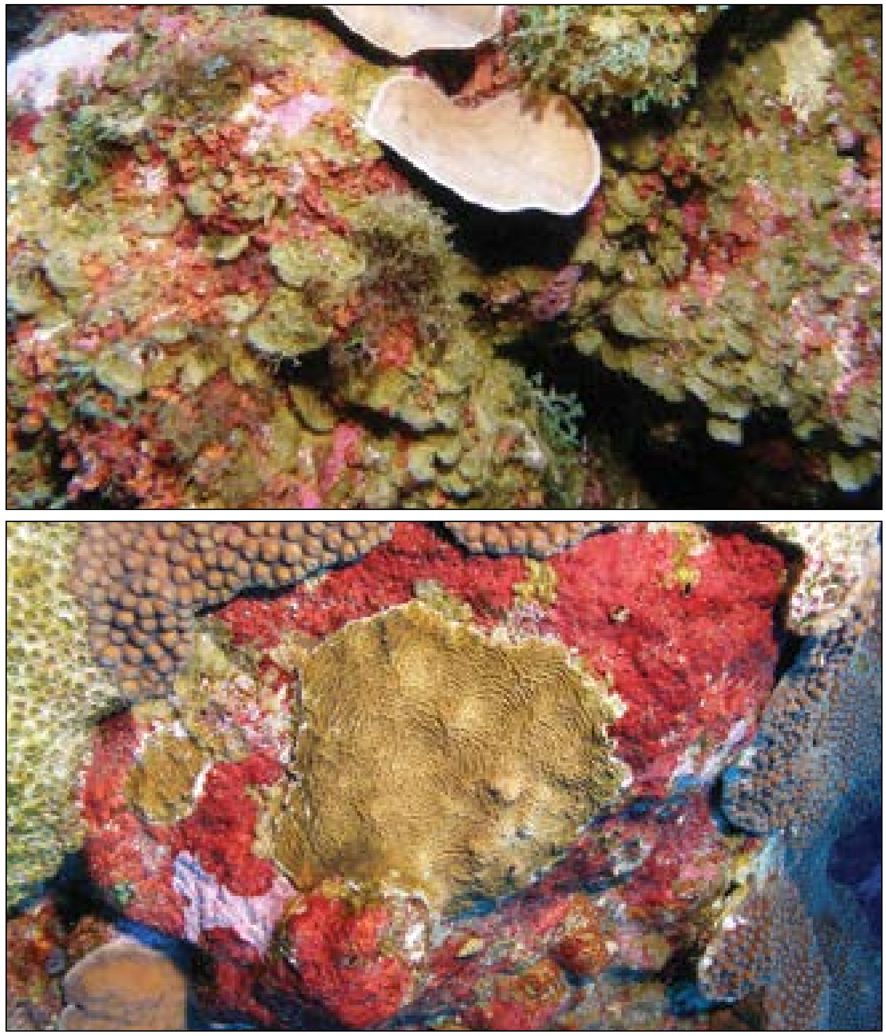

Figure 3.39. Algal cover, including Lobophora *species and* Dictyota *species (top); and crustose coralline algae (including* Peysonnelia *spp.) surrounded by* Montastraea cavernosa *and* Agaricia agaricites *in the middle (bottom) at FGBNMS. Photos: C.A. Buckel (NOAA NOS/NCCOS/CCFHR)*

Table 3.8. Mean percent cover (±SE) of the algal functional groups and the three morphological groups within the functional group: macroalgae, crustose coralline algae (CCA) and turf algae, by strata (N = 275 sites for all algal groups)

Bank	Relief	N surveys	Algae	Macroalgae	CCA	Turf algae
East Bank	Low	32	62.5 (±4.0)	45.6 (±3.9)	9.1 (±1.4)	7.8 (±2.3)
	High	138	35.0 (±1.6)	20.8 (±1.4)	3.7 (±0.4)	10.5 (±1.0)
West Bank	Low	10	76.1 (±4.8)	65.2 (±5.5)	6.6 (±3.0)	4.3 (±1.1)
	High	95	35.7 (±1.9)	22.2 (±1.5)	4.1 (±0.5)	9.5 (±1.0)
Coral Reef		275	39.9 (±1.2)	25.8 (±1.2)	4.6 (±0.3)	9.6 (±0.7)

Macroalgae, defined as algae greater than 1 cm tall, dominated the algal composition of the banks and were reported from all surveys. The highest stratum cover (mean: 65 ± 5.5%) was reported on WB low relief (Table 3.8). Macroalgal cover was not significantly different between banks, but was positively correlated with depth ($r = 0.485$, $p < 0.0001$, consistent with multivariate analyses, Table 3.6) as UM cover was higher than shallow (40.1 ± 2.7% and 20.2 ± 1.1%, respectively; $t = 24.8134$, $p = 0.0001$). Due to the increase with depth, macroalgae was the second highest species (or species group) driving differences observed between depth strata, contributing 10.8% to strata differences (Table 3.6). Macroalgae

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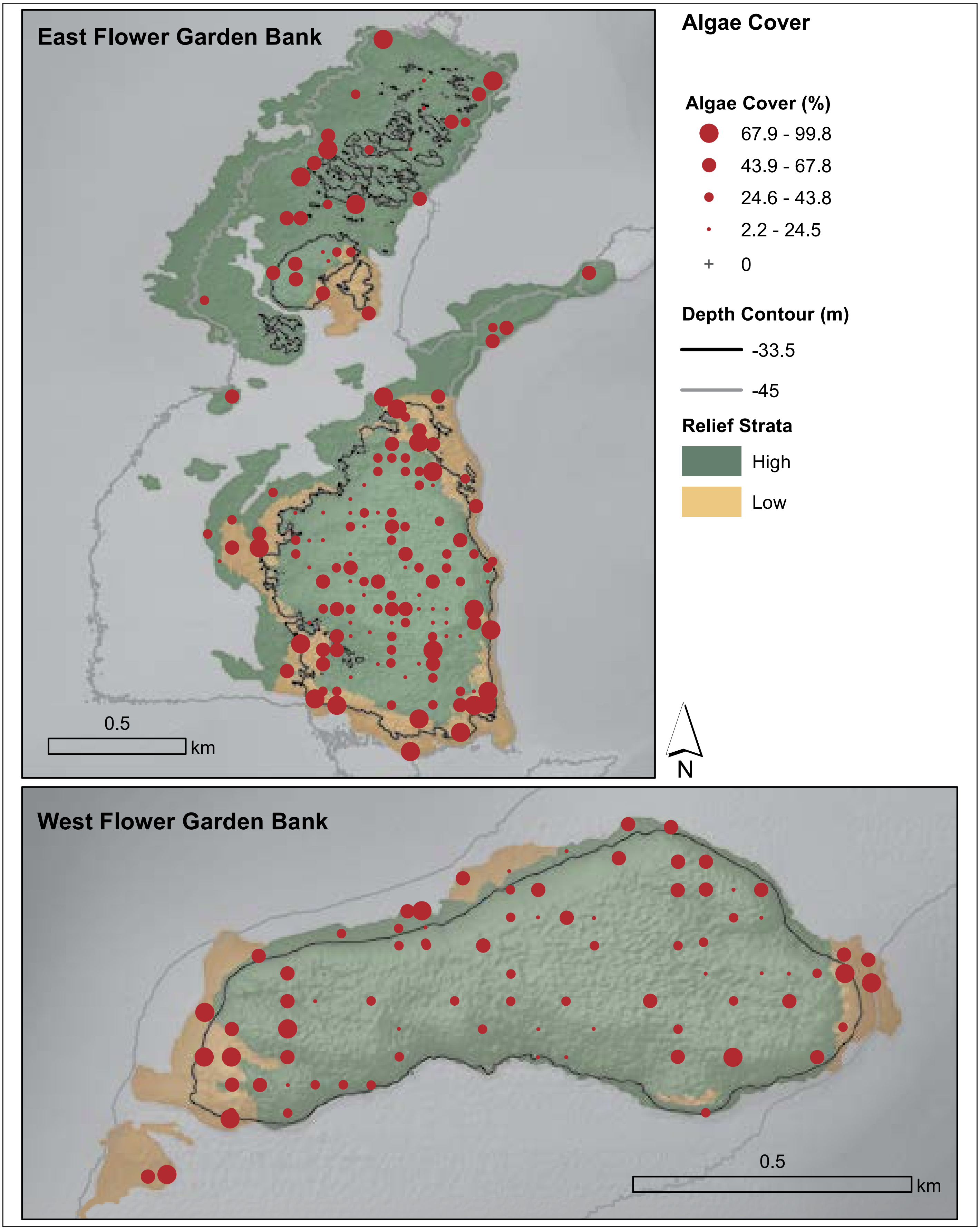

Figure 3.40. Observed (dots) mean algal cover (%, combined macroalgae, turf algae, and crustose coralline algae cover) recorded during dive surveys from 2010 – 2012 (N = 275 sites).

cover was significantly higher in the low relief stratum than high relief (50.2 ± 3.5% and 21.3 ± 1.0%, respectively; Z = -21.608, p = 0.0001), and negatively correlated with rugosity (r = -0.4408, p < 0.0001). While species specific information was not recorded during benthic surveys, review of *in situ* photos identified two dominant algae genera: *Lobophora* spp. and *Dictyota* spp. (Figure 3.39). No significant inter-annual differences in macroalgae cover were recorded.

Turf algae, defined as algae less than 1 cm tall, cover was reported at 255 of the 275 sites. There was no significant difference in turf algae between banks (Table 3.8), but cover differed significantly by relief (Z = -2.70089, p = 0.0069) and depth strata (Z = -4.02949, p < 0.0001), with higher cover on high relief than low and in the shallow strata versus the UM. Multivariate analyses also showed higher cover in shallow compared to UM, and turf algae contributed 8.3% to the differences observed between depth strata (Table 3.6). Turf algal cover was different across all three study years within the shallow stratum, with the highest cover recorded in the first year of the study (2010). Annual differences in turf cover may be influenced by the grazing pressure from the high herbivore densities within the shallow stratum. Timing of shallow surveys were generally within the same season (late summer-early fall) for all survey years, but seasonal differences in cover cannot be excluded as a potential causal factor of observed shallow cover differences. Similar to turf algae, CCA cover was uniform between banks and significantly different by relief (Z = 4.51158, p < 0.0001) and depth strata (Z = 3.02403, p = 0.0025; Table 3.6). CCA cover was higher in low relief habitat than high and at deeper depths. There were no inter-annual differences in CCA cover within depth strata.

Macroalgae cover results from this study contradict Zimmer et al. (2010), who reported macroalgae as typically less abundant on the banks compared to CCA, turf algae or bare rock. This may be attributed to methodology differences, spatial differences based on scale of sampling (i.e., different study areas), large scale removal by hurricanes, or to seasonality. Zimmer et al. (2010) conducted video surveys across both summer and fall seasons (June-November) over four years (2004-2008) within a limited spatial area (100 m x 100 m) and depth range (maximum depth 26 m). Hurricanes can create wave energies strong enough to remove algae from the substratum (Precht et al., 2008). Hurricane Ike was the most recent hurricane to pass over the Sanctuary (2008); post-hurricane surveys documented extensive coral damage but did not address algal community changes (http://flowergarden.noaa.gov/science/ike2008.html; Hickerson, 2008; Johnston et al., 2013). No major hurricanes affected FGBNMS during the time of this study (2010-2012).

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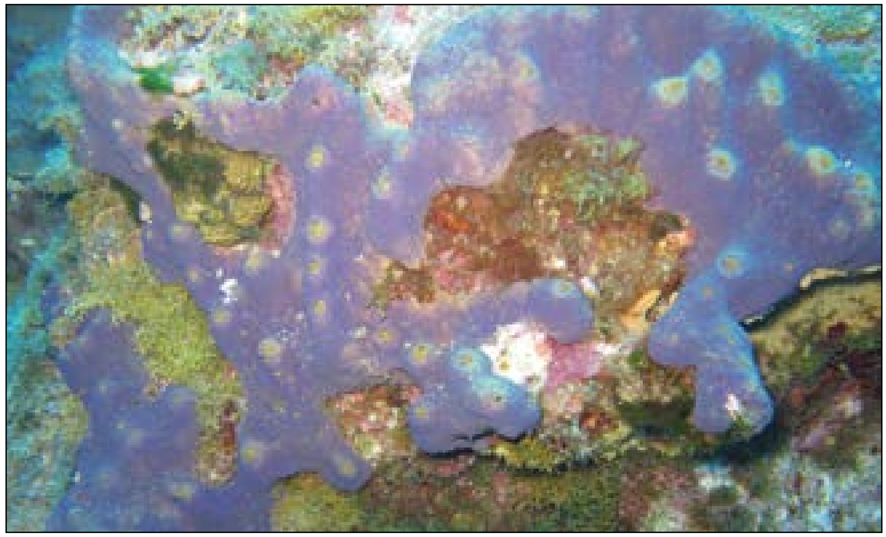
3.3.5. Sponge (Porifera) and Octocoral Cover

The sponge (Porifera) group was made up of two different morphologies: upright (i.e., barrel, tube, vase form or b/t/v) and encrusting. Porifera (Figure 3.41) were reported at 212 of the 291 sites, with the highest reported cover (21.5%) at an UM site. Porifera cover was significantly higher on WB compared to EB (Table 3.9; Figure 3.42; $Z = 2.739$, $p = 0.0062$), and in the low versus high relief stratum ($Z = 4.70260$, $p < 0.0001$), supported by a negative correlation with rugosity ($\tau = -0.1182$, $p = 0.0037$). The correlation between Porifera cover and depth was not significant ($\tau = 0.0713$, $p = 0.0804$), although an increase with depth is suggested and consistent with multivariate analyses (Table 3.3). According to Rezak et al. (1985), sponge cover was greater in the lower mesophotic depths (>46 m), which exceed the diving limits of this portion of the study.

Porifera cover was comprised of similar fractions of b/t/v and encrusting morphologies (mean: 0.52%, 0.53%; respectively). Depth strata differences in cover remained non-significant for both morphologies. Significantly higher cover of b/t/v and encrusting Porifera were recorded within the high relief strata (1% each) compared to low (0.4% each; $Z = 4.304$, $p < 0.0001$; $Z = 2.8$, $p = 0.0048$, respectively). Encrusting Porifera cover was significantly different between WB (0.7%) and EB (0.4%). Upright (b/t/v) sponge cover was greater on WB (0.7%) than EB (0.4%; $Z = 3.07$, $p = 0.0021$; Table 3.9).



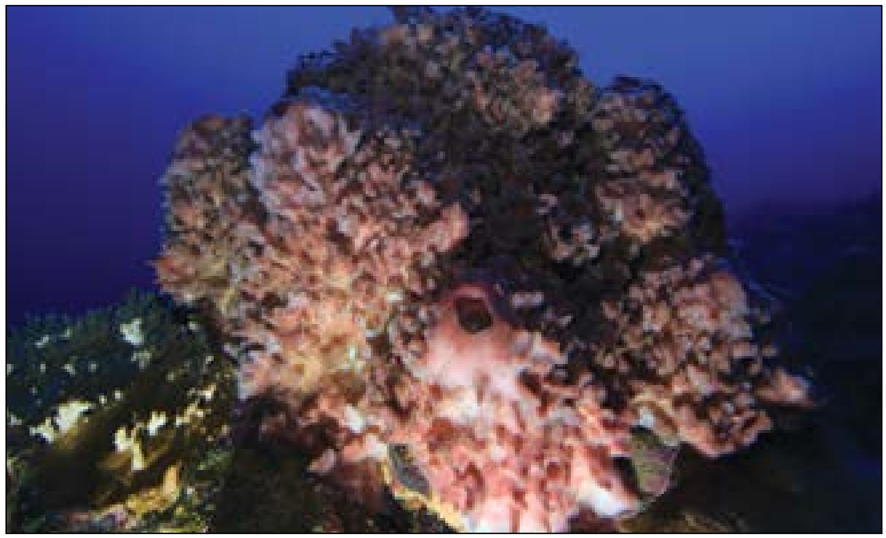


Figure 3.41. Encrusting sponge (top) and tube sponge (bottom). Photos: J. Voss (HBIO-FAU and NOAA/CIOERT) and G. McFall (NOAA NOS/ONMS/GRNMS)

Table 3.9. Mean percent cover by bank and relief strata (±SE) of Porifera, and individual morphologies of barrel/tube/vase sponges (Porifera btv.) and encrusting (Porifera enc.).

Bank	Relief	N Sites	Sponges	Porifera BTV	Porifera ENC
East Bank	LOW	34	1.9 (±0.3)	1.2 (±0.3)	0.7 (±0.2)
	HIGH	145	0.6 (±0.1)	0.2 (±0.1)	0.3 (±0.1)
West Bank	LOW	11	3.2 (±1.9)	0.5 (±0.2)	2.7 (±1.9)
	HIGH	101	1.2 (±0.2)	0.7 (±0.2)	0.5 (±0.1)
Coral Reef		291	1.1 (±0.1)	0.5 (±0.1	0.5 (±0.1)

Octocorals were sighted at two of 291 sites, at 0.3% and 0.8% cover, respectively. Both sites were within the shallow stratum. Octocorals occurrences were rare and in low cover at depths <45 m on the banks, and therefore are not addressed further in this section, but are addressed in the lower mesophotic section (see Chapter 5) where they were more abundant.

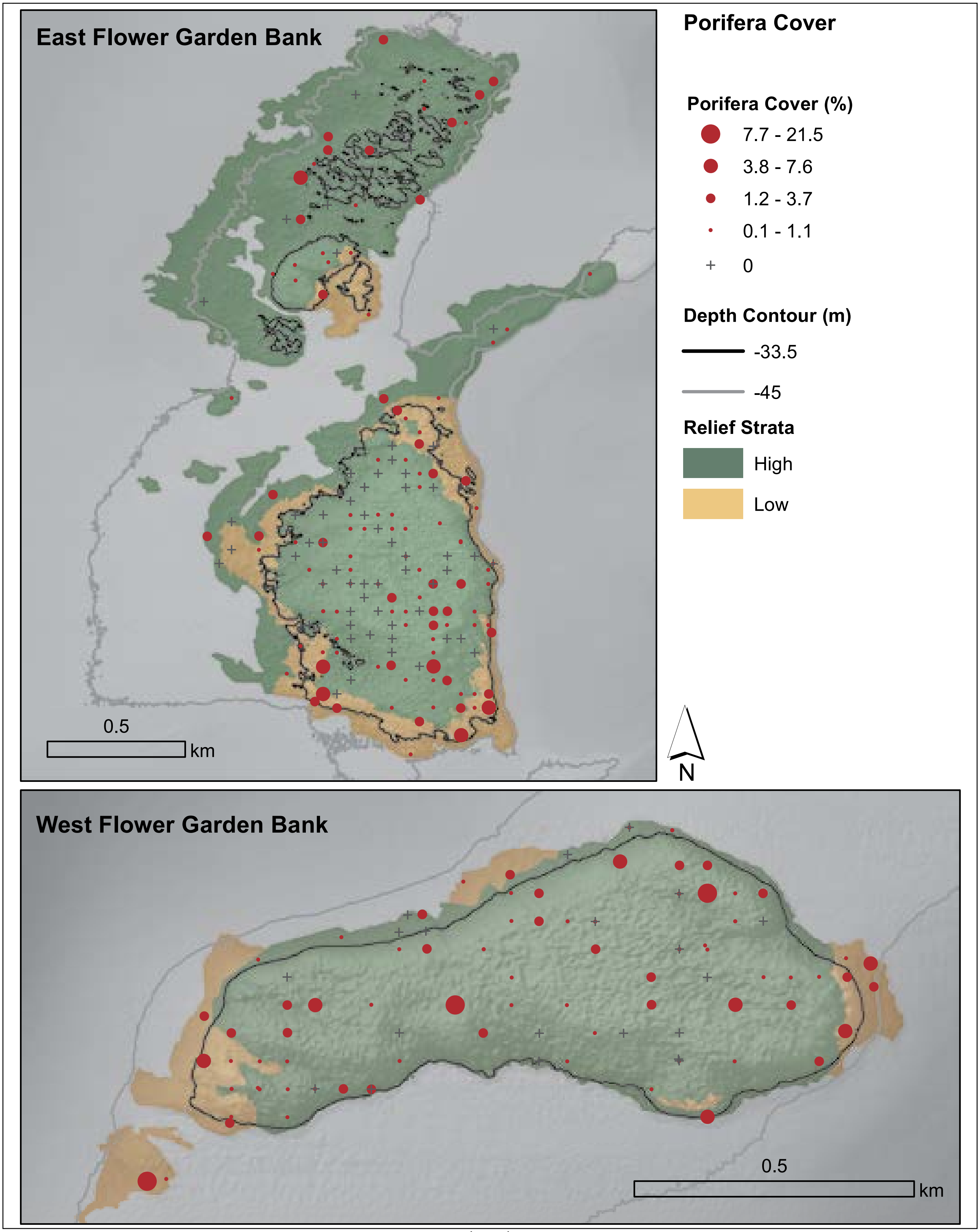


Figure 3.42. Observed (dots) mean Porifera (sponge, combined barrel/tube/vase and encrusting) cover (%) recorded during dive surveys from 2010 – 2012.

3.3.6. Macroinvertebrates

All macroinvertebrate reports during this study were sighted within the shallow strata. A total of 17 *Diadema antillarum* (long-spined sea urchin) and three *Panulirus argus* (Caribbean spiny lobster) were reported in 225 shallow surveys (0.0007 *D. antillarum*/m^2; 0.0001 *P. argus*/m^2; Figure 3.43); an increase over Caldow et al. (2009), who reported only one of each *D. antillarum* and *P. argus* in 95 surveys (0.0001 individuals/m^2) within the same depth range. Surveys of the LTM site 10 years earlier (2002-2003) reported slightly higher macroinvertebrate densities during daytime surveys, with seven *D. antillarum* individuals (0.012/m^2) and one *P. argus* (0.0018/m^2), (Precht et al., 2006). The most recent LTM survey reported *D. antillarum* densities of 0.005/m^2 on EB, and 0.11/m^2 on WB in 2010 (Johnston et al., 2013). Due to the low sample size of macroinvertebrates, no statistical analyses were carried out. While earlier studies reported *P. argus* as a rare species within FGBNMS (Pequenet and Ray, 1974; Dokken et al., 2003; Precht et al., 2006; Precht et al., 2008; Johnston et al., 2013), historic *D. antillarum* densities were higher (range: 0.54-1.63/m^2 prior to 1983; Gittings and Bright, 1987).

Figure 3.43. Diadema antillarum *(long-spined sea urchin; top) and* Panulirus argus *(Caribbean spiny lobster; bottom) at FGBNMS. Photos: A. Uhrin (NOAA NOS/NCCOS/CCFHR)*

In 1983, *D. antillarum* experienced a significant die off event throughout the Caribbean and Gulf of Mexico (Lessios et al., 1988; Pattengill-Semmens et al., 2000). Following the die off, *D. antillarum* densities dropped to 0 individuals/m^2 within FGBNMS (Gittings and Bright, 1987). While numbers still remain low throughout Atlantic coral reefs, studies throughout the Caribbean (Chiappone et al., 2013) and recent FGBNMS studies (Johnston et al., 2013; this study) indicate increasing abundance.

All surveys in this study were conducted during daylight hours, which likely underestimated the abundance of *D. antillarum*, as these organisms hide in crevices during the day and forage on the reef at night. Many individuals in a range of sizes were observed during a 2012 night dive in the shallow strata (C.A. Buckel, pers. comm.). LTM survey protocols for urchins and lobsters includes a night time survey, however, the spatial extent of these surveys is limited. Extending these protocols to a larger area and depth range should be considered if macroinvertebrates are identified as a management concern within the sanctuary.

3.3.7. Marine Debris

The occurrence of marine debris on coral reefs is becoming widespread, and commonly results in entanglement and abrasion of benthic organisms (Chiappone et al., 2005). The FGBNMS offshore location has protected the banks from frequent human interaction, but marine debris is still a concern in the sanctuary, either through direct damage from the impact or potential entanglement of marine life (Figure 3.44).

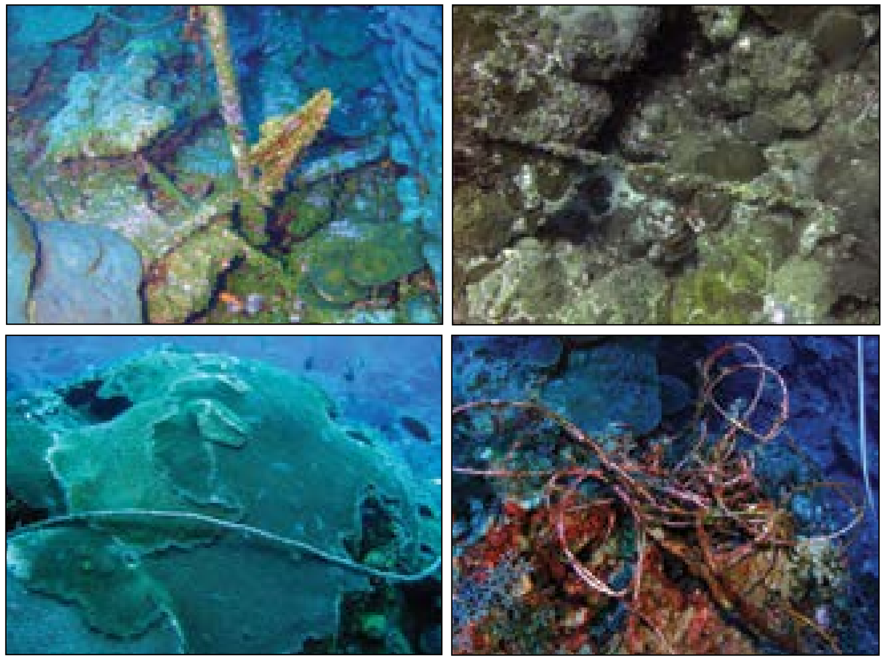


Figure 3.44. Images of marine debris in FGBNMS. Photos: J. Voss (HBOI-FAU; top left), G. McFall (NOAA NOS/ONMS/GRNMS; top right), NOAA/NCCOS CCMA (bottom left) and C.A. Buckel (NOAA/NCCOS CCFHR; bottom right)

A total of 11 marine debris items (Table 3.10) were reported during this study: eight on WB, four on EB. Two of these reports were during UM surveys, with the remaining 10 reports within the shallow surveys (Figure 3.45). Similar to past studies, debris items included: oil and gas exploration materials (i.e., seismic cable), fishing gear (monofilament line, lures), food debris (soda bottles), and boat equipment (i.e., anchors and rope; Gittings et al., 1992; Caldow et al., 2009). Anchoring has been prohibited within the sanctuary since 1992 (NOAA, 2001), and further strengthened in 2001 through the designation as a 'no anchoring area' by the International Maritime Organization; u-bolts and mooring buoys were installed to diminish further anchor impacts. While these buoys are used by fishing and diving vessels, debris reports were not specifically associated with them.

Table 3.10. Marine debris area (cm^2), area of benthos affected (cm^2) by debris, and dominant flora and fauna colonizing debris from 2010-2012 surveys.

Types	Area	Area Affected	Debris Colonized By
Anchor	50	1525	*Millepora*
Soda bottle	12	12	Crustose coralline algae
Fishing gear	50	50	None
Fishing line	100	100	None
Fishing line	150	10	Crustose coralline algae, *Millepora*
Line	10	10	Crustose coralline algae
Line/buoy	70	100	Cyanobacteria and crustose coralline algae
Rope	30	30	Coral and algae
Rope	610	610	Algae
Seismic cable	200	200	*Millepora*, turf, sponge, macroalgae, *P. astreoides*, crustose algae
Seismic cable	288	288	Turf algae

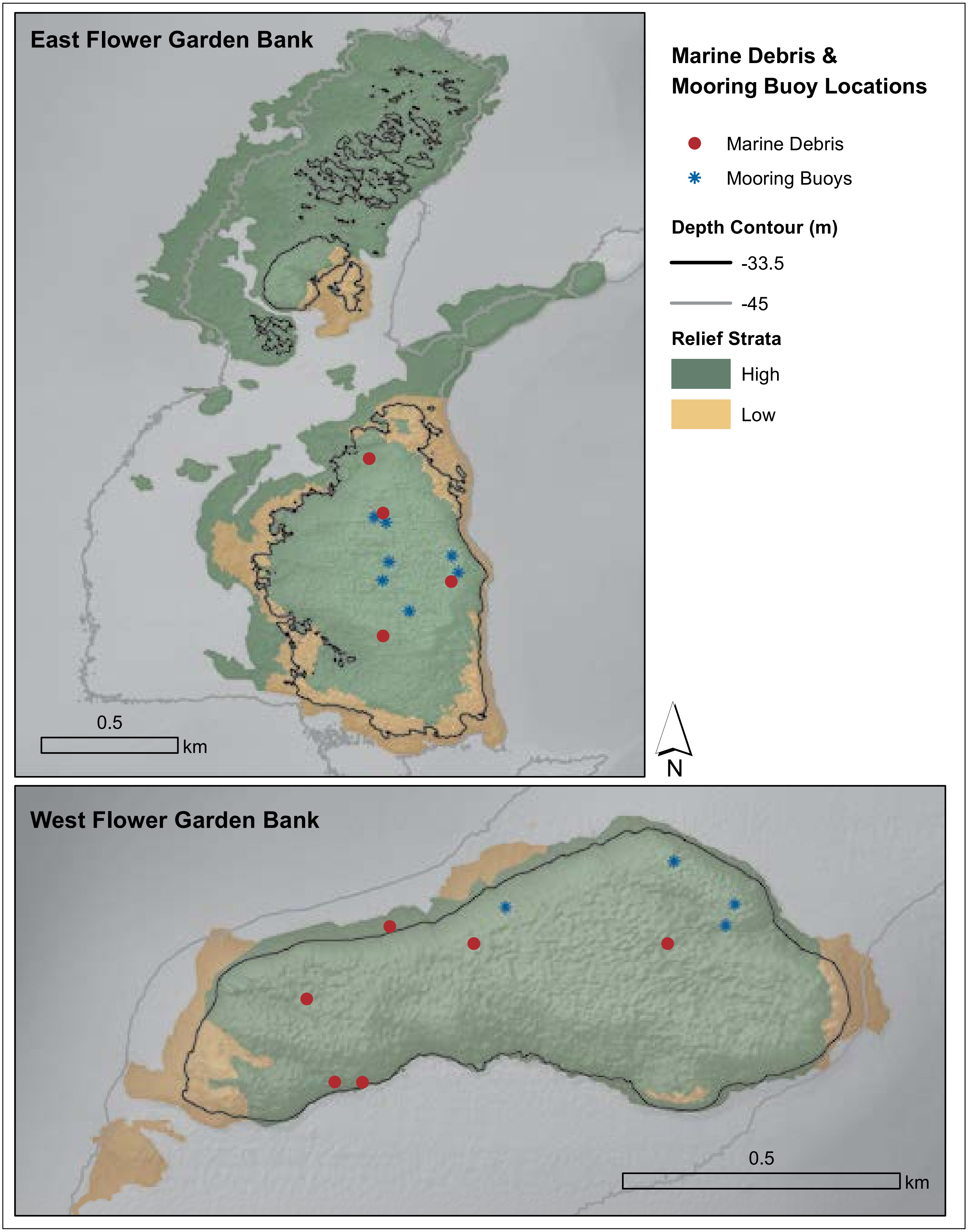


Figure 3.45. Locations of marine debris reports during 2010 – 2012 diver surveys. Locations of mooring buoys are noted for reference.

3.3.8. *Gambierdiscus* Investigation

As an ancillary effort and in response to new U.S. Food and Drug Administration (FDA) guidance on ciguatera fish poisoning (CFP), we sampled for presence and distribution of the causative organisms, *Gambierdiscus* spp. (Figure 3.46). CFP is a human illness caused by eating reef fish that consume and bioconcentrate toxins produced by benthic microalgae in the genus *Gambierdiscus* (Yasumoto, 2005). Older, larger carnivorous fish tend to accumulate the highest levels of ciguatoxin. While rarely fatal, CFP can cause a variety of adverse gastrointestinal, neurological and cardiovascular symptoms that may persist for days, months or years (Freudenthal, 1990). However, death can occur due to respiratory failure. Illnesses associated with CFP have been reported more frequently over the years and it is now considered the most common non-bacterial seafood related illness.

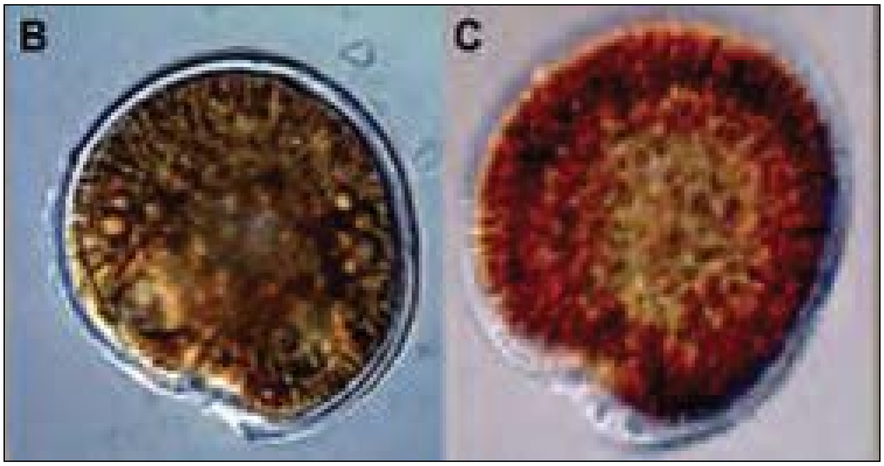


Figure 3.46. Light micrographs of newly described Gambierdiscus *species. Example images of two of four species found in the Caribbean Sea and at FGBNMS:* G. carolinianus *(left) and* G. carpenteri *(right). Source: GEOHAB, 2012*

In 2008, a case of CFP was confirmed by the FDA when fishers became ill after consuming a *Mycteroperca microlepis* (gag grouper) caught at the FGBNMS. Follow-up by FGBNMS, FDA, and the University of Texas Marine Science Institute, led to a regional seafood advisory released by the FDA. Seafood harvesters and processors who purchased fish caught in and around FGBNMS were requested to reassess the risks associated with fish caught within 16 to 80 km of the FGBNMS depending on the species of fish (http://www.fda.gov/NewsEvents/Newsroom/PressAnnouncements/2008/ucm116851.htm ; this advisory was reinstated in 2013 by FDA: http://www.fda.gov/Food/GuidanceRegulation/GuidanceDocumentsRegulatoryInformation/Seafood/ucm375214.htm). Were these fish actually becoming toxic from *Gambierdiscus* at the FGBNMS or did the fish migrate to the FGBNMS from other locations?

To address this question, in part, divers collected samples of opportunity during this 3-year study to determine whether *Gambierdiscus* species capable of introducing ciguatoxins into the local food chain were present in FGBNMS. Assay results showed that while FGBNMS is only a 150 km^2 area, the habitats there support six of the seven *Gambierdiscus* species found in the Caribbean (Tester et al., 2013; Table 3.11). The toxicity of three of the *Gambierdiscus* found in FGBNMS has been established using either receptor-binding (Chinain et al., 2010) or N2A cytotoxicity assays (Lartigue et al., 2009), both of which are considered reliable functional assays for

Table 3.11. Gambierdiscus *species found during 2010-2012. Lartigue et al. (2009) assayed the Caribbean strains 1651 and 1655 for toxicity and these were later identified to species by Litaker et al. (2009). *CCMP is now the National Center for Marine Algae and Microbiota (https://ncma.bigelow.org/).*

Species	Toxicity	Reference
Gambierdiscus carolinianus	Not tested for ciguatoxins	
Gambierdiscus caribaeus	Strain CCMP 1651* N2A cytotoxicity assay	Lartigue et al., 2009; Litaker et al., 2010
Gambierdiscus carpenteri	Not tested for ciguatoxins	
Gambierdiscus belizeanus	Caribbean strain STB-1 Receptor binding assay	Chiain et al., 2010
Gambierdiscus ribotype 2	Strain CCMP 1655* N2A cytotoxicity assay	Lartigue et al., 2009; Litaker et al., 2010
Gambierdiscus ruetzleri	Not tested for ciguatoxins	

cytotoxicity (Caillaud et al., 2010). All three of these toxic species were relatively common in the sanctuary, which may account for the 2008 CFP event. The high diversity of *Gambierdiscus* species observed at FGBNMS is consistent with the regional scale sampling indicating that overlapping distributions of toxic and non-toxic species within a small geographic region is common (Figure 3.47; Tester et al., 2013).

Temperature and salinity data were collected from 2010-2012 by Sea-Bird CTD and HOBO conductivity loggers from various bottom locations in the sanctuary. These data were crucial to predict how *Gambierdiscus* species may respond to climate change. Predictions, based on laboratory studies (Kibler et al., 2012), showed that *Gambierdiscus* growth rates typically increase significantly as temperatures rise from 25-31°C and ceases and cells begin to die between 15-20°C, depending on the species. It is this vulnerability to temperatures less than 20°C which restricts the range of *Gambierdiscus* species to subtropical and tropical regions. Based on the combined laboratory and field data, conditions for optimal *Gambierdiscus* growth occur approximately half the year in FGBNMS (Tester et al., 2013). Wintertime temperatures in the southern most section of the sanctuary near the shelf break, however, routinely drop to 17.5°C which severely limits growth and likely kills some portion of the resident *Gambierdiscus* populations. Water temperatures at Stetson Bank, in the northern area of the FGBNMS, can drop to 15.0°C causing even greater mortality.

Climate models indicate the northern Gulf of Mexico will experience an increase of 2.1-2.2°C to depths of 200 m by 2100. A one degree increase in water temperature is projected to add about 26 days of optimal growth conditions for *Gambierdiscus* species per year in the northern Gulf of Mexico seaward of the 100 meter isobaths (Tester et al., 2013). The more substantial projected increase of approximately 2°C could add an average of 51-55 days of optimal growth conditions. Increasing temperatures would also reduce or eliminate temperatures cold enough to cause *Gambierdiscus* mortality. The combination of increased days of optimal growth and reduced/eliminated low temperature mortalities means that even slight temperature increases projected for the northern Gulf of Mexico within the next few decades portend greater CFP risk. The well-characterized habitats across FGBNMS are ideal sentinel sites for observing changes in *Gambierdiscu*s distribution. Seasonal monitoring of FGBNMS could provide forewarning of increased CFP risk affecting public health and potential economic impacts resulting from effects of climate change (see http://Gambierdiscuswiki.wikispaces.com for more details).

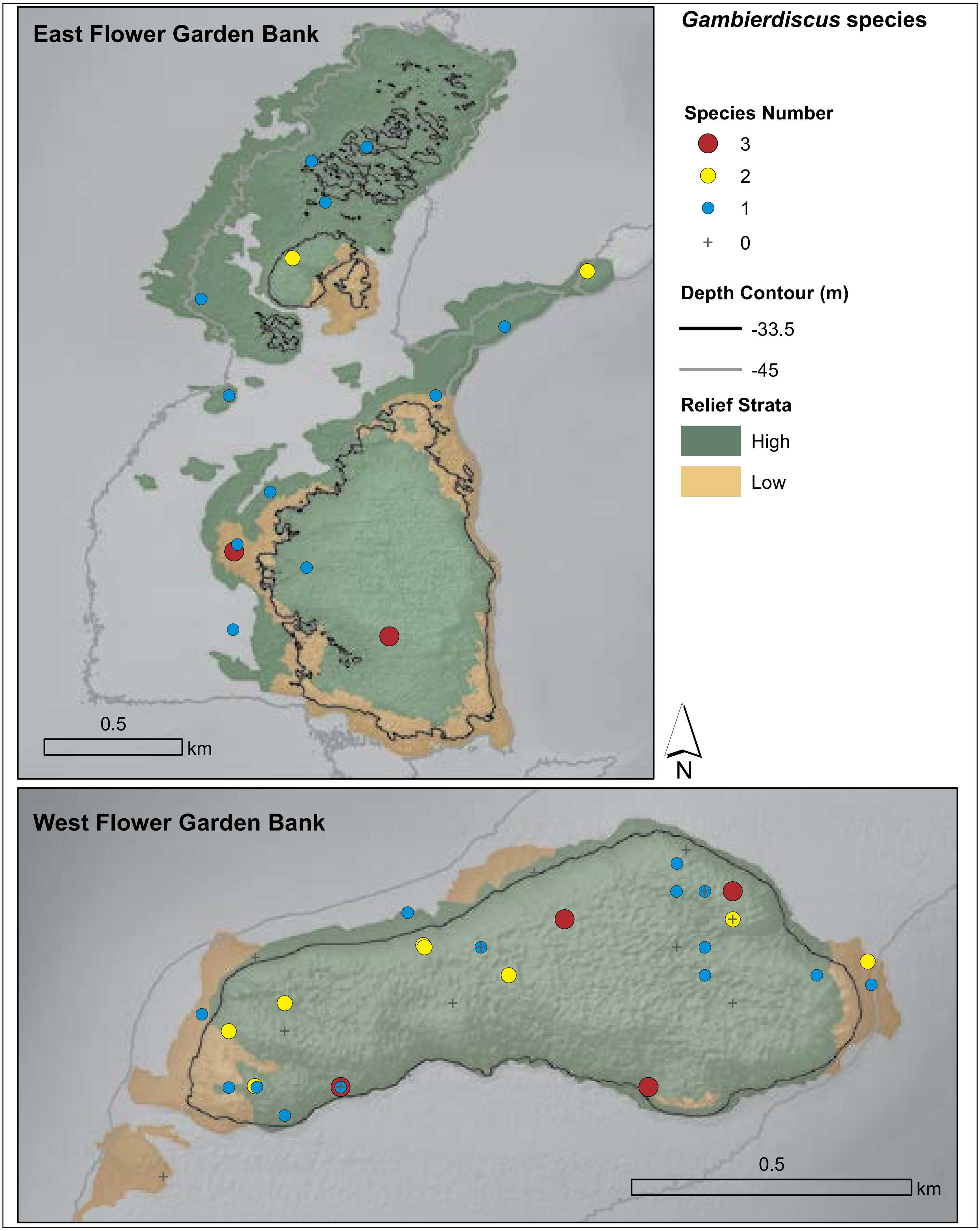


Figure 3.47. Distribution of Gambierdiscus *species in algae samples collected from a subset of dive sites in 2010 – 2012.* Gambierdiscus *species data are presented as graduated symbols where the size is proportional to the number of species observed at each site.*

3.4. SUMMARY

- Coral reef communities of FGBNMS are dominated by coral (52% mean cover per site). Coral cover recorded here is comparable to previous FGBNMS studies that span 20 years, identifying these communities among the healthiest and thriving in the tropical and subtropical western Atlantic. Coral cover declines reported throughout the Caribbean have not been observed here.
- Thirty one species of scleractinian coral were identified in this study. Coral cover exceeds 90% at some sites and is among the highest of U.S. reefs. Incidence of coral bleaching was generally low (1.3% per site). Coral disease occurrences were rare (0.3% per site).
- Few differences in community composition between banks were recorded, although benthic communities did vary based on relief (high and low), and by depth. High relief habitats had high coral cover, whereas lower relief habitats had more algae and rubble, although some low relief sites contained extensive *Madracis auretenra* cover.
- These surveys provide the first comprehensive assessment of UM depths (33.5-45 m). This depth range supports a rich benthic community, with high coral cover (39% mean per site) and species richness (20 species or species complexes). Some increases in species specific cover were recorded with depth, notably: *Montastraea cavernosa*, *Stephanocoenia intersepta*, sponge and macroalgae. Coral morphology transitioned from mounding corals to plating forms with increased depth, but continue to provide structural complexity for a diverse fish community.
- High site cover (>50%) of the ESA candidate species within the *Orbicella* genus was frequently recorded; these were the dominant corals on the coral reef.
- *Diadema antillarum*, the long-spined sea urchin, was rarely recorded in diver surveys (17 individuals), but numbers were higher than in other recent studies. However, many individuals in a range of sizes were observed during a night dive, suggesting this species is rebounding from the die-off in 1983.
- Majority of the marine debris reported were located within shallow surveys (10 of 12 items) and included, oil and gas exploration materials, fishing gear, food debris (soda bottles), and boat equipment.
- Benthic microalgae in the genus *Gambierdiscus*, which produce toxins known to cause the human illness CFP, were found in all strata, with the deepest record to date (45 m). Six species were reported within FGBNMS; three of these are known to be toxic species and were relatively common in the sanctuary. Predicted temperature increases of the Gulf of Mexico waters will make the habitats of FGBNMS more suitable for *Gambierdiscus* species growth, further increasing the risk of CFP. Seasonal sampling across a broad depth range within FGBNMS would provide the sanctuary with a better understanding of CFP risks.
- Further monitoring of the benthic communities of FGBNMS will improve our understanding of benthic and fish community linkages and impacts of natural events (e.g., hurricanes, bleaching) and anthropogenic effects (e.g., fishing, diving, oil/gas exploration). It is recommended that extending quantitative surveys to 45 m depth or greater is needed to effectively monitor potential changes over time.

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Chapter 4
Fish Communities of the Coral Reef

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Photo courtesy of G. McFall (NOAA NOS/ONMS/GRNMS)

Fish Communities

4.1. INTRODUCTION

Figure 4.1. Fish community on a reef habitat at Flower Garden Banks National Marine Sanctuary (FGBNMS). Photo: G. McFall (NOAA NOS/ONMS/GRNMS)

Located approximately 180 km south of Galveston, Texas, Flower Garden Banks National Marine Sanctuary (FGBNMS) is one of the most pristine coral reef ecosystems in U.S. waters (Figure 4.1). Further description of its geographic setting and general habitat characteristics can be found in Chapter 1, and detailed descriptions of benthic communities of the coral reef are contained in Chapter 3. Surveys of fish assemblages have been conducted at FGBNMS since the early 1980s; however, these initial surveys were mostly qualitative (Boland et al., 1983; Rezak et al., 1985; Dennis and Bright, 1988; Pattengill, 1998). Quantitative fish surveys were added to the FGBNMS long-term monitoring (LTM) program in 2002, but spatial coverage was limited (Precht et al., 2006). Long-term monitoring and earlier studies identified fish assemblages of the coral reef zone (17-35 m) at the East and West Banks (hereafter, EB and WB, respectively) to be composed of a subset of Caribbean reef species (Boland et al., 1983; Rezak et al., 1985; Dennis and Bright, 1988; Pattengill, 1998; Precht et al., 2006; Johnston et al., 2013). While species diversity at FGBNMS is lower than Caribbean reef ecosystems, the sanctuary supports significantly more apex predator biomass. The biomass structure of the sanctuaries trophic groups strongly resemble those in remote Pacific Islands that receive less fishing intensity due to their distance from population centers.

The LTM study identified herbivores as the dominant fish guild, with Scarinae (parrotfishes) and Acanthuridae (surgeonfishes) comprising the majority of herbivores recorded; species diversity of piscivores was also lower than herbivores (Precht et al., 2006; Johnston et al., 2013). Certain families, such as the Lutjanidae (snappers) and Haemulidae (grunts), have been reported as less abundant or completely absent compared to Caribbean fish communities (Jones and Clark, 1981; Lukens, 1981; Rezak et al., 1985; Precht et al., 2006; Johnston et al., 2013), presumably due to a lack of nearby seagrass and mangrove habitats (Mumby et al., 2004).

The first spatially comprehensive fish surveys of the coral reef (17-33.5 m) were conducted from 2006-2007 (Caldow et al., 2009). Fish community results were similar to those reported by previous authors and the LTM program: lower diversity of fishes were reported in FGBNMS than the Caribbean but occasionally high densities and larger sized individuals have been recorded in the sanctuary. The larger spatial coverage from Caldow et al. (2009) suggested fish communities changed with depth. Based on these findings they recommended fish community surveys be standardized and conducted over the entirety of the coral reef (max depth 46 m). Data from this study are intended to inform sanctuary management and potential designation of a research only area. This three year study (2010-2012) uses the methods of Caldow et al. (2009) and provides the first comprehensive and quantitative fish community data across the majority of the coral reef (to 45 m). Remotely operated vehicle (ROV) surveys were conducted at depths greater than 45 m; these results are presented in Chapter 5.

Fish Communities

In this chapter we present fish community data collected throughout the range of the coral reef (17-45 m), utilizing scuba to accomplish the following objectives:

- Quantify the density and biomass of all fish species encountered
- Examine the prevalence and distribution of apex predators (fishes from the families Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae >50 cm FL)
- Describe trophic group structure and composition
- Identify the presence and describing the distribution of the invasive *Pterois volitan*s (red lionfish)
- Assess the relationship between fish density, biomass and species diversity with depth, benthic biotic community cover, and reef complexity
- Understand similarities and differences across banks and depth to aid sanctuary management plans

Results from this study can serve as baseline information to develop a spatial monitoring plan, either as an independent monitoring program or to supplement the LTM activities, and can also provide a foundation for assessing impacts associated with natural and/or anthropogenic events in the future.

4.2. METHODS

4.2.1. Field

Fish communities were surveyed to a depth of 45 m using a stratified random sampling approach within a pre-defined sampling frame as described in Chapter 3 (Figure 3.2). To examine spatial and structural trends, data were assigned a strata designation based on bank (East or West), reef complexity (high or low relief), and depth (shallow or upper mesophotic, hereafter UM). The shallow stratum extends from 18-33.5 m, while the UM stratum depth range was 33.5-45 m. Definitions and derivations of the strata are described in Chapter 3. Strata sample size varied based on areal extent and logistic constraints due to depth (see Table 3.1 for sample size by strata and year). The shallow stratum was surveyed during each year of the study (2010-2012). Due to vessel availability limitations resulting from the Deepwater Horizon spill in 2010, the UM depth stratum was surveyed in 2011 and 2012 only.

Figure 4.2. Divers conducting fish and benthic composition surveys in the FGBNMS. Photo: G. McFall (NOAA NOS/ONMS/GRNMS)

Visual fish surveys were conducted simultaneously with benthic surveys (Chapter 3) along 100 m^2 transects at randomly selected sites (see Figure 3.2). During each 15 minute survey, a diver swam a 25 x 4 m transect and identified, counted, and measured all fish species (Figure 4.2). All fish were identified to species or the lowest possible taxon, with densities expressed as the number of fish per 100 m^2. All fish were sized using fork length (FL) in 5 cm categories up to 35 cm; actual values were used for fish greater than 35 cm (more detailed descriptions of methods are provided in Appendix B).

Summary statistics calculated for all species observed included the following: total species occurrence, percent occurrence, total abundance, mean abundance with standard error (± SE), total biomass, and mean biomass (± SE). Biomass (g) was calculated using the length-weight power function ($W = a \times L^b$) and converted to kilograms (kg). Length was determined using the midpoint of 5 cm categories or the actual fish length (where FL >35 cm). A fork length of 3 cm was used for the smallest size class (0-5 cm) midpoint, as fish <1 cm FL were not observed. FishBase (www.fishbase.org) was used to obtain *a* and *b* parameters. For species without published a and b parameters, values from the closest congener, based on morphology, were used. Further, where *a* and *b* parameters were only available for lengths other than FL, the appropriate length conversion factors were used to convert to the appropriate length prior to calculating biomass. Fish were also assigned to a trophic group (piscivore, invertivore, planktivore or herbivore) based on published information or from information provided from FishBase.

Parrotfishes were previously classified in the family Scaridae, but recent genetic analyses have placed them in the subfamily Scarinae within the wrasse family, Labridae (Westneat and Alfaro, 2005). Hereafter referring to the parrotfishes as Scarinae, we recognize their subfamily status but analyzed parrotfish data separately from other labrids and made comparisons between this subfamily and other families to enable comparisons to previous studies.

4.2.2. Statistical Analysis

Univariate

Trends occurring across the surveyed depth gradient were analyzed using correlative analyses. The relationship between actual site depth (m), rugosity (see Chapter 2 for explanation of derivation), benthic community parameters (percent cover of rock, rubble, sand, total biotic cover, macroalgae, and coral), and fish trophic and taxonomic groups, were completed using non-parametric Spearman's ρ (rho) rank correlations. When fish data were compared to algal and total biotic cover data, sample sizes were reduced from 291 to 275, due to benthic field sampling irregularities at 16 sites. Habitat relief, or structure, has been identified as an important factor influencing fish communities (e.g., Connell and Kingsford, 1998; Sluka et al., 1998). Here we examined two reef structure parameters, relief strata and rugosity, as they related to fish density and biomass. While trends were generally similar between relief strata comparisons and rugosity correlations, some differences did occur. It is important to note, rugosity values within the UM zone may be low due to habitat configuration (e.g., reef-sand interface) with half the site containing moderate relief reef and the other sand. Finer resolution rugosity data are needed to confidently identify distribution patterns by rugosity.

Fish community differences among strata were evaluated by comparing overall abundance, biomass, species richness (number of species), and species diversity (Shannon diversity Index). Differences by bank, depth and relief strata were examined using the non-parametric Wilcoxon test (Z). Inter-annual differences in fish species or species group density and biomass were examined using a Wilcoxon test for the UM stratum, and a Kruskal-Wallis test for the shallow stratum. Any significant between year differences within the shallow stratum were examined further using a sequential Bonferroni correction to control the group wide Type 1 error rates (Rice 1989).

Multivariate

Fish density and biomass data were 4th root transformed to down-weight the importance of highly abundant species prior to analysis with PRIMER v6 software (Clarke and Gorley, 2006). Non-metric multi-dimensional scaling (MDS) plots of fish community structure (based on biomass or density) were visually examined for evidence of community differences by four categorical variables: year, depth strata (shallow and UM), relief strata (high and low), and bank (East and West). The importance of each categorical variable to fish community structure was determined simultaneously with permutational multi-way analysis of variance (PERMANOVA).

Fish Communities

The design was unbalanced, with depth strata nested in year and relief nested in bank. Significant differences in fish community structure were examined with the similarity percentages (SIMPER) routine to identify those species that contributed most to the observed dissimilarity.

To determine the role of nine continuous variables (depth (m), rugosity, and percent cover of habitat variables including: rock, rubble, sand, algae, hard corals, hydrocorals and sponges) in structuring fish communities, the global BEST and LINKTREE procedures were combined (Clarke et al., 2008). First, the global BEST procedure was conducted with 999 permutations to determine which community variable(s) 'best' explained the pattern of fish community structure (based on density or biomass). The variables that had the highest spearman rank correlation with the corresponding fish community resemblance matrix reflected those factors most important in structuring the fish communities. The BEST analyses were conducted three different ways: all data combined, shallow stratum only, and UM stratum only. Those variable(s) with the highest Spearman rank correlation from the global BEST procedure were then used within the LINKTREE multivariate regression procedure to determine the actual values of the variables that constituted thresholds for defining fish community differences. The significance level within LINKTREE was set with the similarity profile (SIMPROF) procedure at 0.05, with the additional constraint of limiting group separation to no less than four sites (Harborne et al., 2012a). Absolute group differences at each threshold of division are given by the B% level. B% provides a general measure of the degree of separation of the groups and its overall importance within the tree. Thus, significant separation can be considered hierarchical within the 'tree', where the most important variables (and respective values) are located higher up in the tree, with a higher B% denoting a greater degree of separation.

4.3. RESULTS AND DISCUSSION

Similar to benthic surveys (Chapter 3), fish surveys were conducted at a total of 291 sites; 225 within the shallow strata (<33.5 m) and 66 within the UM strata (33.5-45 m; see Table 3.1 and Figure 3.2).

4.3.1. General Community Metrics

Overall, 123,064 fish totaling 10,207.3 kg from 129 species or species groups and 36 families were observed (n = 291). On EB, a total of 70,667 individuals from 121 species and 35 families weighing a collective 5,770.2 kg were recorded. Totals were lower in all categories for WB (52,397 individuals, 4,437.1 kg, 101 species, and 32 families) however total sampling effort was less than EB (Table 3.1). Examining the fish community by bank, relief and depth indicated higher species richness, density and biomass in the UM stratum than in the shallow with the exception of species diversity (Table 4.1). The highest species richness, density, and biomass were recorded within the WB UM, high relief stratum although the standard error was large due to the smaller number of surveys.

Table 4.1. Summary statistics (mean [SE]) for fish community metrics by bank, depth and relief strata for diver surveys conducted from 2010-2012.

Depth	Bank	Relief	N sites	# Families	Diversity	Richness	Density (# / 100 m^2)	Biomass (kg / 100 m^2)
Shallow	EAST	HIGH	114	11.3 (0.20)	2.14 (0.03)	23.69 (0.41)	304.64 (22.15)	23.32 (2.46)
	WEST	HIGH	86	13.0 (0.22)	2.11 (0.05)	26.40 (0.49)	414.14 (40.51)	32.45 (4.67)
	EAST	LOW	21	10.0 (0.45)	1.90 (0.11)	21.19 (1.05)	591.29 (127.8)	24.05 (8.61)
	WEST	LOW	4	12.0 (1.29)	2.22 (0.06)	26.00 (1.78)	288.25 (42.20)	15.03 (4.74)
Shallow Total			***225***	***12.2 (0.33)***	***2.11 (0.03)***	***24.53 (0.32)***	***372.96 (23.11)***	***26.73 (2.33)***
Upper Mesophotic	EAST	HIGH	31	14.5 (0.45)	2.12 (0.07)	26.58 (0.95)	591.23 (129.10)	63.62 (17.98)
	WEST	HIGH	15	11.2 (0.54)	2.00 (0.12)	28.33 (0.75)	836.60 (207.58)	92.81 (33.27)
	EAST	LOW	13	12.3 (0.71)	2.14 (0.12)	23.77 (1.30)	399.46 (136.07)	48.82 (12.43)
	WEST	LOW	7	11.9 (0.15)	1.81 (0.25)	24.57 (2.05)	439.86 (143.61)	27.68 (7.18)
Upper Mesophotic Total			***66***	***12.5 (0.26)***	***2.07 (0.06)***	***26.21 (0.60)***	***593.17 (83.41)***	***63.53 (11.67)***

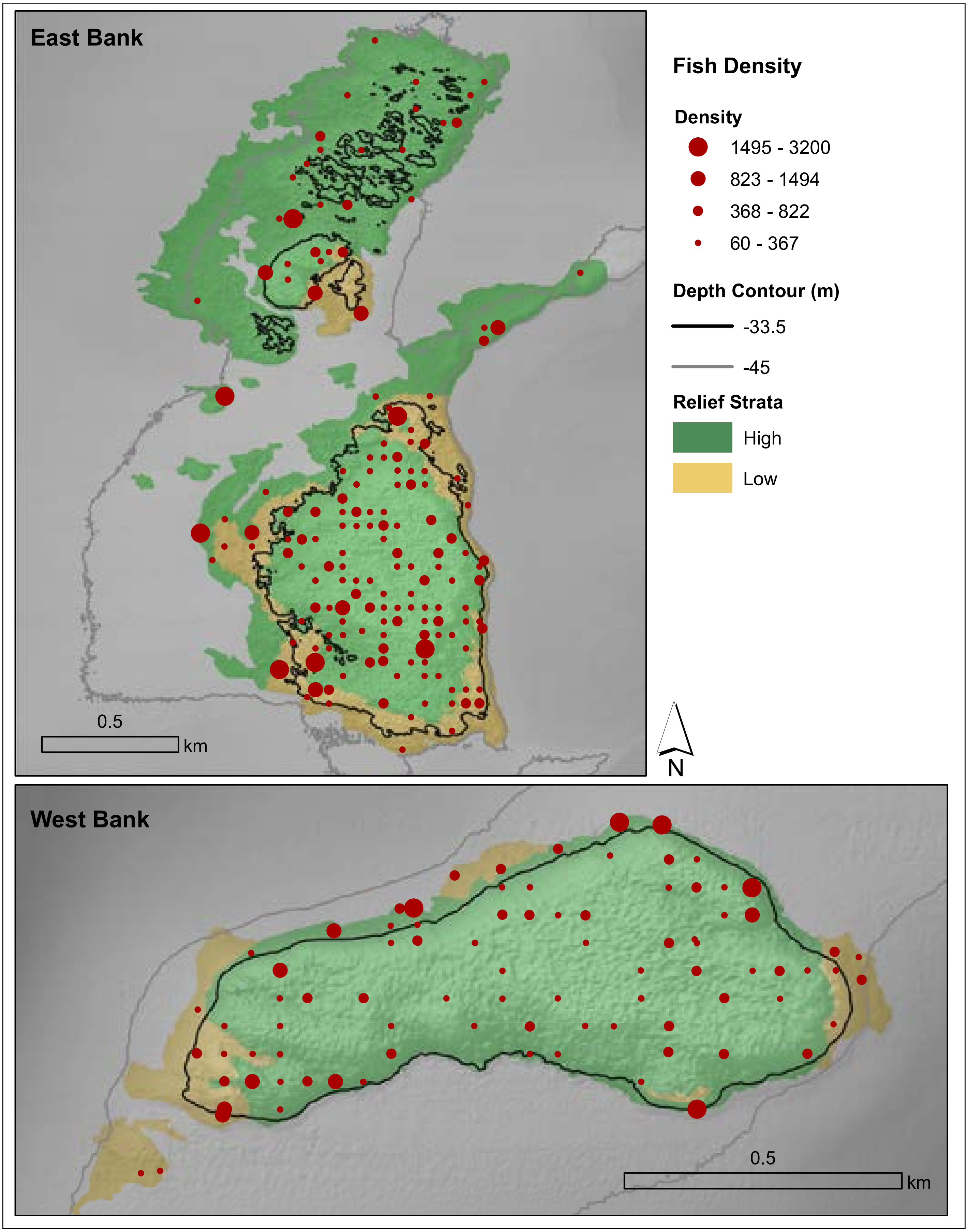


Figure 4.3. Observed fish density (#/100 m²) recorded during diver surveys from 2010-2012.

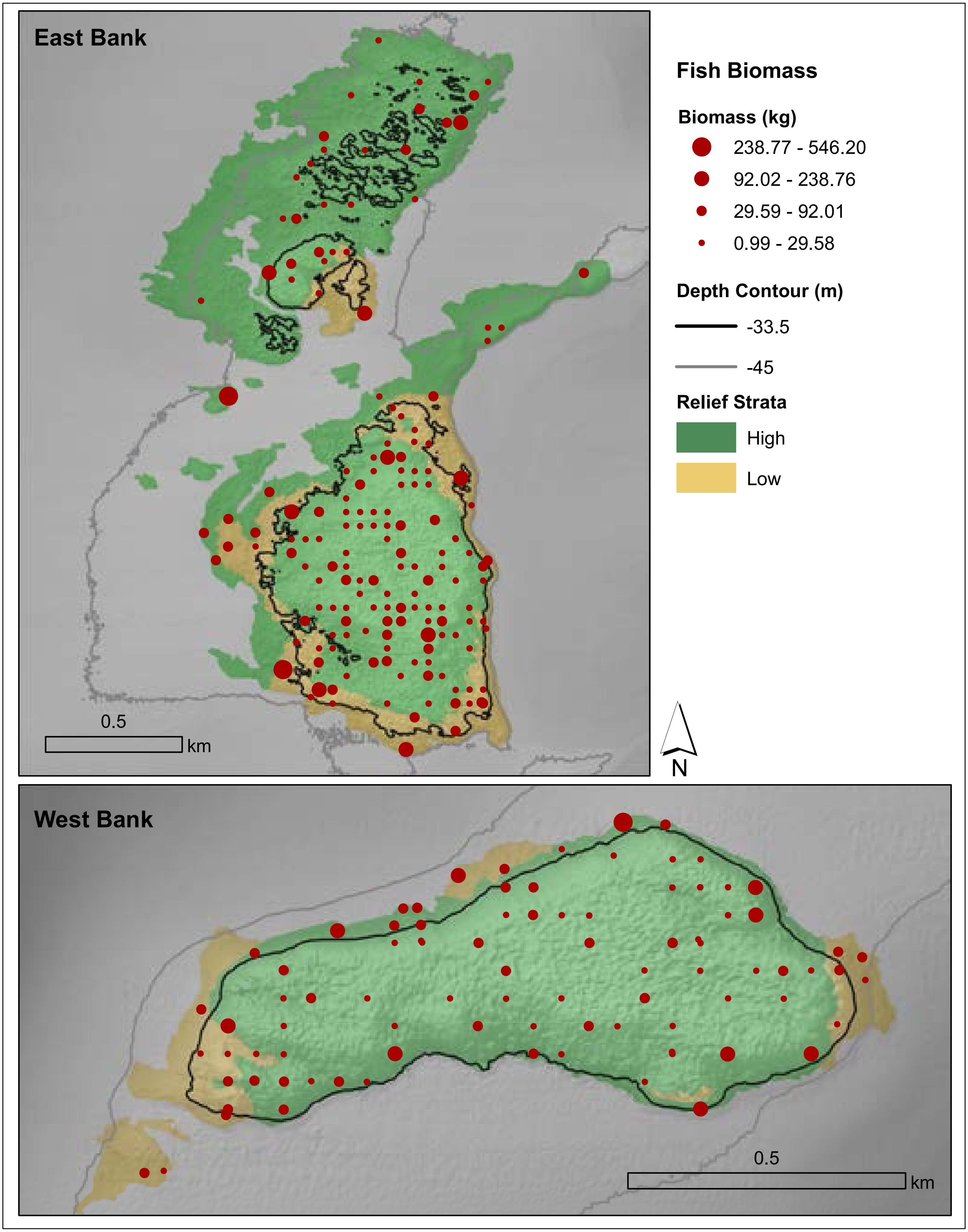


Figure 4.4. Observed fish biomass (kg/100 m²) recorded during diver surveys from 2010-2012.

Within the shallow stratum, fish density and species richness were significantly greater on WB than EB (t-test, t = 2.09, p = 0.0378; t = 5.05, p < 0.0001, respectively). Biomass and species diversity were not different between banks. A previous study within the same depth strata reported higher density, biomass, and species richness on EB compared to WB (Caldow et al., 2009). While such differences may be due to inter-annual variability, the sample size and spatial coverage of WB was more limited in Caldow et al. (2009) compared to this study.

Generally, higher fish densities (Figure 4.3) and biomass (Figure 4.4) were recorded in the UM strata and along depth strata transitions for both banks. High relief sites typically had higher fish density and biomass than low relief sites, but this difference was only significant for biomass at UM sites (t-test, t = 1.729, p = 0.0443). Some large biomass values were recorded at low relief sites which were often near the transition between high and low relief habitats (Figure 4.4).

Despite this study's greater number of surveys (n = 291) and expanded depth range (45 m maximum depth) than a previous study (n = 105 sites; 33 m maximum depth; Caldow et al., 2009), the family richness was similar between the two studies, 35 families versus 37 families in Caldow et al. (2009), and species richness was slightly higher here at 129 species versus 117 species in Caldow et al. (2009). Notable species differences between these two studies were high numbers (>100 individuals) of *Inermia vittata* (boga), *Haemulon melanurum* (cottonwick), and *Parablennius marmoreus* (seaweed blenny) in this study and their absence in the previous (Caldow et al., 2009). Only one species in the 2009 study was recorded with >100 individuals and not reported here, *Haemulon parra* (sailors choice). *I. vittata* were found exclusively within the shallow stratum (eight sites), while *H. melanurum* and *P. marmoreus* were found in both depth strata, they were nearly 10 times more abundant within the UM stratum. Species richness was uniform on the WB, with slightly lower levels at deeper, low relief sites (Figure 4.5; Table 4.1). On EB, species richness was variable by bank and relief strata with many low relief sites having a high number of species recorded, particularly on the edges of the coral reef. The spatial pattern of species diversity was similar to species richness (Figure 4.6).

Table 4.2. Top five families (or subfamily Scarinae) in density and total biomass from diver surveys (2010-2012). Shown is percent of total density and biomass with number of species within each family in parentheses.

Family	Density	Family	Biomass
Pomacentridae (13)	31.66%	Serranidae (17)	38.77%
Labridae (9)	26.49%	Kyphosidae (1)	12.20%
Serranidae (17)	24.89%	Labridae (9)	11.84%
Inermiidae (2)	3.07%	Carangidae (7)	8.40%
Scarinae (7)	3.00%	Lutjanidae (5)	6.33%
Other (81)	10.89%	Other (90)	22.46%

Table 4.3. Top 15 species in density and total biomass from diver surveys (2010-2012). Percent of total is shown. Members of apex predator families and Pterois volitans *are highlighted in gray.*

Species	Density	Species	Biomass
Paranthias furcifer	24.12%	*Paranthias furcifer*	28.37%
Clepticus parrae	16.03%	*Kyphosus sectator*	12.20%
Chromis multilineata	15.27%	*Clepticus parrae*	11.20%
Thalassoma bifasciatum	7.81%	*Caranx latus*	5.73%
Chromis insolata	7.65%	*Sphyraena barracuda*	5.71%
Kyphosus spectator	2.56%	*Mycteroperca bonaci*	4.55%
Stegastes partitus	2.34%	*Lutjanus griseus*	3.32%
Emmelichthyops atlanticus	2.21%	*Lutjanus jocu*	2.89%
Stegastes planifrons	2.20%	*Mycteroperca interstitialis*	2.20%
Bodianus rufus	1.62%	*Mycteroperca tigris*	2.08%
Canthigaster rostrata	1.48%	*Galeocerdo cuvier*	1.87%
Chromis scotti	1.27%	*Manta birostris*	1.60%
Chromis cyanea	1.11%	*Sparisoma viride*	1.53%
Stegastes variabilis	1.05%	*Mulloidichthys martinicus*	1.50%
Carangoides ruber	1.04%	*Melichthys niger*	1.18%

Fish Communities

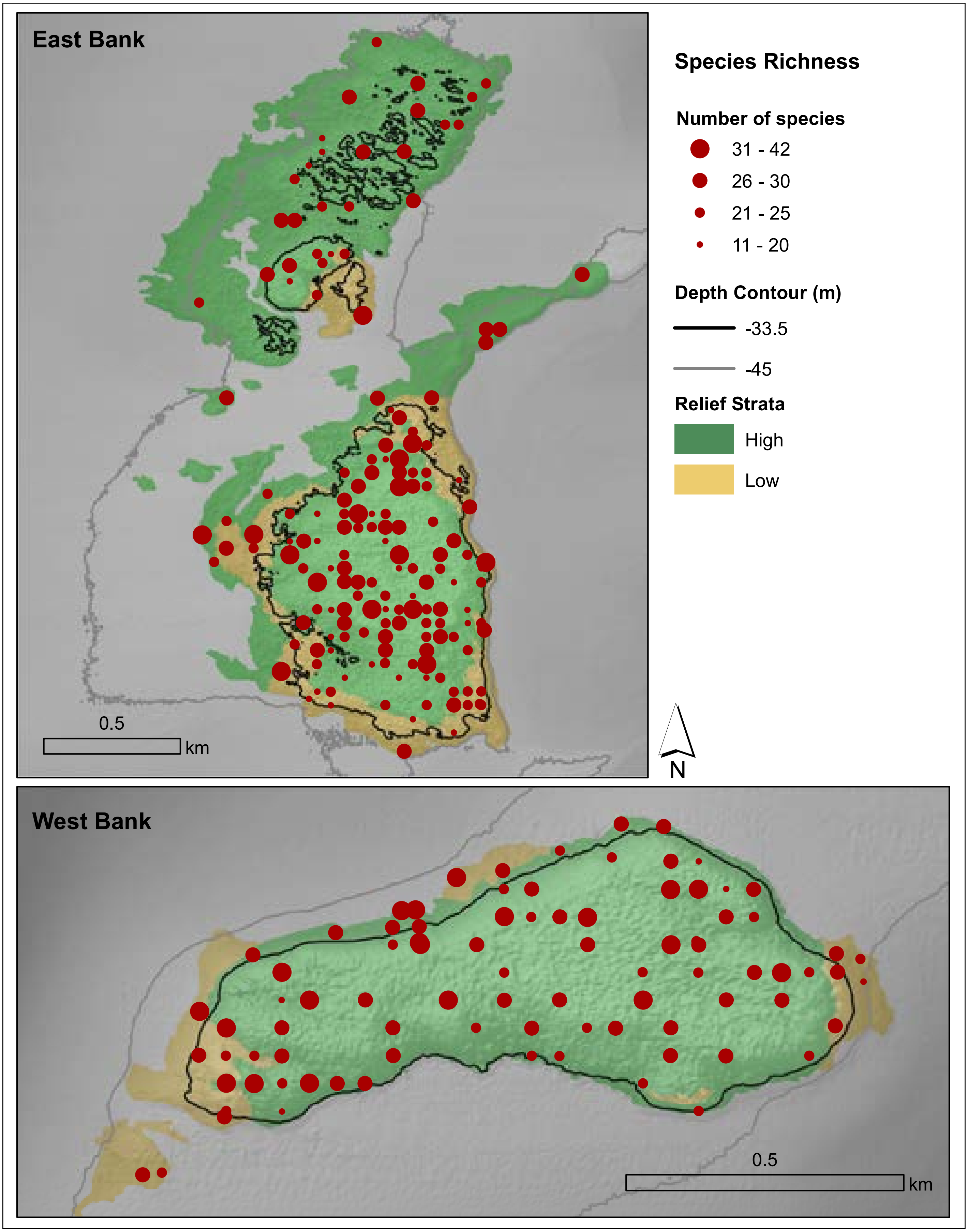


Figure 4.5. Observed fish species richness recorded during diver surveys from 2010-2012.

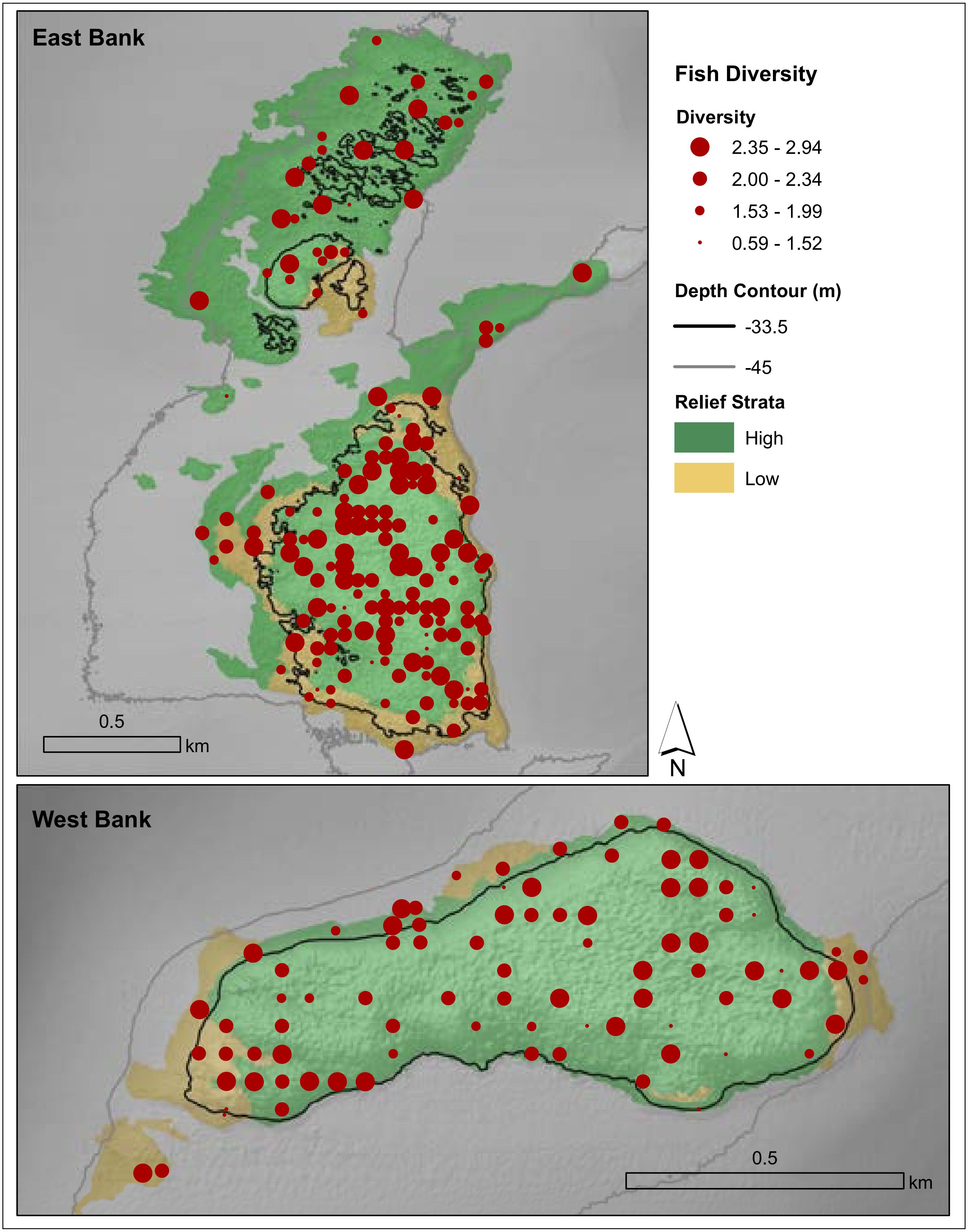


Figure 4.6. Observed fish species diversity recorded during diver surveys from 2010-2012.

Fish Communities

More than 50% of the total fish density was comprised of two fish families, Pomacentridae (Damselfish) and Labridae (Wrasse; Table 4.2). Four species from these two families encompassed four of the top five most abundant species (Table 4.3). Serranidae (Groupers) was the 3rd most abundant family, predominantly represented by *Paranthias furcifer* (Atlantic creolefish), the top species by density and biomass (Table 4.3). The top five most abundant families recorded here are consistent with Caldow et al. (2009); percentages were similar between the two studies except for a small decline in Labridae (35% Caldow et al., 2009; 26% this study) and an increase in Serranidae (14% to 25%, respectively). This study's increase in serranid density is likely due to the extension of survey depth; within the shallow stratum, serranid densities comprised 20% of total fish density, more similar to the previous study within the same depth range (Caldow et al., 2009). Within the UM stratum, serranids made up 33% of the total fish density. Two schooling, pelagic inermiid species, *I. vittata* and *Emmelichthyops atlanticus* (bonnetmouth), were abundant at just a few sites (n sites = 8, 16, respectively) and primarily found only within the shallow stratum (*E. atlanticus* occurred at one UM site; yet the family ranked 4th overall [Table 4.2]). *E. atlanticus* was the 8th most abundant fish species in this study and ranked second in a previous study (Hickerson et al., 2008). A complete list of mean density for each fish species by depth strata is provided in Appendix C.

Biomass was dominated by serranids (38.7% of total), again driven by the most numerically abundant *P. furcifer* (Tables 4.2 and 4.3). Other heavy bodied Serranidae species among the top 15 species in total biomass included: *Mycteroperca interstitialis* (yellowmouth grouper), *Mycteroperca tigris* (tiger grouper) and *Mycteroperca bonaci* (black grouper; Table 4.3). It should be noted that Serranidae totals included all serranid species, including the smaller bodied species from *Liopropoma* and *Serranus* genera. The 24 individuals from these genera made up a small portion of the total serranid biomass (<0.01%) and density (0.08%) presented here. Spatial distribution and trends of these species are explored in more detail within the Serranid section below. Kyphosidae (chubs), comprised of one species, *Kyphosus sectator* (bermuda chub), ranked second in total biomass, followed by Labridae, Carangidae (jacks), and Lutjanidae. The top families by biomass were similar to Caldow et al. (2009) except for Scarinae (previously Scaridae). Scarinae, tied for the 4th greatest biomass in the 2009 study (6%; Caldow et al., 2009), were 9th in this study with 2.64% of the total biomass. This difference may be attributed to higher Labrid and Carangid biomass in this study compared to those reported in the previous study (Caldow et al., 2009). A complete list of mean biomass for each fish species by depth strata is provided in Appendix C. Some larger bodied species were not numerically abundant but nevertheless, amassed considerable biomass, ranking them within the top 15 species by biomass including: *Galeocerdo cuvier* (tiger shark, n individuals = 3) and *Manta birostris* (giant manta, n individuals = 2; Table 4.3; Figure 4.7).

While some significant between year differences were observed for fish density and biomass, no precipitous declines of the dominant species were observed during the study period. Inter-annual differences in fish density for the dominant fish species were largely observed for fishes with schooling behaviors (e.g., *E. atlanticus* and

Figure 4.7. Larger, migratory species such as Galeocerdo cuvier *(tiger shark; left) and* Manta birostris *(manta ray; right) were observed on a small number of fish surveys (2010 -2012). Photos: G. McFall (NOAA NOS/ONMS/GRNMS) and NOAA NOS/NCCOS/CCMA*

Thalassoma bifasciatum [bluehead wrasse]) and differences were generally identified in only one year, with the exception of *Bodianus rufus* (spanish hogfish; Figure 4.8). Within the shallow stratum, significant between year differences were recorded for *Clepticus parrae* (creole wrasse), *T. bifasciatum*, *Stegastes partitus* (bicolor damselfish), *E. atlanticus*, *Stegastes planifrons* (threespot damselfish), and *B. rufus*. Significant differences were identified for *Chromis multilineata* (brown chromis), *Chromis insolata* (sunshinefish), and *Canthigaster rostrata* (sharpnose puffer) within the UM stratum. Fewer between year differences were identified by species biomass (Figure 4.9).

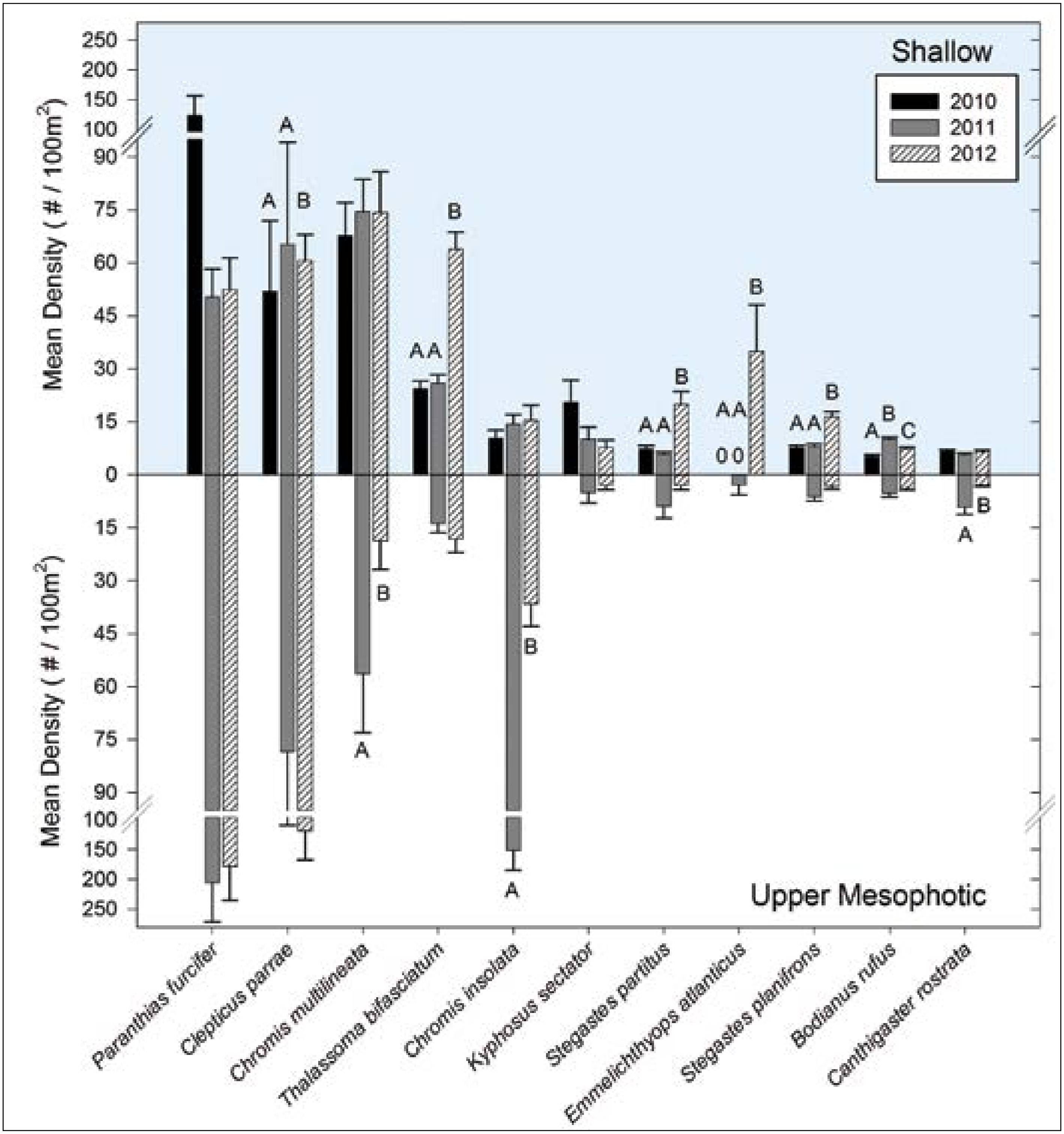


Figure 4.8. Annual mean densities (SE) for the most abundant fish species by depth strata (Table 4.3). Letters above the bars indicate significant between year differences for each species within depth stratum. For example, A and B bars are not significantly different, but A bars are significantly different from B and/or C bars. NOTE: No surveys were conducted in the Upper Mesophotic during 2010.

Fish Communities

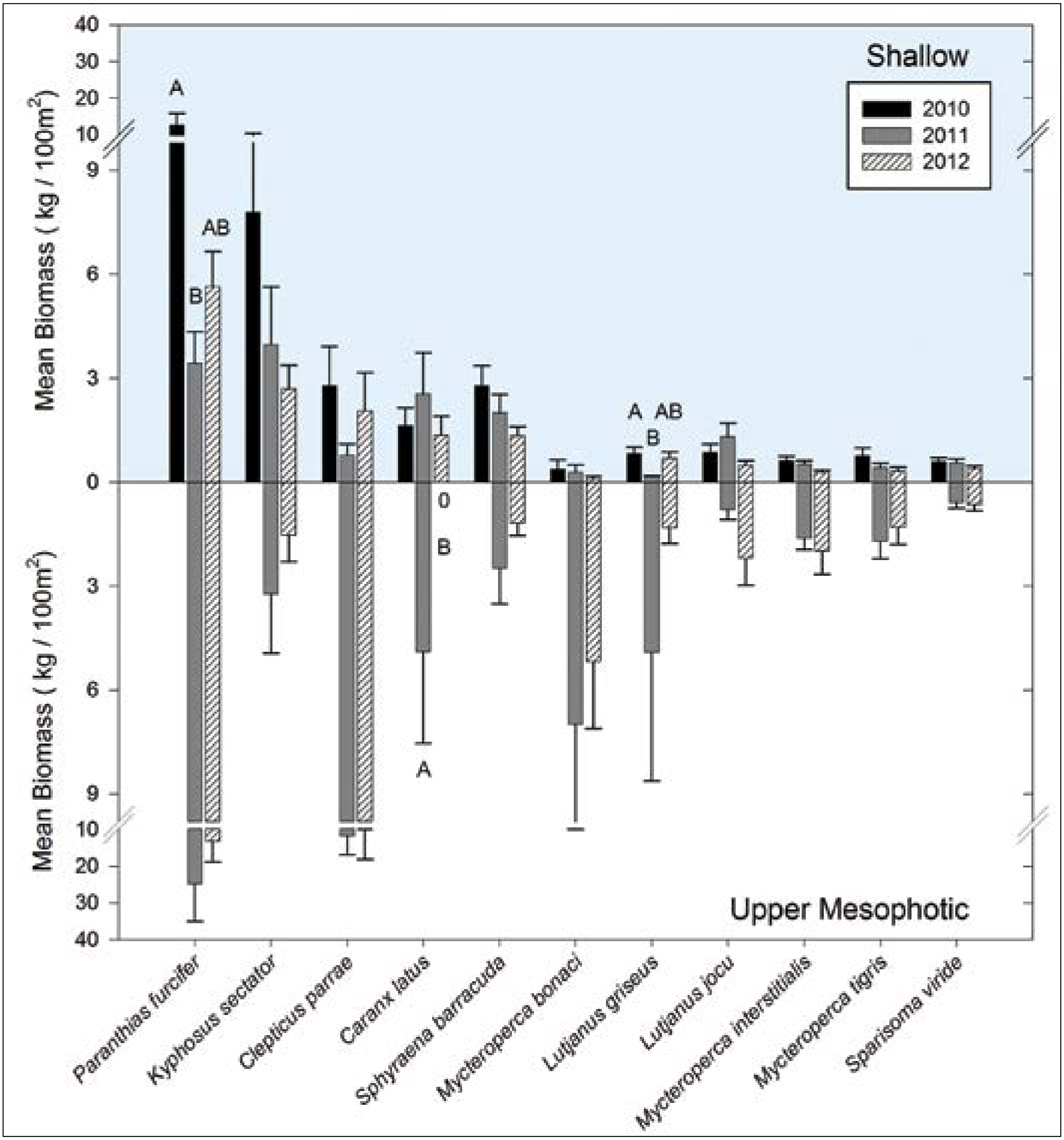


Figure 4.9. Annual mean biomass (SE) for the top 10 species by depth strata (Table 4.3). For example, A and B bars are not significantly different, but A bars are significantly different from B and/or C bars. NOTE: No surveys were conducted in the Upper Mesophotic during 2010.

P. furcifer and *Lutjanus griseus* (gray snapper) biomass means within the shallow stratum were significantly different between 2010 and 2011 surveys; however, 2012 biomass of both species was similar to the previous two years. Within the UM stratum, only *Caranx latus* (horse-eye jack) biomass differed between survey years (2011 vs. 2012). For all three of these species, it is likely the differences are due to chance encounters of larger groups, as each of these species are known to exhibit schooling behavior and in the case of *C. latus*, tend to lack site fidelity.

4.3.2. Fish Assemblages (Multivariate)

Depth strata (shallow versus UM) and relief strata (high versus low) significantly influenced fish community structure based on densities (Table 4.4; Figure 4.10). Based on the square root of estimates of components of variation, depth strata, then relief strata, followed by their interaction, most affected fish community structure (square root = 16.87, 11.42, 8.11, respectively). The average dissimilarity between shallow and UM surveys (based on density) was 54.1%. The top four species responsible for this difference between depth strata were *C. insolata*, *C. parrae*, *C. multilineata*, and *P. furcifer*. Nine species of grouper, snapper and *P. volitans* (the invasive Indo-Pacific red lionfish; Figure 4.11) occurred at higher densities in the UM stratum than the shallow. These include: *L. griseus*, *M. interstitialis*, *Cephalopholis cruentata* (graysby), *M. tigris*, *Lutjanus jocu* (dog snapper), *P. volitans*, *Epinephelus adscensionis* (rock hind), *M. bonaci*, *Mycteroperca venenosa* (yellowfin grouper) and *Epinephelus guttatus* (red hind). A list of species contributing to the dissimilarity of fish communities by depth strata is provided in Table 4.5.

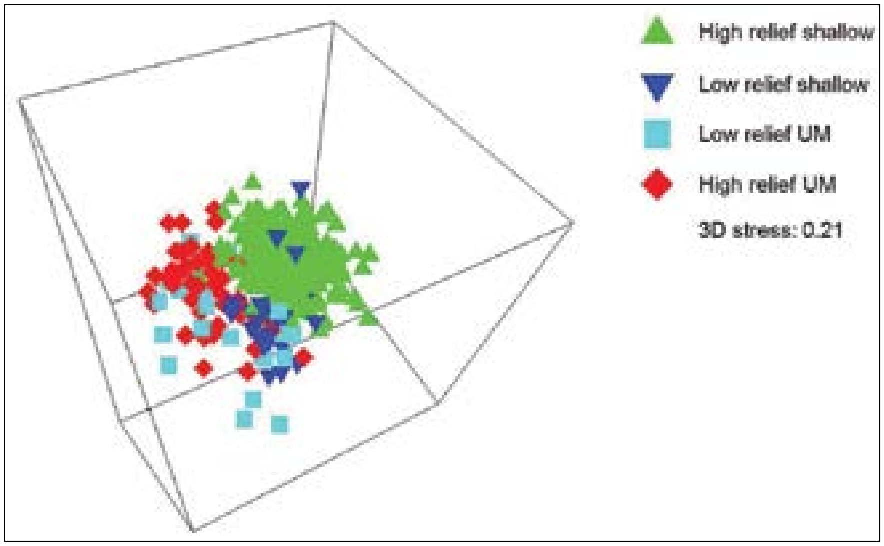


Figure 4.10. Fish community structure at FGBNMS based on densities differed by depth and relief strata (Table 4.4).

*Figure 4.11. An invasive species, the red lionfish (*Pterois volitans*), in the FGBNMS. Photo: C.A. Buckel (NOAA NOS/NCCOS/CCFHR)*

*Table 4.4. PERMANOVA results based on Bray-Curtis dissimilarities of the density and biomass (4th root transformed) of 129 fish species observed from 2010-2012 at FGBNMS as affected by survey year, bank, depth strata, relief strata, and their interactions. Parentheses show those factors within which a particular factor is nested. Significant factors are denoted with *.*

Source	df	Density			Biomass		
		MS	F	P(perm)	MS	F	P(perm)
Year	2	6633.6	0.65514	0.8028	6161.4	0.69913	0.7398
Bank	1	2870.4	0.72119	0.6691	4015	0.98382	0.4254
Depth (Year)	2	7773.4	5.6645	0.0029*	8506	2.2154	0.0654
Relief (Bank)	2	5458.6	3.7275	0.0183*	5179.6	1.3117	0.2919
Year x Bank	2	2233.2	1.2132	0.2742	6361.6	1.3219	0.2357
Year x Relief (Bank)	3	1620.3	1.0519	0.4427	4293.9	1.0586	0.4063
Depth (Year) x Bank	2	1138.5	0.82966	0.6448	3014.8	0.78519	0.6752
Depth (Year) x Relief (Bank)	4	1471.4	1.468	0.0053*	3957.2	1.1637	0.1533
Residual	272	1002.3			3400.4		
Total	*290*						

Fish Communities

Table 4.5. List of fish species that are most responsible (top 88.6% for density, top 90.5% for biomass) for the dissimilarity between the shallow and upper mesophotic (UM) strata. Results are from two-way SIMPER controlling for relief strata. Values presented are 4th root transformed and species are listed in decreasing order of percent contribution. Members of apex predator families and Pterois volitans *are shown in gray.*

Density Average Dissimilarity = 54.1			
Species	**Shallow Ave. Density**	**UM Ave. Density.**	**% Cont.**
Chromis insolata	1.18	2.66	4.11
Clepticus parrae	1.60	1.68	4.08
Chromis multilineata	2.52	1.38	4.08
Paranthias furcifer	2.34	2.92	3.18
Bodianus pulchellus	0.36	1.41	2.78
Chromis scotti	0.68	1.35	2.72
Stegastes partitus	1.54	0.75	2.70
Kyphosus sectator	0.98	0.53	2.42
Chromis cyanea	0.86	0.98	2.37
Mulloidichthys martinicus	0.14	0.84	2.31
Scarus taeniopterus	0.49	0.74	2.00
Thalassoma bifasciatum	2.29	1.70	1.97
Lutjanus griseus	0.51	0.56	1.95
Stegastes variabilis	0.65	0.58	1.87
Sparisoma atomarium	0.54	0.94	1.84
Carangoides ruber	0.49	0.54	1.80
Stegastes planifrons	1.63	1.06	1.75
Acanthurus coeruleus	1.05	0.94	1.73
Sparisoma viride	0.79	0.57	1.69
Sparisoma aurofrenatum	0.96	1.13	1.69
Holacanthus tricolor	0.32	0.76	1.68
Holocentrus adscensionis	0.07	0.59	1.67
Elacatinus oceanops	0.62	0.26	1.62
Melichthys niger	0.57	0.40	1.55
Pseudupeneus maculatus	0.11	0.74	1.55
Mycteroperca interstitialis	0.62	0.93	1.55
Scarus vetula	0.63	0.27	1.55
Prognathodes aculeatus	0.36	0.68	1.55
Sphyraena barracuda	0.62	0.42	1.44
Canthigaster rostrata	1.47	1.23	1.39
Cephalopholis cruentata	0.46	0.59	1.38
Scarus iseri	0.48	0.26	1.37
Chaetodon sedentarius	0.92	0.98	1.32
Mycteroperca tigris	0.37	0.47	1.31
Halichoeres garnoti	0.31	0.43	1.21
Bodianus rufus	1.55	1.25	1.19
Pomacanthus paru	0.24	0.44	1.15
Lactophrys triqueter	0.40	0.23	1.10
Lutjanus jocu	0.27	0.35	1.07
Balistes vetula	0.06	0.32	1.01
Acanthurus bahianus	0.14	0.40	0.98
Stegastes adustus	0.25	0.25	0.98
Caranx latus	0.25	0.20	0.94

Biomass Average Dissimilarity = 85.9			
Species	**Shallow Ave. Biomass**	**UM Ave. Biomass**	**% Cont.**
Paranthias furcifer	7.21	19.28	21.27
Clepticus parrae	1.87	10.95	9.83
Kyphosus sectator	4.82	2.43	8.19
Mycteroperca bonaci	0.26	6.14	5.81
Sphyraena barracuda	2.04	1.87	5.29
Lutjanus griseus	0.56	3.22	5.18
Mycteroperca interstitialis	0.47	1.79	3.71
Mycteroperca tigris	0.50	1.51	3.65
Lutjanus jocu	0.89	1.45	3.53
Caranx latus	1.84	2.60	3.51
Galeocerdo cuvier	0.43	1.42	3.08
Mulloidichthys martinicus	0.06	2.13	2.35
Mycteroperca venenosa	0.01	1.13	1.96
Sparisoma viride	0.51	0.62	1.95
Pomacanthus paru	0.31	0.59	1.64
Acanthurus coeruleus	0.25	0.71	1.24
Carangoides ruber	0.06	0.73	1.18
Melichthys niger	0.49	0.16	1.13
Epinephelus guttatus	0.05	0.39	1.07
Balistes vetula	0.10	0.30	1.06
Canthidermis sufflamen	0.42	0.26	0.95
Chromis multilineata	0.42	0.17	0.88
Scarus vetula	0.28	0.05	0.69
Carangoides bartholomaei	0.01	0.19	0.65
Caranx hippos	0.36	0.38	0.65

Table 4.5. Continued from previous page

Density				Biomass			
Species	**Shallow Ave. Density**	**UM Ave. Density**	**% Cont.**	**Species**	**Shallow Ave. Biomass**	**UM Ave. Biomass**	**% Cont.**
Mycteroperca bonaci	0.04	0.39	0.92				
Halichoeres maculipinna	0.23	0.24	0.91				
Acanthurus chirurgus	0.32	0.09	0.86				
Chaetodon ocellatus	0.27	0.11	0.83				
Mycteroperca venenosa	0.02	0.35	0.82				
Epinephelus adscensionis	0.07	0.33	0.80				
Calamus nodosus	0.03	0.26	0.80				
Pterois volitans	0.01	0.32	0.80				
Holocentrus rufus	0.12	0.26	0.78				
Epinephelus guttatus	0.07	0.27	0.73				
Halichoeres bivittatus	0.28	0.02	0.71				
Total contribution			**88.6**				**90.5**

The average dissimilarity between the high and low relief strata (based on density) was 50.9%. The top four species responsible for this difference between habitat strata were *C. parrae*, *C. multilineata*, *C. insolata* and *P. furcifer*. Among the economically important species (highlighted in gray Table 4.6), nearly all species had higher densities in high relief habitats compared to low relief except for *M. intersitialis, C. cruentata,* and *E. adscensionis*, a list of fish contributing to the dissimilarity of fish communities by relief strata is provided in Table 4.6.

Fish community structure based on biomass was not influenced to the same degree by depth and relief strata as for density (Table 4.4; Figure 4.12). However, given the effect of depth and relief strata on density, we also examined the effect of these variables on biomass separately from bank and year and did find significant, but weaker (relative to density) effects (two-way ANOSIM: depth strata R=0.198, p=0.001, habitat relief R=0.161, p=0.003). The average dissimilarity between shallow and UM surveys (based on biomass) was 85.9%. The top four species responsible for this difference between depth strata were *P. furcifer*, *C. parrae*, *K. sectator* and *M. bonaci*. Most of the species contributing to the differences between depth strata displayed higher biomass in UM depths, and many of these species were apex predators: e.g., *M. bonaci*, *L. griseus*, *M. interstitialis*, *M. tigris*, *L. jocu*, *C. latus*, *G. cuvier* and *M. venenosa*. In contrast, only six of these 25 species displayed higher biomass in the shallow strata, including *K. sectator*, *Sphyraena barracuda* (Great barracuda) and *Melichthys niger* (black durgon). Compared with community structure based on density, far fewer species contributed to differences between shallow and UM community structure based on biomass (Table 4.5).

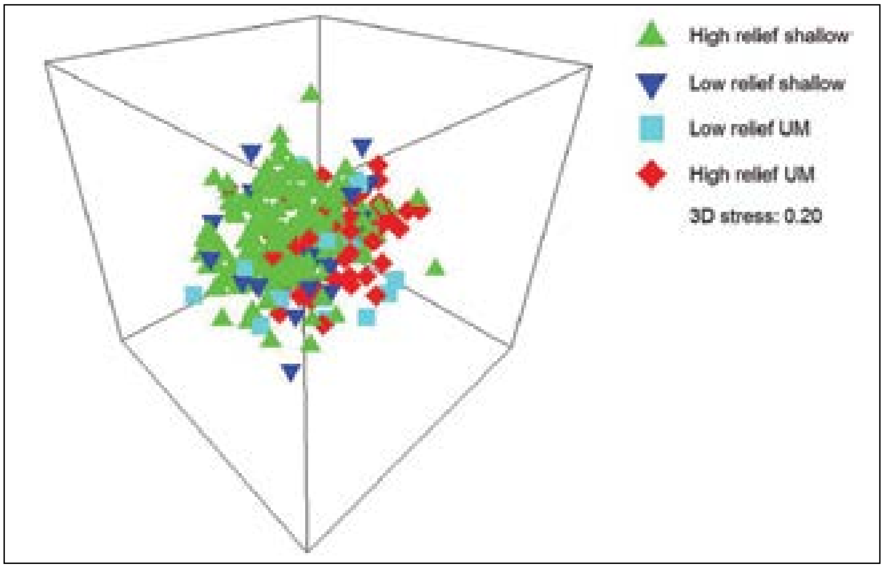


Figure 4.12. Fish community structure at FGBNMS based on biomass as affected by depth and relief strata (Table 4.4).

Fish Communities

Table 4.6. List of fish species that are most responsible (top 88.2% for density, top 90.3% for biomass) for the dissimilarity between the high and low strata. Results are from two-way SIMPER controlling for depth strata. Values presented are 4th root transformed and species are listed in decreasing order of percent contribution. Members of apex predator families are shown in gray.

Density Average Dissimilarity = 50.88				Biomass Average Dissimilarity = 84.9			
Species	**Shallow Ave. Density**	**UM Ave. Density.**	**% Cont.**	**Species**	**Shallow Ave. Biomass**	**UM Ave. Biomass**	**% Cont.**
Clepticus parrae	1.70	1.15	5.15	*Paranthias furcifer*	10.25	8.33	22.79
Chromis multilineata	2.38	1.60	4.33	*Kyphosus spectator*	4.85	1.16	10.78
Chromis insolata	1.34	2.50	4.10	*Sphyraena barracuda*	1.85	2.85	10.11
Paranthias furcifer	2.49	2.37	3.46	*Caranx latus*	1.67	3.88	7.96
Sparisoma atomarium	0.51	1.28	2.89	*Clepticus parrae*	4.54	0.60	5.69
Kyphosus sectator	0.98	0.37	2.74	*Pomacanthus paru*	0.30	0.77	3.17
Chromis scotti	0.80	1.00	2.46	*Mycteroperca interstitialis*	0.72	1.07	3.12
Chromis cyanea	0.89	0.89	2.30	*Lutjanus jocu*	0.99	1.17	3.00
Stegastes planifrons	1.60	0.99	2.30	*Mycteroperca bonaci*	1.02	4.75	2.91
Stegastes variabilis	0.61	0.82	2.21	*Lutjanus griseus*	1.36	0.11	2.64
Stegastes partitus	1.32	1.56	2.20	*Mycteroperca tigris*	0.81	0.29	2.36
Bodianus pulchellus	0.51	1.07	2.12	*Melichthys niger*	0.43	0.31	2.30
Sparisoma viride	0.80	0.41	2.02	*Sparisoma viride*	0.58	0.32	2.28
Acanthurus coeruleus	1.05	0.87	1.98	*Chromis multilineata*	0.31	0.67	1.87
Sparisoma aurofrenatum	1.00	0.98	1.93	*Canthidermis sufflamen*	0.37	0.50	1.71
Thalassoma bifasciatum	2.16	2.15	1.84	*Balistes vetula*	0.13	0.22	1.34
Carangoides ruber	0.52	0.40	1.82	*Acanthurus coeruleus*	0.32	0.53	1.31
Melichthys niger	0.54	0.47	1.81	*Scarus vetula*	0.26	0.05	1.15
Scarus taeniopterus	0.59	0.29	1.76	*Caranx lugubris*	0.20	0.23	0.81
Scarus vetula	0.62	0.15	1.74	*Galeocerdo cuvier*	0.78	0.00	0.81
Elacatinus oceanops	0.58	0.35	1.73	*Mulloidichthys martinicus*	0.58	0.25	0.78
Sphyraena barracuda	0.60	0.44	1.70	*Bodianus rufus*	0.12	0.16	0.73
Mycteroperca interstitialis	0.67	0.79	1.69	*Caranx hippos*	0.35	0.44	0.71
Chaetodon sedentarius	0.96	0.80	1.68				
Lutjanus griseus	0.60	0.09	1.64				
Canthigaster rostrata	1.46	1.15	1.60				
Scarus iseri	0.47	0.23	1.56				
Cephalopholis cruentata	0.48	0.53	1.53				
Halichoeres garnoti	0.31	0.50	1.50				
Pomacanthus paru	0.25	0.46	1.39				
Holacanthus tricolor	0.41	0.50	1.36				
Acanthurus chirurgus	0.27	0.26	1.33				
Mycteroperca tigris	0.42	0.26	1.28				
Prognathodes aculeatus	0.47	0.27	1.25				
Lactophrys triqueter	0.40	0.16	1.19				
Caranx latus	0.25	0.20	1.16				
Bodianus rufus	1.50	1.38	1.09				
Pseudupeneus maculatus	0.19	0.57	0.99				
Lutjanus jocu	0.30	0.23	0.98				
Halichoeres maculipinna	0.24	0.22	0.98				
Mulloidichthys martinicus	0.31	0.25	0.94				
Chaetodon ocellatus	0.26	0.09	0.92				
Stegastes adustus	0.26	0.15	0.89				

Table 4.6. Continued from previous page

Density				Biomass			
Species	Shallow Ave. Density	UM Ave. Density.	% Cont.	Species	Shallow Ave. Biomass	UM Ave. BIomass	% Cont.
Acanthurus bahianus	0.17	0.35	0.88				
Canthidermis sufflamen	0.20	0.13	0.84				
Halichoeres bivittatus	0.25	0.07	0.83				
Holacanthus ciliaris	0.13	0.24	0.76				
Holocentrus rufus	0.14	0.20	0.70				
Stegastes diencaeus	0.14	0.11	0.70				
Caranx lugubris	*0.15*	*0.13*	*0.69*				
Stegastes leucostictus	0.10	0.13	0.66				
Epinephelus adscensionis	0.10	0.26	0.63				
Total contribution			**88.2**				**90.3**

The average dissimilarity between the high and low relief strata (based on biomass) was 84.9%. The top four species responsible for this difference between habitat relief strata were *P. furcifer*, *K. sectator*, *S. barracuda* and *C. latus*. In this case, roughly half (10) of the 23 species contributing to biomass differences between habitat strata showed elevated biomass in high relief habitat. Species here included *P. furcifer*, *K. sectator*, *C. parrae* and *L. griseus*. Some species like *S. barracuda* and *Acanthurus coeruleus* displayed higher density in high relief habitats but higher biomass in low relief habitats (consistent with smaller bodied fishes showing affinity for structurally complex habitat, DeMartini and Anderson, 2007; Harborne et al., 2012b), while other species, like *M. interstitialis* and *M. tigris,* showed both higher abundance and biomass in low or high relief habitat, respectively. A list of fish contributing to the dissimilarity of fish communities by habitat relief is provided in Table 4.6.

Depth, measured as actual site depth, was the most important of nine continuous variables in explaining fish community structure based on density and biomass (global BEST analysis, density: $\rho = 0.485$, $p = 0.001$; biomass: $\rho = 0.389$, $p = 0.001$; Table 4.7). Three main depth category clusters were evident for fish community structure (based on density) with the LINKTREE procedure when all sites were considered. The largest amount of separation among the groups at 82% occurred for sites less than 36.5 m and greater than 36.8 m; a second significant break in community structure was at depths less than 30.9 m and greater than 31.5 m, roughly equivalent to the shallow and UM depth designations, yielding three depth categories 18-30.9, 31.5-36.5 and 36.8-45.0 m. Similarly, the largest break (80% separation) identified from the LINKTREE procedure for fish community structure (based on biomass) resulted in two major depth categories, 18-32.8 and 32.9-45.0 m, again very similar to the depth strata designations. For UM strata only, depth, cover of rubble, sand, algae and sponge had the highest correlation with the fish community structure based on density (BEST, $\rho = 0.214$, $p = 0.006$). For fish community structure based on biomass; depth, algae and sponge variables were the most

Table 4.7. Results of the global BEST (999 permutations) and follow up LINKTREE analysis examining the role of nine continuous variables (site depth, rugosity, percent cover of hard substrate, rubble, sand, algae, hard coral, hydrocoral and sponges) in explaining fish community structure based on density and biomass.

	Density			Biomass		
Sites	Important variable(s)	Spearman correlation/	p-value	Important variable(s)	Spearman correlation/	p-value
All	depth	48.5%	0.001	depth	39.0%	0.001
Upper Mesophotic	depth, rubble, sand, algae, sponges	21.4%	0.006	depth, algae, sponges	18.6%	0.001
Shallow	depth, rugosity, hard substrate, rubble, algae	29.8%	0.001	depth, rugosity, hard substrate, rubble, hard corals	25.3%	0.001

important ($\rho = 0.186$, $p = 0.001$). The variables most responsible for describing the shallow community structure based on density were depth, rugosity, cover of hard substrate, rubble and algae ($\rho = 0.298$, $p = 0.001$), while depth, rugosity, hard substrate, rubble and hard coral cover were the most important variables in explaining the shallow community structure based on biomass ($\rho = 0.253$, $p = 0.001$).

Although the overall story of the fish community is a complex one, it is clear that depth (shallow versus UM) and relief (high versus low) strata have a large influence on fish community structure based on both density and biomass. Variables year and bank did not significantly affect fish community structure. When taking a multivariate perspective, the single most important factor in structuring FGBNMS fish communities is depth. In addition, depth strata designated by shallow and UM was a consistently significant variable in describing differences in the fish community for both biomass and density (Figures 4.12 and 4.10).

4.3.3. Size Frequency

Mean fish densities were greatest in the smaller length categories and generally declined with increasing fish size, consistent with Caldow et al. (2009; Figure 4.13). Mean fish density per site was 414.0 ± 26.0/100 m^2, where all size classes were combined, and primarily comprised of fish less than 10 cm (251.8 ± 15.7/100 m^2). Mean site density within the shallow stratum (372.9 ± 23.1/100 m^2) was similar to a previous study (311/100 m^2; Caldow et al., 2009) and significantly lower than UM density (593.2 ± 83.4/100 m^2; t test, $t = 2.544$, $p = 0.013$). For sizes classes less than 35 cm FL (98% of all fish recorded), UM density exceeded shallow stratum where FL was greater than 15 cm (15-20 cm: $Z = 2.81$, $p = 0.0049$; 20-25 cm: $Z = 3.25$, $p = 0.001$; 25-30 cm: $Z = 4.688$, $p < 0.0001$; 30-35 cm, $Z = 3.667$, $p = 0.002$); for size classes <15 cm, densities were similar between depth strata, however, UM stratum density was consistently higher than shallow stratum. Densities by bank were similar for all size classes except 20-25 cm and 30-35 cm, where WB densities were greater than EB for both ($Z = 2.711$, $p = 0.0067$; $Z = 2.072$, $p = 0.038$, respectively). Lower relief sites had higher fish densities of the smallest fish (0-5 cm size class; 264.3 ± 51.3/100 m^2) than high relief sites (174.1 ± 12.4/100 m^2; $Z = 2.029$, $p = 0.0425$), consistent with Caldow et al. (2009). However, high relief sites had significantly higher fish densities in the 10-15 cm size class (38.03 ± 5.7/100 m^2) compared to low relief (37.3 ± 10.3/100 m^2; $Z = -1.962$, $p = 0.0498$). The density peak in the 0-10 cm size class in the EB shallow low relief stratum was due to a high abundance of *C. parrae,* comprising 32.5% of the total fish abundance within this stratum. The high density within the 10-30 cm size classes of the WB high relief UM stratum was largely due to two species, *P. furcifer* and *C. parrae*. Additional size frequency plots are provided in the following sections for highlighted species.

4.3.4. Apex Predators and Large Fishes

Human impacts to terrestrial and marine communities are widespread and typically begin with the local extirpation of large-bodied animals. Large-bodied species play an important ecological role (Pandolfi et al., 2005), and many are key species that are important for maintaining long-term ecosystem stability (Bellwood et al., 2003; Sadovy et al., 2003). Large predators can shape the number, distribution and behavior of their prey, while large herbivores can act as ecological engineers by shaping the structure and species composition of benthic plant communities (Morrison et al., 2007; McCauley et al., 2010). Removal of large-bodied animals can have immediate short and cascading long-term effects across multiple trophic levels (Berger et al., 2001; Baum and Worm, 2009). For large-bodied species that are generally first to be extirpated following human contact (Jackson et al., 2001; Morrison et al., 2007), no-take protected areas and remote locations relatively free of human impact can provide important baselines of ecosystem function, goals for restoration efforts, and the persistence of historical conditions that can reveal remarkable behavioral and ecological processes (Muñoz et al., 2012). For example, recent comparisons of remote and/or protected coral reefs versus impacted sites suggest remote systems are dominated by apex predators (Stevenson et al., 2007; DeMartini et al., 2008; Sandin et al., 2008; Friedlander et al., 2010), and this pattern can be most pronounced at greater depths (Friedlander et al., 2010).

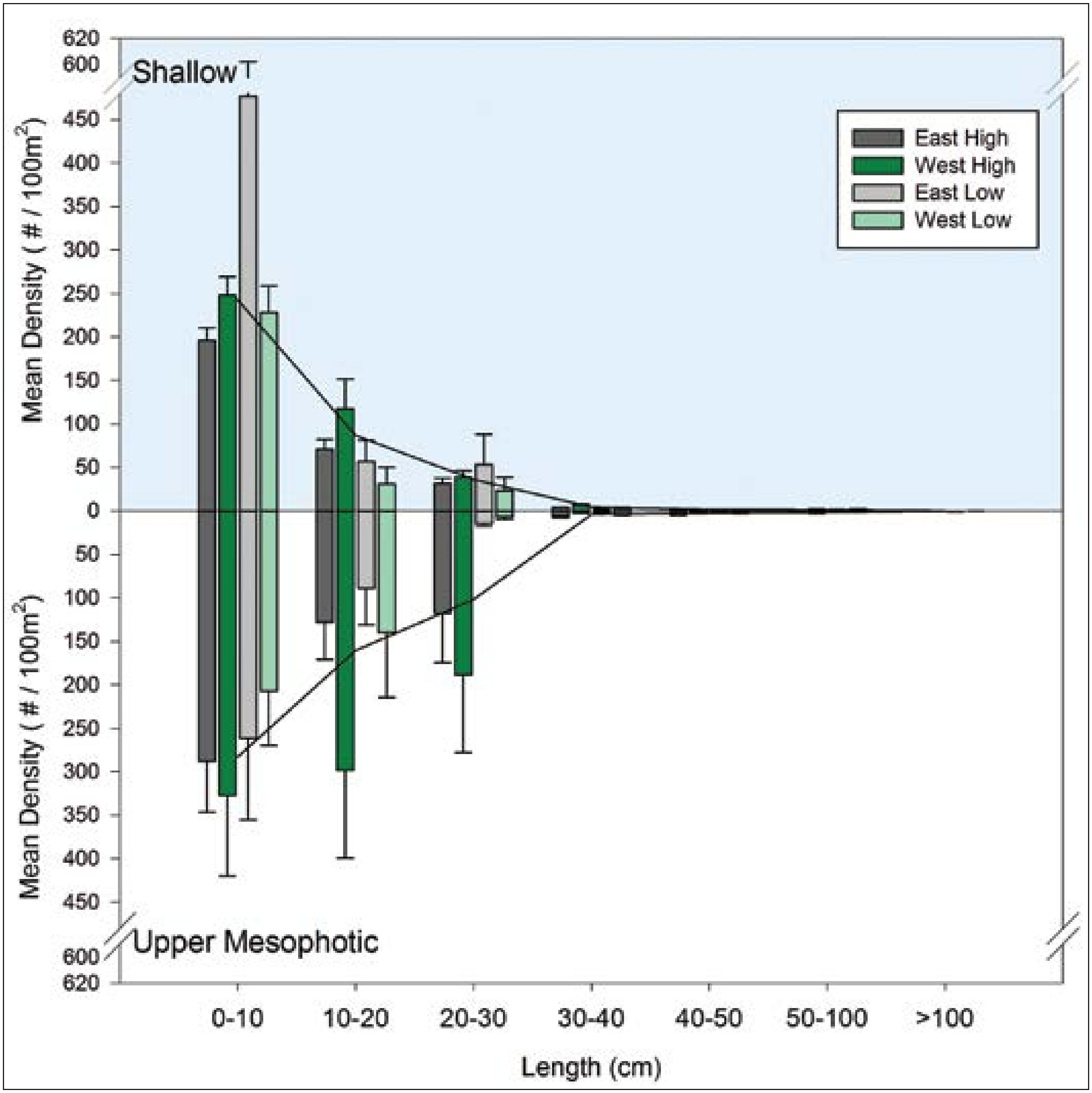


Figure 4.13. Mean fish density per size class (cm FL) for all species combined by strata from diver surveys (2010-2012). Solid line represents overall mean density per size class.

At FGBNMS, large fish ≥ 50 cm FL were observed from the families Balistidae (triggerfishes), Carangidae (jacks), Carcharhinidae (sharks), Labridae (wrasses and parrotfishes), Lutjanidae (snappers), Muraenidae (moray eels), Myliobatidae (manta rays), Serranidae (groupers), and Sphyraenidae (barracudas) from 136 shallow (73 EB, 63 WB) and 54 UM sites (35 EB, 19 WB). A total of 756 large fish were observed at FGBNMS, and the bulk (96%) of these large fish could be considered apex predators (Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae ≥50 cm FL; Table 4.8; Stevenson et al., 2007). Forty large fish ≥100 cm FL were observed on 34 sites (including fish seen off transect on five sites), and were nearly exclusively apex predators, except for two (5%) *M. birostris*. The remaining ≥100 cm FL fish consisted of 7.5% Carangidae (*Caranx hippos* [crevalle jack], and *C. latus*), 12.5% Carcharhinidae (*Carcharhinus perezii* (reef shark), *Carcharhinus plumbeus* (sandbar shark), G. cuvier), 45% Serranidae (mostly *M. bonaci* and one *M. interstitialis*), and 30% *S. barracuda*. Twenty-three of these sites were on the EB (nine shallow and 14 UM), while eleven were on the WB (three shallow and eight UM).

Species composition of apex predators differed significantly between shallow (133 sites) and UM (54) sites (Figure 4.14). Apex predators overall were more numerous in shallow depths (487 fish ≥50 cm FL versus 241 in the UM), primarily the result of greater numbers of carangids and sphyraenids at shallow sites, but these larger numbers also reflect the differing sample size between depth strata. When accounting for sampling effort, apex predators were encountered on a greater proportion of UM (82%) than shallow sites (59%). Despite the disproportional sampling effort in shallow depths, greater numbers of lutjanids, and more than twice as many serranids ≥50 cm FL, were observed on UM strata compared to shallow (78 snappers and 110 groupers from UM sites, versus 54 snappers and 50 groupers from shallow sites (Figure 4.15).

*Table 4.8. Apex predators observed at FGBNMS from 2010-2012. *Species that were not observed ≥50 cm FL but that were included in the analysis where size was not considered.*

Carangoides bartholomaei	*Lutjanus griseus*
*Caranx crysos**	*Lutjanus mahogoni*
Caranx hippos	*Cephalopholis cruentata**
Caranx latus	*Dermatolepis inermis*
Caranx lugubris	*Epinephelus adscensionis*
*Carangoides ruber**	*Epinephelus guttatus*
Seriola dumerili	*Mycteroperca bonaci*
Carcharhinus perezi	*Mycteroperca interstitialis*
Carcharhinus plumbeus	*Mycteroperca phenax*
Carcharhinus species	*Mycteroperca tigris*
Galeocerdo cuvier	*Mycteroperca venenosa*
*Lutjanus cyanopterus**	*Sphyraena barracuda*
Lutjanus jocu	

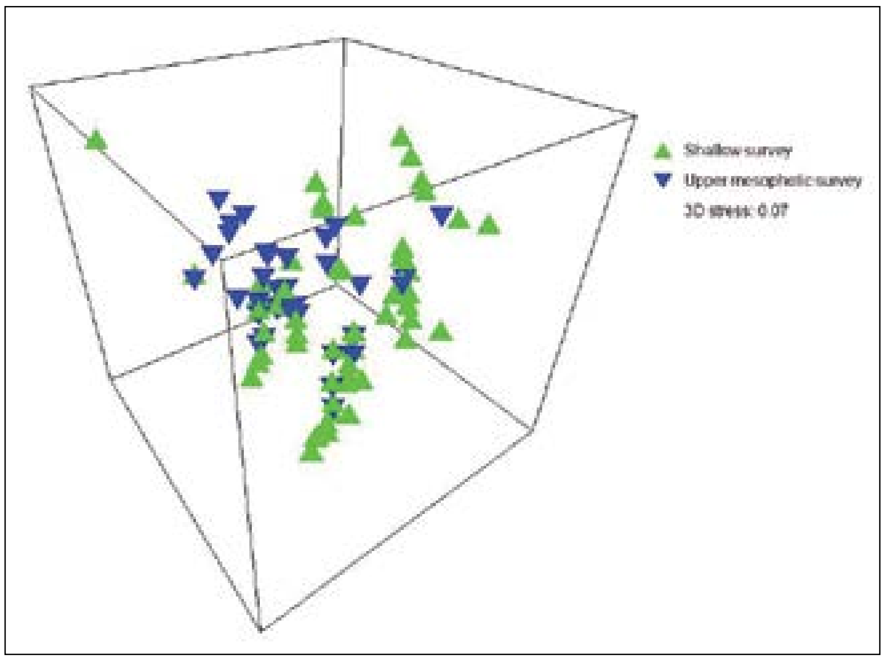


Figure 4.14. Apex predator species (from Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae families ≥ 50 cm FL) community composition at FGBNMS differed between shallow and upper mesophotic sites (two-way ANOSIM: depth strata R=0.226, p=0.001; habitat relief R=0.024, p=0.342).

Overall biomass of apex predators totaled 2664.02 kg, and ranged from 26% (when only fish ≥ 50 cm FL were considered) to 34% (3422.40 kg including all size classes) of total fish biomass (10,207.3 kg). Mean (±SE) apex predator biomass per site (100 m^2) was 9.15 ± 1.04 kg/100 m^2, or 0.915 metric tons/hectare (MT/ha). Apex predator/total fish biomass of 26-34% approximates the result of 36% reported by Caldow et al. (2009). The lower value from the present study may result from the larger sample size over a broader depth range of the current study (291 sites to 45 m depth versus 73 sites <33.5 m depth; Caldow et al., 2009), where a larger sample size would reduce the effect of large fish (>50 cm FL) on total biomass. Caldow et al. (2009) compared apex predator/total fish biomass at FGBNMS with sites at similar depths in the US Virgin Islands that experience fishing pressure/human impacts (Figure 4.16). In general, these sites show a lower proportion of apex/total fish biomass than FGBNMS (36%), compared to St. John (20%), Puerto Rico (16%), and St. Croix (6%; Caldow et al. 2009). Other locations relatively free from

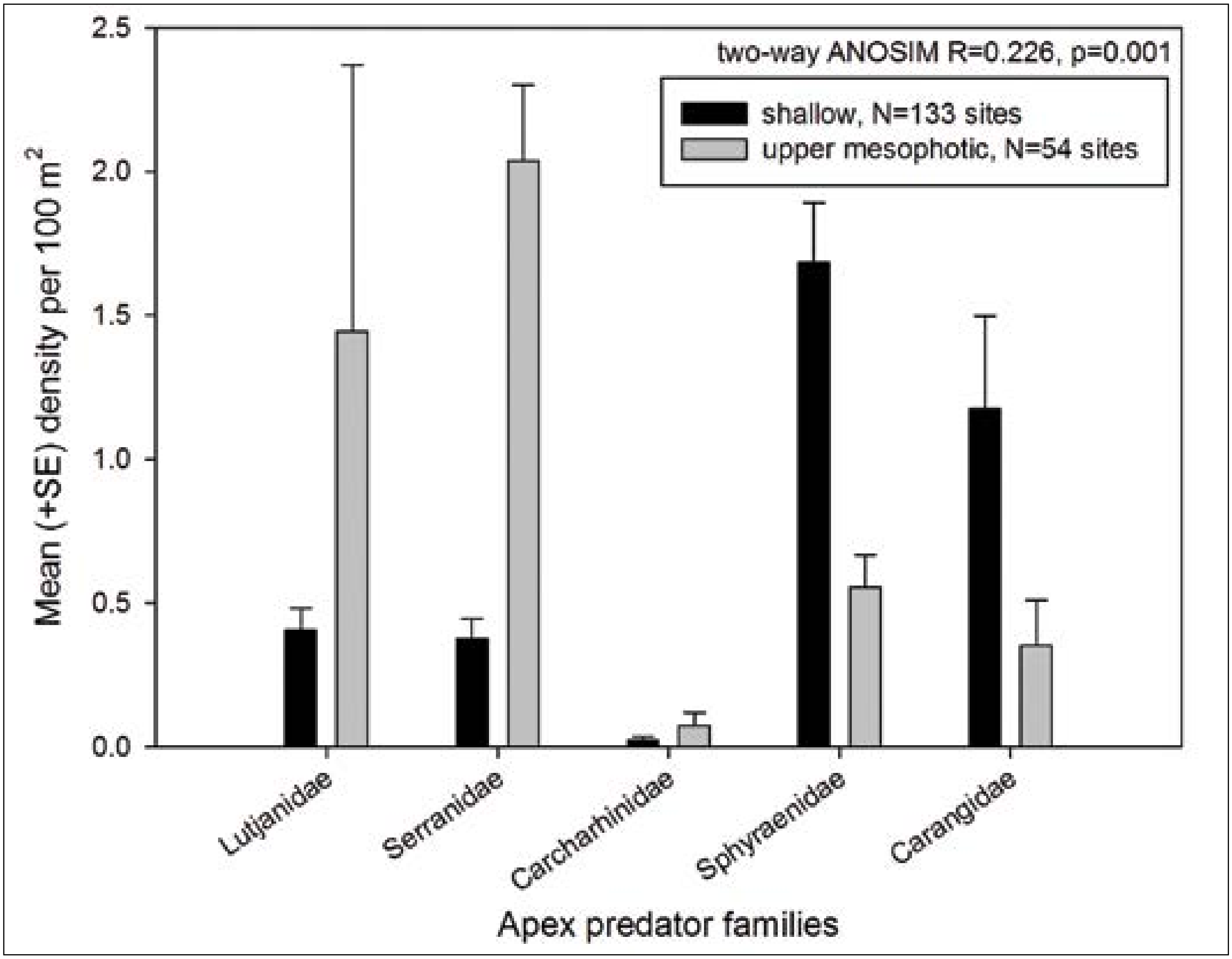


Figure 4.15. Apex predator species (from Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae families ≥50 cm FL) composition at FGBNMS differed significantly between shallow and upper mesophotic sites (two-way ANOSIM R=0.226, p=0.001).

human impact show similar patterns of elevated apex predator/total fish biomass (Newman et al., 2006; Sandin et al., 2008; Friedlander et al., 2010), and apex predator biomass (0.92 MT/ha) at FGBNMS is in the range of reports from fully protected marine reserves in Mexico and Cuba (0.93, 0.77-2.18 MT/ha, respectively; Newman et al., 2006).

Overall biomass of apex predators was greatest for serranids, then distributed approximately equally between carangids and sphyraenids, followed by lesser but approximately equal percent contributions from lutjanids and carcharhinids (Figure 4.17).

*Figure 4.16. Great barracuda (*Sphyraena barracuda*) is an example of an apex predator recorded at FGBNMS, U.S. Virgin Islands and Puerto Rico. Photo: M. Winfield (UNCW)*

Individual carcharhinids contributed much more to overall biomass, however, with individual serranids and carangids also important (Figure 4.17). Although sphyraenids were abundant in shallow depths, their individual contribution to biomass was less than that of serranids and lutjanids. The larger sizes (and biomass contribution per individual) of many fishes in UM depths (see below), particularly serranids and lutjanids, resulted in dramatic differences in biomass of apex predators between depth strata, with significantly greater mean apex predator biomass in UM depths (mean ± SE; 18.5 ± 3.2 versus 6.4 ± 0.9 kg/100 m^2, t-test, $t = -5.093$, $p < 0.001$; Figure 4.18A). In contrast, apex predator biomass between banks was not significantly different (Figure 4.18B), though UM sites on EB tended to support greater apex predator biomass. Friedlander et al. (2010) also showed significantly greater biomass of apex predators at 20 m depths of a pristine atoll. compared to 5 m depths, and numerous species of reef fish are known to make ontogenetic migrations to deeper habitats (Lindeman et al., 2000).

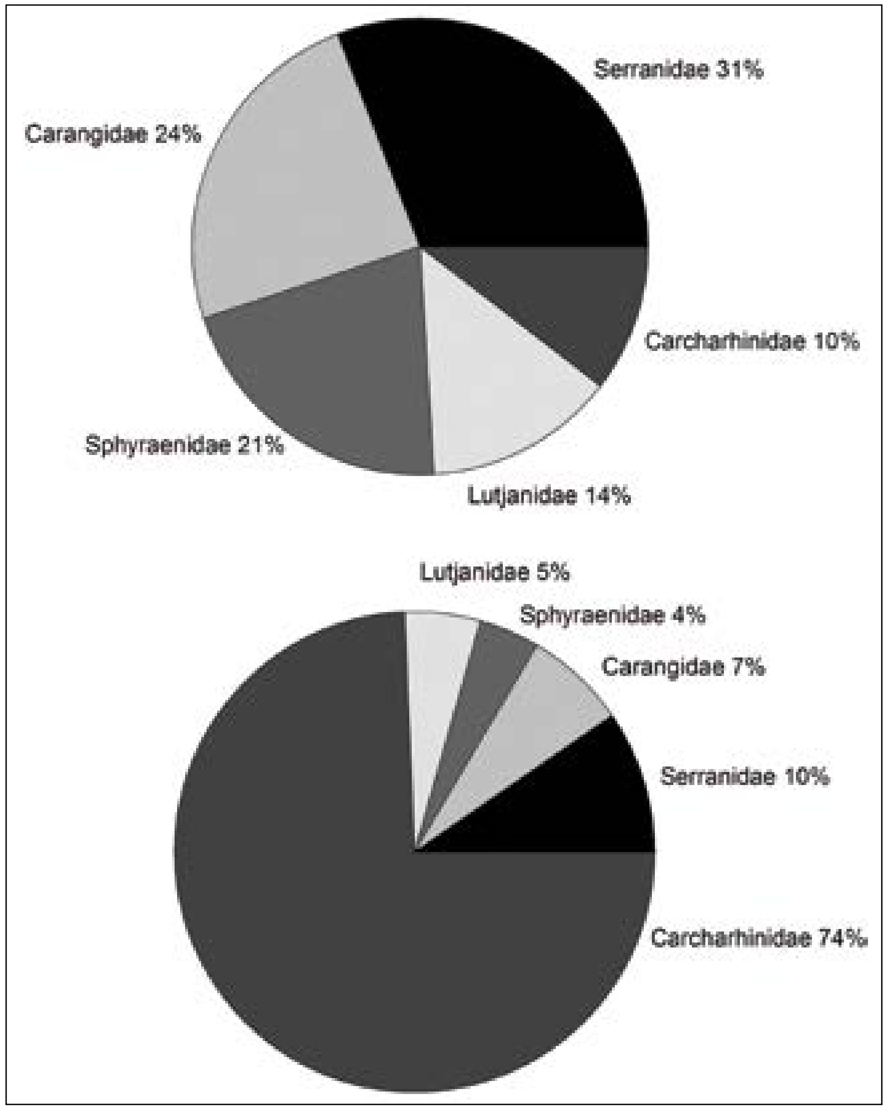


Figure 4.17. Apex predator species (from Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae families ≥50 cm FL) biomass at FGBNMS. Top: Overall percent contribution to apex predator biomass. Bottom: Percent contribution to overall apex predator biomass per individual.

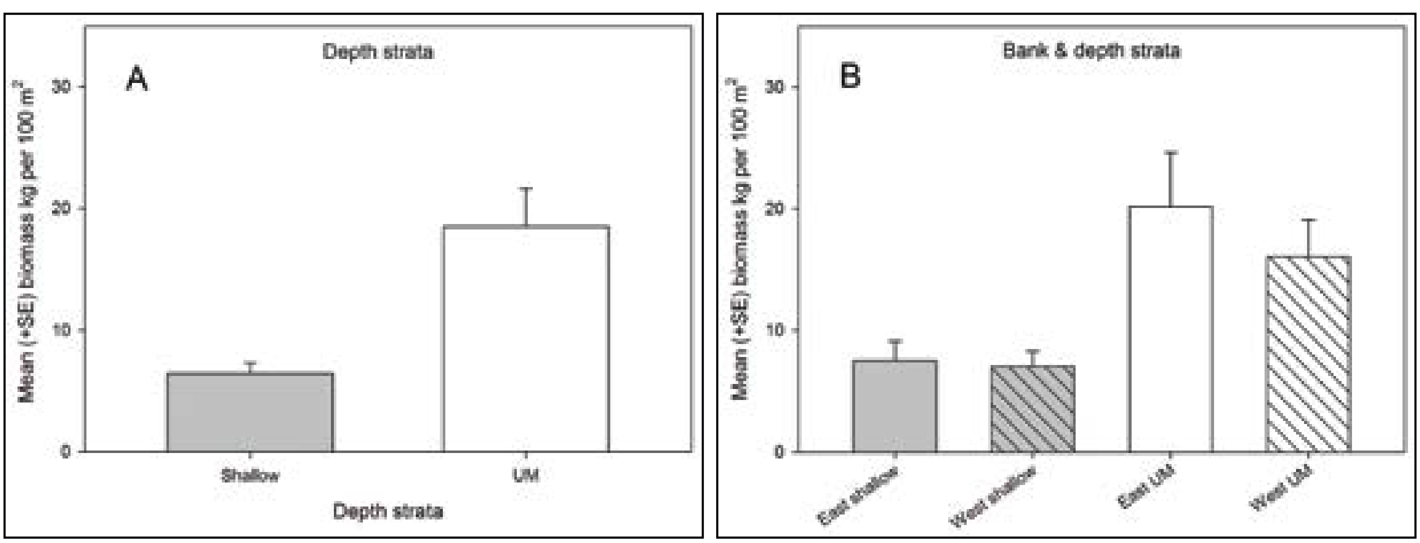


Figure 4.18. Apex predator (Carangidae, Carcharhinidae, Lutjanidae, Serranidae, and Sphyraenidae ≥50 cm FL) biomass at FGBNMS. A) differs significantly between shallow (n = 225) and upper mesophotic (n = 66) sites (t-test, t = -5.093, p <0.001) but B) not between banks.

When only benthic apex predators were considered (lutjanids/serranids), this group was found on significantly more UM strata compared to shallow ($\chi^2 = 38.966$, $p < 0.001$). Within the UM zone, the benthic composition on those sites with apex predators (lutjanids/serranids) was distinct from sites devoid of these fishes (Figure 4.19). Benthic apex predators were more often associated with sites characterized by higher relief (90.4 ± 11.9, n = 51 versus 68.3 ± 16.8 cm, n = 15), greater percent cover of *Orbicella franksi* (17.2 ± 2.9 versus 9.5 ± 3.0 %; previously known as *Montastraea franksi*), a mounding coral species, and lower percent cover of *Madracis auretenra* (1.0 ± 0.9% versus 11.3 ± 5.4%; previously known as *Madracis mirabilis*), common to low relief habitats of FGBNMS. This finding is consistent with previous studies that found large piscivores (including serranids) positively associated with greater structural complexity or rugosity (e.g., Connell and Kingsford, 1998; Sluka et al., 1998).

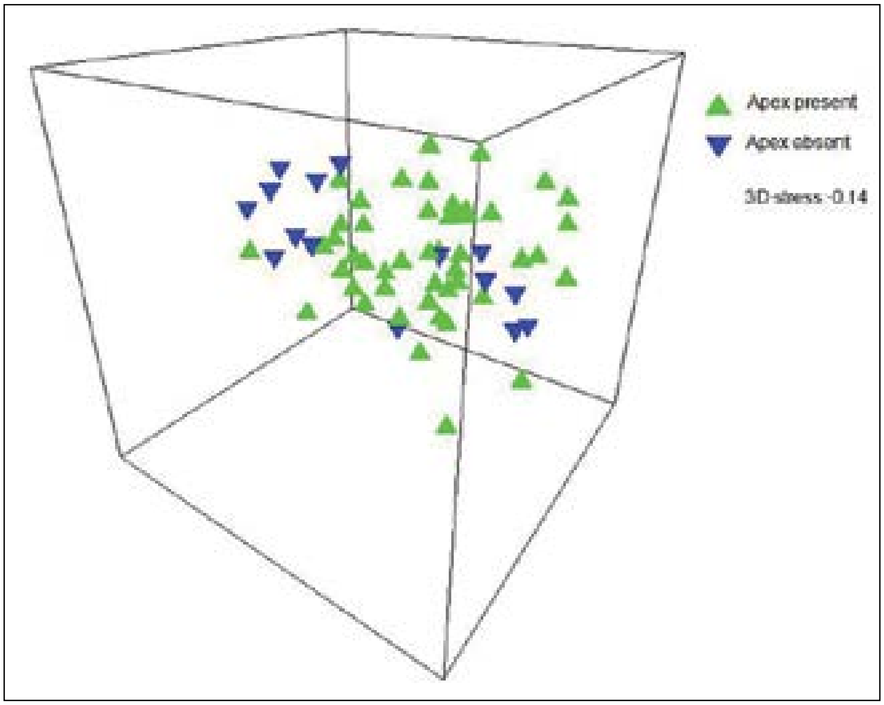


Figure 4.19. Benthic community composition on FGBNMS upper mesophotic sites supporting snapper/grouper ≥50 cm FL differed from those sites devoid of these apex predators (one-way ANOSIM R = 0.192, p = 0.007).

Given the importance of apex predators to trophic flow in marine communities (Duffy, 2003; Byrnes et al., 2007) and the association of apex predators with high coral cover and reef resilience (Knowlton and Jackson, 2008; Sandin et al., 2008; Sandin et al., 2010; Singh et al., 2012), the significantly greater biomass of apex predators on the UM stratum warrants continued study and conservation of fishes and habitats in this zone. Additionally, apex predator biomass on the UM stratum was dominated by serranids, of which many species (except in the case of seasonal reproductive migrations; McGovern et al., 2005) are known to exhibit relatively high site fidelity (e.g., Afonso et al., 2011). This suggests that specific UM sites may be particularly important to the conservation of apex predators there, in contrast to the high vagility (e.g., Carangidae and Sphyraenidae; Afonso et al., 2009; O'Toole et al., 2011) of apex predators in shallow depths at FGBNMS, where the spatial conservation of habitat and predator biomass may be relatively decoupled and more difficult to manage.

4.3.5. Trophic Groups

Fish communities of both EB and WB were dominated by planktivores (51% overall) and invertivores (33% overall; Table 4.9) based on density. Planktivore/zooplanktivores were identified as the dominant trophic groups in previous studies (Pattengill et al., 1997; Caldow et al., 2009). Another study identified herbivores as the dominant trophic group, but only examined herbivores and carnivores (Precht et al., 2006).

*Table 4.9. Number of species and percent of total fish density for all strata combined and by depth strata (shallow and upper mesophotic [UM] sites) by trophic group. * Zooplanktivores were not included in further trophic group analyses as they were a minor portion of total fish density.*

Trophic group	Species #	% of total	% total Shallow	% total UM
Herbivores	20	11.03	12.09	8.75
Invertivores	62	33.21	39.61	19.51
Piscivores	32	4.66	5.70	2.44
Planktivores	12	51.08	42.60	69.28
Zooplanktivore*	2	0.01	0.00	0.03

Fish Communities

Higher invertivore and piscivore densities occurred on WB (Z = 2.09, p = 0.036; Z = 3.89, p = 0.0001, respectively), while planktivore and herbivore densities were similar by bank. A detailed list of fish species and mean density by trophic group is provided in Appendix C. Planktivores comprised more of total fish density within the UM stratum than in the shallow surveys (Figure 4.20; Z = 5.54, p < 0.0001), primarily due to the abundant *P. furcifer*. The opposite pattern was found for invertivores and herbivores, with a larger percent of the total density in the shallow depth strata (Figure 4.20; Z = -3.484, p = 0.0005, Z = -2.37, p = 0.0177, respectively). When depth strata were combined, piscivores were more abundant in high relief habitats, consistent with this finding for large benthic apex predators in the UM zone (see section 4.3.4).

While piscivores made up a small percentage of the community in number (4.6% overall; Figure 4.20), they were second in total biomass (30% overall; Figure 4.21). In some instances, less numerically abundant species dominated total biomass. For example, the peak in total biomass within the EB deep low relief stratum (Figure 4.21) is largely due to *M. bonaci* and *C. latus*, comprising 25% and 14%, respectively, of the total piscivore biomass for this stratum. Caldow et al. (2009) reported a higher ratio of piscivore biomass to total biomass

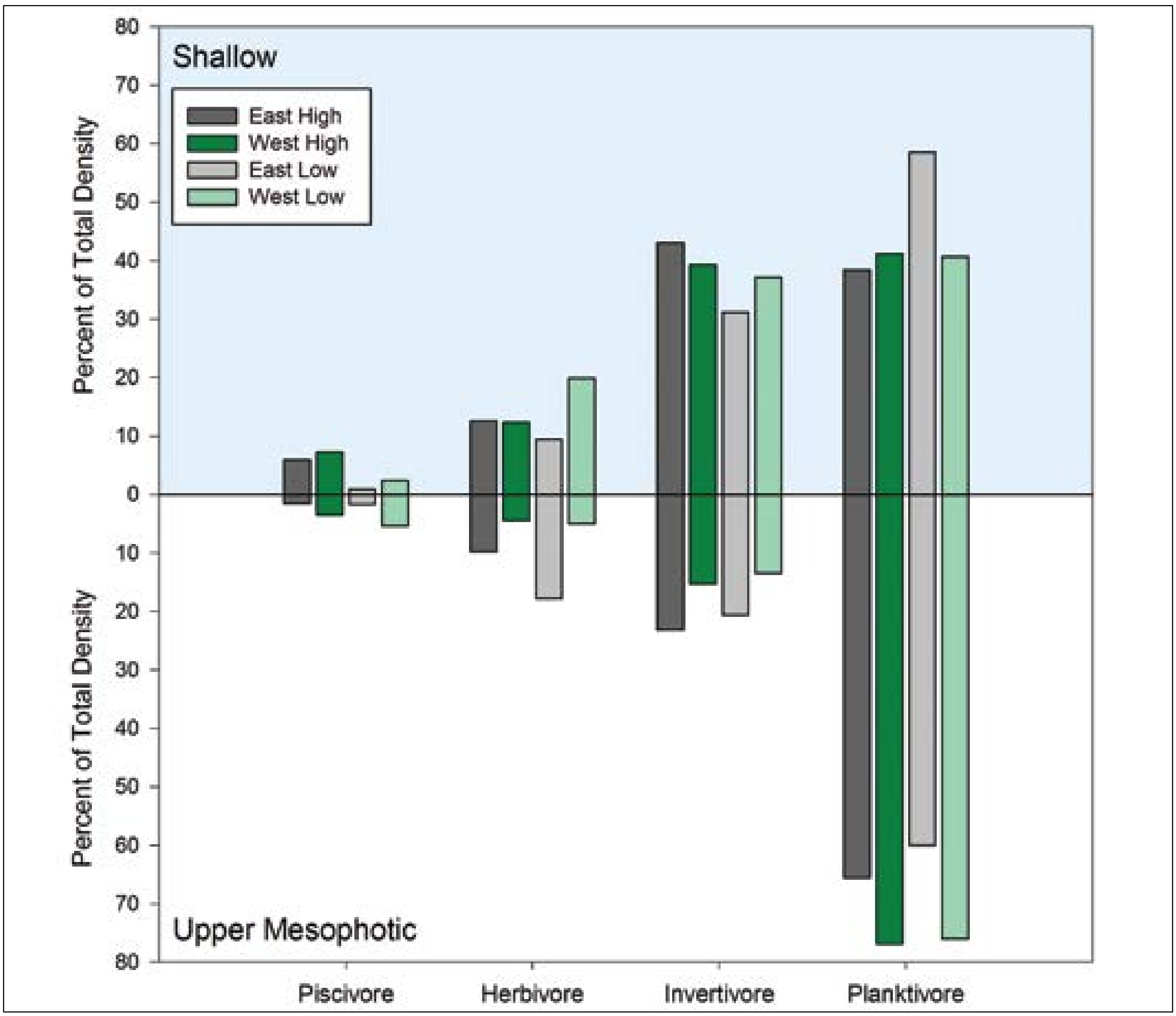


Figure 4.20. Percent of total mean density for four dominant trophic groups by strata for diver surveys (2010-2012).

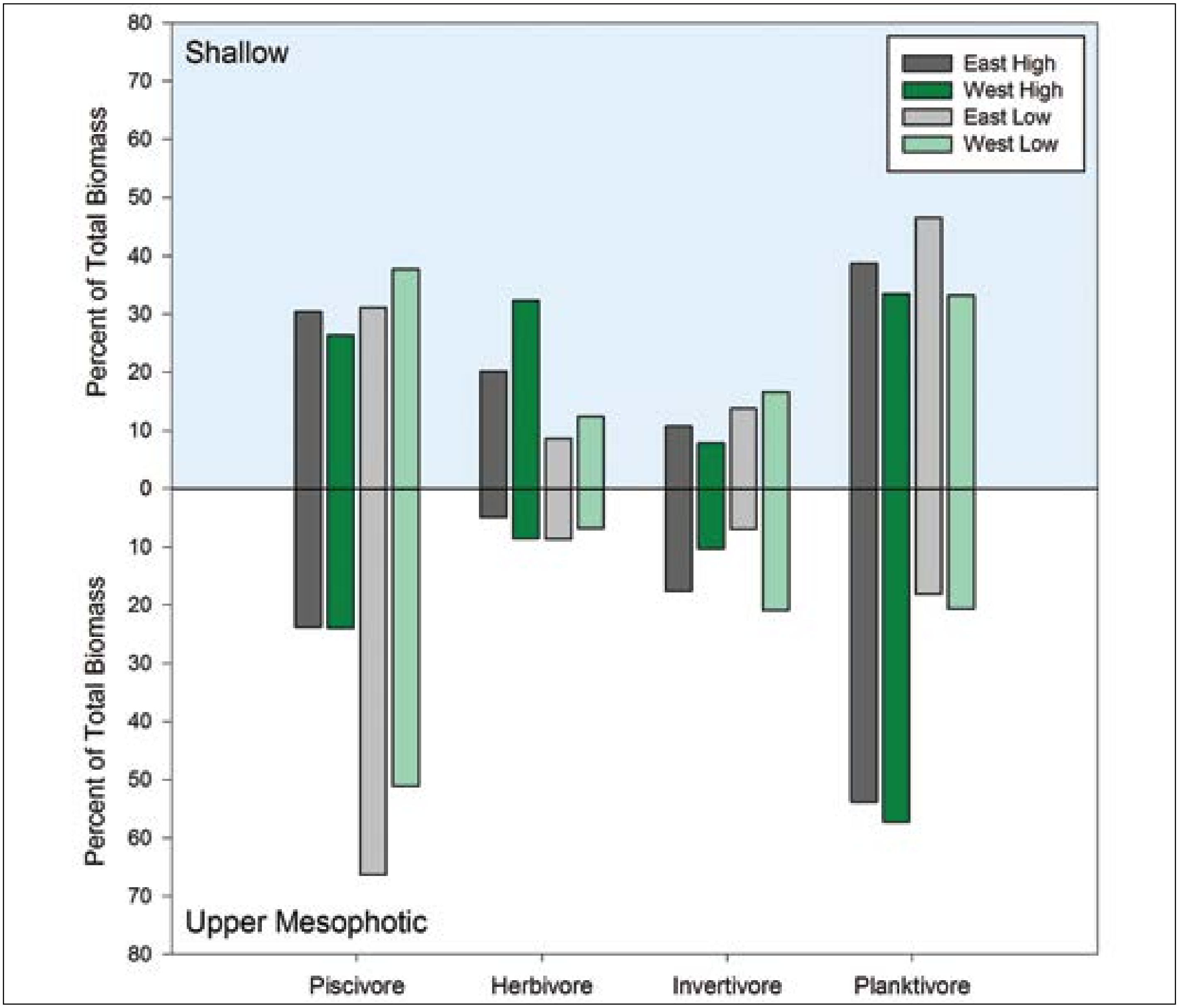


Figure 4.21. Percent of total mean biomass for four dominant trophic groups by strata for diver surveys (2010-2012).

(46%). This difference may be an artifact of sample size. This study had a larger sample size (291 versus 73 sites; Caldow et al., 2009), which reduces the effect of large fish (>50 cm total length [TL]) on total biomass. Further, more planktivores were observed in the extended depths of this study (33.5 – 45m) than in the depth range of the previous study (<33.5 m).

When all sites were combined, all trophic group densities, except piscivores, were strongly correlated with depth. All significant correlations with depth were negative, except for planktivores which were positively correlated (Figure 4.22), indicating an increase in planktivore density with increasing depth (Figure 4.20). Strong correlations with depth are consistent with BEST analyses identifying depth as the primary parameter structuring fish communities when all sites were combined (Table 4.7). Correlations between density and cover of abiotic groups were similar for herbivores and invertivores. Both were strongly negatively correlated with depth, sand and rubble cover, and strongly positively correlated with rugosity and rock cover, although only invertivores were strongly correlated with rugosity (Figure 4.22). Fish densities for the other two trophic groups, piscivores and planktivores, showed similar correlation trends with some benthic parameters (rugosity, depth, sand, rubble, rock cover, total biotic cover and hydrocoral cover), although only planktivores were significant for some

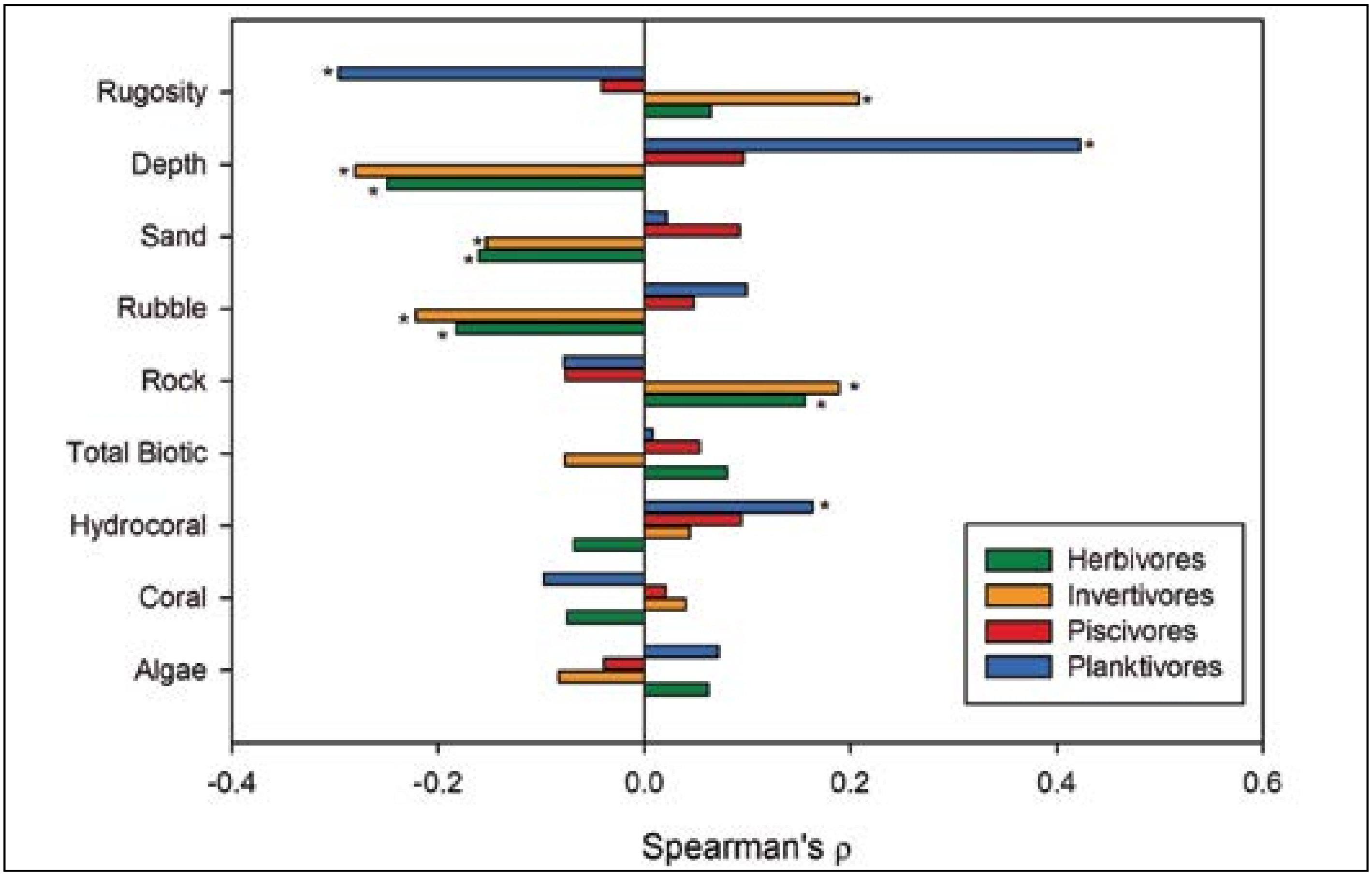


*Figure 4.22. Spearman's rho (ρ) correlations between fish density by trophic groups and benthic parameters where all strata were combined. * Indicates significance probability where α = 0.05.*

parameters (Figure 4.22). Correlation patterns observed for planktivores was largely driven by the abundant *P. furcifer* and *C. insolata* (Figure 4.23). These two species comprised 62.2% of all planktivores by density. *Clepticus parrae*, while also numerically abundant (31.4% of all planktivores), were not significantly correlated with rugosity, depth, or hydrocorals.

*Figure 4.23. Atlantic creolefish (*Paranthias furcifer*), an example of a planktivorous fish in FGBNMS. Photo: M. Winfield (UNCW)*

Correlation patterns from this study's shallow surveys varied somewhat from those reported in a previous study across identical depths (Caldow et al., 2009). For example, all trophic groups were negatively correlated with depth in the previous study; here planktivores were strongly positively correlated with depth (Figure 4.24), reflecting the increased densities of planktivores (primarily driven by *P. furcifer*) within the UM stratum (Figures 4.20 and 4.30 left). Within each depth strata, herbivores and invertivores were negatively correlated with depth (Figure 4.24), reflecting their increased densities in the shallow zone (Figure 4.20), consistent with BEST analyses identifying depth as a component structuring fish communities within each depth strata (Table 4.7). Within the shallow strata where herbivores were more abundant, they were positively correlated with algae (Figure 4.24). The weakly negative correlation between herbivores and algae in the UM may reflect the lower abundance of herbivores and increased abundance of algae in that zone (Figures 4.24 and 3.38). Within the UM stratum, piscivores were positively correlated with sand cover, suggesting an increase in density at sites

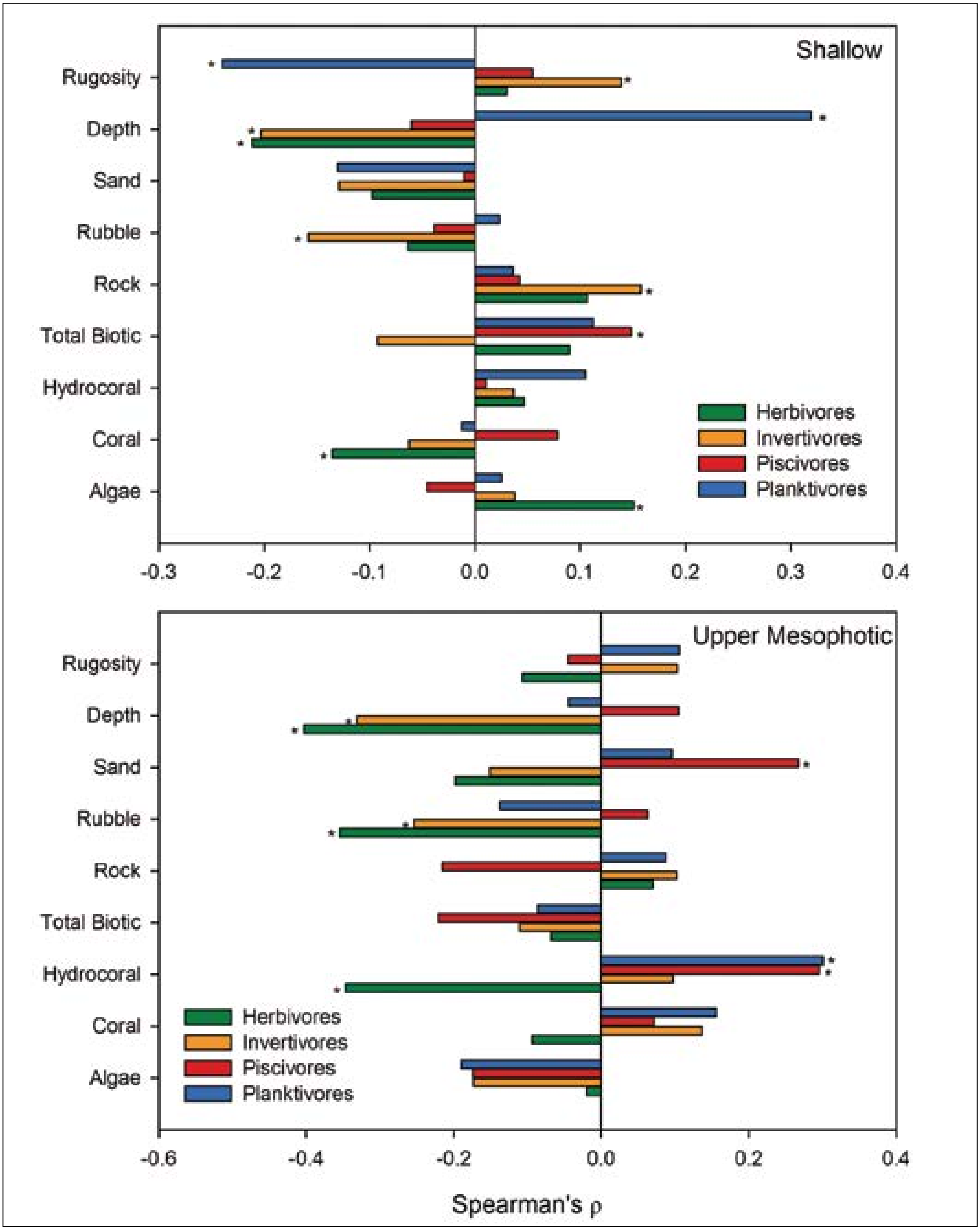


*Figure 4.24. Spearman's rho (ρ) correlations between fish density by trophic groups and benthic parameters by depth strata. * indicates significance probability where α = 0.05.*

with large sand areas, such as reef-sand interfaces, that were abundant in the UM (Figure 3.5). The negative correlations between rubble cover with invertivores and herbivores in the shallow strata were magnified in the UM stratum, with strong negative correlations for both trophic groups (Figure 4.24). This may reflect a preference for greater rugosity than rubble affords, as well as the fact that both trophic groups were more abundant in shallow depths while rubble was more abundant in the UM.

4.3.6. Taxonomic Groups

Six fish families and one subfamily were selected for additional in-depth analyses because of their ecological, commercial, or recreational importance to FGBNMS (Serranidae, Lutjanidae, Carangidae, Scarinae [previously Scaridae; Westneat and Alfaro, 2005], Acanthuridae, Pomacentridae and Scorpaenidae). When all strata were

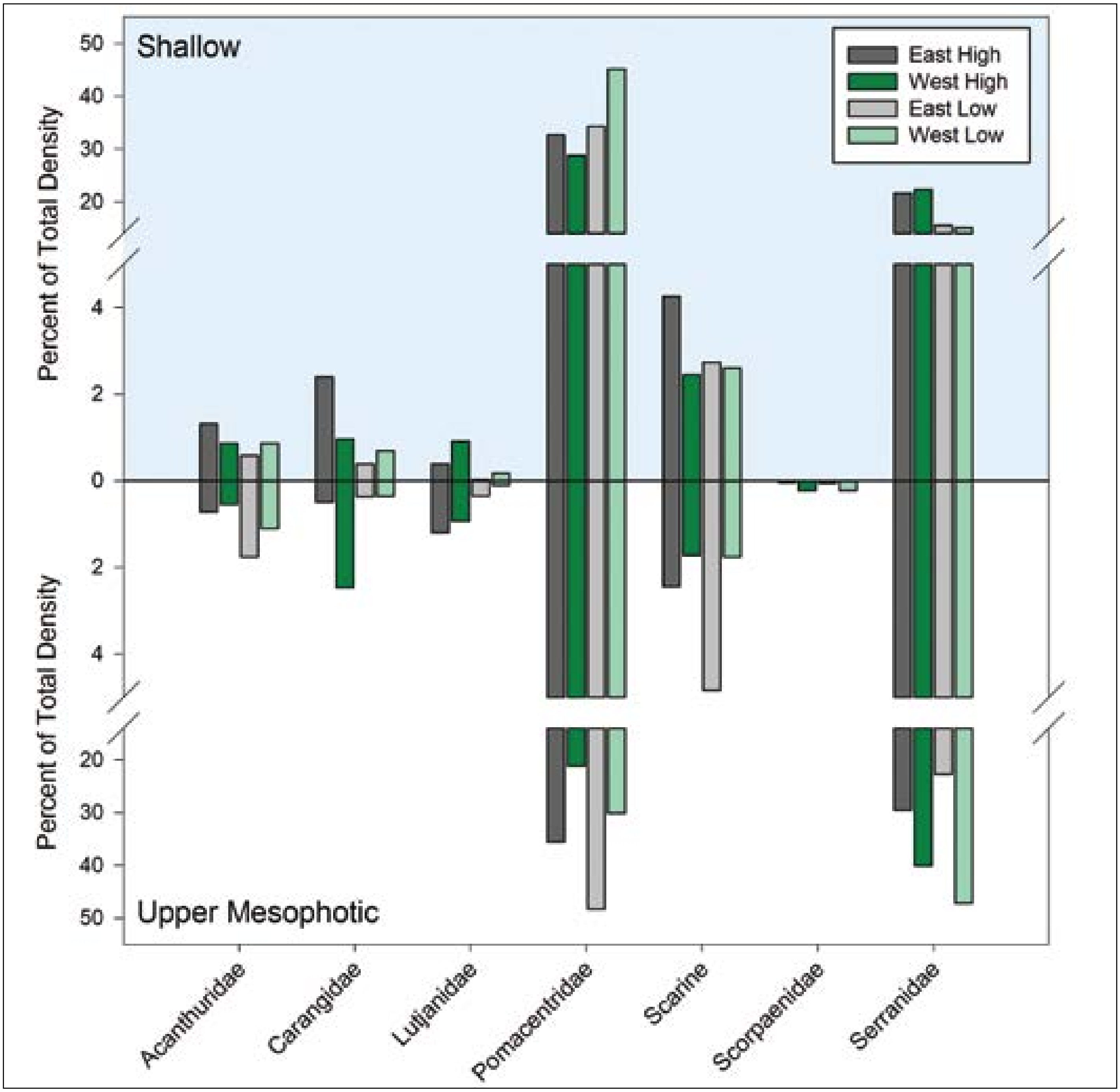


Figure 4.25. Percent of total density (#/100 m²) for seven ecologically, commercially, or recreationally important families by strata observed during diver surveys (2010-2012).

combined, these families comprised 62.5% of total fish density and 58.9% of total fish biomass. Pomacentridae and Serranidae made up the largest percent of total density among the target families in both shallow and UM strata (Figure 4.25). Serranidae encompassed the greatest percent of total biomass for all strata compared to the other six selected families (Figure 4.26). More detailed analysis of spatial patterns of density and biomass for these target families and selected species within each family follows.

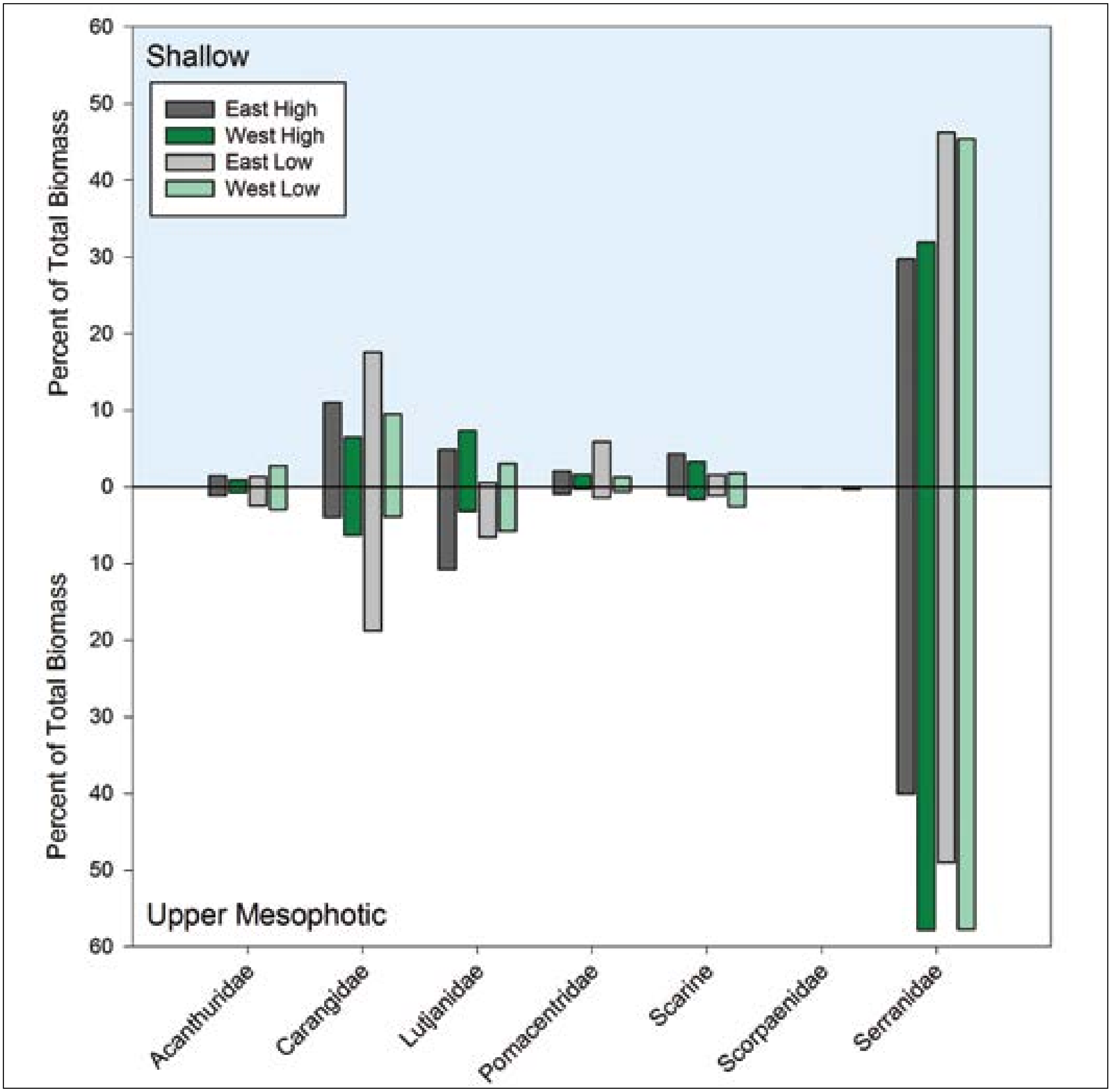


Figure 4.26. Percent of total biomass (kg/100 m²) for seven ecologically, commercially, or recreationally important families by strata observed during diver surveys strata (2010-2012).

Fish Communities

Serranidae (Groupers)

Among the 36 families identified during diver surveys, serranids were the dominant family by biomass (38.8% of total), and the third most abundant family (24.9%), preceded by labrids (26.5%) and pomacentrids (31.6%). Serranids (including groupers) were observed on all banks and relief types for both shallow and UM surveys (Figures 4.27 and 4.28), with none reported at only four sites (three EB, one WB). Highest densities and biomass of serranids occurred at deeper sites (>30 m), often on the edges of the coral reef (Figures 4.28 and 4.29). The maximum site-specific serranid density, excluding *P. furcifer*, reported here (n = 15 individuals/100 m^2) is similar to a previous study (n = 13; Caldow et al., 2009). although this study's maximum site-specific total biomass, also excluding *P. furcifer*, was slightly lower (103 kg/100 m^2; 134 kg/100 m^2; Caldow et al., 2009).

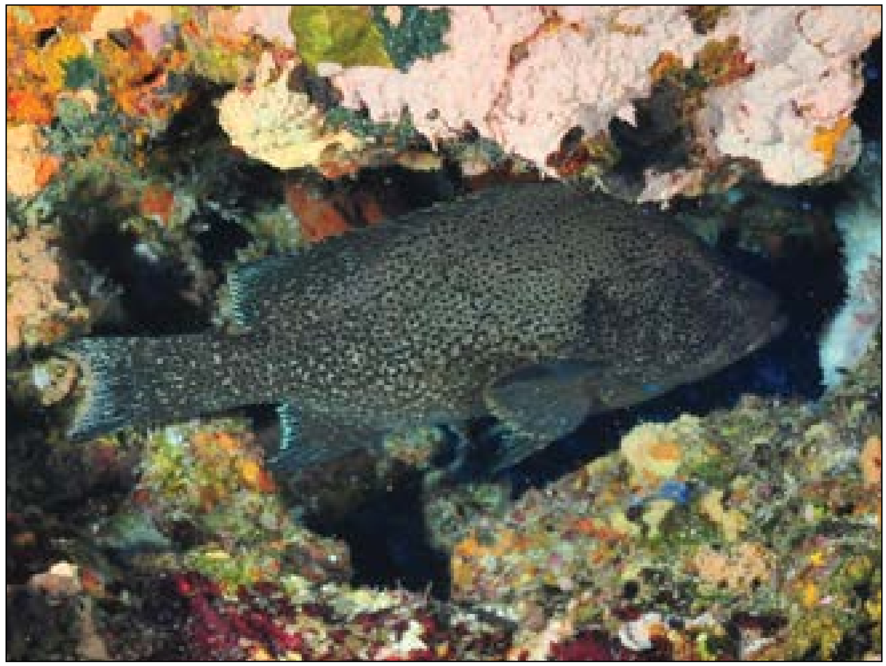


*Figure 4.27. Marbled grouper (*Dermatolepis inermis*), a species from the family Serranidae, in FGBNMS. Photo: M. Winfield (UNCW)*

Fifteen Serranidae species, including members of the genera *Serranus* and *Hypoplectrus*, were identified during diver surveys. *P. furcifer* was observed in the highest abundance and biomass (Figure 4.30). Its biomass and density were greater in the UM stratum, with the highest density and biomass occurring in the high relief stratum of WB. *Mycteroperca interstitialis* was the second most abundant serranid in all strata (Figure 4.31). While *M. bonaci* was not one of the most abundant serranids, it was the second highest in biomass for nearly all strata (Figure 4.32). Mean biomass was greater in the UM stratum for nearly all serranid species, especially *M. bonaci*, *M. interstitialis*, *M. venenosa* and *M. tigris* (Figure 4.32). In addition to *P. furcifer*, seven species of commercially and/or recreationally important species were recorded in these surveys belonging to the genera *Cephalopholis*, *Dermatolepis*, *Epinephelus* and *Mycteroperca*. Spatial trends of these eight species are examined in further detail in the individual species sections below.

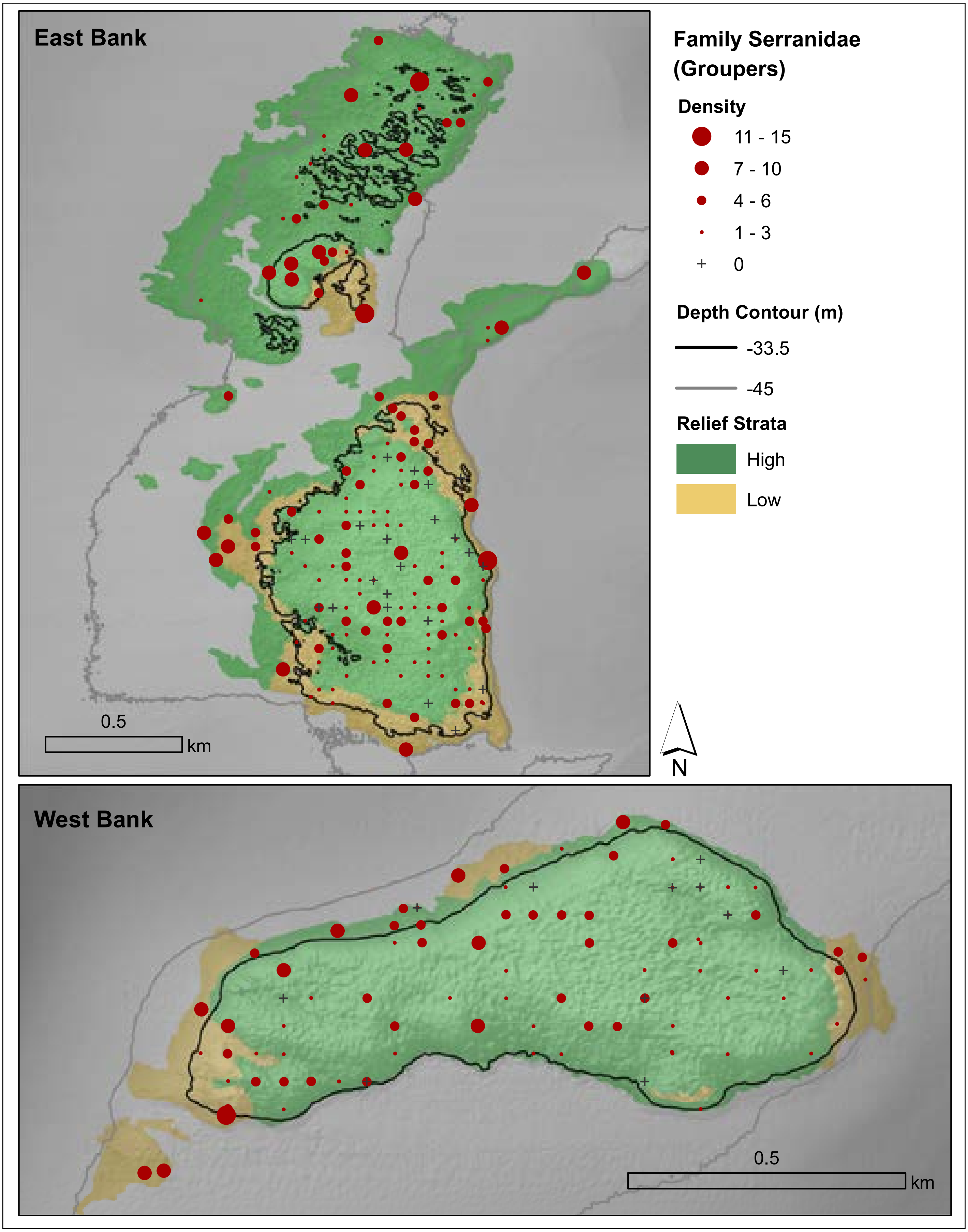


Figure 4.28. Observed density (#/100 m²) of all Serranidae, except Paranthias furcifer, *recorded during diver surveys.*

Fish Communities

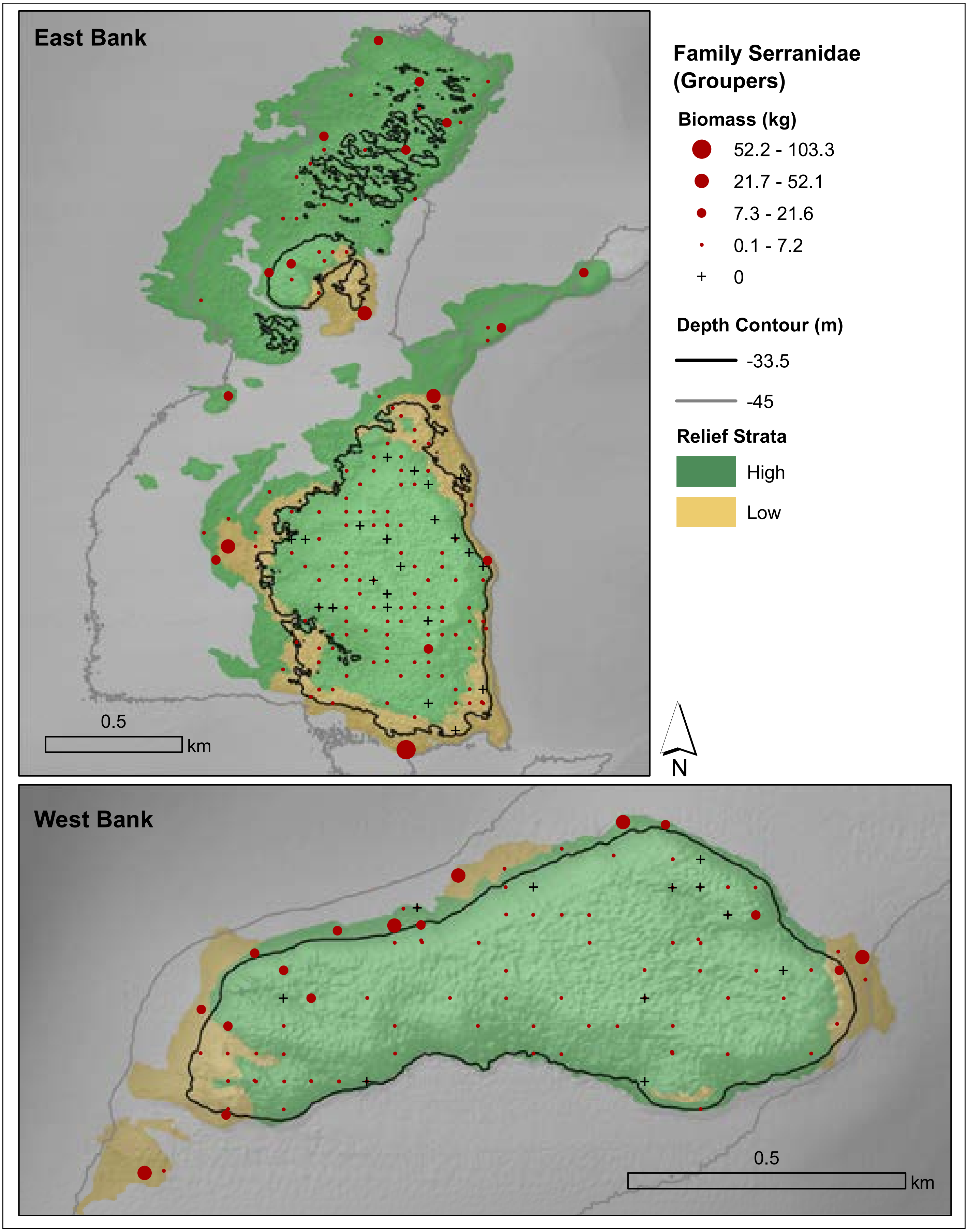
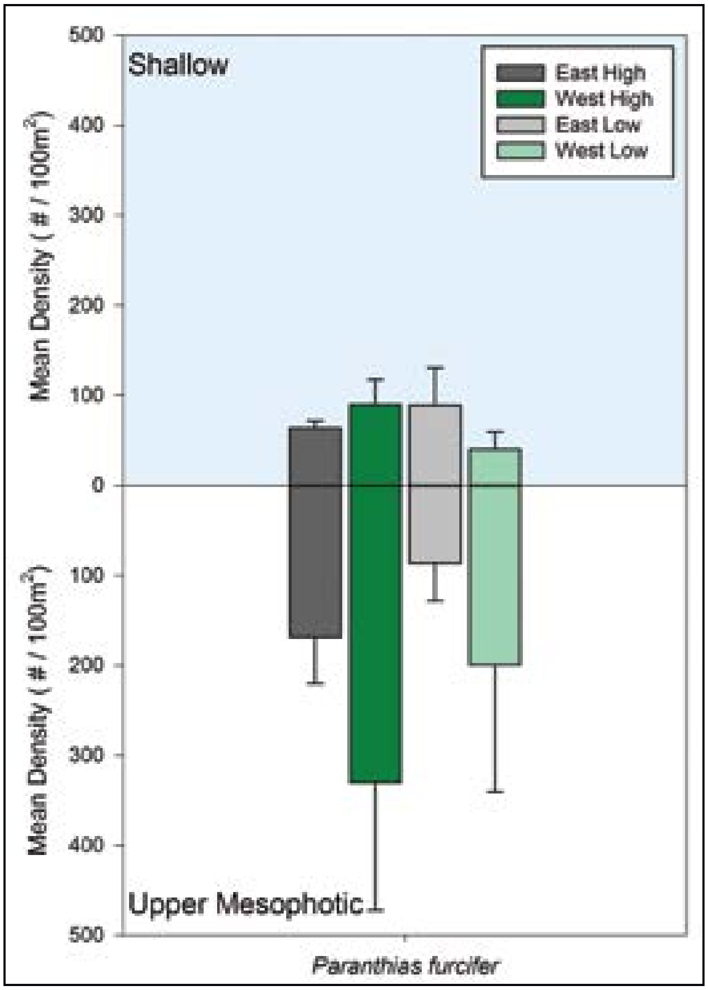


Figure 4.29. Observed biomass (kg/100 m²) of all Serranidae except Paranthias furcifer*, recorded during diver surveys.*



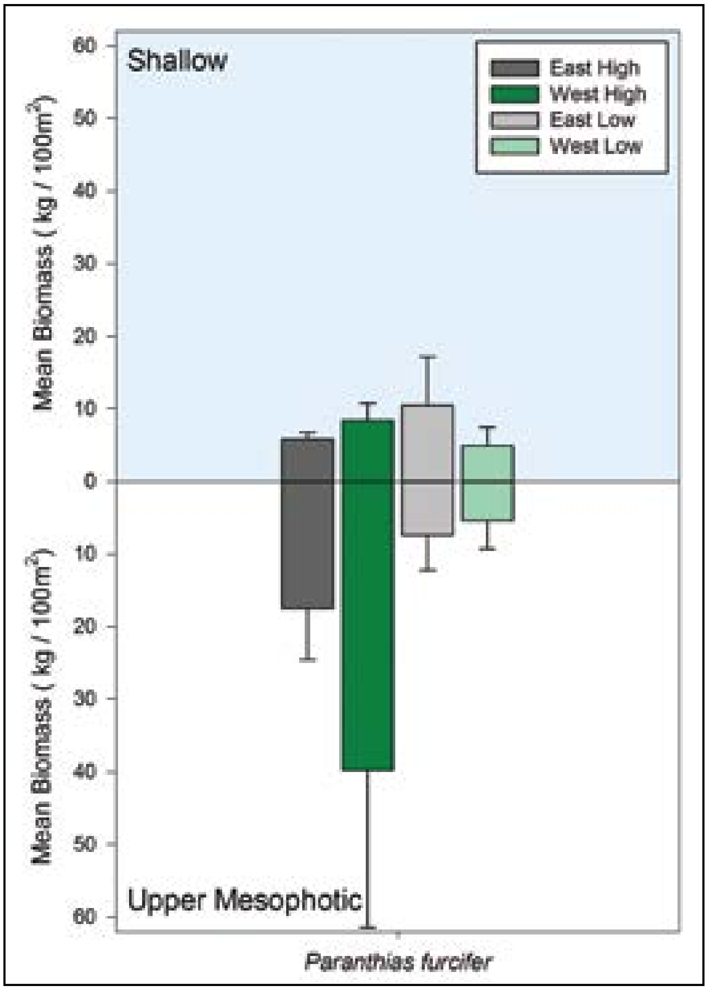


Figure 4.30. Mean density (#/100 m^2; left) and mean biomass (kg/100 m^2; right) of Paranthias furcifer *(Atlantic creolefish) by strata for dive surveys (2010-2012).*

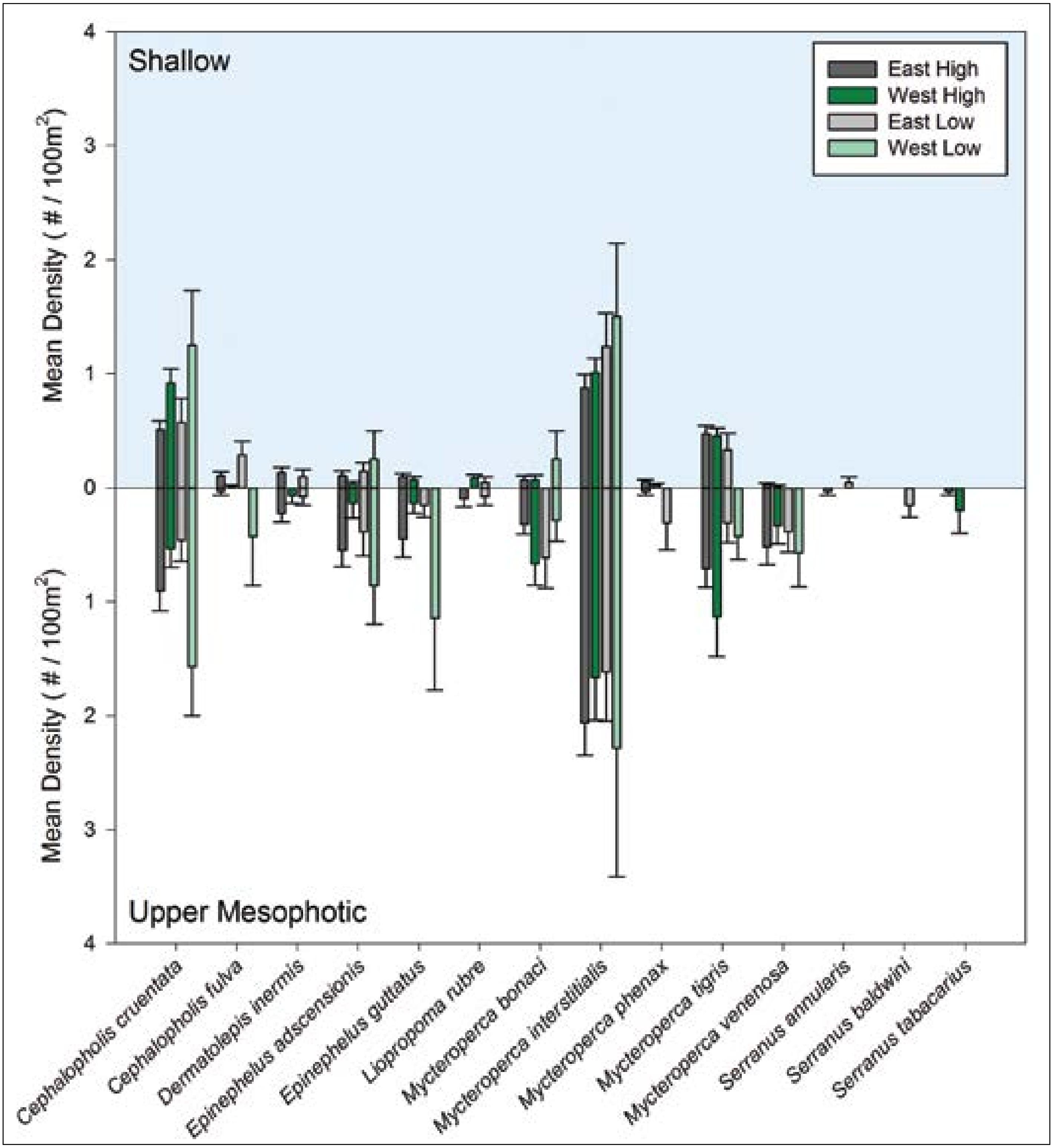


Figure 4.31. Mean density (#/100 m^2) of Serranidae species, except Paranthias furcifer*, by strata for dive surveys (2010-2012).*

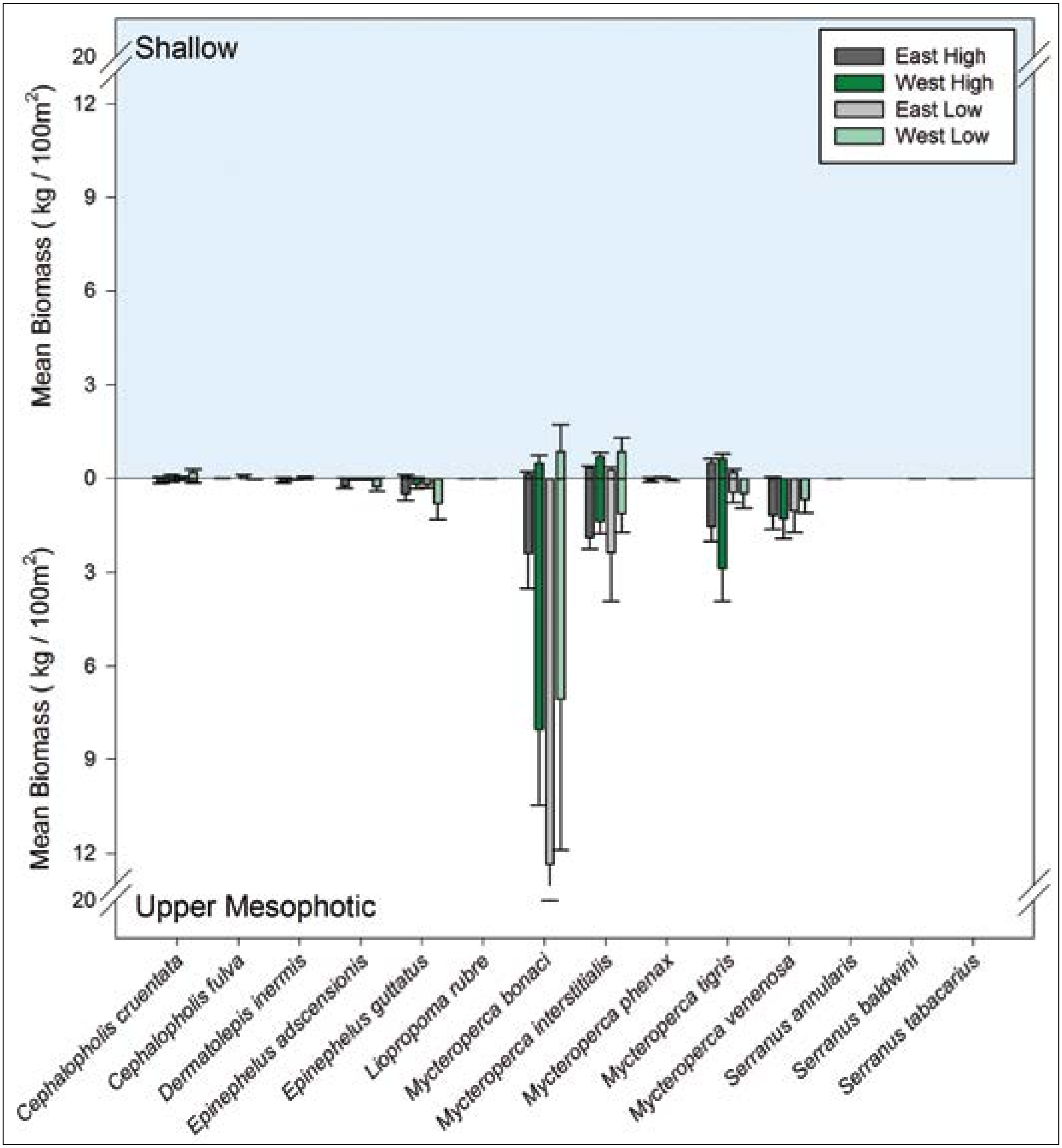


Figure 4.32. Mean biomass (kg/100 m^2) of Serranidae species, except Paranthias furcifer*, by strata for dive surveys (2010-2012).*

<u>*Cephalopholis cruentata* (graysby)</u>

A total of 207 *Cephalopholis cruentata* were recorded at 130 of 291 surveyed sites on both banks and across a range of depths (Figure 4.33). Mean (± SE) density was 0.7 ± 0.6/100 m^2 (Figure 4.31), with highest densities observed on both the WB low relief shallow (1.25 ± 0.48/100 m^2) and UM strata (1.57 ± 0.43/100 m^2; Figure 4.34), consistent with multivariate analyses showing elevated densities in the UM (Table 4.5). Similar to previous studies (Pattengill-Semmens et al., 2000; Caldow et al, 2009), densities were significantly greater on WB than EB (WB: 0.92 ± 0.10/100 m^2, EB: 0.58 ± 0.06 /100 m^2; Z = 2.85, p = 0.0044;), and there was no relationship with depth or rugosity. Within the shallow strata, mean density by bank reported here (WB: 0.93 ± 0.12/100 m^2, EB: 0.52 ± 0.07/100 m^2) were slightly less than observed by Caldow et al. (2009; WB: 1.14/100 m^2, EB: 0.71/100 m^2). There were no inter-annual density differences by depth within the UM or shallow strata.

*Figure 4.33. Graysby (*Cephalopholis cruentata*) in FGBNMS. Photo: J. Voss (HBOI/FAU and NOAA/CIOERT)*

Biomass patterns for *C. cruentata* were similar to those observed for density. Mean biomass, for all sites combined was 0.06 ± 0.009 kg/100 m^2 (Figure 4.32), with highest biomass occurring within the shallow WB low relief stratum (0.19 ± 0.10 kg/100 m^2; Figure 4.35). Biomass was significantly greater on WB than EB (WB: 0.10 ± 0.02 kg/100 m^2, EB: 0.05 ± 0.009 kg/100 m^2; Z = 3.4485, p =0.0006), and was not correlated with depth or rugosity. There were no differences in biomass by year within the UM or the shallow strata.

C. cruentata were found in a range of sizes for all strata, with the majority occurring on low relief habitats for all size classes (Figure 4.36). For both depth strata, the majority of individuals were between 10-20 cm FL, similar to the findings of a previous study (Caldow et al., 2009). The maximum size reported was within the 30-35 cm size class, occurring at six sites, primarily within the shallow survey depths of the WB. Mean length was only slightly greater in the UM (16.6 cm) compared to the shallow strata (15.3 cm). The majority of *C. cruentata* recorded (49% shallow; 55% UM) were larger than the age at sexual maturity (14 cm TL; Nagelkerken, 1979).

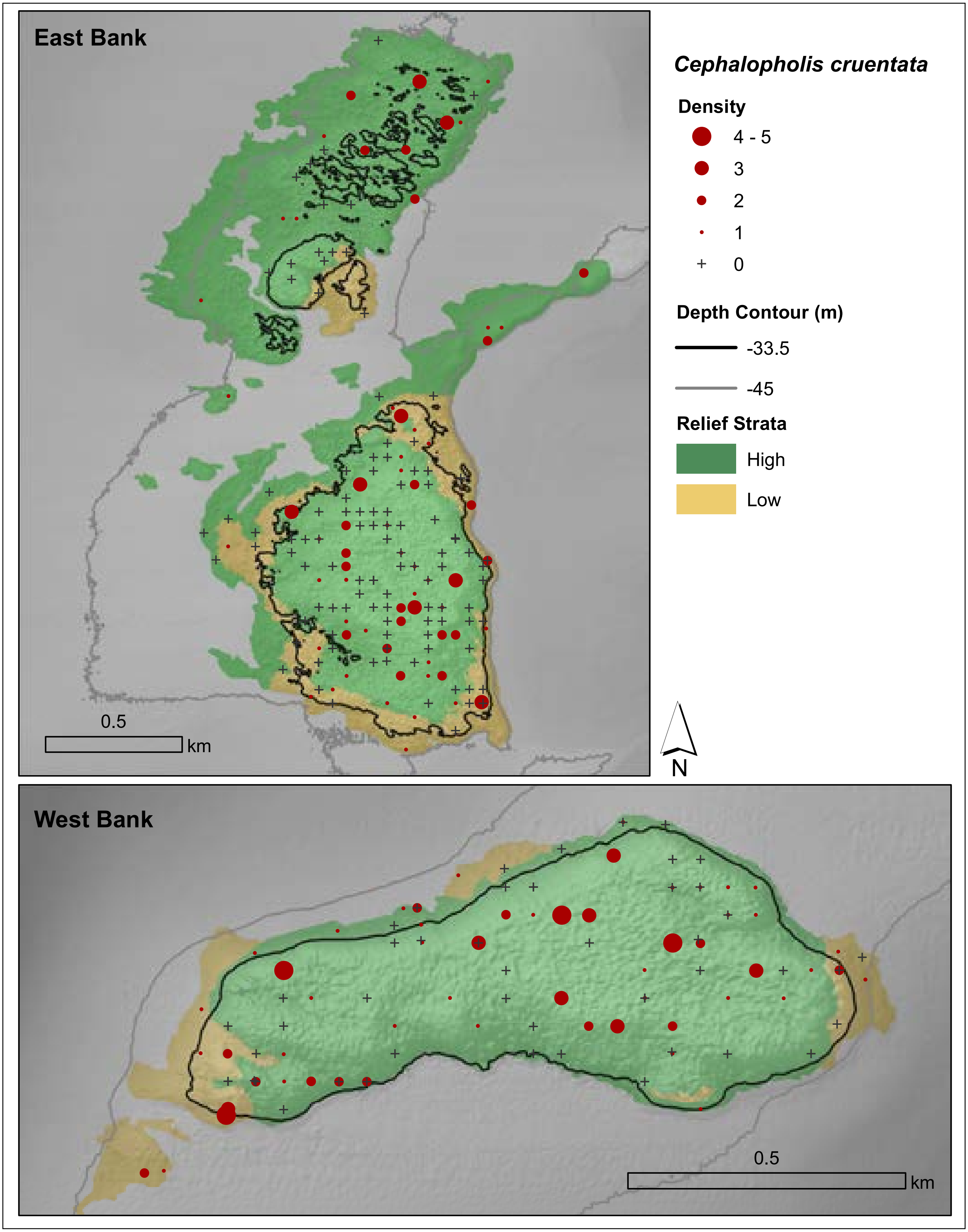


Figure 4.34. Observed density (#/100 m²) of Cephalopholis cruentata *(graysby) recorded during diver surveys from 2010-2012.*

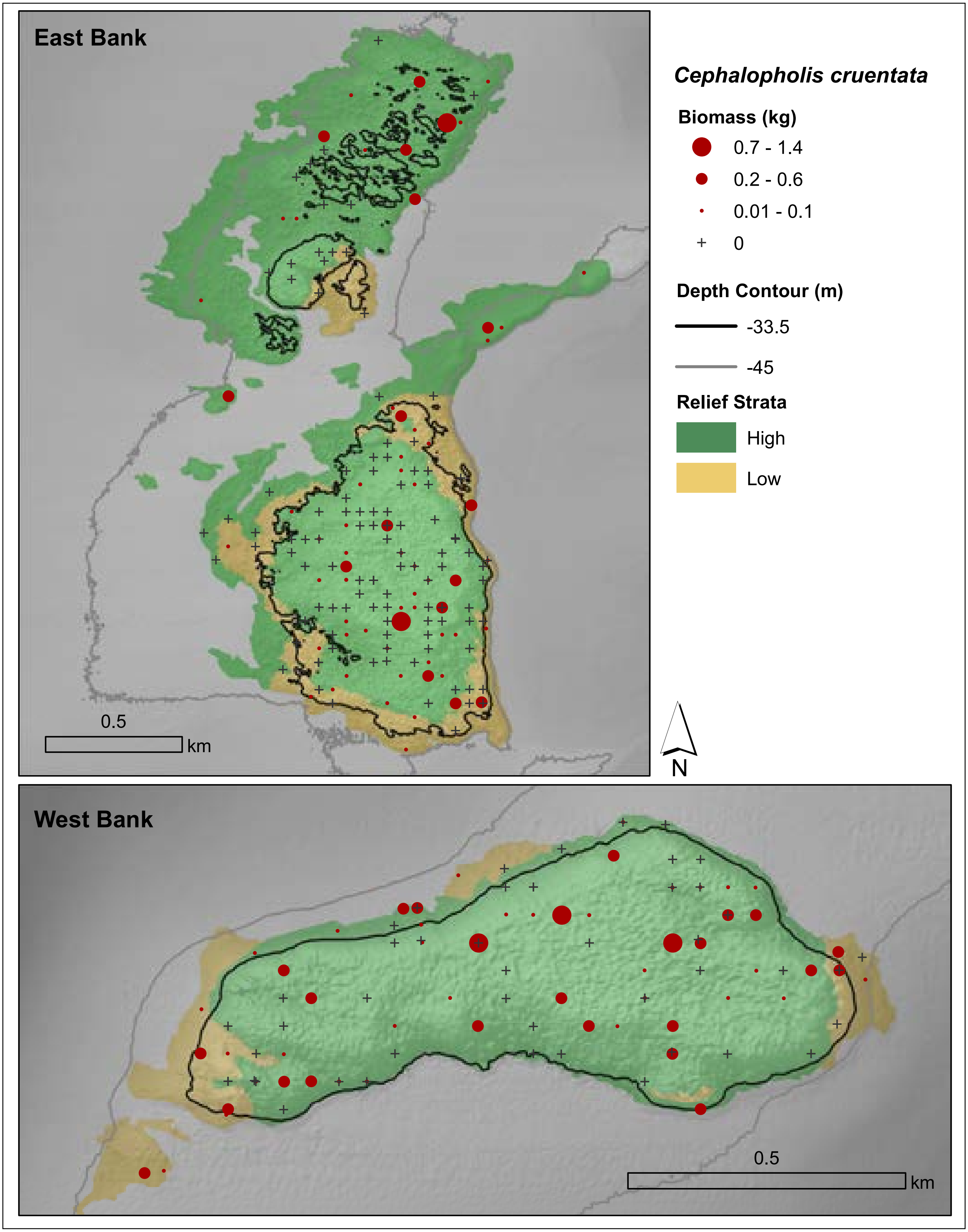


Figure 4.35. Observed biomass (kg/100 m²) of Cephalopholis cruentata *(graysby) recorded during diver surveys from 2010-2012.*

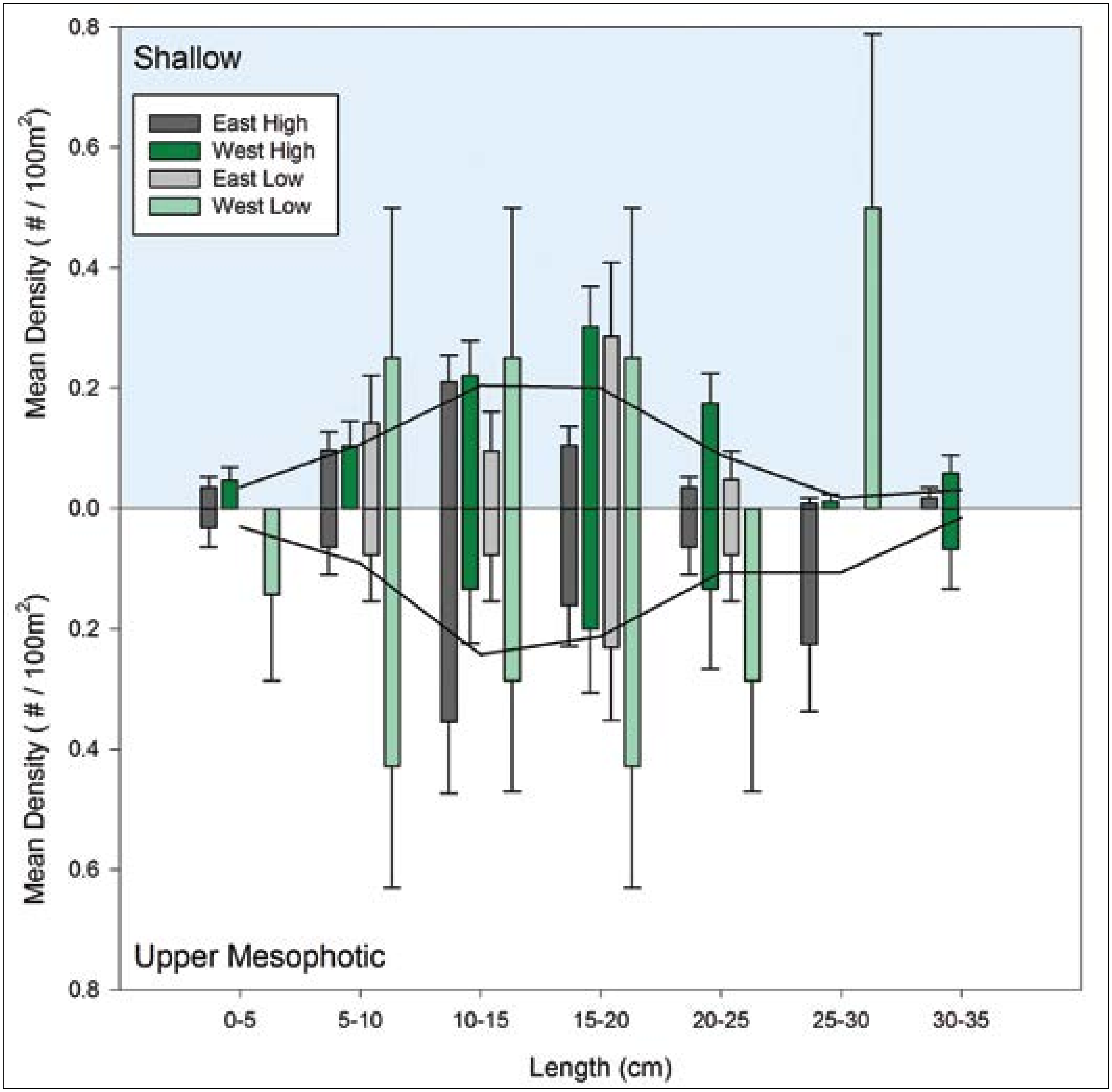


Figure 4.36. Cephalopholis cruentata*, mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* C. cruentata *density per size class.*

<u>*Dermatolepis inermis* (marbled grouper)</u>

A total of 26 *Dermatolepis inermis* were recorded within 22 of 291 sites (Figure 4.37). Mean (± SE) density was 0.09 ± 0.02/100 m^2. Highest densities were observed within the UM (0.14 ± 0.04/100 m^2) and EB (0.14 ± 0.03/100 m^2; Figure 4.31). Densities were significantly higher within the UM (Z = 2.08, p = 0.0371), and increased significantly with depth (ρ = 0.1413, p = 0.0159) and decreased with rugosity, although this result was not significant. This trend with decreasing rugosity may also be related to an edge effect; where *D. inermis* density is higher at reef-sand interfaces, however, further study is needed to answer this question. Caldow et al. (2009) also found higher densities at reef-sand edges. There was no density difference by relief strata, but it was significantly different by bank (WB: 0.009 ± 0.009/100 m^2, EB: 0.14 ± 0.03/100 m^2; Z = -3.39, p = 0.0007), especially within the shallow strata (WB: 0.00 ± 0.00/100 m^2, EB: 0.13 ± 0.04/100 m^2; Z = -3.02, p = 0.0025; Figure 4.38). In contrast, Caldow et al. (2009) found similar densities by bank (WB: 0.12/100 m^2, EB: 0.16/100 m^2) within the shallow strata, despite identical survey methods. The fact that multiple sites within the isolated shallow portion on the north edge of the EB contained *D. inermis*, with four individuals recorded on one transect, may explain the differences between this study and Caldow et al. (2009). There were no interannual differences within the UM or the shallow strata. This species is listed as "Near Threatened" on the International Union for Conservation of Nature (IUCN) Red List of Threatened Species. Within FGBNMS, the EB area may be of interest for potential future management/monitoring considerations for this species.

*Figure 4.37. Marbled grouper (*Dermatolepis inermis*) in FGBNMS. Photo: J. Voss (HBOI/FAU and NOAA CIOERT)*

Biomass results were similar to those observed for density. Mean biomass for all sites combined was 0.022 ± 0.005 kg/100 m^2 and was highest within the UM (0.05 ± 0.01 kg/100 m^2; Figure 4.32). Biomass was also significantly higher on the EB (WB: 0.003 ± 0.003 kg/100 m^2, EB: 0.03 ± 0.009 kg/100 m^2; Z = -3.37, p = 0.0007), especially within the shallow strata (WB: 0.00 ± 0.00 kg/100 m^2, EB: 0.02 ± 0.01 kg/100 m^2; Z = -3.02, p = 0.0025; Figure 4.39). Biomass significantly increased with depth (ρ = 0.1483, p = 0.0113) and was not significantly related to rugosity. Similar to other serranids and apex species, biomass was greater in UM than shallow strata (Z = 2.26, p = 0.0239; Figure 3.49). There were no differences by relief strata or by year within the UM or the shallow strata.

D. inermis mean length was greater within the UM (59.4 cm) than the shallow strata (45 cm). There were more large individuals (≥50 cm) recorded within UM sites (Figure 4.40). *D. inermis* were recorded across a similar size range as that reported by Caldow et al. (2009); however, within shallow depths, mean length in the previous study (57 cm) was greater than mean length recorded here (45 cm).

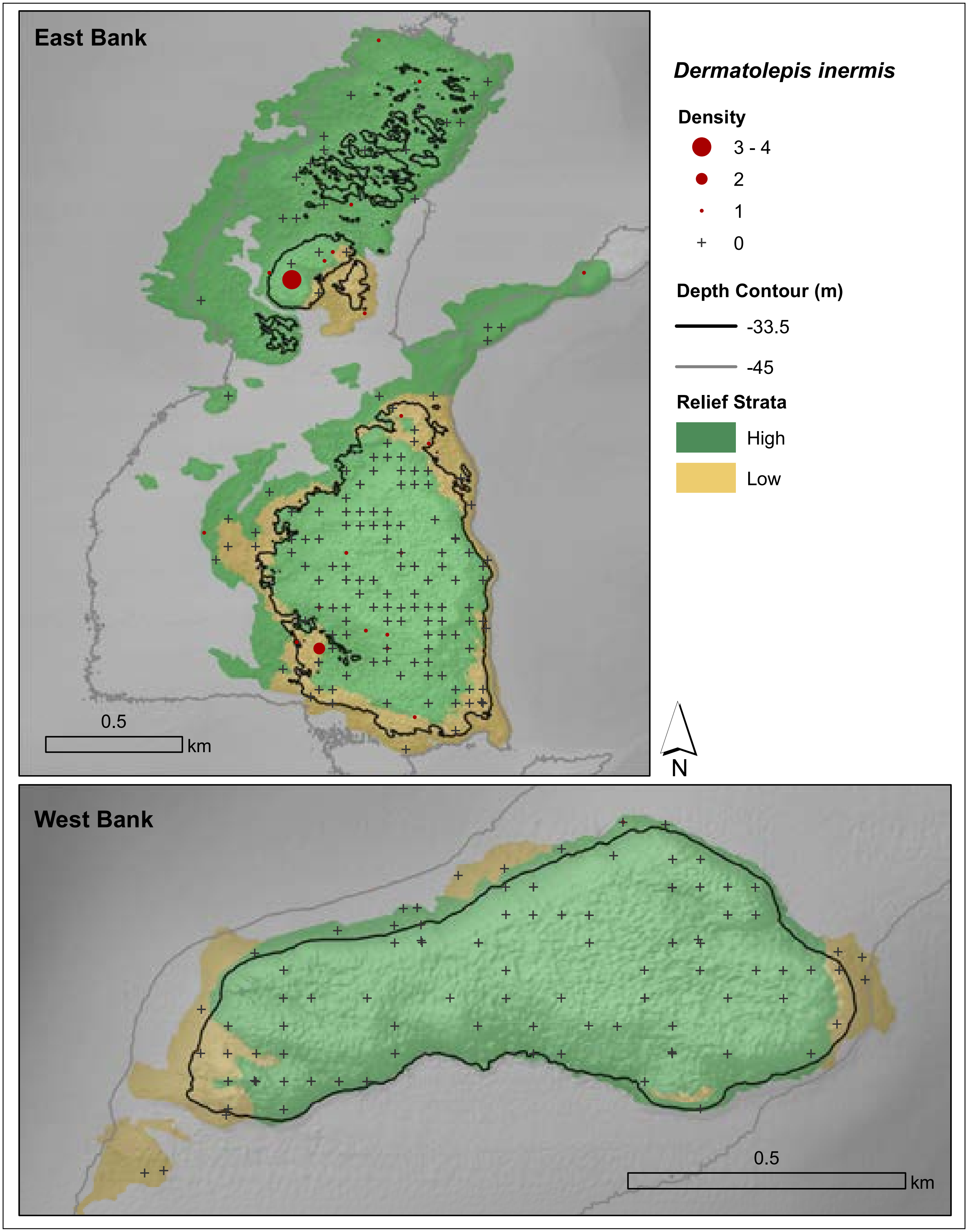


Figure 4.38. Observed density (#/100 m²) of Dermatolepis inermis *(marbled grouper) recorded during diver surveys from 2010-2012.*

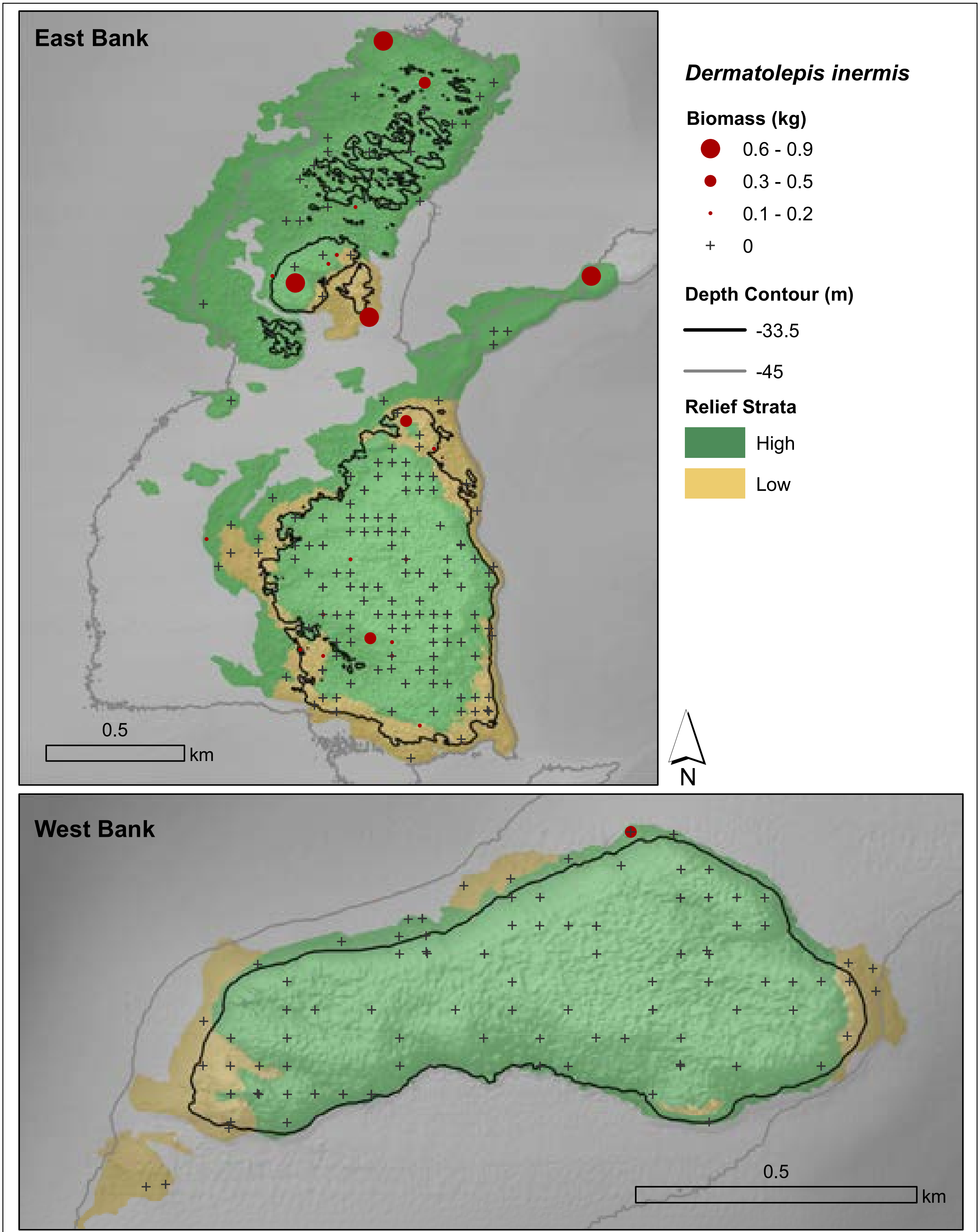


Figure 4.39. Observed biomass (kg/100 m^2) of Dermatolepis inermis *(marbled grouper) recorded during diver surveys from 2010-2012.*

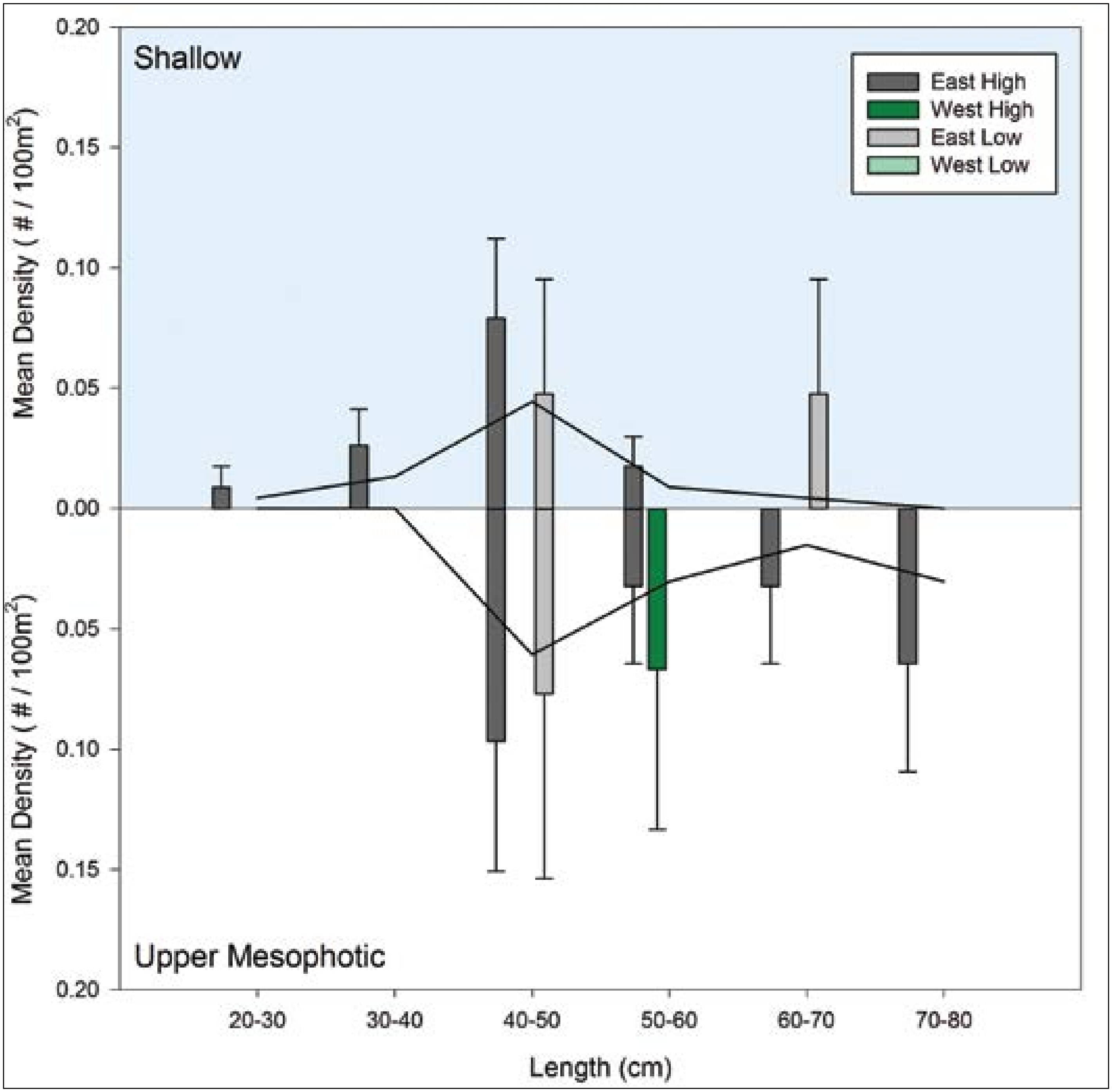


Figure 4.40. Dermatolepis inermis *mean density (#/100 m^2) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* D. inermis *density per size class.*

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<u>*Epinephelus adscensionis* (rock hind)</u>

*Figure 4.41. Rock hind (*Epinephelus adscensionis*) in FGBNMS. Photo: M. Nuttall (NOAA NOS/ONMS/FGBNMS)*

A total of 49 *Epinephelus adscensionis* were recorded at 35 of 291 surveyed sites (Figure 4.41). Overall mean density was 0.17 ± 0.03/100 $m^{2,}$ with the highest densities occurring in the UM (0.45 ± 0.09/100 m^2) and low relief strata (0.33 ± 0.64/100 m^2; Figure 4.31). There were significant differences by depth (UM: 0.45 ± 0.09/100 m^2; shallow: 0.08 ± 0.02/100 m^2; Z = 5.29, p = 0.0001) and relief strata (High: 0.14 ± 0.03/100 m^2, Low: 0.33 ± 0.09/100 m^2; Z = 2.79, p = 0.0052; Figure 4.42). Density was positively correlated with depth (ρ = 0.2621, p < 0.0001) and negatively correlated with rugosity (ρ = -0.1876 p = 0.0013). Although the highest densities were on EB, there were no significant differences between banks (WB: 0.11 ± 0.04/100 m^2; EB: 0.21 ± 0.04/100 m^2). Individuals were reported on only nine WB sites, located on the western edge (Figure 4.42), while sightings on EB exhibited no spatial patterns. Within FGBNMS, *E. adscensionis* were found over deep, low relief habitats, consistent with findings of *E. adscensionis* distribution in the Florida Keys (Sluka et al., 1998) and by Caldow et al. (2009). Within the shallow stratum, mean density by bank reported here (WB: 0.04 ± 0.04/100 m^2, EB: 0.10 ± 0.02/100 m^2) was similar to those of Caldow et al. (2009; WB: 0.0/100 m^2, EB: 0.12/100 m^2). There were significant inter-annual density differences within the shallow strata (χ^2 = 12.069, p = 0.0024), but not in the UM strata. Densities in 2012 were significantly greater than 2010 and 2011. No differences were detected between 2010 and 2011.

Biomass results were similar to those observed for density. *E. adscensionis* mean biomass was 0.04 ± 0.01 kg/100 $m^{2,}$ with total contribution to serranidae biomass being low compared to other serranids (Figure 4.32). Biomass was higher within the UM stratum (0.14 ± 0.04 kg/100 m^2) than in the shallow stratum (0.008 ± 0.002 kg/100 m^2; Figure 4.43). This pattern is consistent with the trend of greater biomass of larger bodied serranids with increasing depth. Similar to density results, biomass inter-annual differences were only found within the shallow stratum, with significantly higher biomass in 2012 than in 2010 and 2011 (χ^2 = 11.94, p = 0.0026). No differences were detected between 2010 and 2011.

E. adscensionis were recorded across a range of sizes, with a greater range of individuals within the UM stratum than shallow (Figure 4.44). In general, mean fish length within the UM (22 cm) was greater than that observed in shallow stratum (15.6 cm). Caldow et al. (2009) reported a mean length of 18.6 cm on shallow reef surveys. Bullock and Smith (1991) estimate size at maturity of 25 cm standard length (SL) for individuals from the Florida Middle grounds. Length frequency indicates that mostly juveniles were found on the shallow stratum, while both adults and juveniles were observed on the UM.

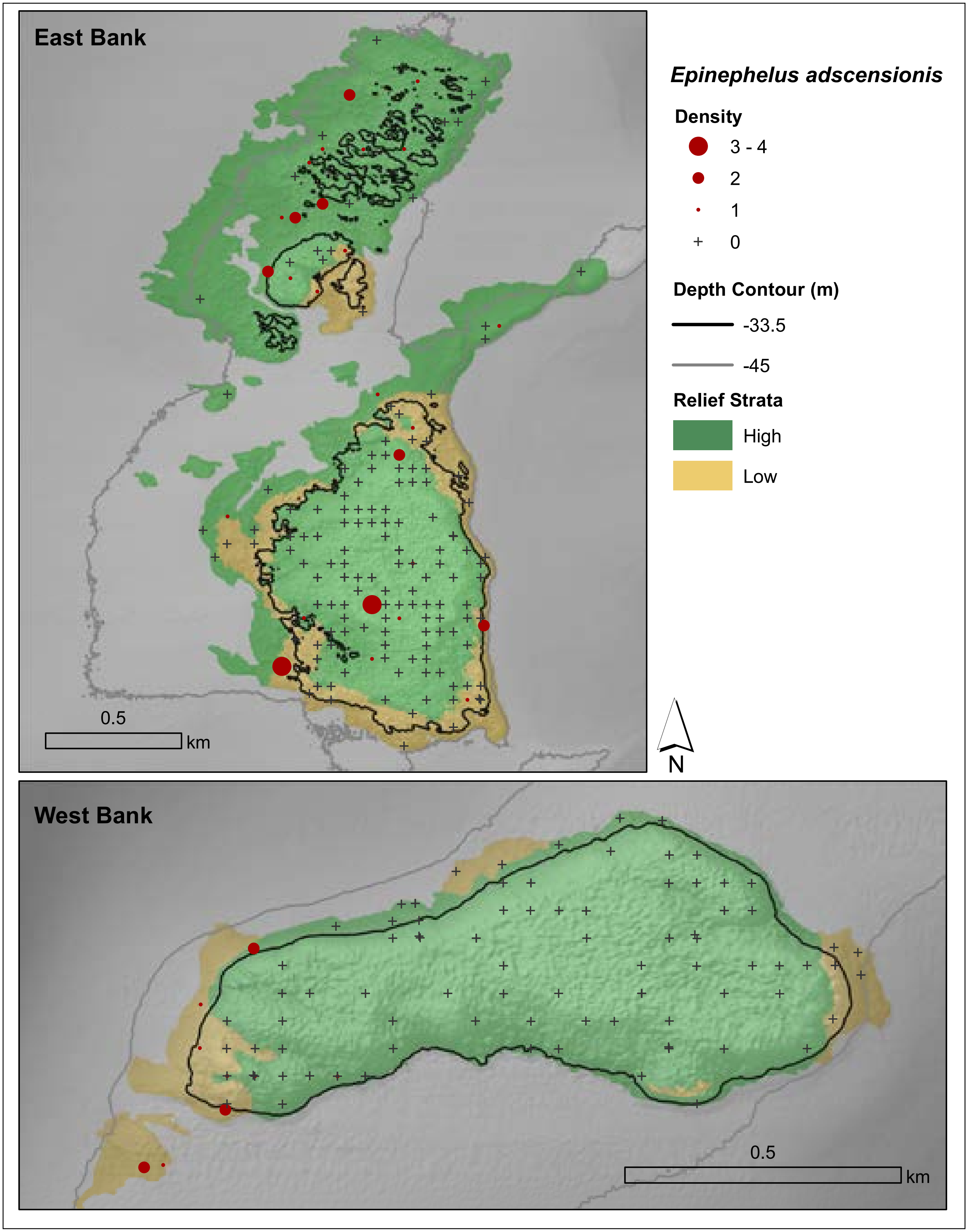

Figure 4.42. Observed density (#/100 m²) of Epinephelus adscensionis *(rock hind) recorded during diver surveys from 2010-2012.*

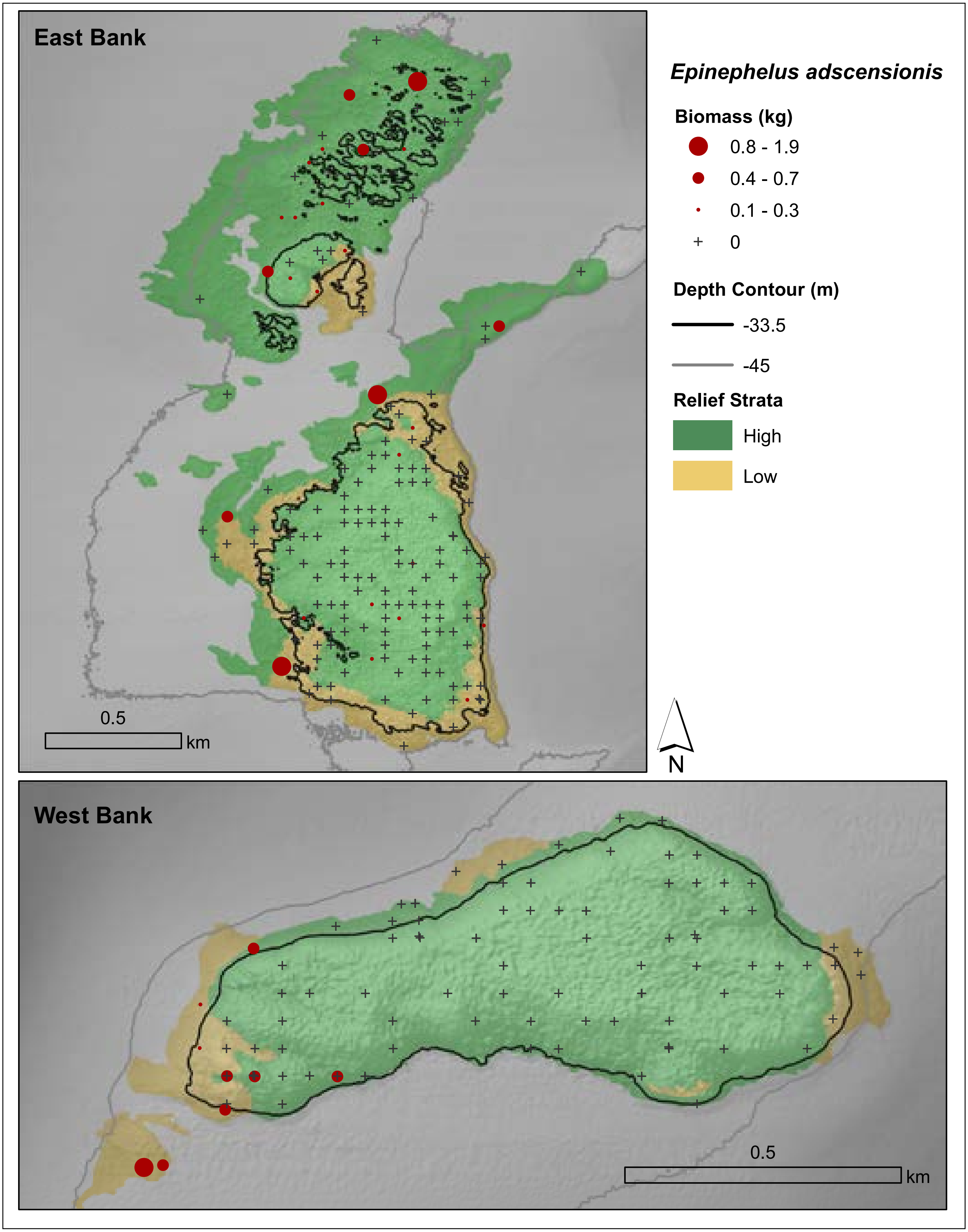

Figure 4.43. Observed biomass (kg/100 m²) of Epinephelus adscensionis *(rock hind) recorded during diver surveys from 2010-2012.*

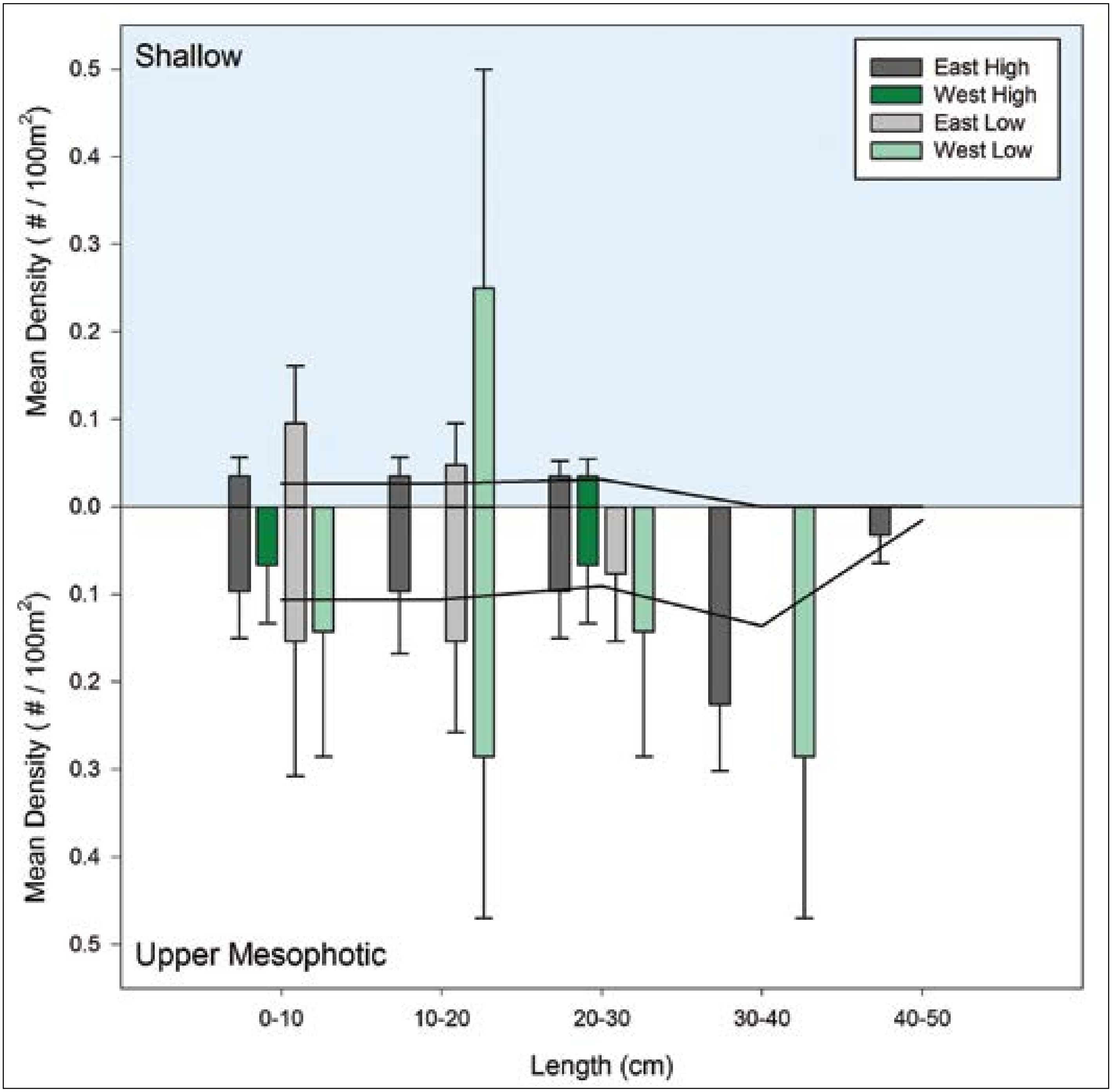

Figure 4.44. Epinephelus adscensionis *mean density (#/100 m^2) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* E. adscensionis *density per size class.*

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<u>*Epinephelus guttatus* (red hind)</u>

*Figure 4.45. Red hind (*Epinephelus guttatus*) in FGBNMS. Photo: M. Winfield (UNCW)*

A total of 43 *Epinephelus guttatus* were recorded at 32 of 291 surveyed sites (Figure 4.45). Overall mean density was 0.14 ± 0.03/100 m^2, with density being significantly greater in the UM (0.39 ± 0.11/100 m^2) than in the shallow strata (0.08 ± 0.02/100 m^2; Z = 4.00218, $p < 0.0001$; Figure 4.31). There was no significant differences between relief strata (High: 0.13 ± 0.03/100 m^2; Low: 0.22 ± 0.11/100 m^2) or by bank (WB: 0.15 ± 0.03/100 m^2; EB: 0.14 ± 0.05/100 m^2; Figure 4.46). Density significantly increased with depth (ρ = 0.2242, $p < 0.0001$) and decreased with rugosity (ρ = -0.1699, p = 0.0036), suggesting *E. guttatus* can often be found in deep, low relief habitats, similar to findings of Sluka et al. (1998). No inter-annual density differences were found. Caldow et al. (2009) reported higher sighting frequency and density on WB (0.24/100 m^2) than on EB (0.08/100 m^2).

Biomass results were similar to those observed for density. Overall mean *E. guttatus* biomass was 0.12 kg/100 $m^{2,}$ with significantly greater biomass in the UM (0.39 ± 0.12 kg/100 m^2) than in the shallow strata (0.04 ± 0.02 kg/100 m^2; Z = 4.10001, $p < 0.0001$; Figure 4.32). Biomass increased significantly with depth (ρ = 0.2691, p = 0.001) and decreased with rugosity (ρ = -0.1888, p = 0.0012). There were no significant differences by relief strata (High: 0.11 ± 0.03 kg/100 m^2; Low: 0.17 ± 0.09 kg/100 m^2) or by bank (WB: 0.08 ± 0.04 kg/100 m^2; EB: 0.14 ± 0.04 kg/100 m^2; Figure 4.47). There were no significant biomass inter-annual differences.

E. guttatus were observed in a range of sizes between 7.5 and 60 cm FL (Figure 4.47). Mean length was 38.8 cm in the UM compared to 26.4 cm in the shallow strata. Mean lengths observed by Caldow et al. (2009) on the shallow strata were similar to that reported here.

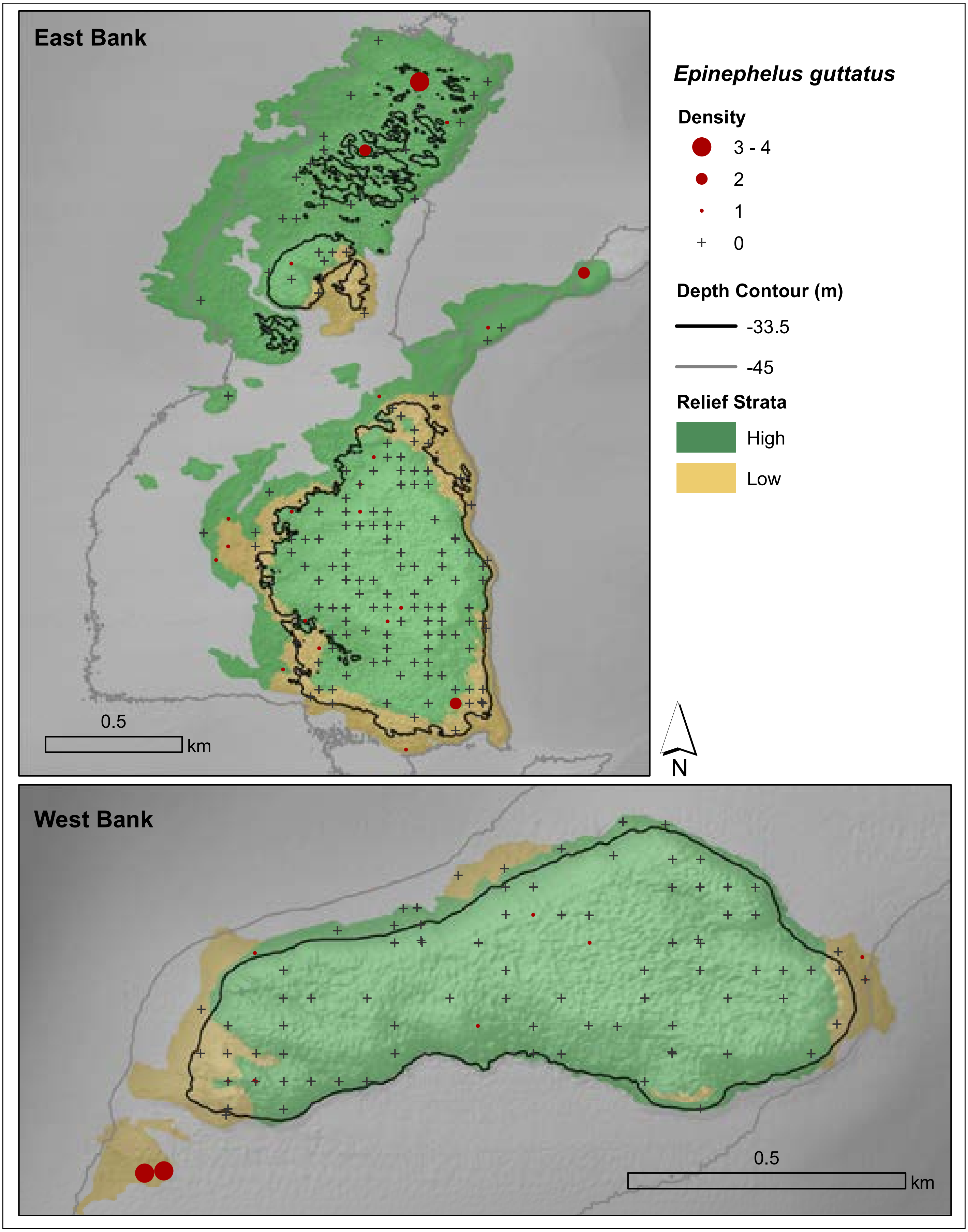

Figure 4.46. Observed density (#/100 m²) of Epinephelus guttatus *(red hind) recorded during diver surveys from 2010-2012.*

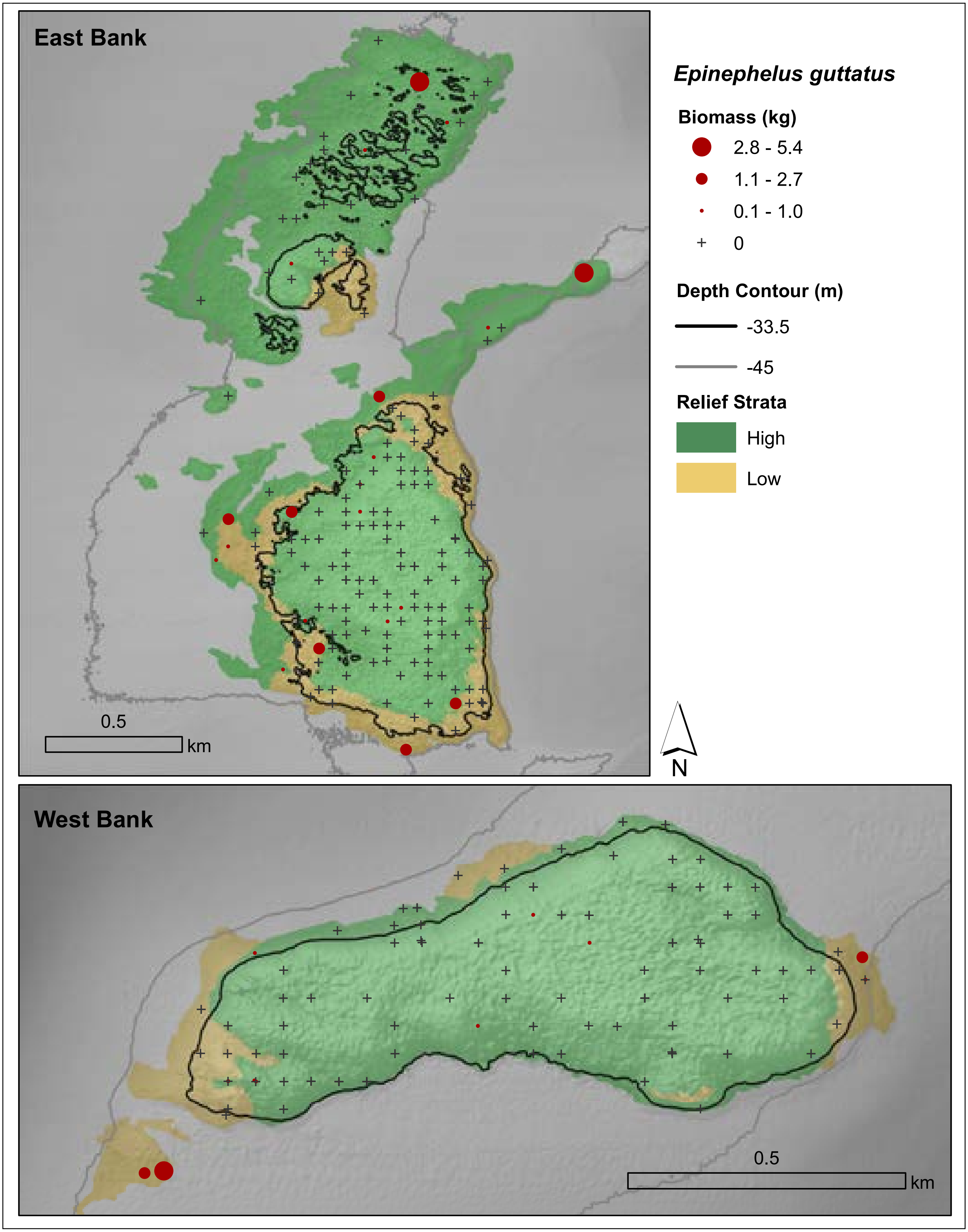

Figure 4.47. Observed biomass (kg/100 m²) of Epinephelus guttatus *(red hind) recorded during diver surveys from 2010-2012.*

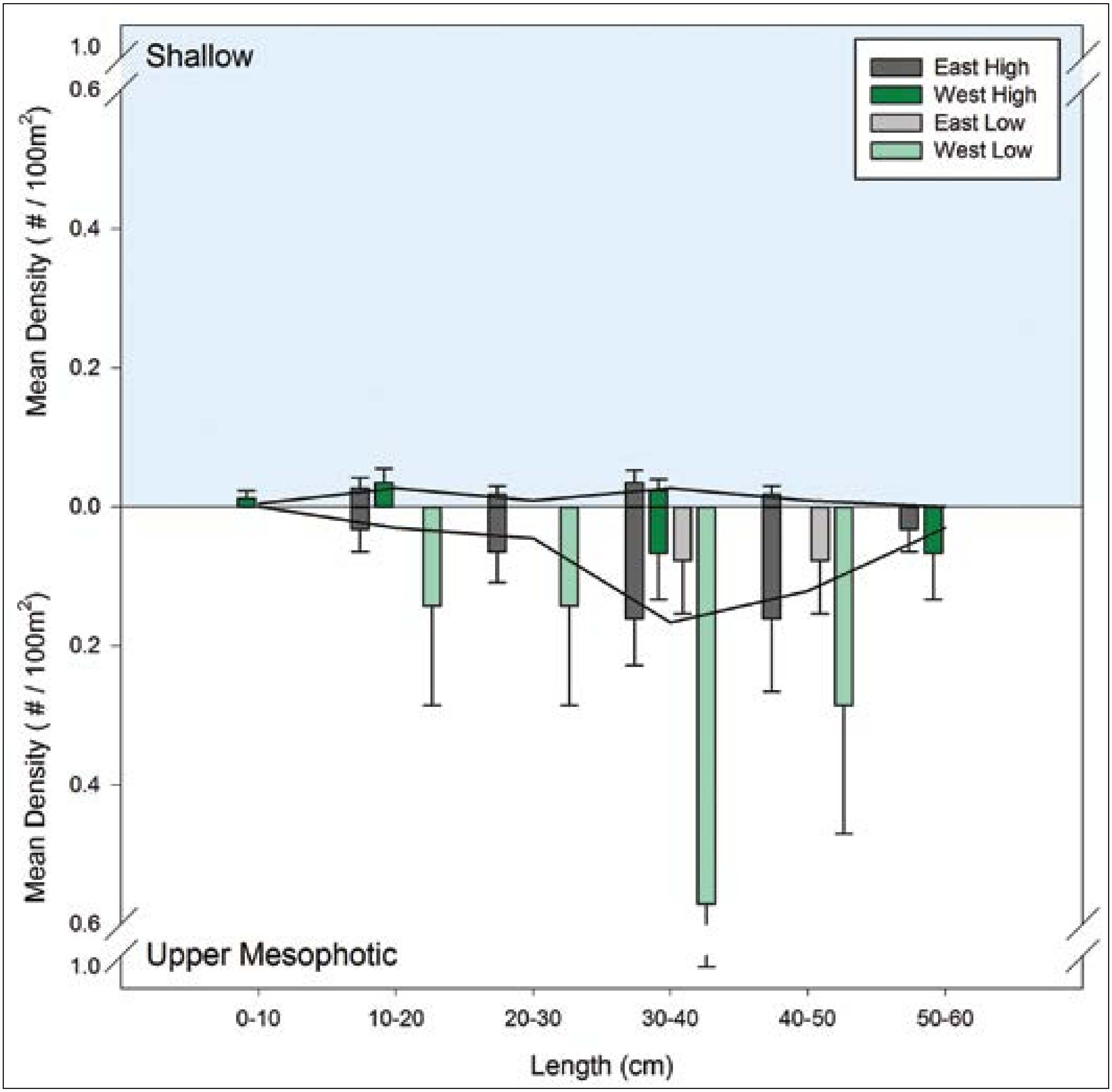

Figure 4.48. Epinephelus guttatus *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* E. guttatus *density per size class.*

Mycteroperca bonaci (black grouper)

A total of 45 *Mycteroperca bonaci* were recorded at 34 of 291 sites; all but nine sites were within the UM strata (Figure 4.49). Mean (±SE) density over all sites was 0.15 ± 0.03 individuals/100 m^2 (Figure 4.31). Highest densities recorded (site maximum 3 individuals/100 m^2; Figure 4.50) were observed on both depth and relief strata, and from both banks. Caldow et al. (2009) reported similar maximum densities (3-4 individuals/100 m^2) between 27-33 m depth. *M. bonaci* distribution was similar to other large bodied serranids, with highest densities occurring in low relief strata at depths greater than 30 m, similar to distribution patterns reported in the Florida Keys (Sluka et al., 1998). More than twice as many *M. bonaci* were observed in this study than by Caldow et al. (2009; 45 versus 21), likely related to the extended depth range of the present study (Figure 4.53). Density increased significantly with depth ($\rho = 0.3145$, $p < 0.0001$), consistent with multivariate analyses (Table 4.5), and was inversely related to rugosity ($\rho = -0.1555$, $p = 0.0079$). Densities were not significantly different between banks. Mean density of *M. bonaci* in the UM was significantly lower in 2012 (0.23 ± 0.08 individuals/100 m^2) compared to 2011 (0.66 ± 0.13 individuals/100 m^2; $Z = -2.5596$, $p = 0.0105$).

*Figure 4.49. Black grouper (*Mycteroperca bonaci*) in FGBNMS. Photo: NOAA NOS/NCCOS/CCMA/Biogeography Branch*

Biomass distribution patterns were similar to density distribution (Figure 4.51) with no difference between banks, negatively correlated with rugosity ($\rho = -0.1661$, $p = 0.0045$), positively correlated with depth ($\rho = 0.3293$, $p < 0.0001$), and consistent with multivariate analyses (Tables 4.5 and 4.6). Biomass of *M. bonaci* was the second highest among serranids (Figure 4.32); mean biomass over all sites was 1.6 ± 0.43 kg/100 m^2. Sites with higher biomass (> 17.7 kg/100 m^2; Figure 4.51) were all at depths >33.5 m. Maximum biomass (95.4 kg/100 m^2) was observed at a UM low relief site on EB. Fish sizes at these sites ranged from 15-20 cm to 120 cm.

M. bonaci were recorded in a range of sizes (7.5-120 cm; Figure 4.52), with the majority of larger individuals (>50 cm) occurring at UM sites. For example, of the 45 *M. bonaci* recorded in this study, 12 individuals were recorded with fork lengths >100 cm (n = 9 UM sites, 1 shallow). Caldow et al. (2009) reported adults (>70 cm) were observed on high relief sites, while juveniles (<70 cm) were found on both relief strata. In the present study, individuals >70 cm within shallow strata were only on high relief sites. Within the UM, however, *M. bonaci* of all sizes were observed in all strata (Figure 4.52). Fish were larger in the UM strata (mean length: 81.2 cm) versus shallow (54.3 cm). Mean fish length (73 cm) from shallow strata observations in Caldow et al. (2009) was greater than that reported here. Individuals of 120 cm FL (maximum size observed during this study) were recorded from UM depths on both EB and WB.

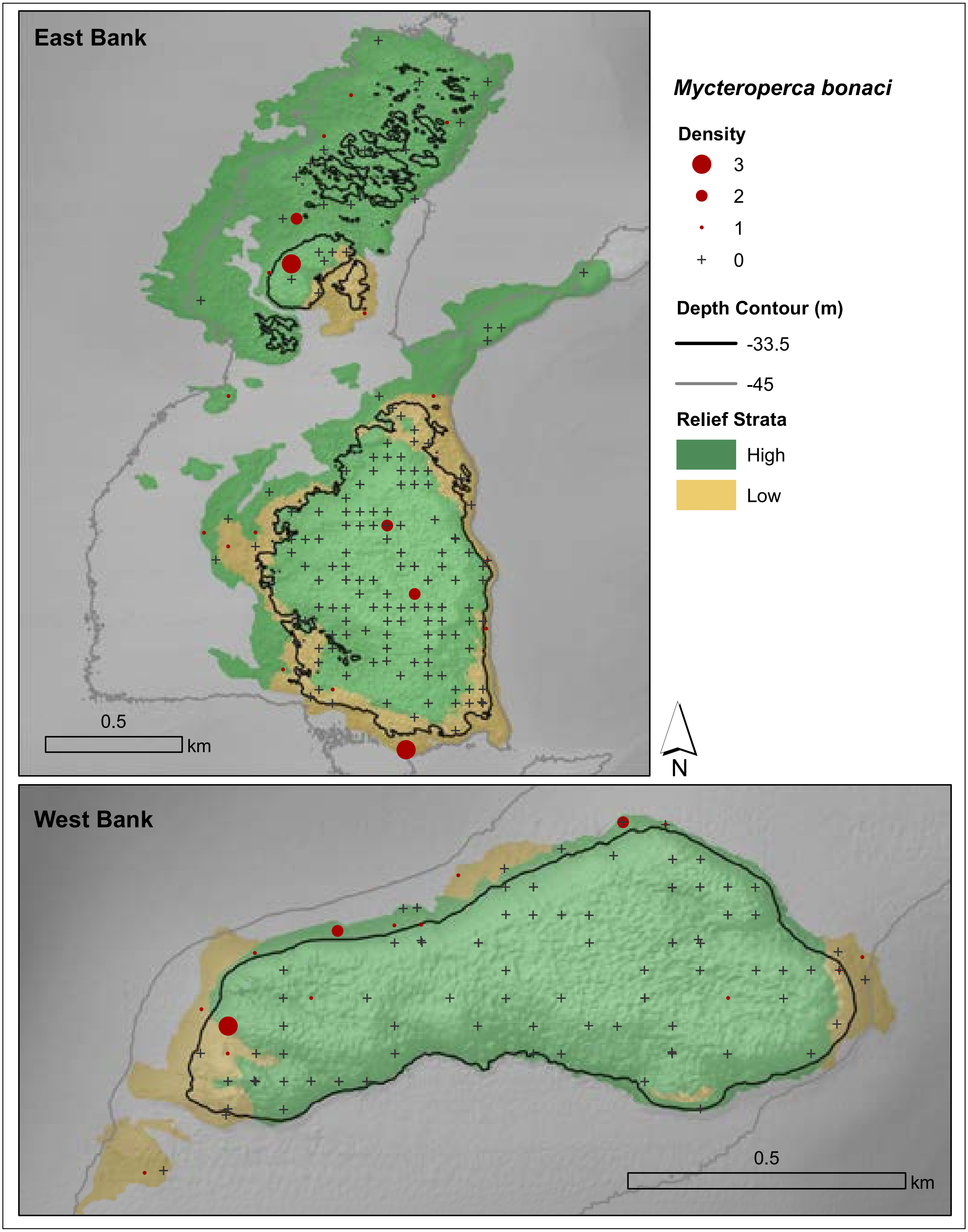

Figure 4.50. Observed density (#/100 m²) of Mycteroperca bonaci *(black grouper) recorded during diver surveys from 2010-2012.*

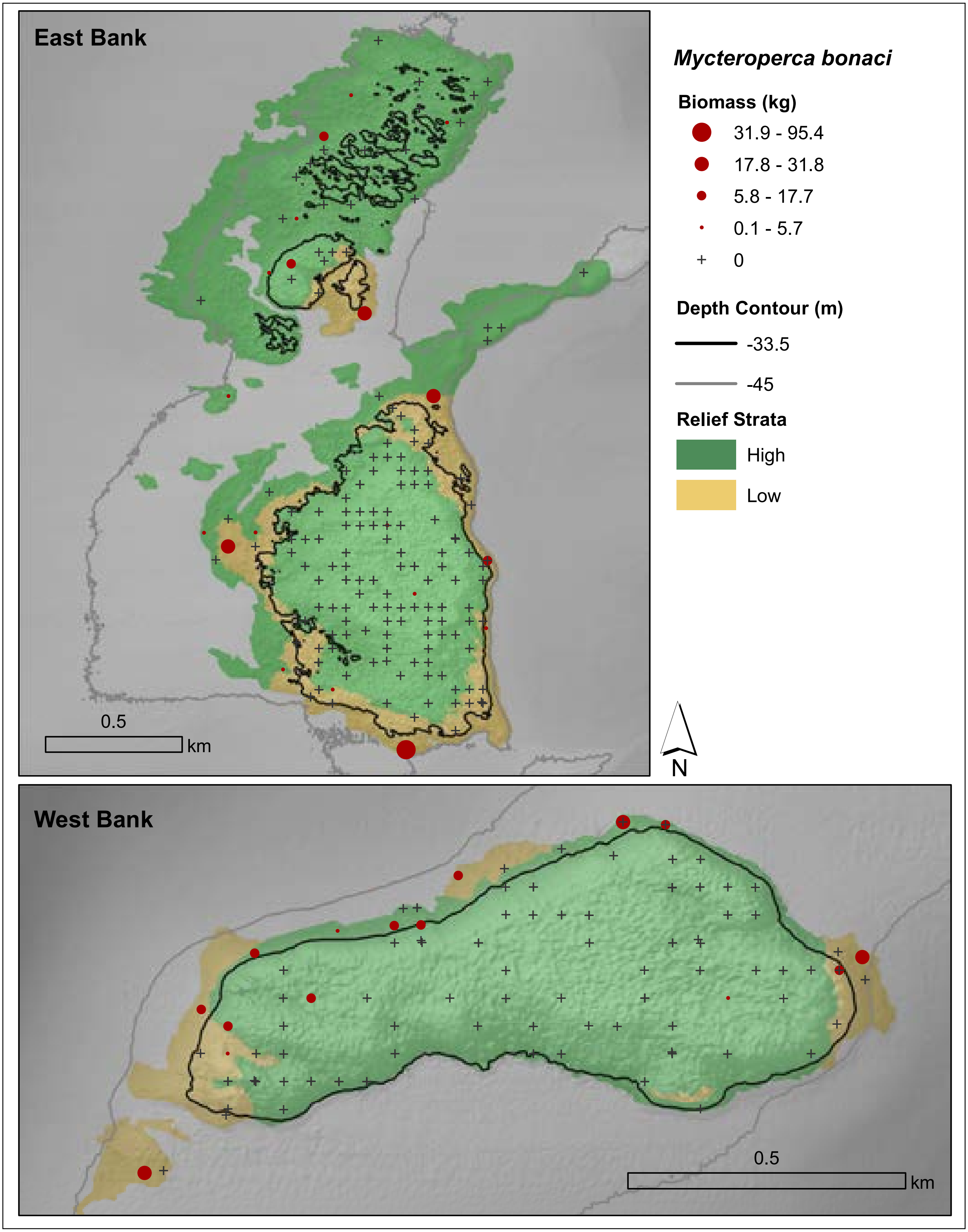

Figure 4.51. Observed biomass (kg/100 m²) of Mycteroperca bonaci *(black grouper) recorded during diver surveys from 2010-2012.*

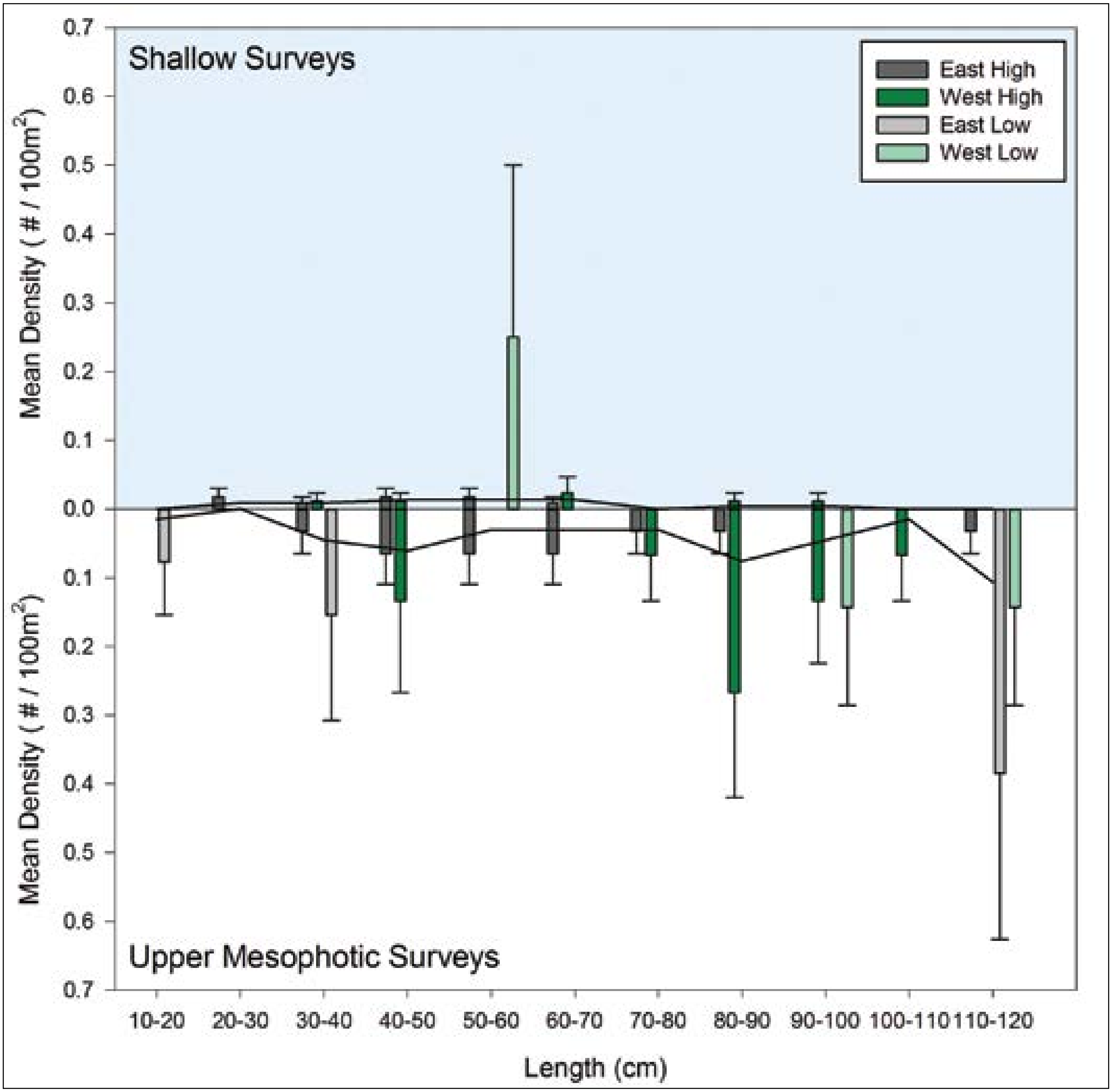

Figure 4.52. Mycteroperca bonaci *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* M. bonaci *density per size class.*

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<u>*Mycteroperca interstitialis* (yellowmouth grouper)</u>

*Figure 4.53. Color and marking variations of yellowmouth grouper (*Mycteroperca interstitialis*) in FGBNMS. Photos: M. Winfield (UNCW; top) and NOAA NOS/NCCOS/CCMA (bottom)*

Mycteroperca interstitialis was one of the more abundant grouper species; a total of 345 individuals were recorded from 174 of 291 sites (Figures 4.53 and 4.31). *M. interstitialis* were observed on both banks and within all strata (Figure 4.54), but was more frequently observed in the UM (76%, 50 of 66) than shallow (55%, 124 of 225). Overall mean density of *M. interstitialis* was highest among all serranid species (1.19 ± 0.08/100 m^2), excluding the most abundant serranid (*P. furcifer*). Density was similar between WB and EB (WB: 1.20/100 m^2; EB: 1.18/100 m^2), and maximum density of 9 individuals/100 m^2 occurred on a low relief shallow site on EB. *M. interstitialis* density positively correlated with depth ($\rho = 0.2872$, $p < 0.0001$) and were roughly two times higher in UM (1.91/100 m^2) than shallow (0.97/100 m^2). *M. interstitialis* densities were slightly higher in a previous study (1.12/100 m^2; Caldow et al., 2009) within the shallow strata. Density was negatively correlated with rugosity ($\rho = -0.7054$, $p < 0.0001$), with increasing density at low relief sites; multivariate analyses were consistent with these patterns (Tables 4.5 and 4.6). Again, these sites may be located near or at the reef-sand interface and were not generally homogeneously low relief. No significant inter-annual density differences were found.

Biomass generally increased with depth (Figure 4.55) and did not differ between banks, similar to density distribution patterns. Mean biomass over all sites was 0.77 ± 0.10 kg/100 $m^{2,}$ with higher biomass in the UM (1.7 ± 0.35 kg/100 m^2) compared to the shallow stratum (0.47 ± 0.06 kg/100 m^2; Z = 4.758, $p < 0.0001$; Figure 4.32). Biomass was positively related to depth ($\rho = 0.2955$, $p < 0.0001$) and negatively related to rugosity ($\rho = -0.1863$, $p = 0.0014$), consistent with multivariate analyses (Tables 4.5 and 4.6). The relationship with rugosity was contrary to Caldow et al. (2009) that reported lower biomass on low relief sites than high relief. Similar to *M. bonaci*, maximum biomass (20.4 kg/100 m^2) was observed at a UM low relief site on EB. No significant inter-annual biomass differences were found by depth strata (Figure 4.9).

Mean length of *M. interstitialis* was greater in UM (30.8 cm) than shallow (25.3 cm), which is lower than that reported by Caldow et al. (2009; 32 cm FL). More small individuals (<30 cm) were recorded here compared to Caldow et al. (2009; Figure 4.56). In that study, no *M. interstitialis* <10 cm were observed, here they were reported in all strata. Also, large (>60 cm FL) individuals were recorded across a wider range of strata than in Caldow et al. (2009), which recorded *M. interstitialis* >60 cm only on EB high relief. A 100 cm FL (maximum size observed during this study) *M. interstitialis* was recorded from UM depths on the EB. The majority of individuals in both depth strata were juveniles (<40 cm FL; SAFMC, 2005), while more adults were recorded in UM than shallow.

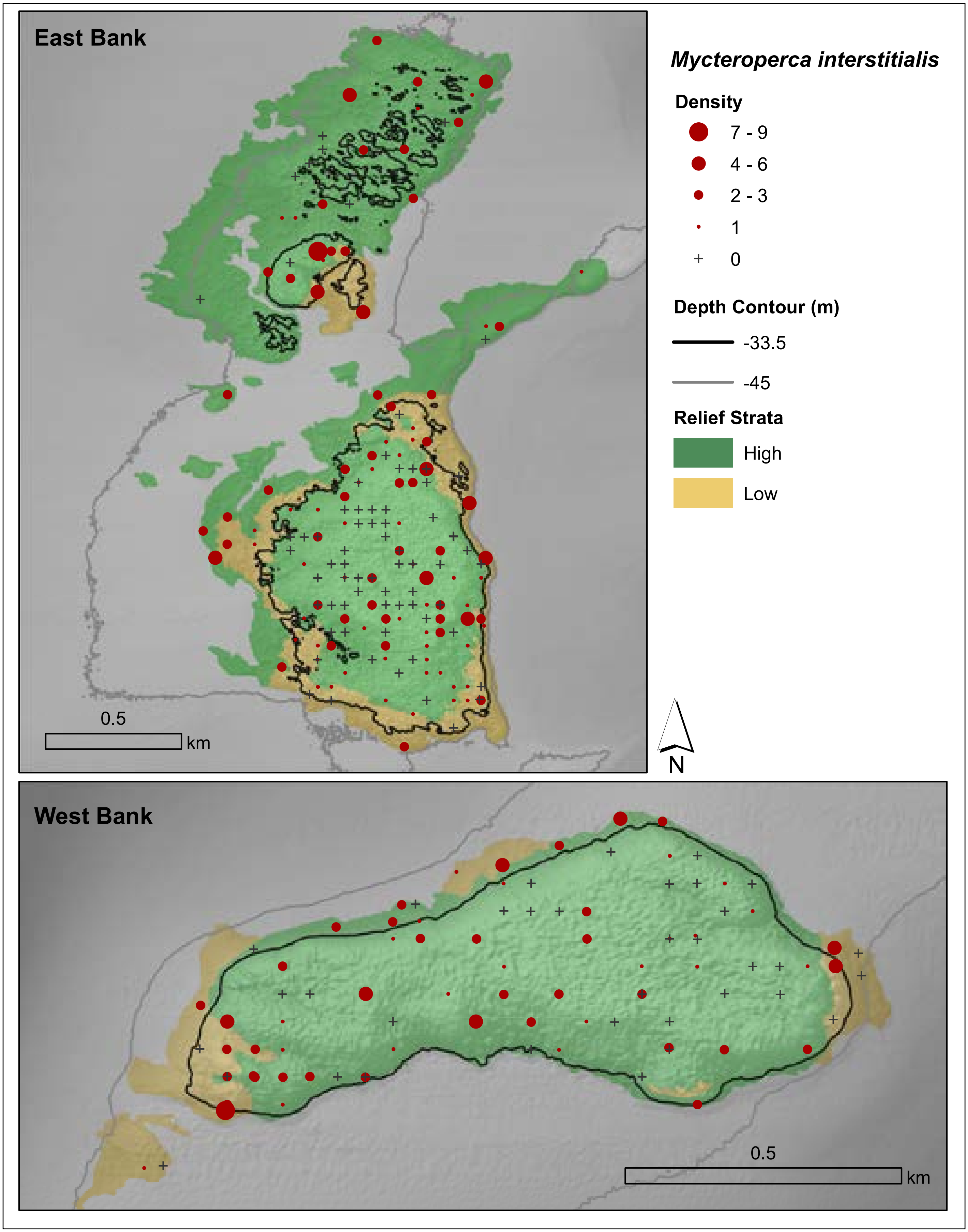

Figure 4.54. Observed density (#/100 m²) of Mycteroperca interstitialis *(yellowmouth grouper) recorded during diver surveys from 2010-2012.*

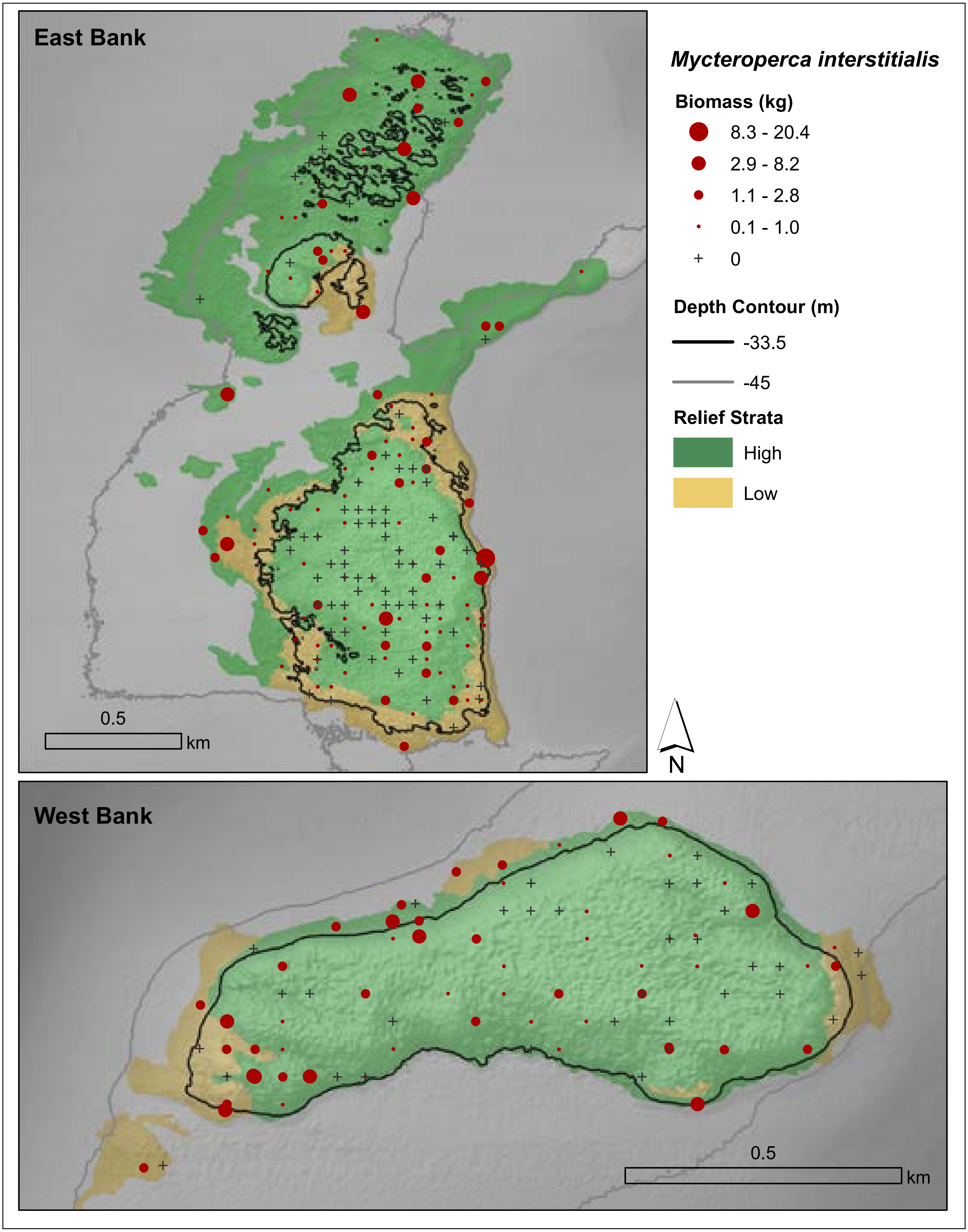

Figure 4.55. Observed biomass (kg/100 m²) of Mycteroperca interstitialis *(yellowmouth grouper) recorded during diver surveys from 2010-2012.*

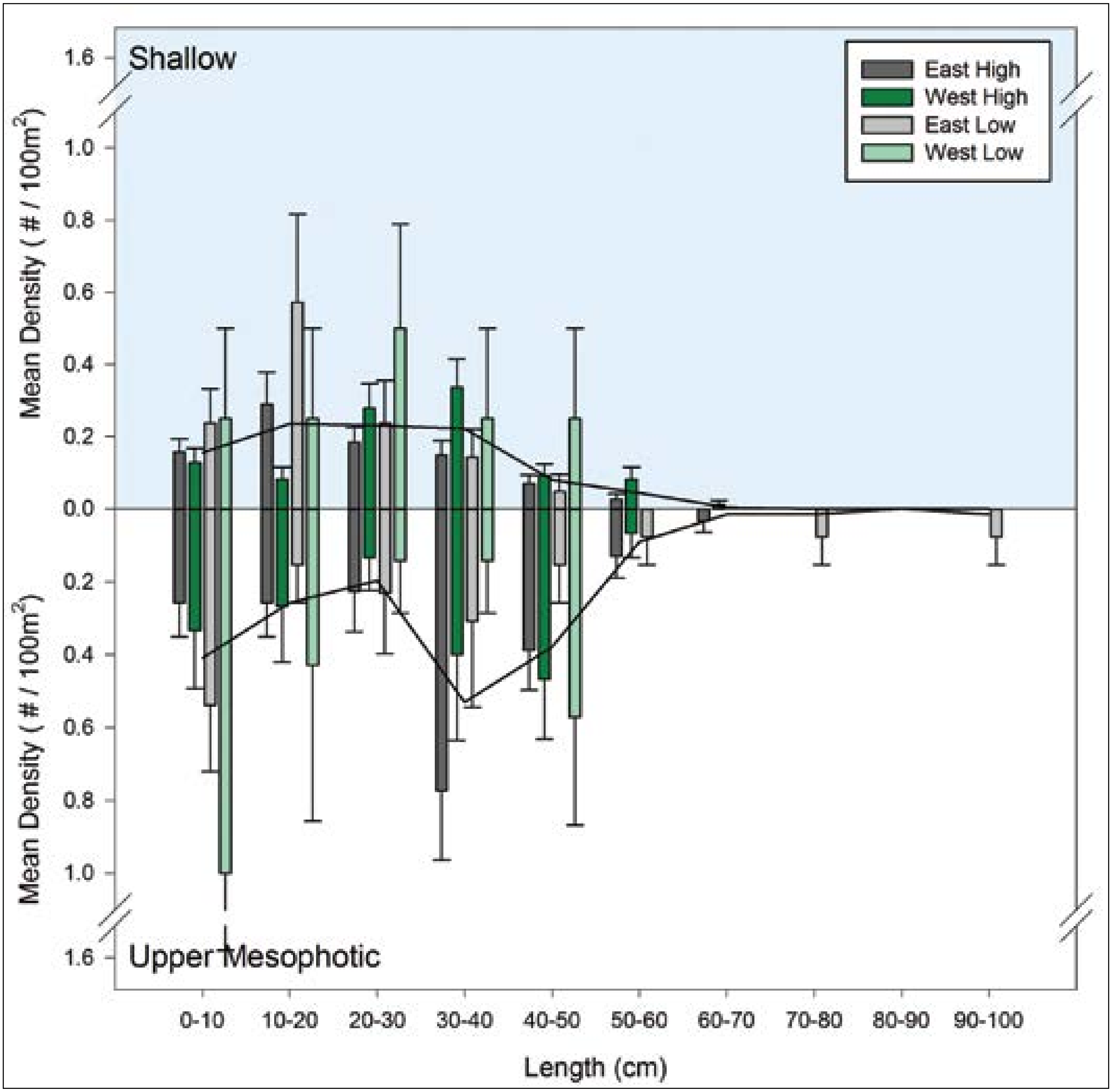

Figure 4.56. Mycteroperca interstitialis *mean density (#/100 m^2) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* M. interstitialis *density per size class.*

<u>*Mycteroperca tigris* (tiger grouper)</u>

A total of 146 *Mycteroperca tigris* were recorded at 108 of 291 sites (Figure 4.57). *M. tigris* was the fourth most abundant serranid, with mean density of 0.5 ± 0.05 individuals/100 m^2, and maximum density of 4 individuals/100 m^2 occurred on a high relief shallow site on EB (Figure 4.58). *M. tigris* were distributed on all strata (Figure 4.31) and there was no significant density difference between banks (WB: 0.52/100 m^2; EB: 0.48/100 m^2), similar to Caldow et al. (2009). Multivariate analyses showed elevated densities in UM depths and on high relief habitats (Tables 4.5 and 4.6; Figure 4.58). Correlations with depth and rugosity were consistent with the multivariate patterns (positive), but were not significant. No significant inter-annual differences in density were found.

*Figure 4.57. Color and marking variations of tiger grouper (*Mycteroperca tigris*) in FGBNMS. Photos: G. McFall (NOAA NOS/ONMS/GRNMS; top) and B. Degan (NOAA NOS/NCCOS/CCFHR (bottom)*

Total mean biomass was 0.73 ± 0.11 kg/100 m^2. Maximum biomass (13.4 kg/100 m^2) was observed at a shallow high relief site on EB, though sites with >2 kg/100 m^2 occurred within all strata; biomass did not differ between banks or depth strata (Figure 4.59). However, six of seven sites with high biomass (>6 kg/100 m^2) were in UM depths, suggesting an increasing trend with depth. Similar to density, multivariate analyses showed elevated biomass in UM depths and on high relief habitats (Tables 4.5 and 4.6), while positive correlations with depth and rugosity were not significant. No significant inter-annual differences in biomass were found within depth strata (Figure 4.9).

Although there were no univariate density or biomass trends with depth, mean fish length was larger within the UM strata (44.6 cm FL) than shallow (34.5 cm). A 90 cm FL (maximum size observed during this study) individual was recorded on the EB in the UM. Mean length recorded in shallow depths of a previous study (45.9 cm FL; Caldow et al., 2009) was greater than that recorded from either depth strata here. The larger sample size and depth range of this study likely contribute to the lower mean length estimates. *M. tigris* sizes ranged from 7.5-90 cm and were recorded in all strata (Figure 4.60). While some small *M. tigris* (<10 cm FL; n = 8) were observed, primarily within the UM strata, most were larger than 30 cm (n = 93) and could be considered adults (Heemstra and Randall, 1993). No individuals <10 cm FL were recorded by Caldow et al. (2009).

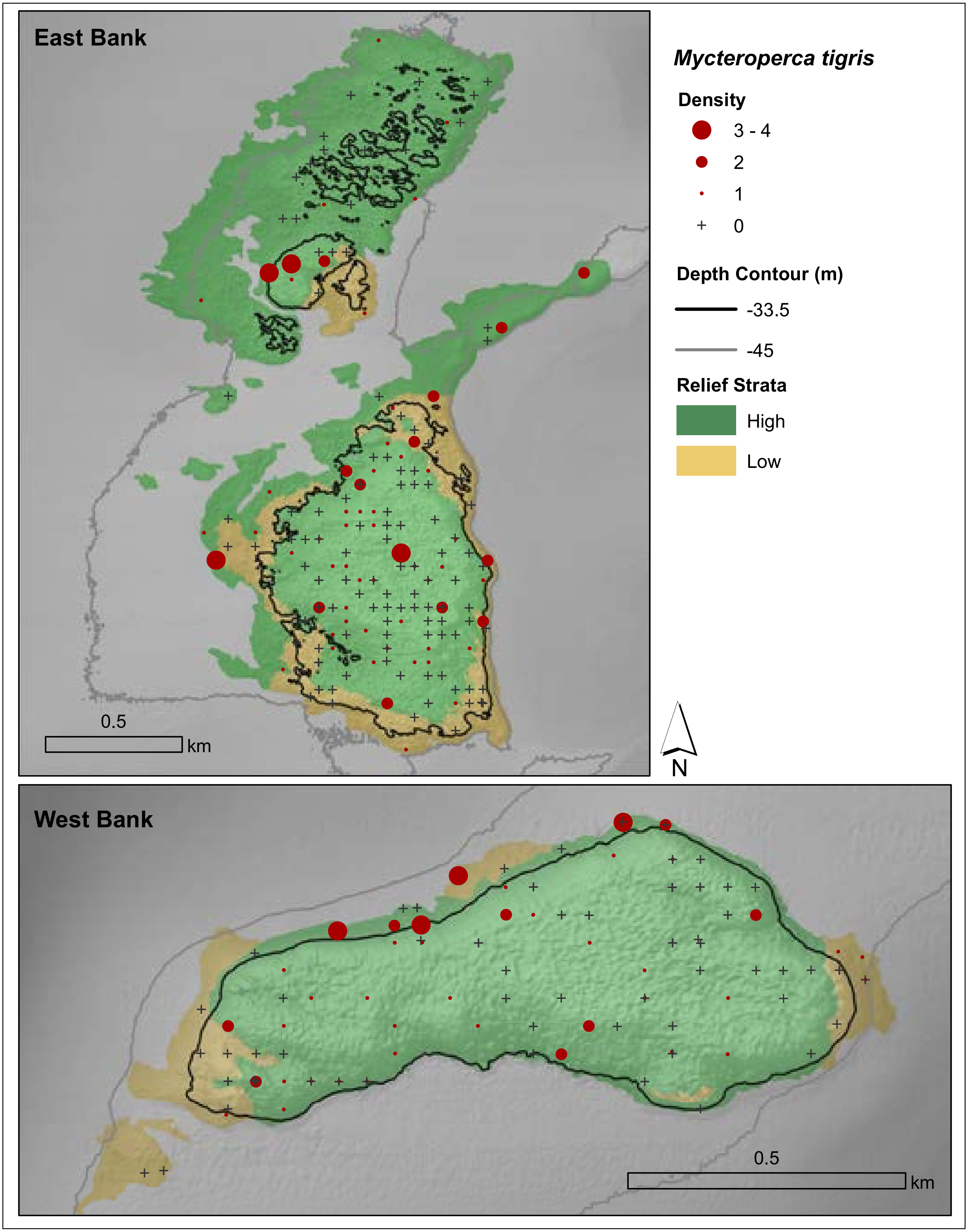

Figure 4.58. Observed density (#/100 m²) of Mycteroperca tigris *(tiger grouper) recorded during diver surveys from 2010-2012.*

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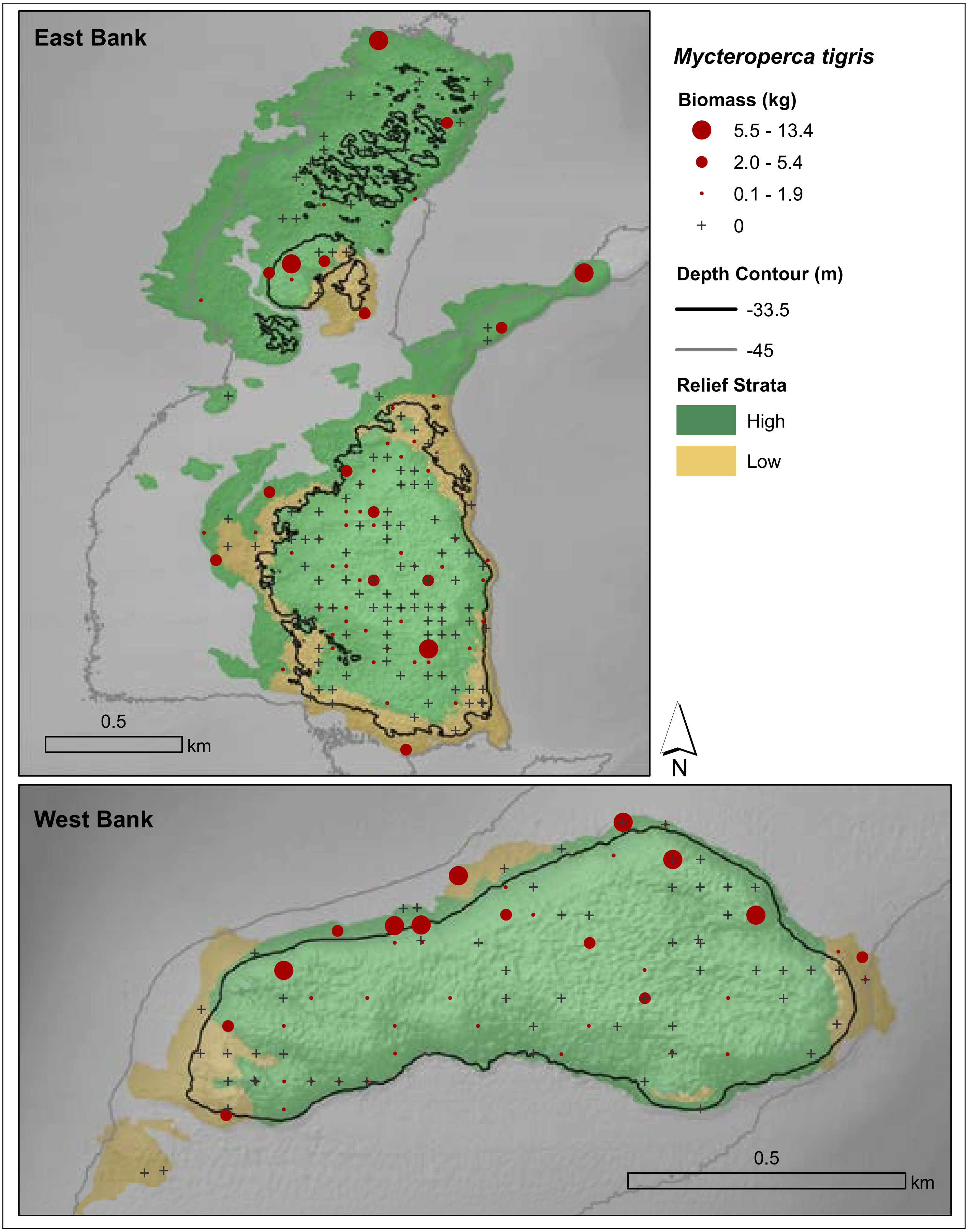

Figure 4.59. Observed biomass (kg/100 m²) of Mycteroperca tigris *(tiger grouper) recorded during diver surveys from 2010-2012.*

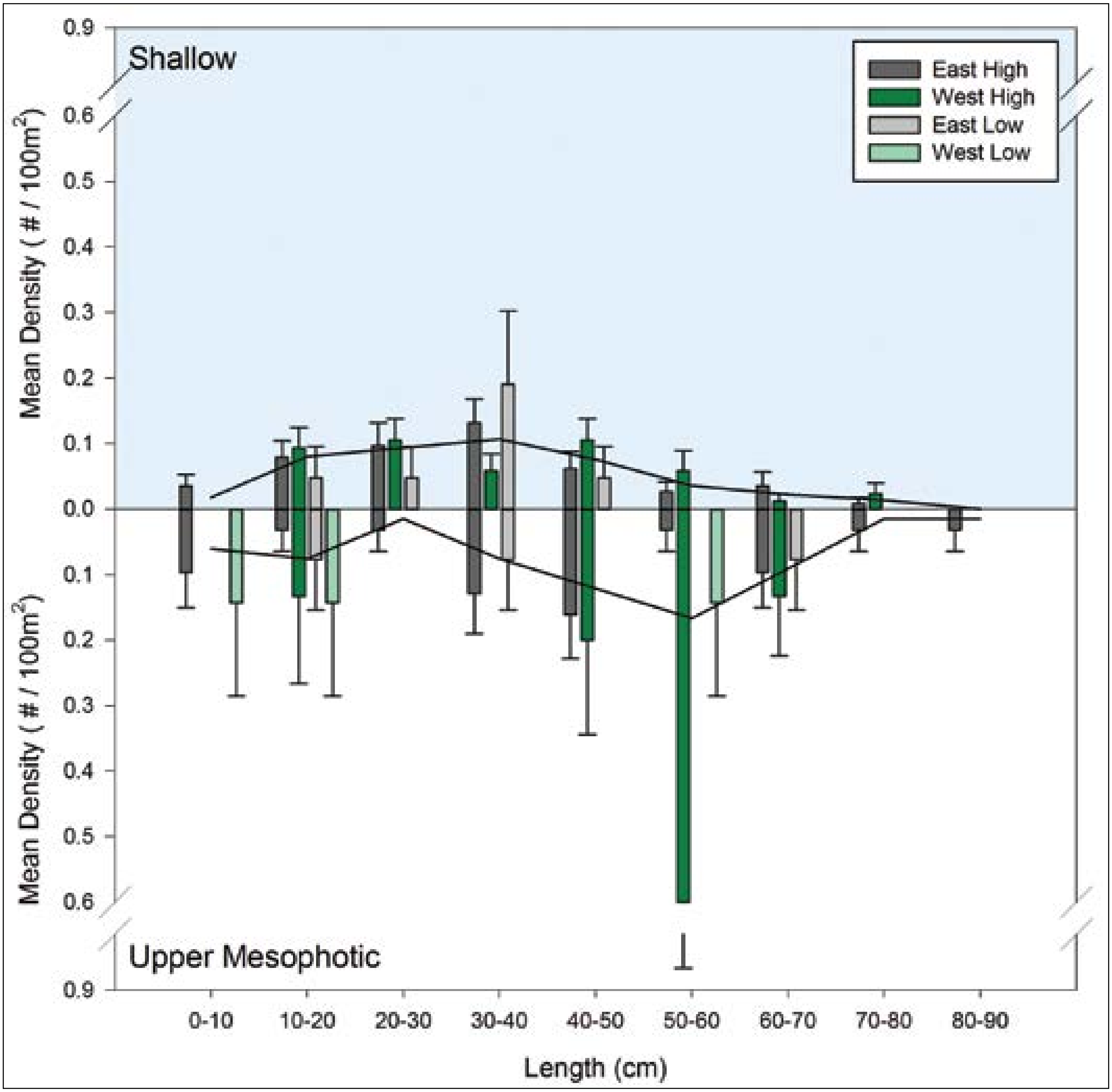

Figure 4.60. Mycteroperca tigris *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* M. tigris *density per size class.*

<u>*Paranthias furcifer* (Atlantic creolefish)</u>

*Figure 4.61. Atlantic creolefish (*Paranthias furcifer*) in FGBNMS. Photo: G. McFall (NOAA NOS/ONMS/GRNMS)*

Paranthias furcifer was the most abundant fish in this study and comprised the majority of Serranid abundance (97%). A total of 29,681 individuals were recorded at 274 of 291 sites (Figure 4.61). Overall mean density was 102 ± 13.8/100 m^2. Density was significantly higher in the UM (192 ± 11.9/100 m^2) than in the shallow strata (101 ± 31.3/100 m^2; Z = 2.58899, p = 0.0096; Figure 4.30). There were no significant density differences between banks (WB: 127 ± 30.4/100 m^2; EB: 86.3 ± 12.0/100 m^2) or relief strata (High: 102 ± 15.4/100 m^2; Low: 101 ± 31.3/100 m^2; Figure 4.62). Multivariate analyses showed elevated density in the UM strata and also on high relief habitats (Tables 4.5 and 4.6), consistent with the positive correlation with depth (ρ = 0.2015, p = 0.0005); however, density decreased with rugosity based on univariate analyses that did not control for depth (ρ = -0.119, p = 0.0425). Density within the shallow strata was greater (WB: 87.4 ± 26.5/100 m^2; EB: 67.5 ± 9.3/100 m^2) than that observed by Caldow et al. (2009) for the same depths (WB: 32.57/100 m^2; EB: 48.53/100 m^2). There were no significant inter-annual density differences observed (Figure 4.8).

Biomass results were similar to those observed for density. *P. furcifer* comprised 73.2% of all serranid biomass. Overall mean biomass was 9.95 ± 1.7 kg/100 $m^{2,}$ with significantly higher biomass in the UM strata (19.3 ± 6.1 kg/100 m^2) than in the shallow strata (7.2 ± 1.2 kg/100 m^2; Z = 2.078, p = 0.0376; Figure 4.30). There was no significant biomass difference by bank (WB: 12.3 ± 3.5 kg/100 m^2; EB: (8.5 ± 1.6 kg/100 m^2) or by relief strata (High: 10.2 ± 1.9 kg/100 m^2; Low: 8.3 ± 3.4 kg/100 m^2; Figure 4.63). Biomass increased significantly with depth (ρ = 0.1544, p = 0.0083), with no significant relationship to rugosity. In the shallow strata, only 2011 biomass was significantly greater than in 2010 (χ^2 = 7.2478, p = 0.0267); 2012 biomass was similar to previous years for both depth strata (Figure 4.9).

P. furcifer were recorded in a range of sizes (3-32.5 cm FL), with most size classes occurring in all strata (Figure 4.64). Density was highest in the 15-20 cm size class, especially in the UM strata (Figure 4.64). Mean length was similar in both depth strata (shallow: 17.25 cm; UM: 17.5 cm). These patterns are similar to observations made by Caldow et al. (2009), which identified a greater number of individuals between 15-25 cm FL, and reported a mean length of 18.3 cm.

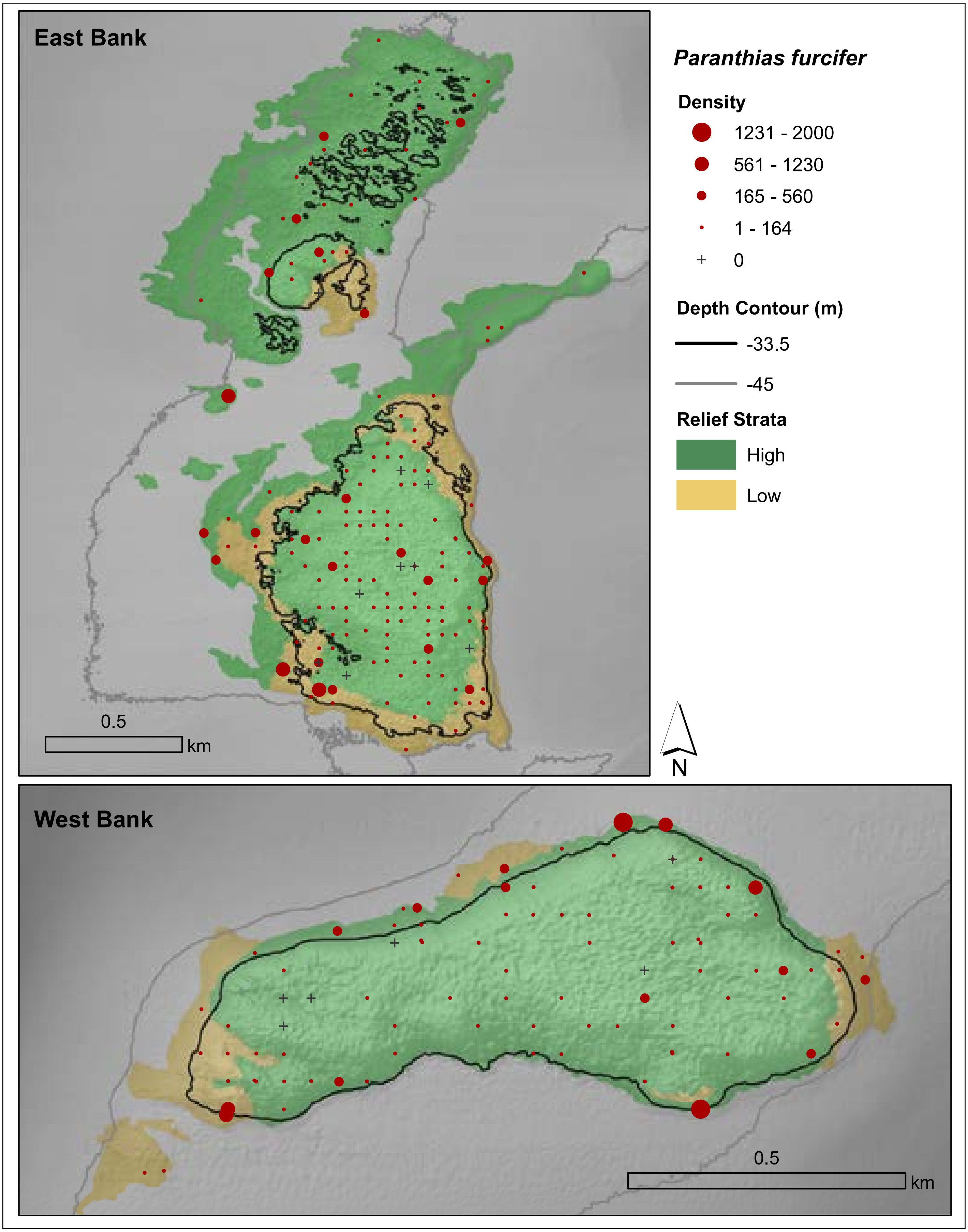

Figure 4.62. Observed density (#/100 m²) of Paranthias furcifer *(Atlantic creolefish) recorded during diver surveys from 2010-2012.*

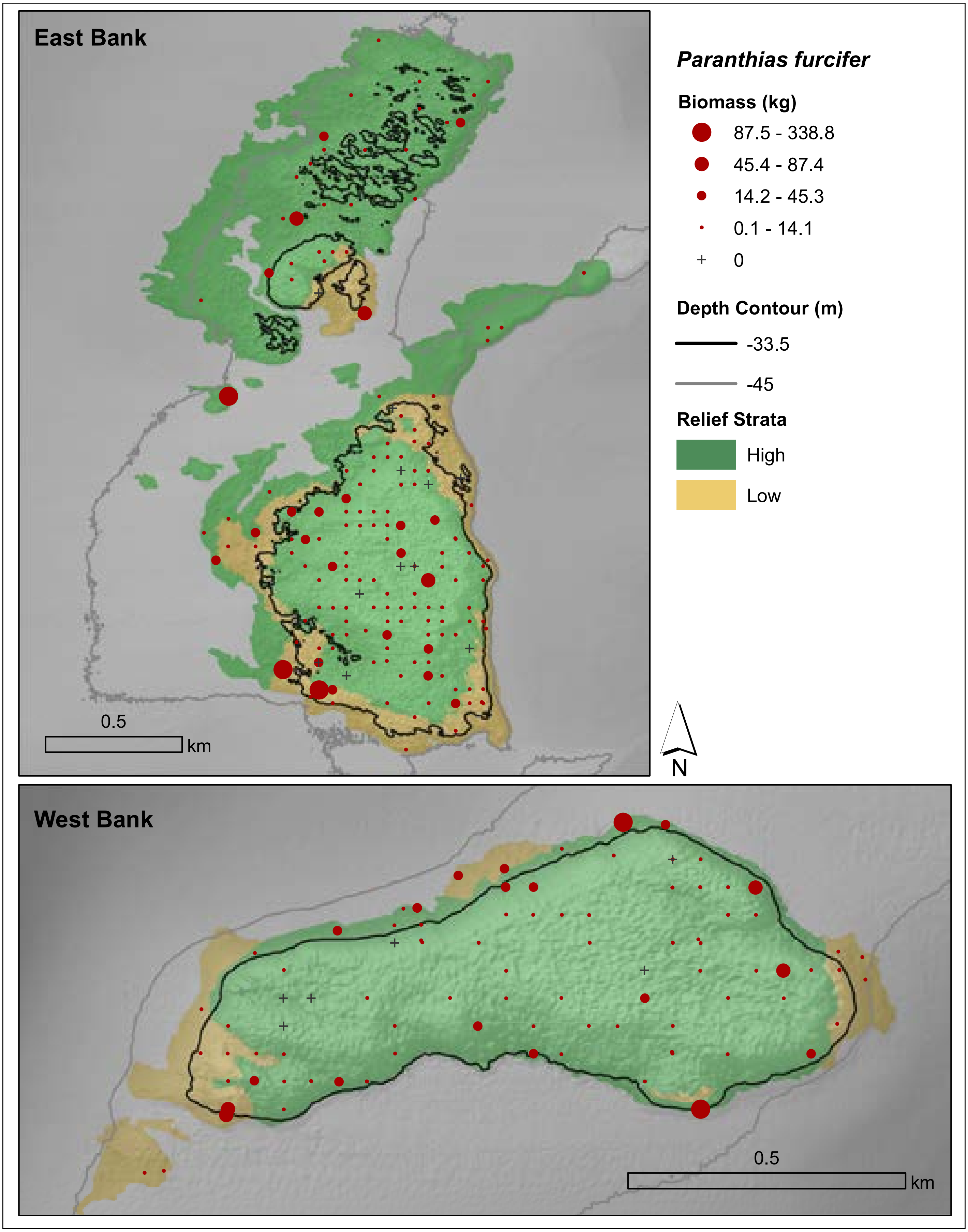

Figure 4.63. Observed biomass (kg/100 m²) of Paranthias furcifer *(Atlantic creolefish) recorded during diver surveys from 2010-2012.*

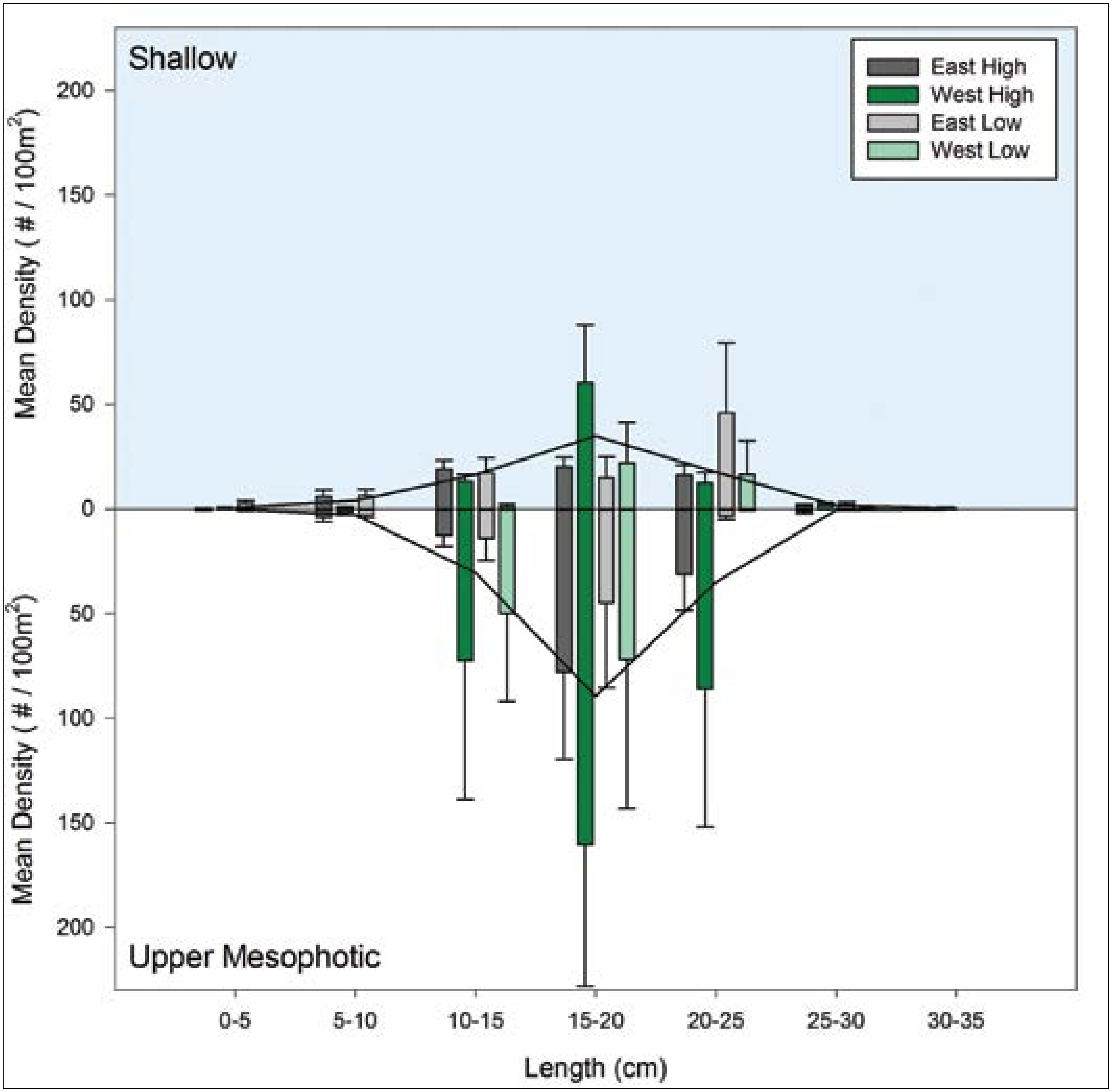

Figure 4.64. Paranthias furcifer *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* P. furcifer *density per size class.*

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Lutjanidae (Snappers)

*Figure 4.65. Dog snapper (*Lutjanus jocu*), an example of a species from the Family Lutjanidae in FGBNMS. Photo: NOAA NOS/NCCOS/CCMA*

Lutjanids were the 13th most abundant family (0.66% of total density) and ranked 6th in biomass (6.3% of total; Figure 4.65). Five lutjanid species were recorded during diver surveys (Figures 4.66 and 4.67), all but *Ocyurus chrysurus* (yellowtail snapper) have been reported in previous surveys (Caldow et al., 2009; Pattengill-Semmens and Semmens, 1998). Two *O. chrysurus* were reported on a single EB shallow high relief survey site. Nearshore *O. chrysurus* densities have been increasing (Fodrie et al., 2010), so it is possible they will become a more regular occurrence within FGBNMS.

Highest lutjanid densities and biomass occurred at depths greater than 27 m (Figures 4.68 and 4.69); however, some intermediate densities (21-45 individuals/100 m^2) did occur at sites less than 27 m depth. Maximum lutjanid density (100 individuals/100 m^2) and biomass (129.4 kg/100 m^2) recorded here were higher than a previous survey (maximum n = 12/100 m^2; max biomass = 23.9 kg/100 m^2; Caldow et al., 2009). Spatial patterns of biomass concentrations (Figure 4.69) were similar to Caldow et al. (2009). The additional sample size and depth range of this survey clearly identify the highest lutjanid densities and biomass along the transition between depth strata and at deeper UM sites, similar to the distribution patterns of serranids. The relationship between lutjanid density or biomass and depth was not significant; however, density did significantly increase with rugosity ($\rho = 0.1308$, $p = 0.0256$), but no significant trend was observed with biomass.

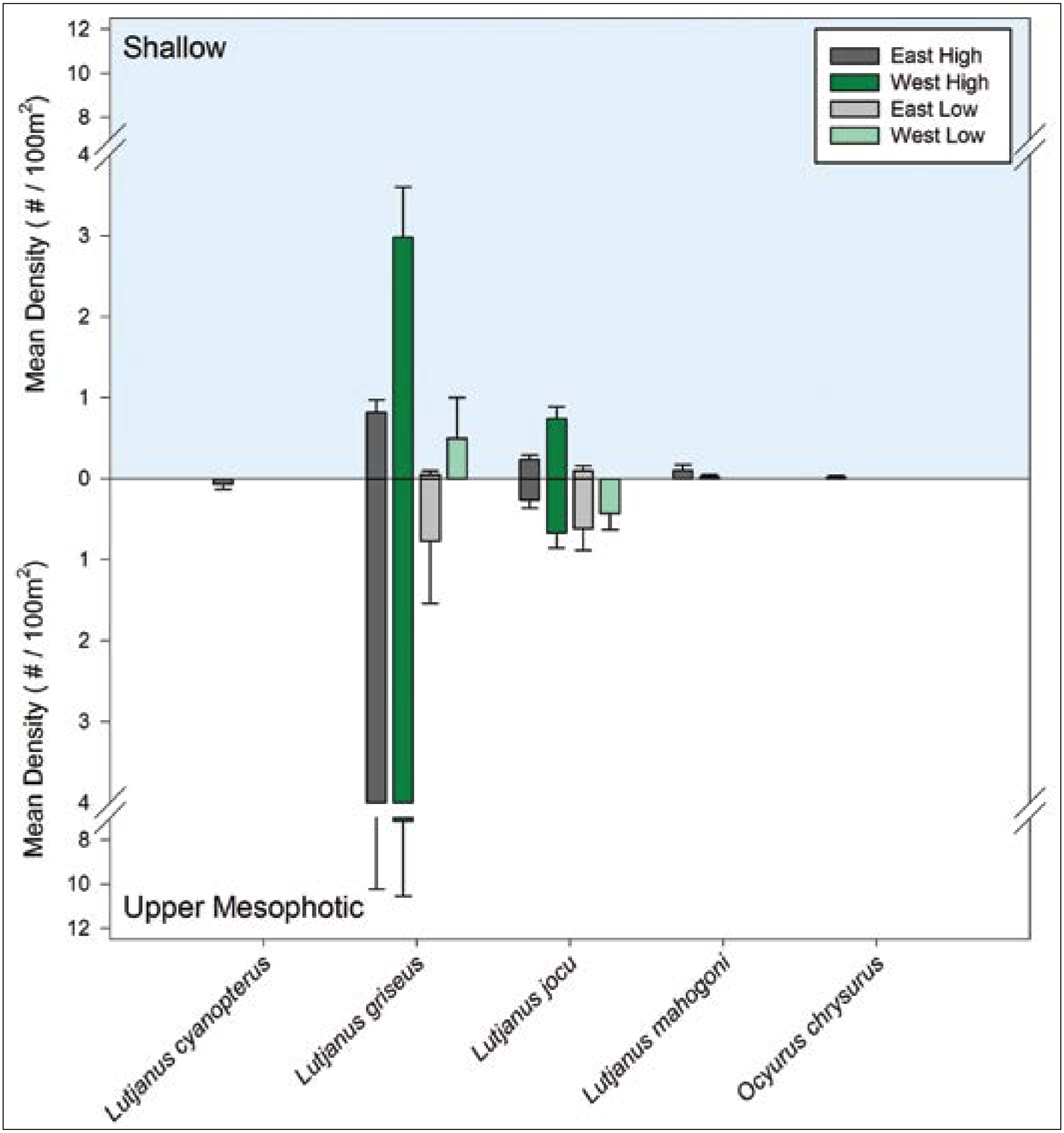

Figure 4.66. Mean density (#/100 m²) of Lutjanidae species by strata for dive surveys (2010-2012).

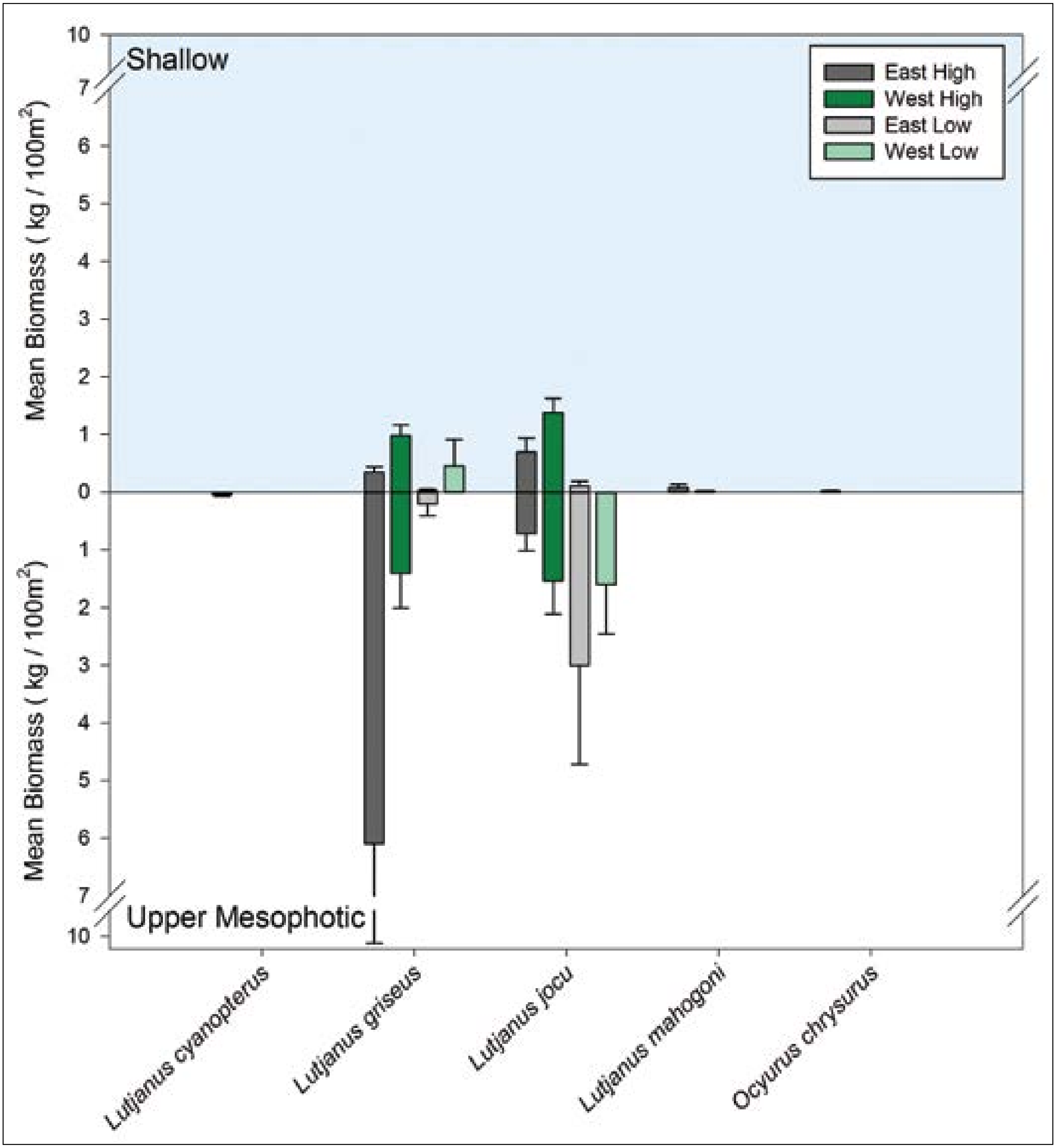

Figure 4.67. Mean biomass (kg/100 m^2) of Lutjanidae species by strata for dive surveys (2010-2012).

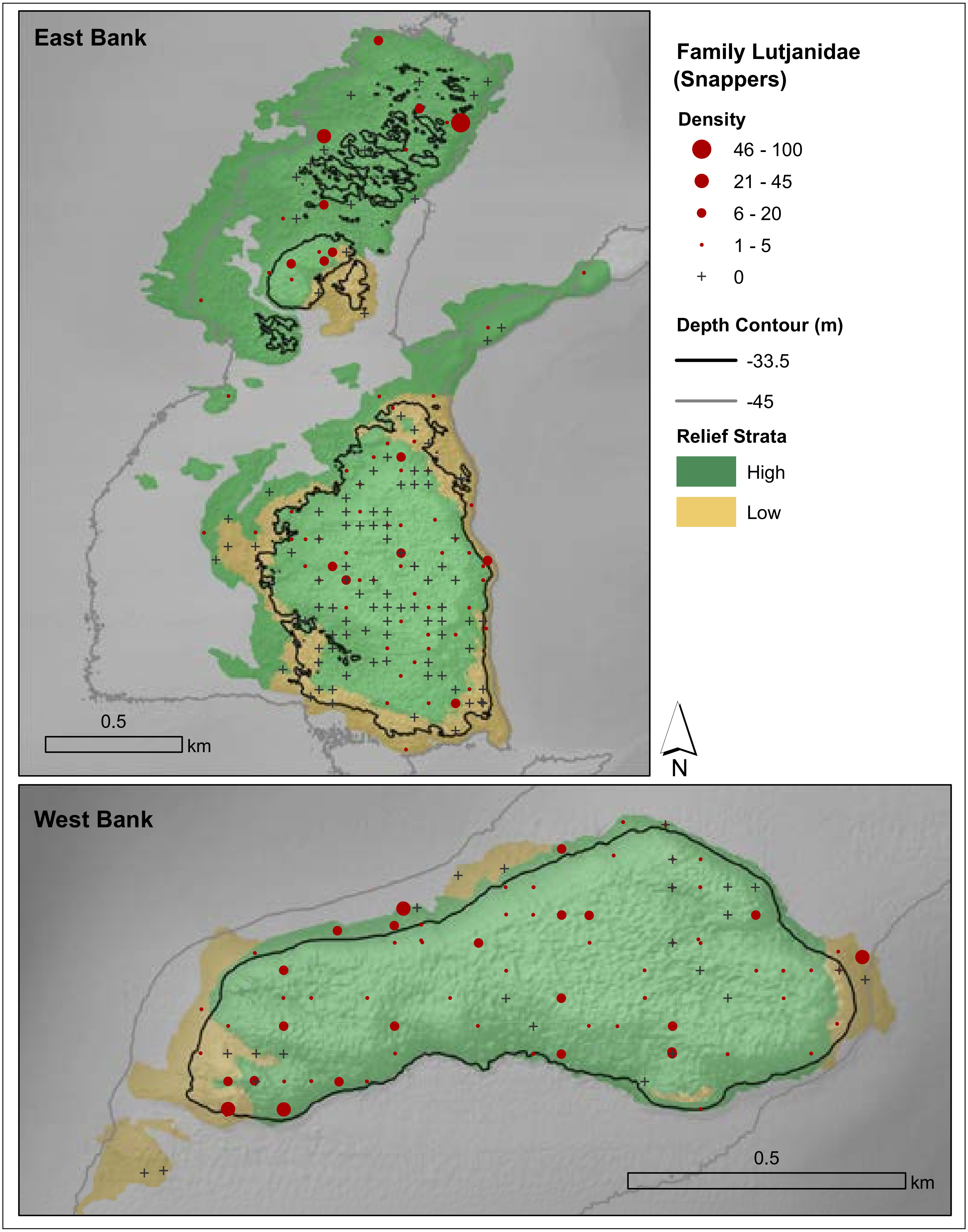

Figure 4.68. Observed density (#/100 m²) of Lutjanidae recorded during diver surveys from 2010-2012.

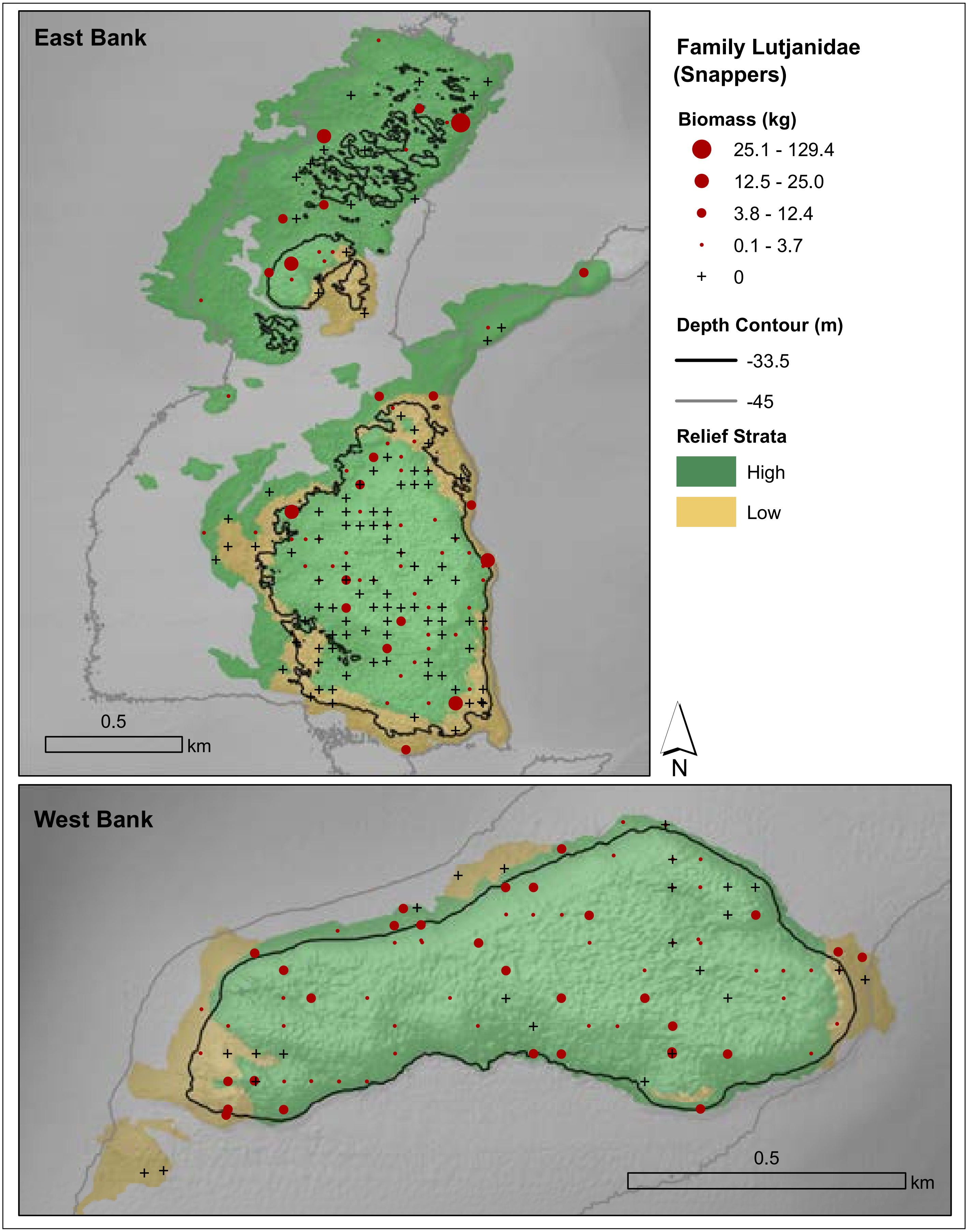

Figure 4.69. Observed biomass (kg/100 m²) of Lutjanidae recorded during diver surveys from 2010-2012.

<u>*Lutjanus griseus* (gray snapper)</u>

Lutjanus griseus was the most abundant lutjanid; a total of 678 individuals were recorded from 110 of 291 sites (Figure 4.70). They were predominantly found on high relief habitats (ρ = 0.1372, p = 0.0192; Figure 4.66). While there was no significant density trend with depth, some higher densities were recorded in the UM strata, particularly in high relief habitats (Figure 4.71). *L. griseus* exhibited an overall mean density of 2.33 ± 0.47/100 m^2, and were significantly more abundant on WB than on EB (WB: 3.26 ± 0.67/100 m^2, EB :1.75 ± 0.63/100 m^2; Z = 4.653, p < 0.0001). Maximum density of 100 individuals/100 m^2 occurred on a high relief UM site on EB. Shallow stratum density was significantly lower in 2011 compared to 2010 (χ^2 = 9.2243, p = 0.0099), while 2012 density was similar to previous years. No inter-annual differences were found within the UM stratum.

*Figure 4.70. Gray snapper (*Lutjanus griseus*) in FGBNMS. Photo: J. Voss (HBOI/FAU and NOAA CIOERT)*

Within both shallow and UM strata, densities in high relief habitat were significantly greater than in low relief (shallow: Z = -3.379, p = 0.0007; UM: Z = -3.061, p = 0.002). Shallow strata densities at WB were significantly greater than on EB (WB: 2.9 ± 5.7/100 m^2; EB: 0.70 ± 1.6/100 m^2; Z = 5.217, p < 0.0001), but were not significantly different at UM depths. Similar patterns of density, relief, and bank were observed by Caldow et al. (2009). Mean density of *L. griseus* in shallow depths was significantly lower in 2011 (0.55 ± 0.11 individuals/100 m^2) compared to 2010 (2.28 ± 0.59 individuals/100 m^2; χ^2 = 9.22, p = 0.01). Interestingly, shallow strata *L. griseus* occurrences and densities have consistently been low (five individuals in four surveys in 1999, Marks, 2007; 2 individuals in 96 surveys in 2006-2007, Zimmer et al., 2010; 65 individuals in 73 surveys from 2006-2007, Caldow et al., 2009; 21 individuals in 32 LTM surveys in 2009-2010, Johnston et al., 2013) but may be increasing (this study 352 individuals in 225 surveys from 2010-2012). While this may be an artifact of sampling approach or effort, it is consistent with dramatic increases of *L. griseus* densities in seagrass beds within the Gulf of Mexico. A 105% increase in *L. griseus* relative abundance from 1971-1979 surveys to 2006-2007 was recorded in a recent study (Fodrie et al., 2010), which also proposed warming ocean waters may result in a continued increase in *L. griseus* densities within the Gulf of Mexico.

Among Lutjanids, *L. griseu*s biomass was highest (Figure 4.67), with overall mean 1.16 ± 0.46 kg/100 m^2. Biomass was greatest in depths below 33.5 m (Figure 4.72), but the trend with depth was not significant. Maximum site biomass (129.4 kg/100 m^2) was observed at a UM high relief site on EB. Mean EB biomass (1.29 ± 0.74 kg/100 m^2) was significantly greater than on WB (0.96 ± 0.17 kg/100 m^2, Z = 4.52, p < 0.0001). Maximum biomass and density reported here were higher than in Caldow et al. (2009; 15.5 kg/100 m^2; 12/100 m^2). Like density, biomass significantly increased with rugosity (ρ = 0.1294, p = 0.0273). Multivariate analyses showed elevated density and biomass in UM depths and on high relief habitats (Tables 4.5 and 4.6). Shallow biomass was significantly lower in 2011 compared to 2010 (χ^2 = 10.925, p = 0.004; Figure 4.9), while 2012 biomass was similar to previous years. No inter-annual differences were found within the UM stratum.

Individuals of *L. griseus* ranged in size from 10 to 60 cm (Figure 4.73). In the UM strata, smaller fish were found on WB high relief habitats, while larger fish were recorded on EB high relief; overall, mean fish length was larger in the UM (33.3 cm FL) than shallow (26.8 cm). Caldow et al. (2009) recorded a mean length of 37.8 cm FL from shallow depths, exceeding those recorded here from both depth strata. This difference may be due to the extended spatial coverage and greater number of smaller individuals observed in this study. Length distribution within the shallow strata here peaked at a smaller size (20-25 cm) compared to the previous report (30-40 cm; Caldow et al., 2009); however, in the present study more large fish were observed in the UM strata which was not surveyed by Caldow et al. (2009).

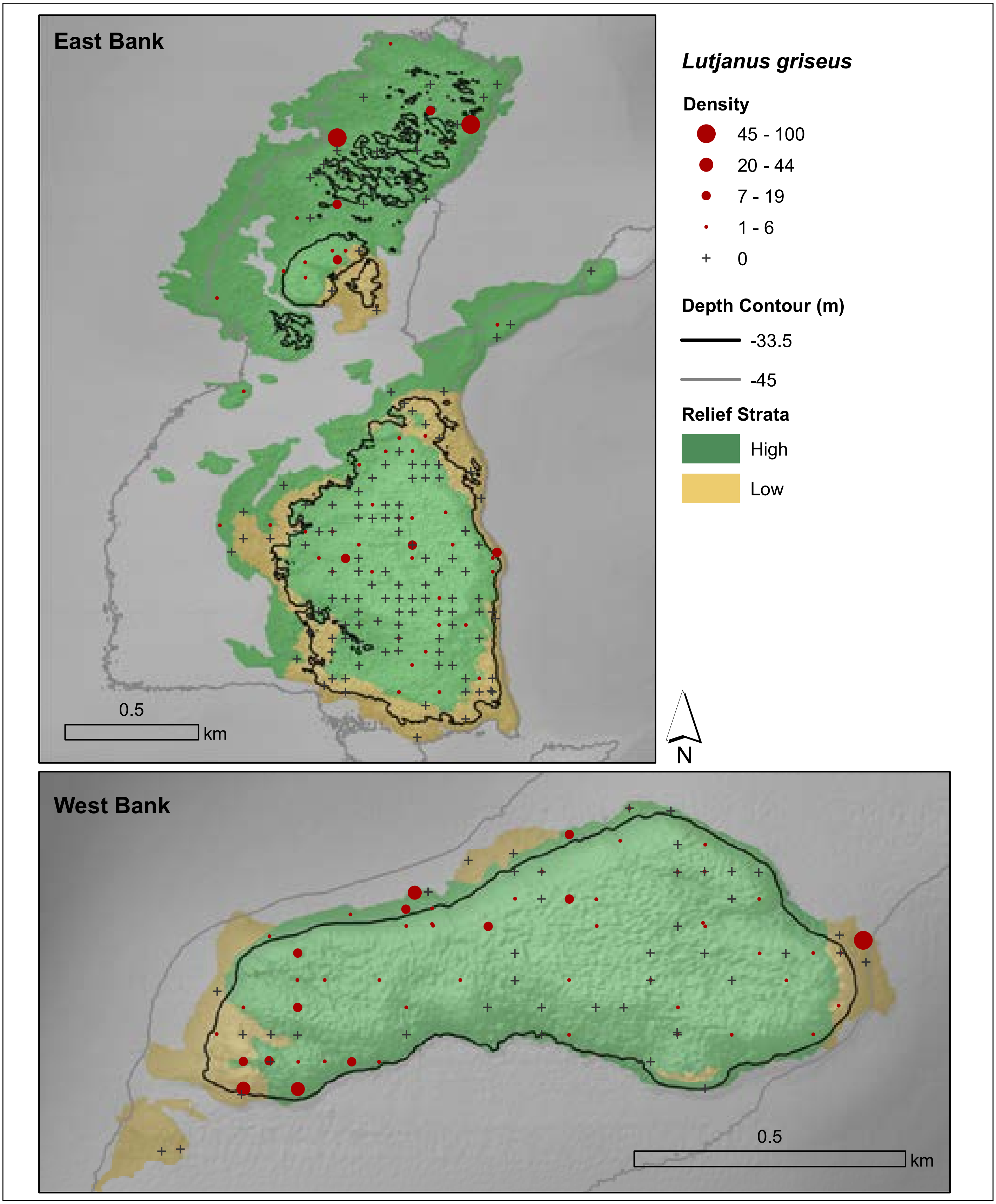

Figure 4.71. Observed density (#/100 m²) of Lutjanus griseus *(gray snapper) recorded during diver surveys from 2010-2012.*

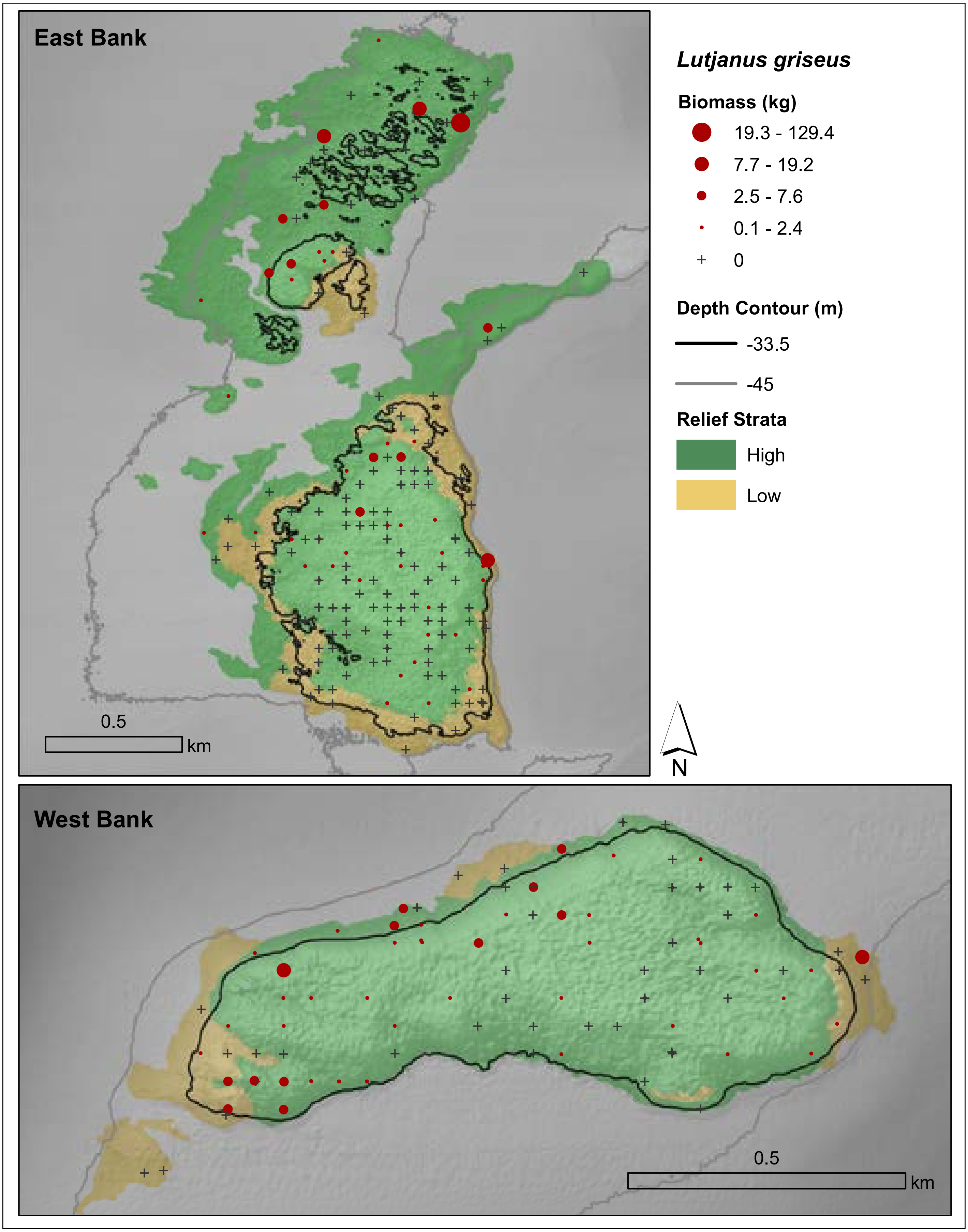

Figure 4.72. Observed biomass (kg/100 m^2) of Lutjanus griseus *(gray snapper) recorded during diver surveys from 2010-2012.*

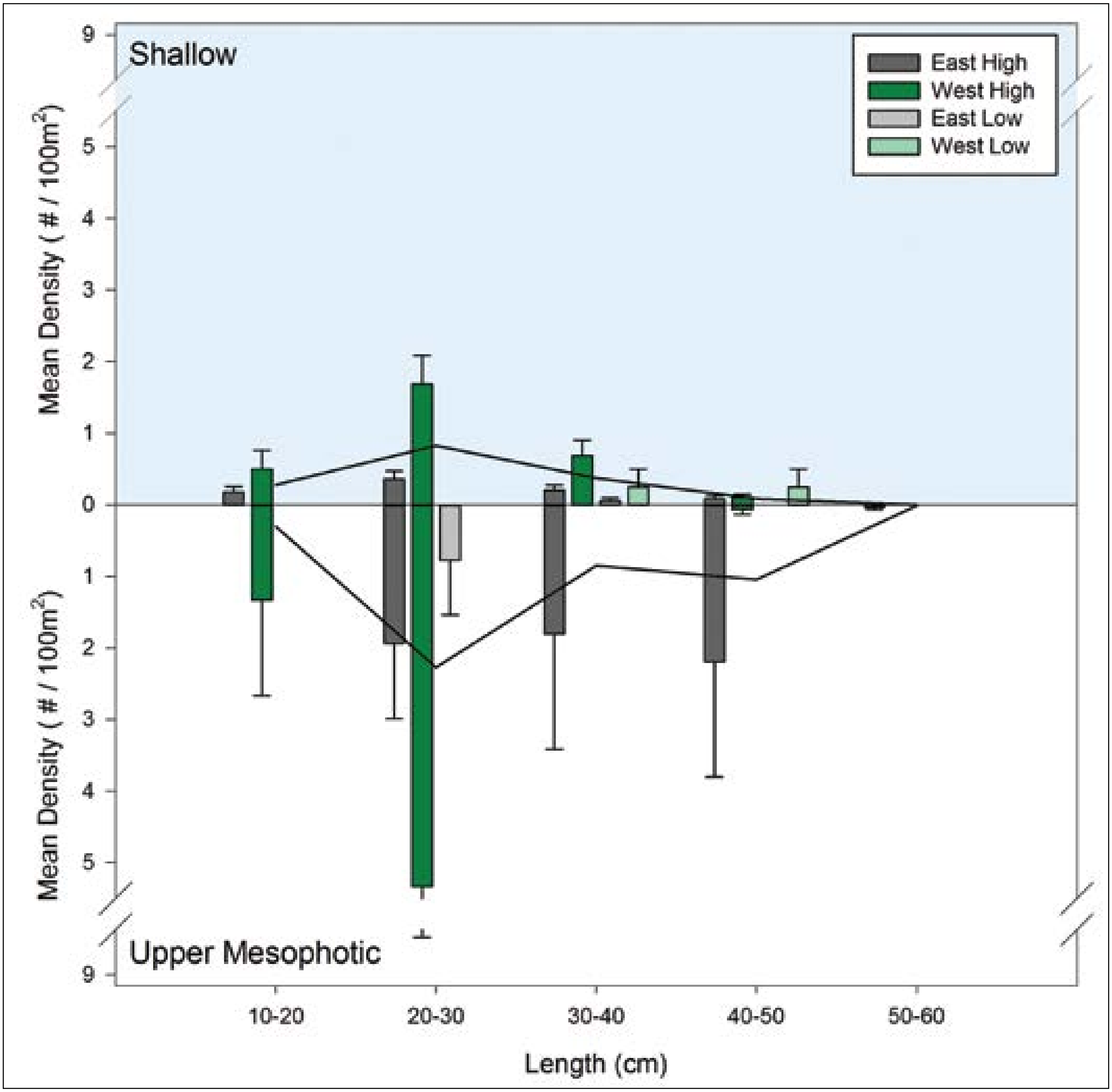

Figure 4.73. Lutjanus griseus *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* L. griseus *density per size class.*

Lutjanus jocu (dog snapper)

A total of 122 *Lutjanus jocu* were recorded at 78 sites (Figure 4.74). *L. jocu* were reported more frequently in the UM strata (33.3%) than shallow (24.8%). While *L. jocu* was the second most abundant lutjanid (0.42 ± 0.05/100 m^2; Figure 4.66), its overall mean biomass (1.01 ± 0.15 kg/100 m^2) was comparable to *L. griseus* (1.2 kg/100 m^2). For all strata combined, *L. jocu* density was higher on WB (Figure 4.75; 0.69 ± 0.11/100 m^2) than on EB (0.25 ± 0.04/100 m^2; Z = 4.1025, p < 0.0001). Maximum density of 7 individuals/100 m^2 occurred on a high relief shallow site on WB. UM sites had similar densities on each bank, whereas within the shallow strata, WB densities (0.71 ± 0.14/100 m^2) were greater than EB (0.21 ± 0.05/100 m^2; Z = 3.76, p = 0.0002). The opposite pattern was reported by Caldow et al. (2009), likely driven by the limited number of surveys completed on WB in the previous study. Some of the higher density sites (≥ 6 individuals/100 m^2) were found on WB, particularly within the high relief strata (Figure 4.75). Multivariate analyses showed elevated densities in UM depths and on high relief habitats (Tables 4.5 and 4.6). Correlations with depth and rugosity were consistent with the multivariate patterns (positive), but were not significant. No significant inter-annual differences in density were found by depth strata.

*Figure 4.74. Dog snapper (*Lutjanus jocu*) in FGBNMS. Photo: J. Voss (HBIO/FAU and NOAA CIOERT)*

L. jocu biomass was significantly greater on WB (Figure 4.76 ; 1.36 ± 0.21 kg/100 m^2) than on EB (0.80 ± 0.21 kg/100 m^2; Z = 4.11582, p < 0.0001). Biomass ranged from 0-22.4 kg/100 m^2, which is greater than that observed by Caldow et al. (2009; 0-12.7 kg/100 m^2). Biomass was greater in UM (1.45 ± 0.41 kg/100 m^2) than shallow strata (0.89 ± 0.16 kg/100 m^2), however, the difference was not significant. Within the shallow strata, mean biomass of *L. jocu* was significantly greater in high (0.99 ± 0.18 kg/100 m^2) versus low (0.09 ± 0.07 kg/100 m^2) relief habitats (Z = -2.18, p = 0.03). The opposite pattern occurred in UM strata but the difference was not significant (Figure 4.67). Although the highest biomass site occurred on a low relief site within the UM, biomass was not significantly different between relief strata or correlated with depth or rugosity. Multivariate analyses showed elevated biomass in UM depths on low relief habitats (Tables 4.5 and 4.6). No significant inter-annual differences in biomass were found (Figure 4.9).

Overall, there were larger *L. jocu* in the UM strata (mean FL: 51.6 cm; Figure 4.77). Mean length within the shallow strata (45.6 cm) was similar to that observed by Caldow et al. (2009; 44.6 cm FL). Within all size classes, more fish were recorded in high relief habitats than low relief (Figure 4.77), similar to Caldow et al. (2009). However, the largest individuals (size class 70-80 cm FL) were primarily in low relief habitats of the UM. Nevertheless, an 80 cm FL (maximum size observed during this study) *L. jocu* was recorded from shallow depths on the EB.

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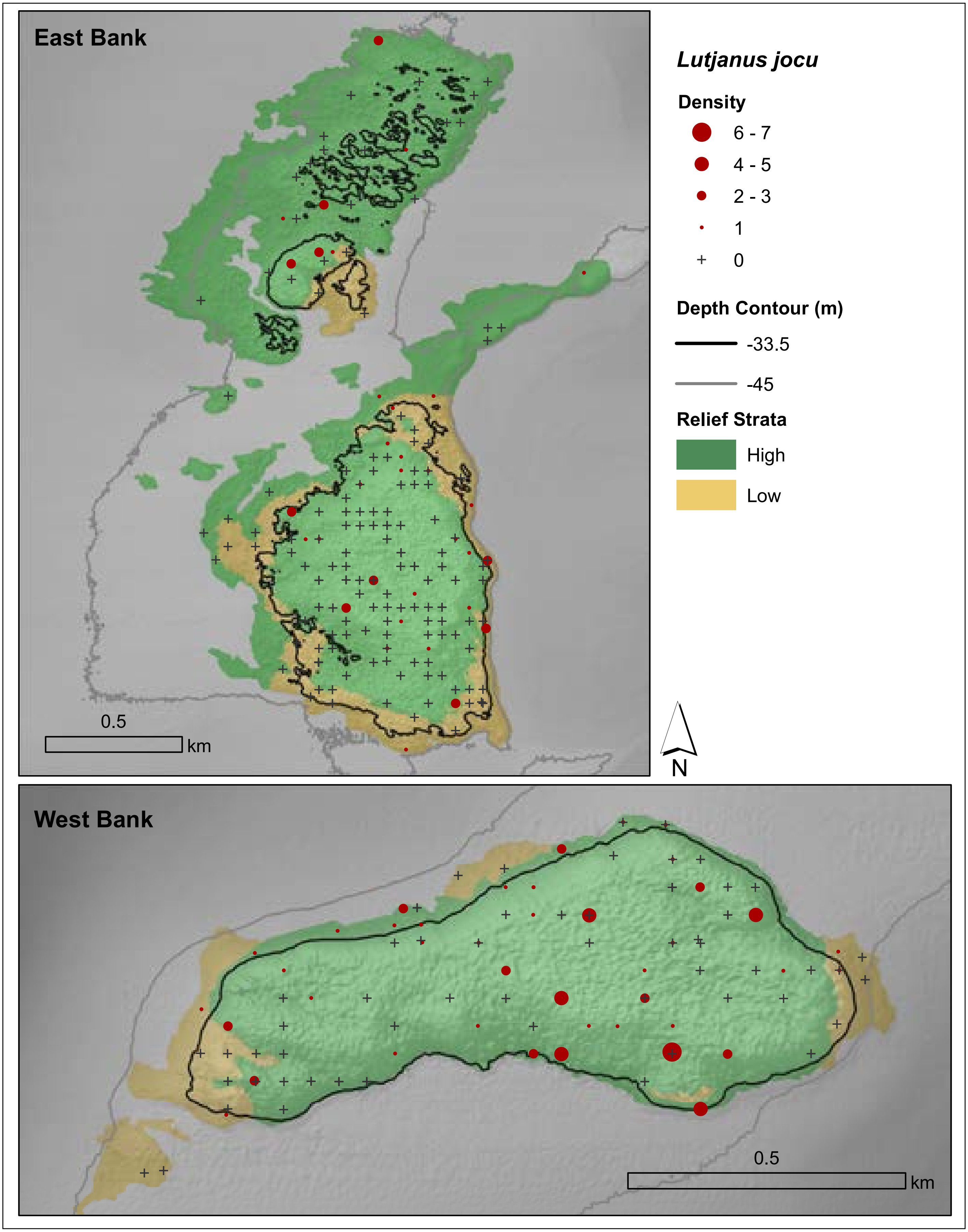

Figure 4.75. Observed density (#/100 m²) of Lutjanus jocu *(dog snapper) recorded during diver surveys from 2010-2012.*

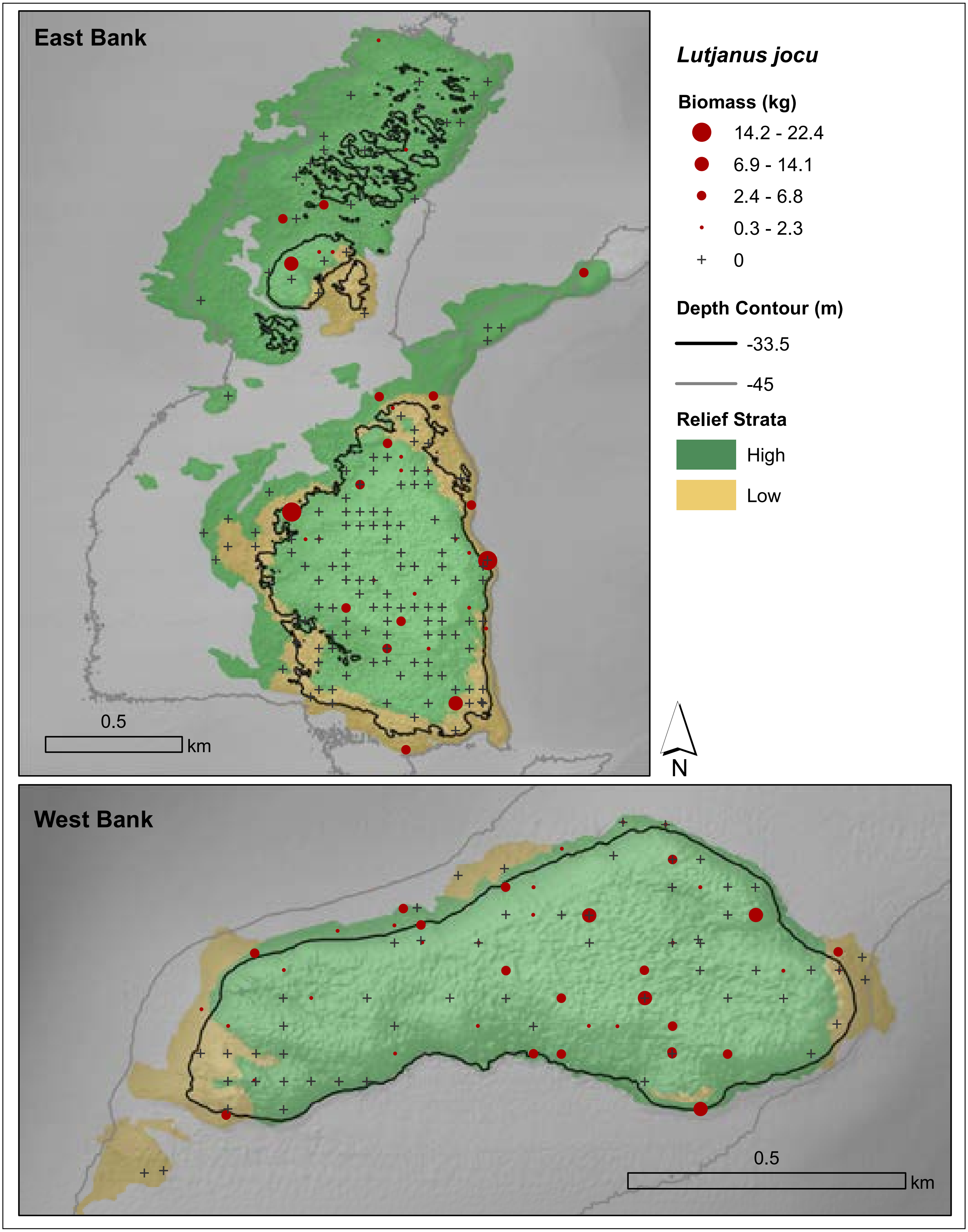

Figure 4.76. Observed biomass (kg/100 m²) of Lutjanus jocu *(dog snapper) recorded during diver surveys from 2010-2012.*

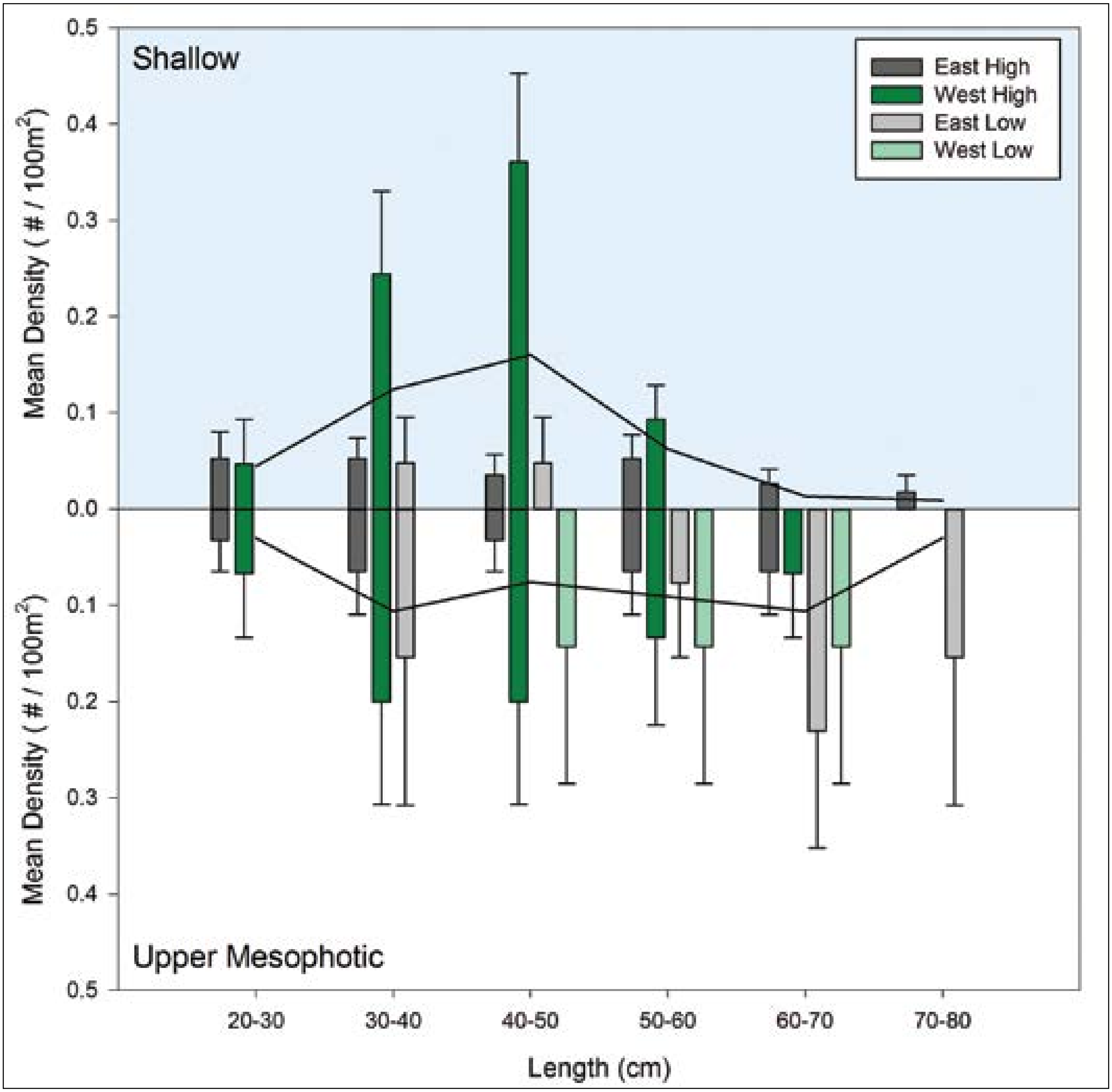

Figure 4.77. Lutjanus jocu *mean density (#/100 m^2) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* L. jocu *density per size class.*

Carangidae (Jacks)

Carangids (Figure 4.78) were recorded at 143 of 291 surveyed sites, occurring in 49.3% of sites on shallow stratum and 48.5% of sites in the UM stratum. A total of 1659 carangids (mean ± SE: 5.7 ± 1.3/100 m^2) totaling 857.5 kg (2.9 ± 0.58 kg/100 m^2) were recorded in this study. Seven carangid species were recorded during diver surveys (Figure 4.79); only four species were observed by Caldow et al. (2009). Species not reported in Caldow et al. (2009) were *Carangoides bartholomaei*, *C. hippos* and *Seriola dumerili*; these species were previously recorded by Pattengill et al. (1997).

Carangids were found on all strata (Figure 4.80), with no significant differences by bank, relief strata, or depth strata when all sites were combined. Within the UM stratum, WB densities (14.5 ± 6.4/100 m^2) were significantly greater than on EB (2.4 ± 1.2/100 m^2; Z = 2.3833, p = 0.0172), largely driven by two sites with schools of *Carangoides ruber* (n = 73 and 125 individuals/100 m^2; Figure 4.81). Where carangids were recorded, density ranged from 1-303 individuals/100 m^2, similar to a previous report (1-405 individuals/100 m^2; Caldow et al., 2009). The site with the highest number of carangids was comprised nearly entirely of *C. ruber* (n = 300/100 m^2). Although not significantly different, high relief habitats (6.4 ± 1.5/100 m^2) had higher carangid densities than low relief (1.9 ± 0.4/100 m^2).

*Figure 4.78. Examples of jacks observed in FGBNMS: greater amberjack (*Seriola dumerili*; top left), horse-eye jacks (*Caranx latus*; top right), blue runner (*Caranx crysos*; bottom left), and black jacks (*Caranx lugubris*; bottom right). Photos: (Clockwise from top left) D. Kesling (NOAA CIOERT), G. McFall (NOAA NOS/ONMS/GRNMS), E. Hickerson (NOAA NOS/ONMS/FGBNMS) and J. Voss (HBOI/FAU and NOAA CIOERT)*

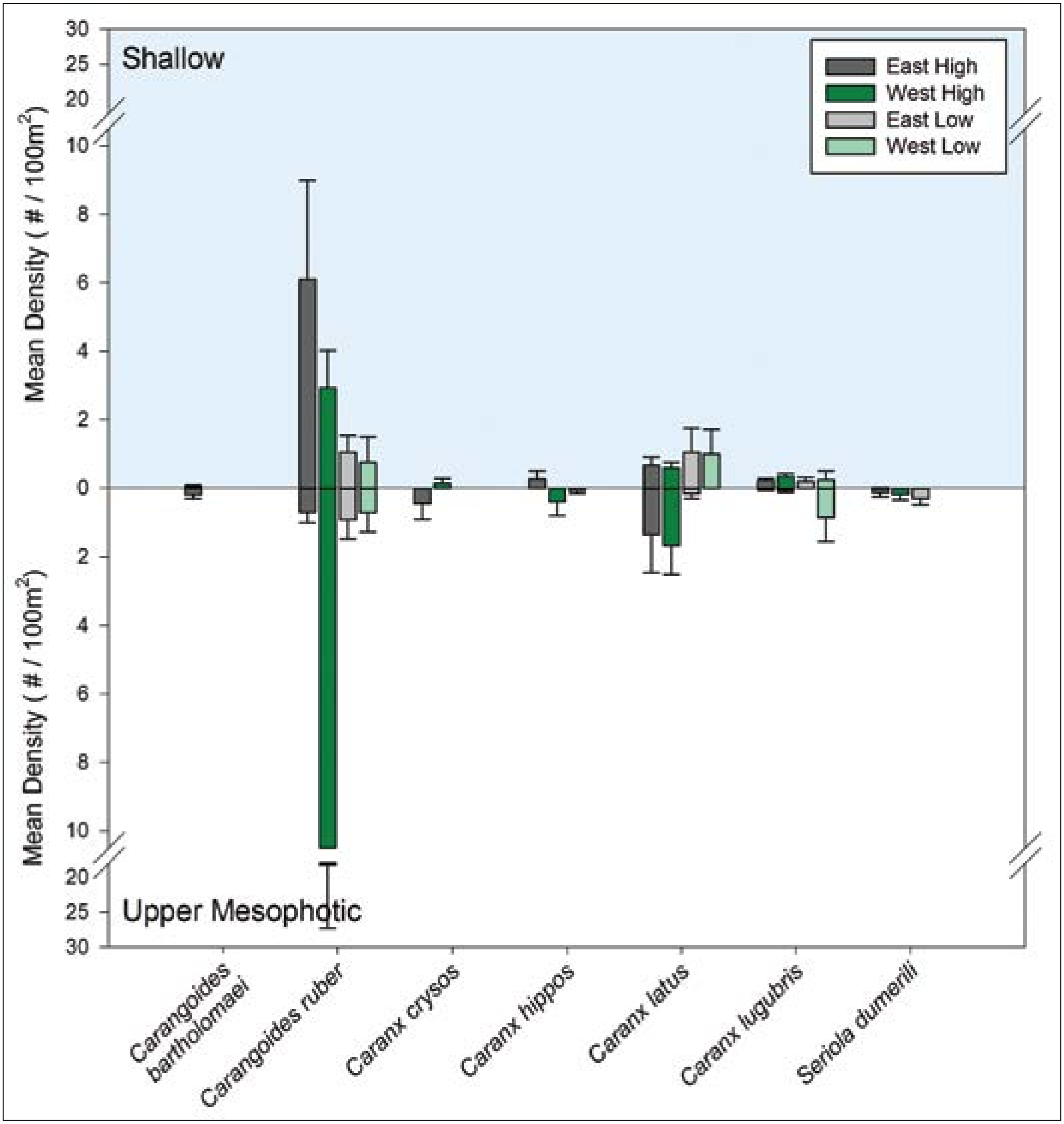

Figure 4.79. Mean density (#/100 m²) of Carangidae species by strata for dive surveys (2010-2012).

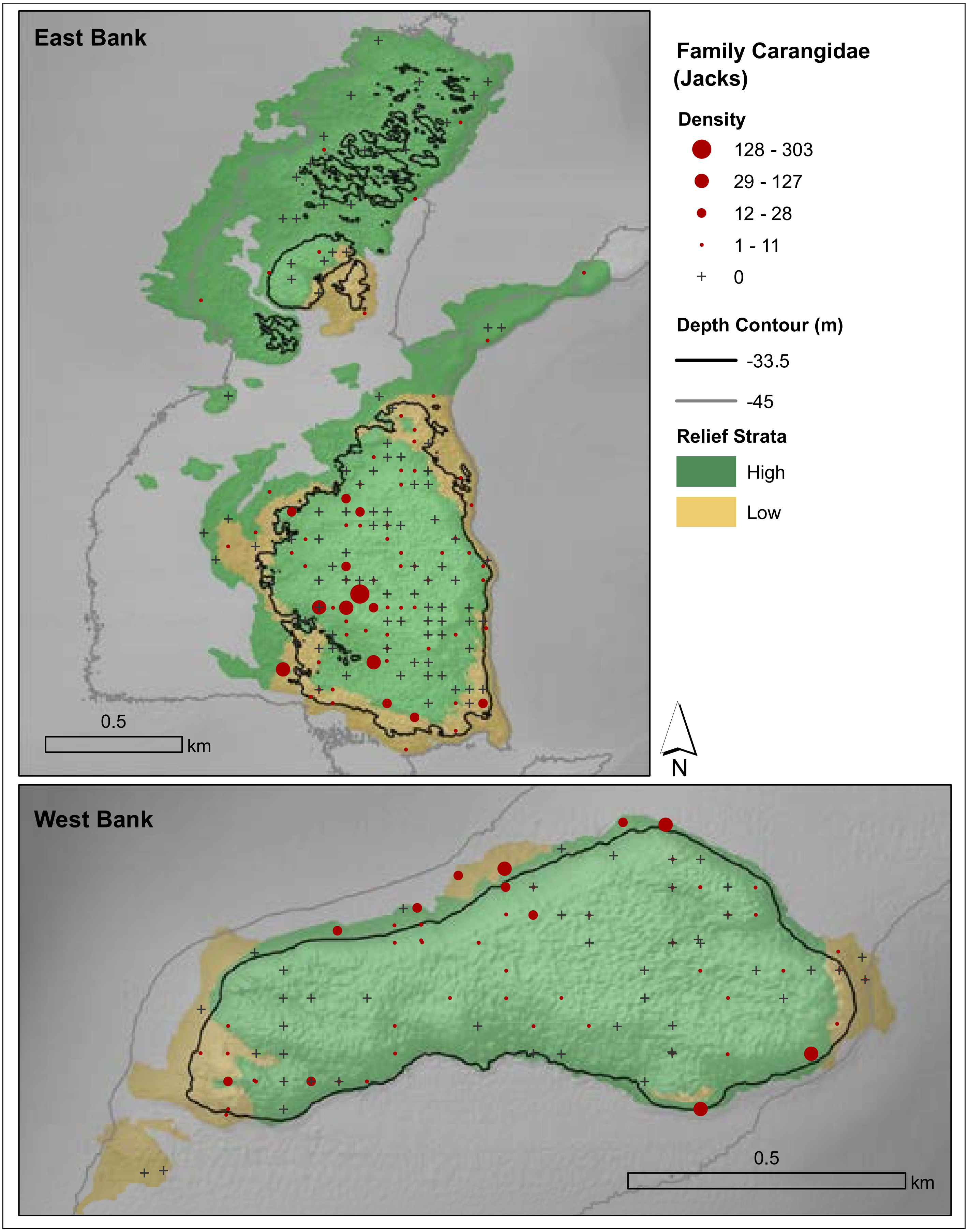

Figure 4.80. Observed density (#/100 m²) of Carangidae recorded during diver surveys from 2010-2012.

Biomass was greatest near depth strata transitions, particularly on EB, and was comprised of larger bodied carangids, mainly *C. latus* and *C. hippos* (Figure 4.82). Carangid biomass and density distribution were similar to Caldow et al. (2009), with individuals occurring throughout the reef and highest on reef edges (approximately 30-32 m). However, there was no significant correlation with density or biomass and rugosity or depth. In contrast, multivariate analyses showed elevated biomass in low relief, UM depths (consistent with large carangid schools observed on the edges of the coral reef), and species-specific depth zone effects on densities (Tables 4.5 and 4.6).

*Figure 4.81. Bar jack (*Carangoides ruber*) in FGBNMS. Photo: NOAA NOS/NCCOS/CCMA*

C. ruber was found at 98 sites and was the most numerically abundant carangid (4.4 ± 1.3/100 m^2), comprising 78% of the total abundance (Figure 4.81). Density and biomass of *C. ruber* were evenly spread across the banks, relief strata, and depth strata when all sites were combined; only density within the UM was significantly different by bank (WB: 12.6 ± 6.3/100 m^2, EB: 0.77 ± 0.26/100 m^2; Z = 2.449, p = 0.0143; Figure 4.79). Mean *C. ruber* densities by bank (WB: 4.76/100 m^2; EB: 4.2/100 m^2) for all sites and for shallow strata only (WB: 2.8/100 m^2, EB: 5.3/100 m^2) reported here were lower than those of a previous report (WB: 3.8/100 m^2; EB: 11.1/100 m^2). Inter-annual differences in *C. ruber* density were identified within the shallow strata (χ^2 = 14.8195, p = 0.0006) and UM (Z = -2.01, p = 0.0443), with higher densities in 2012 than previous years for both depth strata. Biomass was also greater in 2012 compared to previous years for the shallow strata (χ^2 = 12.0399, p = 0.0024) and UM strata (Z = -2.099, p = 0.0357). Density and biomass differences were likely due to the patchy nature of these highly mobile schooling fish. There was no correlation with density or biomass and depth, in contrast to multivariate analyses that indicated elevated density and biomass in the UM (Table 4.5). Within the shallow stratum no significant correlations were observed with density by depth or rugosity, similar to Caldow et al. (2009). For all strata combined, density was not correlated with rugosity, in contrast to multivariate analyses that showed elevated densities at high relief sites (Table 4.6); however, a significant negative relationship with biomass was identified (ρ = -0.1396, p = 0.0172). This negative relationship is somewhat surprising given the higher biomass within the high relief stratum (Figure 4.83), but it is largely driven by a few sites with high biomass and low rugosity values, although the site was classified as high relief strata, possibly due to a reef-sand interface configuration at the site.

C. latus was the most dominant carangid by biomass (2.0 ± 0.48 kg/100 m^2; Figure 4.78), comprising 68.25% of the total biomass followed by *C. hippos* (0.36 ± 0.22 kg/100 m^2; 12.4% of total carangids; Figure 4.83). *C. latus* were recorded at 54 sites; biomass and density were not significantly correlated with depth or rugosity. Although differences were not significant, density was greater within high relief UM (Figure 4.79), and biomass was greater in the low relief for both depth strata (Figure 4.83). Multivariate analyses revealed elevated *C. latus* densities in the shallow, high relief strata and elevated biomass in the low relief UM. Multivariate biomass patterns for *C. hippos* were similar (Tables 4.5 and 4.6). No between year density differences occurred within the shallow stratum; however *C. latus* were more abundant in 2011 (1.97 ± 1.03/100 m^2) than 2012 (0 individuals/100 m^2; Z = -2.994, p = 0.0028), and occurred in higher biomass in 2011 than 2012 (Z = -2.994, p = 0.003; Figure 4.9) within the UM.

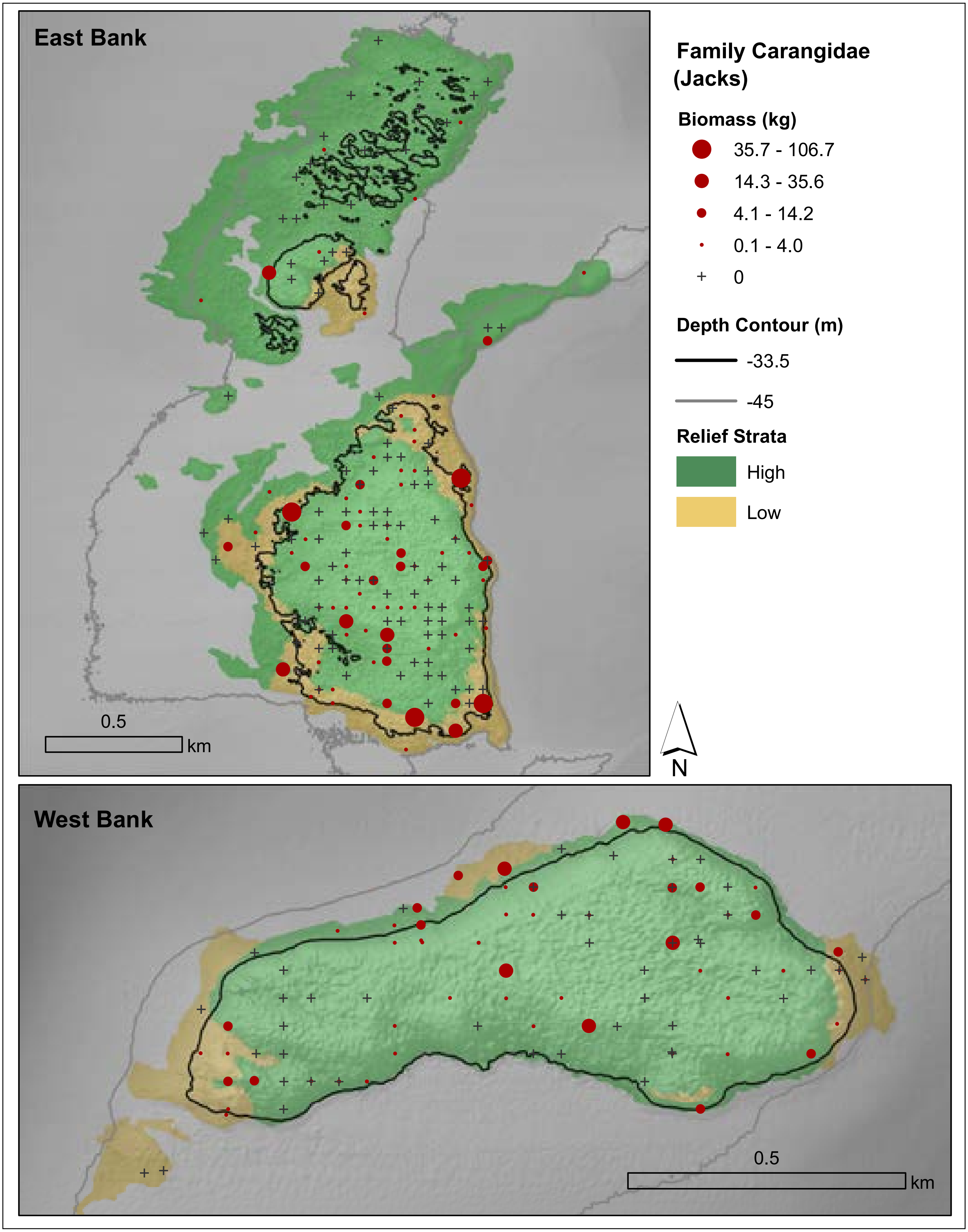

Figure 4.82. Observed biomass (kg/100 m²) of Carangidae recorded during diver surveys from 2010-2012.

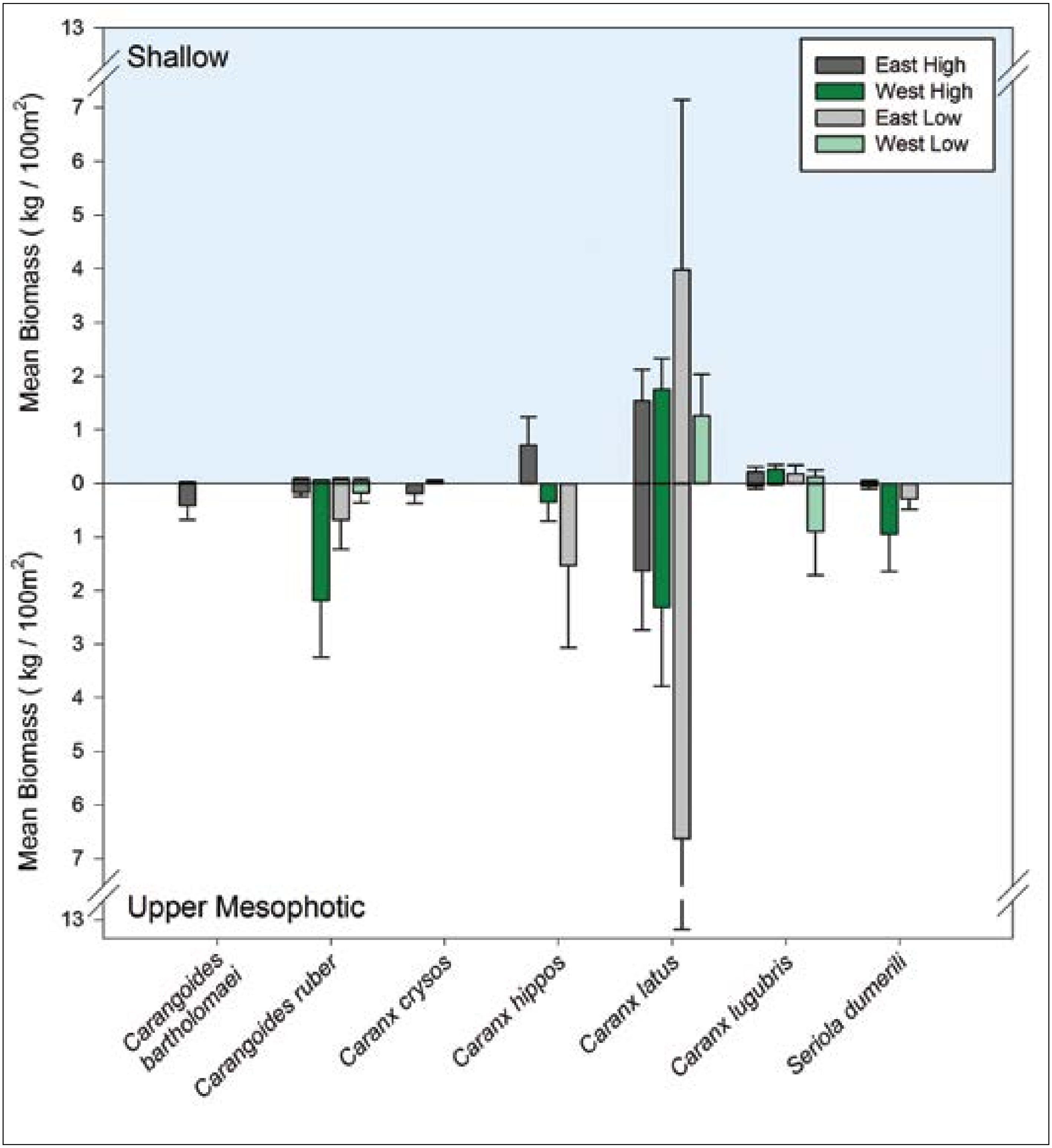

Figure 4.83. Mean biomass (kg/100 m²) of Carangidae species by strata for dive surveys (2010-2012).

Scarinae (Parrotfishes)

Scarinae was the 5th most abundant (sub)family in diver surveys and were recorded at 285 of 291 sites (Figure 4.84), with densities ranging from 1-93 individuals/100 m^2 where they occurred (Figure 4.85). Seven species were identified (Figure 4.86); one species, *Sparisoma radians* (bucktooth parrotfish), was not recorded in previous surveys (Pattengill et al., 1997; Precht et al., 2006; Caldow et al., 2009; Zimmer et al., 2010). This species was found at five sites (four shallow, one UM), all on EB, with densities ranging from 1-14 individuals/100 m^2.

*Figure 4.84. Terminal phase stoplight parrotfish (*Sparisoma viride*) at FGBNMS. Photo: A. Uhrin (NOAA NOS/NCCOS/CCFHR)*

Overall 3,687 individual Scarinae were observed within the sanctuary. They were recorded within all stratum, with slightly higher densities in the UM strata than in the shallow, although the difference by depth strata was not significant (Figure 4.85). There was no significant relationship between density and rugosity or depth (ρ =-0.1040, p = 0.0765), but the depth trend is suggestive of decreased density at increasing depth. Multivariate analyses showed species-specific patterns for depth and rugosity effects on density (Tables 4.5 and 4.6). Scarinae density was slightly higher on EB (14.0 ± 1.1/100 m^2) than WB (10.5 ± 0.76/100 m^2), and elevated in high relief habitats (Figure 4.86), but the differences were not significant.

Biomass patterns were similar to density with greater biomass in the high relief shallow strata (Figure 4.87). Biomass correlations were significant with depth (ρ = -0.2126, p = 0.0003) and rugosity (ρ = 0.1795, p = 0.0021). As for density, multivariate analyses showed species-specific patterns for depth and rugosity effects on biomass (Tables 4.5 and 4.6). Scarinae maximum site biomass recorded here (7.6 kg/100 m^2) was similar to Caldow et al. (2009; 10.0 kg/100 m^2), and this study's maximum density (93/100 m^2) was greater (46/100 m^2).

Sparisoma atomarium (greenblotch parrotfish) and *Sparisoma aurofrenatum* (redband parrotfish) were the most abundant Scarinae species (Figure 4.86); making up 21% and 26% of the total Scarinae density, respectively. *S. aurofrenatum* was among the most abundant Scarinae in Caldow et al. (2009), and among the top three in other studies (Marks, 2007; Zimmer et al., 2010; Johnston et al., 2013). *S. atomarium* was not among the top Scarinae species reported in previous surveys (Patengill-Semmens, 2006; Caldow et al., 2009; Zimmer et al., 2010; Johnston et al., 2013), likely due to the previous studies' depth limitation (maximum 32 m). In this study, *S. atomarium* density was significantly higher in the UM (Z = 3.911, p < 0.0001) and in the low relief stratum (Z = 6.089, p < 0.0001; Figure 4.86), consistent with multivariate analyses (Tables 4.5 and 4.6). Multivariate analyses indicated *S. aurofrenatum* was more abundant at UM depths on high relief habitats (Tables 4.5 and 4.6); similar to density correlations with depth (ρ = -0.0317, p = 0.5905) and rugosity (ρ = 0.0282, p = 0.6314), although these relationships were not significant. Mean biomass for *S. atomarium* (0.005 kg/100 m^2) and *S. aurofrenatum* (0.08 kg/100 m^2) were similar to a previous study (0.002 kg/100 m^2 and 0.07 kg/100 m^2, respectively; Caldow et al., 2009). The larger bodied *Sparisoma viride* (spotlight parrotfish) and *Scarus vetula* (queen parrotfish) were dominant by biomass (Figure 4.88) and are discussed in more detail below.

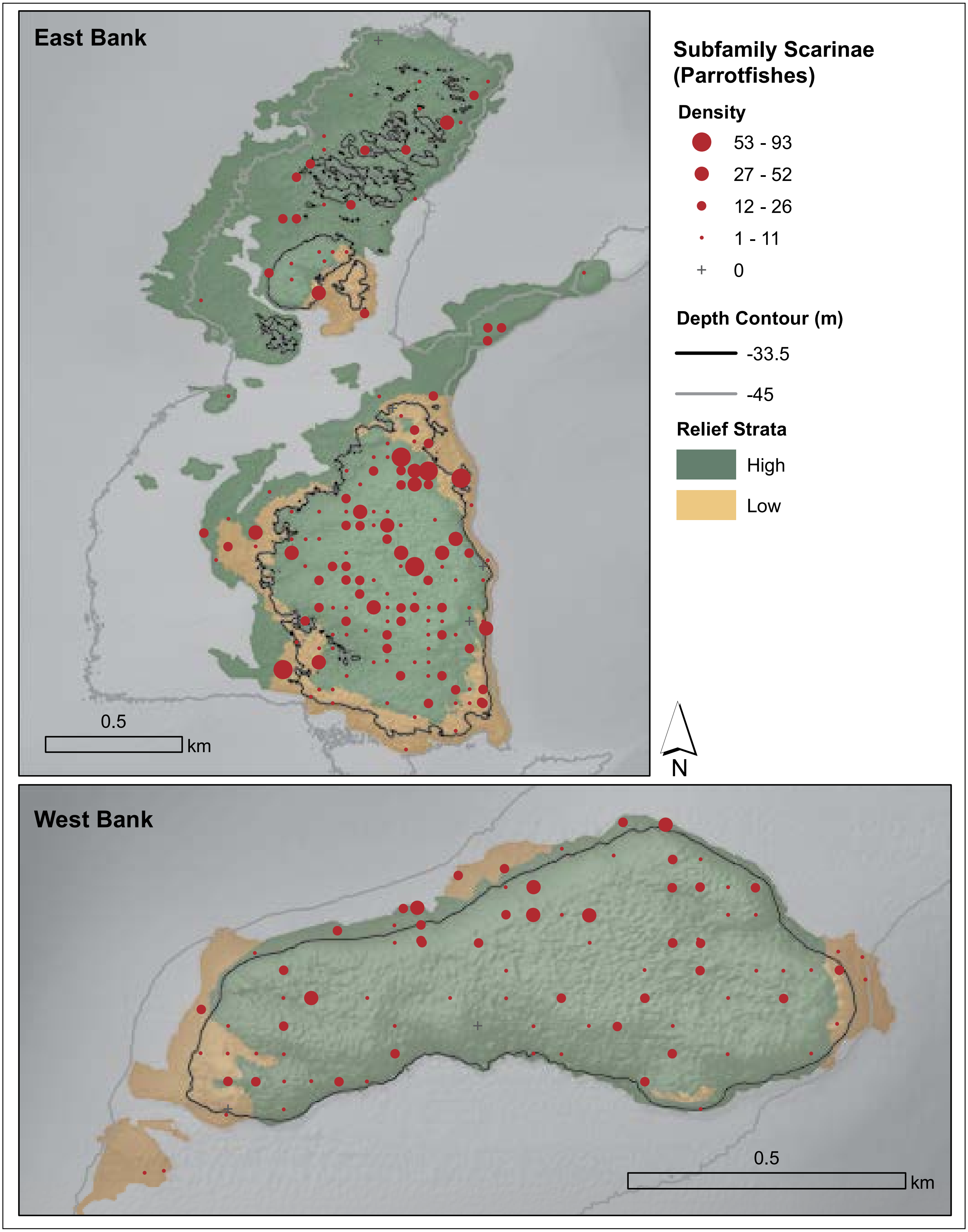

Figure 4.85. Observed density (#/100 m^2) of Scarinae recorded during diver surveys from 2010-2012.

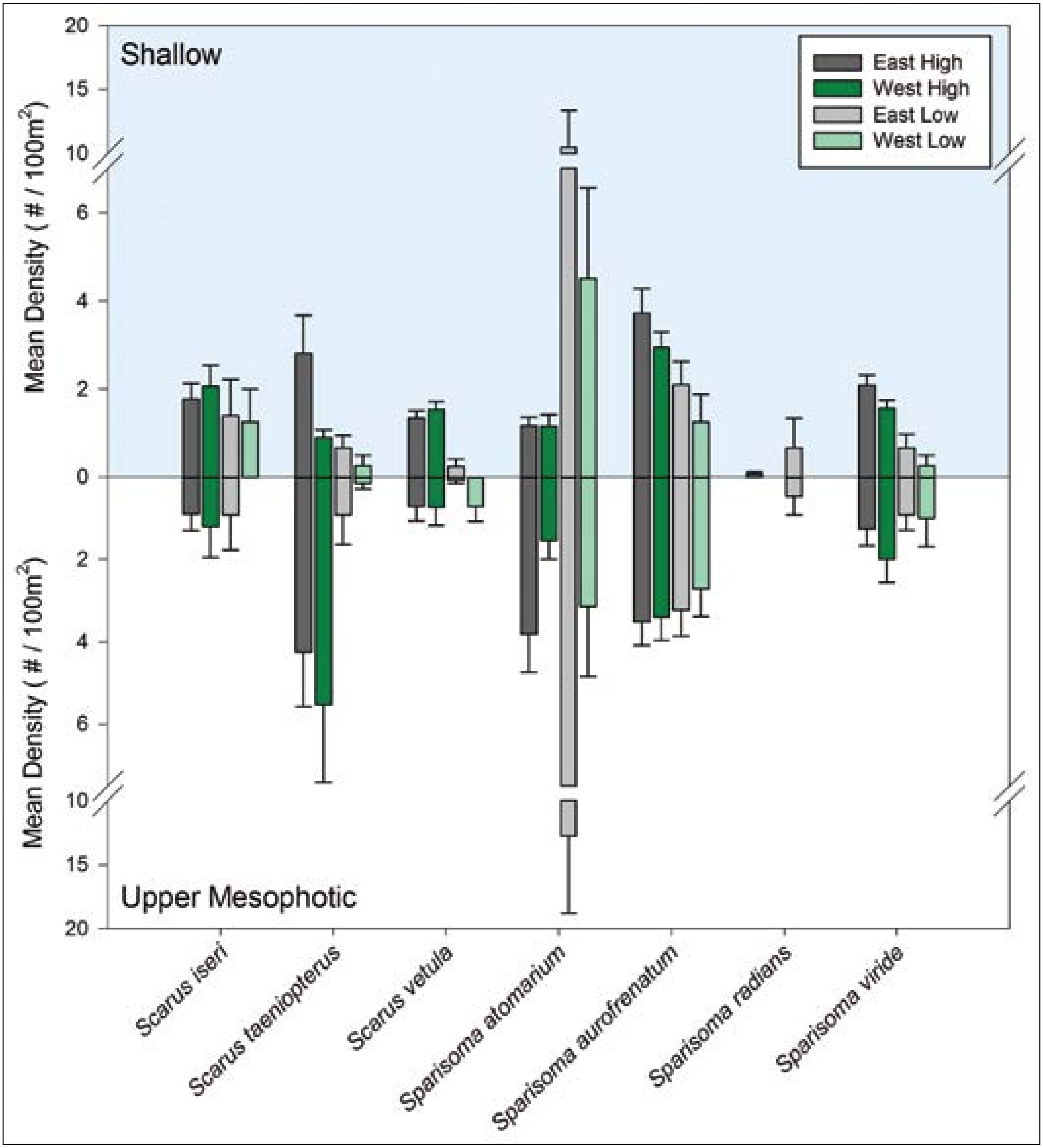

Figure 4.86. Mean density (#/100 m^2) of Scarinae species by strata for dive surveys (2010-2012).

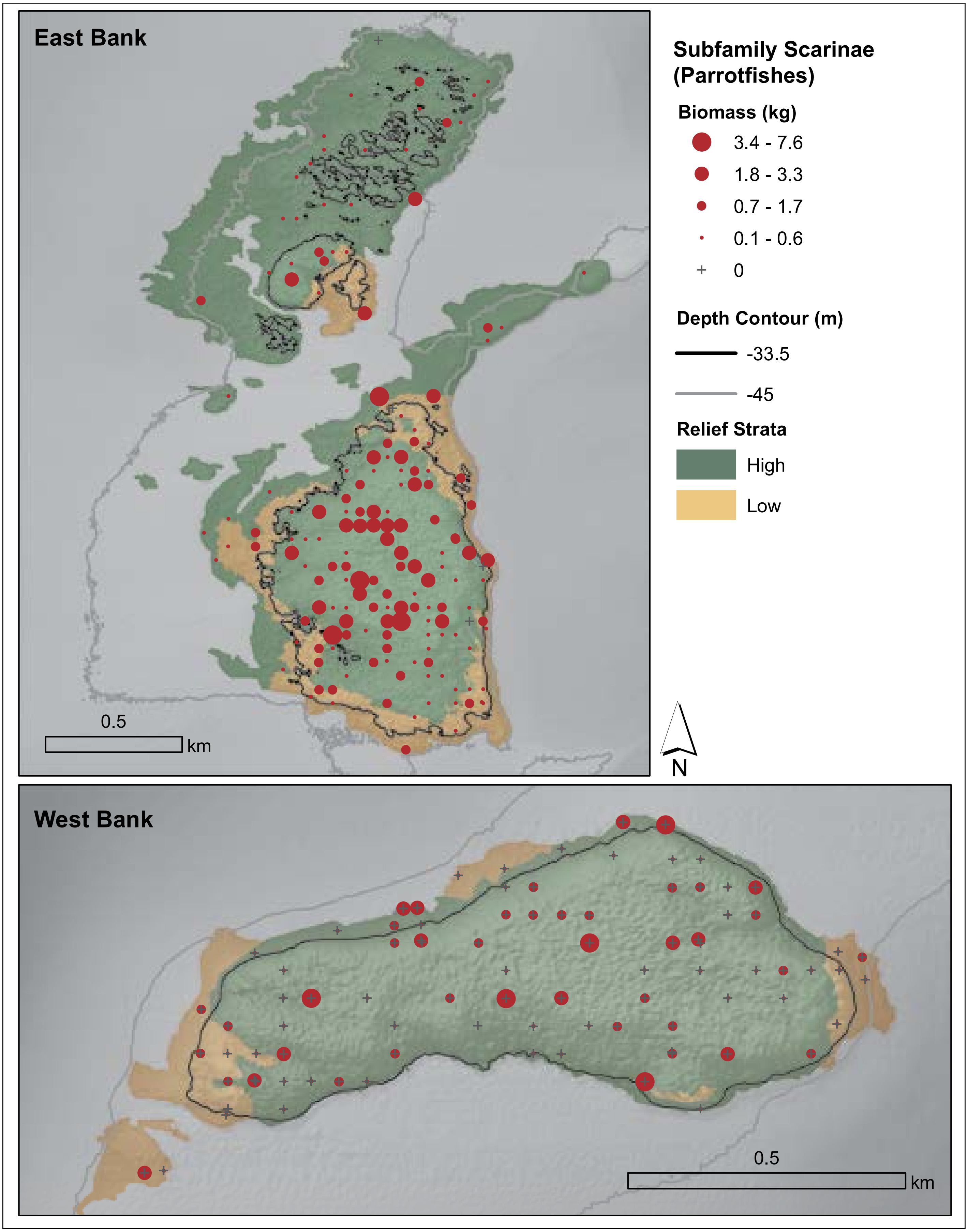

Figure 4.87. Observed biomass (kg/100 m^2) of Scarinae recorded during diver surveys from 2010-2012.

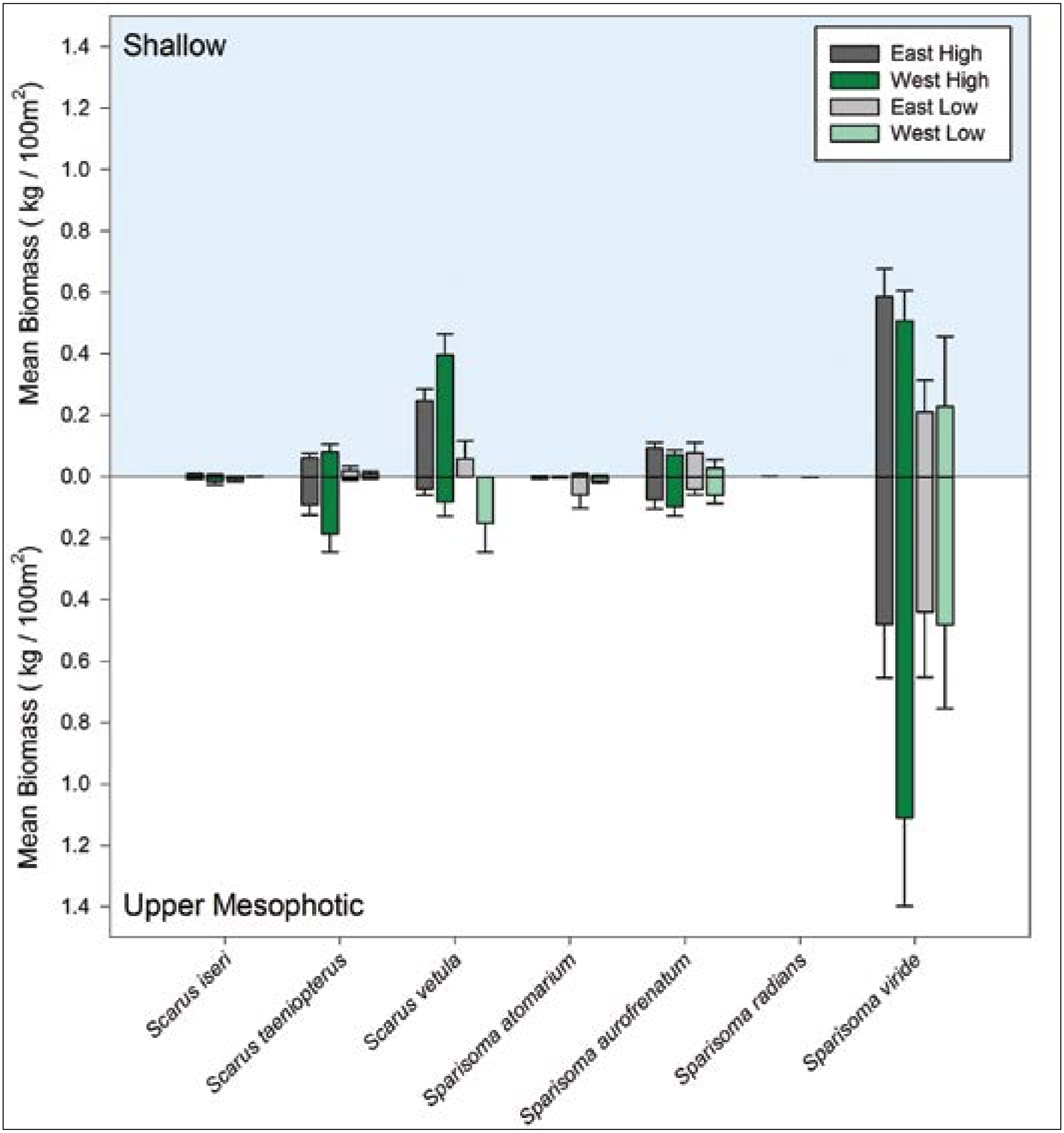

Figure 4.88. Mean biomass (kg/100 m²) of Scarinae species by strata for dive surveys (2010-2012).

Sparisoma viride (stoplight parrotfish)

Sparisoma viride was recorded at 177 of 291 surveyed sites. With 475 total individuals, *S. viride* was the 4th most abundant scarine in this study (12.9% of total; Figure 4.89). This species was also abundant in previous studies (Caldow et al., 2009; Pattengill et al., 1997; Precht et al., 2006; Johnston et al., 2013), although sighting frequencies varied among studies. *S. viride* were primarily recorded in shallow, high relief habitats (Figure 4.86); density was significantly higher within shallow stratum (1.72 ± 0.14/100 m^2) than UM (1.33 ± 0.25/100 m^2; Z = -2.19, p = 0.0281), and in the high relief stratum (1.78 ± 0.14/100 m^2) compared to low relief (0.75 ± 0.21/100 m^2; Z = -3.785, p = 0.0002; Figure 4.90). This trend was supported by a significant negative correlation with depth (ρ = -0.3428, p < 0.0001), and positive trend with rugosity (ρ = 0.265, p < 0.0001). Multivariate analyses for density also indicated greater densities in shallow depths (Table 4.5). Densities were not different between banks. Within the shallow stratum, densities were similar by bank (EB: 1.87 ± 0.2/100 m^2; WB: 1.5 ± 0.18/100 m^2), whereas Caldow et al. (2009) reported over 1.5 times higher densities on EB compared to WB. Densities were consistent between survey years within depth strata.

*Figure 4.89. Terminal phase (top) and initial phase (bottom) stoplight parrotfish (*Sparisoma viride*) at FGBNMS. Photos: G.P. Schmahl and E. Hickerson (NOAA NOS/ONMS/FGBNMS)*

Biomass patterns were similar to those reported for density. No biomass differences were recorded between banks (EB: 0.51 ± 0.07 kg/100 m^2, WB: 0.58 ± 0.09 kg/100 m^2) and depth strata (shallow: 0.51 ± 0.06 kg/100 m^2, UM: 0.61 ± 0.12 kg/100 m^2; Figure 4.88); however, there was a significant negative correlation with depth (ρ = -0.2497, p < 0.0001). The site with the highest biomass (6.4 kg/100 m^2) was within the shallow, high relief stratum (Figure 4.91). Significantly higher biomass was recorded in the high relief strata (0.57 ± 0.06 kg/100 m^2) compared to low relief (0.32 ± 0.09 kg/100 m^2; Z = -3.0073, p = 0.0026), consistent with the positive correlation with rugosity (ρ = 0.2237, p = 0.0001) and multivariate analyses (Table 4.6). These patterns are similar to those observed by Caldow et al. (2009), with larger biomass on the shallow portions of the coral reef. No significant inter-annual differences in biomass were found (Figure 4.9).

S. viride occurred in a range of sizes and in all strata (3-50 cm FL; Figure 4.92), except for the shallow, low relief stratum of WB where only individuals within the 30-35 cm size class were identified. On average, *S. viride* were larger in the UM stratum (21.7 cm FL) than shallow strata (13.9 cm). Mean length for the shallow surveys was similar to Caldow et al. (2009; 11.5 cm FL). However, length histograms between the two studies differ. Here, both survey types have a bi-modal length distribution, peaking in the smallest size (0-5 cm) and between 25-35 cm (Figure 4.92). Length distribution reported by Caldow et al. (2009) shows a single peak between 10-20 cm; no individuals within the 0-5 cm size class were reported.

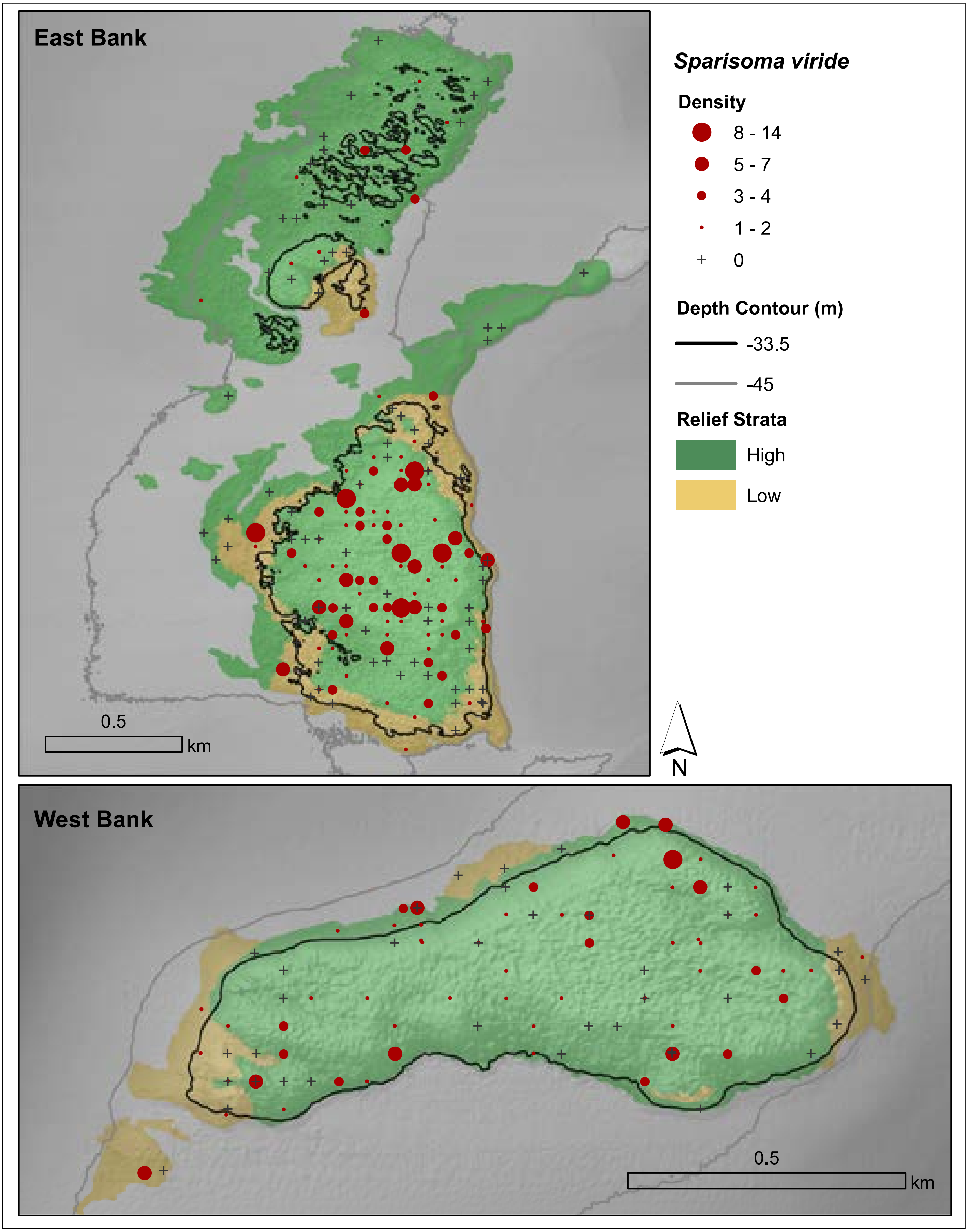

Figure 4.90. Observed density (#/100 m²) of Sparisoma viride *(stoplight parrotfish) recorded during diver surveys from 2010-2012.*

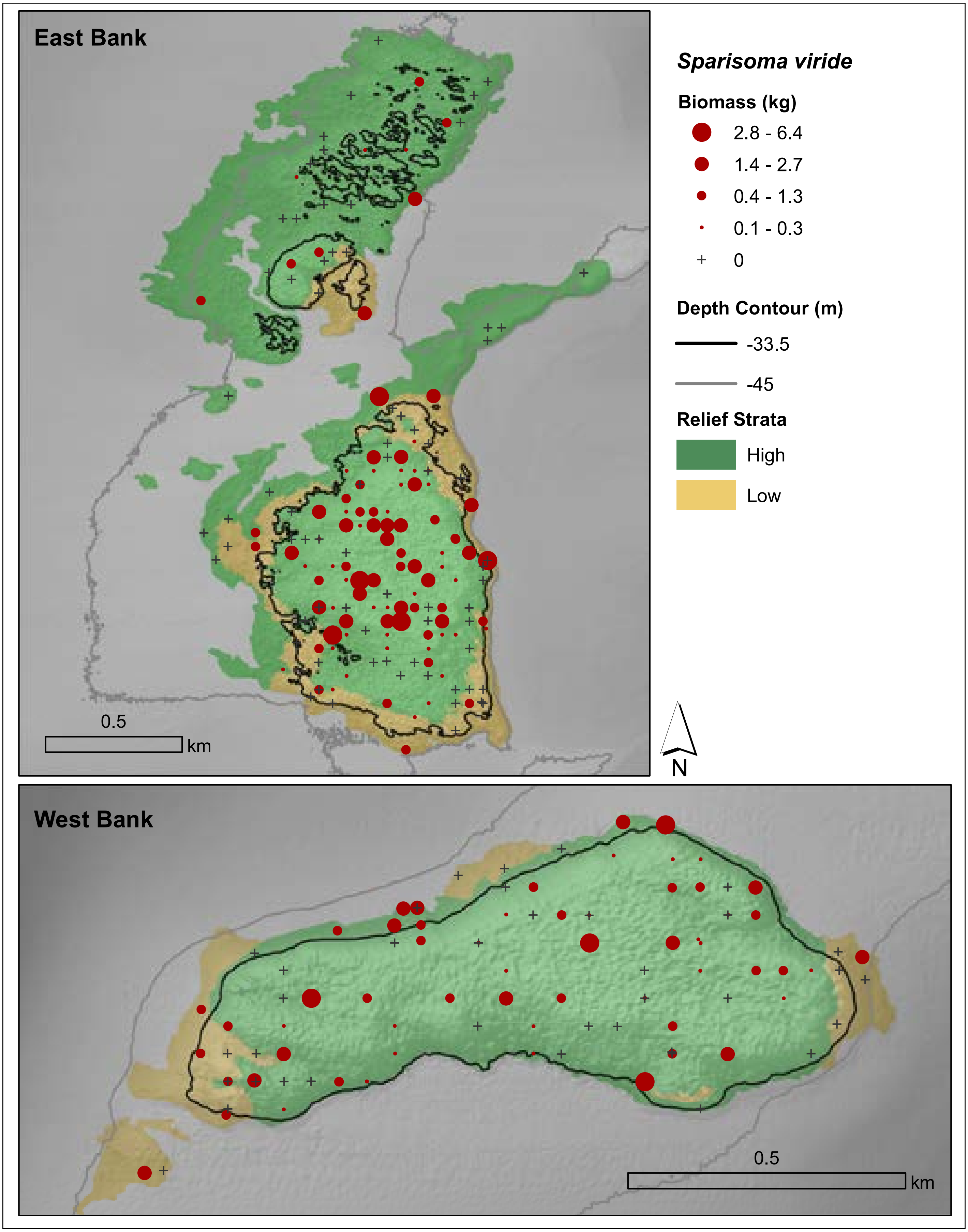

Figure 4.91. Observed biomass (kg / 100 m^2) of Sparisoma viride *(stoplight parrotfish) recorded during diver surveys from 2010-2012.*

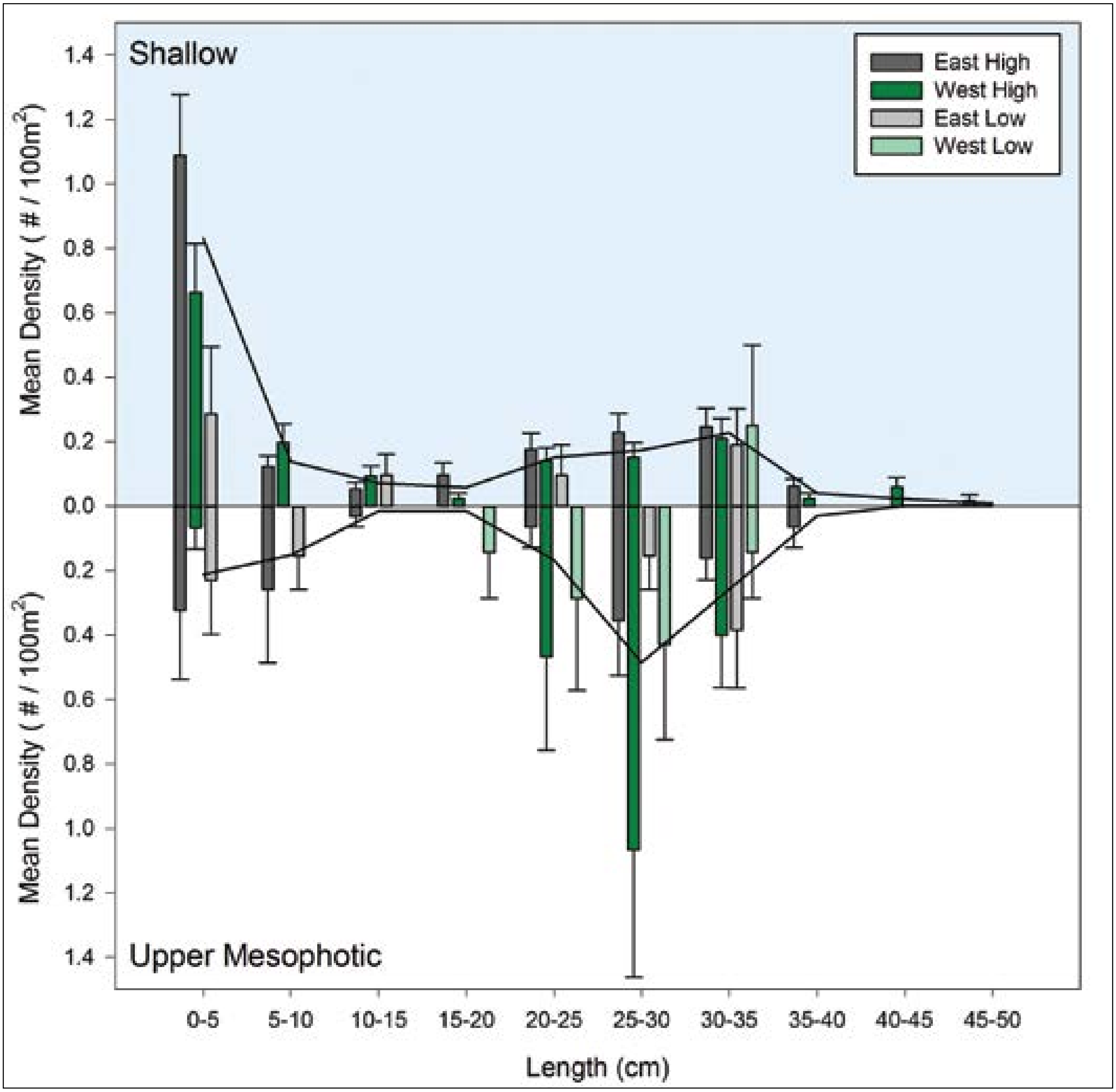

Figure 4.92. Sparisoma viride *mean density (#/100 m^2) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* S. viride *density per size class.*

<u>*Scarus vetula* (queen parrotfish)</u>

*Figure 4.93. Terminal phase of queen parrotfish (*Scarus vetula*) at FGBNMS. Photo: G.P. Schmall (NOAA NOS/ONMS/FGBNMS)*

Scarus vetula were not numerically abundant (ranked 6th among Scarinae; 8.9% of total; Figure 4.93), but had the second greatest biomass (24.8% of total), preceded by *S. viride* (57.9%; Figures 4.86 and 4.88). *S. vetula* were recorded at 133 of 291 sites and occurred more frequently on shallow strata sites (52.4%) than UM sites (22.7%); however, shallow occurrences were lower than previous studies across a similar depth range (Pattengill-Semmens, 2006; Caldow et al., 2009). Higher *S. vetula* densities were found on shallow, high relief habitats, on EB, although some high density sites occurred at sites deeper than 30 m (Figure 4.94), with significant differences by depth strata ($Z = -4.6035$, $p < 0.0001$), relief strata ($Z = -3.785$, $p = 0.0002$), and bank ($Z = 2.133$, $p = 0.0329$); consistent with significant correlative trends with depth and rugosity ($\rho = -0.4263$, $p < 0.0001$, $\rho = 0.3651$, $p < 0.0001$, respectively). Multivariate analyses for density were in agreement with univariate patterns (Table 4.5 and 4.6). Mean density within the shallow high relief stratum (1.79 ± 0.14/100 m^2), and where all sites were combined (1.12 ± 0.09/100 m^2), was less than previous studies (Precht et al., 2006; Caldow et al., 2009; Johnston et al., 2013). *S. vetula* density was similar for all surveyed years by depth strata.

Shallow stratum biomass (0.28 ± 0.03 kg/100 m^2) was significantly higher than UM (0.05 ± 0.01 kg/100 m^2; $Z = -4.6035$, $p < 0.0001$), with a shallow stratum site maximum (3.8 kg/100 m^2) over six times higher than the UM site maximum (0.6 kg/100 m^2; Figure 4.95). The shallow site maximum reported here is similar to Caldow et al. (3.4 kg/100 m^2; 2009). Distribution patterns were similar to density, with higher biomass in shallow depths ($\rho = -0.4139$, $p < 0.0001$) and in high relief habitats ($\rho = 0.3586$, $p < 0.0001$; Figure 4.86). Multivariate analyses for biomass were consistent with the univariate patterns (Tables 4.5 and 4.6). Where all sites were combined, biomass was significantly higher on WB (0.32 ± 0.05 kg/100 m^2) compared to EB (0.17 ± 0.03 kg/100 m^2; $Z = 2.6004$, $p = 0.0093$), and in high relief habitats compared to low ($Z = -4.6212$, $p < 0.0001$). Differences by bank and relief strata were largely driven by shallow stratum sites, where the majority of biomass occurred; there was no difference between strata within the UM stratum. No significant inter-annual differences in biomass were found.

S. vetula mean length was greater within shallow surveys (19.3 cm) than in UM surveys (15.4 cm). Caldow et al. (2009) recorded a similar mean length of 20.6 cm FL within the shallow stratum. Individuals were recorded across a range of sizes in both depth strata, with individuals spanning a greater size range in the shallow surveys (Figure 4.96). Peak density for both depth strata was between 10-15 cm, and maximum individual length was 42.5 cm within the shallow stratum, and 32.5 cm within UM. Length distributions were similar to Caldow et al. (2009), which identified peak density between 10 and 20 cm, and maximum individual length occurred between 40-50 cm.

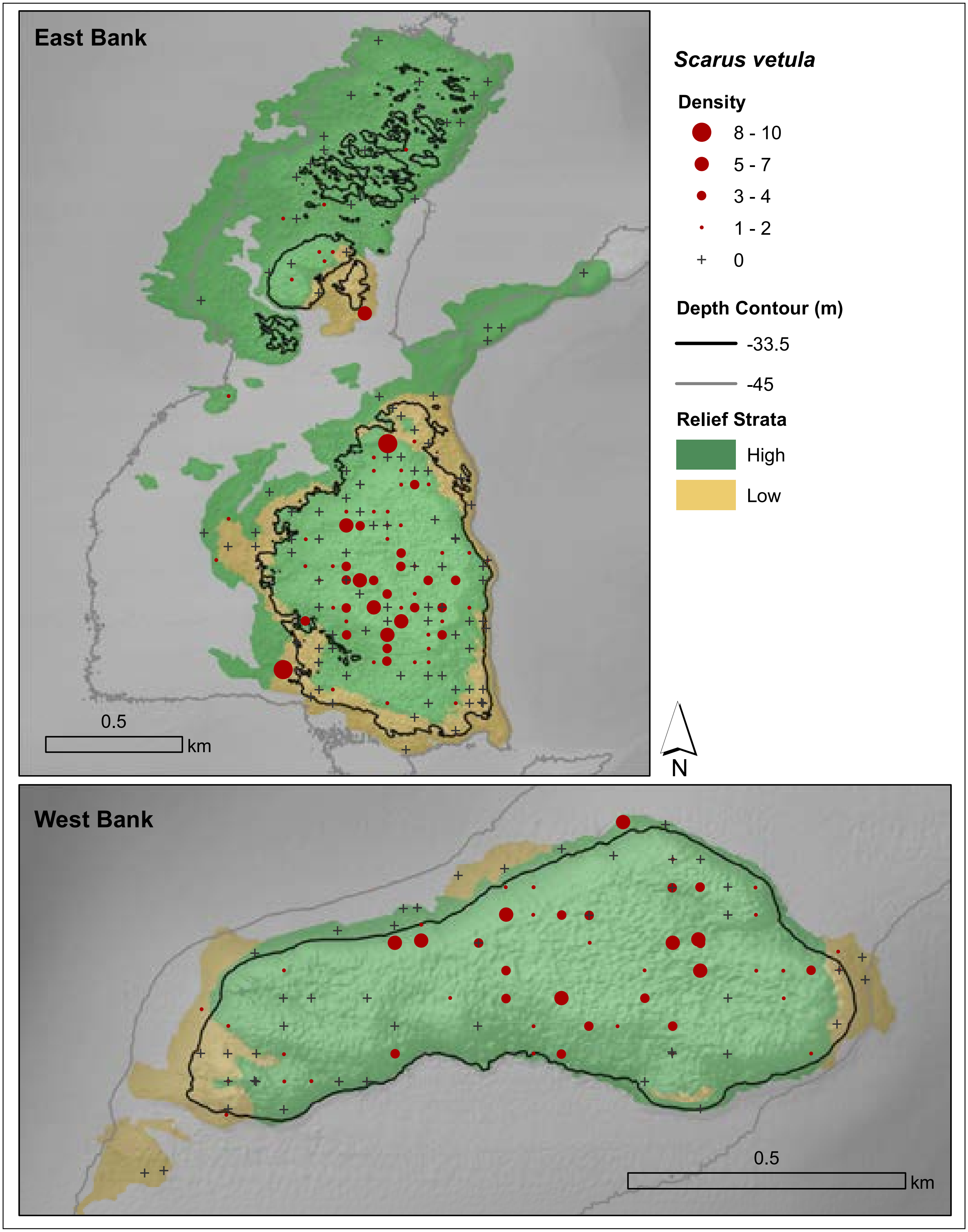

Figure 4.94. Observed density (#/100 m²) of Scarus vetula *(queen parrotfish) recorded during diver surveys from 2010-2012.*

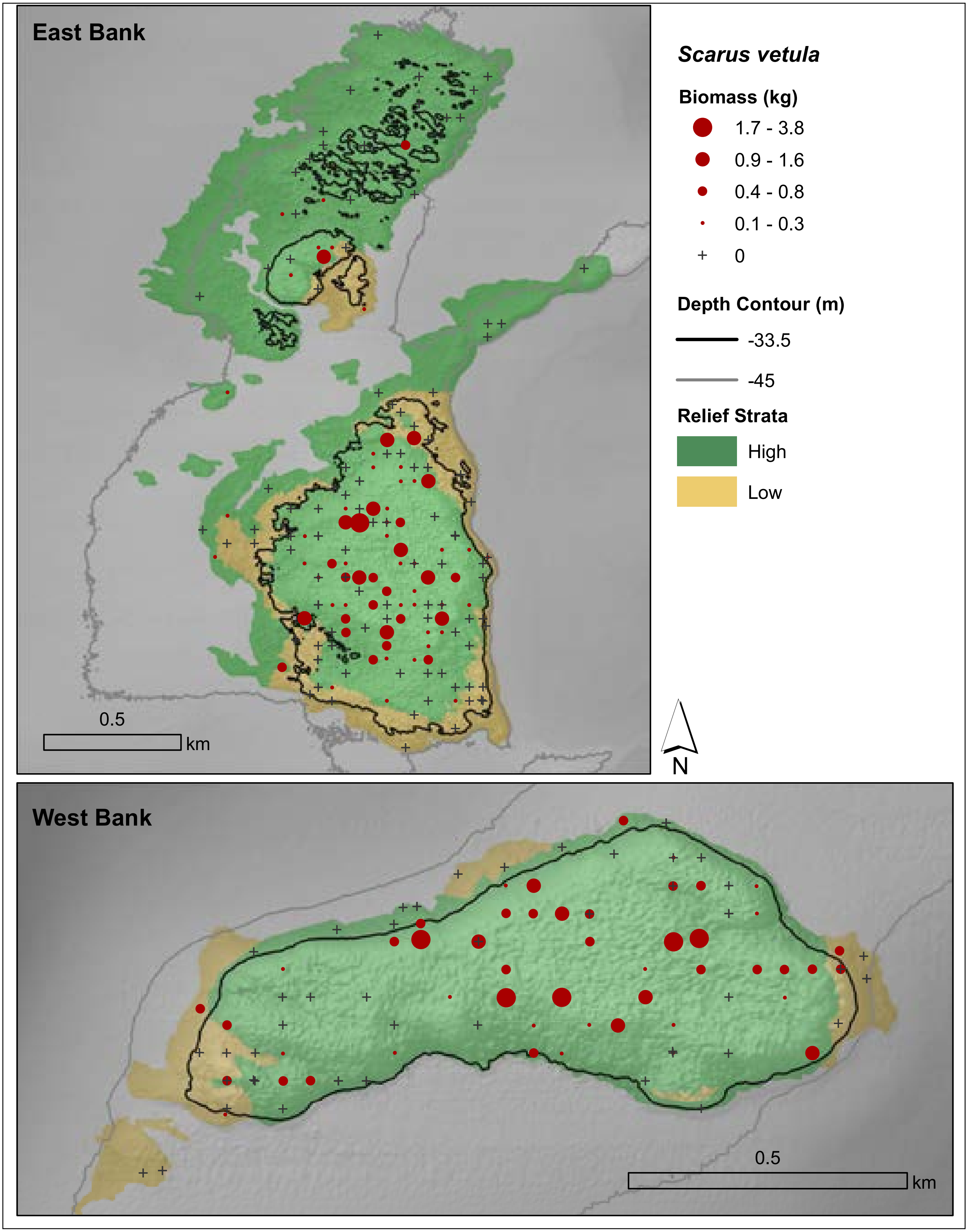

Figure 4.95. Observed biomass (kg/100 m²) of Scarus vetula *(queen parrotfish) recorded during diver surveys from 2010-2012.*

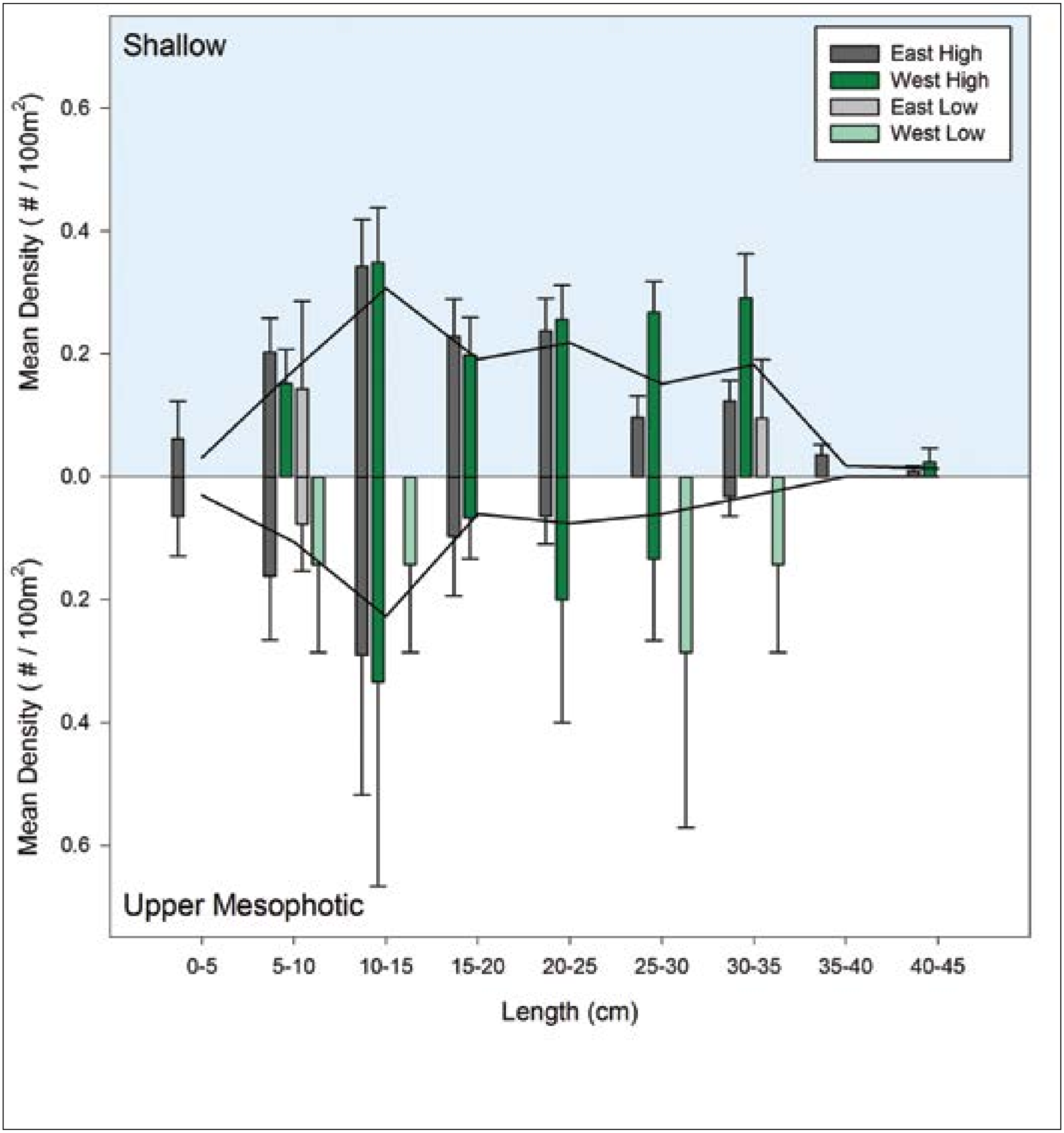

Figure 4.96. Scarus vetula *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* S. vetula *density per size class.*

Acanthuridae (Surgeonfishes)

*Figure 4.97. Species of the Acanthuridae Family, the blue tang (*Acanthurus coeruleus*) at FGBNMS. Photo: G.P. Schmall (NOAA NOS/ONMS/FGBNMS)*

Acanthurids were recorded at a majority of sites (249 of 291), however, they ranked 9th in overall family abundance (0.94% of total fish number) and 13th in total biomass (1.2% of total; Figure 4.97). They were recorded in all strata and were distributed across a range of depths (19-44 m) on both banks, with higher densities on the shallow portions of the coral reef in high relief habitats (Figure 4.98). Density was negatively correlated with depth (ρ = -0.1572, p = 0.0072), and positively correlated with rugosity (ρ = 0.1189, p = 0.0426), largely driven by patterns within the shallow stratum as correlations were not significant within the UM stratum. Acanthurid density was similar by bank (EB: 4.19/100 m^2; WB: 3.75/100 m^2) and by depth strata (shallow: 3.7 ± 0.21/100 m^2; UM: 4.9 ± 0.53/100 m^2). Many high biomass sites were at depths >30 m (Figure 4.99); UM biomass (0.81 ± 0.14 kg/100 m^2) was significantly greater than shallow (0.31 ± 0.02 kg/100 m^2; Z = 2.4285, p = 0.0152), with no differences by bank or relief strata, though species-specific patterns for density and biomass were apparent in the multivariate analyses (Tables 4.5 and 4.6).

Three acanthurid species, *Acanthurus bahianus* (ocean surgeon), *Acanthurus chirurgus* (doctorfish) and *Acanthurus coeruleus* (blue tang), were recorded in this study (Figure 4.100) and previously (Hickerson et al., 2008; Caldow et al., 2009; Zimmer et al., 2010; Johnston et al., 2013). The most ubiquitous of the acanthurid species, *A. coeruleus*, was the most abundant species, comprising 75% of all acanthurid abundance. It was recorded at 221 of 291 sites, with a maximum site density of 21/100 m^2. Higher *A. coeruleus* densities were found in shallow sites (ρ = -0.1966, p = 0.0007), with higher rugosity (ρ = 0.1552, p = 0.008), consistent with multivariate analyses (Tables 4.5 and 4.6). *A. bahianus* and *A. chirurgus* densities were similar (0.47/100 m2; 0.51/100 m^2; respectively), however, their distributions were different. *A. bahianus* was recorded more frequently in the UM stratum (Z = 3.8769, p = 0.0001), while *A. chirurgus* was predominantly found within the shallow stratum (Z = -3.267, p = 0.0011), suggesting a stratification by depth (Figure 4.100). For each Acanthuridae species, mean biomass was higher in the UM stratum, but only *A. bahianus* and *A. chirurgus* differed significantly between depth strata (Z = 3.73, p = 0.0002; Z = -3.03, p = 0.0024, respectively; Figure 4.101). Multivariate analyses for *A. bahianus* and *A. chirurgus* density and biomass were consistent (Table 4.5). Only *A. chirurgus* biomass (Z = -2.306, p = 0.0211) and density (Z = -2.244, p = 0.0248) were different by bank, with higher values on EB (0.04 ± 0.01 kg/100 m^2; 0.65 ± 0.12/100 m^2, respectively) compared to WB (0.01 ± 0.004 kg/100 m^2; 0.28 ± 0.07/100 m^2, respectively). Densities of these three species were consistent across surveyed years within each depth strata, except for *A. coeruleus,* where density was higher in 2011 compared to 2010 within the shallow stratum (χ^2 = 7.929, p = 0.019).

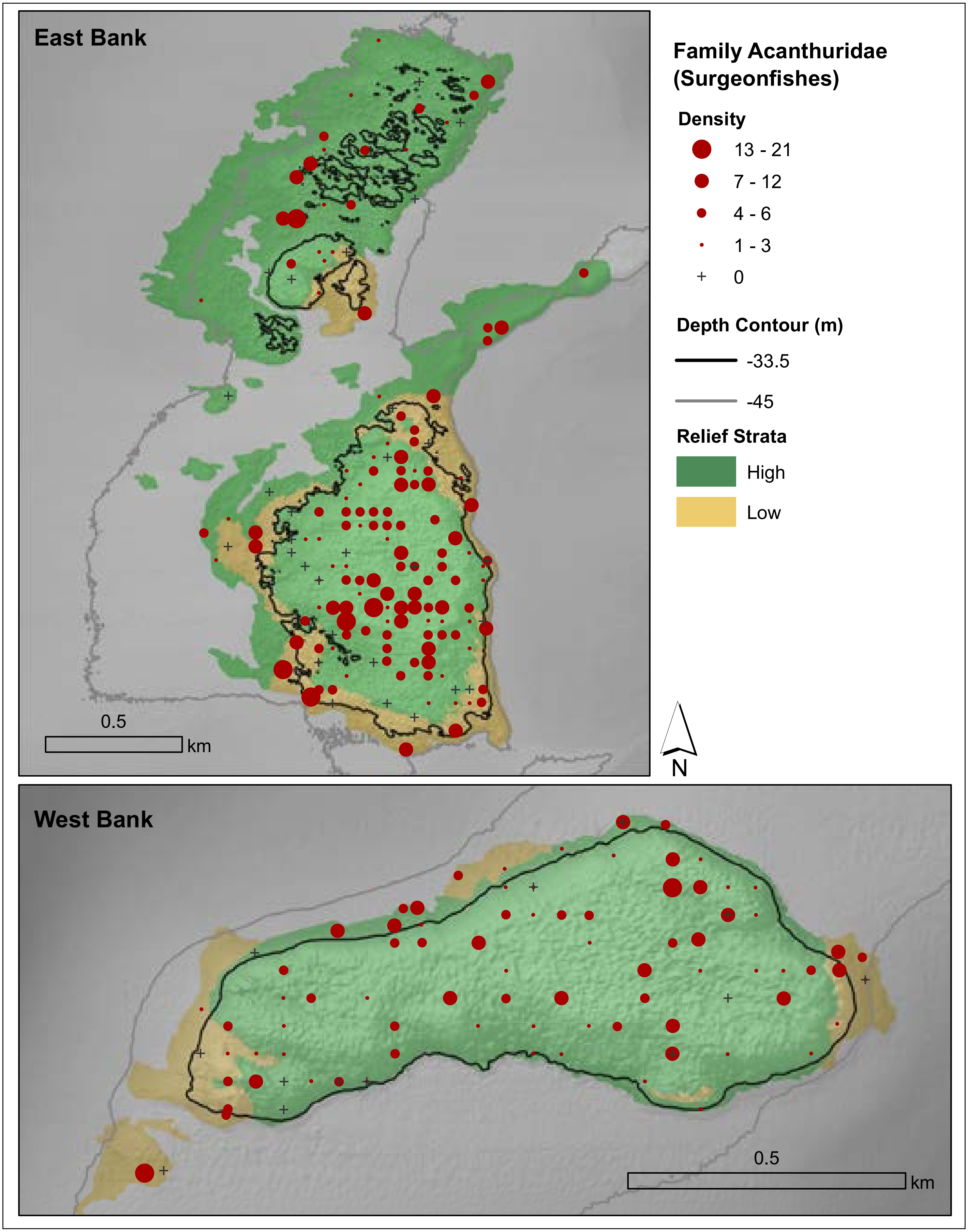

Figure 4.98. Observed density (#/100 m²) of Acanthuridae recorded during diver surveys from 2010-2012.

Fish Communities

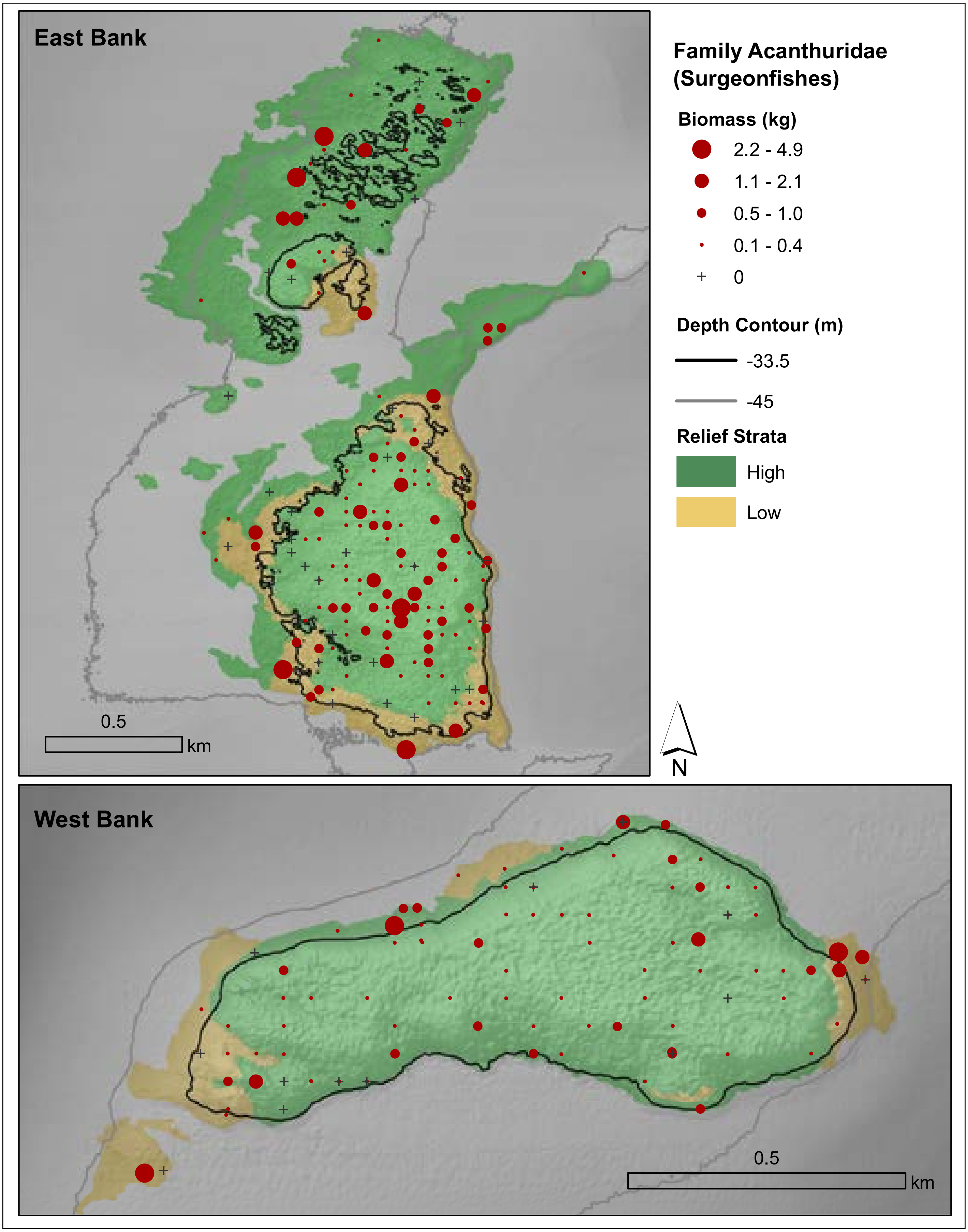

Figure 4.99. Observed biomass (kg/100 m²) of Acanthuridae recorded during diver surveys from 2010-2012.

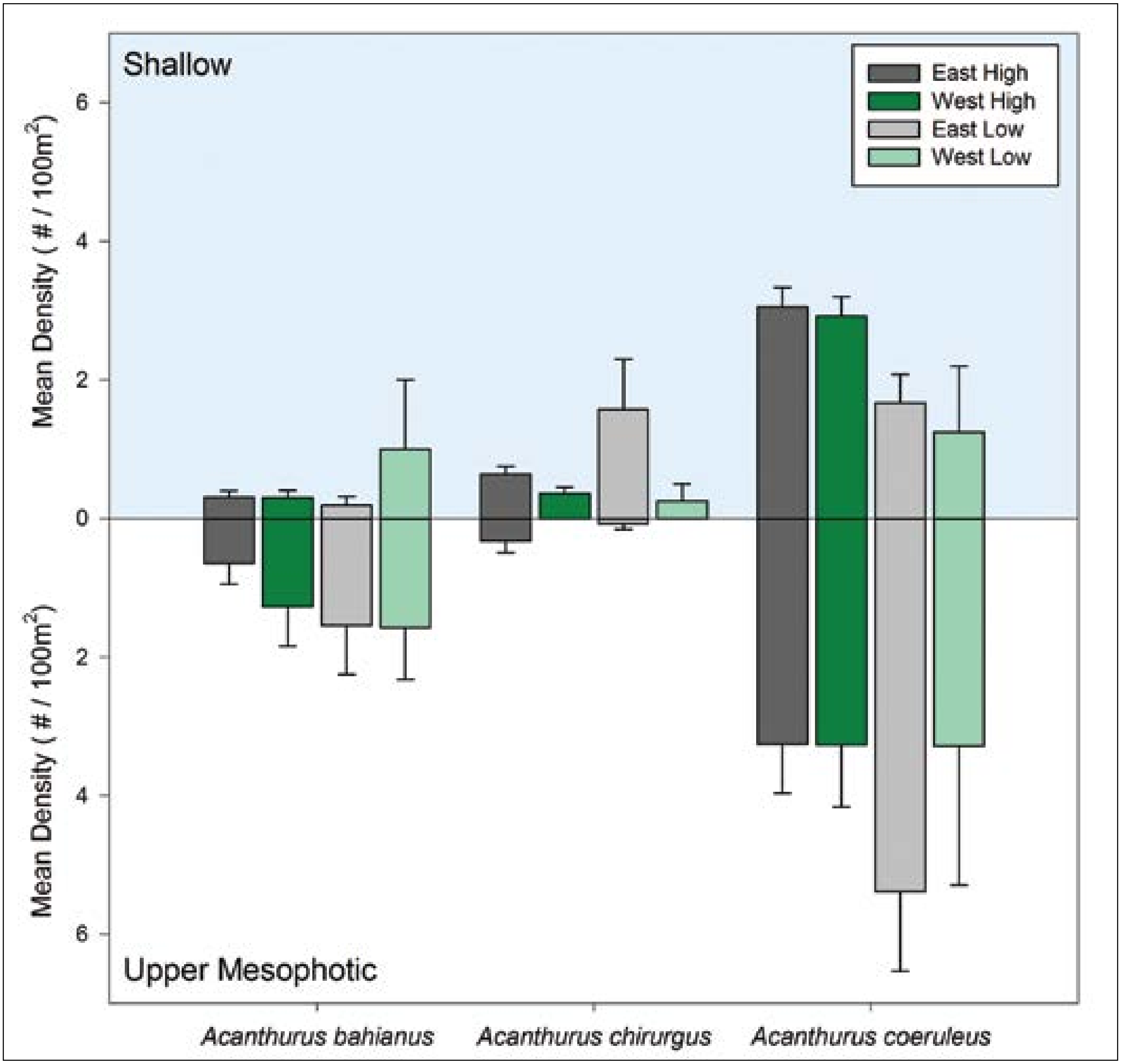

Figure 4.100. Mean density (#/100 m²) of Acanthuridae species by strata for dive surveys (2010-2012).

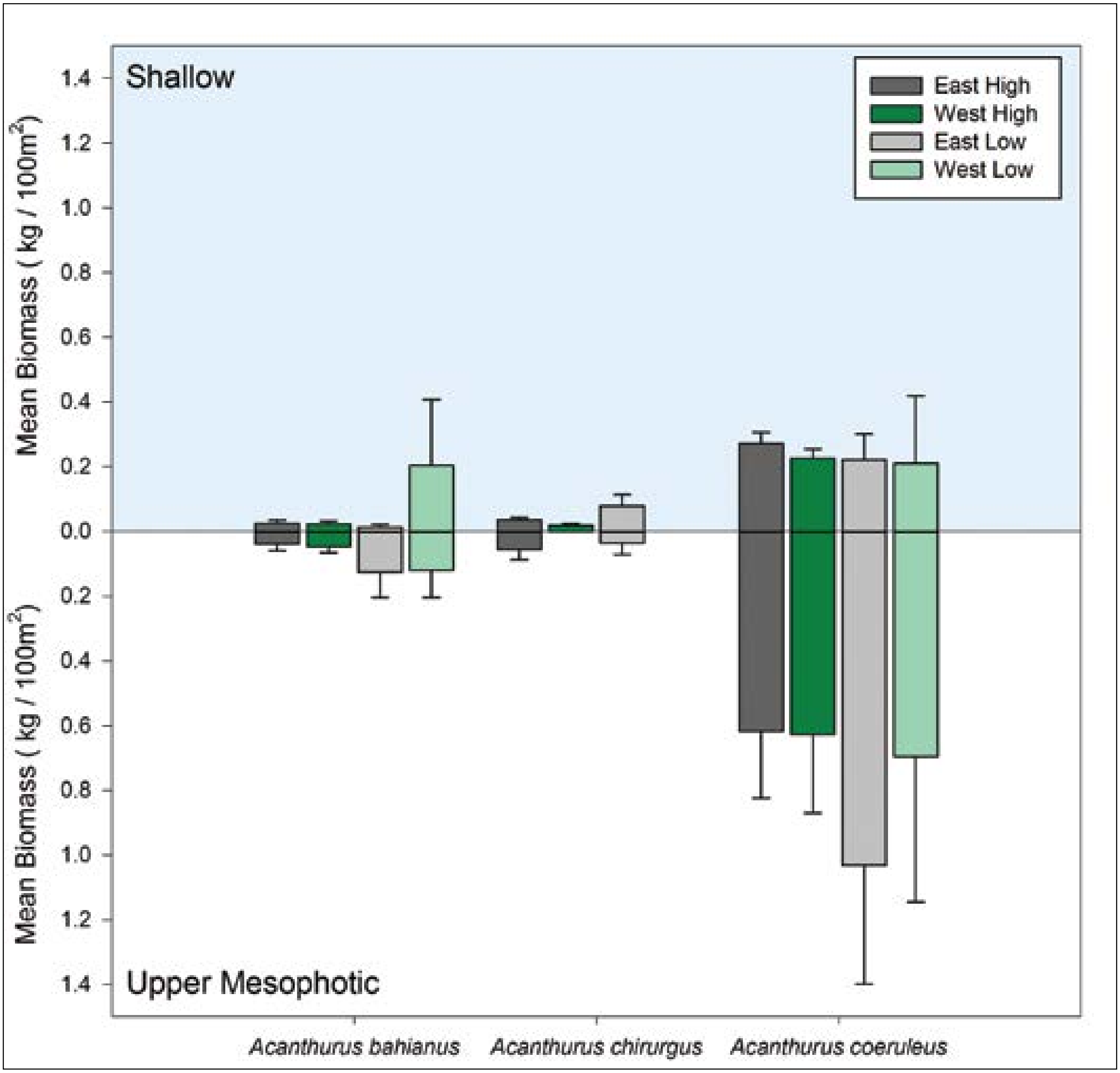

Figure 4.101. Mean biomass (kg/100 m^2) of Acanthuridae species by strata for dive surveys (2010-2012).

Pomacentridae (Damselfishes)

Pomacentrids were found at all surveyed sites, with site densities ranging from 5-1,268 /100 m^2. This was a species rich family (Figure 4.102), with 13 species recorded (Figures 4.103 and 4.104); all of which were documented in previous surveys (Caldow et al., 2009; Pattengill et al., 1997; Precht et al., 2006; Johnston et al., 2013). Pomacentridae was the most abundant family by density, comprising 31.4% of total fish density and ranked 12th among families by biomass (1.58% of total biomass). Density was uniformly distributed across all strata (Figure 4.105), with no significant differences by bank, relief or depth strata, or correlations with depth or rugosity, similar to Caldow et al. (2009). Highest site density (1,268/100 m^2) was dominated by two species: *C. insolata* (n = 639) and *Stegastes adustus* (dusky damselfish; n = 346); this site was within the EB UM high relief strata.

Generally, pomacentrid biomass decreased with depth ($\rho = -0.2569$, $p < 0.0001$), although some sites with high to moderate biomass were recorded at depths >30 m (Figures 4.106 and 4.107). Biomass also increased in lower rugosity areas ($\rho = -0.705$, $p < 0.0001$), consistent with differences by relief strata (Low: 0.91 ± 0.47 kg/100 m^2, High: 0.48 ± 0.04 kg/100 m^2; Z = -2.188, p = 0.0287). Species-specific patterns for density and biomass were apparent in the multivariate analyses (Tables 4.5 and 4.6). Biomass was similar between banks (EB: 0.62 ± 0.12 kg/100 m^2, WB: .0.43 ± 0.04 kg/100 m^2).

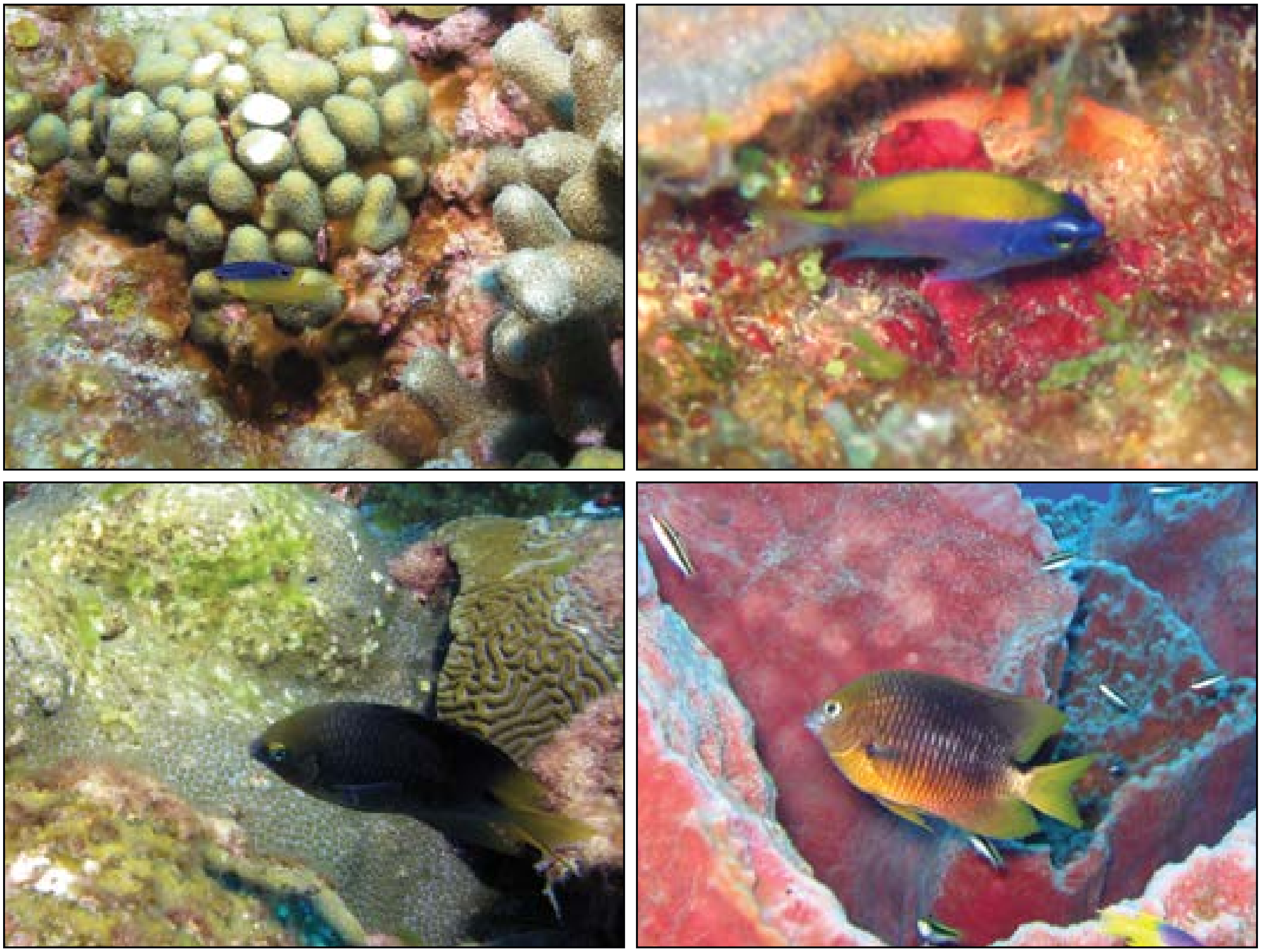

*Figure 4.102. Various examples of species in the Family Pomacentridae: cocoa damselfish (*Stegastes variabilis*; top left), sunshinefish (*Chromis insolata*; top right) and threespot damselfish (*Stegastes planifrons*; bottom row) at FGBNMS. Photos: G.P. Schmahl (NOAA NOS/ONMS/FGBNMS; top left and bottom left), NOAA NOS/NCCOS/CCMA (top right) and E. Hickerson (NOAA NOS/ONMS/FGBNMS)*

C. multilineata was the most abundant pomacentrid (48.2% of total) and displayed the highest biomass (Figure 4.104). They were primarily found on shallow high relief habitats. Densities were different by relief strata (Low: 58.4 ±17.5; High: 65.7 ± 5.1/100 m^2; Z = -3.42, p = 0.0006), consistent with the positive correlation with rugosity (ρ = 0.2885, p < 0.0001). Shallow stratum density (72.1 ± 5.8/100 m^2) was greater than UM (38.7 ± 9.8/100 m^2; Z = -5.801, p < 0.0001), and negatively correlated to depth (ρ = -0.4047, p < 0.0001). Multivariate analyses were consistent, indicating elevated densities in shallow high relief habitats (Tables 4.5 and 4.6). There were no density differences by bank. No inter-annual differences were detected within the shallow stratum, however, within the UM, 2012 densities were significantly less than 2011 (Figure 4.8); biomass was also greater in 2011 than 2012 within the UM (Z = -2.209, p = 0.027).

Chromis insolata was the second most abundant pomacentrid (24.15% of total) and second by biomass (11.79% of total; Figure 4.104). Unlike *C. multilineata*, *C. insolata* densities, *Chromis insolata* densities were greater in low relief habitats (73.0 ± 17.1/100 m^2) than in high relief habitats (24.9 ± 4.9/100 m^2; Z = 6.34, p < 0.0001), supported by a significant correlation with rugosity (ρ = -0.5154, p < 0.0001), and also greater in deeper depths (ρ = 0.6869, p < 0.0001), consistent with multivariate analyses (Tables 4.5 and 4.6). Depth and rugosity were similarly correlated to biomass (ρ = 0.6197, p < 0.0001; ρ = -0.4655, p < 0.0001, respectively). Density was greater on WB (33.8 ± 8.7/100 m^2) compared to EB (31.4 ± 6.0/100 m^2; Z = 2.871, p = 0.0041), and within the UM (97 ± 19.1/100 m^2) compared to shallow stratum (13.3 ± 1.9/100 m^2; Z = 8.906, p < 0.0001). Within the shallow stratum, there were no density differences between years, however, 2011 densities were higher than 2012 within the UM (Figure 4.8). No significant inter-annual biomass differences were found. *Chromis scotti* (purple reeffish) also preferred deeper, low relief habitats (depth: ρ = 0.3658, p < 0.0001; rugosity: ρ = -1.452, p = 0.0132; Figure 4.103). Density of all other Pomacentridae species were either significantly positively related to depth (*Abudefduf saxatilis* [sergeant major], *C. multilineata*, *Microspathodon chrysurus* [yellowtail damselfish], *Stegastes adustus*, *Stegastes diencaeus* [longfin damselfish], *Stegastes partitus* [bicolor damselfish], *S. planifrons*, *Stegates variabilis* [cocoa damselfish]), or there was no significant relationship (*Chromis cyanea*, *Chromis enchrysurus*, *Stegates leucostictus*; Figure 4.104). Similar patterns were observed with rugosity, with a significant density increase with increased rugosity for many pomacentrids (*A. saxatillis*, C. multilineata, M. chrysurus, S. adustus, S. planifrons). For the remaining species, (*C. cyanea*, *C. enchrysurus*, *S. diencaeus*, *Stegastes leucostictus* [beaugregory], *S. partitus*, *S. variabilis*), the trend with rugosity was not significant.

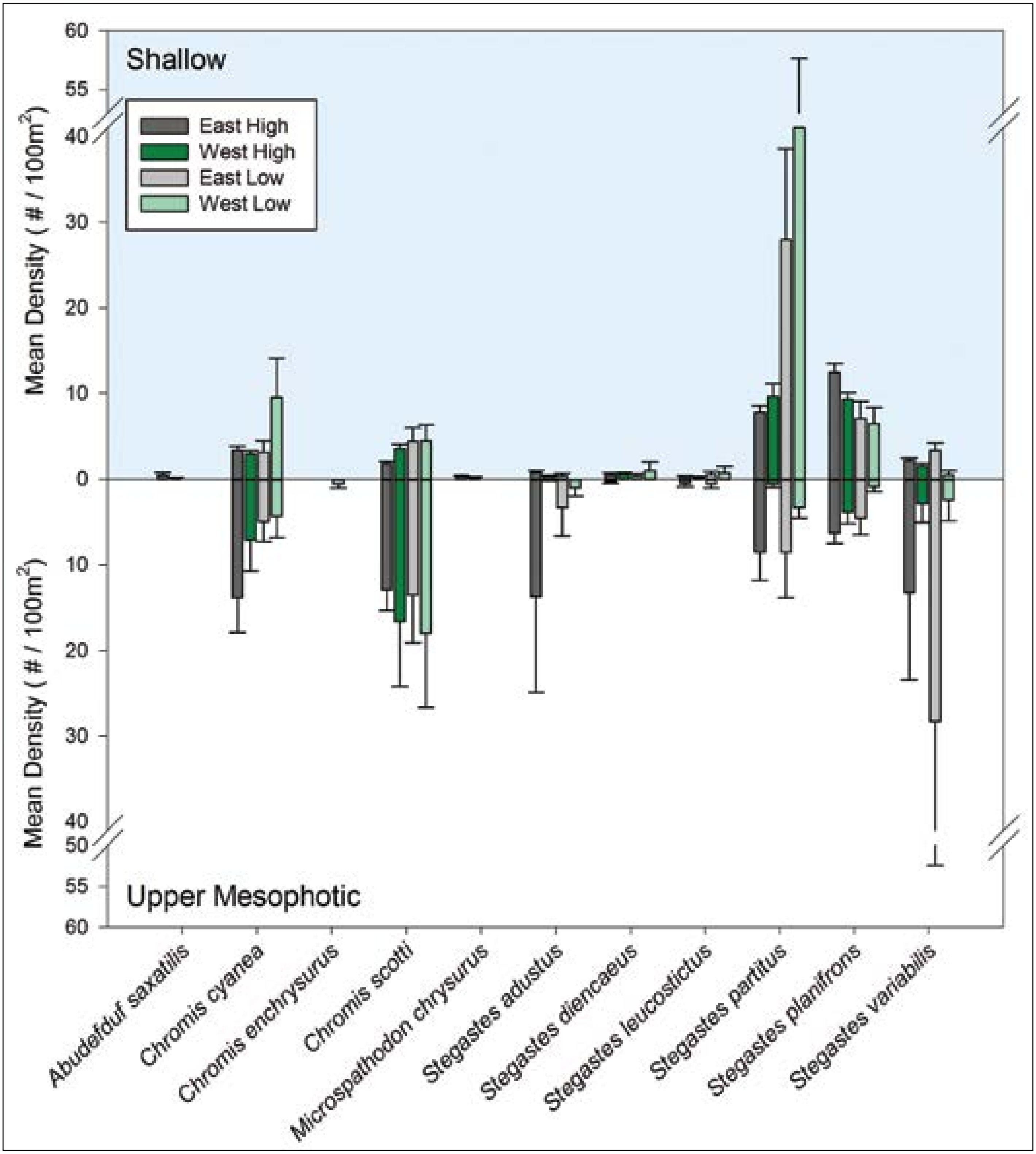

Figure 4.103. Mean density (#/100 m^2) of Pomacentridae species, except Chromis insolata *and* Chromis multilineata*, by strata for dive surveys (2010-2012).*

Fish Communities

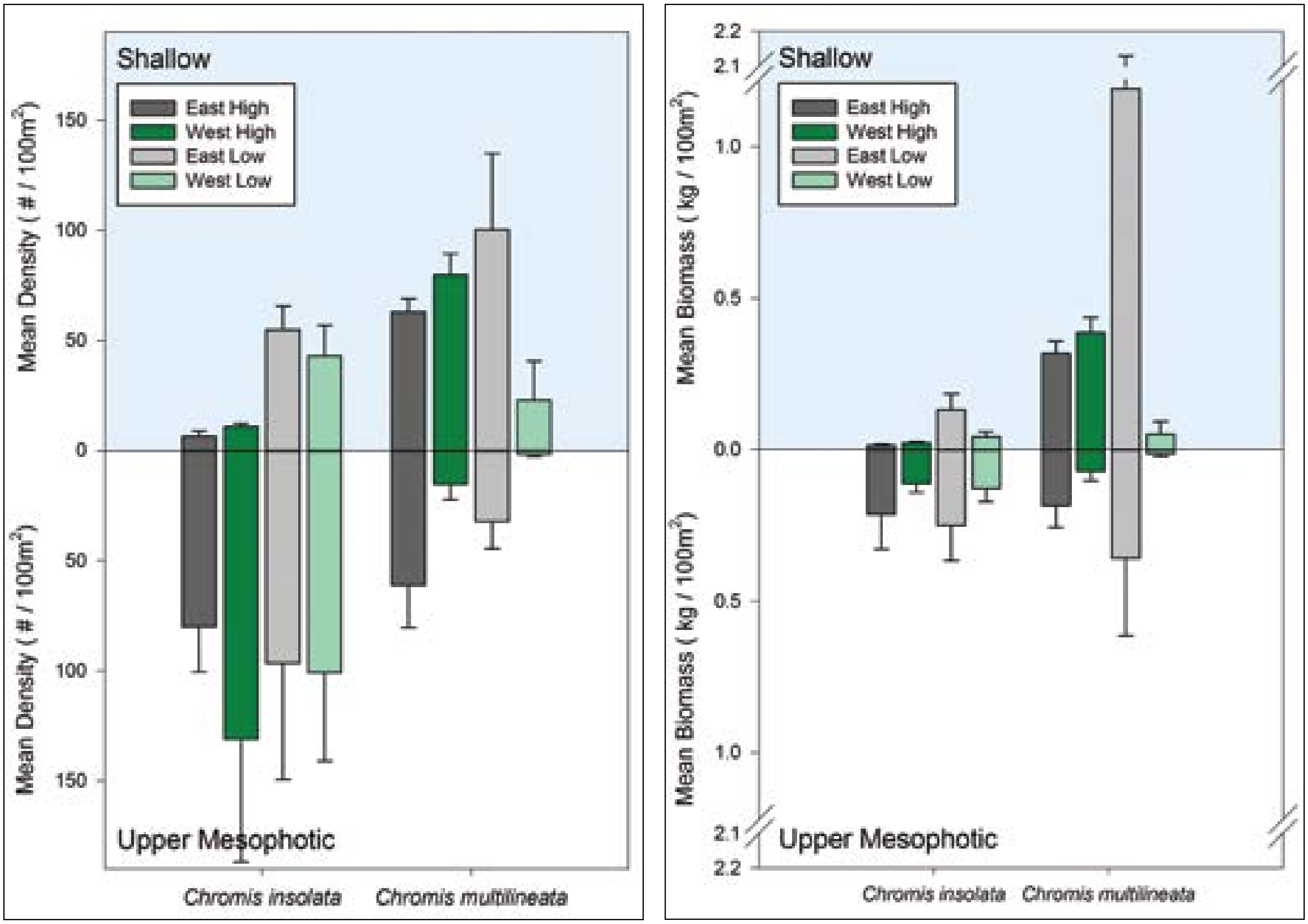

Figure 4.104. Mean density (#/100 m²; left) and mean biomass (kg / 100 m²; right) of Chromis insolata *and* Chromis multilineata *by strata for dive surveys (2010-2012).*

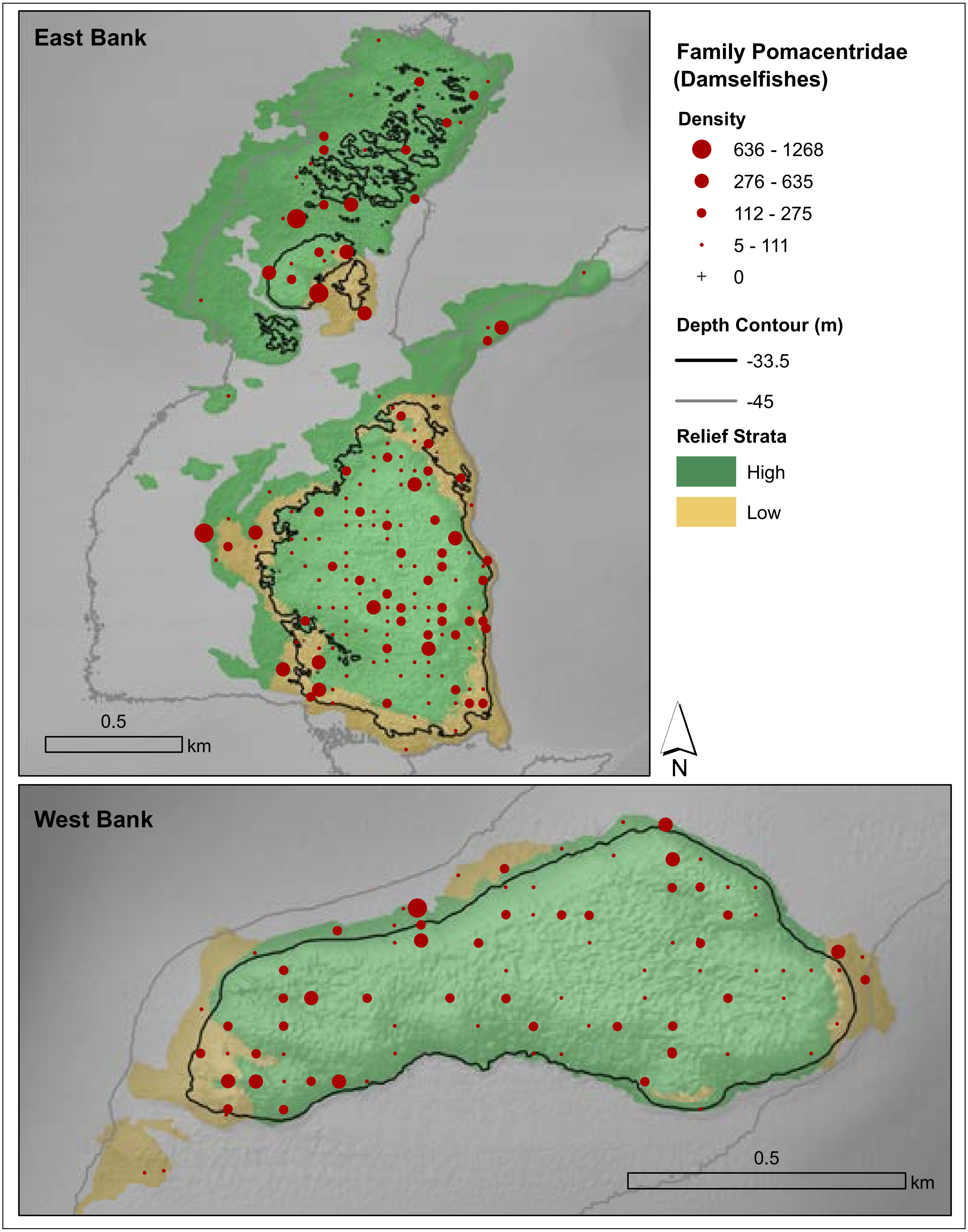

Figure 4.105. Observed density (#/100 m²) of Pomacentridae recorded during diver surveys from 2010-2012.

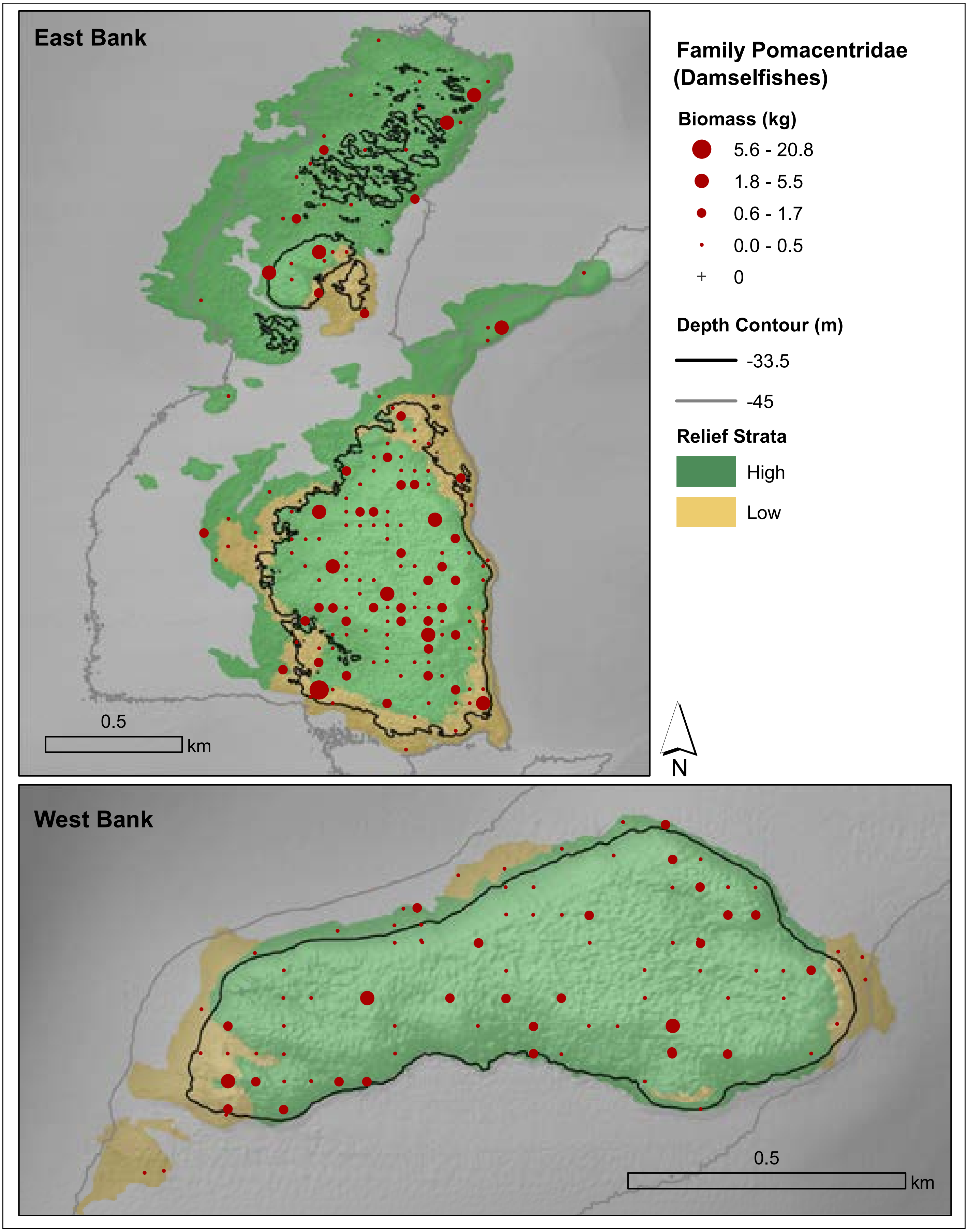

Figure 4.106. Observed biomass (kg/100 m²) of Pomacentridae recorded during diver surveys from 2010-2012.

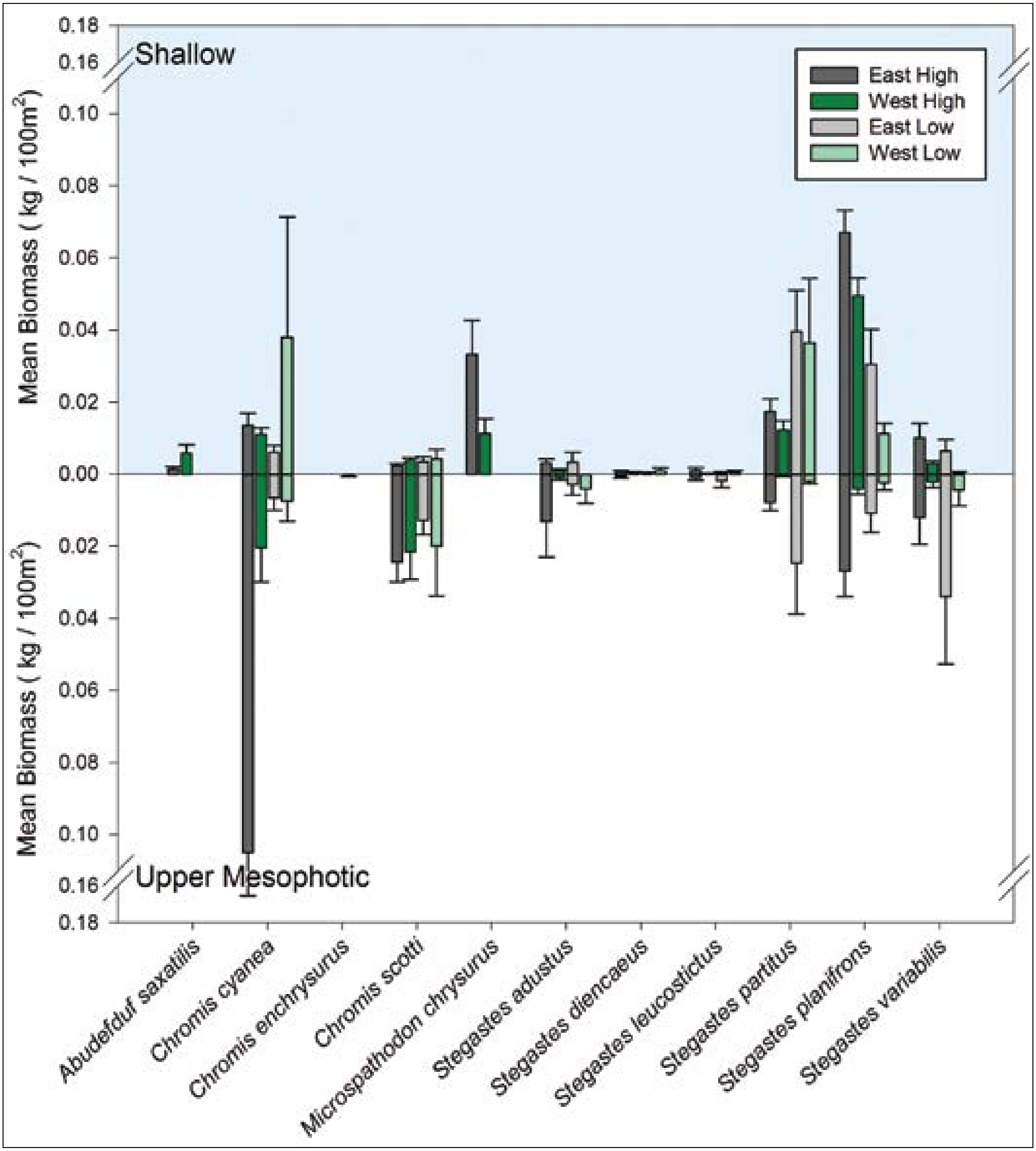

Figure 4.107. Mean biomass (kg/100 m²) of Pomacentridae species, except Chromis insolata *and* Chromis multilineata*, by strata for dive surveys (2010 – 2012).*

Scorpaenidae (Scorpionfish)

Pterois volitans (red lionfish)

*Figure 4.108. An invasive species, red lionfish (*Pterois volitans*) in FGBNMS. Photo: G. McFall (NOAA NOS/ONMS/GRNMS)*

Along the United States Southeast Atlantic coast, and in Bermuda and the Bahamas, the invasive red lionfish (*Pterois volitans*) are now established and are continuing to expand their range throughout the Gulf of Mexico and Caribbean (http://nas.er.usgs.gov/taxgroup/fish/lionfishdistribution.aspx; Figure 4.108). Native to the subtropical and tropical regions of the South Pacific, Indian Ocean and the Red Sea, lionfish are venomous predators whose popularity in the aquarium trade may have contributed to their introduction to Atlantic waters (Whitfield et al., 2002; Semmens et al., 2004; Ruiz-Carus et al., 2006). Recent studies of lionfish in the invaded range (Western Atlantic) are beginning to shed light on their ecology and biology (Morris et al., 2009; Morris and Whitfield, 2009; Muñoz et al., 2011). Lionfish have been documented at five to 177 times higher densities than in their native range (Grubich et al., 2009; Kulbicki et al., 2012), and are capable of reducing reef fish recruitment by up to 79% from experimental patch reefs, preying mostly on fishes (Albins and Hixon, 2008; Green and Côté, 2009; Morris and Akins, 2009; Muñoz et al., 2011).

During this study no lionfish were observed on fish transects prior to 2012, though lionfish were observed off transect on EB and WB in 2011, as well as on nearby Sonnier Banks, Stetson Bank, and neighboring oil and gas platforms beginning in 2010 (Johnston et al., 2013; http://nas.er.usgs.gov/taxgroup/fish/lionfishdistribution.aspx). In 2012, lionfish were recorded from 25 of 291 surveyed sites, sightings at 6 of these sites were off transect (Figures 4.109 and 4.110). Lionfish were seen at 12 and 13 sites on WB and EB, respectively (includes individuals off transect), though densities of lionfish were greater on WB than on EB (WB: 3.27 ± 0.541/100 m^2, n = 11 sites; EB: 1.38 ± 0.183, n = 8 sites, t-test, t=-3.468, p=0.003). Within the UM stratum, lionfish were recorded on all relief strata and only on high relief habitats within the shallow stratum (Figure 4.111). Two 5 cm FL individuals were observed in shallow depths, and 45 larger individuals (14 ± 1.11 cm FL) were observed at UM sites (Figure 4.112), consistent with multivariate analyses (Table 4.5).

Within the UM stratum, sites with lionfish were characterized by lower relief habitats compared to sites without lionfish (lionfish present: rugosity = 0.028 ± 0.0035, n = 23; lionfish absent: 0.035 ± 0.0023, n = 43; Mann-Whitney U = 605, p < 0.026). The maximum number of lionfish recorded during a single fish survey was seven, at a relatively low rugosity site on WB (rugosity = 0.018) in 39.2 m water depth.

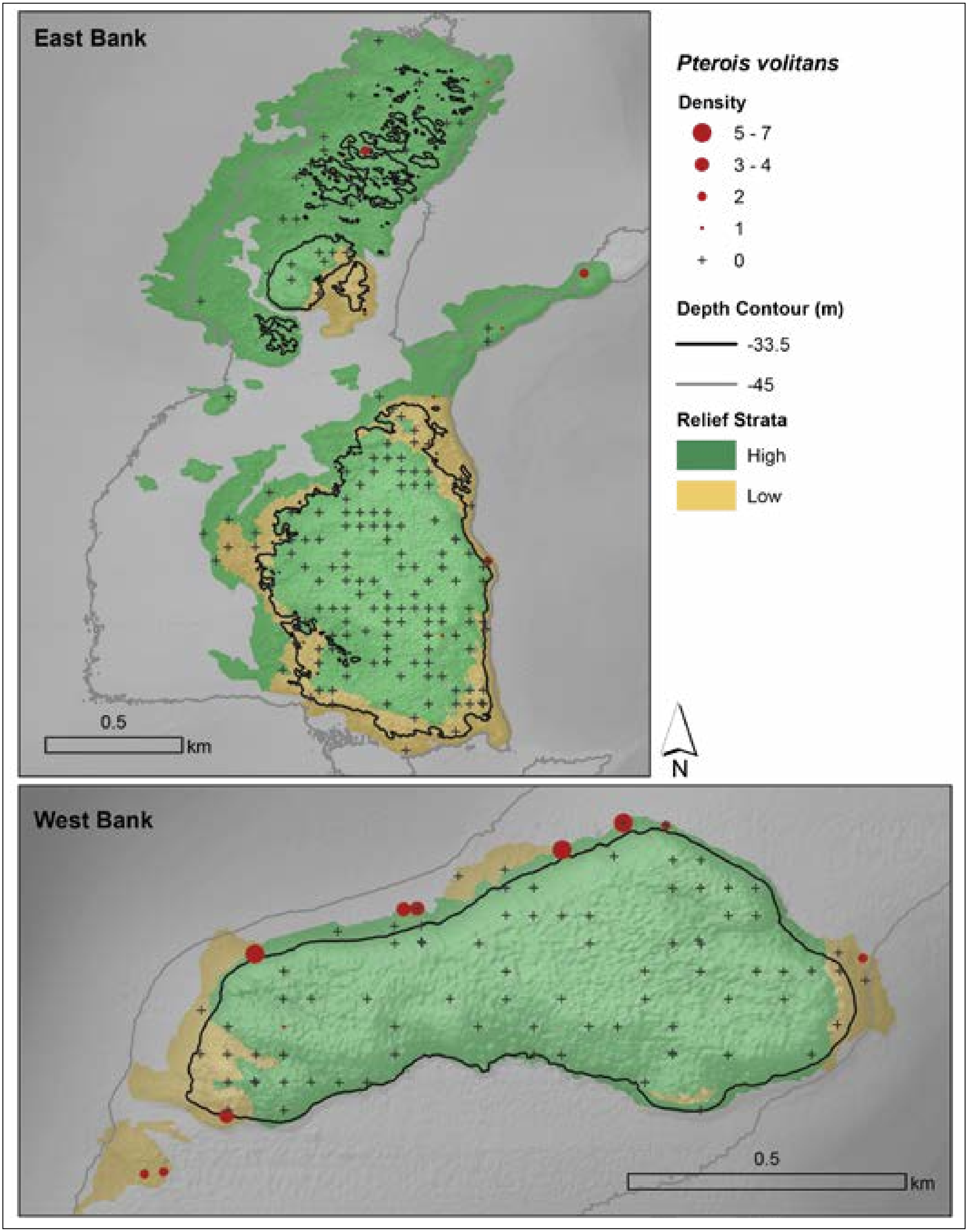

Figure 4.109. Observed density (#/100 m²) of Pterois volitans *(red lionfish) recorded during diver surveys from 2010-2012.*

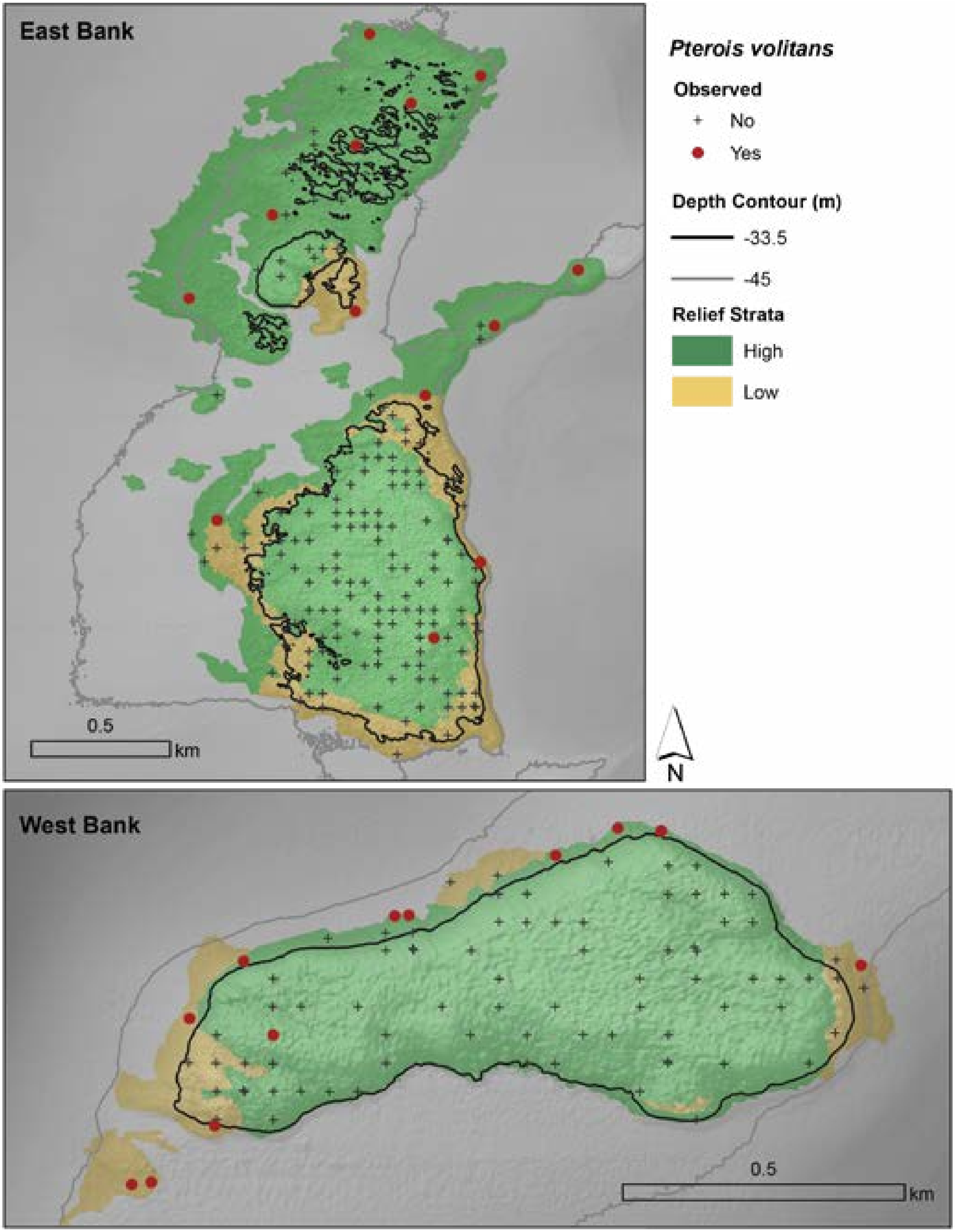

Figure 4.110. Observed Pterois volitans *(red lionfish), on or off transect, during diver surveys (2010-2012).*

Management Opportunity at FGBNMS

Current theory pertaining to invasive species impacts, the expanding lionfish distribution, observations that lionfish appear capable of settling to many different habitat types, and the overall pattern of generalist piscivory, all indicate the potential for significant impacts to the invaded community (Muñoz et al., 2011; Côté et al., 2013). Indeed, Morris and Akins (2009) found economically important *O. chrysurus* and *Epinephelus striatus* (Nassau grouper) in the stomachs collected from Bahamian lionfish. Given their planktonic larval dispersal and opportunistic colonization of habitats and use of food resources, large scale eradication of lionfish will not be feasible. Although sustained control measures can reduce lionfish size and densities at the local scale (Frazer et al., 2012), the restricted spatial scale of targeted removals means the costs of these efforts will need to be carefully evaluated against the effort and minimal ecological benefit that can be gained. Isolated and highly rugose habitats, such as FGBNMS, further complicate efforts of eradication.

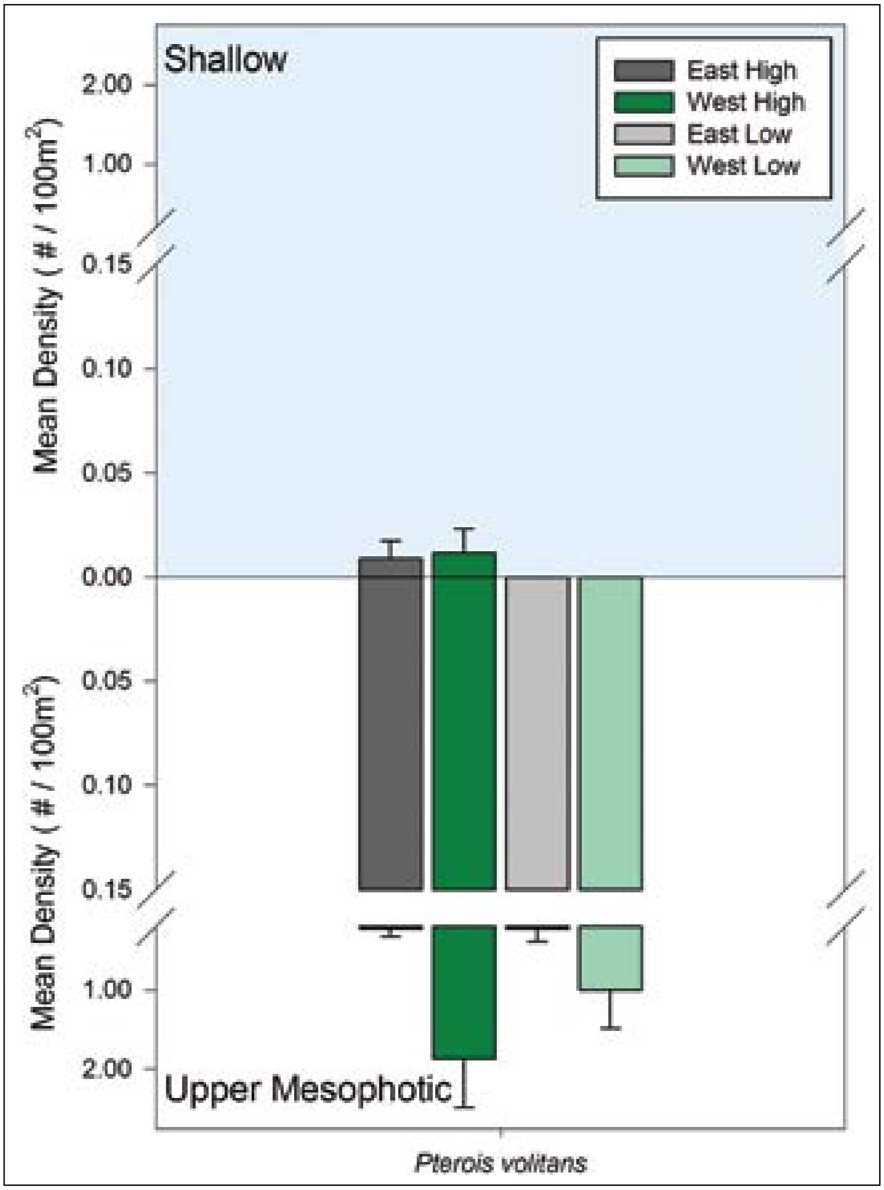

Figure 4.111. Mean density (#/100 m^2) of Pterois volitans*, red lionfish, by strata for dive surveys (2010-2012).*

Lionfish are believed to have few natural predators (Bernadsky and Goulet, 1991), reportedly due to their venomous spines, but conclusions from earlier studies are hampered by small sample sizes and suffer from the paucity of investigations in the native range. In the invaded range, lionfish have been found in the stomach contents of piscivorous serranids (Maljković et al., 2008), while Mumby et al. (2011) recently demonstrated a 7-fold reduction in lionfish biomass relative to grouper biomass in a Bahamian marine reserve. Anton (2013) has found similar results in other locations in the Bahamas and Belize, though both of these studies may be confounded by varying levels of lionfish removals from study sites. Additionally, numerous Atlantic fishes are capable of consuming venomous scorpaenids, including *Lophius americanus* (goosefish) and *Lutjanus analis* (mutton snapper), which are known to consume the venomous scorpaenid *Helicolenus dactylopterus* (blackbelly rosefish) and *Scorpaena plumieri* (spotted scorpionfish), respectively (Randall, 1967; Bowman et al., 2000).

Predation by large piscivores such as serranids and carcharhinids may represent one of the best controls for invasive lionfish (Albins and Hixon, 2008), as low densities (approximately 2.2 individuals/ha) of lionfish were observed in their native range on Palauan reefs with robust grouper populations (Grubich et al., 2009).

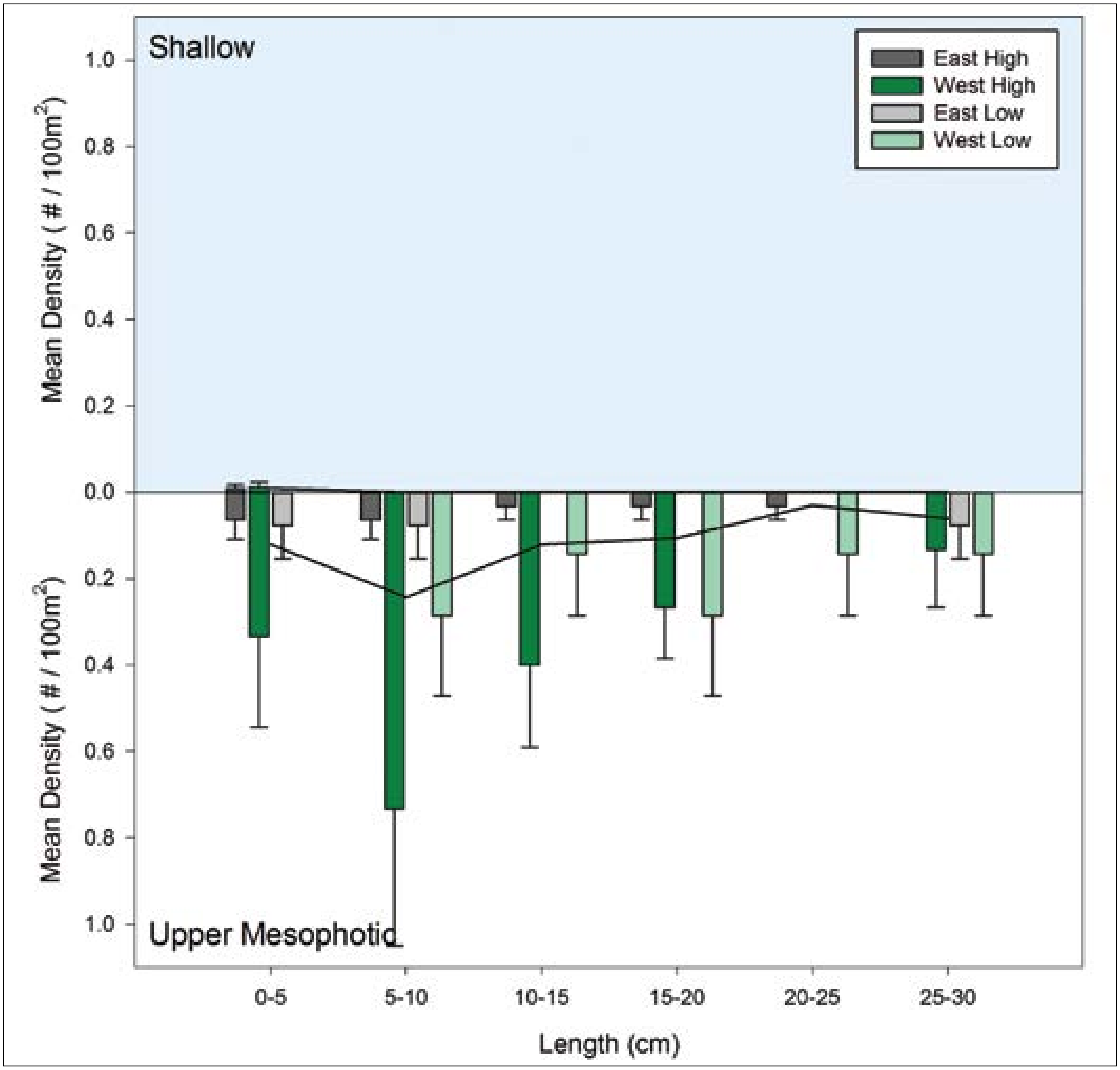

Figure 4.112. Pterois volitans *mean density (#/100 m²) and SE per size class (cm FL) by strata for diver surveys (2010-2012). Solid lines represent overall mean* P. volitans *density per size class.*

Reduced numbers of large predators in many invaded locations means that predation on lionfish may not occur at levels high enough to provide effective control. However, increased densities of exploited predators in marine reserves are often the first signs of positive responses to protection from fishing (Roberts and Polunin, 1993; Mosqueria et al., 2000; Côté et al., 2001). If predation on lionfish is a controlling mechanism, marine reserves may act as refugia where community assemblages are maintained with low densities of invaders by healthy populations of large predators. Reserves should thus be one of the first places to search for evidence of lionfish population control by predators. Currently, the effect of lionfish on native predators and the potential role of predation in controlling the number of lionfish is unknown and the subject of an active debate (Mumby et al., 2011; Hackerott et al., 2013; Mumby et al., 2013). This is due in part because previous studies of lionfish in reserves suffer from the confounding effects of ongoing lionfish removals (Hackerott et al., 2013; Mumby et al., 2013). Thus, controlled laboratory and correlative field studies that are not subject to confounding removals are an important research need and may shed light on natural lionfish control by native predators.

Remote and protected locations such as FGBNMS may be the most appropriate sites to examine the effects of native predators on lionfish. Their relative inaccessibility is often associated with an abundance of apex predators (Stevenson et al., 2007; DeMartini et al., 2008; Sandin et al., 2008; Friedlander et al., 2010), so these areas may have greater success controlling lionfish populations via predation or biocontrol. Importantly, the remote location should minimize the confounding effects of lionfish removals relative to nearshore locations. Compared to other Caribbean sites, FGBNMS has higher densities and biomass of apex predators. For example, at FGBNMS 32 sites supported apex predators [serranids, sphyraenids, carangids, carcharhinids] ≥100 cm FL. We hypothesize that this may lead to increased predation on lionfish, allowing large predators to act as a natural control. Outside of remote or protected locations, there is a relative paucity of large predators and so this natural control process should occur less frequently. Data supporting the possibility of natural control might include negative correlations between the density of predators and lionfish, and reduced sizes and numbers of lionfish from sites where large predators are found relative to nearby sites that do not harbor large predators. The lionfish invasion at FGBNMS is in its early stages so searching for these patterns at this time is likely premature, but it may be useful for managers to obtain highly resolved (spatially and temporally) baseline data now and plan for these future analyses by keeping detailed records of any lionfish that are removed from the sanctuary. The remote location of FGBNMS and management designation means limited removals of either predators or lionfish, thus avoiding confounding analyses of relationships between the two. Our ongoing study from a comparable location (remote, protected management status, abundance of large predators) in the Tortugas South Ecological Reserve is examining similar questions and will make a valuable reference site for predator-lionfish comparisons in FGBNMS. Such studies could have important implications for the importance of marine reserves, preservation of biodiversity and facilitation of community resilience to invasion, and for the management and natural control of lionfish.

4.4. SUMMARY

- A total of 129 species from 36 families were observed during the three years of diver surveys (2010-2012). Some species recorded here were not observed in a previous comprehensive survey of the shallow coral reef (max depth 32 m; Caldow et al., 2009), including: *O. chrysurus*, *S. radians*, *P. marmoreus*, *H. melanurum* and *I. vittata*. *H. melanurum* and *P. marmoreus* were recorded primarily in UM depths (32-45 m).
- Extending survey depths to 45 m allowed observation of significantly higher densities of some species (including but not limited to: serranids [*D. inermis*, *M. bonaci*, *M. interstitialis*], lutjanids [*L. griseus*], pomacentrids [*C. cyanea*, *C. scotti*, *C. insolata*], and acanthurids [*A. bahianus*]), compared to shallower depths (<32 m).
- The single most important factor structuring fish communities in the coral reef zone of the FGBNMS is depth, followed by habitat relief. In depths greater than 32 m, economically valuable species and apex predators, such as groupers, snappers and sharks were more numerous and larger. More reef associated fishes were found in high relief coral habitats, while juvenile and herbivorous fishes were more numerous within low relief algae dominated habitats. In contrast, there were few fish community differences between EB and WB.
- Highest fish densities and biomass were more often recorded in the UM stratum and along depth strata transitions on both banks. High relief sites typically had higher fish density and biomass than low relief sites; however, high densities and biomass were recorded in some low relief habitats. Many of these low relief sites were near the transition between high and low relief habitats.

Fish Communities

- The FGBNMS contain a diverse and abundant fish community complete with many large apex predators in the UM. Fishing is still permitted within the sanctuary, but little is known about fishing effort and distribution. Closure of an area or bank within FGBNMS may allow these fishes to remain within and expand beyond the deepest margins of the coral reef.
- Apex predator biomass in the UM strata was dominated by serranids, of which many species are known to exhibit relatively high site fidelity. Given the importance of apex predators to trophic flow in marine communities, and the association of apex predators with high coral cover and reef resilience, the significantly greater biomass of apex predators on UM strata warrants continued study and conservation of fishes and habitats in this zone.
- Planktivores were the dominant trophic group at FGBNMS in fish number (51% of total) and biomass (44.7%). While piscivores made up a small percentage of the fish community in number (4.6%), they ranked second in total biomass (28%).
- Marbled grouper (*D. inermis*), a species listed as Near Threatened on the International Union for Conservation of Nature (IUCN) Red List of Threatened Species, were found at 22 sites. They were recorded more often (13.6% of sites) and were larger (mean size: 59.4 cm FL) in UM depths than in shallow depths (5.8% sites, 45 cm FL). However, mean densities reported here were lower than those by Caldow et al. (2009). Although further investigation is needed, a northern area of East bank with multiple sightings may be a good candidate area for potential future management/monitoring consideration for this species. To further protect and manage this species, pursuing Species of Concern designation through NOAA's Office of Protected Species is recommended.
- Invasive red lionfish, *P. volitans*, densities are currently low but increasing within FGBNMS. There is potential for lionfish to impact native fish communities, but due to the remote location and depth of FGBNMS, large scale eradication of lionfish will not be feasible. However, the presence of larger apex predators within the sanctuary provides an opportunity to examine the potential role of natural predation as a biological control of lionfish. This is difficult to do in areas where large predators are absent or where periodic human removal efforts confound results. FGBNMS could take the lead on spearheading this kind of research and promoting conservation of large predators in the process.
- This study provides the first comprehensive and quantitative documentation of fish communities below 32 m depth at FGBNMS. Sixty-six sites were surveyed within the UM zone, providing complementary information to shallow surveys for fish biomass and species composition (particularly for apex predators). Continued monitoring and exploration of these deeper (>32 m) portions of the sanctuary are recommended to fully understand the connectivity between deep and shallow habitats.
- In addition to continued monitoring, emphasis should be placed on examining sites with abundant, large groupers and snappers. This would increase our knowledge of site fidelity and potential spawning activities of these apex predators, which would in turn improve the sanctuary's ability to make management decisions at a scale that is appropriate for these larger fish.

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Chapter 5
Benthic and Fish Communities in the Mid and Lower Mesophotic Zone of the Sanctuary

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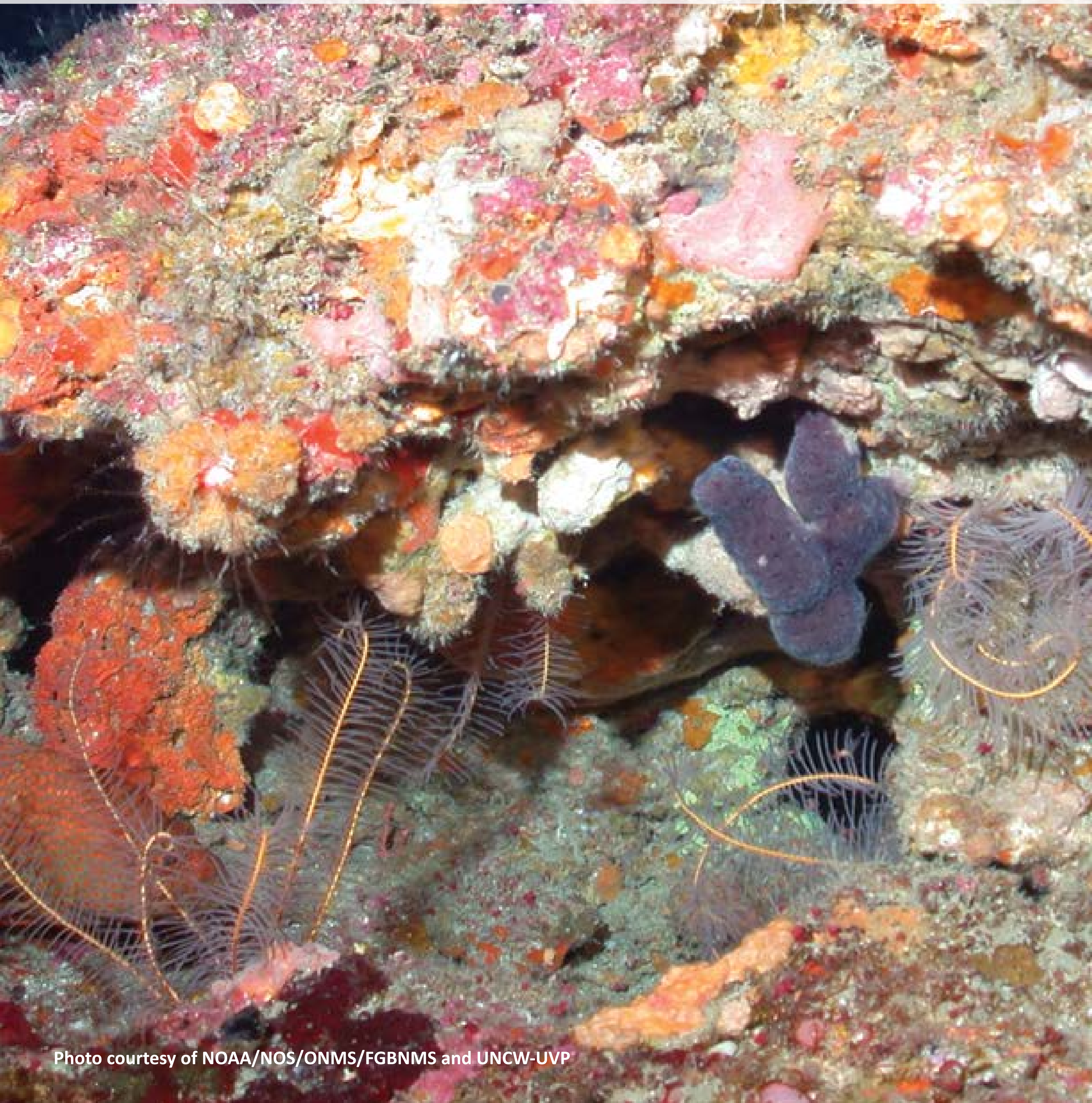


Photo courtesy of NOAA/NOS/ONMS/FGBNMS and UNCW-UVP

5.1. INTRODUCTION

Mesophotic coral reef ecosystems (MCEs) are warm water, light-dependent coral reef communities starting at approximately 30 m to the deepest depth of the photic zone, which varies by location and may extend to 150 m in some regions (Khang et al., 2010). MCEs can serve as an extension of shallow water reef ecosystems and provide a unique opportunity to investigate similarities and differences between the two adjacent systems.

Despite their proximity to shallow reef ecosystems, MCEs are poorly understood due to the logistical difficulties and safety issues associated with working near or below the depths of recreational SCUBA diving (Pyle, 1996; Menza et al., 2008). Recently, MCE studies have revealed extensive and diverse fish and benthic communities that are different than those found on shallow reefs. The composition of fish communities associated with MCEs follows similar patterns of sessile benthic fauna (Khang et al., 2012). In general, fish abundance and species richness tend to be dramatically lower than that observed on the shallow reef system (Lukens, 1981; Nelson and Appeldoorn, 1985; Itzkowitz et al., 1991) with composition and numerical dominance being strongly correlated with live coral cover and habitat complexity.

The Flower Garden Banks shallow water coral reef fish and benthic communities have been characterized (Bright et al., 1985; Gittings et al., 1992; Pattengill-Semmens, 2006; Precht et al., 2006) and spatially analyzed (Caldow et al., 2009), and long term monitoring stations have been established on each bank with annual monitoring conducted since 1998 (Johnston et al., 2013). Due in part to their status as a national marine sanctuary, the Flower Garden Banks are the most well studied banks in the northern Gulf of Mexico (Kahng et al., 2012). However, deep areas of the banks have only been partially characterized and the relationships between benthic habitat and fish community composition are not fully understood.

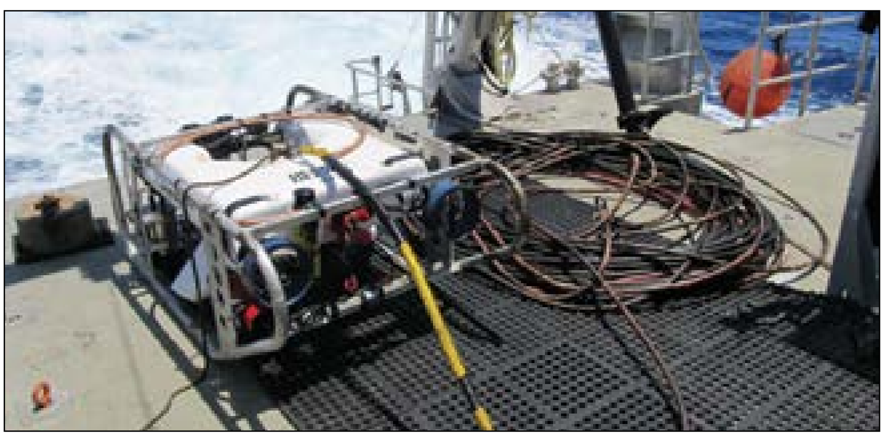


Figure 5.1. Remotely operated vehicles (ROVs) like this Super Phantom S2 are valuable tools to study deep water communities. Photo: L. Horn (UNCW)

Historically, remotely operated vehicle (ROV) surveys (Figure 5.1) have been used in an exploratory role in the deep habitats of the sanctuary (>35 m). This previous work has generated an inventory of fish and benthic organisms, including crustose coralline algal, antipatharians, azooxanthellate gorgonians, fish, azooxanthellate and zooxanthellate scleractinians, sponges and crinoids. However, there was no quantitative baseline of community structure for fish and benthic organisms. This assessment used an ROV to quantify benthic and fish populations among the habitats described in Chapter 2. Fish communities were not quantified in the coral reef zone while all habitats were included for the benthic community assessment.

5.2. METHODS

5.2.1. Survey

Deep water (>46 m) benthic and fish communities were surveyed by conducting 100 m ROV transects with both continuous forward looking video footage and downward looking still photography. A stratified random design similar to that used for the 2,500 m^2 SCUBA surveys was employed with larger (40,000 m^2) sampling frame structure (Figure 5.2). Each frame was classified using the benthic habitat map biological zones (see Figures 2.8 and 2.9). When multiple zones were contained within a grid, the grid was classified with the zone that had the majority of area. Each year site allocation was intended to be equitably distributed. However,

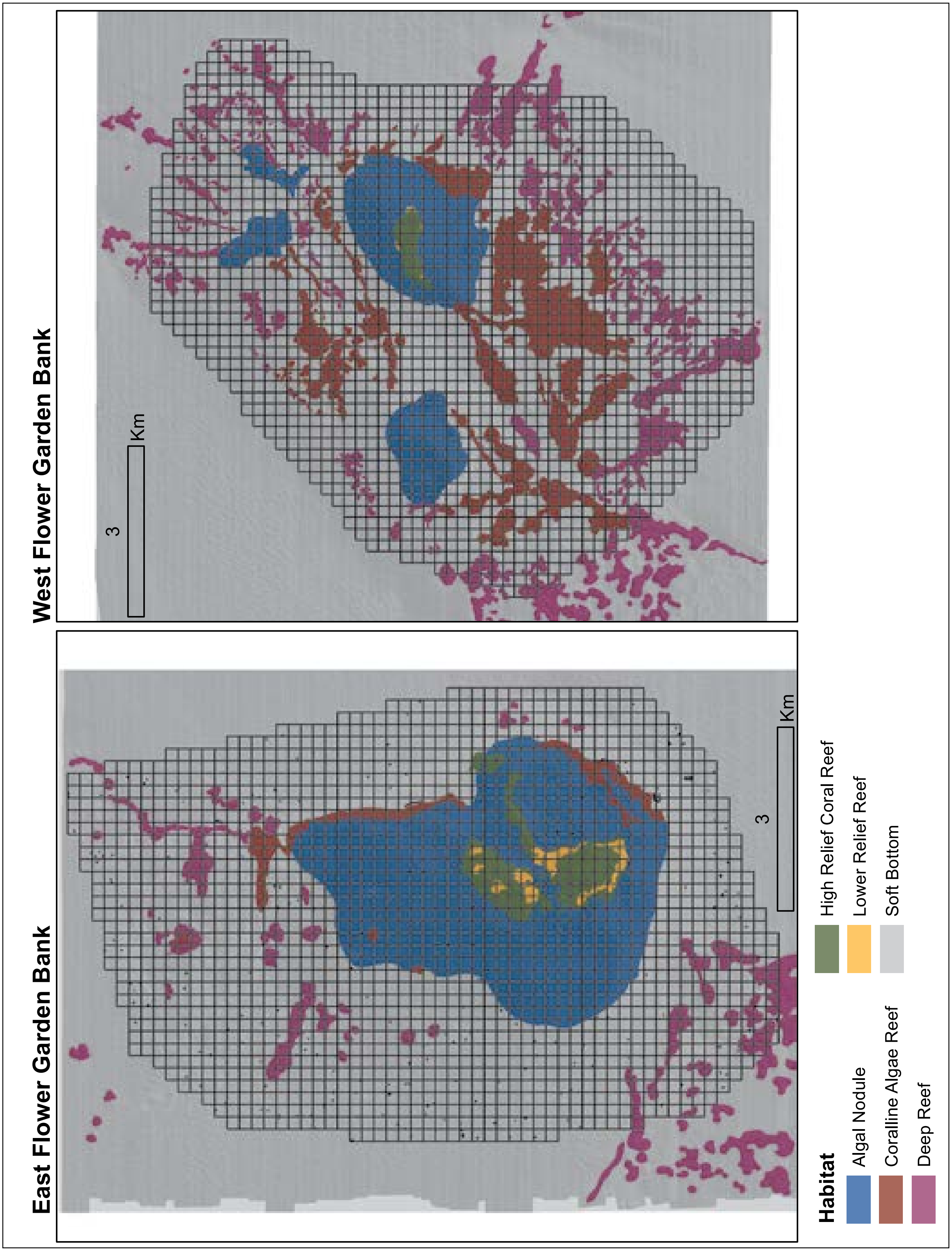


Figure 5.2. Sampling grid (200 x 200 m) for ROV benthic and fish surveys.

Table 5.1. Number of fish and benthic surveys conducted by remotely operated vehicle (ROV) by year and strata.

	2010		2011		2012		Total
	EB	WB	EB	WB	EB	WB	
Coral reef	7	5	-	-	-	-	12
Algal nodule	8	10	7	4	3	4	36
Coralline algae reef	9	13	9	8	14	13	66
Deep reef	12	9	6	7	17	16	67
Soft bottom	5	4	10	4	8	1	32
Total	**42**	**40**	**32**	**23**	**42**	**34**	**213**

poor sampling conditions and benthic habitat classification errors skewed habitat allocations (Table 5.1). Site selection was conducted with an ArcMap GIS sampling design tool (Buja and Menza, 2009). The survey design used a stratified random approach with four habitat types (algal nodule, coralline algae reef, deep reef, and soft bottom; see Figure 2.8 and 2.9) and the two banks, East Bank (EB) and West Bank (WB). ROV specifications and associated equipment are provided in detail in Appendix D. In 2010, 12 surveys on shallow coral reef (18-40 m) were targeted to compare fish data collected by ROV with those by *in situ* diver observations.

In the field and prior to ROV deployment, a select cluster of sites was chosen and conditions defined as to how the ROV will travel (e.g., under its own control or towed by the surface vessel). Under ideal conditions and when the ROV operator had good control of the ROV, sampling commenced as close as possible to the centroid of each sampling point, or if conditions hindered ROV handling, within the 200 m^2 grid cell. Transect speed was ¼ knot and followed the target habitat type for 100 m. At the conclusion of the transect, ROV speed was increased to transit to the next station.

In addition to high resolution video, tracking and depth information were also collected to provide real-time estimates of ROV depth and position on the seafloor. Fish and benthic features were identified and enumerated qualitatively by a team of scientists watching a live feed from the ROV. The video was reviewed later to verify identification and estimate abundance of taxa and benthic habitat types.

5.2.2. Benthic Data

Benthic community information was collected along each transect using a digital still camera (for camera specs see Appendix C) positioned underneath the ROV and perpendicular to the seafloor. Still photos of the seafloor were taken by the ROV approximately every 30 seconds with the ROV positioned approximately 1 m from the bottom. On average, 12.7 images were taken on each transect. Photoshop CS5 was used to adjust images for color and contrast to improve image analysis. Images that were excessively dark or blurry were removed from the analysis. Initially, 2,732 photos among 215 transects were analyzed for use and after post processing, 2,259 photos from 205 transects were included in the analysis. Photos saved for analysis were first scaled using laser points from the ROV in Coral Point Count with Excel extensions (CPCe 4.0; Kohler and Gill, 2006) to determine image area. To ensure capture of low percent cover types and to avoid errors, fifty points were randomly transposed over each image (Dumas et al., 2009; Pante and Dustan, 2012) and the benthic type under the point was recorded. Generally, bare soft bottom, bare hard bottom and biota were identified. Within biota, cnidarians in the Class Alcyonacea and in the Order Antipatharia were identified to the family level. Cnidarians in the Order Scleractinia were identified to species level. Algae were identified to Phylum, and sponges were identified to Class. When the area under a point could not be identified, a label of “unidentifiable” was used. All fish, mollusk, echinoderm, bacterial mat and “unidentifiable” points were omitted from the family and species level analyses.

The percent cover of each benthic type was determined by pooling all images within that transect (except those omitted by the criteria outlined above). In addition to the percent cover data, the density of each coral taxon was quantified by counting individuals within all images for each transect. Transects are the units of replication within each habitat type and bank for the percent cover and density data. Low biota densities in most habitat types prevented quantitative assessment of within transect heterogeneity.

5.2.3. Fish Data

In the lab, videos were visually analyzed by a member of each ROV cruise team to identify and enumerate fish to species or lowest taxon and fork length (FL) in 5 cm bins up to 35 cm. Actual fish FL was used for fish greater than 35 cm. The ROV was equipped with two forward projecting lasers (with a linear distance of 15 cm) positioned in the center of the video frame to estimate fish sizes. Fish biomass was estimated using published length-weight ($W = a \times L^b$) parameters for each species. *W* is weight in grams, and *L* is length in mm and *a* and *b* are constants. If species specific information was not available, information for the most appropriate congener was used. Fish were also assigned to a trophic group (piscivore, invertivore, planktivore or herbivore) based on published information or from information provided from FishBase (www.fishbase. org). Tracking information from the ROV was used to calculate catch per unit effort (CPUE; Barry and Baxter, 1993; Pacunski et al., 2008) as transect length varied as a result of currents, ROV entanglements or other factors. Fish identification and measurement was best conducted at close range. By measuring the width of the monitor in relation to the lasers we determined the most successful identification and measurements came within 5 m of the ROV where the width of the field of view was approximately 1 m on either side of the ROV (Pacunski et al., 2008). CPUE was thus calculated as the *linear distance traveled* x 2 m. For comparisons among ROV surveys and to be consistent with fish metrics reported in Chapter 3, fish metrics derived from ROV surveys were standardized to 100 m^2. There are significant differences regarding field of view using an ROV and that of in-situ divers. Comparisons of fish communities observed with these two techniques should be taken with care in evaluating perceived differences in locations.

5.2.4. Benthic Analysis

The relative abundance of biota, hard bottom and soft bottom were compared using non-parametric techniques using bank and habitat type as factors. Relative abundances based on percent cover data were used to calculate Bray-Curtis similarity matrices (Bray and Curtis, 1957).

The Bray-Curtis similarity values were square root transformed to reduce the impact of extremely dominant common members of the community assemblages. To facilitate analyses of transects with relatively denuded cnidarians assemblages, zero-adjusted Bray-Curtis coefficient were used following the procedure of Clarke et al. (2006). This method uses dummy variables as a means to supplement the abundances of the least frequently observed organism. We used non-metric multidimensional scaling (nMDS; Kruskal, 1964) to visualize dissimilarities in the community composition among transects. Each ordination was run with 100 random starting configurations to determine the best fit model for non-parametric regression between the distance among samples on the plot and Bray-Curtis similarity. In order to test variation in community structure among the various habitat types and between East and West Bank, analysis of similarity was used (ANOSIM; Clarke and Warwick, 2001). For groups discriminated by ANOSIM, the contribution of each category to the average dissimilarity between the different groups was determined using similarity percentage, (SIMPER; Clarke and Warwick, 2001). Non-metric MDS ordinations, ANOSIM and SIMPER were all carried out using Primer 6 (Primer-E Ltd., Plymouth, UK).

5.2.5. Fish Analysis

Domain-wide Population Estimates

Domain-wide estimates were computed using methods described by Cochran (1977) for a stratified sampling design. Summary statistics include: total species occurrence, percent occurrence, total abundance, mean abundance (+/- standard error [SE]), total biomass and mean biomass (+/- SE) were generated for all species and trophic groups for each bank.

Strata Comparisons

Differences and similarities in the species composition of communities between samples were examined using a species-abundance by site data matrix. Infrequently observed fish that were not identified to species level were removed. The matrix was fourth-root transformed to ensure that rare and intermediate abundance species, in addition to the highly abundant species, played a significant role in determining patterns in community composition. The data was then used to construct a matrix of the percentage similarity in community composition between all pairs of sites using the Bray-Curtis Coefficient. An ANOSIM test, a multivariate version of Analysis of Variance (ANOVA; Primer v6; Clarke and Warwick, 1994), with 999 permutations was used to test for significant differences in fish assemblage composition between mapped classes at multiple thematic resolutions including: 1) habitat type – algal nodule, coralline algal reef, deep reef, softbottom; 2) bank; and 3) strata. The R value (ranging from 0-1) is a better relative indicator of the amount of dissimilarity between groups than the significance test and is thus given greater emphasis here. It is usually interpreted as the pairs of fish assemblage composition being: $R<0.25$ = barely separable; $R>0.5$ = overlapping but clearly different and $R>0.75$ = well separated.

For a visual examination of patterns of between site similarity a two-dimensional nMDS was constructed. This information determines whether benthic map classes and thematic levels are delineated in an ecologically meaningful way. The multivariate analyses also provide an assessment of the ability of the benthic habitat map to predict patterns of fish assemblage composition. The similarities/ dissimilarities should not be interpreted as a measure of connectivity between habitat types as has been suggested by Chittaro et al. (2005), although similarity between neighboring habitat types may result from inter-habitat movements and resource utilization that would need to be validated by direct observations of space use patterns.

We also used the ANOSIM test to examine the difference in observation of in-water diver fish surveys and fish surveys conducted using an ROV. In 2010, we co-located 12 ROV and *in situ* SCUBA surveys to provide a preliminary assessment of the differences between the two techniques. See Chapter 3 for SCUBA methods. We compare fish metrics among strata and between years with paired t-tests if the data are normal or Wilcox rank sums test if data are not able to be normalized.

Lastly, we assess the abundance and distribution for the large bodied groupers and snappers. Abundance, size structure and habitat preferences were examined to obtain an estimate of their population size in the sanctuary.

5.3. RESULTS AND DISCUSSION

5.3.1. ROV Surveys

One ROV survey cruise was conducted each year from 2010-2012, totaling three ROV cruises. Harsh sea conditions limited survey effort during the 2011 ROV cruise (N=55; Table 5.1). Figure 5.3 displays the spatial distribution of ROV transects. In total, 215 transects were conducted among the 10 strata, with an area of 58,600 m^2 or 0.06 km^2. On EB a total of 7,727 m^2 of algal nodule, 8,880 m^2 of coralline algal reef, 11,858 m^2 of deep reef and 6,248 m^2 of soft bottom habitat were surveyed by ROV. On WB we surveyed 3,977 m^2 of algal nodule, 8,758 m^2 of coralline algal, 9,152 m^2 of deep reef and 1,997 m^2 of soft bottom habitats. Combined, benthic habitat and fish survey area conducted by ROV totaled was equivalent to 1.1% of sanctuary habitats (Table 5.2).

Table 5.2. Area of ROV surveys by strata.

	East Bank			West Bank		
	Area Surveyed km^2	Habitat area km^2	% of total area	Area Surveyed km^2	Habitat area km^2	% of total area
Algal nodule	0.007	15.21	0.05	0.003	7.45	**0.04**
Coralline algal reef	0.008	1.41	0.57	0.008	9.69	**0.08**
Deep reef	0.01	3.53	0.28	0.009	9.57	**0.09**
Softbottom	0.006	65.81	0.01	0.001	77.47	**0.00**
Coral reef	0.002	2.63	0.08	0.001	0.48	**0.21**
Total	**0.033**	**88.59**	**0.98**	**0.022**	**104.66**	**0.44**

Depth of surveys on algal nodule habitats ranged from 47-102 m on EB and 51-83 m on WB. EB coralline algal reef surveys ranged from 52-97 m, and 77-97 m on WB. Deep reef surveys on EB ranged from 69-107 m while surveys on WB deep reefs were between 92 and 123 m. Survey depth on soft bottom habitats ranged from 89-106 m on EB and 85-118 m on WB.

Benthic Assessment

The relative percent cover of bare hard bottom, bare soft bottom, and biota (any observed macro-organism) differed among habitat types, but were generally consistent between banks (Figure 5.4). Soft bottom and deep reef habitats on both EB and WB were characterized by >90% soft bottom cover with biota comprising less than 1% of soft bottom habitats and about 3% of deep reef habitats. Conversely, coral reef cap habitat on both banks was dominated by >75% biota on average, predominantly scleractinian corals (see detailed analyses below). Biota accounted for roughly half of the benthos in algal nodule habitat on both banks. Whereas, coralline algal reef transects were dominated by biota on the EB, the same habitat type was

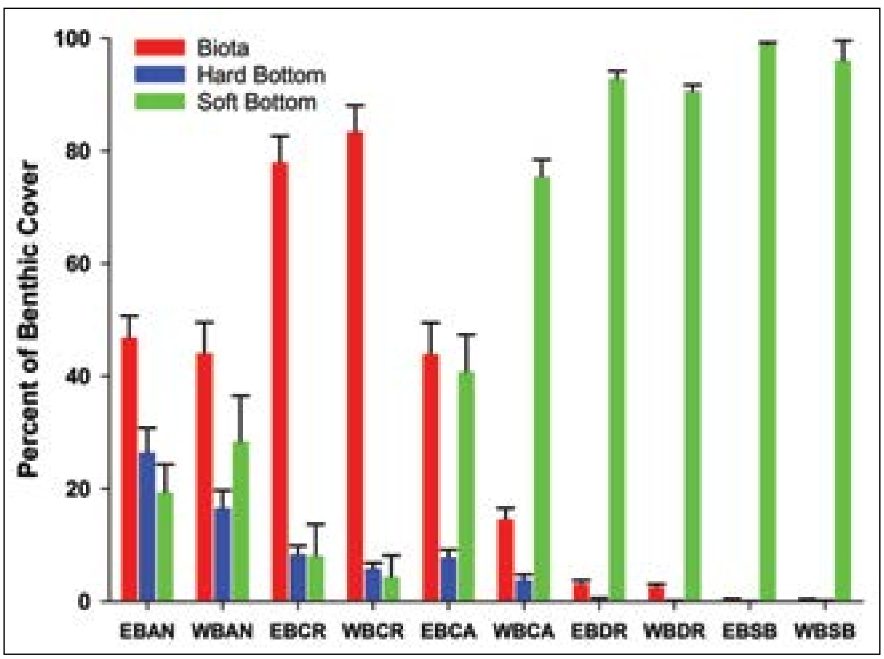


Figure 5.4. Mean ± SD for percent cover of abiotic hardbottom, abiotic softbottom and biota by bank and habitat type. Habitat types: EBAN= EB algal nodule habitats, WBAN= WB algal nodule habitats, EBCR= EB coral reef habitats, WBCR= WB coral reef habitats, EBCA= EB coralline algal habitats, WBCA= WB coralline algal habitats, EBDR= EB deep reef habitats, WBDR= WB deep reef habitats, EBSB= EB softbottom habitats, WBSB= WB softbottom habitats.

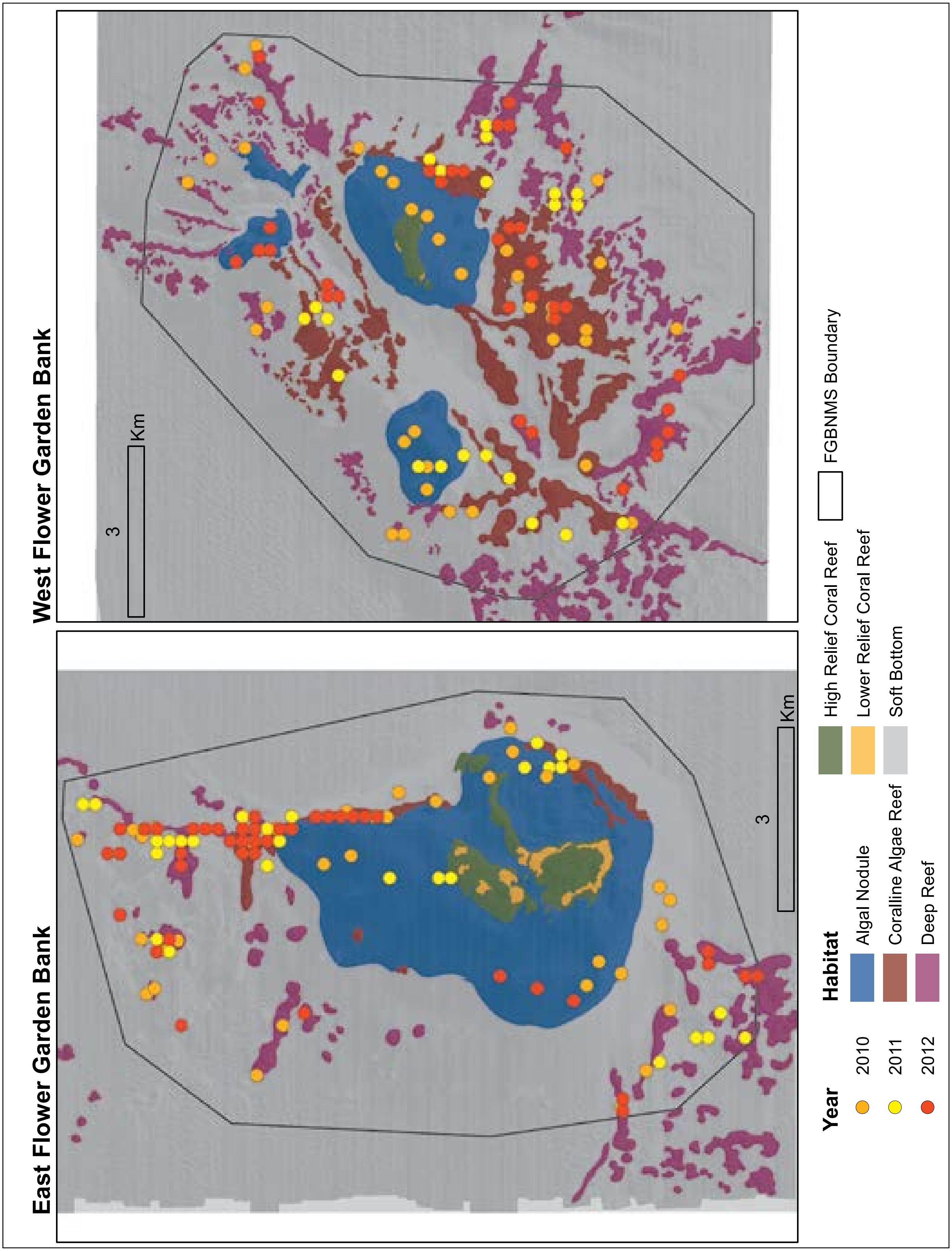


Figure 5.3. Location of benthic and fish ROV surveys, 2010-2012.

dominated by soft bottom on the WB. Percentage based response variables for such broadly defined criteria usually can be analyzed with parametric 2-way ANOVA. However, here the response variables are correlated (i.e., as biota increases, hard and soft bottom must decrease). Therefore a 2-way MANOVA would be the appropriate test. The data failed tests of normality, multivariate normality, and equity, limiting any potential inferences from a parametric approach. Attempts at transformation did not improve the normality of the data. As a result, these response variables were treated like community composition types in the nonparametric approach traditionally applied to characterize similarities in taxonomic diversity among samples or transects (Figure 5.5). In a two-way ANOSIM, both bank ($p < 0.01$, R= 0.127) and habitat ($p < 0.01$, R = 0.561) are significant, with all pairwise comparisons between habitat types significant ($p < 0.01$).

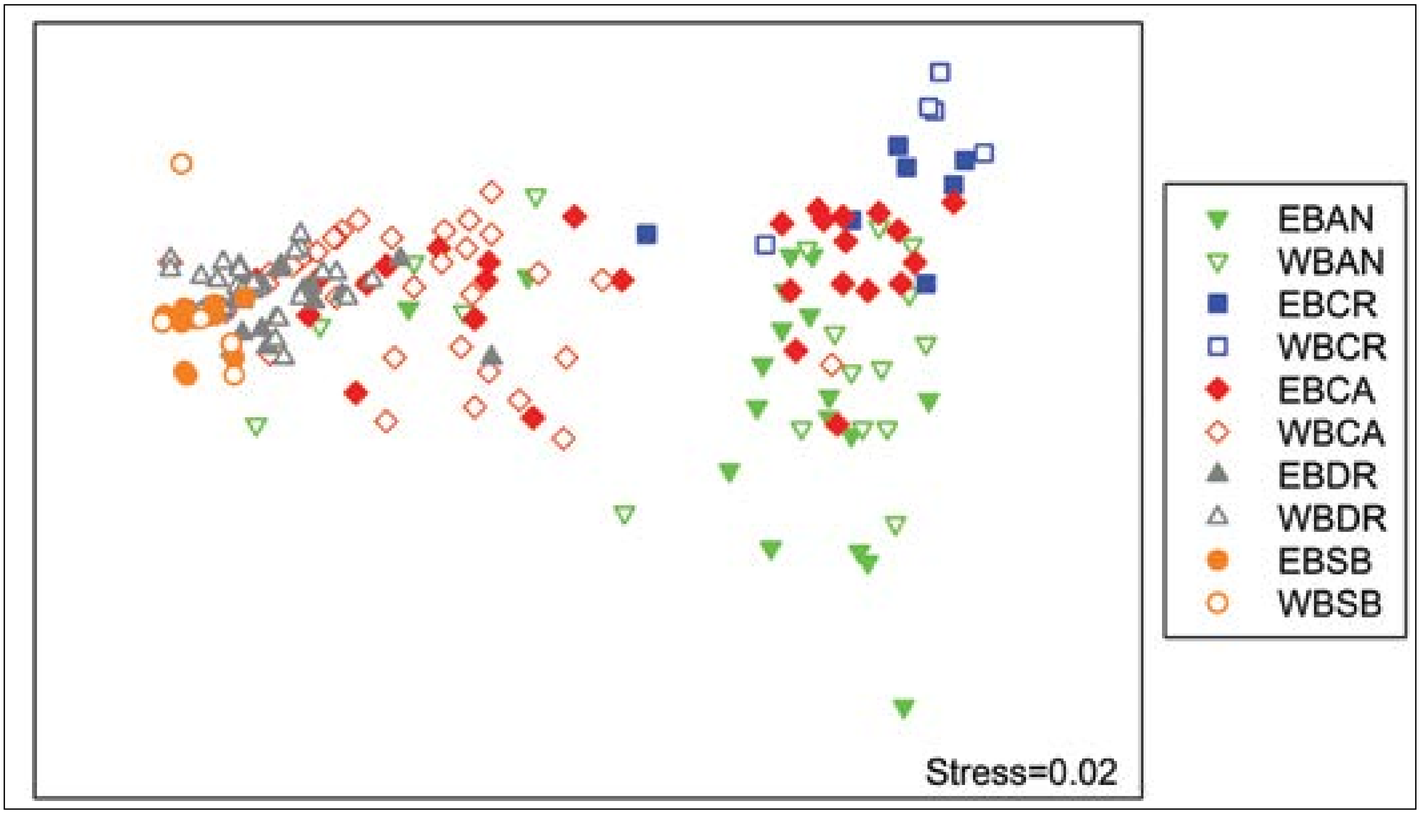


Figure 5.5. Non-metric multidimensional scaling (nMDS) plot based on square root transformed Bray-Curtis similarities from benthic cover data by biota, hard bottom, and soft bottom. Each symbol represents a transect; nearby symbols have similar community structure and distance symbols have disparate community structure. Here both bank and habitat type were significant factors.

Given that both benthic biota and bare substrate cover can dictate fish habitat suitability the data were examined at the level of phyla, including bare substrate (Figure 5.6). Again, all habitats, with the exception of coralline algal reefs, have similar composition between EB and WB. See Figure 5.7 to see the percent cover of hard and softbottom and biota by survey station for each bank. Cnidaria were abundant on the coral reefs, with nearly all observed cnidarians being hard corals. Whereas coralline algal reefs at EB were dominated by CCA (Phylum Rhodophyta), bare soft bottom was far more common in the coralline algal reefs on the WB.

When comparing the benthic composition using nMDS (Figure 5.8) and ANOSIM, both bank and habitat were significant ($p < 0.01$, global R = 0.163; $p < 0.01$, global R = 0.588, respectively), with all pairwise comparisons between habitat significant ($p < 0.01$), except deep reef and soft bottom ($p > 0.05$, R = 0.072). SIMPER analysis identified that the abundance of cnidaria drove differences between coral reef and algal nodule (31.9% contributed dissimilarity by cnidaria) as well as coral reef and coralline algal reef (29.4% contributed dissimilarity by cnidaria). The percent cover of the phylum Rhodophyta contributed most to differences between coralline

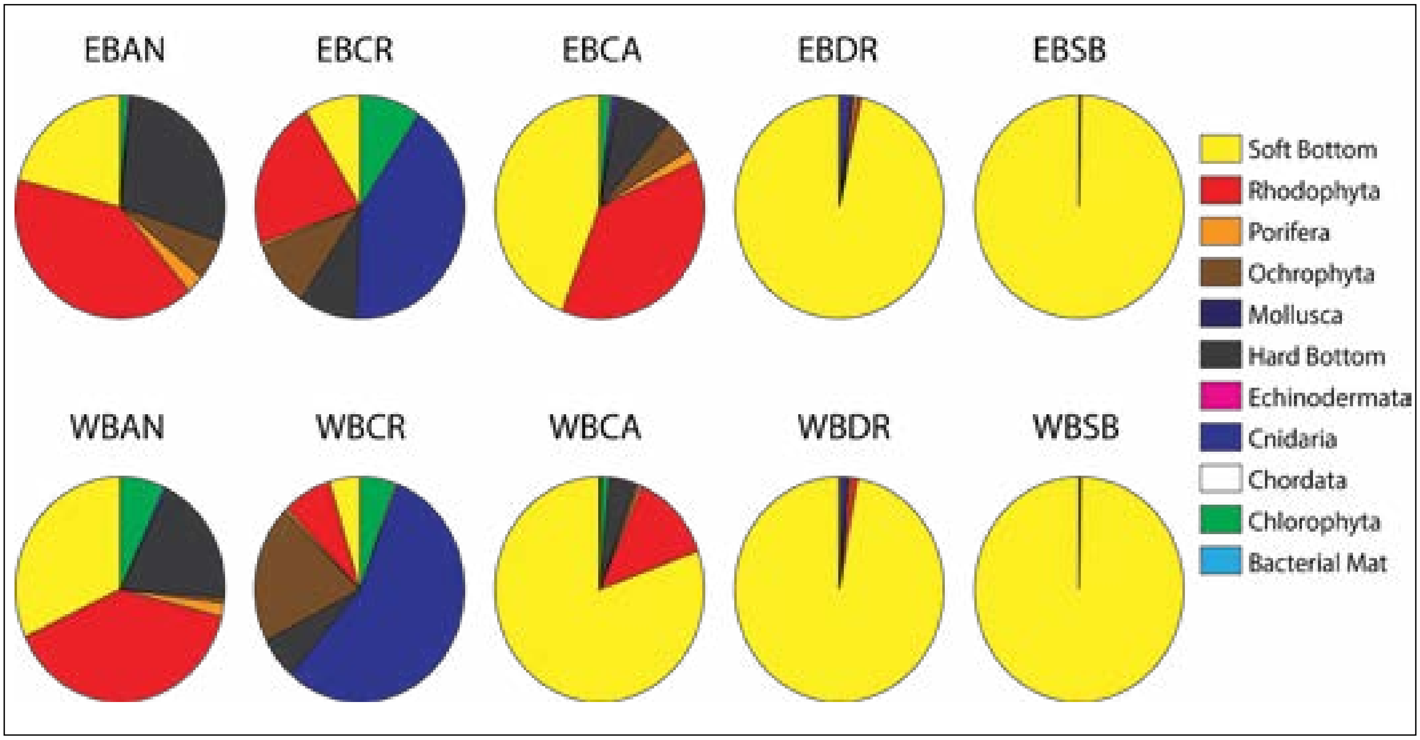


Figure 5.6. Mean percent cover of benthic taxa within strata.

algal reef and soft bottom, and coralline algal reef and deep reef. All other differences were attributable to the percentage cover of soft bottom.

Within Cnidaria, family level identifications were used to further discriminate among strata (Figure 5.9). While community composition of coral reef habitats were similar on East and West Banks, low cnidarian density off the coral caps contributed to disparate assemblages between banks. The family Merulinidae (star corals of the genus *Orbicella*) dominated on the coral cap, accounting for >30% of the observed cnidaria on both banks. Montastraeidae, Mussidae, Astrocoeniidae, Poritidae, Pocilloporidae, and Siderasteridae were all also consistently present on both EB and WB in relatively lower abundance. In contrast, Antipathidae were more abundant in soft bottom habitats. Algal nodule habitats on the EB include multiple occurrences of Astrocoeniidae, entirely comprised of *Stephanocoenia intersepta*, where as this family was absent from the WBAN. Also noteworthy, Antipathidae were more numerous on coralline algal reefs and deep reef habitats at the EB than in similar habitats on the WB. Finally, Aphanipathidae was more abundant on deep reefs and soft bottom habitats in the WB versus the EB.

Mesophotic Communities

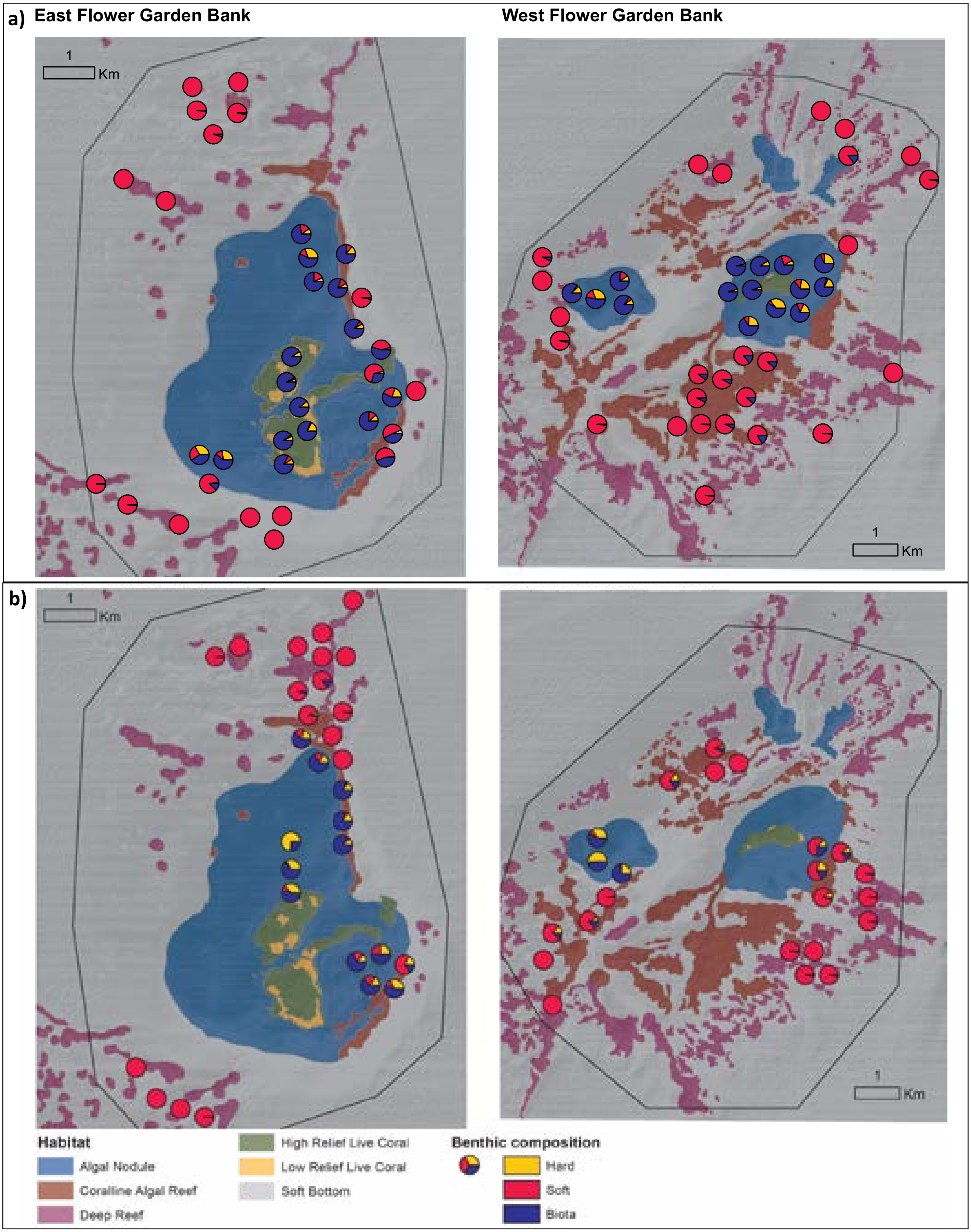


Figure 5.7. Benthic composition (% bare hardbottom, bare soft bottom and biota) for all ROV surveys in a) 2010, b) 2011, and c) 2012 (adjacent page).

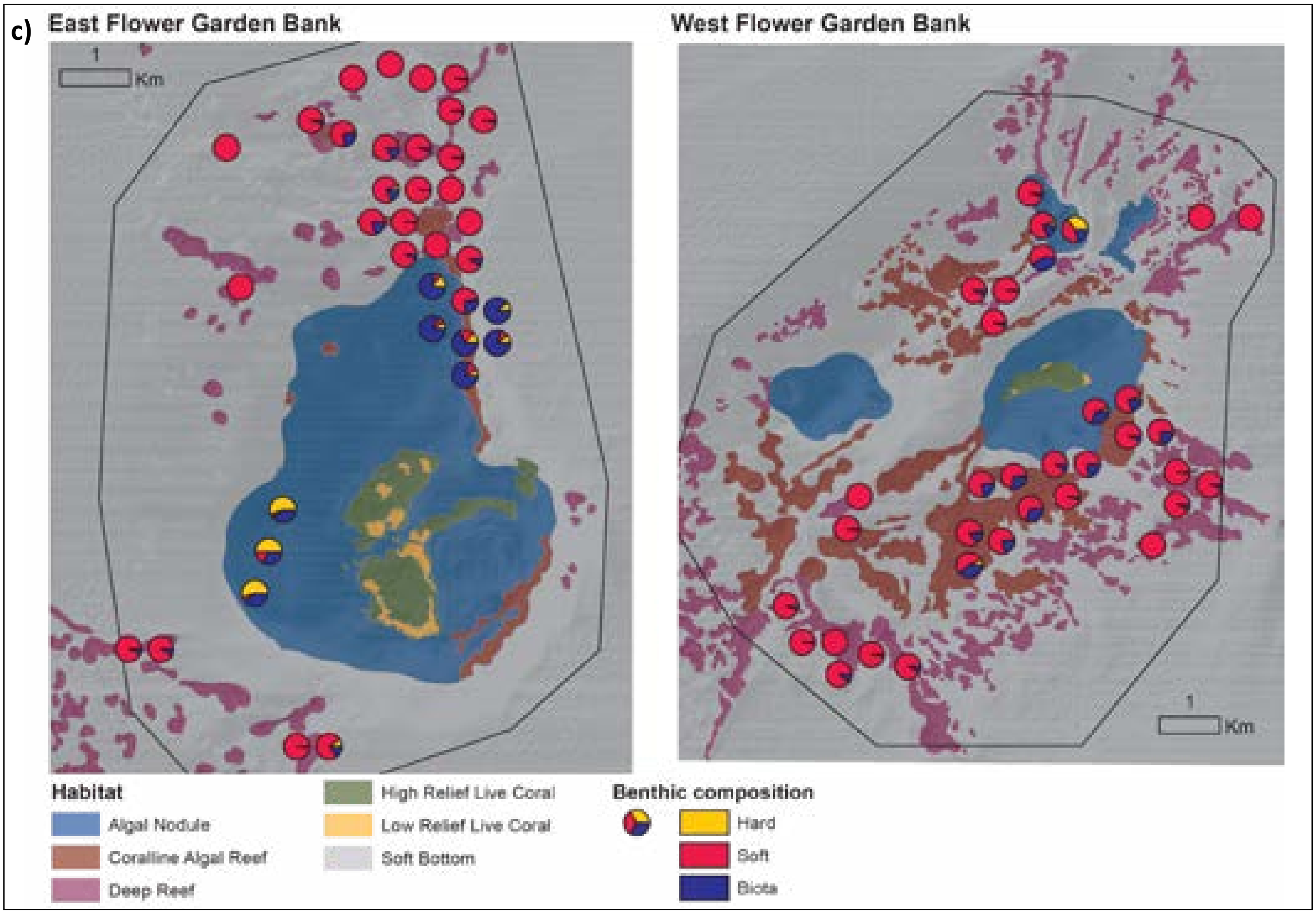


Figure 5.7. Continued from adjacent page. Benthic composition (% bare hardbottom, bare soft bottom and biota) for all ROV surveys in c) 2012.

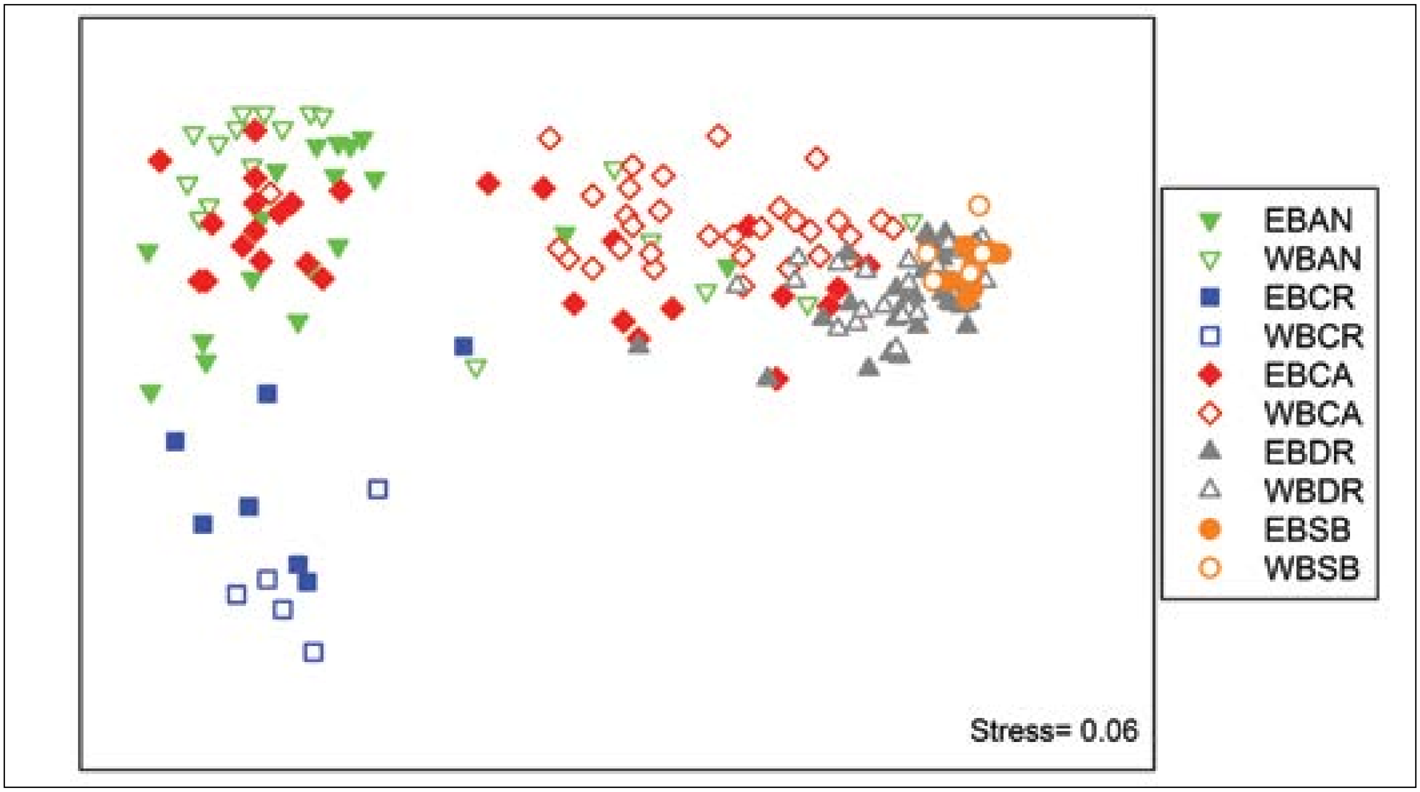


Figure 5.8. nMDS plot based on square root transformed Bray-Curtis similarities from phyla and abiota bottom cover data.

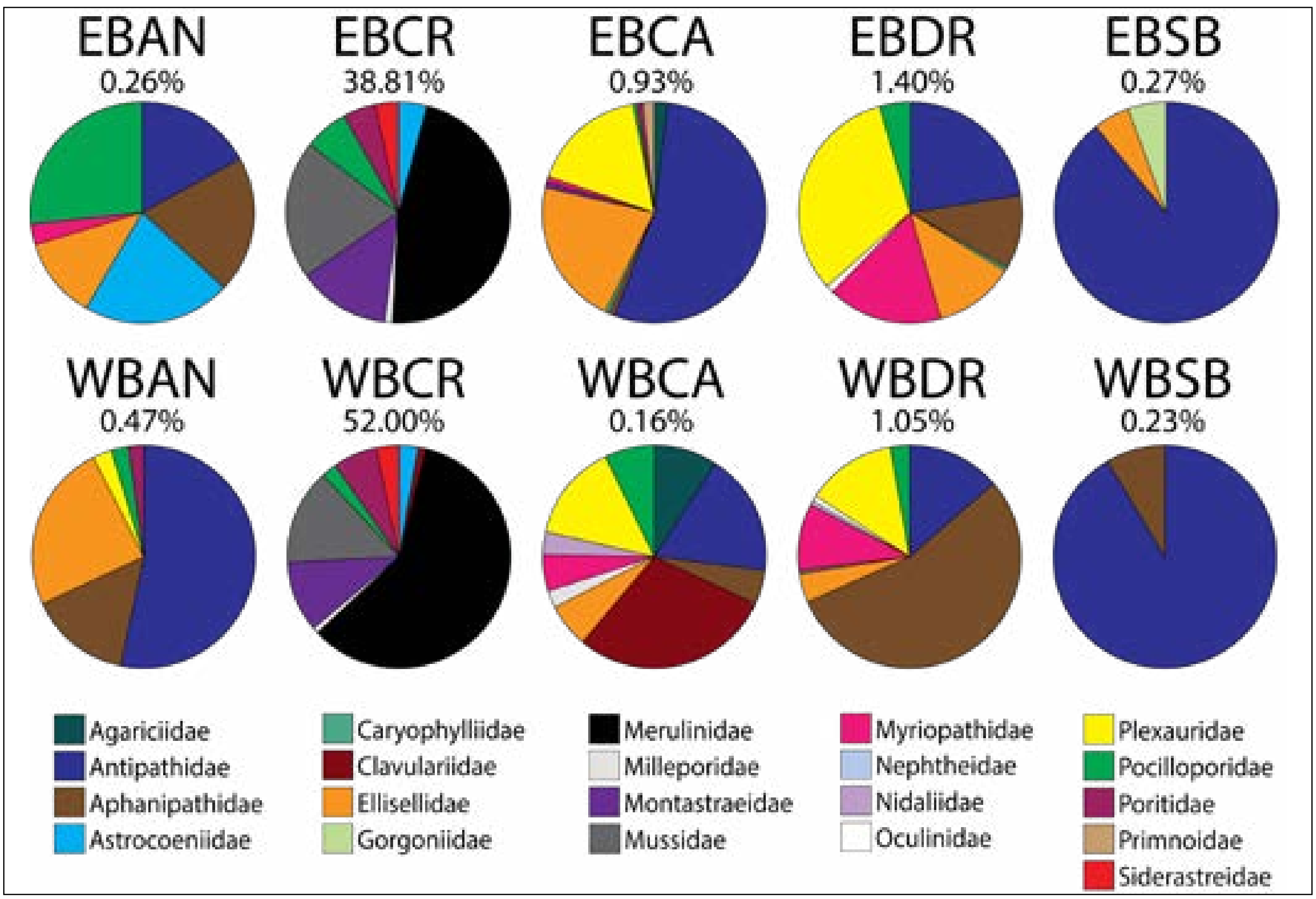


Figure 5.9. Relative abundance of each cnidarian family observed within each habitat type and bank.

Cnidarian family level analysis with nMDS (Figure 5.10) and ANOSIM indicated that both bank and habitat were significant ($p<0.01$, $R=0.118$; $p<0.01$, $R=0.288$, respectively). Although a significant p-value was observed for differences among EB and WB, the relatively low R-value indicates this result is probably not meaningful to the community structure observed. The tight clustering of the coral reef transects indicate strong similarity in community structure. Pairwise comparisons revealed significant differences between all habitats except algal nodule and soft bottom ($p >0.3$, $R = 0.011$), coralline algal reef and soft bottom ($p > 0.05$, $R = 0.09$), and algal nodule and coralline algal reef ($p > 0.05$, $R = 0.064$). The relative paucity of cnidaria in soft bottom habitats resulted in a number of transects with no cnidarian individuals. Such lack of data present difficulties when using the Bray-Curtis similarity index, and contributes to the inability of this analytical approach to differentiate a bottom type that is ecologically distinct. The differences that were observed between coral reef and other habitat types were primarily driven by the percentage cover of corals in the families Merulinidae, Montastraeidae, and Mussidae, with corals in the order Antipatharia the significant factor differentiating deep reef and soft bottom, algal nodule and deep reef, and coralline algal reef and deep reef. Spatial distribution of cnidarian percent cover based on 2010 ROV surveys is displayed in Figure 5.11.

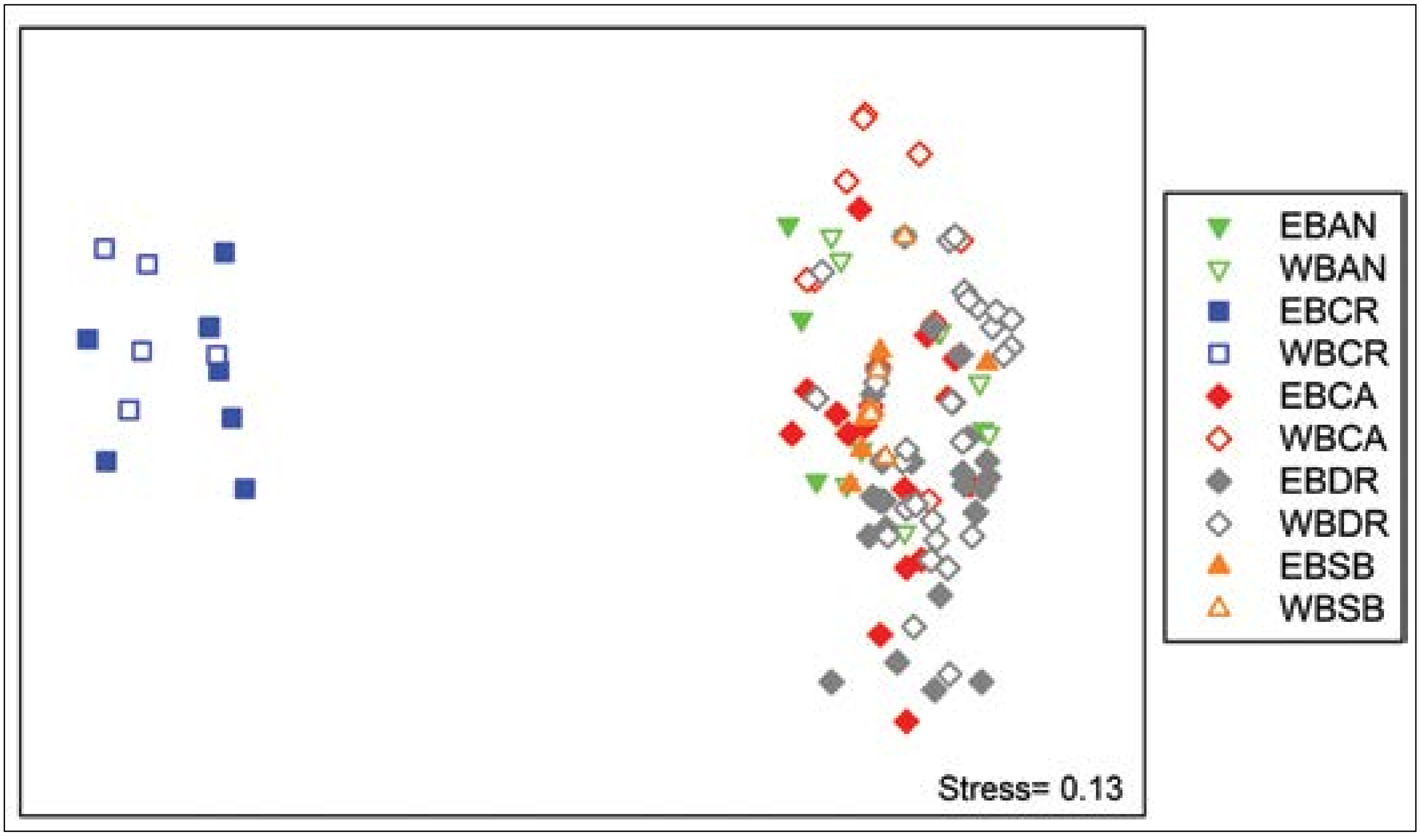


Figure 5.10. nMDS plot based on square root transformed Bray-Curtis similarities from relative family abundance within the phylum Cnidaria. Both bank and habitat type were significant factors.

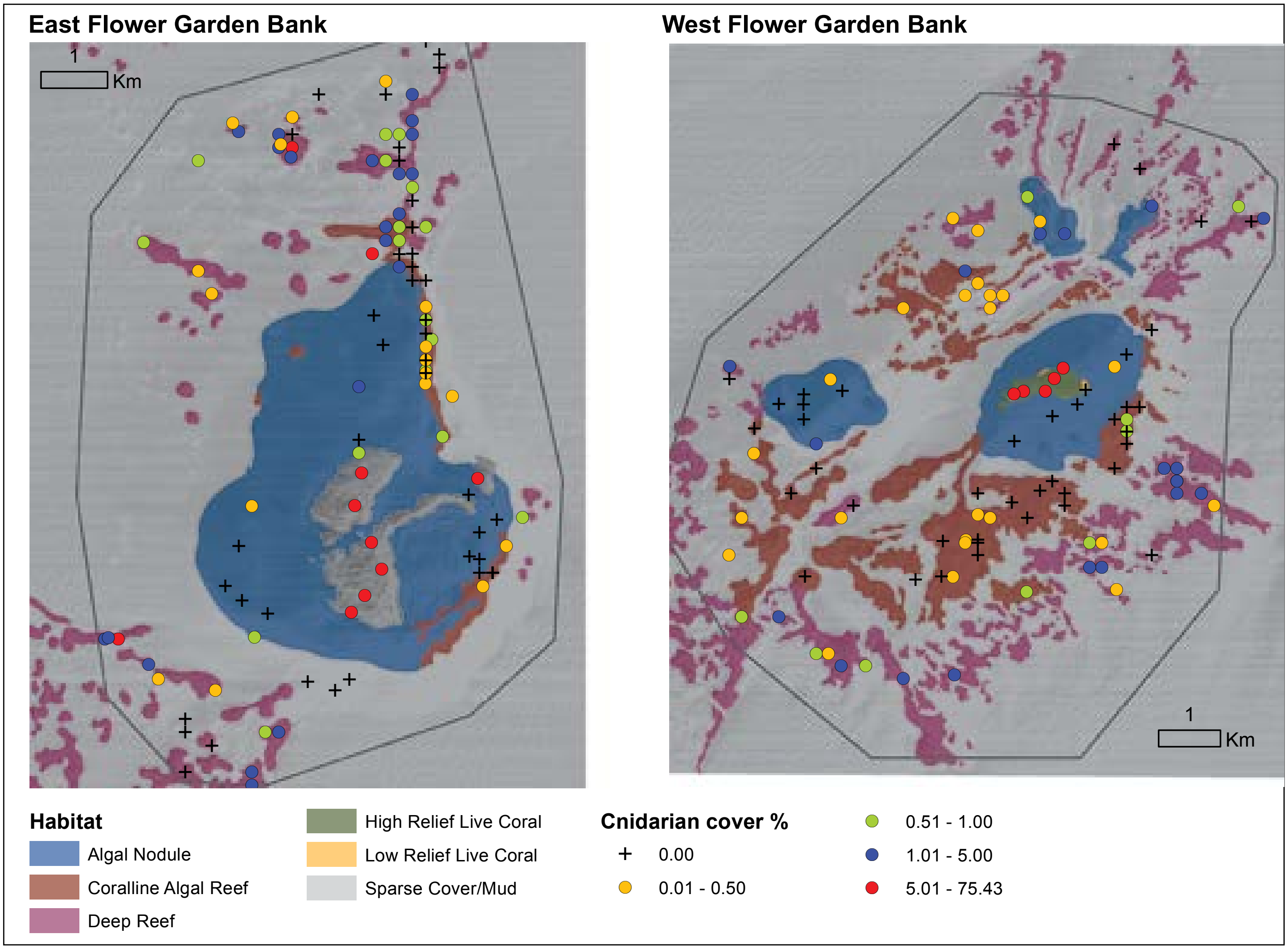


Figure 5.11. Mapped cnidarian density for all ROV surveys, 2010.

High relative density of scleractinians in the coral reef habitats permitted species level community comparisons among transects on the shallow reefs of EB and WB. Similar to results obtained in diver surveys, the coral communities were dominated by *Montastraea cavernosa*, *Orbicella faveolata* and *Orbicella franksi* (previously known as *Montastraea faveolata* and *M. franksi*), and *Porites astreoides* (Figure 5.12). *Stephanocoenia intersepta* was marginally more abundant at EB than WB. Nonparametric analysis of scleractinian community structure indicated no differences between the two banks ($p > 0.8$, $R = 0.121$).

Density Analysis

Density for all cnidarian families and for all hard coral species (scleractinia plus *Millepora alcicornis*; Figure 5.13) exhibited similar community composition among habitat types as concluded by the point count analyses.

Similar to the point count results, Merulinidae, Pocilloporidae, Montastraeidae, and Mussidae were consistently the dominant families in the coral reef habitats. Cnidarian density in the coral reef habitats was >8 individual colonies/m^2; no soft corals were observed. Antipathidae, Aphanipathidae, and Ellisellidae were common in deep reef habitats, and in the soft bottom habitat on EB.

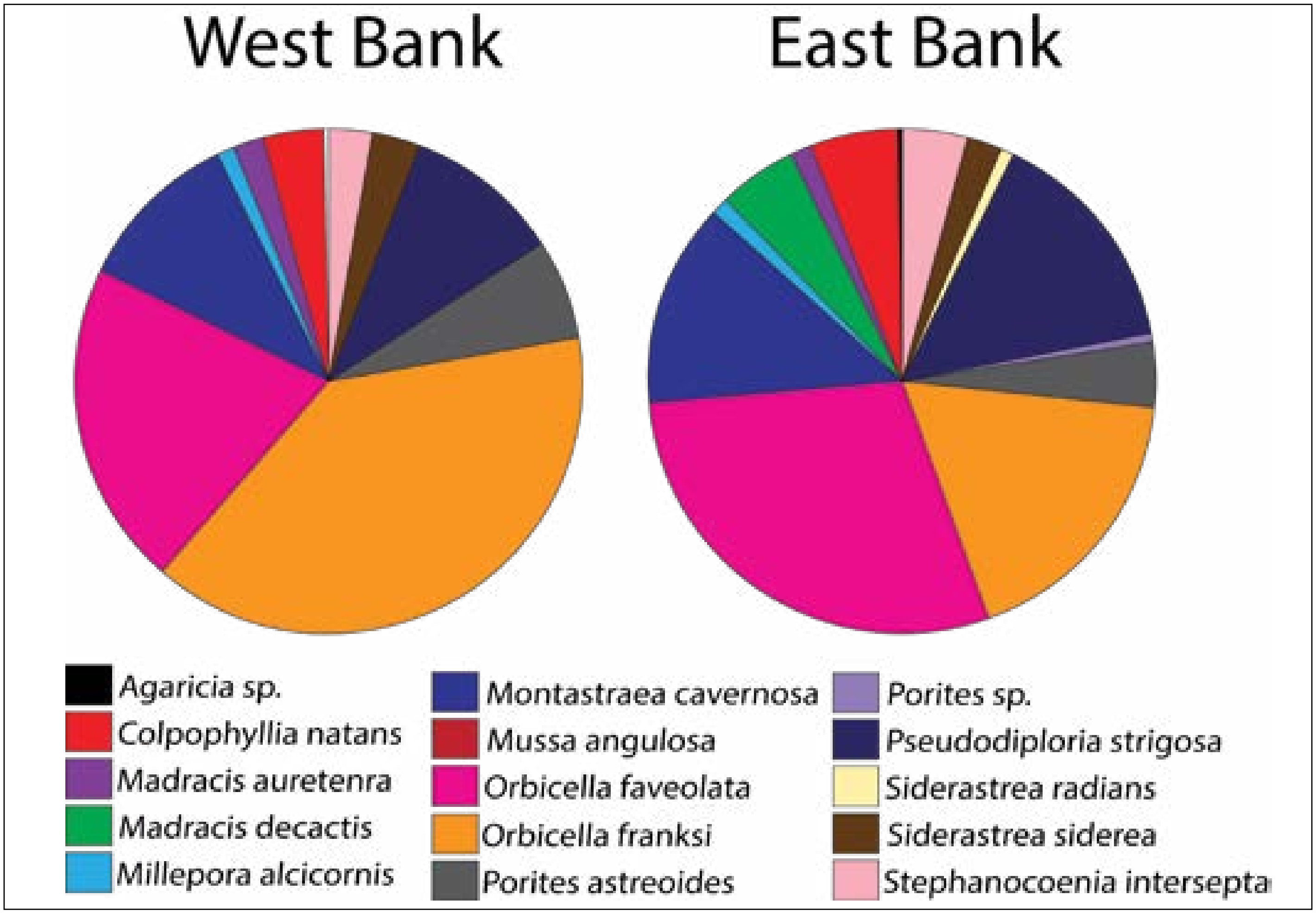


Figure 5.12. Relative abundance of scleractinian species in coral reef habitats of East and West Bank.

Cnidarian community composition data indicated strong differences among habitat types ($p < 0.01$, R = 0.466; Figure 5.13), and a slight difference between the two banks ($p < 0.01$, R = 0.097). Pairwise comparisons revealed significant differences between habitat types ($p < 0.05$), except coralline algal reef and soft bottom ($p > 0.4$). Differences between habitat types were driven primarily by the density of corals in the Merulinidae and Antipathidae families.

Scleractinian families such as Merulinidae (*Orbicella* spp.), Montastraeidae (*Montastraea* spp.), Mussidae (*Psuedodiploria* spp.) and Astrocoeniidae (*Stephanocoenia* spp.) were dominant or more abundant on coral reef than deeper habitats (Figure 5.14). Black corals from the family Antipathidae provided the most benthic cover on coralline algal reefs and were the second most abundant on deep reefs behind species from the family Aphanipathidae.

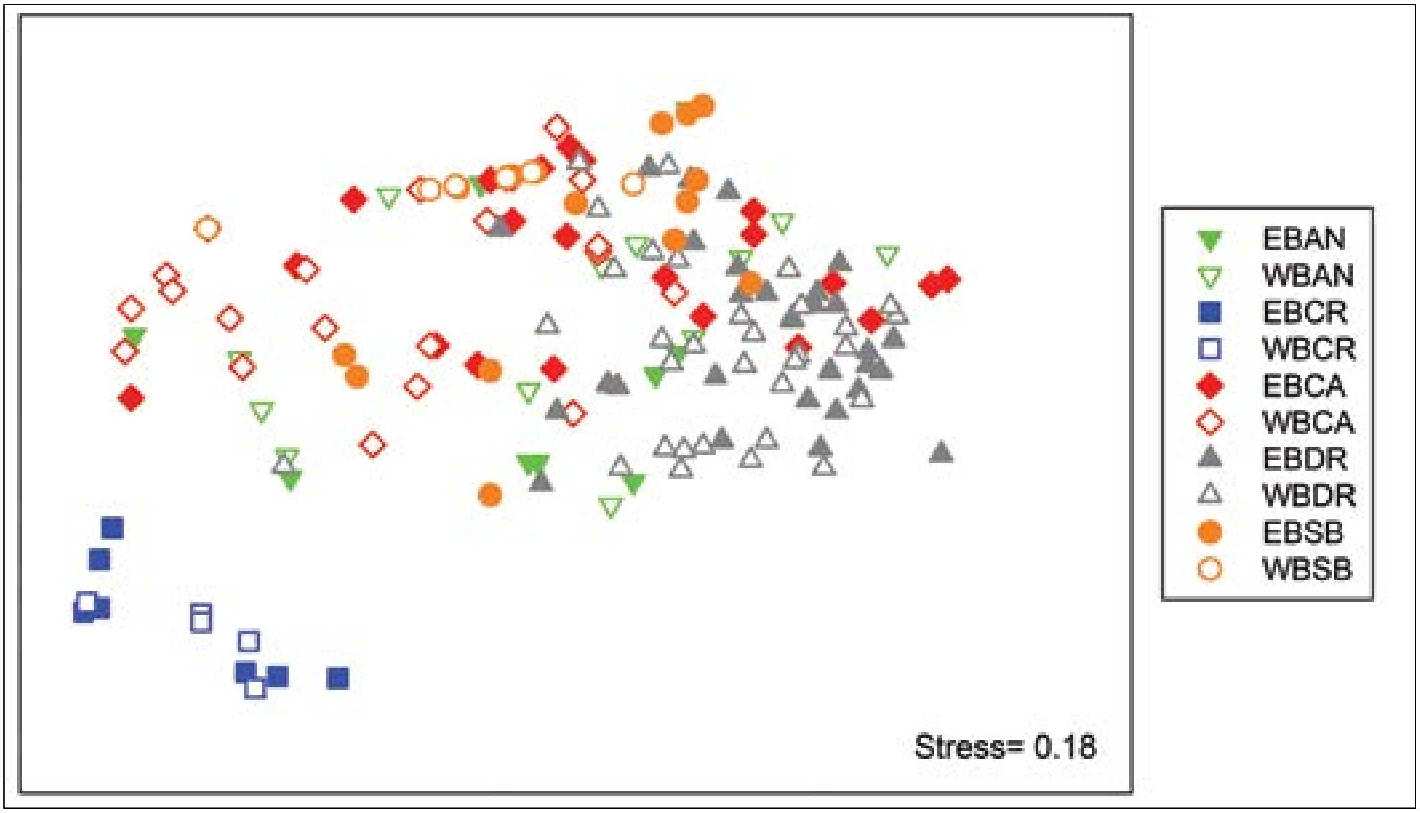


Figure 5.13. nMDS plot based on square root transformed Bray-Curtis similarities from cnidarian family density measures in each transect.

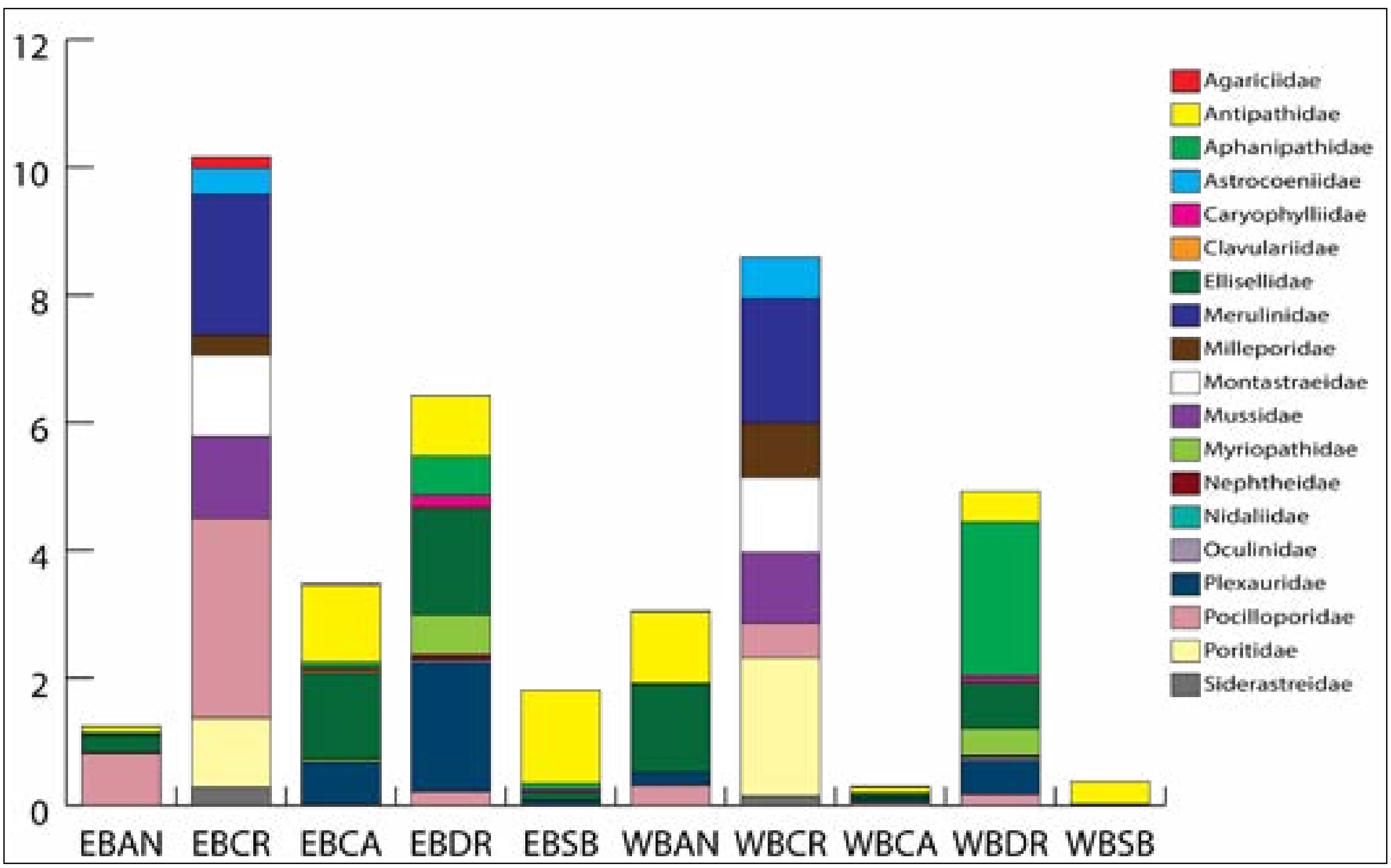


Figure 5.14. Family level identifications of cnidarians indicate strong differences among habitat types, but similarities among banks in each habitat type.

5.3.2. Spatial Distribution of Ecologically Important Benthic Cnidarians

Biota were overwhelmingly more abundant on coral reef transects than deeper hard bottom habitat transects. In most regions within the sanctuary, coralline algal and deep reefs were rarely continuous and were often segregated by various extents of mud or sand between the reef structures. As such, the higher proportion of soft substrate at coralline algal and deep reef habitats portrays patchiness at the scale of the map. Scleractinians were the dominant component of cnidarians on the shallow and upper mesophotic reef but were less abundant at depths greater than 50 m. Overall, cnidarian cover and density declined with depth (Figure 5.15). In terms of density (i.e., number of corals per square meter), scleractinians were most dominant and species rich in the coral reef habitat (Figure 5.16). Within each habitat type, scleractinian species densities were relatively similar between the East and West Banks. The patterns of abundance displayed consistent and widespread abundance of reef-building corals at depths between 18 and 50 m and patchy, upright growth of corals and sponges on deeper habitats (Figure 5.17). Spatially, cnidarians were present on all hardbottom habitats in the sanctuary, but density tended to be greater on the coral and deep reef habitats (Figure 5.18). On habitats deeper than 50 m, colonies from the families Antipathidae, Aphanipathidae and Ellisellidae were dominant.

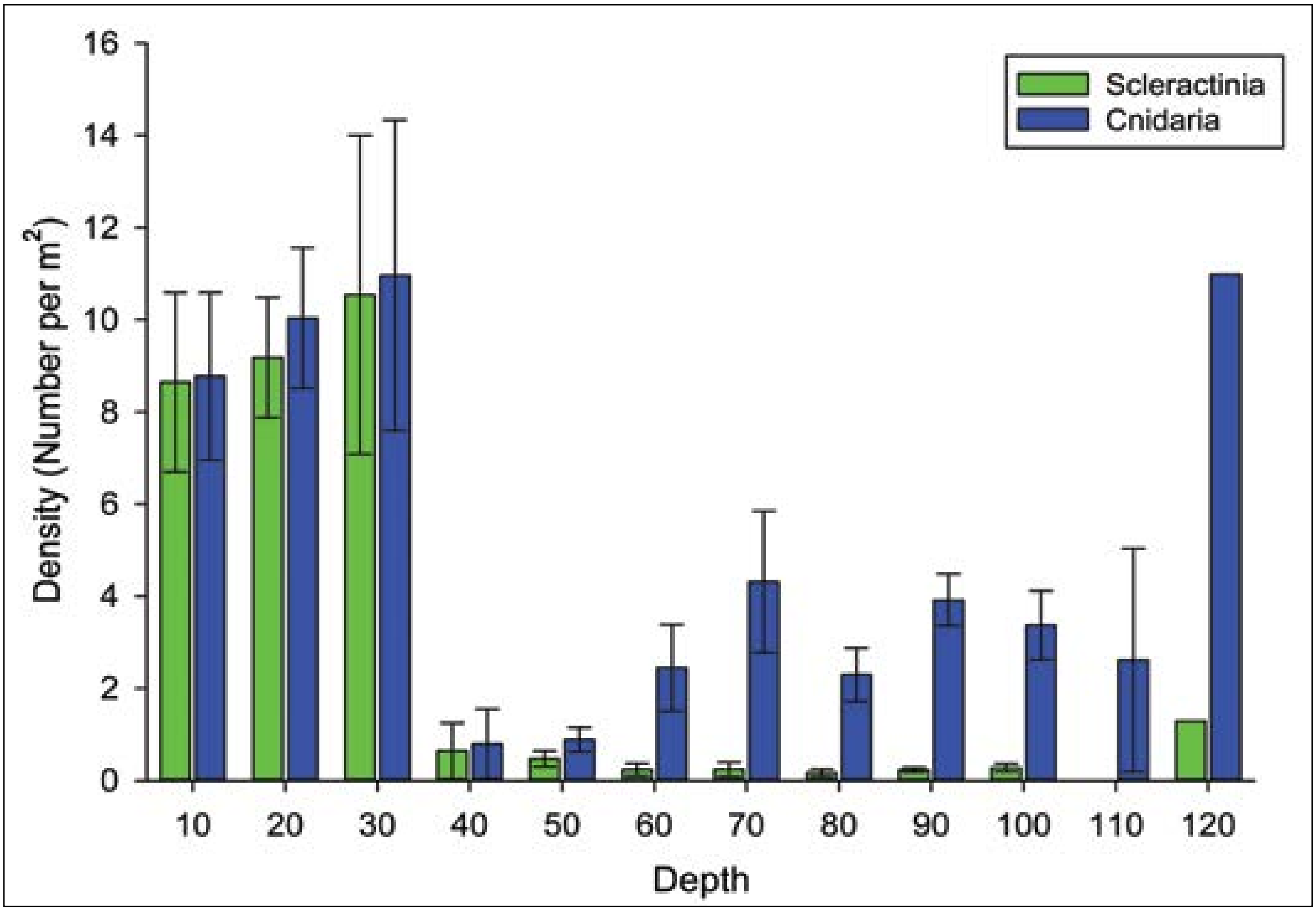


Figure 5.15. Density of cnidarian and scleractinian with depth along ROV transects, 2010.

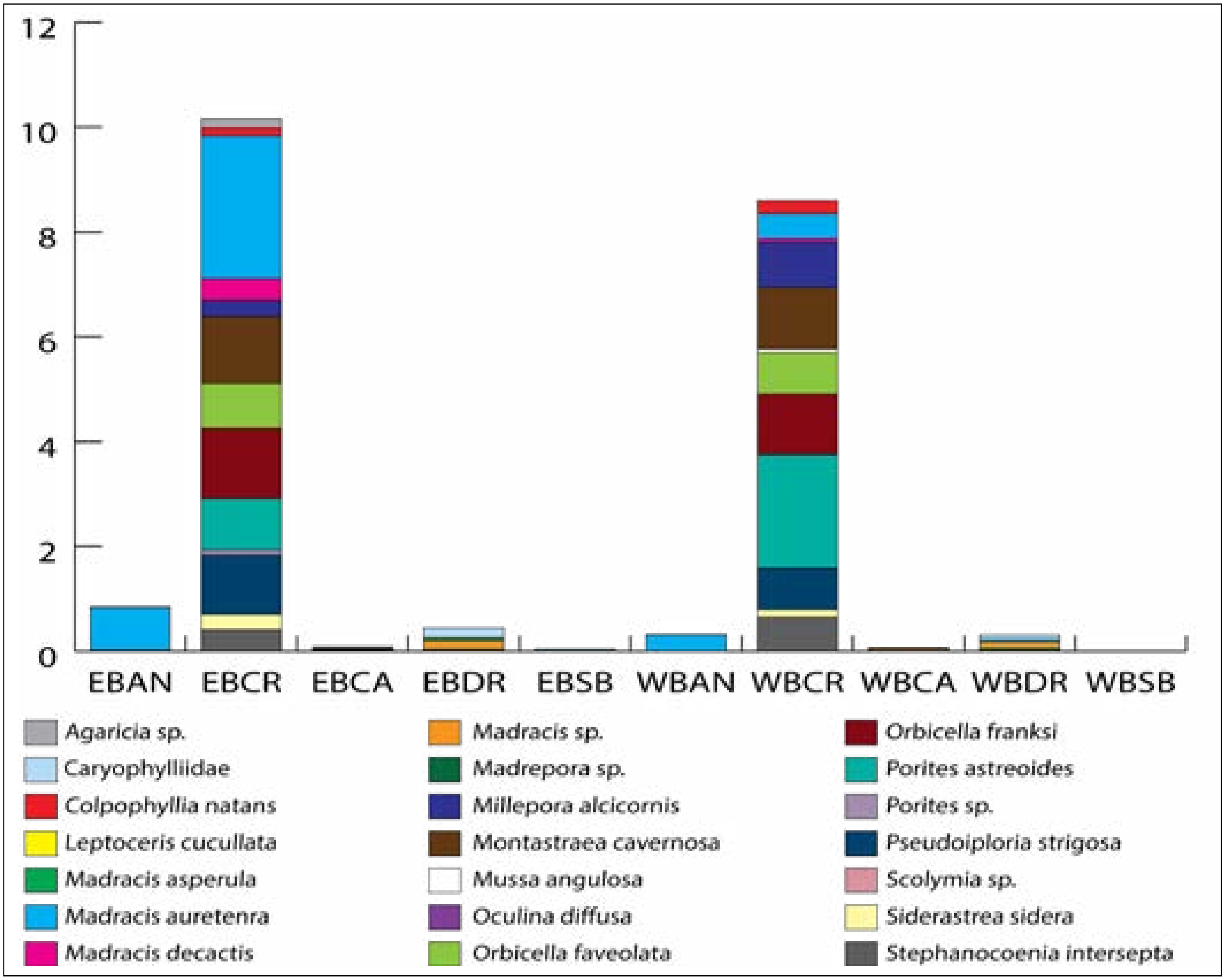


Figure 5.16. Mean density (#/m²) for scleractinian coral species.

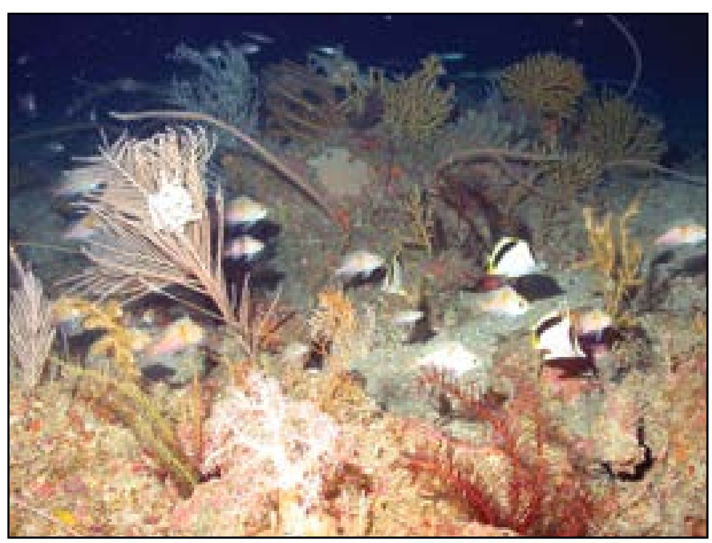


Figure 5.17. Example of coral-dominant reefs at shallower depths (left) and decreased cover dominance at deeper depths (right). Photos: NOAA NOS/ONMS/FGBNMS and UNCW-UVP

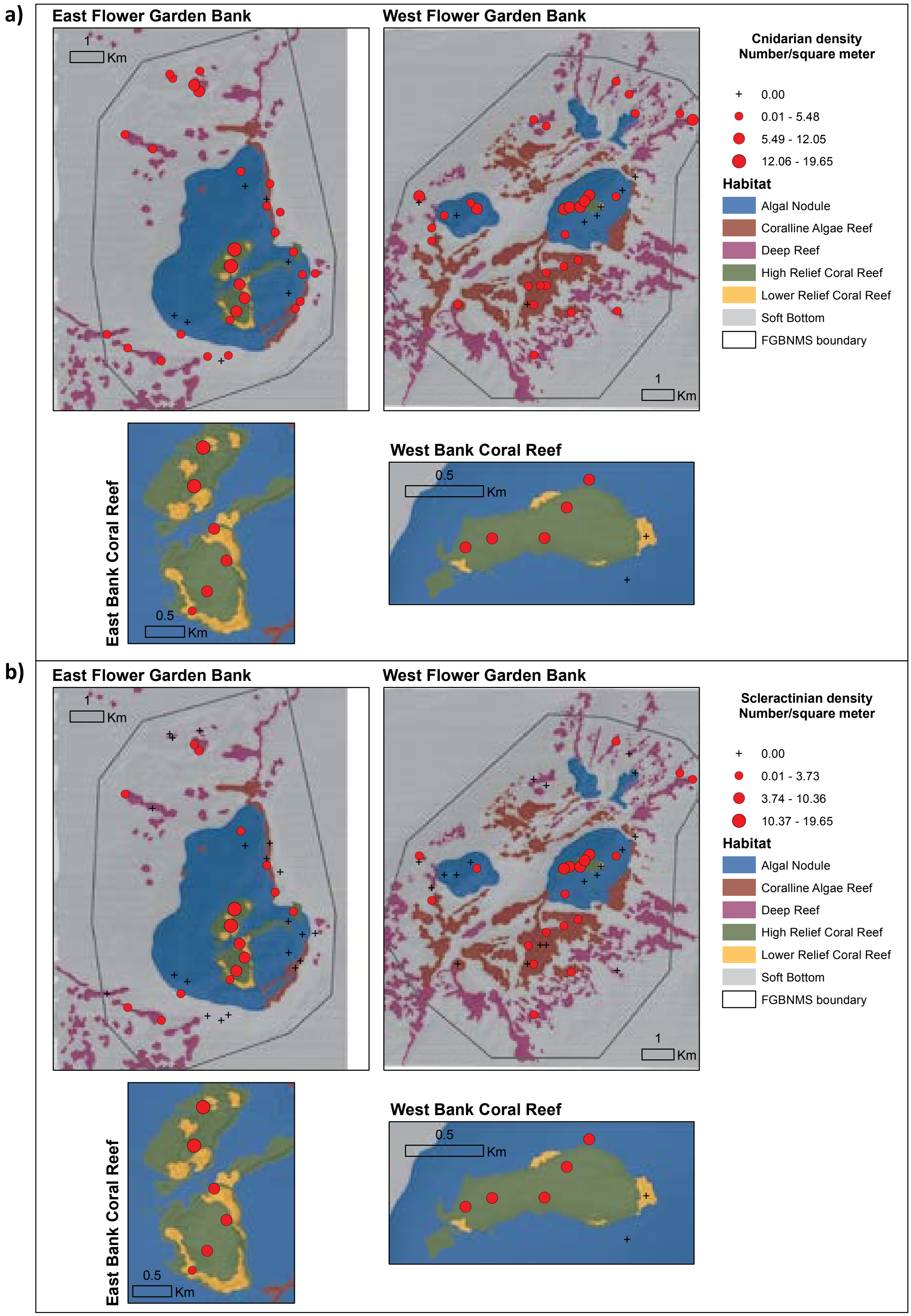


Figure 5.18. a) Cnidarian density at ROV stations on East and West Banks, 2010; b) scleractinian density at ROV stations on East and West Banks, 2010.

Mesophotic Communities

Black Corals

Black corals (order Antipatharia in the class Anthozoa of the phylum Cnidaria) are important long-lived, habitat forming, sessile benthic suspension feeders (Grigg, 1965; Lewis, 1978; Parrish et al., 2002). Dense populations of these corals have been found in the tropical western Atlantic.

Family Antipathidae

In the Gulf of Mexico, coral taxa belonging to the family Antipathidae are considered non-reef building species but can provide habitat for numerous other species (Figure 5.19). There are at least 20 species documented in the Gulf, with at least half of these reported from the Flower Garden Banks (Brooke and Schroeder, 2007). Antipatharians are usually found at depths greater than 20 m, to a maximum depth of nearly 3,000 m (Etnoyer and Morgan, 2005).

Figure 5.19. Type of Antipathidae species found in FGBNMS. Photo: NOAA NOS/ONMS/ FGBNMS UNCW-UVP

Species belonging to the family Antipathidae were found on 55 of 116 EB sites (47%) and 43 of 99 sites on WB (43%). Of the 552 observations, 124 were found on coralline algal reef, 230 on deep reef, 117 on soft bottom and 81 on algal nodule habitats. Depths of observations ranged from 48.2-115.9 m. Figure 5.20 displays the location of observations. Percent cover was low on all habitats: 0-4.5% on coralline algal reefs, 0-1.6% on deep reefs, 0-2.4% on softbottom and 0-2.6% on algal nodule.

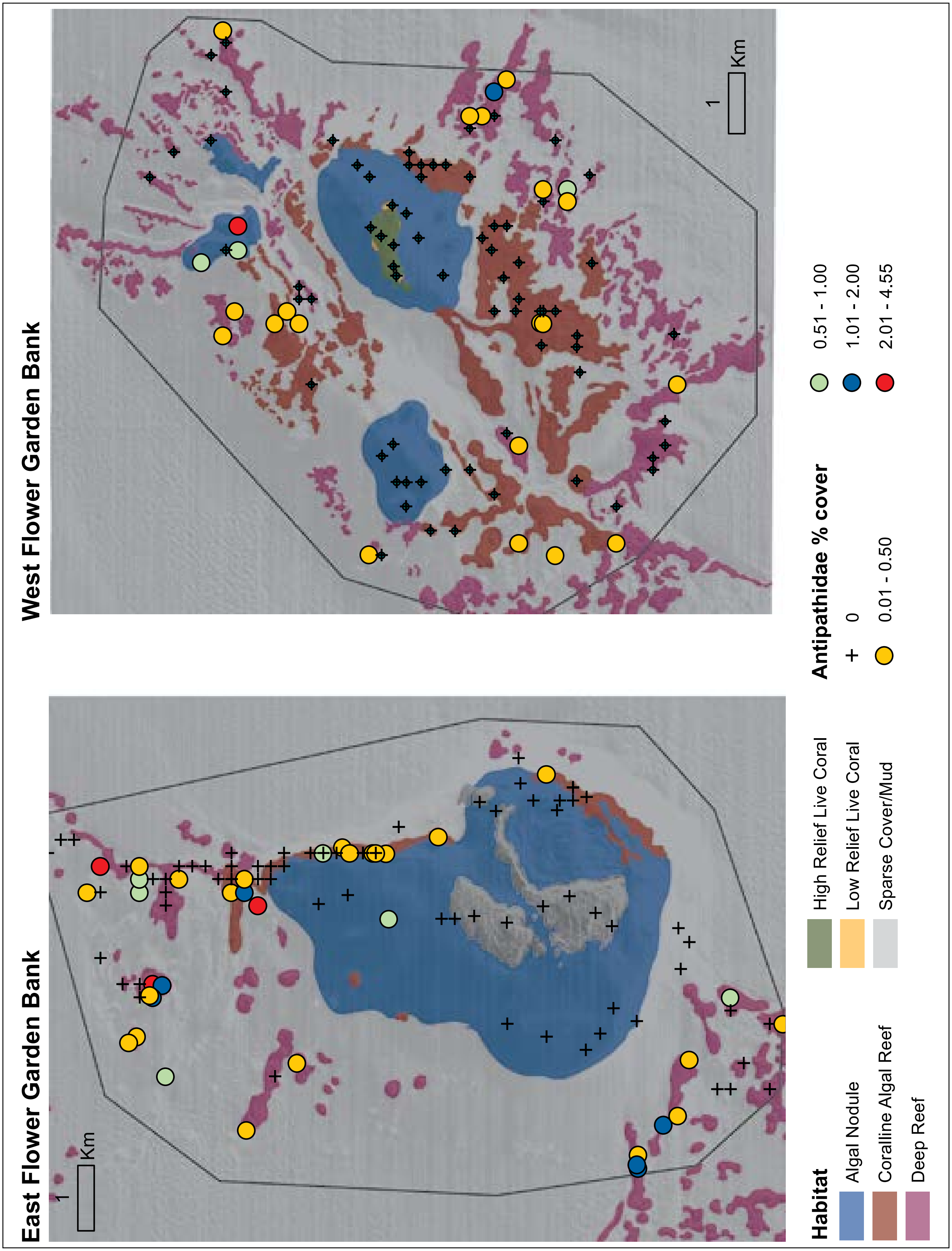


Figure 5.20. Spatial distribution of black coral (Family Antipathidae) observations along ROV transects, 2010.

Family Aphanipathidae

This family is considered non-reef building and ecosystem engineers that provide habitat for a variety of other organisms (Figure 5.21). There are about nine species of this family found in the Gulf, all found at depths from 51-500 m (Opresko, 2009).

Figure 5.21. Type of Aphanipathidae species found in FGBNMS (organisms in green). Photo: NOAA NOS/ONMS/FGBNMS UNCW-UVP

Members of this family were less frequently observed than those from the family Antipathidae. One hundred and fifty-one observations were made on EB, three on algal nodule, nine on coralline algal reef, 128 on deep reef, and 11 on soft bottom. Three hundred and sixty-seven sightings were made on WB, 363 on deep reef, one on algal nodule, one on soft bottom, and two on coralline algal reef. Depths of sightings were mostly deep but ranged from 55-122 m. Percent cover was also low for this family. Percent cover ranged from 0-2.2% on deep reefs and 0-0.26% on coralline algal reefs. Figure 5.22 shows the location of observations of Aphanipathidae seem to be more common on the WB, with high densities in deep reef habitats.

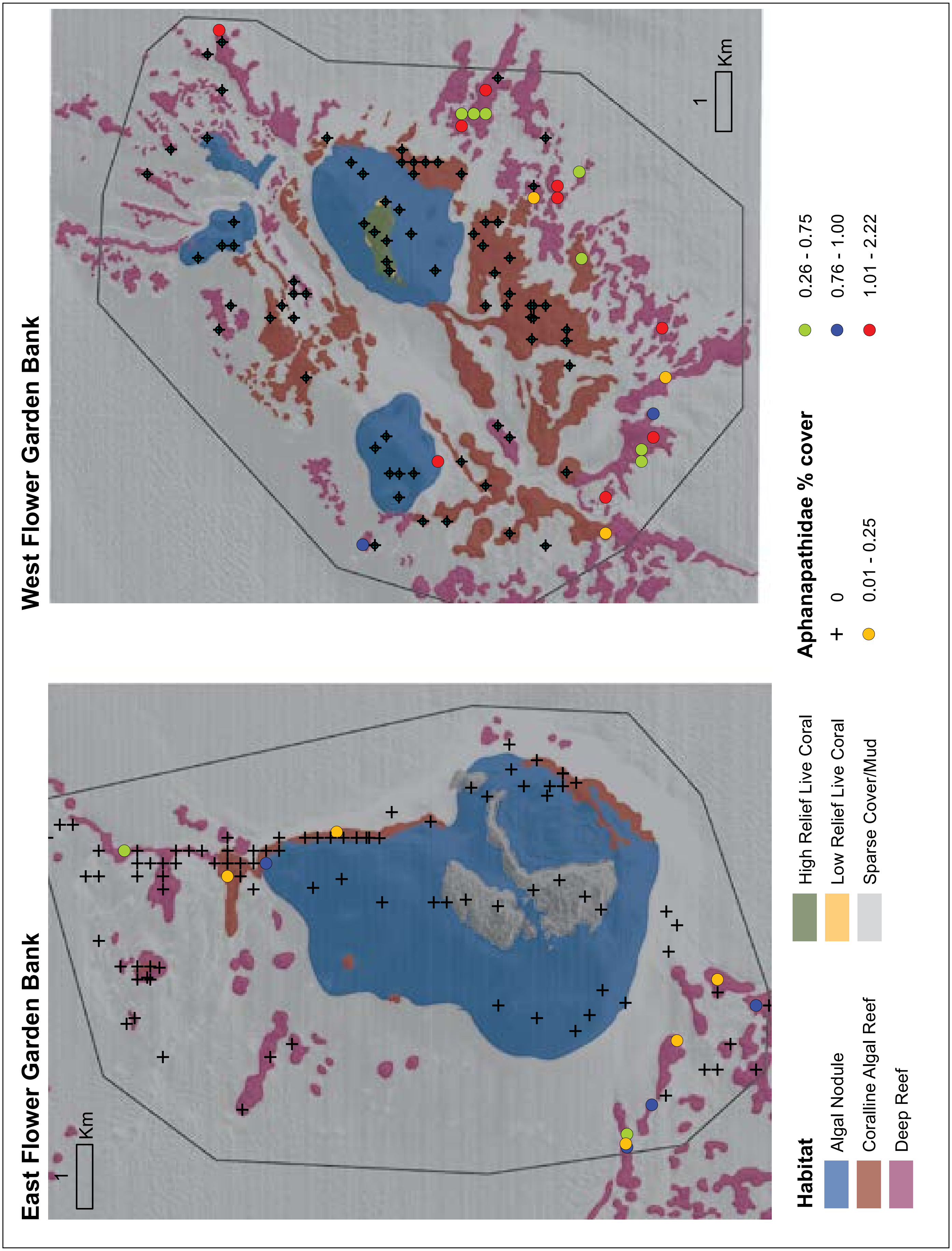


Figure 5.22. Spatial distribution of black coral (Family Aphanipathidae) observations along ROV transects, 2010.

Gorgonia

<u>Family Ellisellidae</u>

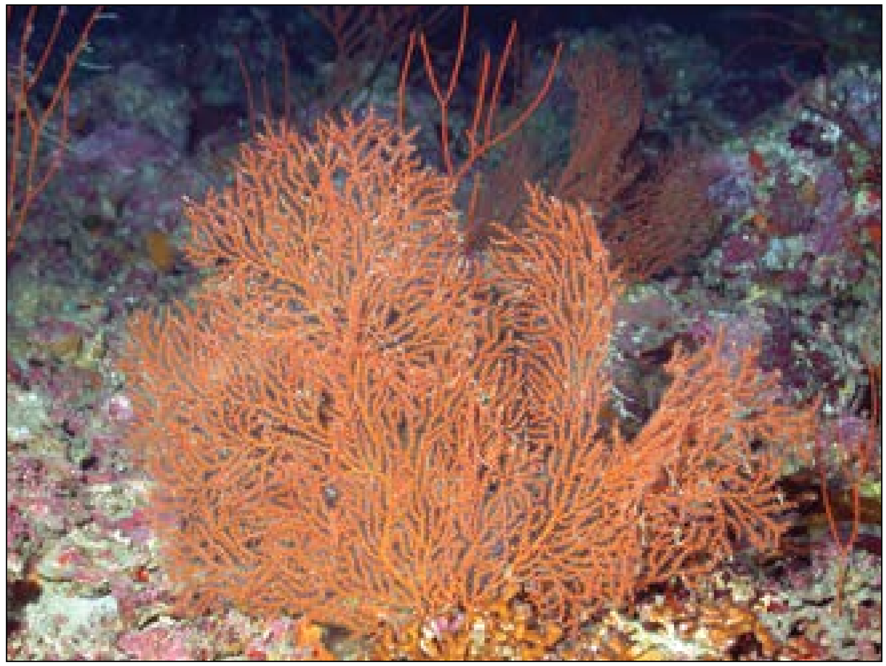


Figure 5.23. Type of Ellisellidae species found in FGBNMS. Photo: NOAA NOS/ONMS/ FGBNMS and UNCW-UVP

Gorgonians are an important component of the shelf edge (50-120 m) reefs and banks of the northwestern Gulf of Mexico. There are numerous species of octocorals in the deep waters of the Gulf of Mexico (many of which are still unidentified), the majority of which belong to the family Plexauridae. There are eight species of gorgonians in the family Ellisellidae found in the northern Gulf of Mexico with depth ranges from 55-390 m (Brooke and Schroeder, 2007; Figure 5.23). Most species are large fan-like gorgonians that provide refuge for a variety of organisms (Etnoyer et al., 2011), while others are more whip-like.

Many individuals of this family were observed by ROV, with a total of 527 observations, (Figure 5.24). Within the EB, a total of 325 were observed, with 26 in algal nodule, 108 in coralline algal reefs, 179 in deep reefs, and 12 in soft bottom habitats. A total of 202 were observed in the West Bank, with 74 in algal nodule, 15 in coralline algal reefs, and 113 in deep reef habitats. Depths of observation ranged from 48-122 m. Ellisellidae were more dense on the EB, especially in deep reef habitats.

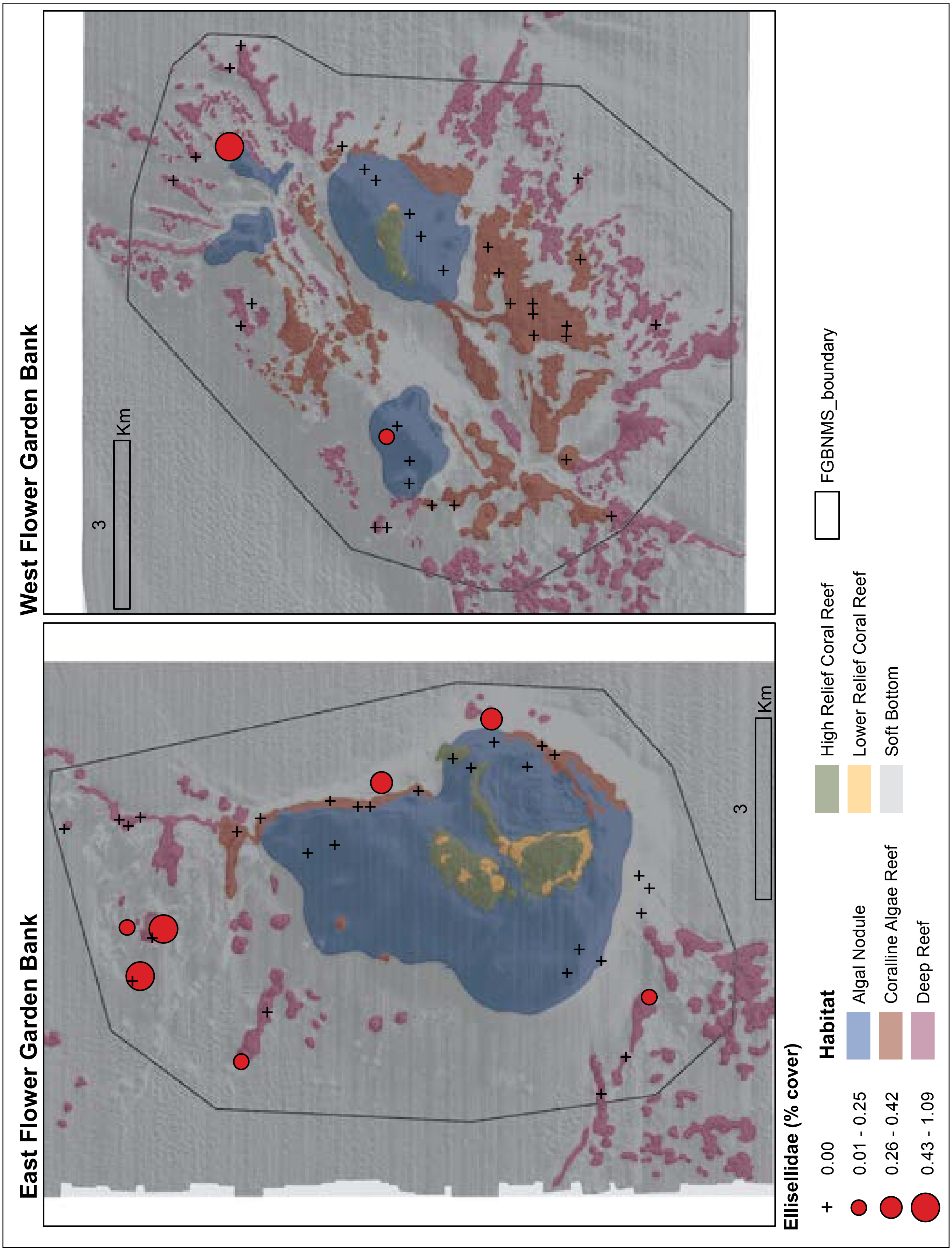


Figure 5.24. Spatial distribution of deepwater gorgonian (Family Ellisellidae) observations along ROV transects, 2010.

5.3.3. Fish Assessment

Comparison of ROV and Diver

Fish CPUE and species richness were compared at 12 sites where surveys were conducted by both divers and ROV (Table 5.3). Diver survey effort was unchanging at 100 m², ROV survey effort ranged from 234-454 m². CPUE ranged from 60-2489/100 m² (mean = 490.1) on diver surveys while ROV CPUE ranged from 59-237, averaging 125.9 individuals/100 m².

Table 5.3. Effort, catch per unit effort (CPUE) and species richness in surveys conducted by divers and ROV.

			Fish CPUE		Fish Species Richness	
	n	Effort (m²)	Mean	SE	Mean	SE
Diver	12	100	490.1	207.6	23.4	1.8
ROV	12	234-448	125.9	18.1	21.1	1.7

Mean CPUE estimated by divers was three times greater than that observed by ROV but high variability on the diver surveys contributed to a lack of statistical significance between the ROV surveys. Species richness ranges were similar between the two methods (diver surveys: 11-33 species/transect, mean = 23.4; ROV surveys: 10-29 species/ transect, mean = 21.1) and were not statistically significant. Despite this relationship, the similarity index revealed that the ROV and diver surveys are capturing different components of the communities surveyed (Figure 5.25).

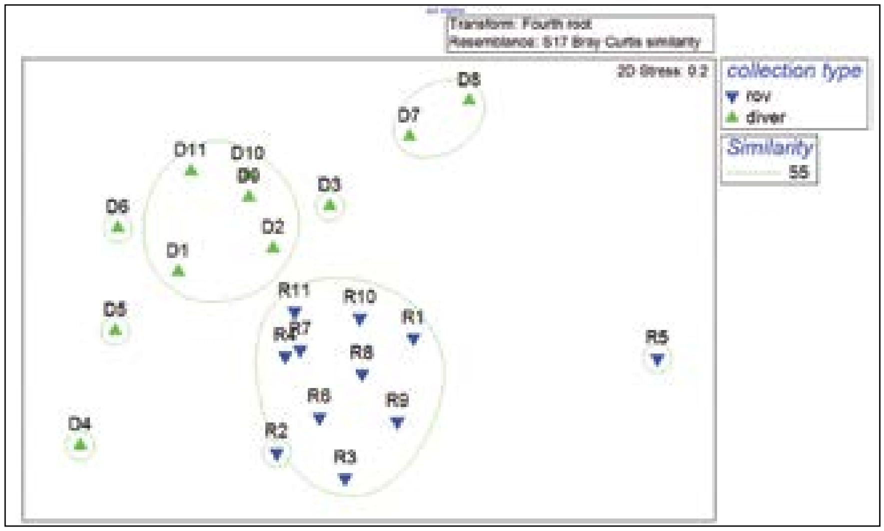


Figure 5.25. nMDS showing site similarity in fish assemblages composition as surveyed by diver and ROV.

Fish Assemblages

Of the 201 surveyed sites, seven did not contain fish. These surveys were all on softbottom habitats and all but one was on EB. Overall, 77 species were identified to species and another 14 fish identified to genera. Fish assemblages were significantly different (ANOSIM R = 0.405, p < 0.01) across strata (Table 5.4). While statistically significant, assemblage comparisons by habitat type between bank, e.g., EB algal nodule versus WB algal nodule, were not distinct. Overall fish assemblages on deep reef and coralline algal habitats were separated from those on algal nodule and soft bottom habitat types (Figure 5.26). Fish assemblages were indistinct for coralline algal and deep reef habitats on East and West Banks, respectively, however some distinction was observed between coralline algal habitat on EB and deep reef on WB (Table 5.4).

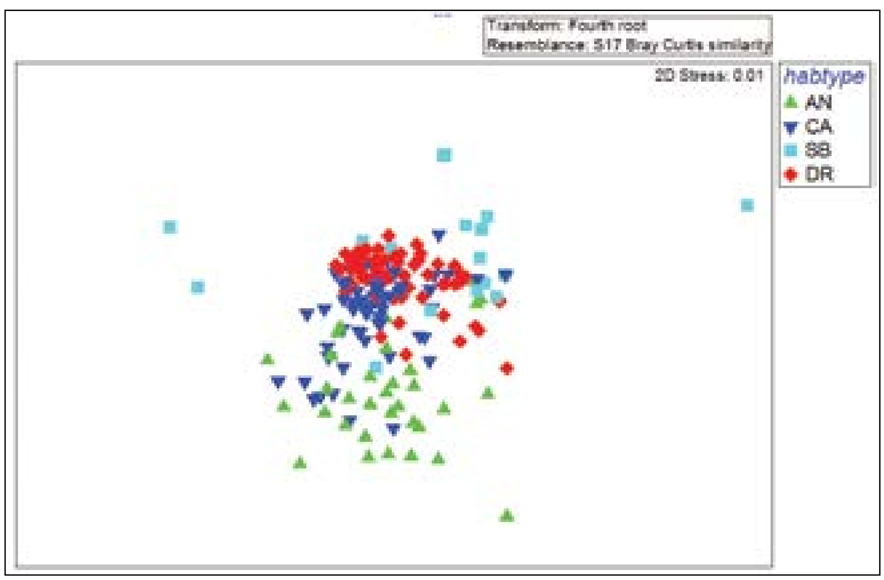


Figure 5.26. nMDS plot for fish community similarity for all deep water benthic habitat types.

Table 5.4. ANOSIM R values measuring fish assemblage similarities between surveys grouped by strata. R values in bold are statistically significant (p=<0.01). Dark gray background=well separated, light gray=some overlap, white=barely separable. Habitat types: E=East Bank, W=West Bank followed by AN=algal nodule, CA=coralline algal reef, DR=deep reef, SB=soft bottom.

Habitat type	EAN	ECA	EDR	ESB	WAN	WCA	WDR
ECA	0.381						
EDR	0.572	0.17					
ESB	0.269	0.617	0.573				
WAN	0.013	0.323	0.626	0.396			
WCA	0.669	0.238	0.11	0.75	0.681		
WDR	0.717	0.302	0.039	0.715	0.772	0.151	
WSB	0.389	0.885	0.873	0.153	0.555	0.93	0.963

Community Metrics

Overall we observed over 58,260 fish on East and West Banks combined. During 2012, fish density was significantly greater (p<0.0001) on coralline algal reefs on EB and deep reefs on EB and WB. This was a result of large populations (nearly 10 times that observed in prior years) of *Prontogrammus martinicensis* (roughtongue bass) and *Choranthias tenuis* (threadnose bass; Figure 5.27). Despite this observed pattern, fish density was comparable during the study. Biomass was not statistically significant among strata by year. Fish density was greatest on coralline algal and deep reef habitats (Figure 5.28). Mean fish density was not significantly different for coralline algae reefs (approximately 130 individuals/100 m^2) on either bank and deep reefs on WB (162 individuals /100 m^2). Mean density was significantly lower (p < 0.0001) on EB deep reefs than density observed on WB and coralline algal reefs on WB. Coralline algal and deep reef density was significantly greater (p < 0.001) than density observed on algal nodule and soft bottom habitats. Algal nodule mean density was not statistically different by bank, but was significantly greater (p < 0.001) than soft bottom habitats on either bank .

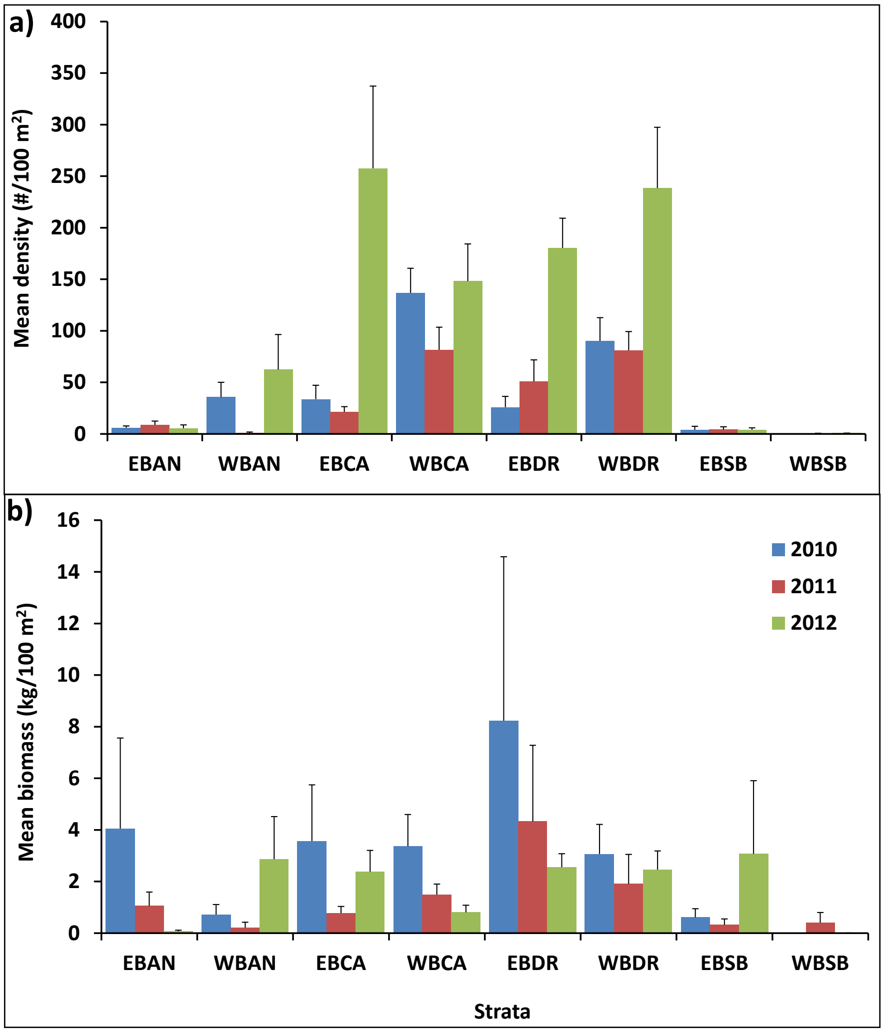


Figure 5.27. Mean fish a) density and b) biomass by year among survey strata.

Spatially, areas of highest density are located in the northeast and southwest quadrants of EB (Figure 5.28) and high density sites were more widely distributed throughout WB. Areas of highest density correlate with the mapped features of coralline algal habitats and deep reef.

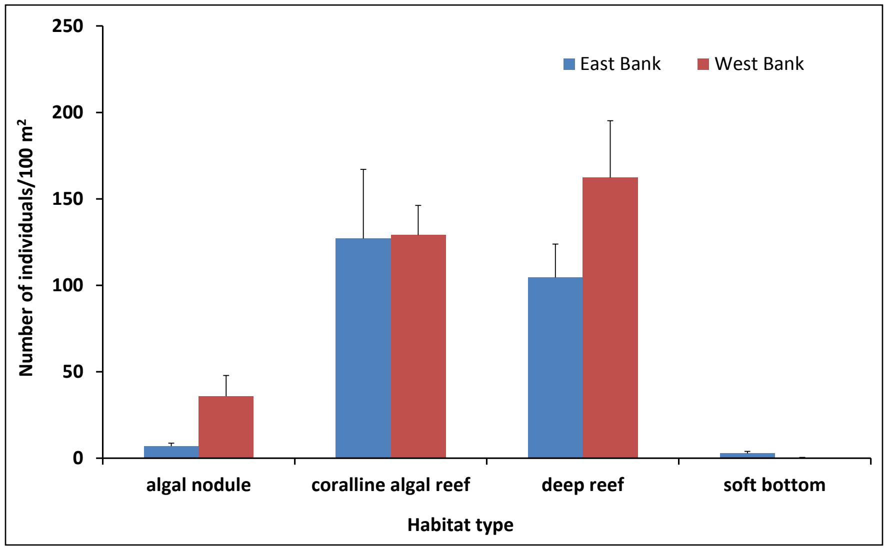


Figure 5.28. Mean density and standard error for fish observed on ROV surveys by strata, 2010-2012.

Total fish biomass exceeded 1,061 kg for all surveys combined. In general, total biomass on EB (627.1 kg) was almost 30% higher than on WB (434.3 kg); this is potentially related to greater effort on EB. Patterns of biomass paralleled that observed with density where coralline algal and deep reef habitats yielded higher values than algal nodule and soft bottom. Mean biomass was not significantly different among coralline algal and deep reef habitats on either bank (Figure 5.29). However, mean biomass for coralline algal and deep reefs (approximately 2-3.3 kg and 2-5 kg, respectively) was significantly greater ($p < 0.0001$) than algal nodule (0.6-1.1 kg) and soft bottom (0.1-1.2 kg). Mean biomass was not significantly different between algal nodule habitats on either bank but both were significantly greater than soft bottom habitats. Mean biomass on WB

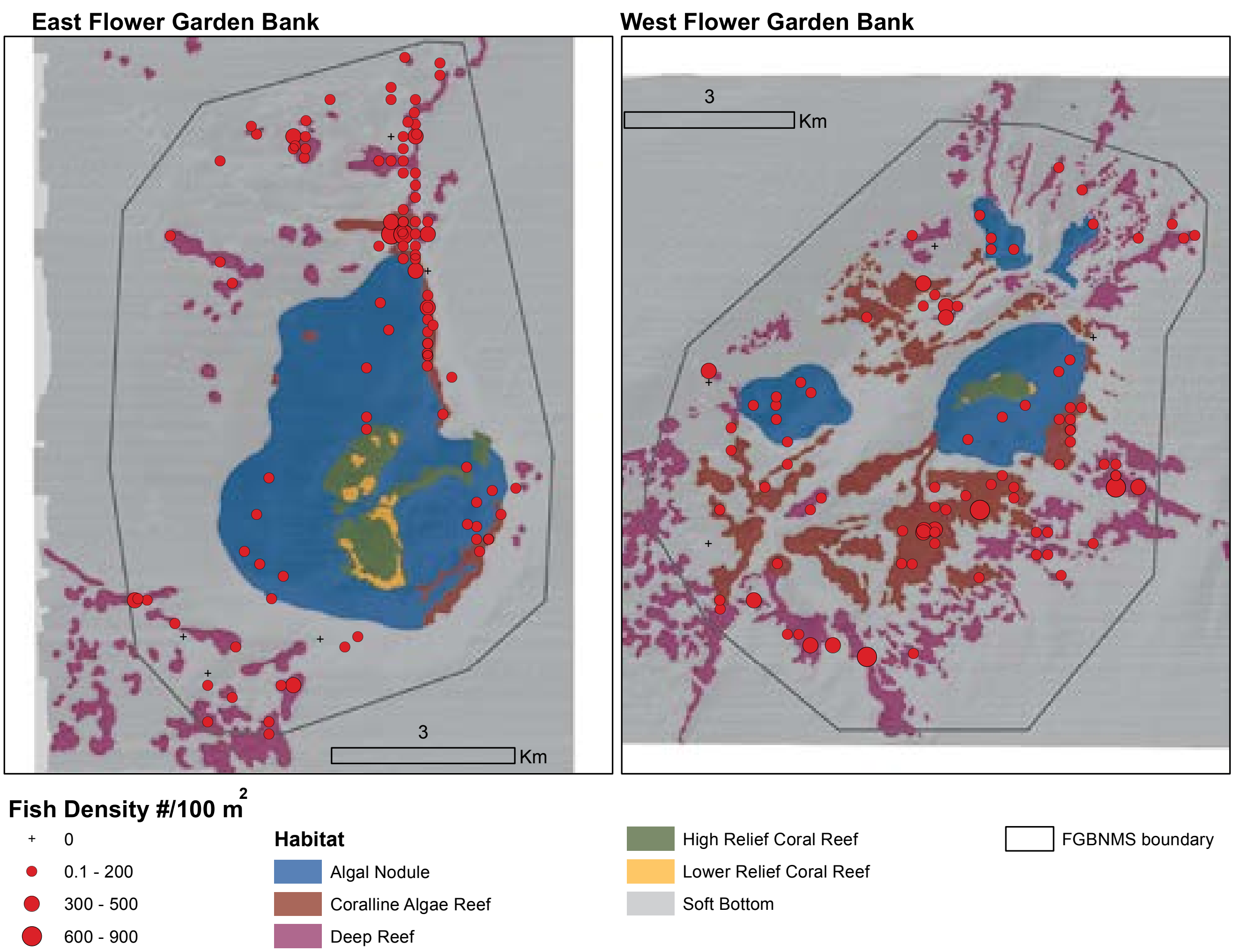


Figure 5.29. Mapped mean density (#/100 m^2) for fish observed on ROV surveys, 2010-2012.

soft bottom was significantly lower than EB soft bottom habitats. This is due to the observation of multiple individuals of *Lutjanus campechanus* (red snapper) on two soft bottom sites on EB. Each ROV station's biomass estimates are displayed in Figure 5.30. Sites with the highest biomass on EB appear to be located in the northeast quadrant on coralline algal and deep reefs. Some high biomass sites are also observed on deep reefs in the southwest portion of the bank. Biomass on WB is less localized but spread around various coralline algal and deep reefs around the bank (Figure 5.31).

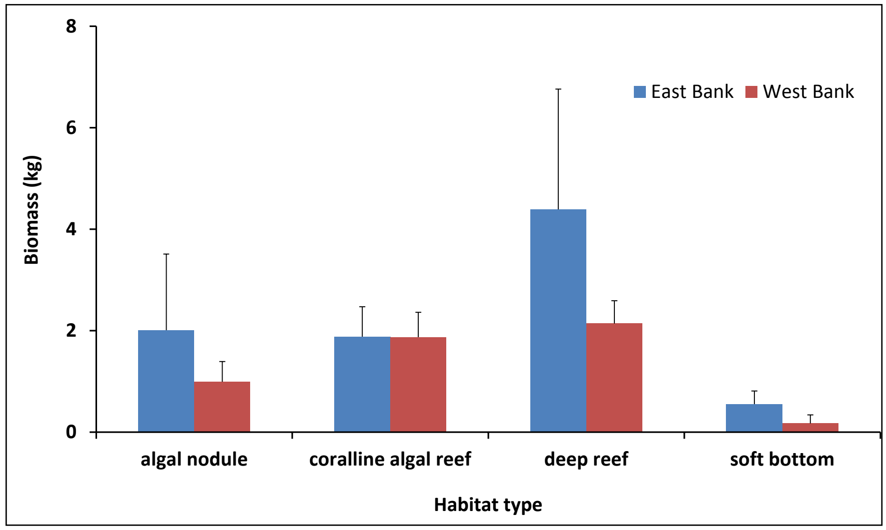


Figure 5.30. Mean biomass and standard error for fish observations on ROV surveys by strata, 2010-2012.

Overall, we observed 73 different fish species on the ROV surveys; 69 species on EB and 53 on WB. Mean richness was low overall with approximately 1.7-2.1 species/100 m^2 on coralline algal reef (Figure 5.32), 1.8-1.2/100

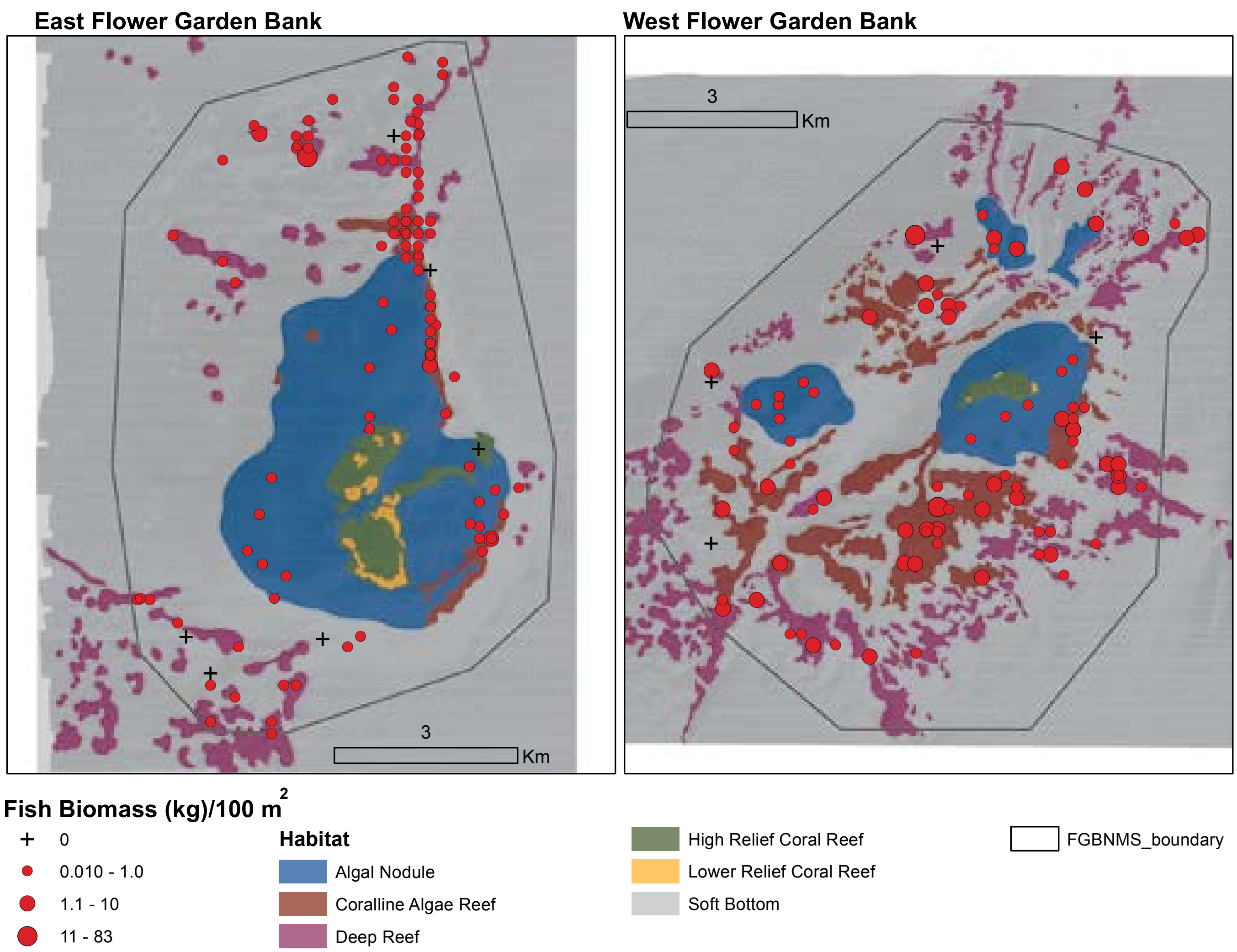


Figure 5.31. Mapped mean biomass (kg/100 m^2) for fish observed on ROV surveys, 2010-2012.

m² on algal nodule, 1.2/100 m² on deep reef an 0.3/100 m² on soft bottom. Richness on coralline algal habitats was significantly greater (p<0.0001) than all other habitats with the exception of WB algal nodule. Richness on soft bottom habitats was significantly lower (p < 0.0001) than that observed on all other habitats.

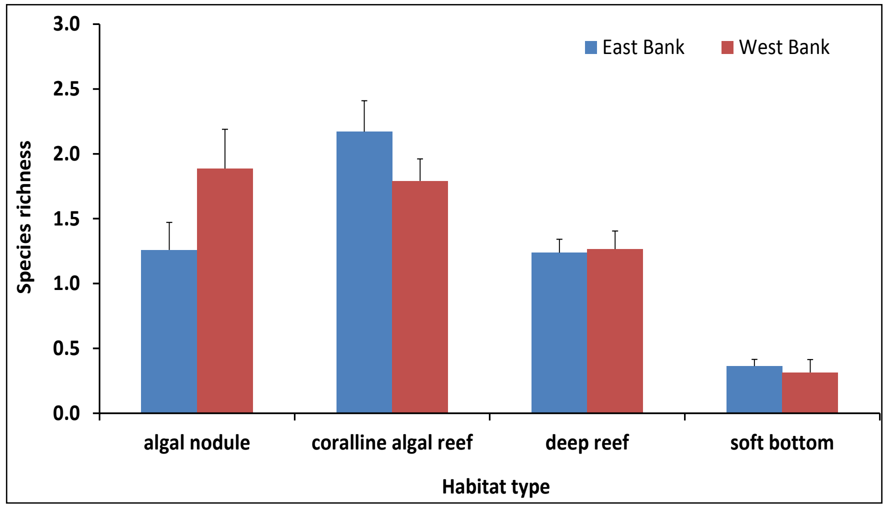


Figure 5.32. Mean species richness and standard error for fish observed on ROV surveys by strata, 2010-2012.

Spatial observation of the ROV stations (Figure 5.33) displaying total species observed highlights the amount of variability associated with the benthic habitats. In general, it appears that there are more high richness sites in the northeast quadrant of EB, while the southwest quadrant of WB contains additional high richness sites.

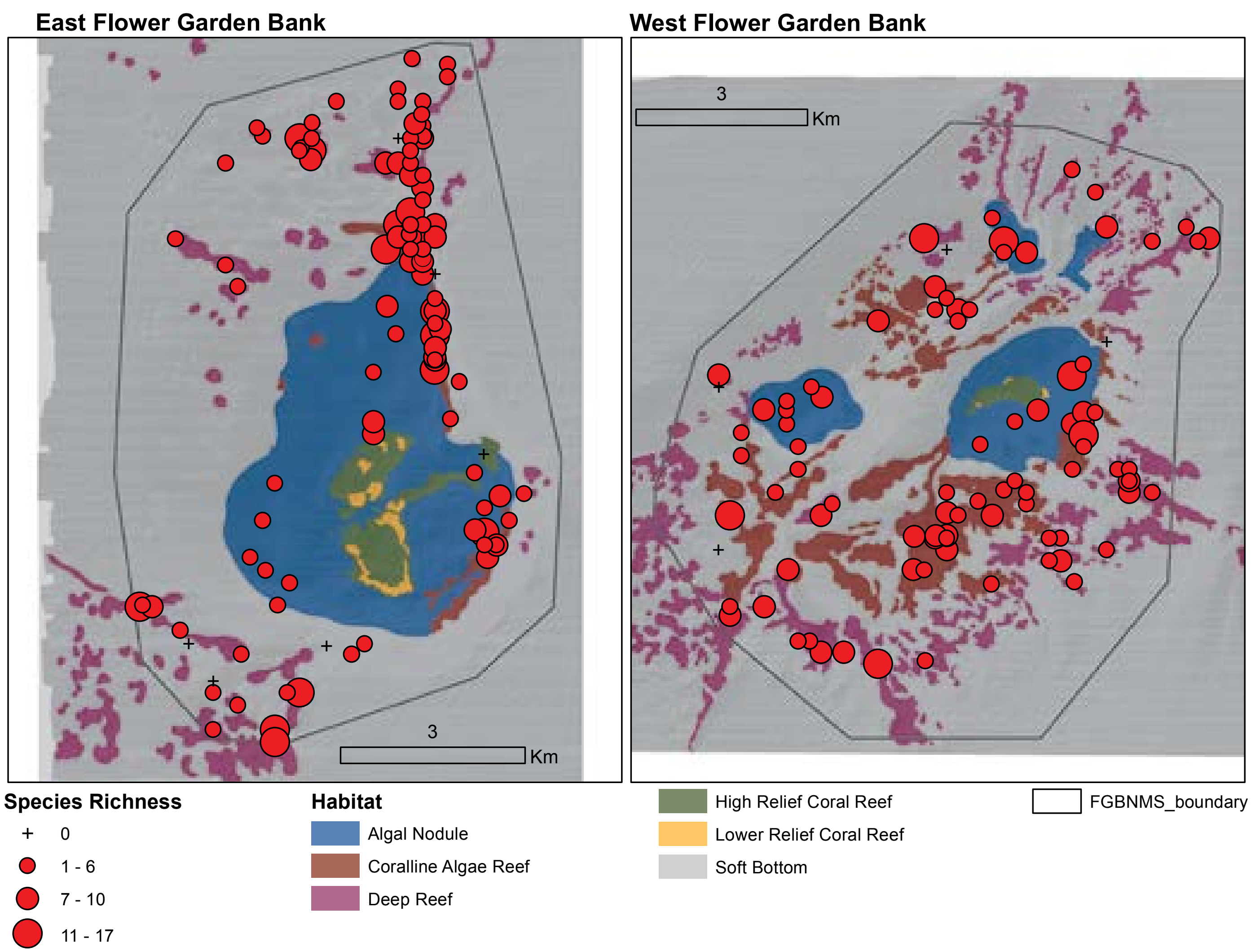


Figure 5.33. Mapped species richness (# of species/100 m²) for fish observed on ROV surveys, 2010-2012.

Trophic Groups

Planktivores were numerically dominant on algal nodule habitats, due to consistent large numbers of *Chromis insolata* (sunshinefish). Invertivores were the dominant group on all other habitats, except for WB coralline algal habitats (Figure 5.34a); most notably *P. martinicensis* and *C. tenuis* were the most dominant fish species on coralline algal and deep reefs. Herbivorous species were uncommon on most habitat types and were only moderately abundant on EB algal nodule habitats.

Piscivores were also less abundant, only contributing approximately 1-13% to total density among the habitats. In contrast, the piscivores (the heavy bodied groupers, snappers and jacks) dominated biomass in some habitat types (Figure 5.34b). Invertivore biomass was also a major component of total biomass at all sites. Herbivore biomass was low at all sites and planktivore biomass was most noticeable in deep reef and soft bottom habitats.

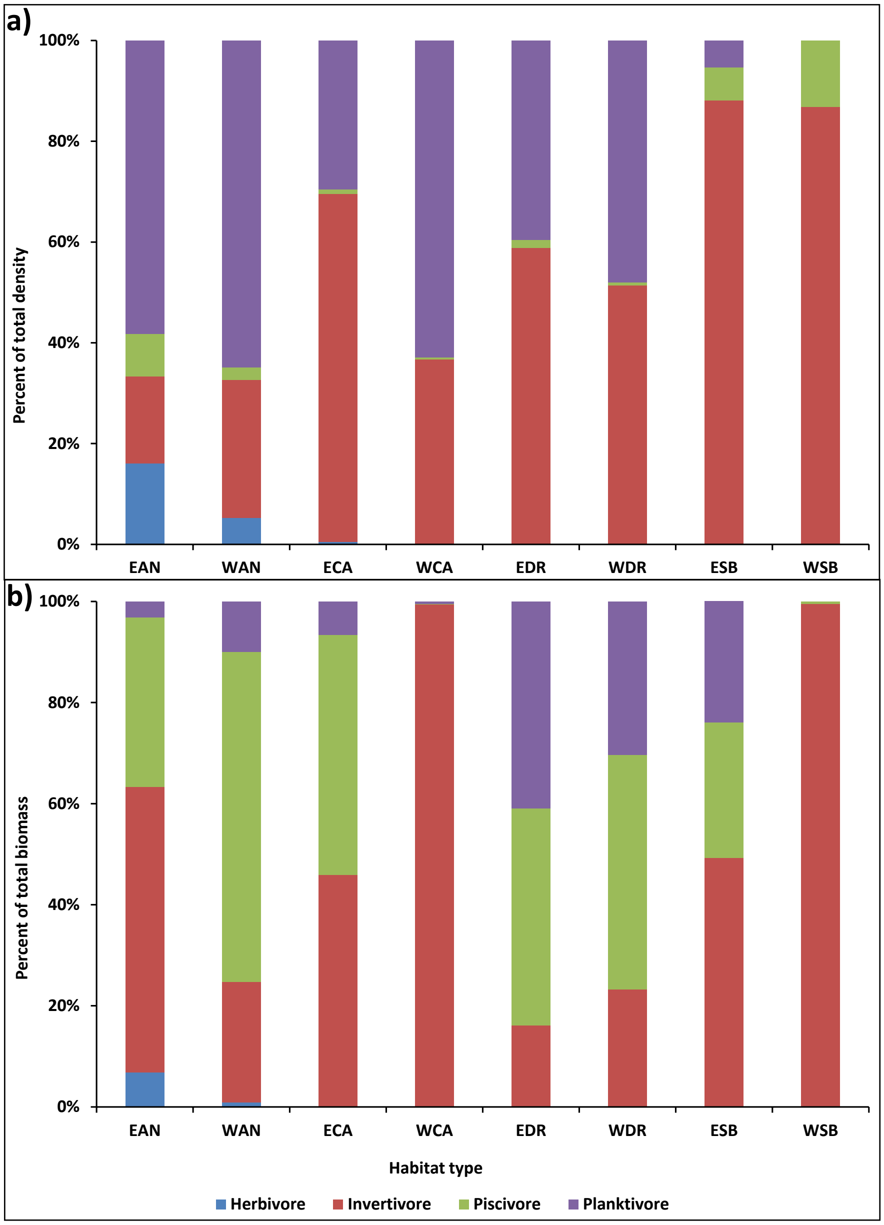


Figure 5.34. Percent contribution of (a) total density and (b) total biomass by fish trophic groups. H=herbivore, I=invertivore, PL=planktivore, P=piscivore; EAN= EB algal nodule habitats, WAN= WB algal nodule habitats, ECA= EB coralline algal habitats, WCA= WB coralline algal habitats, EDR= EB deep reef habitats, WDR= WB deep reef habitats, ESB= EB softbottom habitats, WSB= WB softbottom habitats.

Dominant Taxa by Habitat Type

Algal nodule

Dominant species on algal nodule habitats were similar between East and West Banks (Table 5.5) and were represented by Pomacanthidae: *C. insolata*, *Chromis enchyrsura* (yellowtail reeffish), *Centropyge argi* (cherubfish), *Chromis scotti* (purple reeffish); small species of Serranidae: *Serranus annularis* (orangeback bass), *P. martinicensis* (roughtongue bass), *Paranthias furcifer* (Atlantic creolefish), *C. tenuis* (threadnose bass); Scaridae (Subfamily Scarinae): *Sparisoma atomarium* (greenblotch parrotfish); Chaetodontidae: *Chaetodon sedentarius* (reef butterflyfish); Pomacentridae: *Stegastes partitus* (bicolor damselfish); Holocentridae (squirrelfish); and Labridae: *Bodianus rufus* (spotfin hogfish). These species are generally small bodied and generally reflect low biomass unless they occur in large numbers. *C. insolata* were the most numerically dominant species on EB, while four species (*P. martinicensis, C. tenuis, P. furcifer* and *C. insolata*) were most common on WB. Biomass on EB was enhanced by the presence of *L. campechanus* and four species of grouper while no snapper and only three individuals of grouper were observed on WB.

Table 5.5. Mean density and biomass for the 10 most abundant species, and select grouper and snapper species observed on algal nodule habitats.

	% Occurrence	Total Abundance	Density (#/100 m^2)		Total Biomass (g)	Biomass (g/100 m^2)	
			Mean	SE		Mean	SE
East Bank							
Chromis insolata	42.9	147	2.79	1.48	493.28	8.59	5.19
Chromis enchrysura	19.0	25	0.46	0.40	64.68	1.20	1.18
Sparisoma atomarium	23.8	19	0.37	0.17	47.30	0.94	0.51
Chaetodon sedentarius	33.3	19	0.39	0.15	436.41	8.89	4.12
Serranus annularis	38.1	17	0.35	0.12	35.78	0.79	0.51
Centropyge argi	23.8	15	0.36	0.25	10.21	0.23	0.16
Chromis scotti	9.5	15	0.27	0.24	7.88	0.14	0.12
Prontogrammus martinicensis	23.8	13	0.17	0.09	199.77	3.55	2.06
Lutjanus campechanus	9.5	10	0.15	0.10	8297.04	133.44	81.46
Stegastes partitus	19.0	10	0.23	0.12	25.50	0.52	0.42
Groupers/Snappers							
Mycteroperca phenax	23.8	5	0.10	0.05	6328.98	140.64	64.01
Dermatolepis inermis	4.8	1	0.02	0.02	108.80	2.46	2.46
Mycteroperca venenosa	4.8	1	0.02	0.02	1491.47	32.07	32.07
Mycteroperca interstitialis	4.8	1	0.02	0.02	1858.36	39.05	39.05
West Bank							
Prontogrammus martinicensis	17.6	402	10.44	6.40	3334.47	91.51	75.45
Choranthias tenuis	29.4	329	8.26	5.91	246.11	5.49	3.29
Paranthias furcifer	17.6	280	5.98	3.76	17930.31	373.53	256.36
Chromis insolata	52.9	207	5.48	2.56	345.98	7.65	3.87
Centropyge argi	52.9	49	1.42	0.54	33.35	0.96	0.37
Chromis enchrysura	23.5	30	0.91	0.54	11.79	0.36	0.21
Serranus annularis	41.2	16	0.42	0.23	63.45	1.66	1.43
Bodianus rufus	47.1	16	0.39	0.12	89.73	2.01	0.76
Sparisoma atomarium	41.2	15	0.44	0.16	15.31	0.42	0.21
Holocentrus adscensionis	23.5	12	0.25	0.17	1125.15	21.84	18.55
Groupers/Snappers							
Mycteroperca phenax	5.9	1	0.10	0.10	2914.06	75.84	75.84
Cephalopholis cruentata	5.9	1	0.03	0.03	82.00	2.25	2.25
Mycteroperca interstitialis	5.9	1	0.03	0.03	1858.36	50.97	50.97

Coralline algal reef

The same species that were numerically dominant on algal nodules were also dominant on coralline algal reefs, but organized differently (Table 5.6). *C. tenuis* and *P. martinicensis* were overwhelmingly the dominant taxa comprising 80-95% of the total abundance on each bank and were present on nearly all surveys. Other small species from Pomacanthidae, Labridae and Chaetodontidae were moderately abundant. *L. campechanus* were ranked in the top 10 for abundance on both banks and were more frequently encountered on WB (42.9%) than EB (19.4%). Nine species of groupers were observed on EB coralline algal reefs, with *Mycterpoerca phenax* (scamp) being the only species that was commonly seen. Only three species of grouper were seen on WB. The bulk of biomass was provided by the large numbers of small serranids and also boosted by the larger grouper species and *L. campechanus*. *Rhomboplites aurorubens* (vermilion snapper) were observed infrequently but in moderate abundance on WB.

Table 5.6. Mean density and biomass for the 10 most abundant species, and select grouper and snapper species observed on coralline algal reef habitats.

	% Occurrence	Total Abundance	Density (#/100 m^2)		Total Biomass (g)	Biomass (g/100 m^2)	
			Mean	SE		Mean	SE
East Bank							
Choranthias tenuis	67.7	7649	84.12	33.18	4686.85	56.53	23.75
Prontogrammus martinicensis	64.5	2436	23.75	11.25	7511.62	86.94	28.08
Chromis enchrysura	58.1	525	6.60	3.73	5174.47	70.49	20.69
Chromis insolata	51.6	319	3.26	1.35	968.83	15.22	8.45
Paranthias furcifer	29.0	199	2.82	1.69	11140.72	195.35	148.70
Bodianus rufus	64.5	92	1.06	0.22	1084.41	14.93	5.34
Lutjanus campechanus	19.4	39	0.39	0.21	27460.64	338.35	207.64
Holocentrus adscensionis	45.2	33	0.41	0.14	2906.48	41.66	16.15
Decodon puellaris	25.8	27	0.29	0.11	124.55	1.37	0.63
Centropyge agri	22.6	25	0.33	0.17	17.01	0.23	0.12
Groupers/Snappers							
Mycteroperca phenax	35.5	11	0.25	0.08	21454.10	277.14	107.62
Epinephelus guttatus	9.7	3	0.03	0.02	3188.49	38.49	23.62
Cephalopholis cruentata	6.5	2	0.04	0.03	1378.16	18.09	14.18
Mycteroperca venenosa	6.5	2	0.02	0.02	3503.29	42.92	30.66
Mycteroperca interstitialis	6.5	2	0.02	0.02	2382.07	25.52	19.00
Dermatolepis inermis	6.5	2	0.02	0.01	321.30	3.31	2.44
Epinephelus adscensionis	3.2	1	0.02	0.02	29.88	0.49	0.49
Hyporthodus flavolimbatus	3.2	1	0.01	0.01	2127.97	23.48	23.48
Mycteroperca bonaci	3.2	1	0.01	0.01	1923.30	21.89	21.89
West Bank							
Prontogrammus martinicensis	97.1	6425	77.72	7.66	38703.59	501.80	94.38
Choranthias tenuis	82.9	3546	44.22	12.45	7708.65	104.42	34.75
Chromis enchrysura	62.9	154	1.73	0.56	1532.36	16.83	5.72
Paranthias furcifer	20.0	125	1.16	0.62	6977.44	59.57	31.82
Rhomboplites aurorubens	14.3	102	1.54	1.33	2893.28	43.90	43.15
Chromis insolata	31.4	31	0.38	0.11	12.19	0.14	0.04
Chaetodon sedentarius	48.6	30	0.36	0.08	511.41	5.76	1.70
Lutjanus campechanus	42.9	24	0.32	0.10	21949.29	257.64	73.46
Bodianus rufus	42.9	24	0.26	0.06	163.36	1.77	0.55
Seriola rivoliana	8.6	16	0.22	0.15	38733.83	511.31	376.89
Groupers/Snappers							
Mycteroperca phenax	31.4	16	0.20	0.07	14293.95	171.26	58.67
Mycteroperca bonaci	5.7	4	0.03	0.02	4390.74	29.27	21.82
Mycteroperca interstitialis	11.4	4	0.04	0.02	9343.84	78.69	40.98

Deep reef

Like coralline algal reefs, *C. tenuis* and *P. martinicensis* were the dominant taxa on deep reefs and were observed on at least 66% of the surveys (Table 5.7). Deeper water taxa began to emerge on deep reef surveys including *Liopropoma eukrines* (wrasse bass), *Priacanthus arenatus* (bigeyes), and *Gonioplectrus hispanus* (Spanish flag). Members from Pomacanthidae, Chaetodontidae, Labridae and Serranidae comprised the remainder of the species most commonly observed on deep reefs. *L. campechanus* were more frequently observed on EB (57.1%) and WB (46.9%) deep reefs than other habitats. *M. phenax* were moderate to common on these reefs. Other large grouper and *R. aurorubens* were present but infrequently observed on each bank.

Table 5.7. Mean density and biomass for the 10 most abundant species and select grouper and snapper observed on deep reef habitats.

	% Occurrence	Total Abundance	Density (#/100 m²)		Total Biomass (g)	Biomass (g/100 m²)	
			Mean	SE		Mean	SE
East Bank							
Choranthias tenuis	68.6	7905	57.30	15.16	4737.12	32.42	7.94
Prontogrammus martinicensis	88.6	4985	38.92	6.44	20331.71	197.66	28.71
Lutjanus campechanus	57.1	76	1.36	0.55	82643.43	1654.68	694.14
Hemanthias leptus	11.4	63	0.43	0.28	576.63	4.74	2.24
Paranthias furcifer	20.0	56	1.10	0.67	6510.19	85.83	63.82
Chromis enchrysura	20.0	39	0.39	0.20	1426.78	14.33	8.16
Chaetodon sedentarius	37.1	25	0.59	0.39	732.04	27.38	22.88
Mycteroperca phenax	40.0	25	0.39	0.19	34593.73	423.19	170.43
Lioprpoma eurkrines	34.3	21	0.20	0.06	305.20	3.47	1.37
Priacanthus arenatus	20.0	20	0.21	0.11	1973.08	19.34	14.06
Groupers/Snappers							
Mycteroperca bonaci	2.9	1	0.01	0.01	1923.30	18.80	18.80
Cephalopholis cruentata	2.9	1	0.01	0.01	330.22	4.09	4.09
Hyporthodus flavolimbatus	2.9	1	0.03	0.03	5958.37	42.75	42.75
Mycteroperca venenosa	2.9	1	0.01	0.01	1491.47	12.26	12.26
Mycteroperca interstitialis	2.9	1	0.01	0.01	2459.37	24.30	24.30
Rhomboplites aurorubens	8.6	3	0.06	0.04	364.53	2.42	1.75
West Bank							
Choranthias tenuis	75.0	8402	81.57	23.86	6219.76	57.31	20.89
Prontogrammus martinicensis	100.0	7622	75.76	10.71	31330.76	352.01	59.27
Paranthias furcifer	18.8	164	1.52	1.29	25923.03	235.32	207.64
Gonioplectrus hispanus	46.9	65	0.63	0.39	4465.79	52.33	28.93
Lutjanus campechanus	46.9	49	0.61	0.15	66646.92	812.41	193.82
Decodon puellaris	53.1	41	0.43	0.09	474.81	4.67	1.29
Chromis enchrysura	37.5	24	0.25	0.07	418.59	3.90	1.47
Serranus phoebe	28.1	16	0.18	0.06	389.91	3.98	1.51
Chatodon sedentarius	31.3	15	0.17	0.05	361.74	3.66	1.14
Prognathodes aya	18.8	13	0.15	0.06	308.85	3.16	1.34
Groupers/Snappers							
Mycteroperca phenax	25.0	10	0.13	0.05	13216.36	136.61	57.68
Mycteroperca interstitialis	9.4	5	0.05	0.03	10633.10	95.73	55.45
Hyporthodus flavolimbatus	3.1	1	0.03	0.03	2650.90	37.72	37.72
Mycteroperca microlepis	3.1	1	0.01	0.01	1322.89	19.33	19.33
Rhomboplites aurorubens	3.1	1	0.04	0.04	580.48	8.18	8.18

Soft bottom

Both the number of species and abundance were greatly reduced on soft bottom habitats (Table 5.8). Soft bottom habitats on WB were depauperate of both fish species and abundance. Elevated densities and numbers of species were observed on EB as some areas had steep bathymetric features that tended to provide structure for reef species such as *L. campechanus*. *L. campechanus,* which were moderately common (found on 32% of softbottom surveys) on EB and represent the large majority of biomass for this habitat. *L. campechanus* were rarely seen on WB soft bottom habitats. No large grouper species were observed on soft bottom habitats.

Table 5.8. Mean density and biomass for the 10 most abundant species and select grouper and snapper observed on soft bottom habitats.

	% Occurrence	Total Abundance	Density (#/100 m^2)		Total Biomass (g)	Biomass (g/100 m^2)	
			Mean	SE		Mean	SE
East Bank							
Choranthias tenuis	13.6	122	1.27	0.96	65.18	0.72	0.41
Lutjanus campechanus	31.8	63	0.79	0.59	38568.54	507.50	253.07
Decodon puellaris	31.8	11	0.14	0.05	53.21	0.71	0.23
Serranus notospilus	9.1	8	0.12	0.10	41.00	0.58	0.46
Serranus phoebe	13.6	5	0.09	0.06	62.22	1.20	0.74
Pristigenys alta	4.5	4	0.05	0.05	211.84	2.35	2.35
Hemanthias vivanus	4.5	3	0.07	0.07	23.92	0.50	0.50
Priacanthus arenatus	4.5	2	0.03	0.03	155.80	2.35	2.35
Lizardfish spp.	4.5	2	0.04	0.04	8.34	0.18	0.18
Chaetodon sedentarius	4.5	2	0.03	0.03	24.68	0.37	0.37
West Bank							
Decodon puellaris	14.3	2	0.07	0.07	11.69	0.48	0.48
Malacanthus plumieri	28.6	2	0.11	0.07	66.79	4.30	3.87
Lizardfish spp.	14.3	1	0.04	0.04	19.29	0.79	0.79
Lutjanus campechanus	14.3	1	0.05	0.05	2717.27	169.96	169.96

*Deepwater species: roughtongue bass (*Prontogrammus martinicensis*; left) and yellowfin grouper (*Mycteroperca venenosa*; right) taken by ROV at FGBNMS. Photos: NOAA NOS/ONMS/FGBNMS and UNCW-UVP*

Groupers and Snappers

The groupers and snappers are highlighted here because of their ecological (top level predator) and fishing (targeted by commercial and recreational fishers) importance. Species of interest include groupers from the genus *Cephalopholis, Epinephelus, Dermatolepis* and *Mycteroperca* (Table 5.9). Snappers included here reflect the deeper occurring species, *L. campechanus* and *R. aurorubens,* and shallow species *Lutjanus jocu* (dog snapper), *Lutjanus cyanopterus* (cubera snapper), *Lutjanus mahogani* and *Lutjanus griseus* (gray snapper).

*Table 5.9. Size limits and length at maturity information for grouper and snapper observed on ROV surveys in FGBNMS, 2010-2012. * Indicates species is managed by the Gulf of Mexico Fishery Management Council's Reef Fish Fishery Management Plan. Lengths are total length (TL).*

Species	Size limit for rec fishery (cm-TL)	Size limit for commercial fishery (cm-TL)	Size at maturity (cm-TL)	Reference
Mycteroperca bonaci	55.8	61	*Campeche Bank - 72.1; Florida - 82.6; Cuba - 84.4	Brule et al., 2003
Cephalopholis cruentata	N/A	N/A	*Curacao - 14	Nagelkerken, 1979
Mycteroperca phenax	40.6	40.6	50% at 35.3	Harris et al., 2002
Mycteroperca interstitialis	50.8	50.8	40-45	Bullock and Murphy, 1994
Dermatolepis inermis	N/A	N/A	N/A	
Mycteroperca microlepis	55.8	61	Southeast US 50% at 62.2	McGovern et al., 1998a,b
Epinephelus adscensionis	N/A	N/A	Florida - 25	Potts and Manooch, 1995
Epinephelus guttatus	N/A	N/A	Jamaica 50% at 25	Thompson and Munro, 1978
Hyporthodus flavolimbatus	None	None	Gulf of Mexico 50% at 56.8	Bullock et al., 1996
Mycteroperca venenosa	50.8	50.8	N/A	
Lutjanus campechanus	40.6	38.1	Gulf of Mexico 50% at 37.8	White and Palmer, 2004
Rhomboplites aurorubens	25.4	25.4	Southeast US 100% at 150-200	Cuellar et al., 1996

Abundance was low for groupers and snappers throughout the deep habitats (47-125 m) of the sanctuary accounting for 1% of total abundance. The select groupers were present on 48% of all ROV surveys and on 40% of coralline algal and deep reef habitats, while frequency dropped below 30% on algal nodule habitats (Table 5.10). Groupers were not observed on soft bottom habitats. The select snapper species were present on 34% of all ROV surveys. Snapper frequency of occurrence (Table 5.10), in general, was similar to that of groupers, highest values on coralline algal and deep reefs and diminishing on algal nodule and softbottom habitats.

Table 5.10. Frequency of occurrence for selected grouper and snapper species by strata, 2010-2012. EAN= EB algal nodule habitats, WAN= WB algal nodule habitats, ECA= EB coralline algal habitats, WCA= WB coralline algal habitats, EDR= EB deep reef habitats, WDR= WB deep reef habitats, ESB= EB softbottom habitats, WSB= WB softbottom habitats.

Strata	N	Grouper frequency of occurrence	Snapper frequency of occurrence
EAN	21	28.57	19.05
ECA	29	48.28	13.79
EDR	36	44.44	52.78
ESB	23	0	39.13
WAN	17	11.76	0
WCA	35	42.86	45.71
WDR	32	37.5	50
WSB	8	0	12.5

Grouper and snapper abundance was greater on coralline algal and deep reef habitats (Figure 5.35a, 5.36a). Frequency of occurrence and density were significantly greater on hardbottom habitats than soft bottom. Scamp and red snapper were the most abundant on coralline algal and deep reefs on both banks (Table 5.10). Densities of other groupers and snappers were considerably lower on all habitat types.

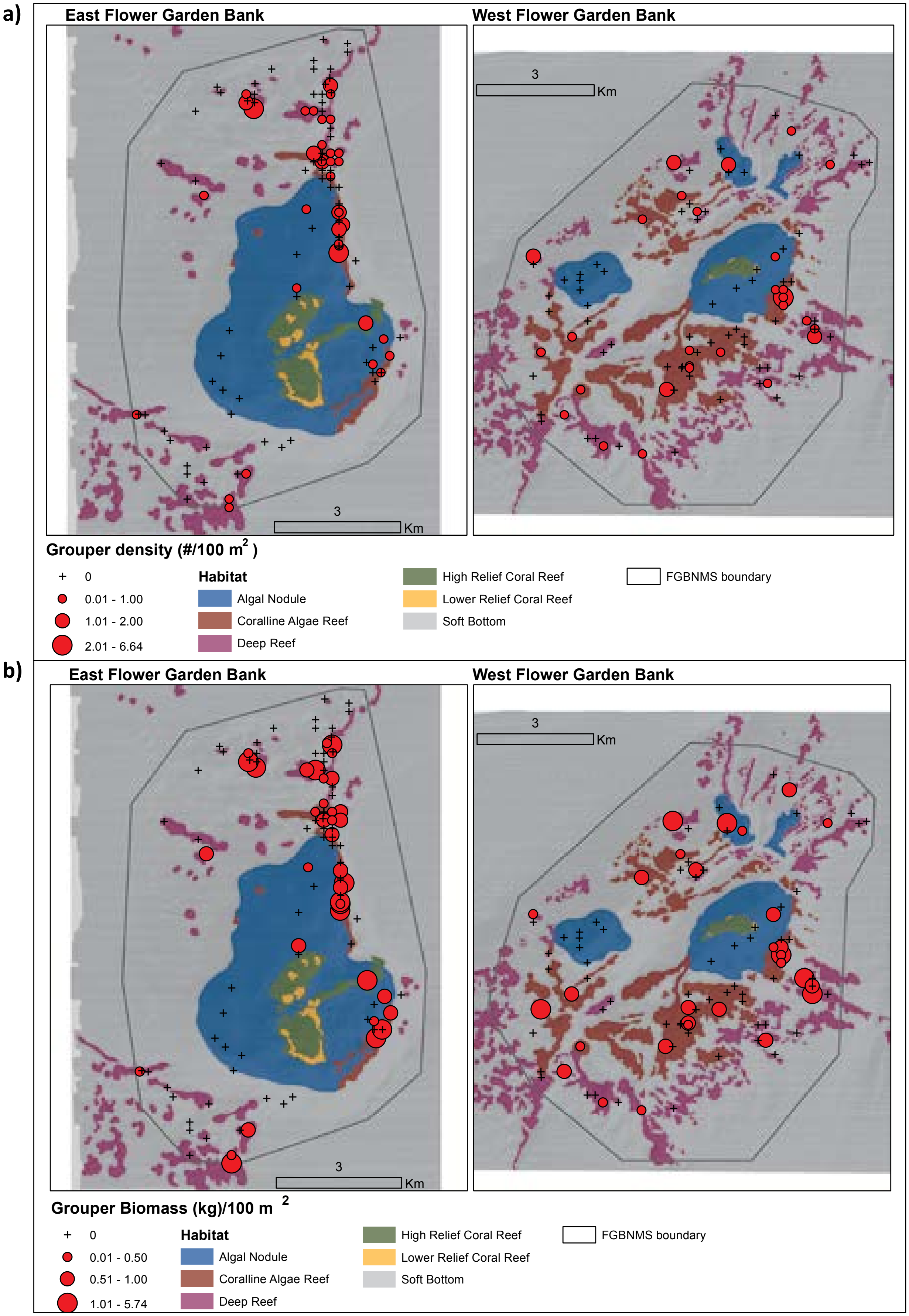


Figure 5.35. Spatial distribution of select grouper a) density and b) biomass on all strata, 2010-2012.

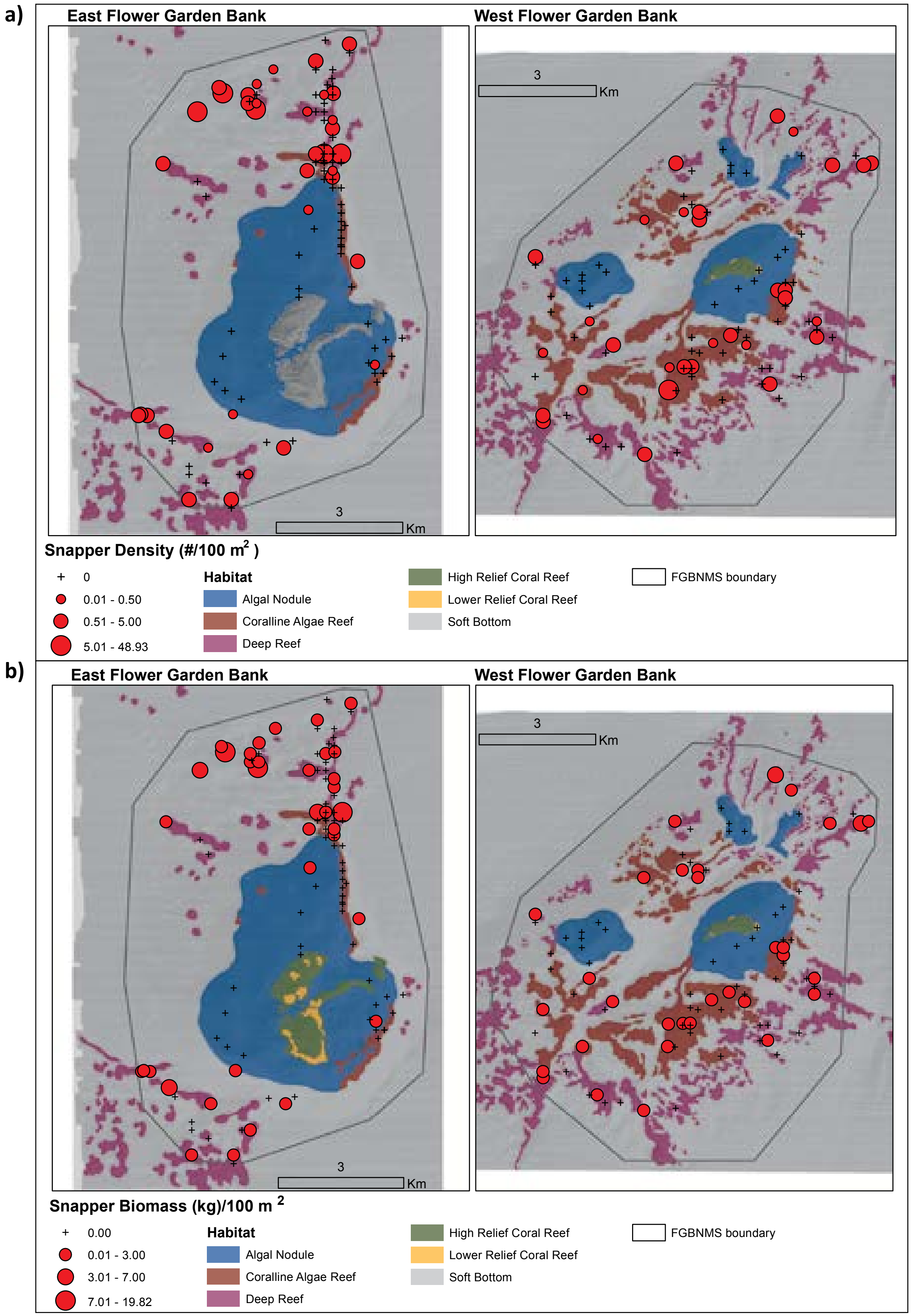


Figure 5.36. Spatial distribution of select snapper a) density and b) biomass observations on all strata, 2010-2012.

Grouper and snapper species composition changed with depth. On the coral reef (to 46 m), diver surveys observed that *Mycteroperca tigris* (tiger grouper), *Mycteroperca interstitialis* (yellowmouth grouper; Figure 5.35), *Epinephelus cruentata* (graysby), *L. griseus* and *L. jocu* were the common species. ROV surveys indicated that *M. phenax* and *L. campechanus* were the most common grouper and snapper species at depths greater than 46 m. Many other species were present, but less abundant.

*Picture of red snapper (*Lutjanus campechanus*) taken by ROV at FGBNM. Photo: NOAA NOS/ONMS/FGBNMS and UNCW-UVP*

Groupers and snappers accounted for 50% of the total biomass observed in the sanctuary. Spatially we observed greatest grouper (Figure 5.35b) biomass on coralline algal and deep reefs primarily in the northern part of EB and the southern part of WB. Snapper biomass was greatest on coralline algal and deep reefs.

Using density estimates (#/100 m^2 minus the SE) from Table 5.11 and area calculations for the habitats (see Table 2.6) grouper and snapper populations within each bank were estimated. Table 5.12 displays estimates for the population size for each grouper and snapper species for sanctuary habitats, excluding the coral caps.

Table 5.11. Mean density and standard error (SE) for grouper and snapper observations on all strata, 2010-2012. AN= algal nodule, CA= coralline algae reef, DR= deep reef, SB= soft bottom.

	East Bank				West Bank			
Species name	**AN**	**CA**	**DR**	**SB**	**AN**	**CA**	**DR**	**SB**
Mycteroperca bonaci	0	0.01(.01)	0.01(.01)	0	0	0.03(.02)	0	0
Mycteroperca microlepis	0	0	0	0	0	0	0.01(.01)	0
Cephalopholis cruentata	0	0.04(.03)	0.01(.01)	0	0.03(.03)	0	0	0
Dermatolepis inermis	0.02(.02)	0.02(.02)	0	0	0	0	0	0
Epinephelus guttatus	0	0.04(.02)	0	0	0	0	0	0
Epinephelus adscensionis	0	0.02(.02)	0	0	0	0	0	0
Mycteroperca phenax	0.12(.05)	0.25(.08)	0.38(.19)	0	0.10(.10)	0.20(.07)	0.13(.05)	0
Hyporthodus flavolimbatus	0	0.01(.01)	0.01(.01)	0	0	0	0.03(.03)	0
Mycteroperca venenosa	0.02(.02)	0.02(.02)	0.01(.01)	0	0	0	0	0
Mycteroperca interstitialis	0.02(.02)	0.02(.02)	0.01(.01)	0	0.03(.03)	0.04(.02)	0.05(.03)	0
Lutjanus campechanus	0.15(.10)	0.35(.22)	1.33(.54)	0.85(.56)	0	0.32(.10)	0.61(.15)	0.05(.05)
Rhomboplites aurorubens	0	0	0.06(.04)	0	0	1.54(1.33)	0.04(.04)	0

Table 5.12. Population estimates for select grouper and snapper species based on observed mean density on strata surveyed by ROV.

Species name	East Bank	West Bank	Species name	East Bank	West Bank
Mycteroperca bonaci	247	969	*Mycteroperca phenax*	19751	27665.75
Mycteroperca microlepis	0	478.5	*Hyporthodus flavolimbatus*	247	1435.5
Cephalopholis cruentata	317.5	1117.5	*Mycteroperca venenosa*	1838.5	0
Dermatolepis inermis	1662	0	*Mycteroperca interstitialis*	1838.5	4969.5
Epinephelus guttatus	282	0	*Lutjanus campechanus*	162112	77910
Epinephelus adscensionis	141	0	*Rhomboplites aurorubens*	706	20349

Mycteroperca bonaci (black grouper)

Adult *Mycteroperca bonaci* are typically found on coral reefs or rocky ledges/ outcrops at depths of 9-30 m, with a maximum depth at 100 m (Heemstra and Randall, 1993; Figure 5.37). They attain maximum size of approximately 150 cm TL and 81 kg (Mowbray, 1950). Size of 50% maturity has regional differences and ranges from 72-84 cm TL (Brule et al., 2003).

*Figure 5.37. Picture of black grouper (*Mycteroperca bonaci*) at FGBNMS. Photo: NOAA NOS/NCCOS/CCMA*

M. bonaci are included here as they have been assessed as "Near Threatened" by the International Union for Conservation of Nature and Natural Resources (IUCN) as they have undergone nearly 30% population decline and is expected to continue to decline in the future (Ferriera et al., 2008).

Overall, six *M. bonaci* were observed on ROV surveys, considerably less than that observed on the coral reef (see Chapter 4). *M. bonaci* were observed in low densities (Figure 5.38a) on coralline algal and deep reefs at depths between 7 8 and 95 m. Individuals were equal to or smaller than 60 cm (Figure 5.38b) and also below the commercial catch limit of 61 cm. Therefore these individuals are considered subadults.

Frequency of occurrence and density were greatest on the coral caps and no clear spatial pattern of abundance on the deepwater habitats (Figure 5.39). Using conservative density values the *M. bonaci* population on deepwater (>46 m) habitats is estimated to be about four times greater on WB (N=969) than EB (N=247; Table 5.12).

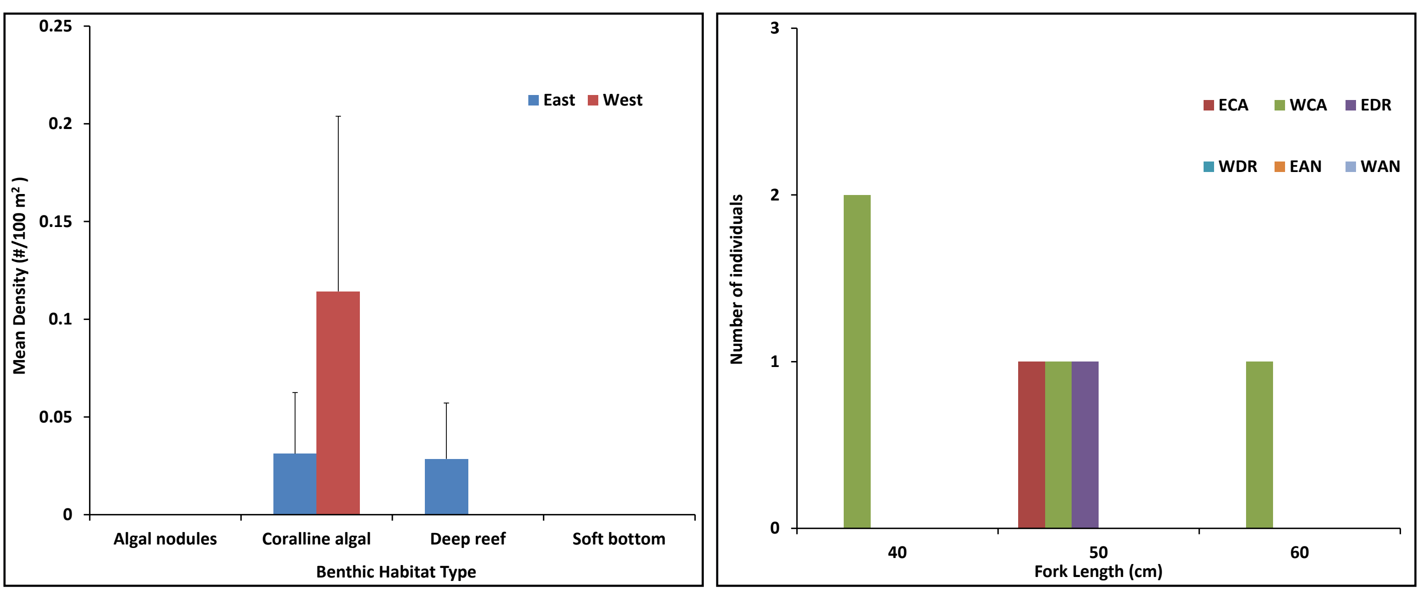


Figure 5.38. (a) Mean density and standard error and (b) length frequency distribution (FL) for M. bonaci *observed in ROV surveys, 2010-2012. ECA= EB coralline algal habitats, WCA= WB coralline algal habitats, EDR= EB deep reef habitats, WDR= WB deep reef habitats, EAN= EB algal nodule habitats, WAN= WB algal nodule habitats.*

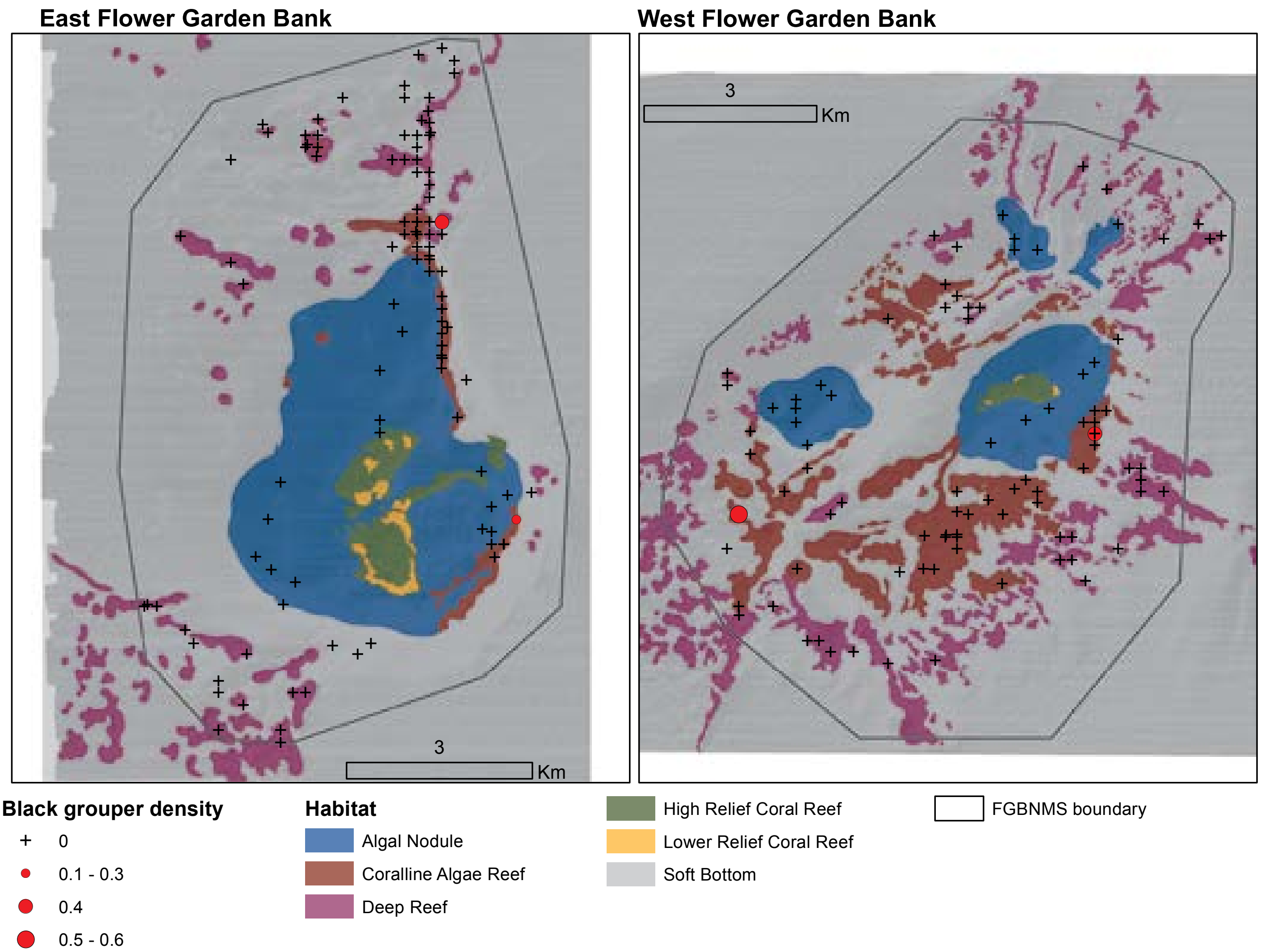


Figure 5.39. Spatial distribution of observed M. bonaci *density (#/100 m²) on deep water habitats surveyed by ROV, 2010-2012.*

<u>*Mycteroperca phenax* (scamp)</u>

Mycteroperca phenax are a reef associated species; found on ledges and high relief rocky bottoms in the Gulf of Mexico (Figure 5.40). In the Oculina Banks off the east coast of Florida, another deep reef area (70-100 m), they are the most abundant grouper. *M. phenax* attain maximum size of 107 cm TL and 14.2 kg. Length of 50% maturity in the Gulf of Mexico is 35.3 cm TL (Harris et al., 2002).

*Figure 5.40. Photo of scamp (*Mycteroperca phenax*) taken by ROV at FGBNMS. Photo: NOAA NOS/ONMS/FGBNMS and UNCW-UVP*

M. phenax were the most abundant grouper observed in the ROV surveys and were uncommon on SCUBA surveys (see Chapter 4). *M. phenax* densities were greatest on coralline algal and deep reefs (Figure 5.41a), and were significantly greater than densities on algal nodule and softbottom habitats (nonparametric Wilcoxon test; X > 0.0036). *M. phenax* densities were also greater on EB, but not statistically significant.

M. phenax length frequency (FL) ranged from 17.5 to 70 cm. Approximately 38% of the individuals were juveniles/subadults with lengths less than 35 cm. (Figure 5.41b). Nearly all the *M. phenax* observed on the coral reef were not marketable size (>40.6 cm TL) however most individuals on deeper habitats were above the minimum size for both fisheries.

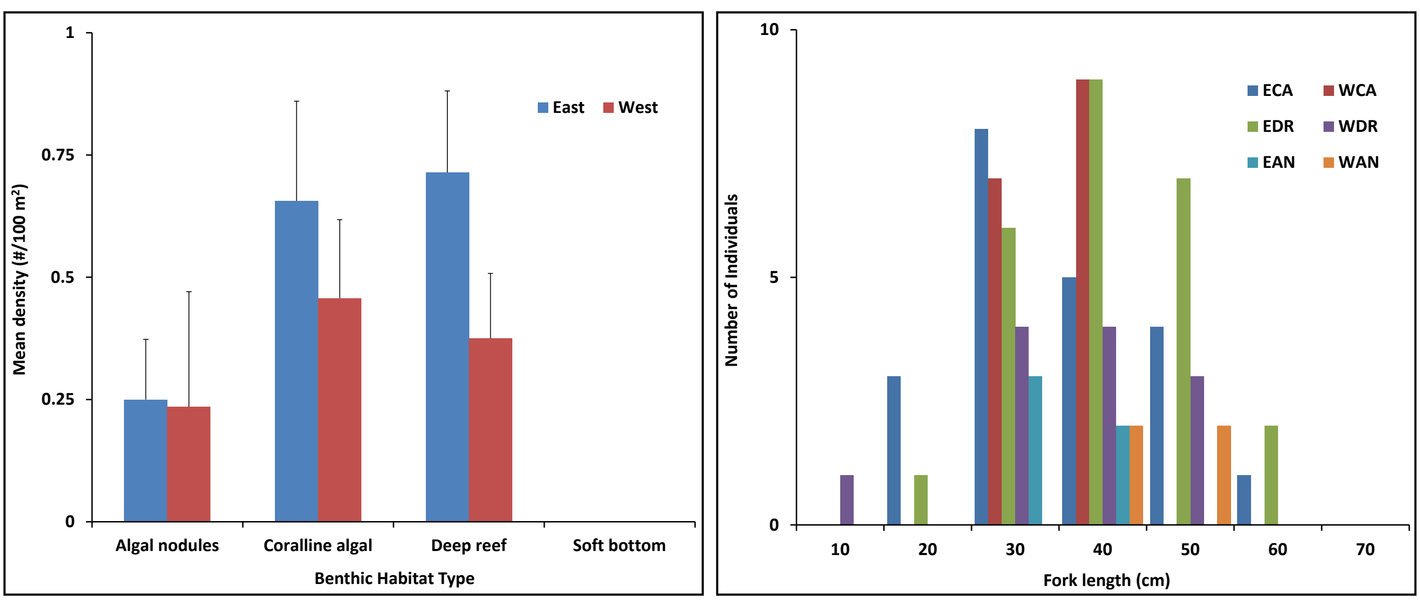


Figure 5.41. (a) Mean density and standard error and (b) length frequency distribution (FL) for M. phenax *observed in ROV surveys, 2010-2012.*

Overall, 88 individuals were observed and density was greatest on EB coralline algal and deep reef habitats (Table 5.9). Depth of observations ranged from 40-120 m, but the majority of observations occurred at 70-100 m on both banks (Figure 5.42). *M. phenax* density was highest on coralline algal and deep reef habitats on EB, primarily those found in the northern portion of the bank. Overall, *M. phenax* density was low on WB with only two observations on the coral reef. *M. phenax* were rare on algal nodule habitats and not present on soft bottom.

Population estimates (Table 5.12) for *M. phenax* rank it the most abundant grouper in the deep water habitats of the sanctuary. Although *M. phenax* density was higher on EB, total population is greater on WB due to greater amount of coralline algal reef and deep reef habitats.

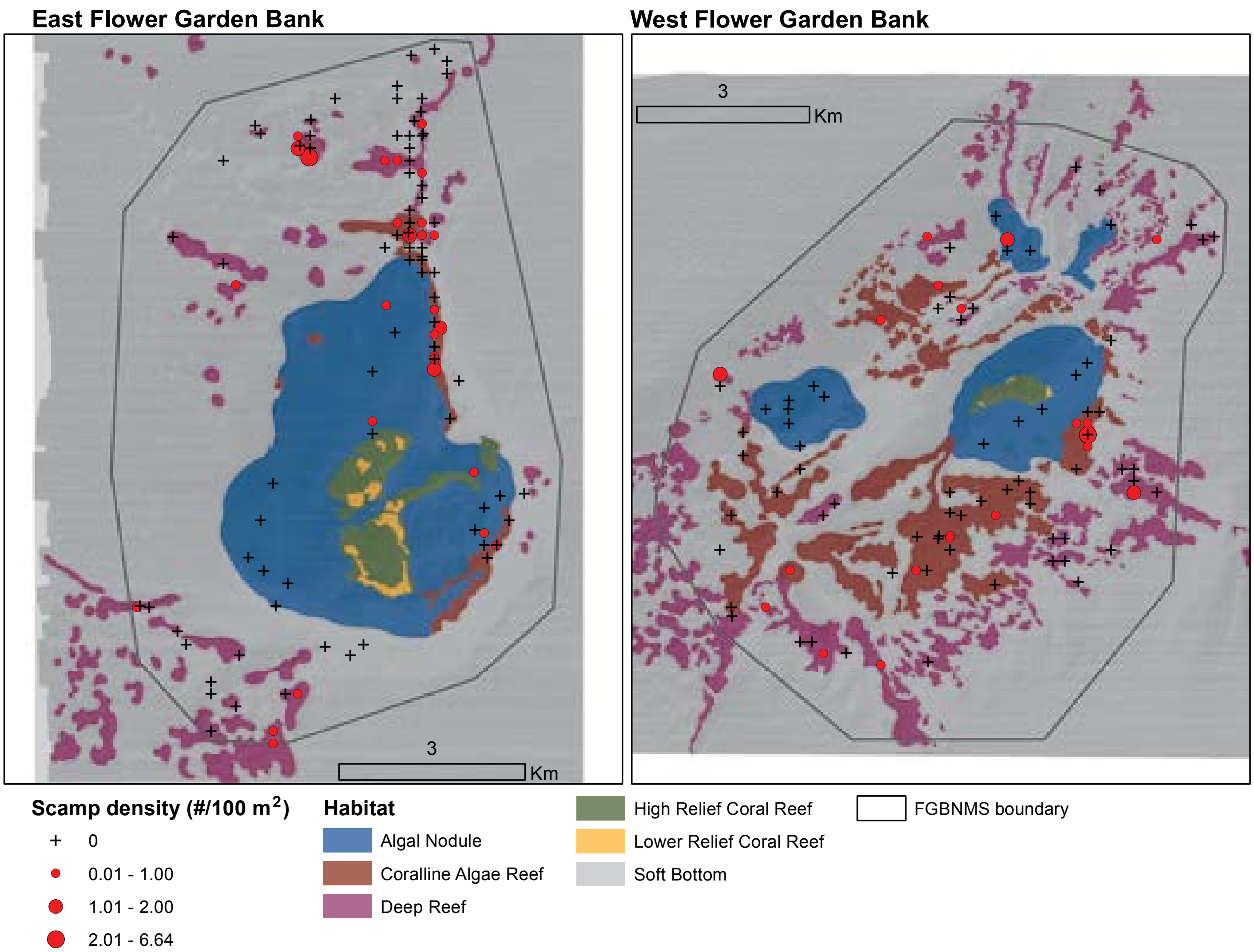


Figure 5.42. Spatial distribution of observed M. phenax *density (#/100 m²) on deep water habitats surveyed by ROV, 2010-2012.*

<u>*Mycteroperca interstitialis* (yellowmouth grouper)</u>

Mycteroperca interstitialis are uncommon throughout their range, including the Gulf of Mexico and reside on hardbottom habitats of rock or coral to depths of 150 m (SAFMC, 2005; Figure 5.43). *M. interstitialis* is listed as vulnerable by the IUCN (Ferriera et al., 2008) because of inferred declines in abundance of at least 30% in the past three generations (generation length is at least 10 years). Additionally, the species has life history characteristics that make this species more vulnerable to overfishing (long lived, protogynous; (Rocha et al., 2008a). Over-exploitation is possible since it is captured along with similar yet more abundant and more persistent species, such as *M. phenax* (Musick et al. 2000) and *M. bonaci* (Ferreira et al. 1998). *M. interstitialis* attain maximum size at 84 cm (TL) and weigh approximately 10.2 kg (Froese and Pauly, 2014). Females are sexually mature at 40-45 cm TL (Bullock and Murphy, 1994).

*Figure 5.43. Yellowmouth grouper (*Mycteroperca interstitialis*) taken by ROV at FGBNMS. Photo: NOAA NOS/ONMS/FGBNMS and UNCW-UVP*

M. interstitialis was the most abundant grouper on the coral reef (Chapter 4), but frequency of occurrence and density were greatly reduced in deeper (>46 m) hard bottom habitats. In general, density was greatest on coralline algal and deep reefs (Figure 5.44a), but the number of observations were not sufficient to statistically compare among strata.

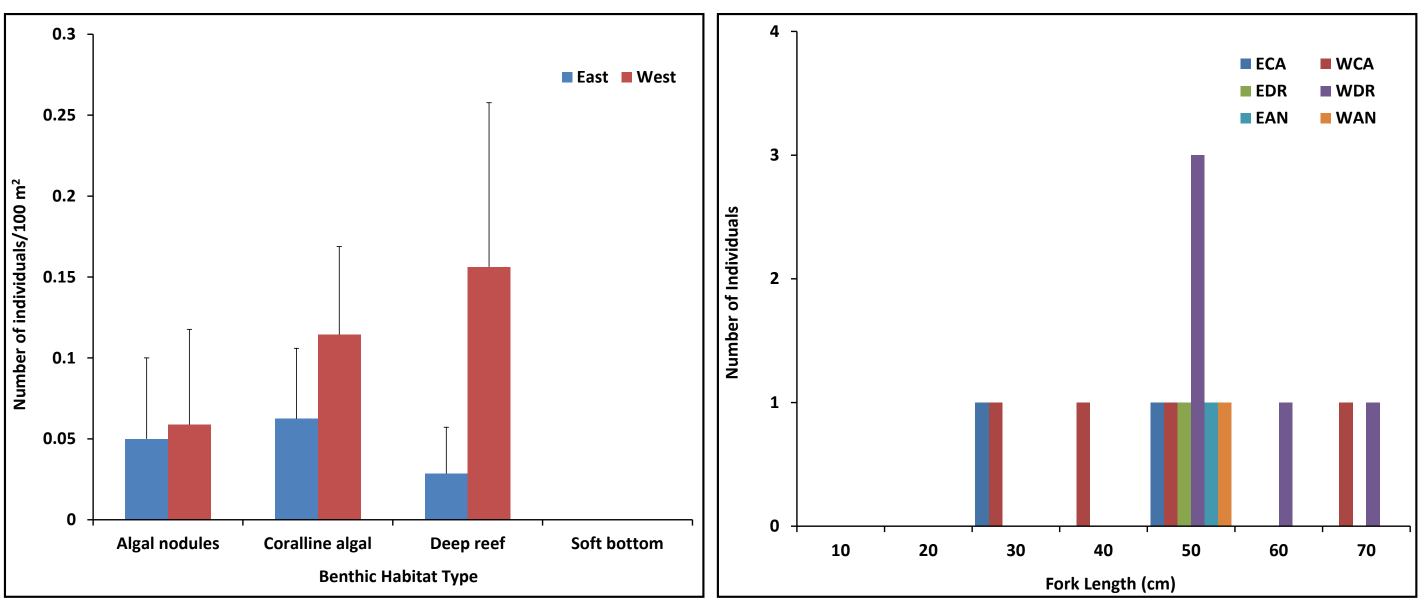


Figure 5.44. (a) Mean density and standard error and (b) length frequency distribution (FL) for yellowmouth grouper, M. interstitialis, *observed in ROV surveys, 2010-2012.*

Interestingly, nearly all *M. interstitialis* observed in ROV surveys were considered adults (Figure 5.44b), while those observed on the coral reef were mostly juveniles or subadults (Chapter 4). About 50% of the *M. interstitialis* observed in ROV surveys were harvestable to both commercial and recreational fisheries.

Overall 14 individuals were observed on habitats deeper than 46 m; most (n = 12) were on coralline algal or deep reef habitats at depths ranging from 61 to 105 m (Figure 5.45). Two individuals were recorded on algal nodule habitats at depths of 51 and 64 m. Most *M. interstitialis* on deep water habits were observed on WB at depths between 50-105 m.

The conservative estimate of *M. interstitialis* population size on deep habitats (> 46 m) is 1,838 individuals on EB and 4,969 on WB (Table 5.12).

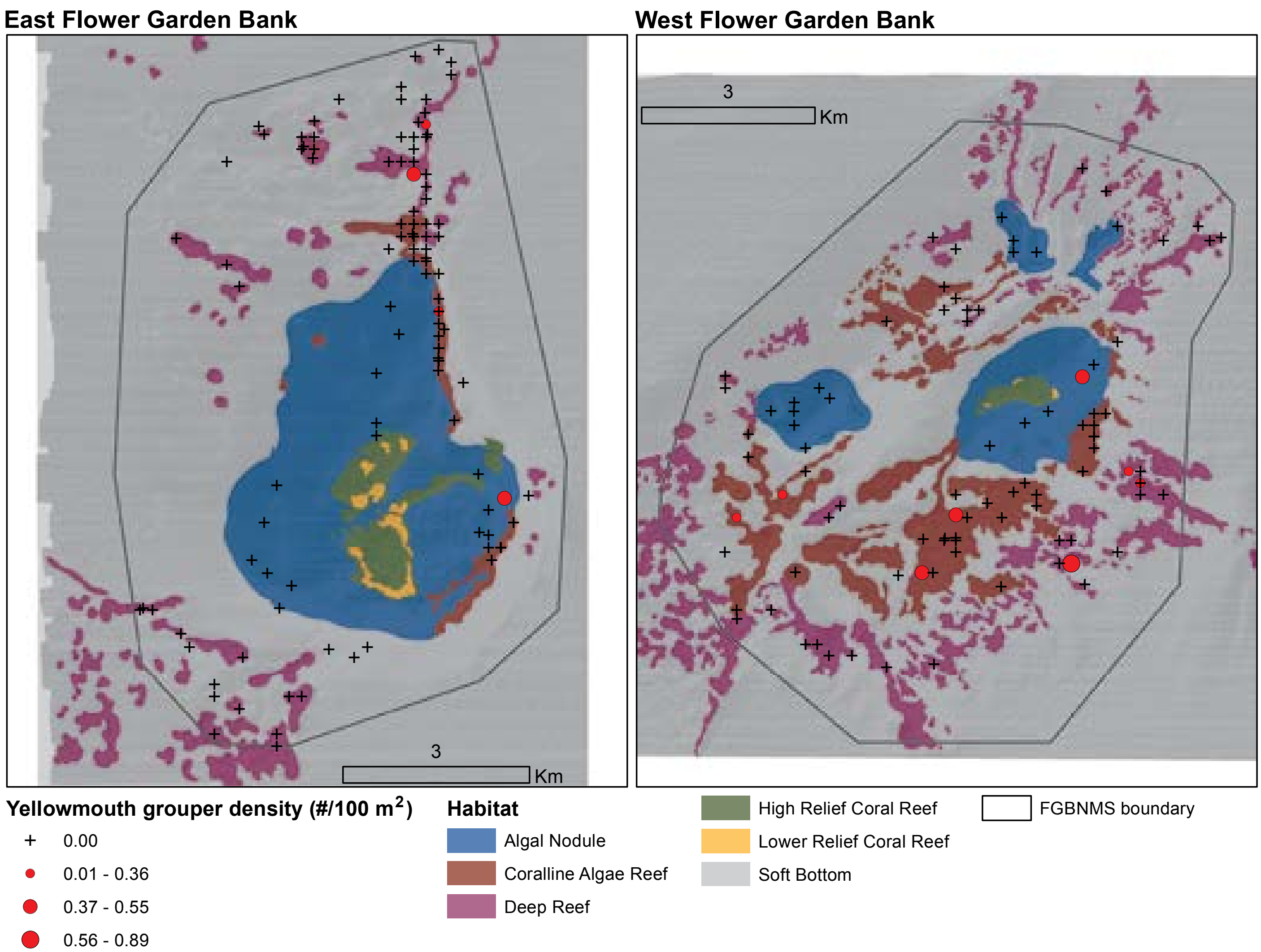


Figure 5.45. Spatial distribution of M. interstitialis *density (#/ 100 m²) on deep water habitats surveyed by ROV, 2010-2012.*

<u>*Dermatolepis inermis* (marbled grouper)</u>

Dermatolepis inermis are considered nearly threatened (Rocha et al., 2008b) based on their rarity, life history characteristics and heavy fishing pressure on spawning aggregations (Figure 5.46). *D. inermis* are a reef-associated species found over large depth ranges, 20-210 m. There are few observations of *D. inermis* in markets, or numbers are low due to historic overfishing. Little life history information is known, such as size at maturity and population status. Maximum size of 91 cm TL has been recorded (Robins and Ray, 1986).

*Figure 5.46. Marbled grouper (*Dermatolepis inermis*) taken by ROV at FGBNMS. Photo: NOAA NOS/ONMS/FGBNM and UNCW-UVP*

Historical ROV surveys (2001-2007) in the sanctuary indicated that the banks of the northwest Gulf of Mexico, specifically East and West Flower Garden Banks, are potential areas of high density for *D. inermis*.

During 2010-2012 three individuals of *D. inermis* were observed, one during each survey year and all on EB. In fact, all ROV observations and the majority of observations by divers were recorded on EB, a pattern similar to that observed in 2006 (Caldow et al., 2009). Low densities were observed overall (Figure 5.47a) on algal nodule and coralline algal reef. Depths of observations ranged from 47-60 m. *D. inermis* on deep habitats ranged in size from 40-50 cm (Figure 5.47b).

Density and frequency of occurrence was low for this species at depths greater than 50 m, but were mostly associated with EB hard bottom features with high relief (Figure 5.48).

The total population estimate for *D. inermis* on EB is 1,662 individuals (Table 5.12). *D. inermis* was not observed on WB deep habitats but have been observed on deep habitats on prior ROV surveys.

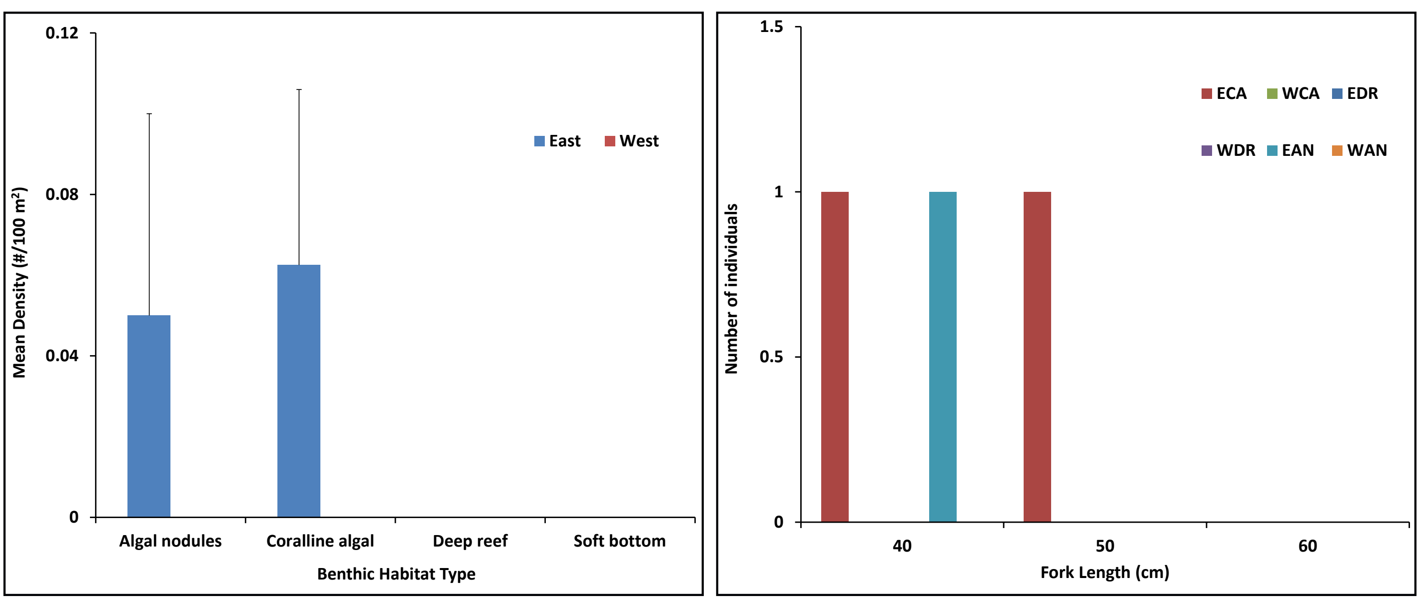


Figure 5.47. (a) Mean density and standard error and (b) length frequency distribution (FL) for D. inermis *observed in ROV fish surveys, 2010-2012.*

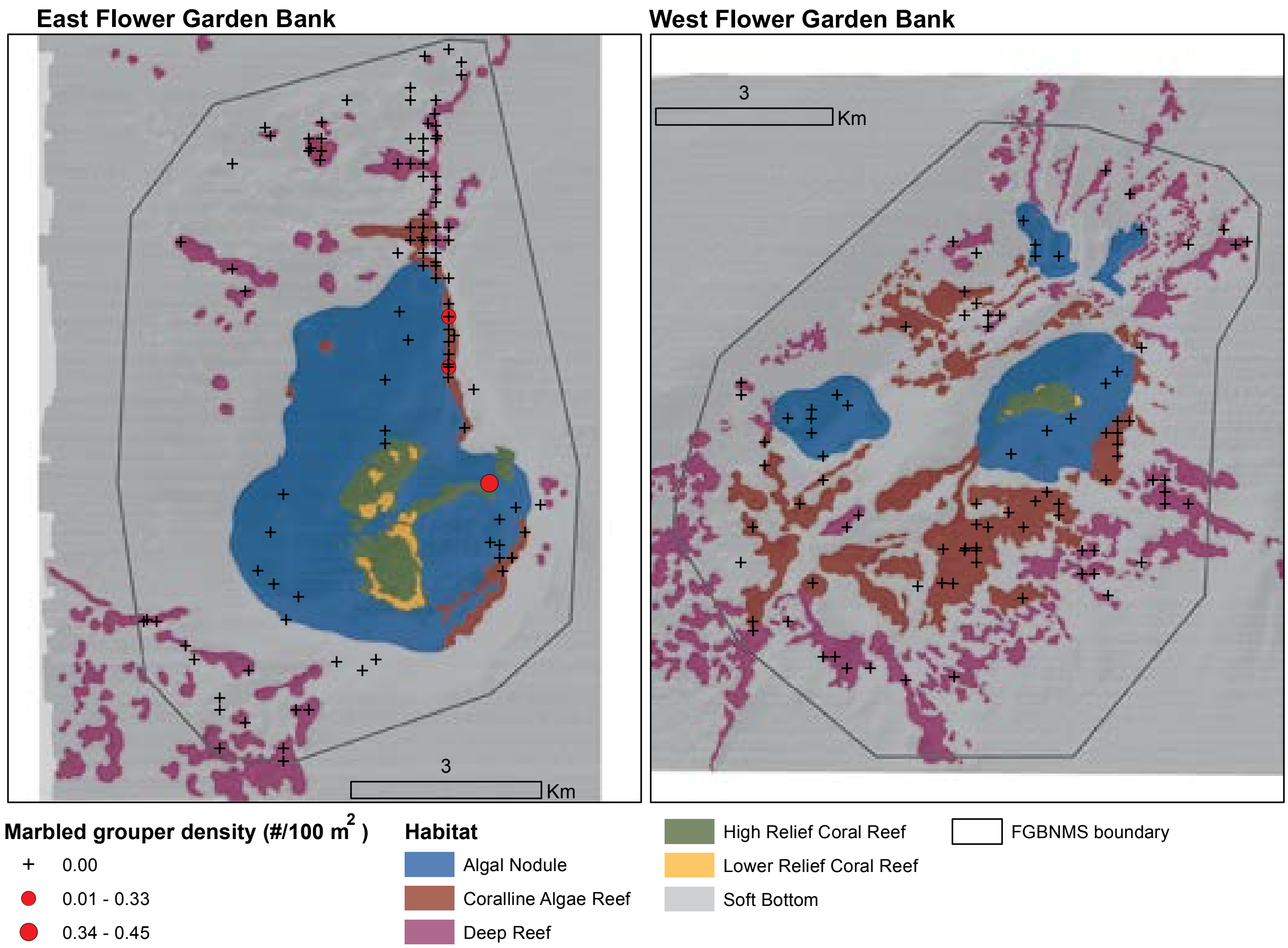


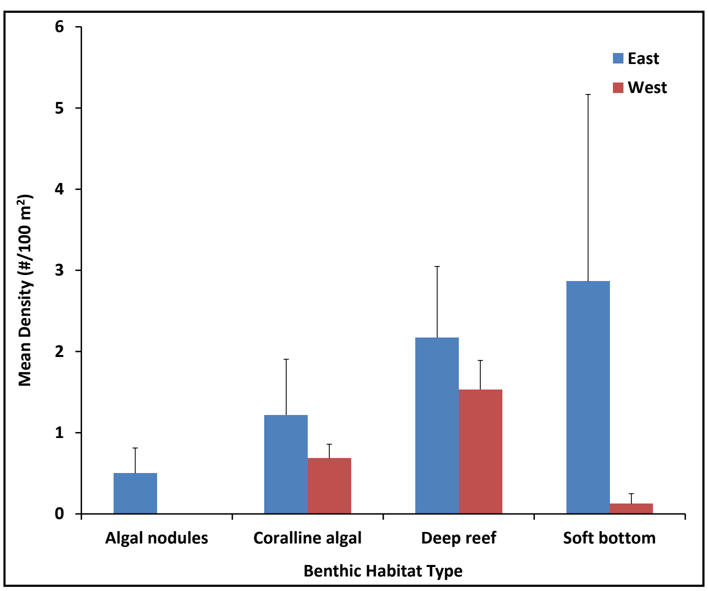
Figure 5.48. Spatial distribution of D. inermis *density (#/ 100 m²) on deep water habitats surveyed by ROV, 2010-2012.*

<u>*Lutjanus campechanus* (red snapper)</u>

Lutjanus campechanus occur across the shelf to the shelf edge and demonstrate an affinity for vertical structures (Patterson et al., 2001), especially between two and 10 years of age (Figure 5.49). Growth is very rapid during the first 8-10 years of life (Szedlmayer and Shipp, 1994; Patterson, 1999; Nelson and Manooch, 1982; Patterson et al., 2001; Wilson and Nieland, 2001; Fischer et al., 2004). After this period, fish continue to grow but at slower rates. Older and larger fish tend to expand habitat choices and may be frequently found on open softbottom areas (Szedlmayer, 2007). Age 1 snapper, generally 10-20 cm TL, typically inhabit many different substrates at depths between 18-55 m, but tend to prefer low-relief, relic shell habitat that protects them from predation (Galloway et al., 2009). At the beginning of age 2, young *L. campechanus* are generally between 200 and 375 mm TL (Goodyear, 1995) and *L. campechanus* begin to enter fishery minimum sizes around 40 cm. Preferred habitats include natural hard substrates with relief on the order of meters; e.g., reef pinnacles, exposed rock ledges, and shelf- edge banks, as well as artificial reefs like offshore oil and gas structures, shipwrecks, and constructed artificial reef areas.

*Figure 5.49. Red snapper (*Lutjanus campechanus*) taken by ROV at FGBNMS. Photo: NOAA NOS/ONMS/FGBNM and UNCW-UVP*

L. campechanus can attain large sizes, the largest reported in the Gulf of Mexico was 104 cm (TL) and weighed 22.8 kg (Wilson and Nieland, 2001). *L. campechanus* are most common at lengths of 60 cm TL (Allen, 1985). White and Palmer (2004) report that 50% of females in the Gulf of Mexico are sexually mature at 37.8 cm (TL). Several other reports provide data that support regional differences on maturity: full maturity 37.5 cm FL (Nelson and Manooch, 1982; Collins et al., 1996) and 49.4 cm FL (Martinez-Andrade, 2003). *L. campechanus* were present on all deep strata with the exception of WB algal nodule. Frequency of occurrence was highest on coralline algal and deep reefs. However, frequency of occurrence was nearly four times higher on WB coralline algal reefs than EB.



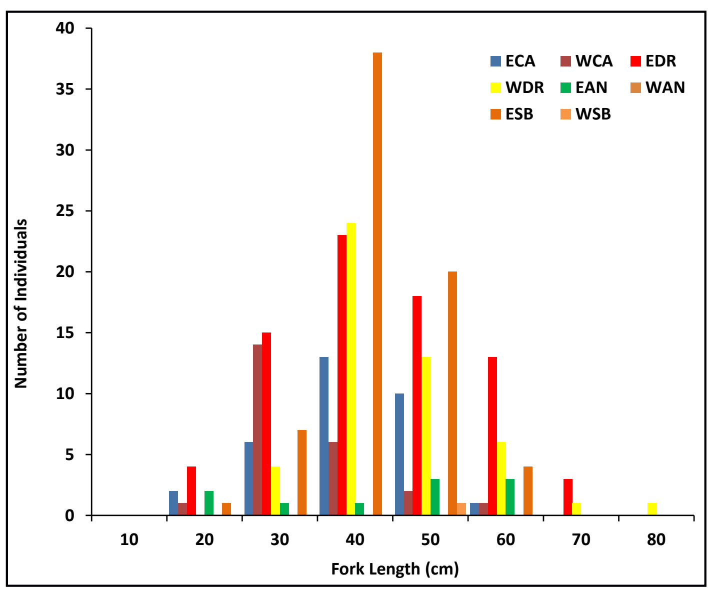


Figure 5.50. (a) Mean density and standard error and (b) length frequency distribution (FL) for red snapper, L. campechanus*, observed in ROV fish surveys, 2010-2012.*

Overall we observed 280 individuals at depths ranging from 55-106 m. *L. campechanus* density was greatest on EB deep reefs, 1.33 individuals/100 m^2. Density was 50% less (0.61 individuals/100 m^2)on WB deep reefs (Table 5.12).

L. campechanus size frequency ranged from 22-80 cm (Figure 5.50). Size structure was not correlated with any particular habitat, and in general, mean snapper size increased with depth but was not statistically significant. Size at maturity information suggests that at least 50% of the individuals observed were adults. The majority of *L. campechanus* observed were large enough for take by both fisheries.

L. campechanus were common at all hardbottom habitats at depths greater than 50 m. There was no obvious spatial pattern of density distribution (Figure 5.51).

Population estimates (Table 5.12) for *L. campechanus* indicate that this species is very abundant throughout most of the sanctuary. EB yields a higher estimate due to the presence of *L. campechanus* over softbottom habitats where effort was the lowest. Additional information is needed to provide a more accurate estimate of *L. campechanus* populations in the sanctuary, especially on softbottom habitats.

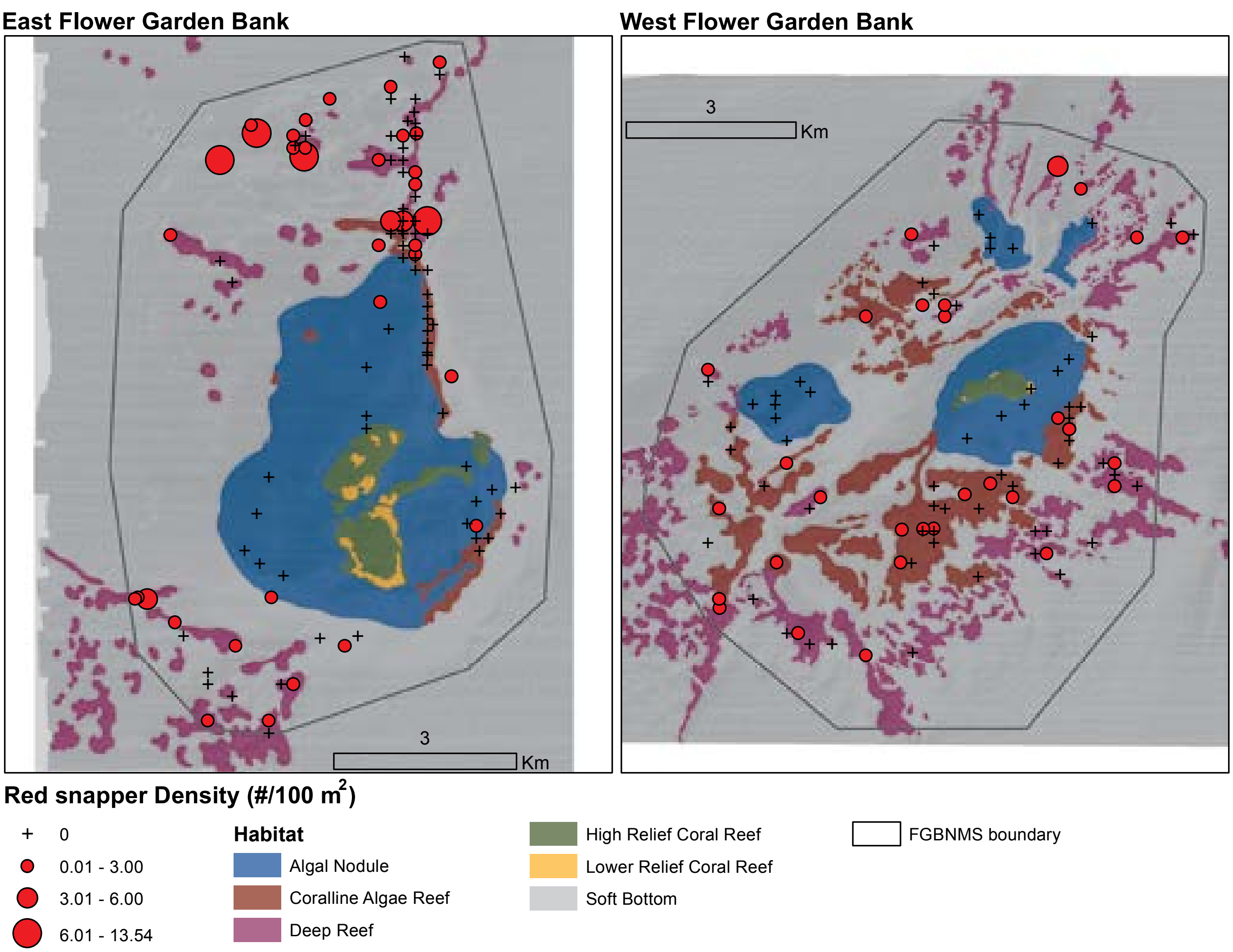


Figure 5.51. Spatial distribution of red snapper, L. campechanus, *density (#/ 100 m^2) on deep water habitats surveyed by ROV, 2010-2012.*

Rhomboplites aurorubens (vermillion snapper)

Rhomboplites aurorubens are a deep water species (40-300 m) found commonly from North Carolina to Brazil, including the Caribbean and Gulf of Mexico (Figure 5.52). They are most commonly found over rock, gravel or sand bottoms near shelf edges. They often form large schools, especially juveniles (Cervigon, 1993). *R. aurorubens* attain maximum size of 60 cm TL with maximum biomass of 3.2 kg. They are most frequently observed or captured at 35 cm TL or less (Allen, 1985). Length at maturity is not well documented but ranges from 20-23 cm (unknown length estimate) in individuals taken from Puerto Rico (Boardman and Weiler, 1980), and 26 cm TL in the Gulf of Mexico (Martinez-Andrade, 2003).

*Figure 5.52. Photo of vermillion snapper (*Rhomboplites aurorubens*). Photo: C. Cox (Mexico Beach Artificial Reef Association)*

ROV surveys yielded 116 individuals ranging in size from 12.5-22.5 cm FL (Figure 5.53). Approximately 15% of the individuals are considered adults and all would not be considered legal size for either commercial or recreational fishery.

Overall, sighting frequency of *R. aurorubens* was low, but were very abundant when present. *R. aurorubens* were only observed on deep reef habitats on EB (Figure 5.54) and coralline algal reefs on WB. Density was overwhelmingly greater on coralline algal reefs, but that was attributed to high abundance on a few surveys. The population estimate has a high degree of uncertainty (Table 5.12) and more information is needed to strengthen this estimate.

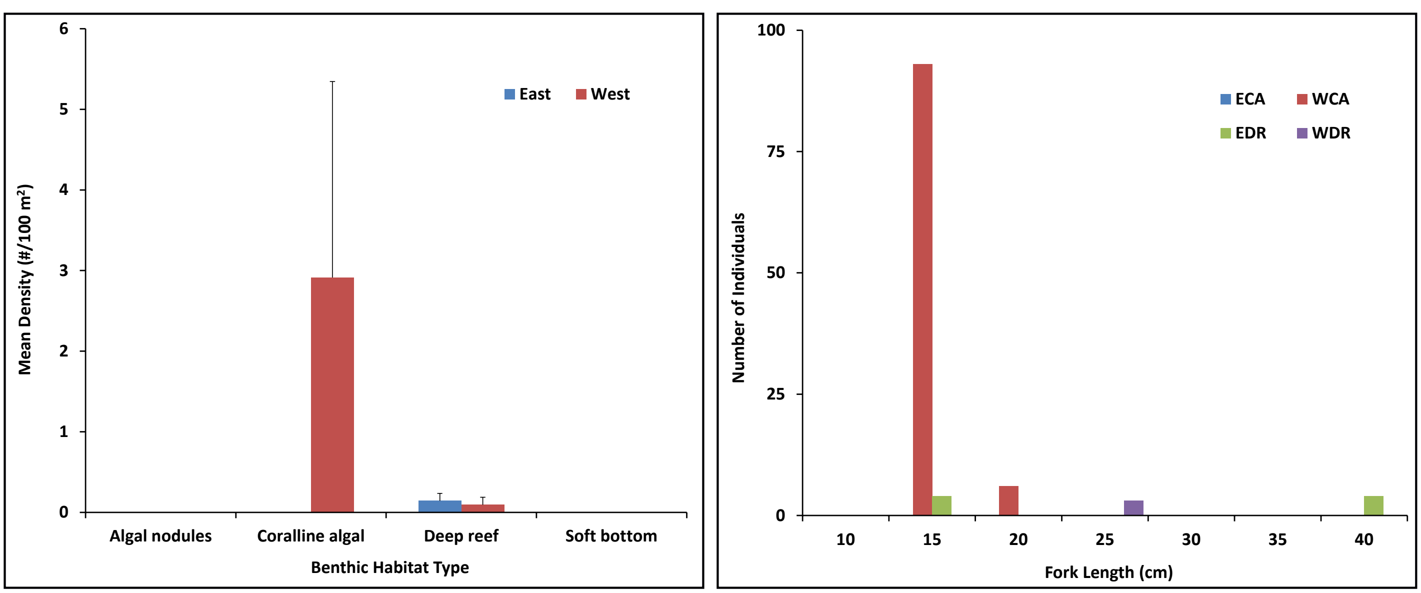


Figure 5.53. (a) Mean density and standard error and (b) length frequency distribution (FL) for R. aurorubens *observed in ROV fish surveys, 2010-2012.*

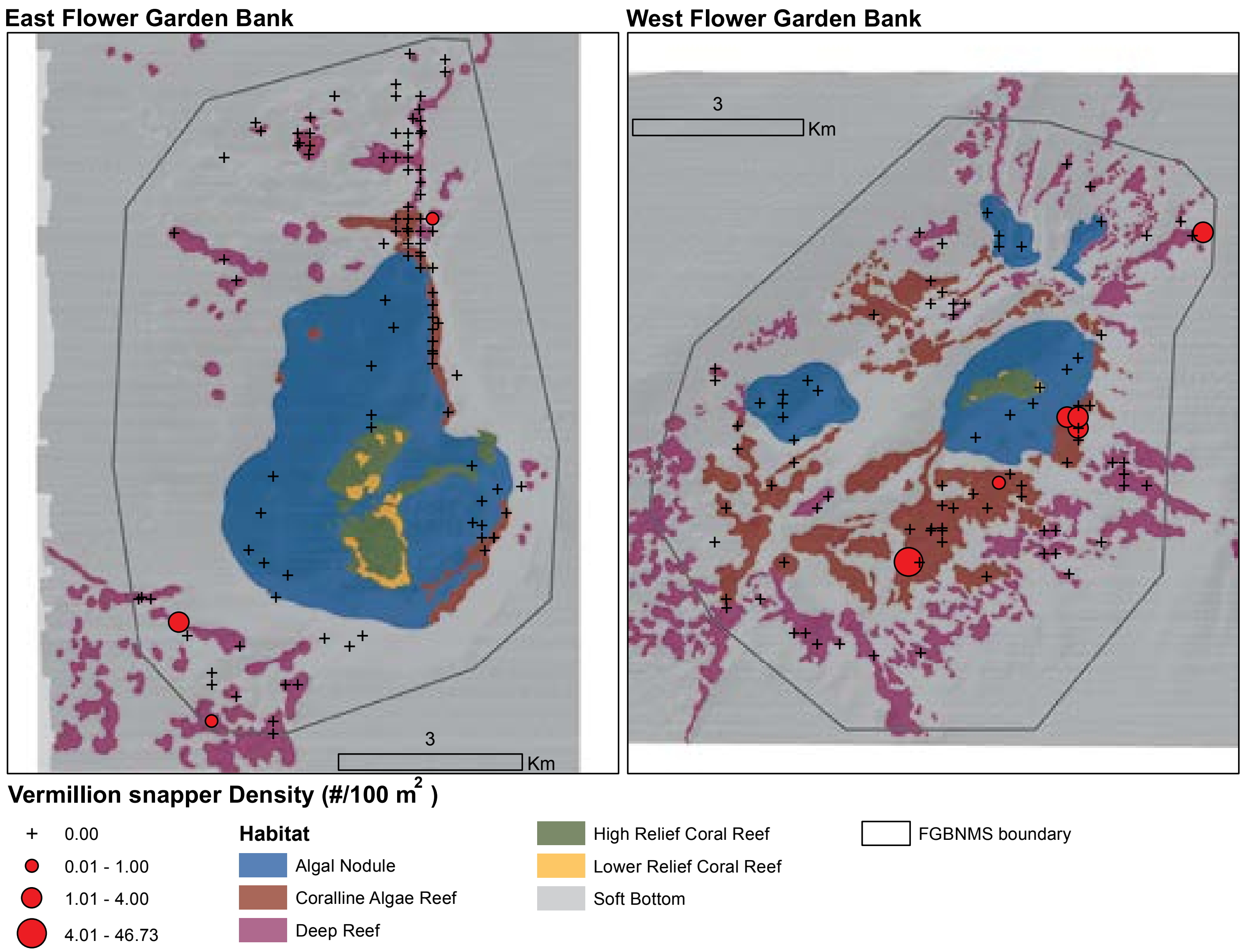


Figure 5.54. Spatial distribution of R. aurorubens *density (#/ 100 m²) on deep water habitats surveyed by ROV, 2010-2012.*

Mesophotic Communities

<u>*Pterois volitans* (red lionfish)</u>

Invasive *Pterois volitans* (red lionfish) in the sanctuary were presented in Chapter 4. *P. volitans* were observed on a few sites on East and WB coral reefs in 2011 and on 19 sites (23-42 m) in 2012 (Figure 5.55). *P. volitans* were not observed in ROV surveys until 2012. One individual was observed on a WB algal nodule transect (68 m depth; Figure 5.57). Five individuals were observed off transect at two sites (Figure 5.56): four on WB coralline algal habitat (75 m depth) and one on an EB coralline algal habitat (79 m depth). Individuals ranged in size from 15-25 cm, however, smaller fish may have been present but were not observed by the ROV.

*Figure 5.55. Invasive red lionfish (*Pterois volitans*) at FGBNMS. Photo: G. McFall (NOAA NOS/ONMS/GRNMS)*

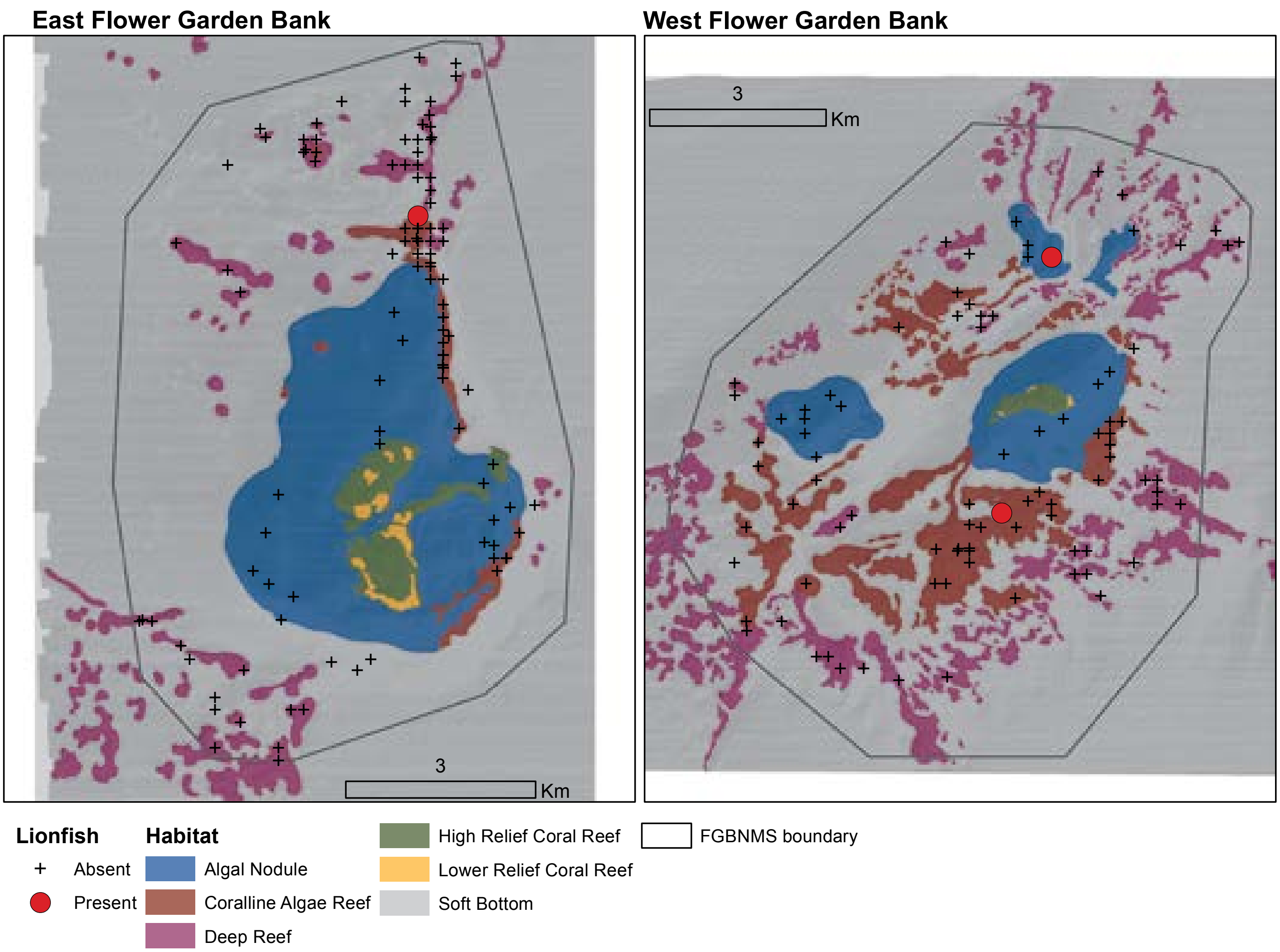


Figure 5.56. Spatial distribution of Pterois volitans *observations on deepwater habitats surveyed by ROV, 2010-2012.*

5.4. SUMMARY

- Deep coral habitats in the Gulf of Mexico are more extensive and important than previously known, particularly with respect to supporting biologically diverse faunal assemblages (Wilkinson, 2004; Roberts et al., 2006). They are threatened by a variety of activities ranging from bottom fishing to energy exploration (Rogers, 1999; Koslow et al., 2000). Over the past decade, science has demonstrated that deep corals are often extremely long-lived, slow growing animals, characteristics that make them particularly vulnerable to physical disturbance, especially from activities such as bottom trawling. Where water, current and substrate conditions are suitable, corals in these habitats can form highly complex reef-like structures, thickets or groves and there is increasing evidence that many areas of deep coral and sponge habitats function as ecologically important habitats for fish and invertebrates.
- The benthic community analysis in the deep habitats (>46 m) provide a broad assessment of deep coral habitats in the sanctuary. The data reported in this study reinforce that FGBNMS has consistently high coral cover on the coral reef caps. However , it is clear that while coral caps are dominated by scleractinians, deep reefs are dominated by numerous families of black and soft corals. Clear differences in biological communities and relative abundance of soft bottom exist between the EB and WB in areas currently defined as coralline algal reef. Likewise, there are notable differences in scleractinian community structure between the EB and WB in all five of the habitat types assessed in this study. As a result, these two distinct areas have the potential to respond differently to species specific environmental or biological stressors such as temperature change, ocean acidification, or disease.
- Fish density is strongly related to the deepwater coralline algal and deep reefs. While density and other fish community metrics are not as diverse as those seen on the coral reef, the deepwater reefs do support more diverse fish communities than soft bottom or algal nodule habitats. Overall, ROV surveys examined a small percentage of area in the sanctuary. Continued monitoring is strongly suggested to provide better fish density estimates.
- The ROV surveys provide a clear picture of habitat partitioning among the grouper and snapper families. Few species were present in large abundance on both the coral reef and deep hardbottom areas. This certainly has fishery management implications and is certainly a critical piece of biological information to assess when developing a research only area.
- We observed a robust *L. campechanus* population in the sanctuary. *L. campechanus* are an important component of the commercial and recreational fisheries in the Gulf of Mexico and may comprise a considerable portion of the recreational fishing harvest in the sanctuary; however, no information exists on the levels of harvest by recreational fishing within the sanctuary or vicinity. *M. phenax* were also very abundant and their status should be monitored to assess any impacts from fishing.
- Other groupers had much lower density and continued monitoring should help understand the variability of these populations. Long term monitoring associated with a no-take area would provide important information about the populations of these species and fisheries impacts.
- Lionfish have been recently recorded on deep habitats in the NW Gulf of Mexico (Nuttall et al., In Press) and we document the first observations on deep habitats within the sanctuary. These data provide a population baseline as their continued recruitment into the sanctuary is inevitable. Continued monitoring of these habitats will also provide recruitment or reproductive information and may be useful to evaluate natural control by other piscivorous species, (e.g., groupers).
- Lastly, documentation of fishing effort is critically needed to accompany this biological baseline. Fishing pressure is present in the sanctuary, but estimates of fishing effort and harvest in the sanctuary are limited and not quantifiable.

5.5. REFERENCES

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Chapter 6
Mapping Fish Densities Using Fishery Acoustics

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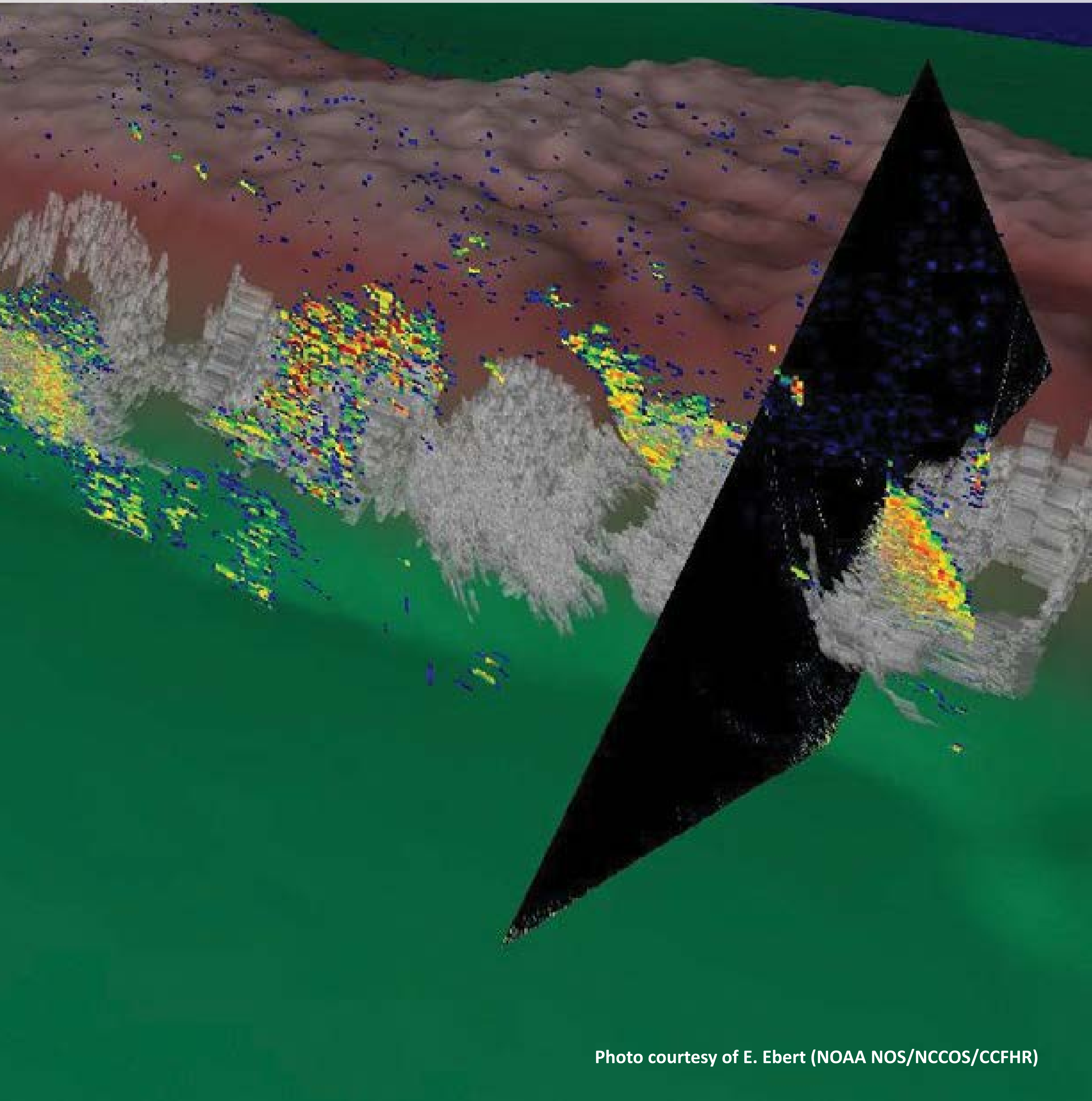

Photo courtesy of E. Ebert (NOAA NOS/NCCOS/CCFHR)

6.1. BACKGROUND

*Photo of Atlantic creolefish (*Paranthias furcifer*) in Flower Garden Banks National Marine Sanctuary (FGBNMS). Photo: M. Whitfield (UNCW)*

Fishery acoustics (fishery sonar, fish finders) are a non-invasive and non-destructive means of using high-frequency sound to survey and detect marine organisms. The fishery acoustic survey was designed to sample fish and other marine organisms throughout the water column, over the full depth range (17-250 m) and across all habitat types within the sanctuary boundaries. Acquisition parameters used in these surveys allowed for the detection of organisms as near as 25 cm from the sea floor to within 5 m of the surface (Appendix F). Fish were detected over a broad range of estimated sizes throughout the water column (6 cm to >150 cm), and were detected throughout the water column over all habitat types (Figure 6.1). The geomorphology and complexity of some habitat structures, such as the high relief reefs on the coral reef and deep reef habitats, did occlude some fishes that resided within the reef matrix. In these cases, detectability may be reduced when compared to unstructured habitat like sand and soft mud; but detectability was assumed to be similar within a habitat type. The use of acoustics complements the visual assessments described in previous chapters and allows for interpretation of the spatial distribution of fishes across the seascape.

6.2. METHODS

6.2.1. Fish Abundance Mapping - Splitbeam Echosounder

The splitbeam echosounder system (SBES) detects fish and other organisms in the water column by rapidly transmitting acoustic pulses (pings) that reflect off objects of differing density than the surrounding water. The primary contributor to the acoustic reflection, or echo, from fish is the swimbladder, which is used for buoyancy control (Simmonds and MacLennan, 2005). SBES data on fish occupying the water column were collected at night during each visual survey mission (remotely operated vehicle, or ROV; shallow and deep-technical diving) in 2010, 2011 and 2012. A single-frequency Simrad EK-60 splitbeam echsounder operating at 120 kHz was used for surveys onboard the R/V *Manta*. The transducer was deployed overboard attached to a rigid pole affixed to the side hull of the vessel at a depth of 1.5 m below the surface. During a single mission in 2011, a three-frequency (38, 120, 200 kHz) Simrad EK-60 splitbeam echosounder system on board the NOAA Ship *Nancy Foster* was used. The transducers on the *Nancy Foster* are hull mounted 3.4 m below the surface. The beam width of the 120 kHz SBES was 7° for both platforms, resulting in a swath under the vessel of about 12% water depth.

Vessel survey speed was about 7 knots (kts; 3.6 m/s) and ship position was recorded using differential GPS or WAAS GPS. The location of the transducer relative to GPS antenna was measured precisely to calculate the horizontal offset for real-world coordinates of individual fish relative to the ship's position and the underlying seafloor habitats.

Fishery Acoustics

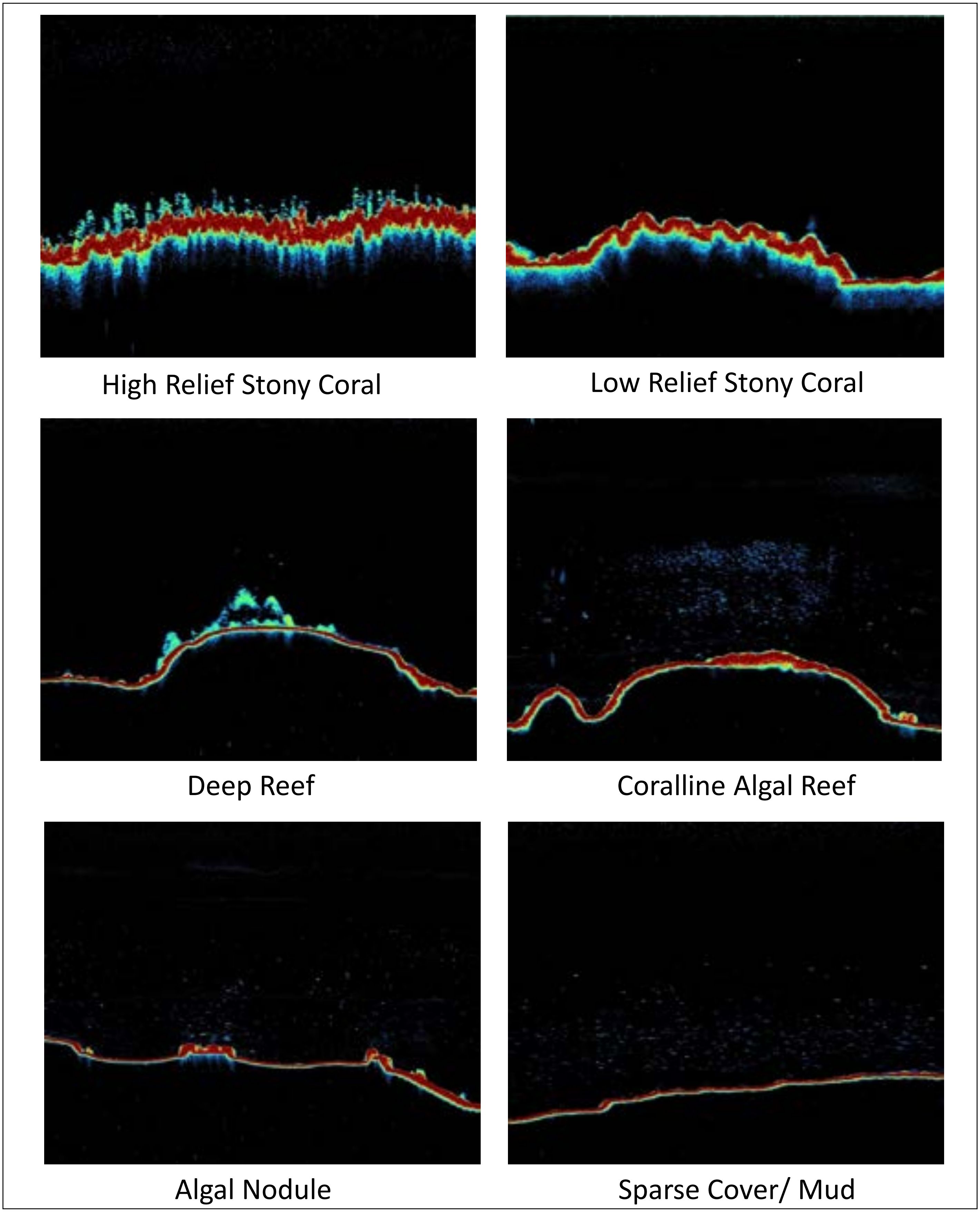

Figure 6.1. Examples of fishery acoustic echograms over the six major habitat zones on East Bank (EB) and West Bank (WB) of Flower Garden Banks National Marine Sanctuary (FGBNMS). Color scale represents acoustic backscatter strength from blue (weak acoustic backscatter, small organisms) to red (strong acoustic backscatter, seafloor and large organisms).

Data were acquired using Simrad ER-60 software using pulse transmission and receive parameters that optimize detection of fishes close to bottom or habitat structure across a range of depths while minimizing noise and interference. Acquisition parameters for the 120 kHz systems on each survey platform were identical (Appendix F).

6.2.2. Fishery Acoustics Survey Design

The survey design was a set of discrete rectangular polygons placed to cover the coral reef on East Bank (EB; 3.7 km^2) and West Bank (WB; about 2 km^2), with seven additional polygons (about 1.5 km^2 each) distributed over deeper habitats on each bank. The location of the additional polygons was designed to coincide with remotely operated vehicle (ROV) and visual surveys to cross-reference acoustic and visual observations. Transects within each survey polygon were spaced 50 m apart over the coral reef and 100 m apart over other habitats.

6.2.3. Acoustic Data Processing

The SBES data were processed to detect individual fishes using Echoview software (version 5.3, Myriax, Pty, Ltd. http://www.echoview.com). Similar protocols were used to analyze the data from both the R/V *Manta* and NOAA Ship *Nancy Foster*. The seafloor was delineated to remove acoustic noise, interference and air bubbles. Faint echoes likely representing plankton and other non-fish targets were masked or eliminated from the data. Vessel speed (ca 7 kts) and rapid ping rate (3-8 pulses per second) resulted in multiple, sequential echoes from each individual fish. These sequential echoes were grouped using a target tracking algorithm and stored in a database. Each fish was assigned a central geographic position referenced to the vessel's positioning system, depth below water surface (corrected for transducer depth below sea surface), depth above bottom, and an average target strength or acoustic size of the fish measured in decibels (dB). A generalized acoustic size to fish size relationship was used to convert the target strength into an approximate measure of fish length (Appendix F). Fish exceeding -50 dB or a length of about 6 cm were retained in the database.

6.2.4. Calculating and Mapping Fish Densities

Data along the survey transect were binned into 100 m intervals. Fish density was calculated for each interval, accounting for the increasing detection of individuals as the transducer beam increases with depth, standardized to a 1-m swath (Appendix F). Final units of fish density were fish per 100 m^2 and exported as a point shapefile in ArcGIS with the centroid of the interval used as the geographic position. Densities were plotted as scaled points proportional to magnitude, with zero density (absence of detected fish) omitted.

It is not possible to differentiate species from acoustic signatures using a single SBES frequency. Instead, individual fish were grouped into three size classes representing ecological or fishery species groups. Small fish, less than 11 cm total length (TL), likely represented small reef resident species and small pelagic planktivores. Medium fish, between 12 and 28 cm, included larger reef residents and juvenile or small adults of targeted fishery species. Large fish, greater than 29 cm, were likely comprised of many fishery important species (Serranidae and Lutjanidae) and other species. The divisions in size classes were selected based on qualitative interpretation of length frequency data from fish visual census data in the region (Caldow et al., 2009; this volume).

We produced maps from the acoustic surveys to show broad patterns of fish distributions at fine spatial resolution for medium and large fish size classes. Because the surveys were conducted repeatedly over the course of the three year study, and in some cases several surveys were conducted over the coral reef in each year, we were able to make some inferences on possible biological "hotspots" in fish density over habitat types within the sanctuary. We created composite density maps by interpolating large fish densities (using kriging with default model parameters in Geostatistical Analyst in ArcGIS 10.1) from all surveys conducted on the coral reef in WB and EB from 2010-2012.

Fishery Acoustics

Fish density of each size class was mapped in relation to habitat type to examine the relative abundance of fish across the seascape of each bank. Statistical comparisons of large fish densities were made between EB and WB and between habitat types using a Tukey's HSD test. Lastly, we compared densities of large fishes on the coral reef habitats on EB and WB with large fish densities found in other stony coral habitats in the Florida Keys National Marine Sanctuary, Puerto Rico and the U.S. Virgin Islands (USVI). Data acquisition parameters, calibration protocols and data processing algorithms used in these studies were identical to the methods used here. For comparison, we pooled the densities of large fish over both high and low relief stony corals in EB and WB as well as large fish densities over stony corals in the other U.S. territories.

6.3. RESULTS AND DISCUSSION

6.3.1. Fishery Acoustic Survey Effort

Acoustic surveys were conducted at both EB and WB during the night in each year of the survey missions (2010, 2011 and 2012). During each mission, the EB and WB coral reef were surveyed along with a selection of polygons distributed over deeper habitats. Total survey effort and area covered varied between years due to mission length and vessel platform used, with a notably lower effort in 2010 due to the cancellation of an upper mesophotic coral reef diving mission resulting from the Deep Water Horizon accident (Table 6.1). Regardless, relative survey effort between habitat types was similar across years (Figure 6.2).

Table 6.1. Fishery acoustic survey effort at FGBNMS. DAS = operating days at sea. Area covered and linear miles surveyed are calculated from survey polygons.

Project Year	Missions (DAS)	Coral Reef Surveys	Total Area Covered Across Banks (km^2)	Line Miles Surveyed (km)
2010	2 (8)	2	27.8	267
2011	3 (24)	3	66.2	717
2012	3 (20)	3	48.6	634

6.3.2. Distribution of Fishes in the Water Column

Depth distribution of fish in the water column varied significantly by habitat types and size class. Over structured habitats such as high and low relief coral reef, coralline algal, and algal nodules, fish were more closely associated with the bottom habitat (Figures 6.3 and 6.4). Over 90% of fish were found within 10 m of the coral reef. Fish density surveyed on the WB by Wilson et al. (2003) using similar echosounder technology, showed similar densities found in close proximity to the coral reef (referred to as "upper terrace", Wilson et al. 2003). For coralline algal reefs and algal nodules, over 80% of the fish were found within 15 m of the seafloor (Figures 6.3 and 6.4). These patterns were consistent over both banks. Wilson et al. (2003) also observed similar patterns over the coralline algal reefs (referred as "middle terrace"), with highest densities observed in the 60-80 m depth strata. Vertical location of fishes over the sparse cover/mud and deep reefs were distributed more broadly through the water column (Figures 6.3 and 6.4). On WB, fish were evenly distributed throughout the water column on mud and deep reef habitats; whereas 80% of the fish on EB in these habitats were found within 30 m of the seafloor.

Fish assemblage of various sizes at FGBNMS. Photo: M. Winfield (UNCW)

The vertical position of fish within the water column was strongly related to fish size. Small fishes were distributed throughout the water column, especially over deep habitats. In contrast, larger fishes were detected closer to the seafloor on stony coral and deep structured habitats. The small fish distributed through the water column are likely small planktivorous fishes that may be following vertically migrating zooplankton in deep waters (Bollens et al., 1992; Checkley et al., 1992). Likewise, over deep structured habitats of coralline algal reefs on WB (Figure 6.5) and algal nodule and sparse cover/mud on EB (Figure 6.6) there were several medium and large fish detected mid-water column. These larger pelagic fishes may also be associated with deep scattering layers of fish and plankton that were observed simultaneously in the sonar echogram (e.g., Coralline Algal Reef in Figure 6.1).

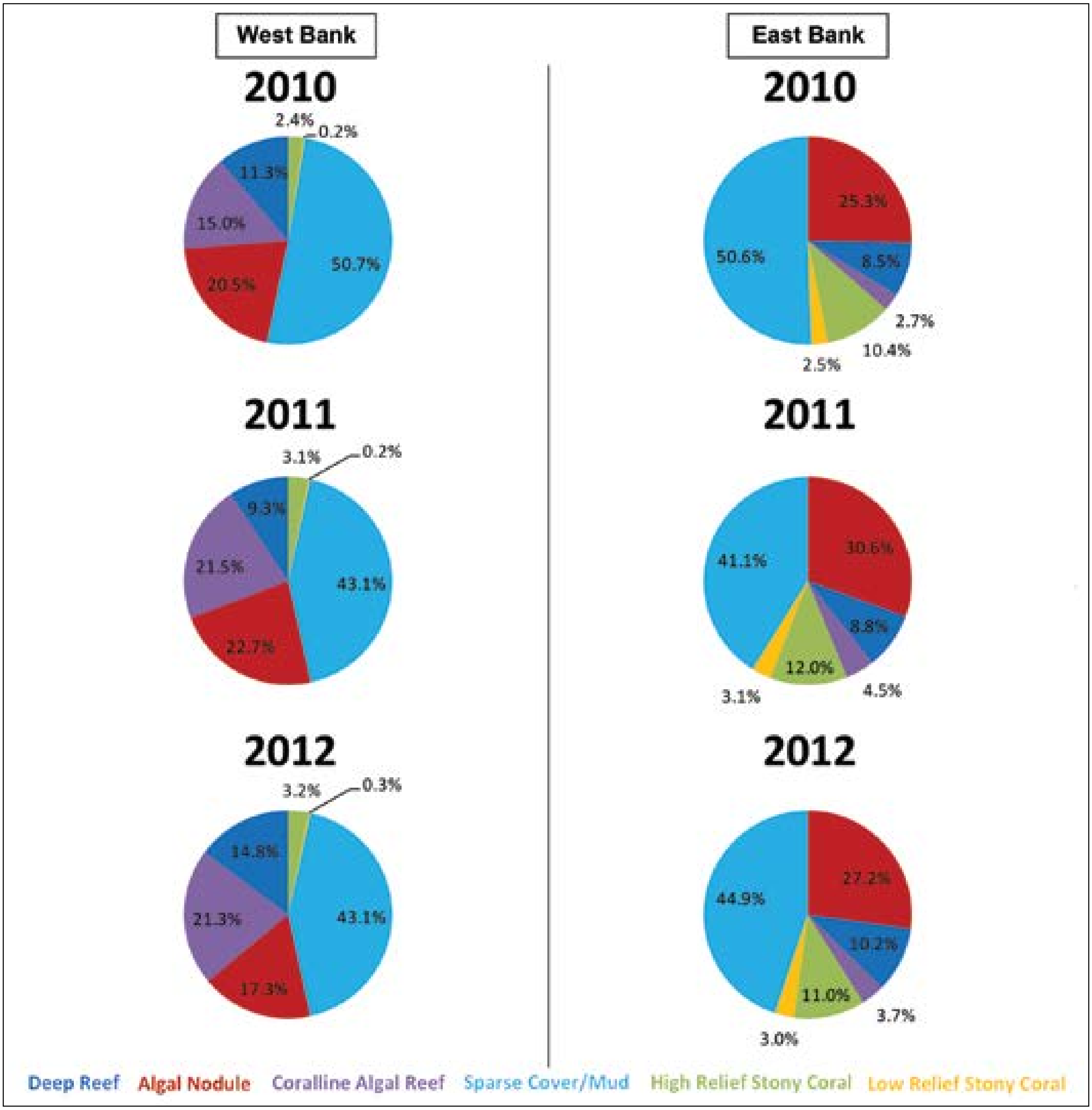

Figure 6.2. Proportion of fishery acoustic surveys by habitat zone, year, and bank.

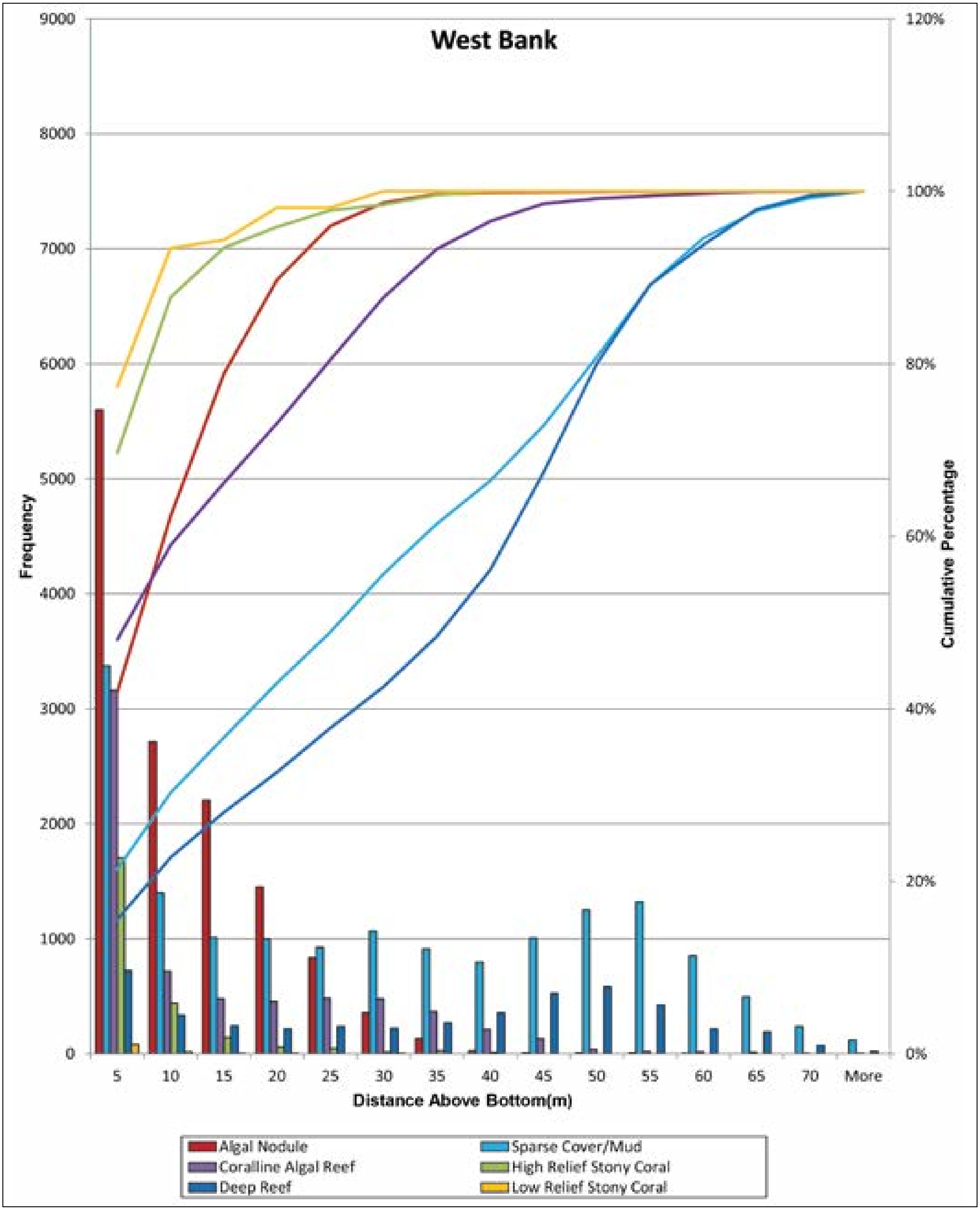

Figure 6.3. Distribution of all sizes of fish relative to distance above bottom across habitat zones on WB. Bars represent total number of fish in each 5 m distance bin. Lines represent cumulative frequencies with increasing distance above bottom.

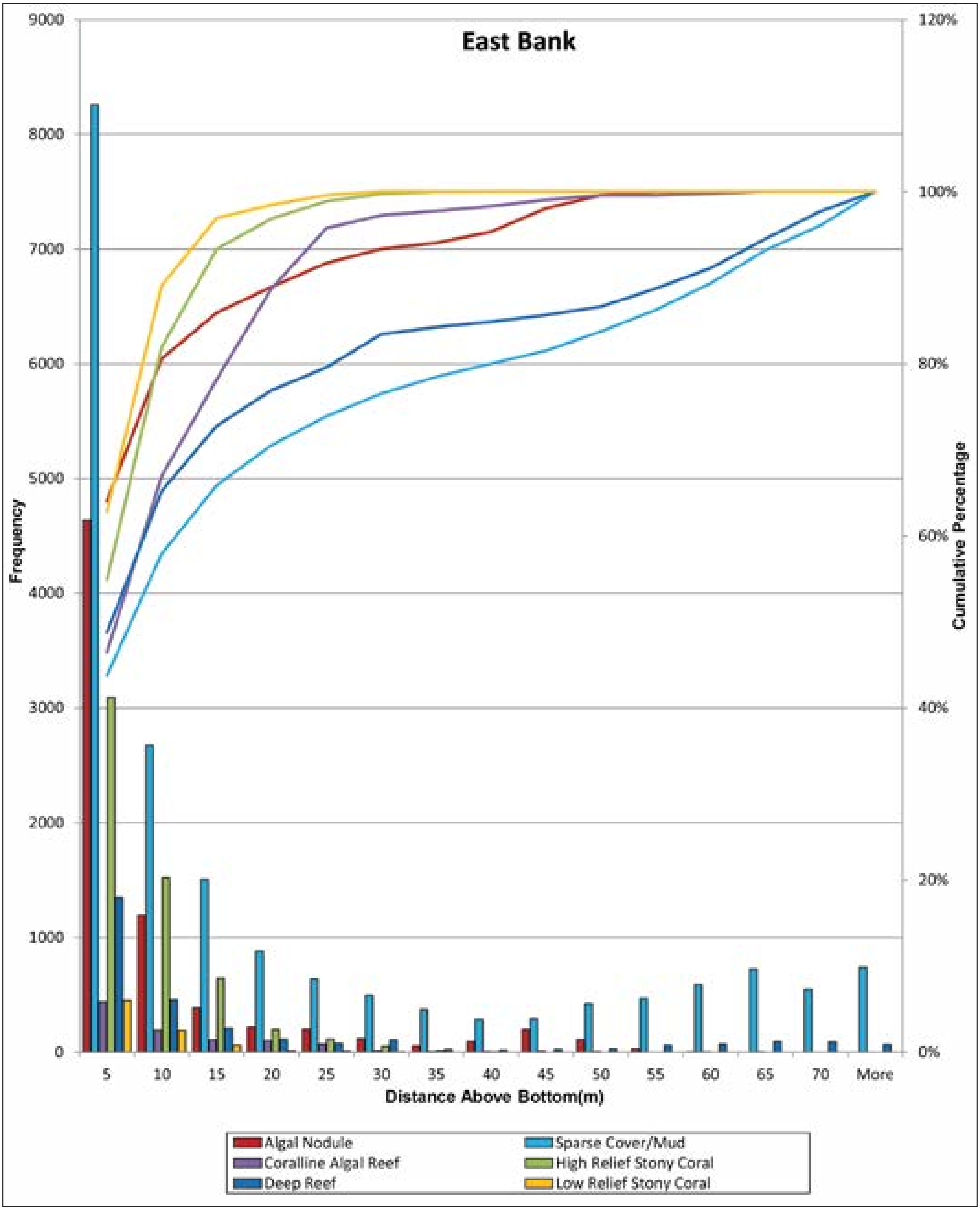

Figure 6.4. Distribution of all sizes of fish relative to distance above bottom across habitat zones on EB. Bars represent total number of fish in each 5 m distance bin. Lines represent cumulative frequencies with increasing distance above bottom.

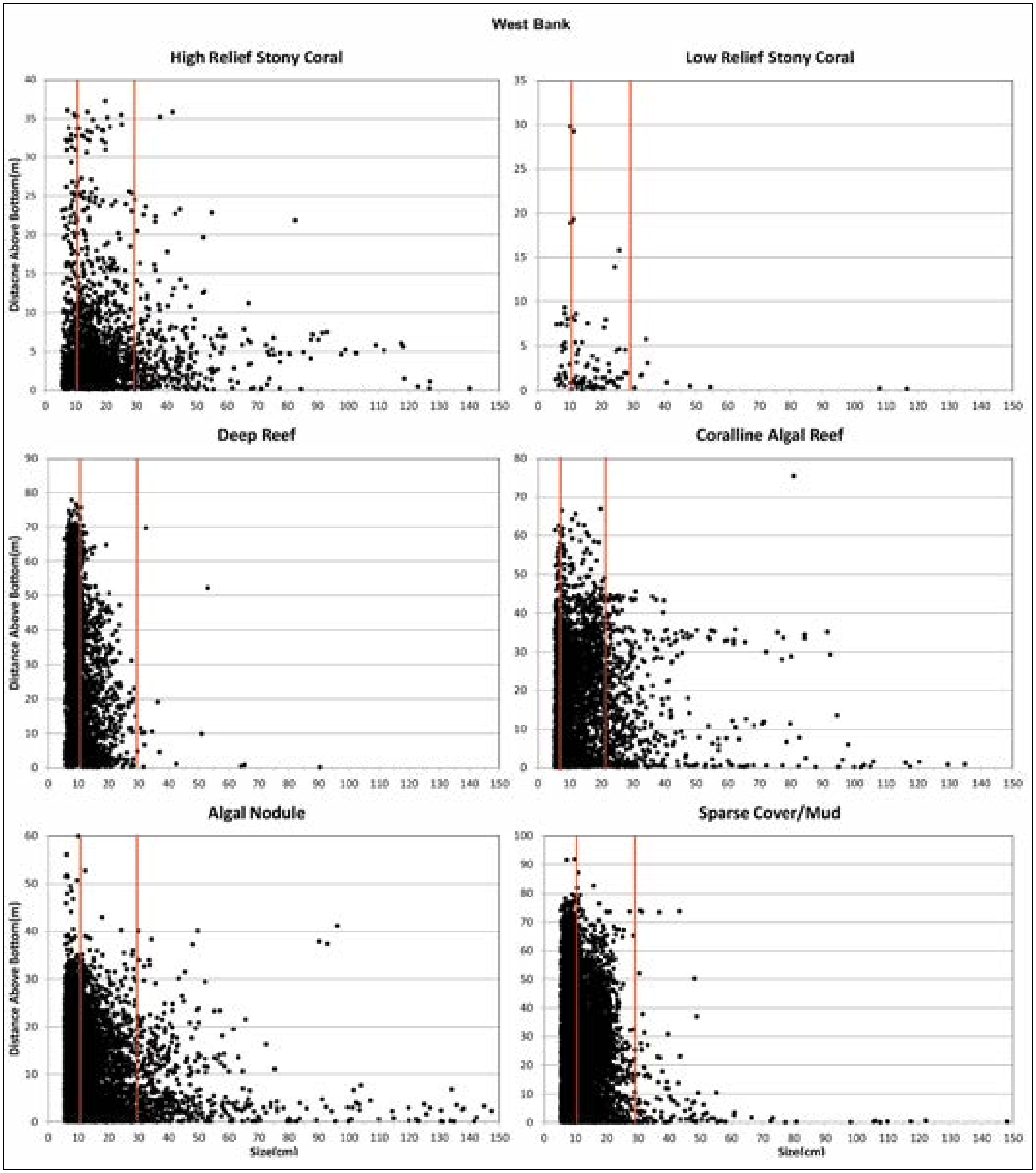

Figure 6.5. Distance above seafloor for individual fish over each of the habitat zones on WB. Red vertical lines are breaks in size classes: small (<11 cm), medium (11-29 cm), large (>29 cm).

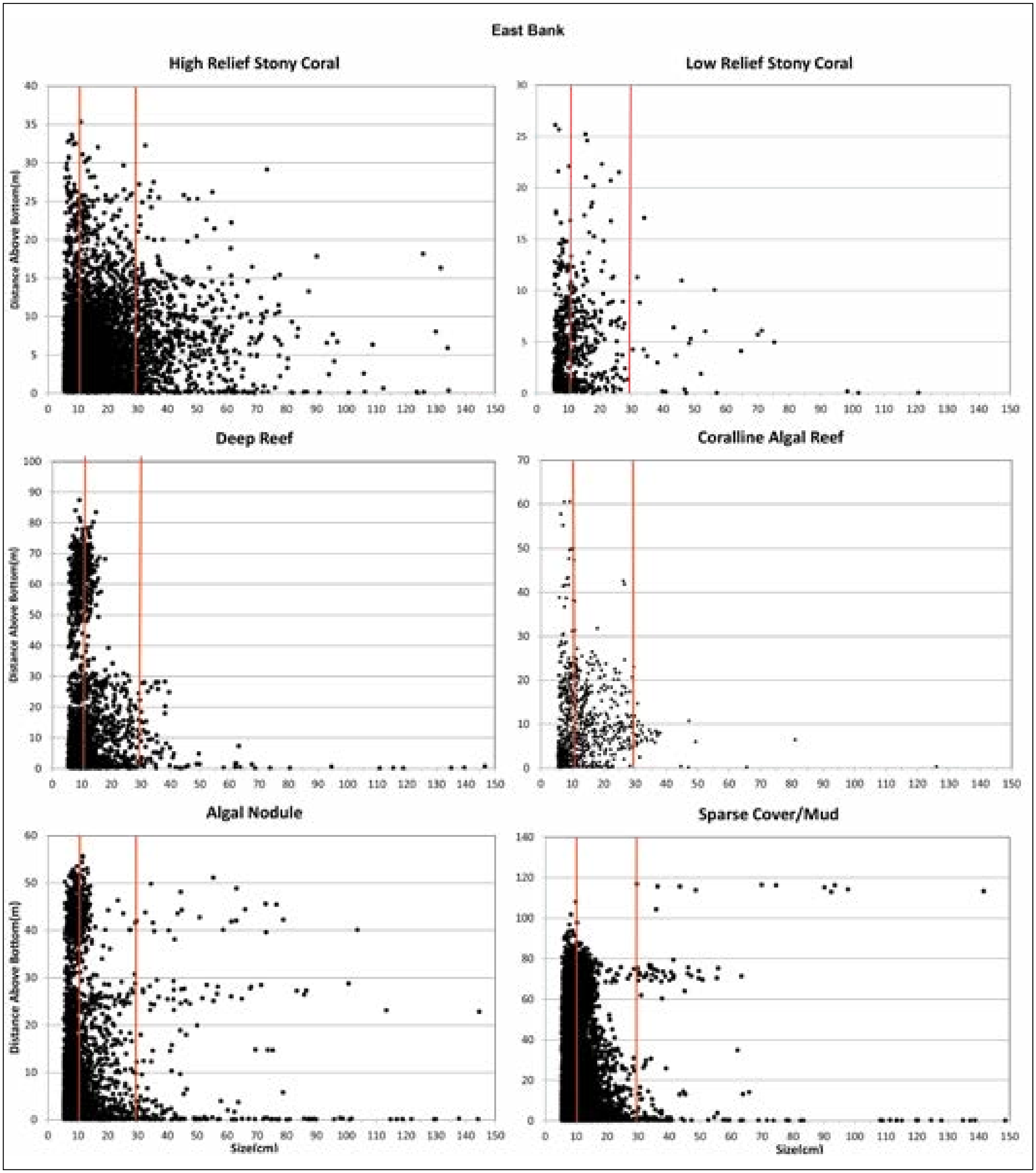

Figure 6.6. Distance above seafloor for individual fish over each of the habitat zones on EB. Red vertical lines are breaks in size classes: small (<11 cm), medium (11-29 cm), large (>29 cm).

6.3.3. Fish Densities Across Habitats and Banks

Due to the lack of species identification from acoustic surveys, it is difficult to infer patterns of species distribution that may be related to bottom habitats or oceanographic features. Instead, the following density maps are classified by size classes, aggregated over species and groups. Small fishes (<11 cm) were found throughout the water column and throughout the sanctuary, likely comprised of pelagic (e.g., Inermiidae) and mesopelagic (e.g., Gonostomatidae) planktivores, small schooling fishes associated with reef structure (e.g., *Chromis* species), and smaller reef resident species (*Stegastes* species and Labridae). Due to their ubiquitous nature, fishes in the small size class were excluded from mapping and analysis (except as an example, see Figure 6.9 and Figure 6.17). Medium fish (between 11 and 29 cm) were likely a mix of reef residents (e.g., Scaridae) and juvenile or sub-adult forms of fishery-important species (e.g., Serranidae, Lutjanidae, Carangidae). Both individual fish and fish schools were detected in this size group. A notable contribution were very large schools on the coral reef (Figures 6.7 and 6.8). Finally, large fish (>29 cm) likely included large reef residents (Scaridae and Labridae) as well as pelagic transients (Carangidae and Sphyraenidae) and demersal species (Serranidae and Lutjanidae). The large size class is of greatest interest to managers as fishes in this size class are likely to be targeted by recreational and commercial fishing in the sanctuary and surrounding waters.

Medium and large fish size class maps are presented by bank and season in 2010, 2011 and 2012 (Figures 6.11-6.15 and 6.18-6.23). Fish density is shown with shaded circles proportional to densities using a natural breaks algorithm (ArcGIS 10.1). Each survey polygon was systematically surveyed. To simplify the maps visually, areas

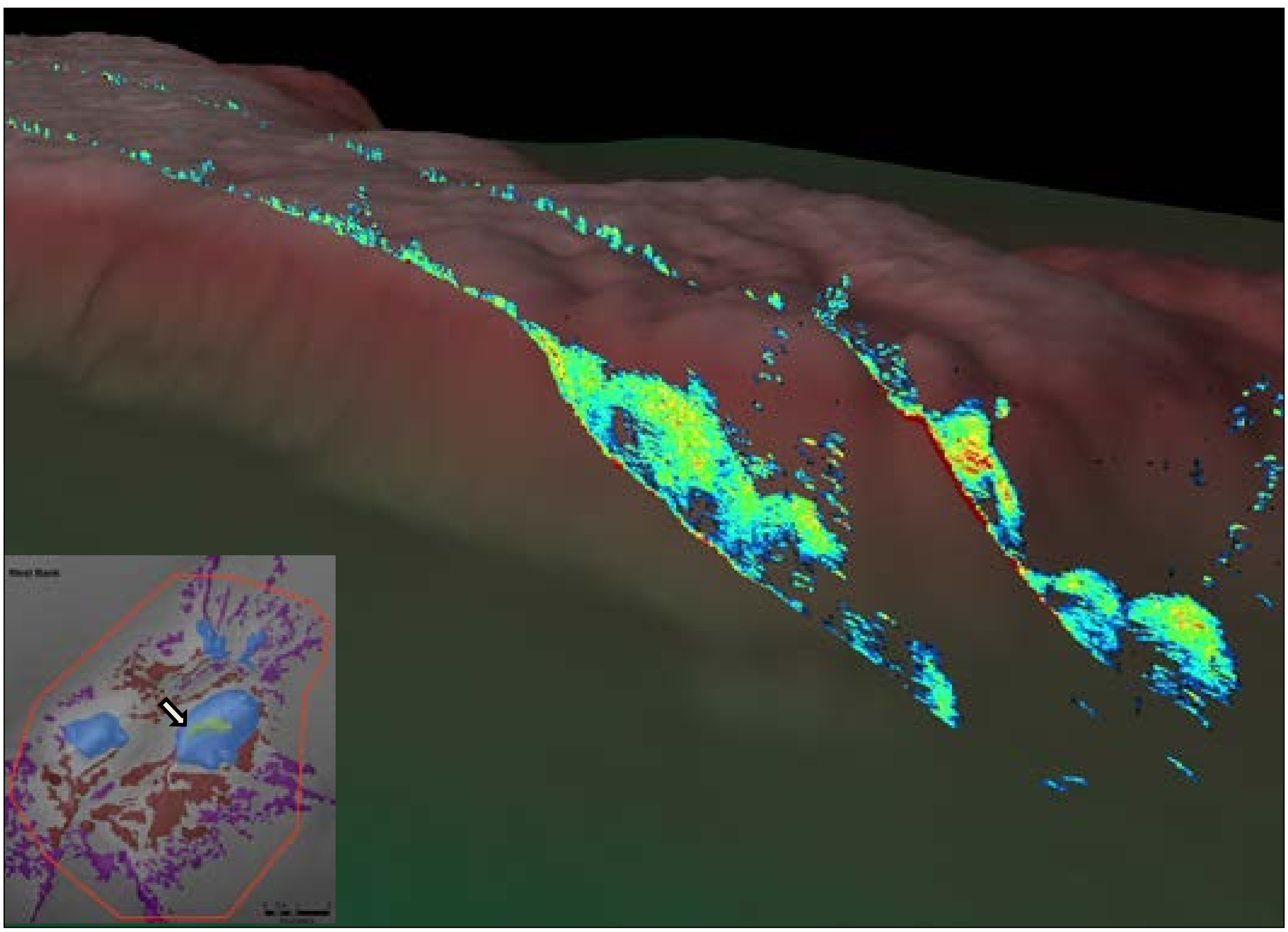

Figure 6.7. Perspective image of large fish schools near northwest edge of coral reef in WB. School appears as green-yellow-red concentrated acoustic returns along two survey lines. Bathymetry of WB in perspective is exaggerated vertically by three times.

in survey polygons without shaded circles were sampled but contained zero density for the size class. In 2012, an electrical problem in the 120 kHz transducer resulted in a decrease in fish detection. The reduction in detections was consistent across all surveys in 2012, such that comparisons of the relative spatial distributions of densities between years were still appropriate (e.g., location of density hotspots).

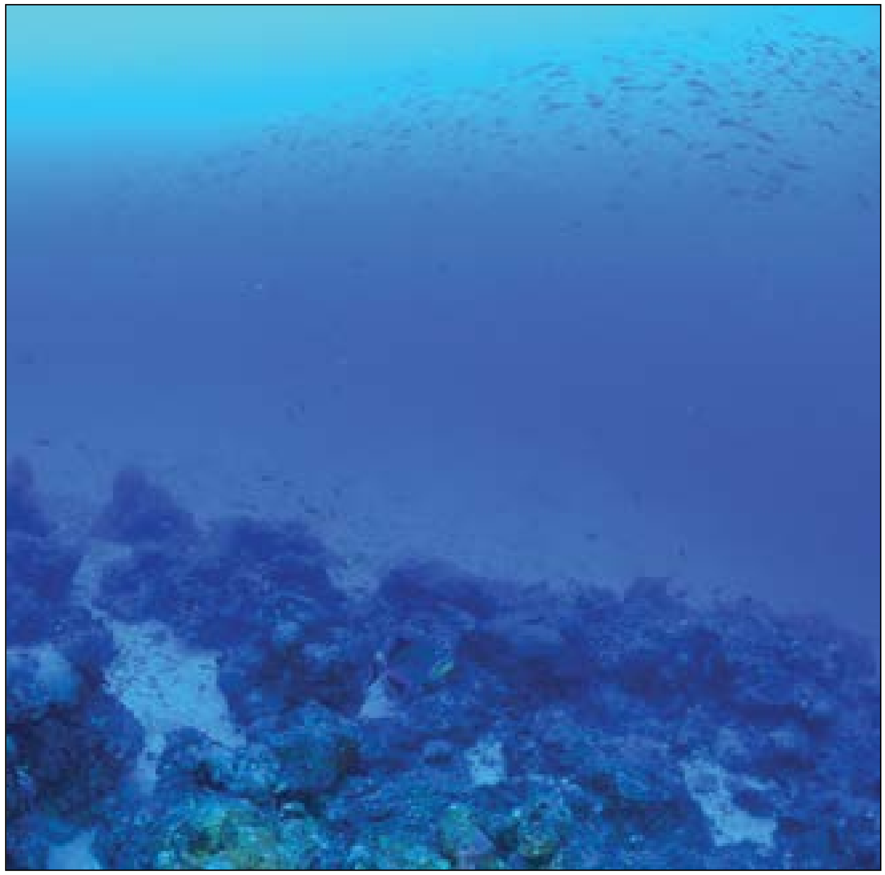

*Figure 6.8. Large school of Atlantic creolefish (*Paranthias furcifer*) over low relief stony coral on coral cap. Photo: M. Winfield (UNCW)*

Acoustic Fish Densities – West Bank

Small fish were broadly distributed across the WB coral reef and deep structured habitats, but less so over sparse cover/mud in 2010 (Figure 6.9). Higher densities of medium and large fish were found on the coral reef than on other habitats in the WB in 2010 (Figures 6.10 and 6.11). Sparsely distributed patches of high large fish densities were also found on other structured habitats such as coralline algal reefs, algal nodules and deep coral zones in year 2010, 2011, and 2012 (Figures 6.11, 6.13 and 6.15, respectively). There was a noticeable absence of large fish over soft sediment on WB.

High densities of large fishes were found in the northwestern region of the coral reef during the 2010 and 2011 surveys and during the September survey in 2012, particularly at the margins of the high and low relief coral habitat (Figures 6.11, 6.13 and 6.15). The distribution of medium sized fish was similar to that of large fishes on the coral cap, with high densities found in the northwestern region of the coral reef during many of the surveys 2010-2012(Figures 6.10, 6.12 and 6.14). It is likely that the high densities of medium sized fishes were comprised of large schools of Atlantic creolefish (*Paranthias furcifer*) and creole wrasse (*Clepticus parrae*), as those species were observed frequently during the diver surveys (Table 4.3 and Figures 6.7 and 6.8). Based on the composite map (2010-2012) of large fish densities, higher densities were observed in the northwestern region of the coral reef (Figure 6.16).

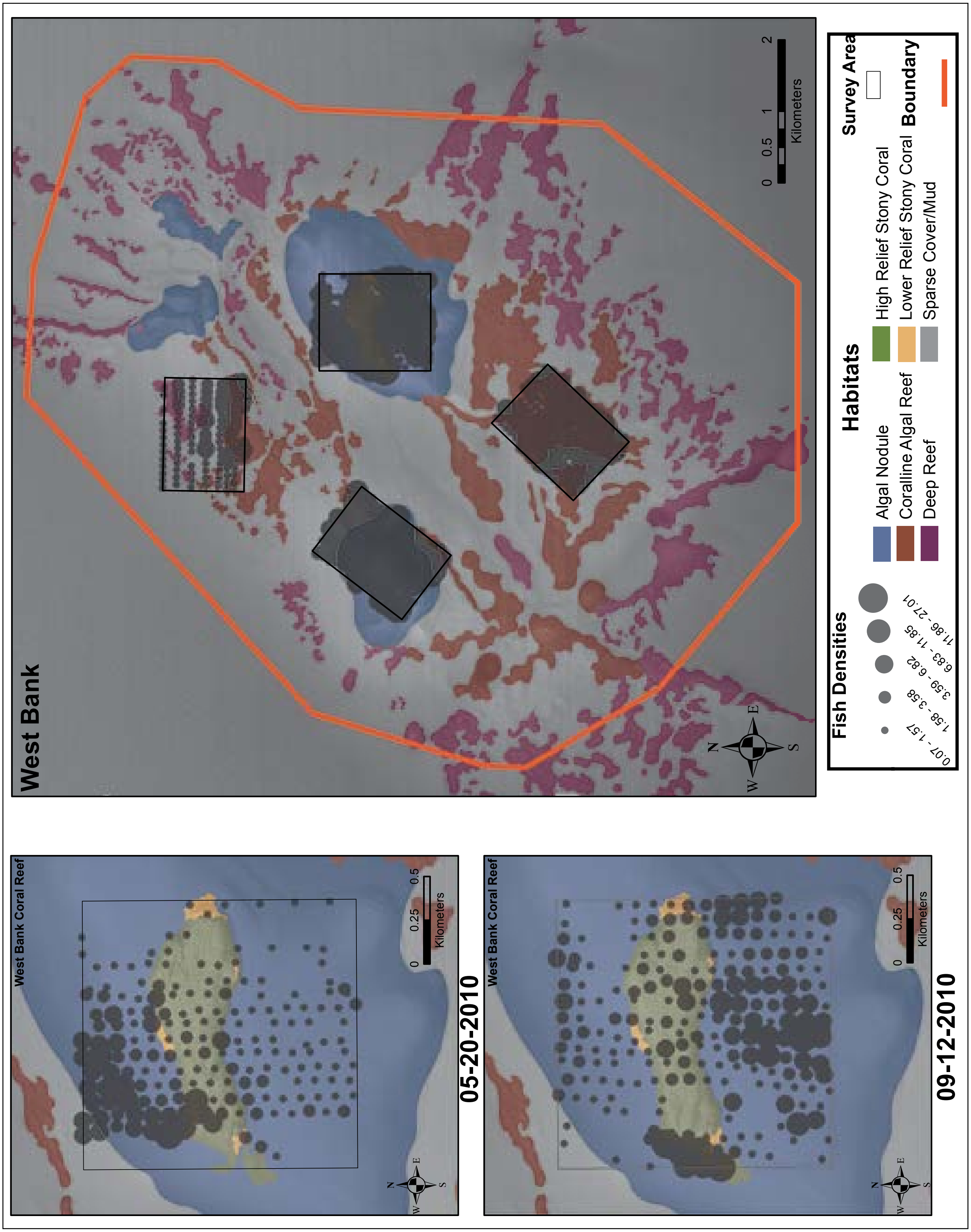

Figure 6.9. Density distribution for small fish (<11 cm) on Coral Reef Zone (left) and WB (right) for all surveys conducted in 2010. Symbols are proportional to density and differ between size classes in following maps.

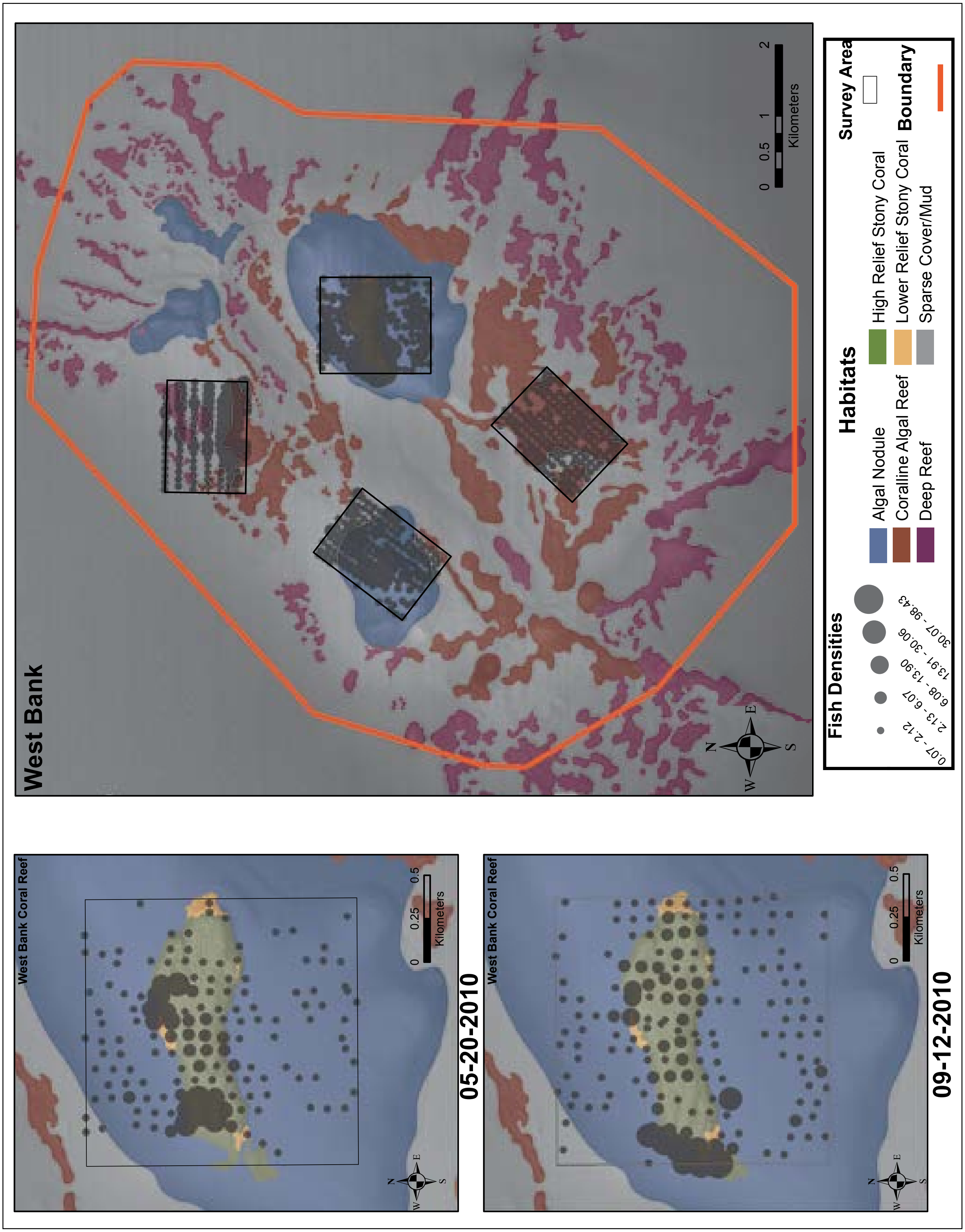

Figure 6.10. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (left) and WB (right) for all surveys conducted in 2010.

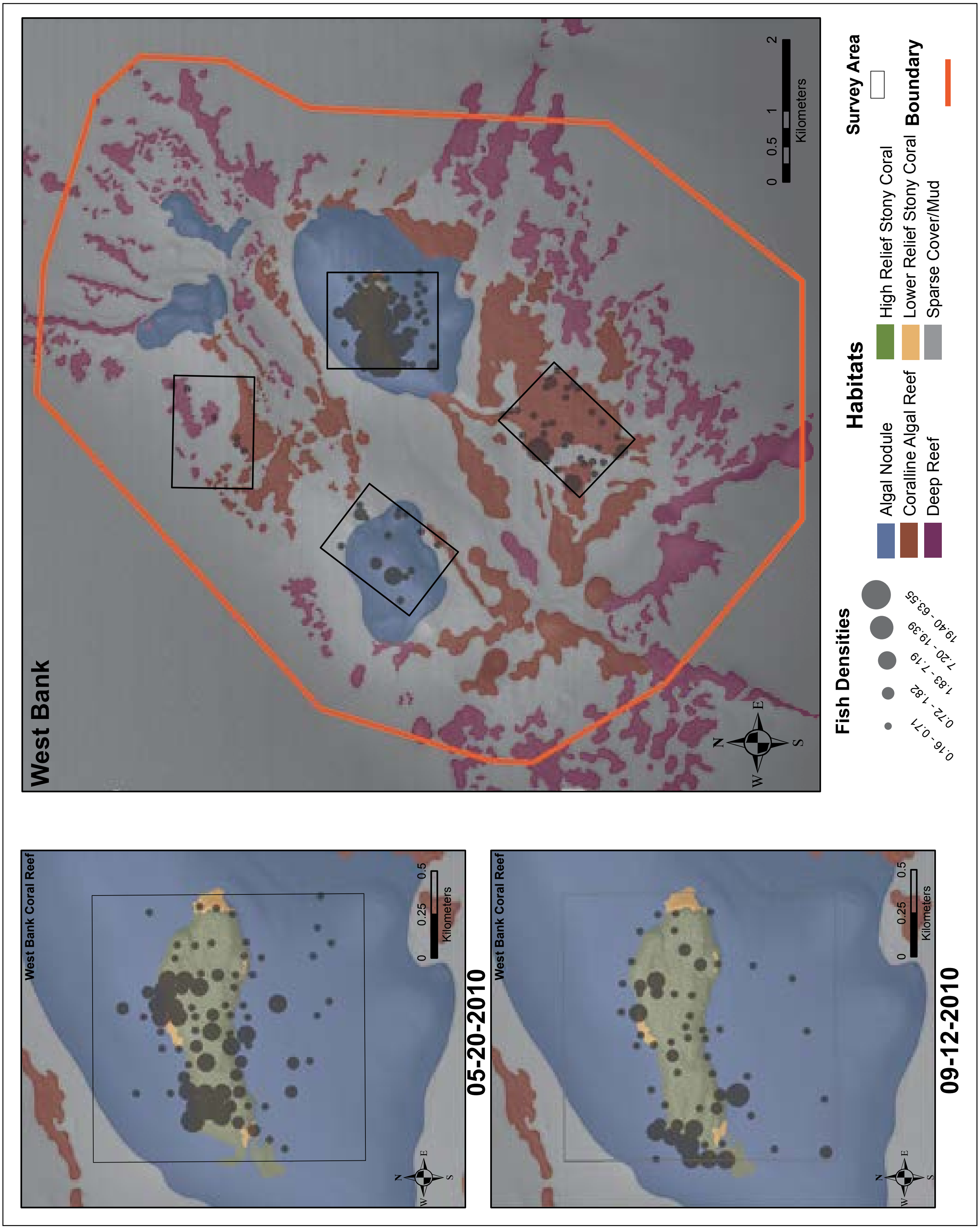

Figure 6.11. Density maps for large fish (>29 cm) for the Coral Reef Zone (left) and WB (right) for all surveys conducted in 2010.

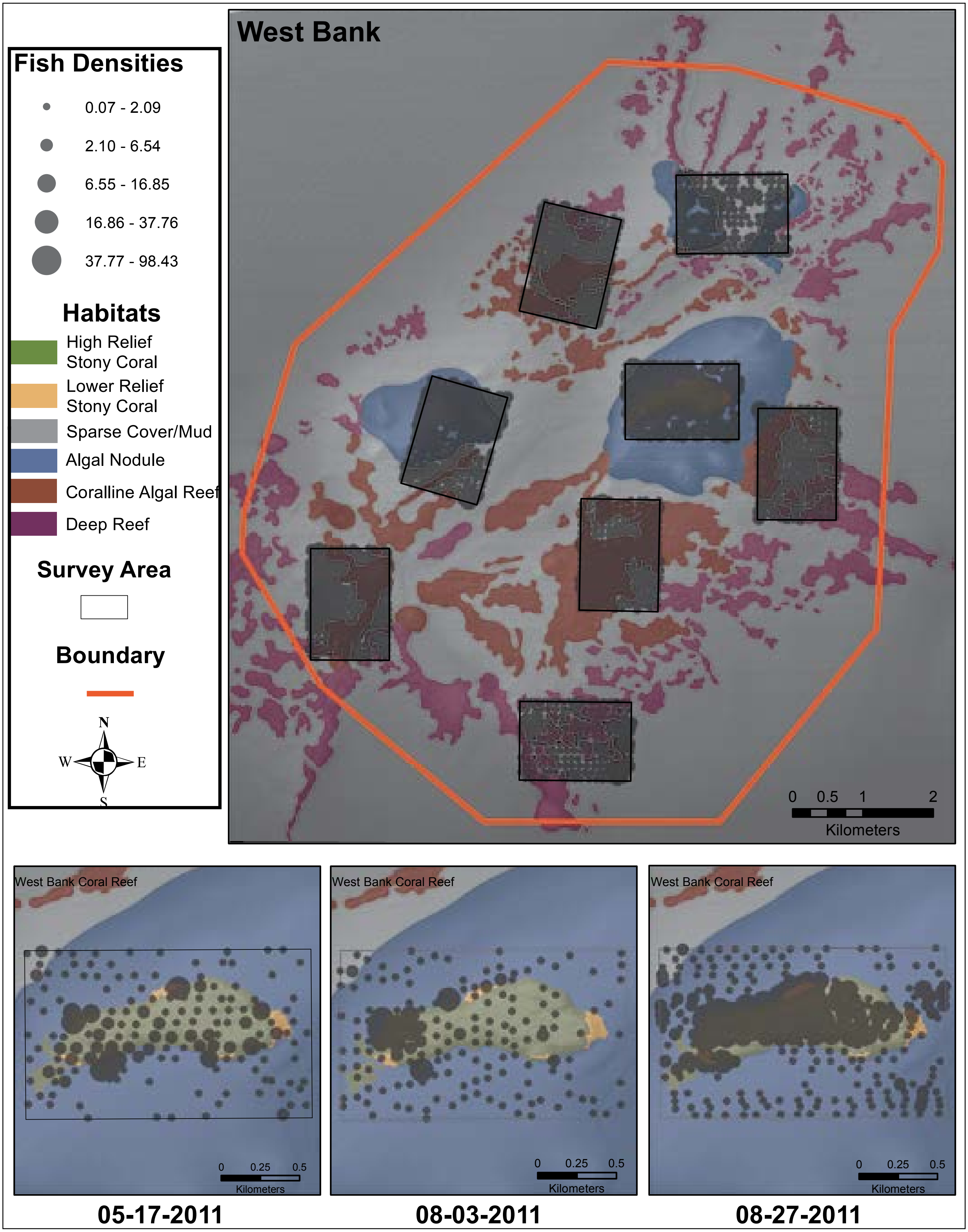

Figure 6.12. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (bottom) and WB (top) for all surveys conducted in 2011.

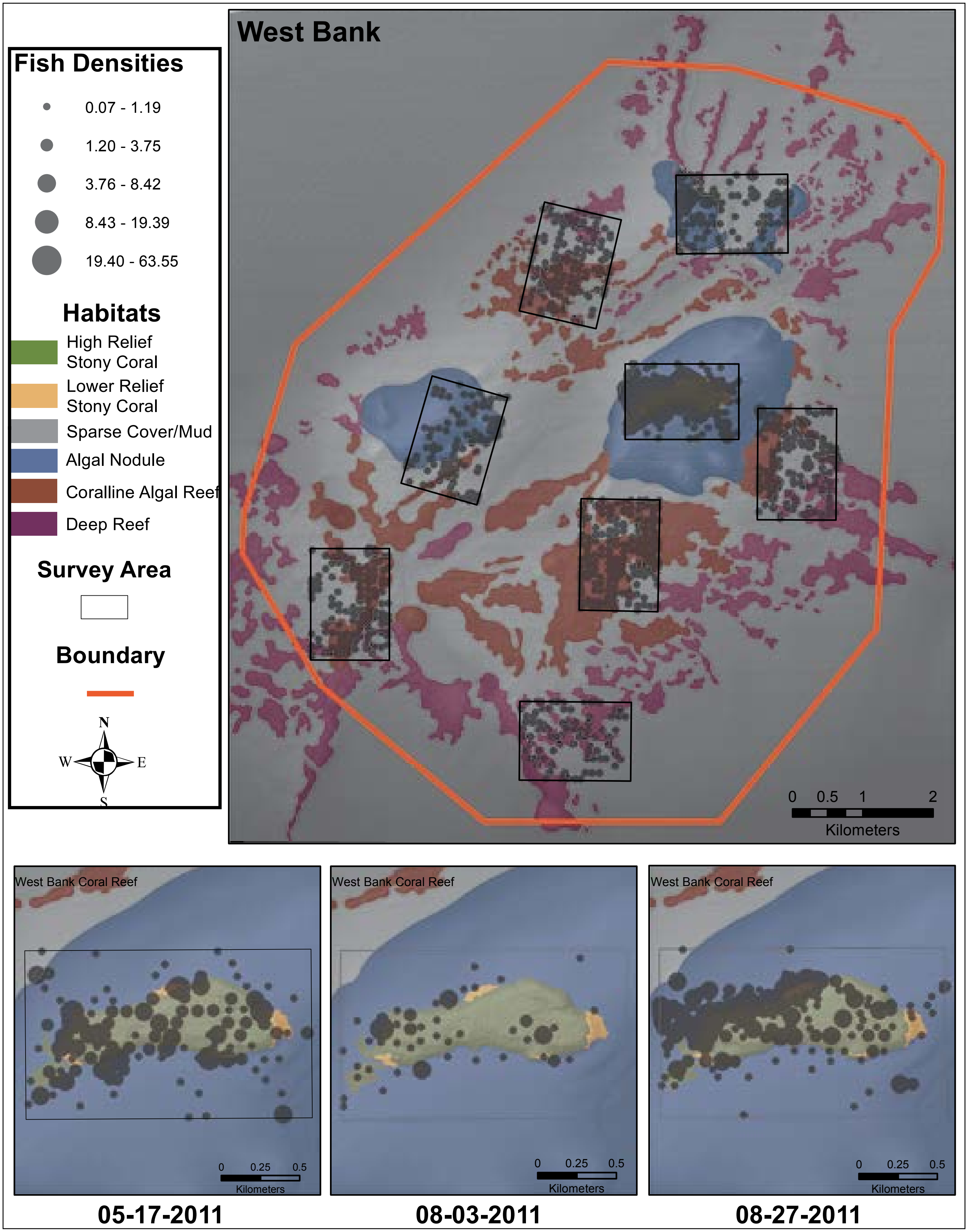

Figure 6.13. Density maps for large fish (>29 cm) for the Coral Reef Zone (bottom) and WB (top) for all surveys conducted in 2011.

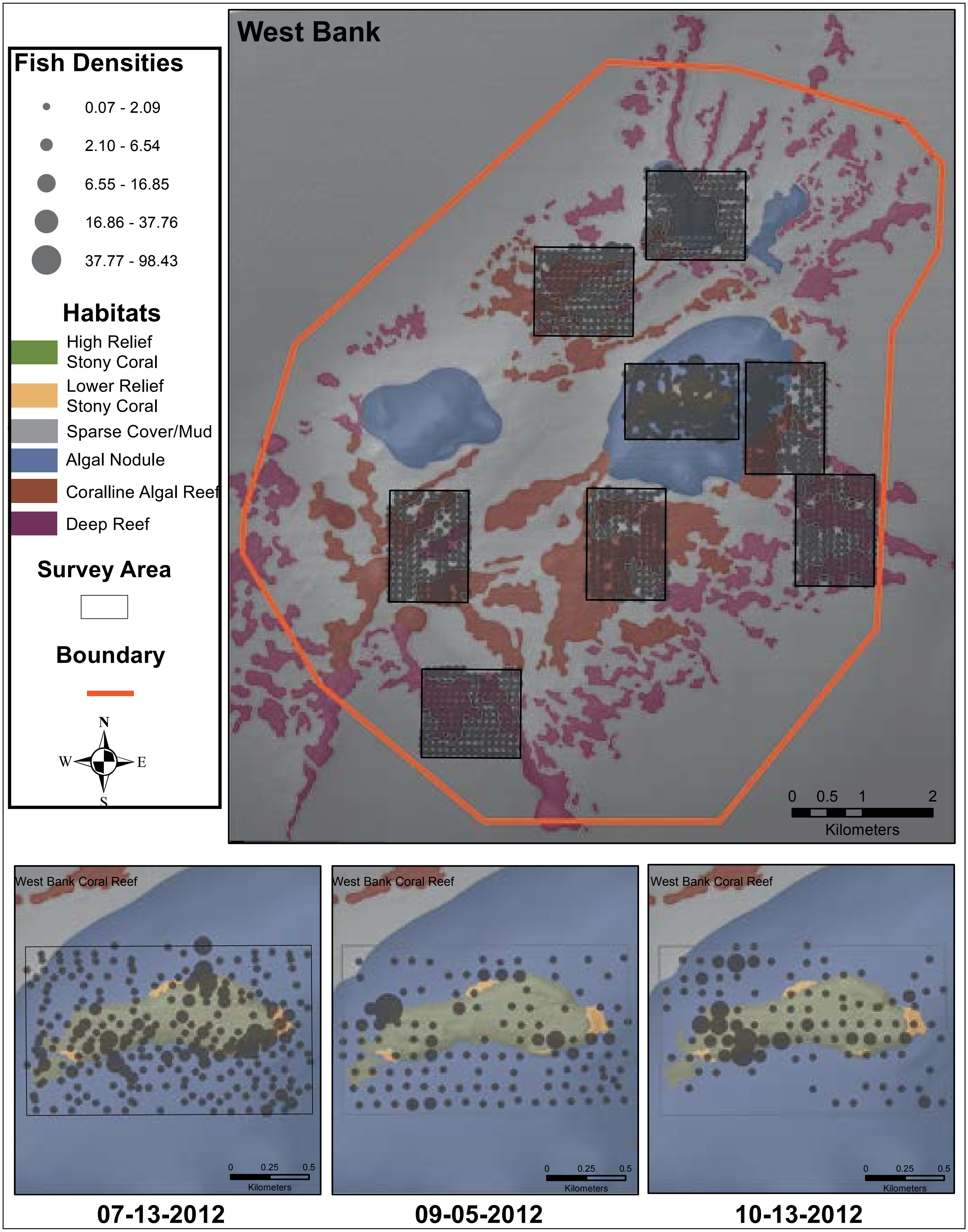

Figure 6.14. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (bottom) and WB (top) for all surveys conducted in 2012.

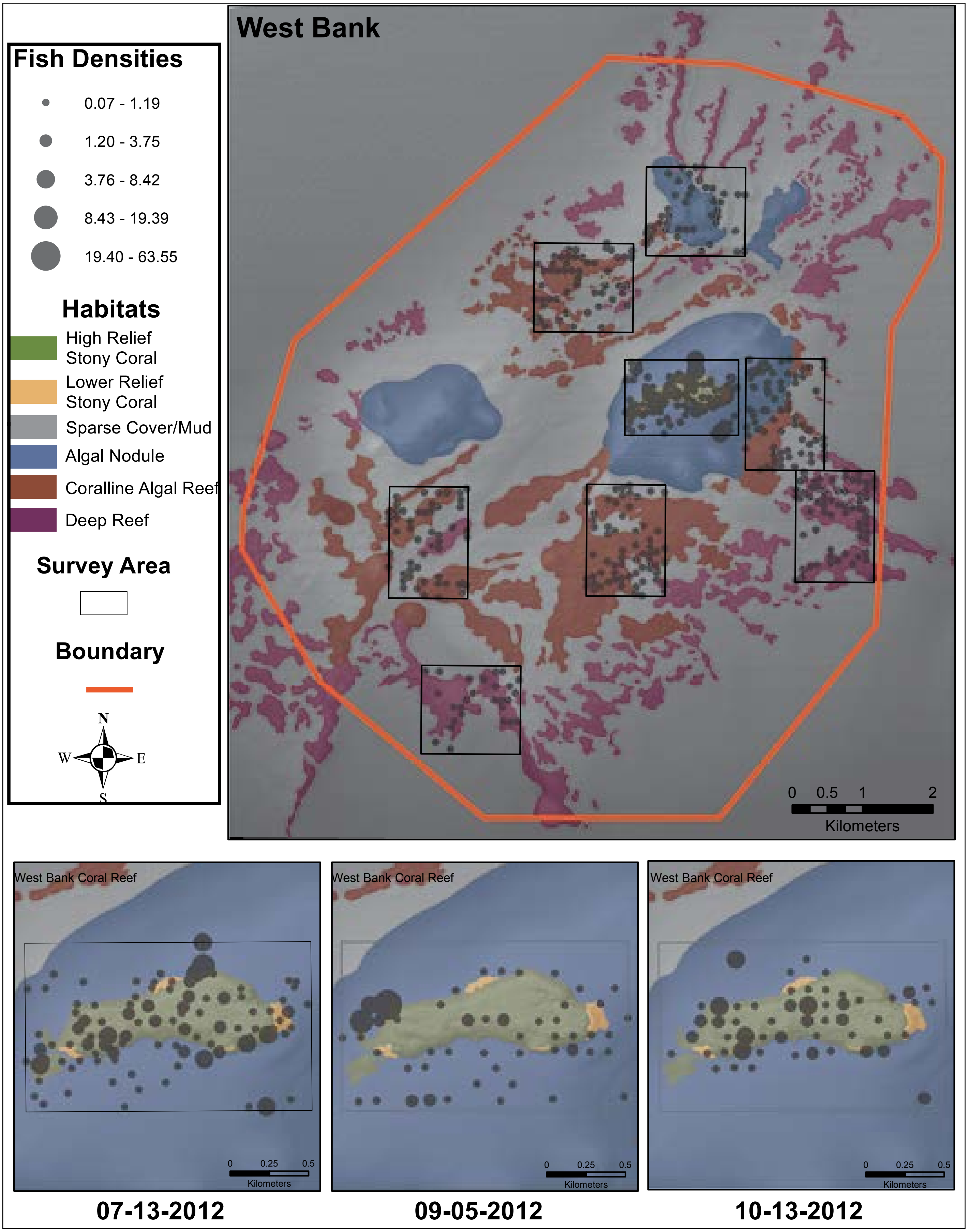

Figure 6.15. Density map for large fish (>29 cm) for the WB and Coral Reef Zone for all surveys conducted in 2012.

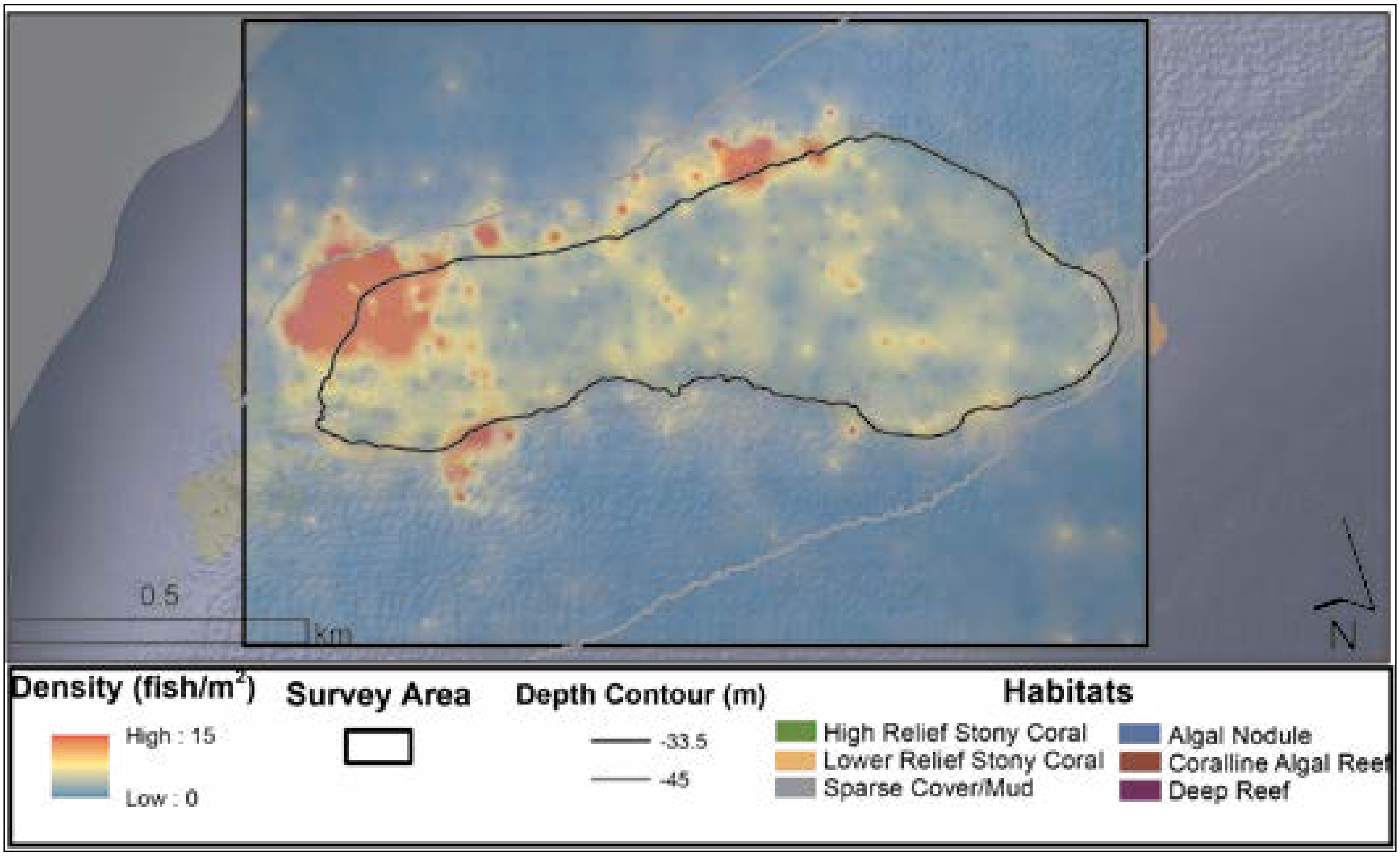

Figure 6.16. Composite interpolation of large fish density for all surveys (2010-2012) conducted on WB coral reef. Densities represented by blue (low) to red (high) color scale.

Fishery Acoustics

Acoustic Fish Densities – East Bank

High densities of small fish on EB were observed over the coral reef and deep structured habitats, in addition to sparse cover/mud zones adjacent to structured habitats in 2010 (Figures 6.17). The highest densities of medium and large fish were found in the southern portion of the EB coral reef. Fish distribution patterns were consistent over the three years of study (though less pronounced in 2012), with high densities of large sized fish found in the southern region of EB coral reef (Figures 6.19, 6.21 and 6.23). Medium sized fish density patterns nearly mirrored the pattern of the large sized fish densities with high densities in the central region of the southern and northern portion of EB coral reef (Figures 6.18, 6.20 and 6.22).

Mulloidichthys martinicus *(yellow goatfish) at FGBNMS. Photo: M. Winfield (UNCW)*

Deep reefs and coralline algal reefs also held relatively high densities of large and medium sized fish which were consistent throughout the three year study. High densities were also observed on deep reefs in the northern region of EB as well as coralline algal reefs adjacent to the algal nodule habitat (Figures 6.18-6.23). Medium fish densities were more broadly distributed across all habitat types, particularly during the surveys conducted in 2011. In 2011, high densities of medium sized fish were found over soft sediments, likely comprised of pelagic planktivores that formed high density scattering layers during night-time (Figure 6.20). In 2010 and 2012, lower fish densities over soft sediments were observed with high densities of medium sized fishes concentrated over deep reefs and the transition zone between the coral reef and algal nodules (Figures 6.18 and 6.22).

Interpolations of large fish density, from repeated surveys on the EB coral reef indicate a pattern of high density in the central region of both the northern and southern area of EB cap, with notably high densities in the central region of the northern edge of the coral reef (Figure 6.24).

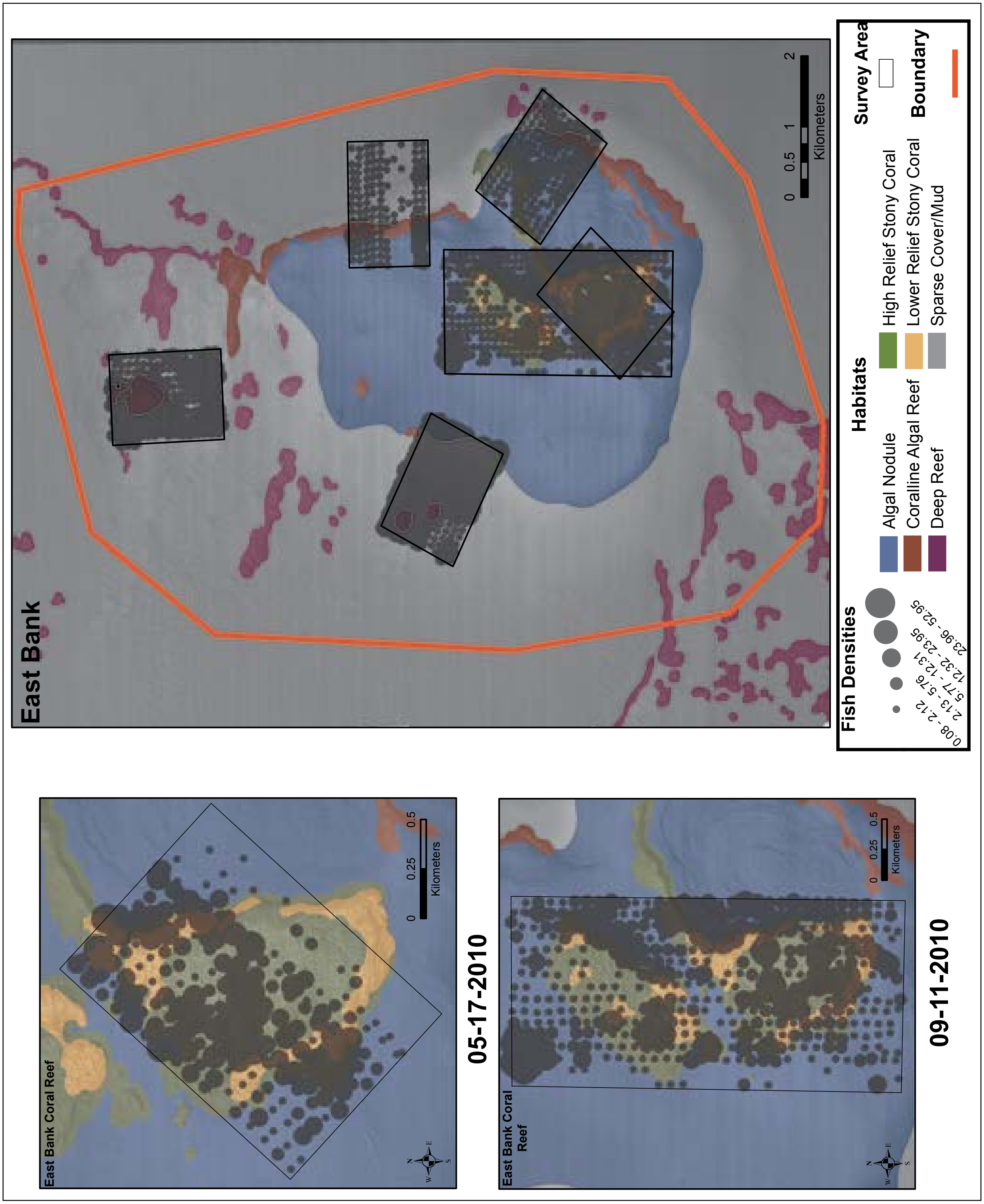

Figure 6.17. Density maps for small fish (<11 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2010.

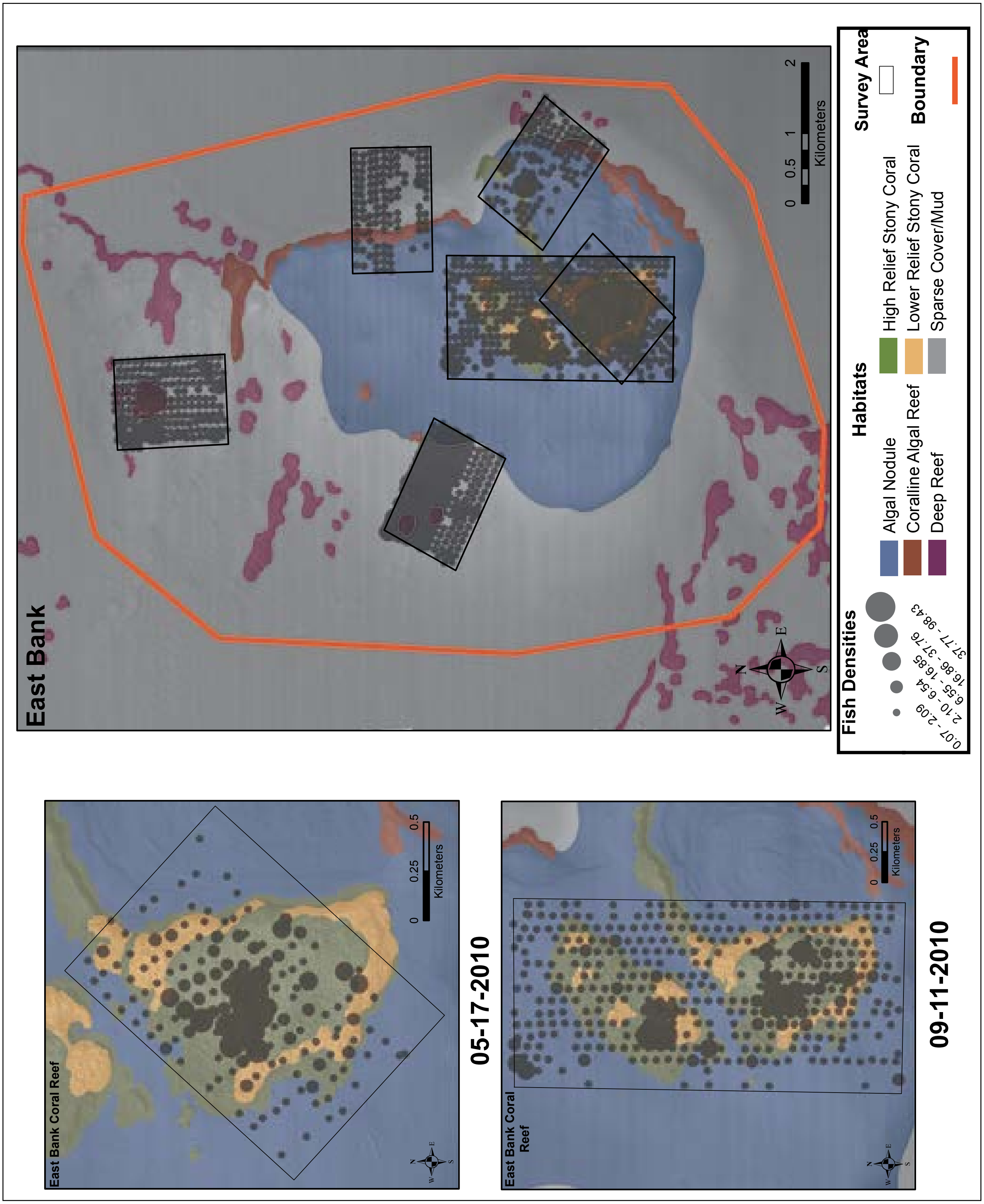

Figure 6.18. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2010.

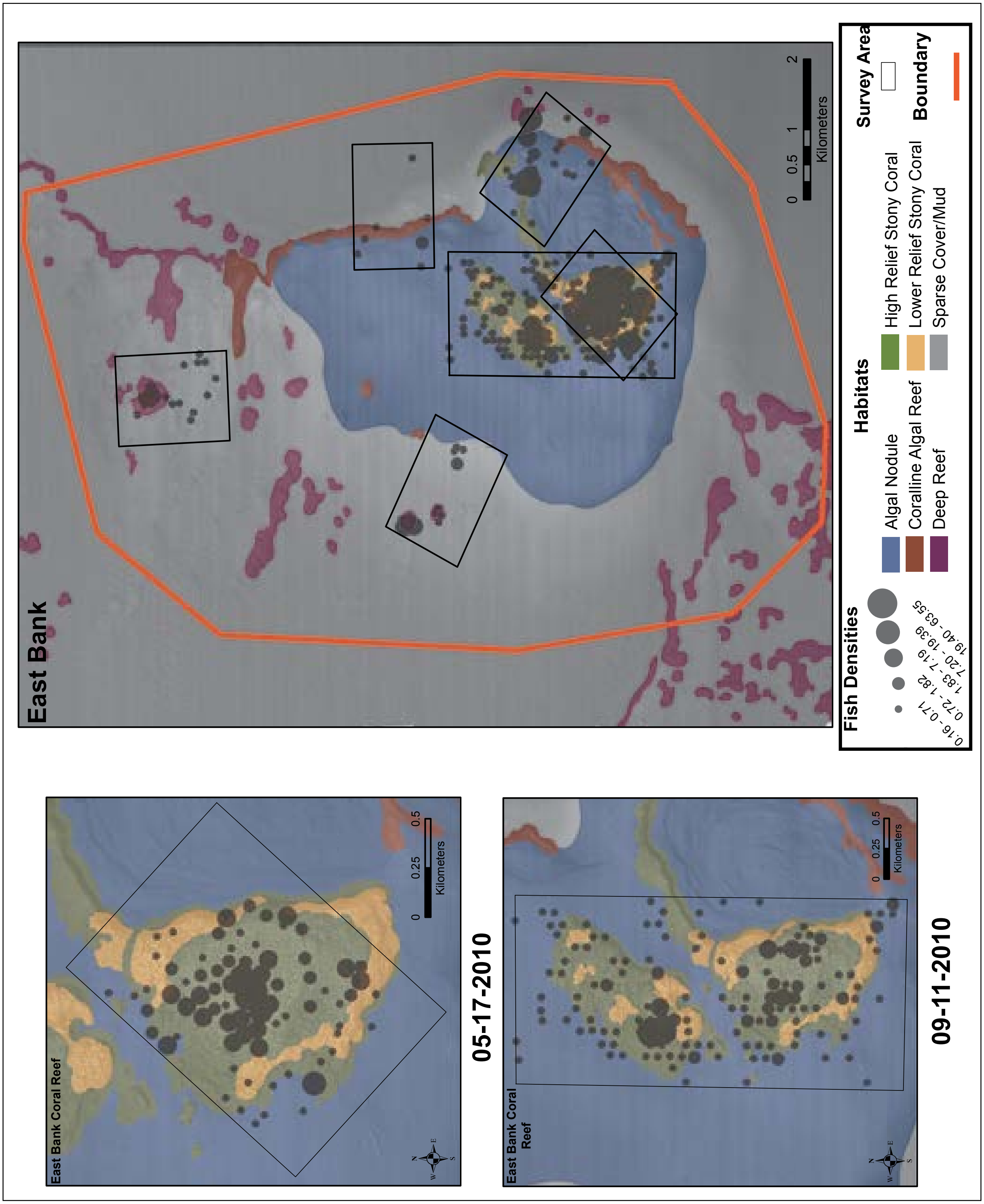

Figure 6.19. Density maps for large fish (>29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2010.

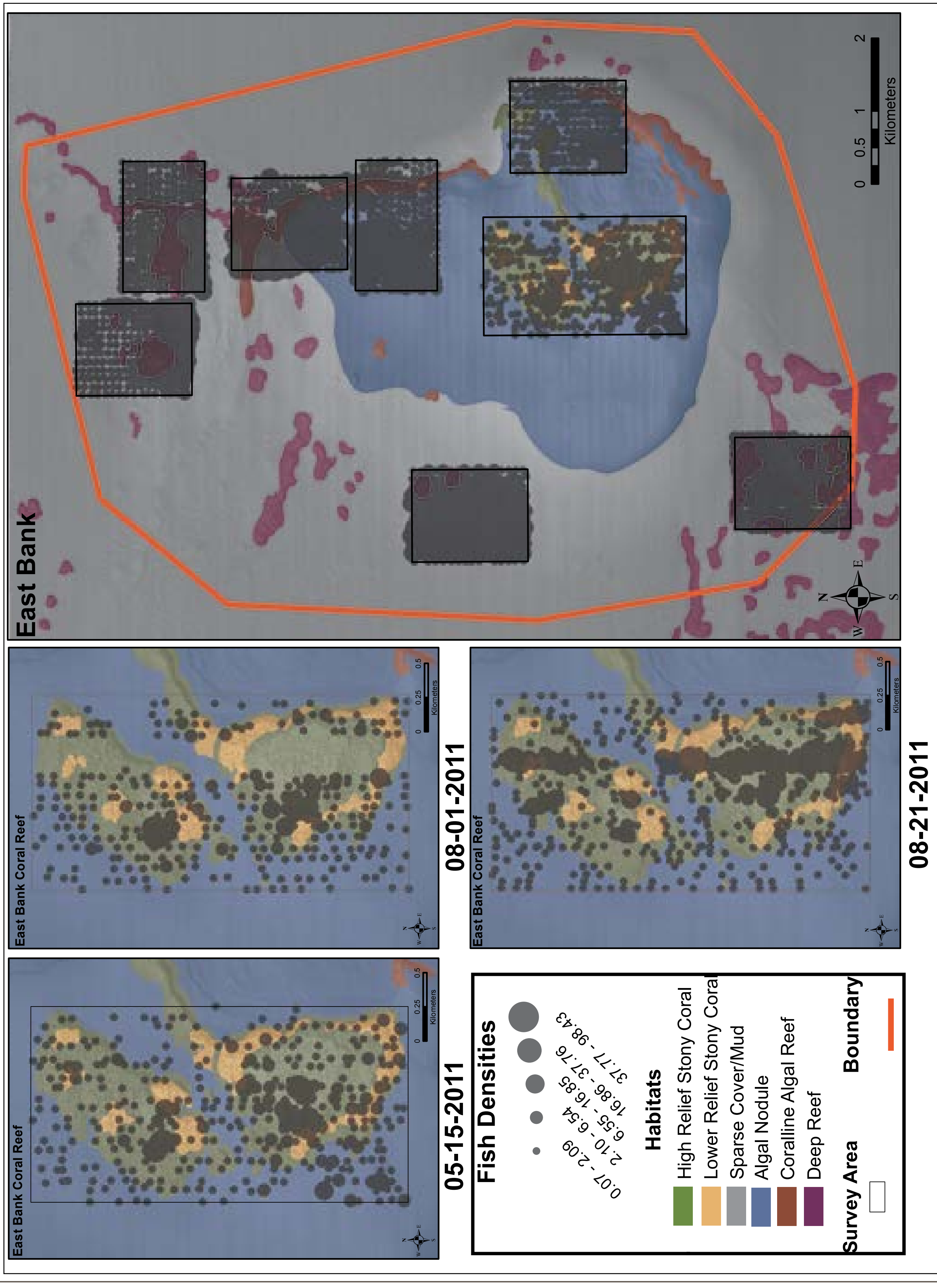

Figure 6.20. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2011.

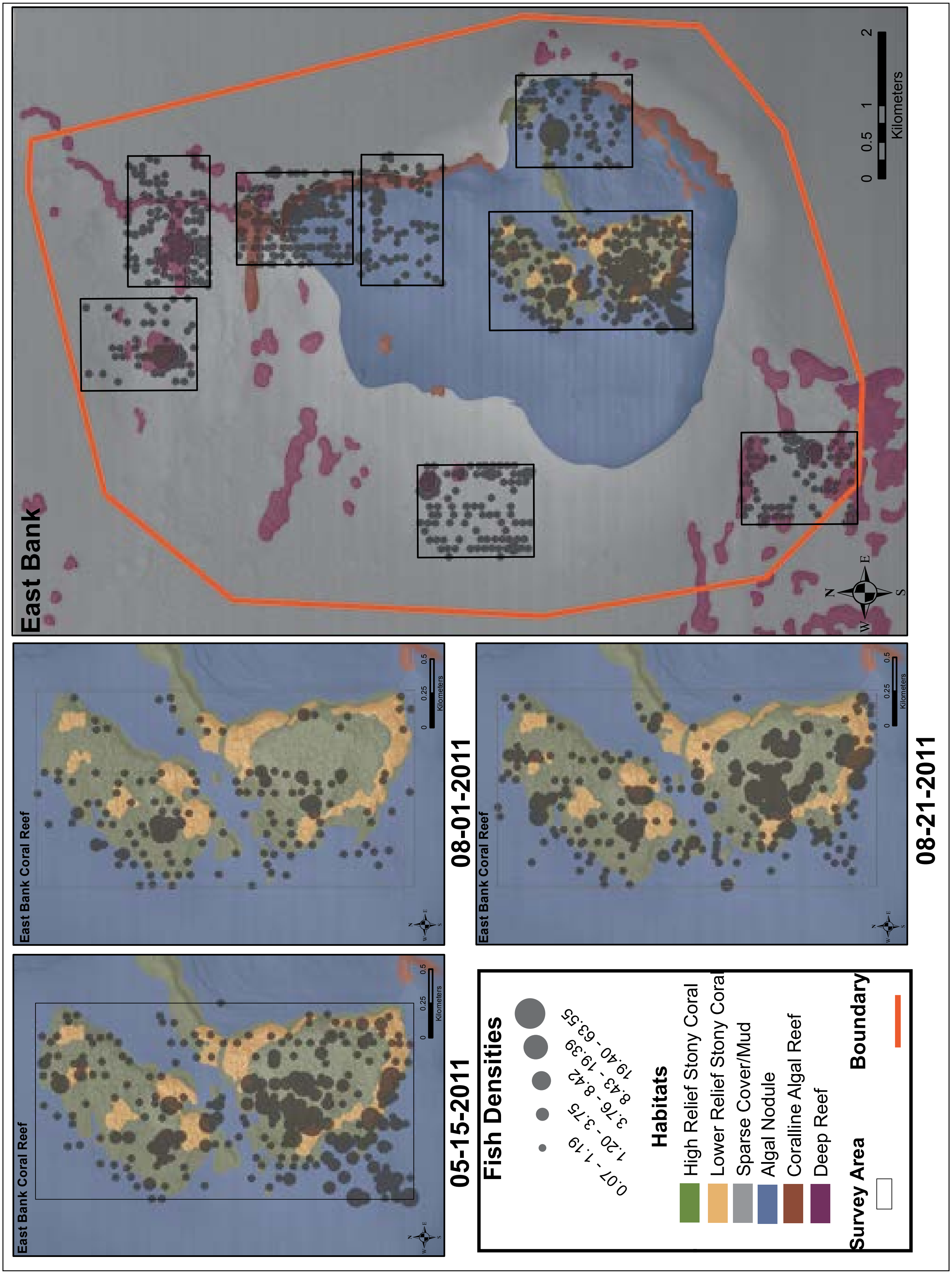

Figure 6.21. Density map for large fish (>29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2011.

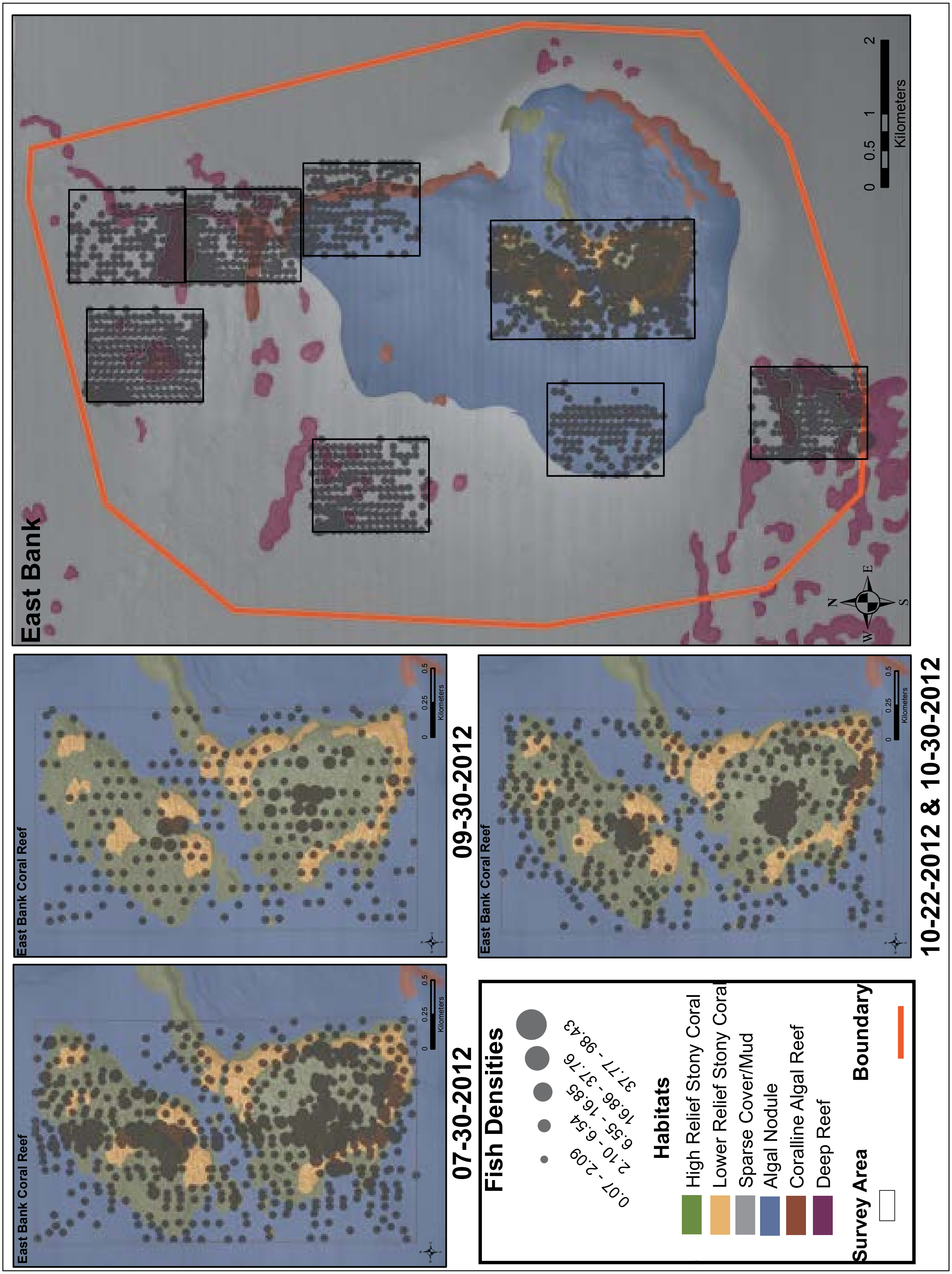

Figure 6.22. Density maps for medium fish (11-29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2012.

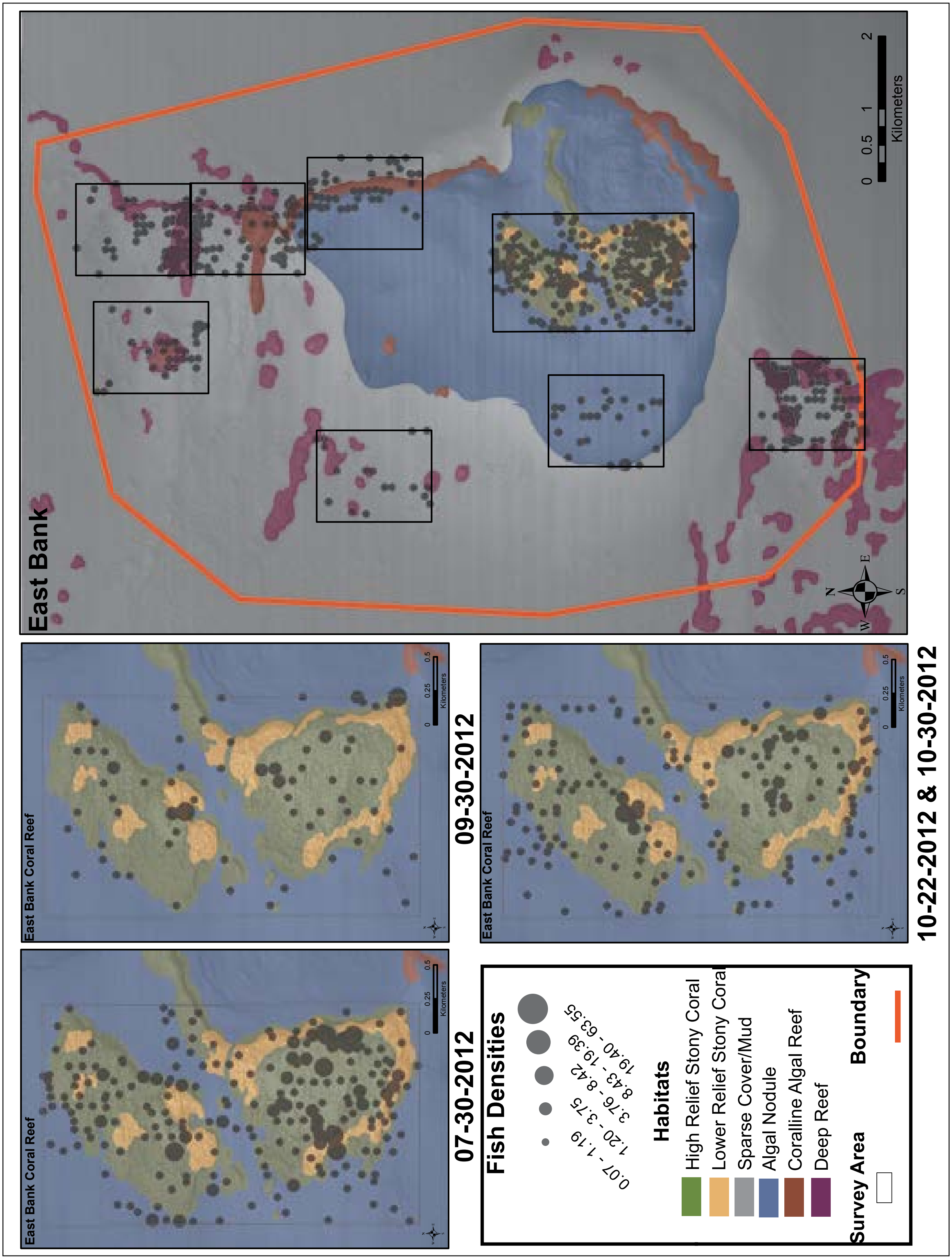

Figure 6.23. Density maps for large fish (>29 cm) for the Coral Reef Zone (left) and EB (right) for all surveys conducted in 2012.

Fishery Acoustics

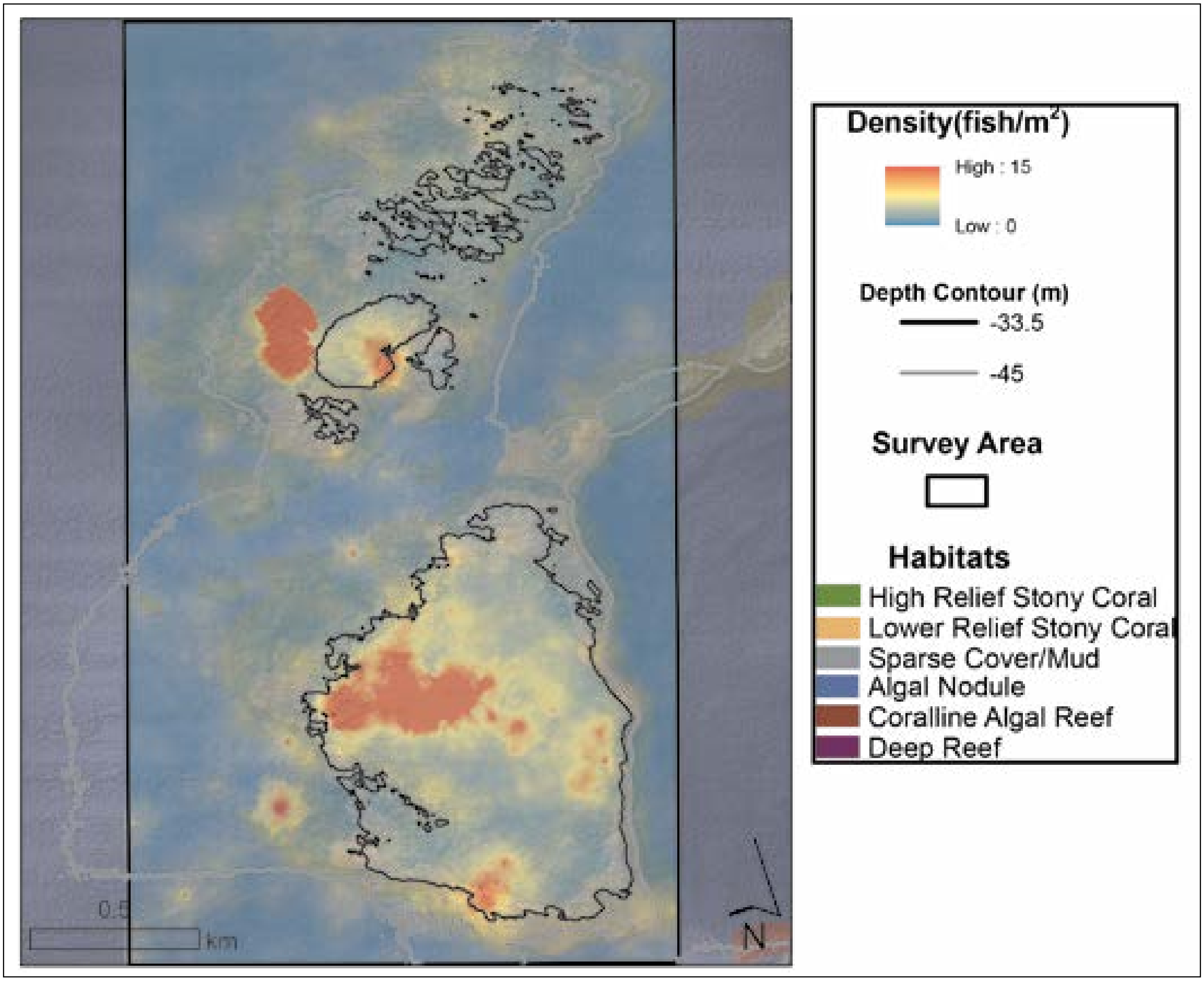

Figure 6.24. Composite interpolation of large fish densities from all fishery acoustic surveys (2010-2012) conducted at EB coral reef.

6.3.4. Fish Densities by Habitat Type

Mycteroperca insterstitialis *(yellowmouth grouper) at FGBNMS. Photo: M. Winfield (UNCW)*

Average densities of all fish size classes between banks, across habitat zones, and over the three years are presented in Figures 6.25 and 6.26. Over the entire bank, independent of habitat zone, EB had significantly higher densities of large fish than WB (Table 6.2). Mean densities of large fish in the WB and EB were significantly higher over the coral reef, followed by the deep reef zone, coralline algae, algal nodule, and finally sparse cover/mud zones (Tukey HSD, adjusted p-value < 0.009). Fish densities were over three times higher on stony coral habitats on the coral reef than on other structured deep habitats. Wilson et al. (2003) also observed significantly higher densities from acoustic surveys on the coral reef (called "upper terrace"), compared to other deep habitats. Their density estimates included all size fish calculated in units of fish per cubic meter and were assigned to depth zones rather than habitat types; however, converting their maximum densities over the upper terrace to fish per 100 m^2 results in densities of about 7.5 fish/100 m^2 which is similar to the overall average densities we observed over stony coral habitats on the coral reef in 2010 and 2011 (mean density: 7.38 fish/100 m^2). Similarly, the densities observed by Wilson et al (2003) over the "middle terrace" were similar to the overall densities we observed during acoustic surveys over coralline algal, algal nodule and deep reef habitats with about 3 fish/100 m^2. Average densities of large fish on the coral reef were significantly higher on the WB than EB (Tukey's HSD, adjusted p-value << 0.0001). In contrast, average densities of large fish over coralline algae, deep coral, and sparse cover/mud were significantly higher in the EB than WB (Tukey's HSD, adjusted p-value <<0.0001). Densities of large fish in algal nodule habitats were statistically similar between the East and West Banks.

Table 6.2. Comparison of large fish densities (>29 cm TL) within habitats between East and West Banks. Symbol indicates Bank with higher densities (Tukey HSD Test, p < 0.05). N.S. indicates no significant difference for that habitat between Banks.

Habitats				Adjusted p-value
Overall	West Bank	<	East Bank	<<0.0001
Coral Reef	West Bank	>	East Bank	<<0.0001
Deep Reef	West Bank	<	East Bank	<<0.0001
Coralline Algae	West Bank	<	East Bank	<<0.0001
Algal Nodule	West Bank	N.S.	East Bank	0.9999
Sparse Cover/Mud	West Bank	<	East Bank	<<0.0001

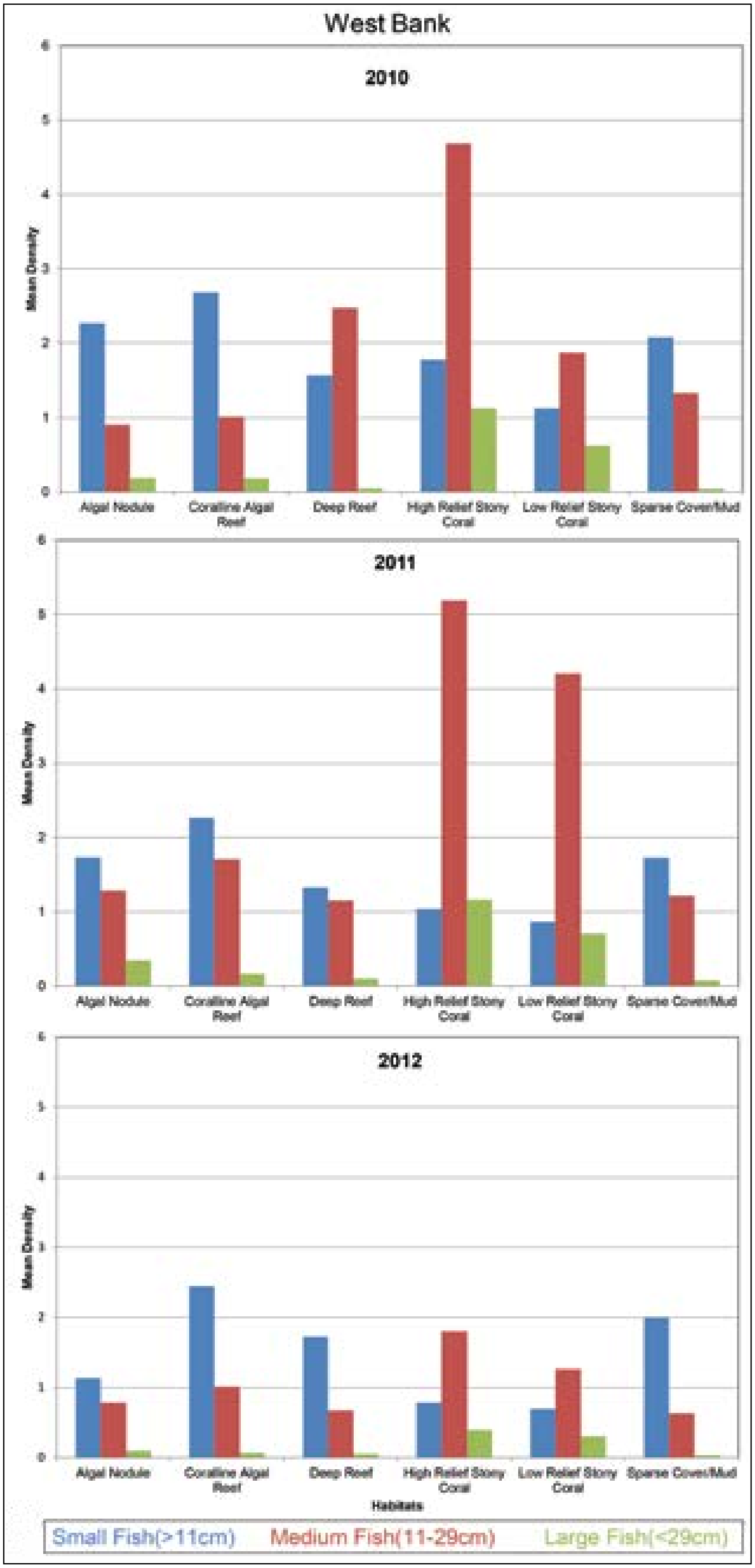

Figure 6.25. Mean densities (# / 100 m²) for small, medium and large fish over each habitat zone in WB for 2010, 2011 and 2012. Note: an electrical problem in the 120 kHz transducer resulted in a decrease in fish detection.

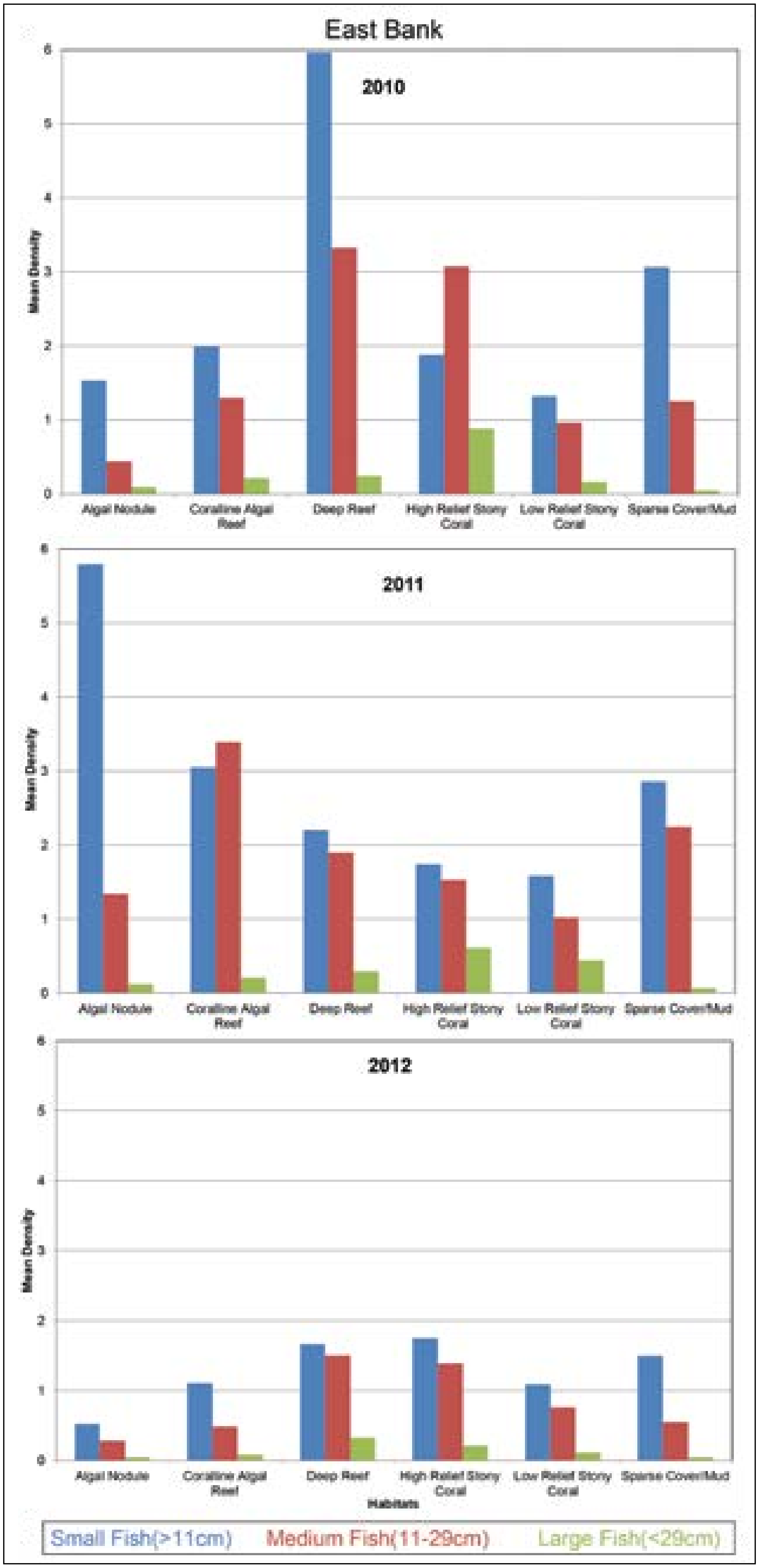

Figure 6.26. Mean densities (# / 100 m²) for small, medium and large fish over each habitat zone in EB for 2010, 2011 and 2012. Note: an electrical problem in the 120 kHz transducer resulted in a decrease in fish detection.

6.3.5. Comparison of Fish Densities at FGBNMS and Other US Coral Reef Ecosystems

Densities of large fish size classes observed over stony coral habitats on the coral reef in the sanctuary from this three-year study were compared with other similar surveys conducted in coral reef ecosystems in U.S. and territorial waters. Densities of large fish were three to ten times higher in the WB coral reef of FGBNMS than other coral reef ecosystems surveyed in the Tortugas Ecological Reserve in the Florida Keys; Vieques Island, PR; and the insular shelf near St. John, USVI (Figure 6.27). The Tortugas Ecological Reserve is the only region in this group that is closed to fishing harvest. The Flower Garden Banks is farther from fishing ports than all the other locations which may help explain the high densities of large fish found there. However, it remains unknown whether the remote location of the Flower Garden Banks is directly related to lower fishing effort.

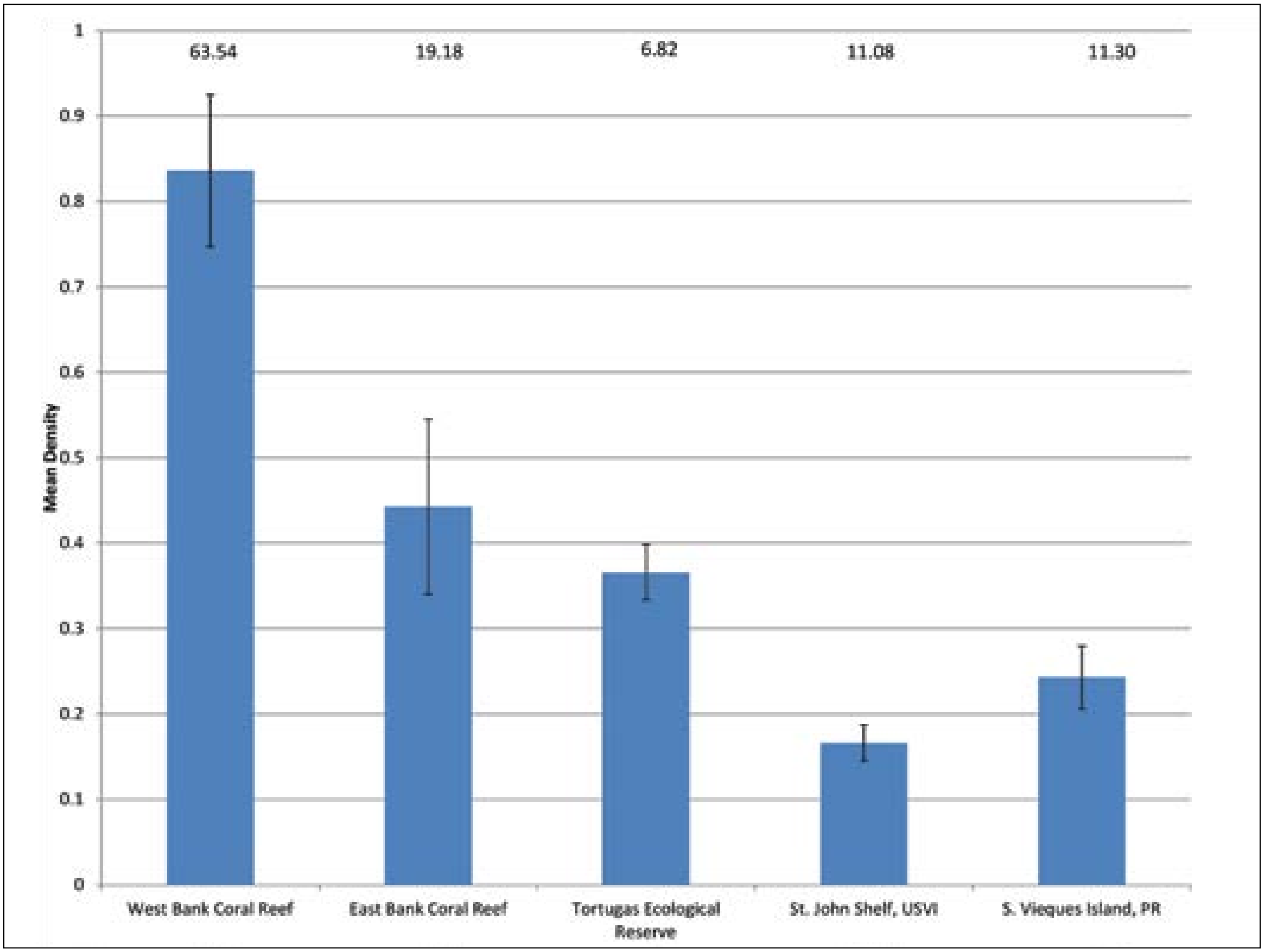

Figure 6.27. Mean densities of large fish (>29 cm estimated total length) in high and low relief stony coral habitats on coral reef in the FGBNMS and other coral reef ecosystems in the US territorial waters. Values at top are maximum densities observed.

6.4. SUMMARY

- The fishery acoustic surveys conducted during this project were the first to systematically survey the Flower Garden Banks in over ten years (Wilson et al., 2003).
- The distribution of fish density conformed to the general patterns observed using visual methods on the coral caps and the deep structured habitats. Densities of large fish were highest on stony coral habitats compared to deeper structured habitats like algal nodules, coralline algal reefs and deep reefs.

- The acoustic surveys were unique among the methods used in this study because they provided repeated surveys over the coral reef and identified consistent fish density hot spots over the three year baseline study. Hot spots were evident in the northern and northwestern region of the WB coral reef and the central region of the shallow coral reef on the EB. The biomass in these hotspots is likely comprised of very large schools of creolefish and creole wrasse, but also includes large apex predators like snappers and groupers that were also observed in the shallow and upper mesophotic visual surveys in these same areas of the coral reefs (Chapter 4, this volume).
- Anecdotal reports suggest relatively higher fishing pressure occurs on the deeper slopes of the coral reefs and deep reefs (E. Hickerson, pers. comm.). Continued monitoring of both the fish communities and fishing effort should be conducted in these high fish density regions of the coral reefs to determine the unique quality of this region to the fish communities and potential impacts from fishing on large apex predators.
- Additional and repeated acoustic surveys should also be conducted in coralline algal and algal nodule reefs to identify other potential biomass hotspots in these deeper habitats.
- We recommend continuing to conduct acoustic surveys with other visual monitoring approaches in future monitoring programs.

6.5. REFERENCES

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Chapter 7
Conclusions and Recommendations

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Photo courtesy of M. Winfield (UNCW)

Conclusion

7.1. CONCLUSIONS

This report presents summaries and analyses of a three-year baseline ecological assessment of the Flower Garden Banks National Marine Sanctuary (FGBNMS) to guide the design and implementation of a research-only area within the sanctuary. The results reported here represent the most comprehensive and synoptic survey of FGBNMS and is consistent with historical data that the sanctuary is a diverse ecosystem with thriving coral reefs, fish, and benthic communities. The shallow coral reef has the highest percent coral cover in the tropical western Atlantic and has exhibited remarkable resilience in a time when many other coral reef ecosystems around the world have become substantially degraded. Some portions of the banks yielded over 90% coral cover and may be among the highest of coral reefs in the U.S. and territorial waters. The scleractinian coral reef extends from 18-52 m and is comprised of mostly high relief coral formations, with low relief areas dominated by algae, sponges and non-structure forming corals. Interestingly, the reef morphology changes with depth (Figure 7.1). At depths from 18-33 m, coral colonies are massive boulder shapes, however at depths greater than 33 m, the colonies flatten. This flattening with increasing depth is likely an adaptation to maximize surface area exposed to sunlight for photosynthesis (Barnes, 1973). Despite this difference in the benthic structure over depth, both morphologies provide ample habitat for benthic invertebrate and fish communities - a quality that has proven important by the evidence of use of the sanctuary by fishing and diving communities. Likewise, the state of the resources also raises challenges for managing the sanctuary, particularly when considering implementation of a research area in the sanctuary.

Figure 7.1. Example of high (left) and low (right) relief stony coral on the shallow and mesophotic caps. Photos: G. McFall (NOAA NOS/ONMS/ GRNMS) and NOAA NOS/NCCOS/CCMA

The sanctuary's living marine resources, both fishes and benthic invertebrate communities, vary significantly with depth. Depth is the predominant factor structuring fish and benthic communities within habitat types. The shallow coral reef (<33 m) has a rich diversity of fishes that is significantly different in terms of species and abundance when compared to reefs >33 m. In the upper mesophotic reef (33-52 m depths) groupers, snappers, sharks and other apex predators are larger and more numerous, equating to significantly higher biomass in this depth zone compared to the shallow coral reef habitats. Habitat complexity is also highly correlated with apex predator density and biomass, both on the coral reef and on the coralline algal and deep reef habitats. Interestingly, species composition among the apex predators is completely different across depth strata. On the shallow reef, *Mycteroperca tigris* (tiger grouper), *Lutjanus griseus* (gray snapper), *Lutjanus jocu* (dog snapper) and *Sphyraena barracuda* (great barracuda) are numerically dominant. On the deeper reefs, *Mycteroperca phenax* (scamp) and *Lutjanus campechanus* (red snapper) dominate.

Conclusion

Information derived from fishery acoustic surveys support the pattern of consistent, high fish density and biomass observed on the complex hardbottom habitats (e.g., shallow and mesophotic stony coral, coralline algal reef and deep reefs) throughout both banks. The high densities of large fish (>29 cm total length) were higher than the densities found during similar fishery acoustic surveys conducted in no-take areas of the Florida Keys National Marine Sanctuary (FKNMS) – Tortugas Ecological Reserve and two other coral reef ecosystems in the U.S. Virgin Islands. Over the three years of this project, at least ten surveys were repeated over the stony coral habitats on the caps in each bank. Taken together, these surveys show consistent hotspots of large fish (>29 cm total length) density, specifically in the northwest region of the cap in the West Bank (WB), and in a few areas on the East Bank (EB) caps. Although there were slightly higher densities of large fish found in the WB compared to the EB cap, high densities of large fish and their distribution relative to habitats was common to both banks. The larger area of upper mesophotic coral reef on EB may have significant importance to the growth and production of grouper/snapper populations in the region.

Our knowledge of the ecological importance of deepwater coral communities is growing. This spatially comprehensive assessment of fish and benthic communities in the sanctuary provides baseline data that can be used for impact assessment and addressing future management questions.

Indo-Pacific Lionfish

The proliferation of non-indigenous species is an emerging threat throughout many marine ecosystems. During the time of this project, we observed the onset of *Pterois volitans* (red lionfish) settlement into the sanctuary. Given the expanding lionfish distribution through the U.S. Atlantic, Gulf and Caribbean waters, the apparent recent arrival to the frequently surveyed shallow and upper mesophotic reef of Flower Garden Banks has raised concern over future impacts to the resident fish community (Muñoz et al., 2011). Indeed, Morris and Akins (2009) found economically important *Ocyurus chrysurus* (yellowtail snapper) and *Epinephelus striatus* (Nassau grouper) in the stomachs collected from lionfish in the Bahamas.

Lionfish are reported to have few natural predators (Bernadsky and Goulet, 1991) due to their venomous spines, but conclusions from earlier studies are hampered by small sample sizes and suffer from the paucity of investigations in the native range. In the invaded range, lionfish have been found in the stomach contents of piscivorous groupers (Maljkoviç et al., 2008). Mumby et al. (2011) recently found a seven-fold lower biomass of lionfish relative to grouper biomass in a Bahamian marine reserve compared to outside the reserve. Numerous Atlantic fishes are capable of consuming venomous scorpaenids, including *Lophius americanus* (goosefish) and *Lutjanus analis* (mutton snapper), which consume the venomous scorpaenid *Helicolenus dactylopterus* (blackbelly rosefish) and *Scorpaena plumieri* (spotted scorpionfish; Randall, 1967; Bowman et al., 2000).

To date, no method of control beyond physical removal on the scale of local reefs has been investigated. Because of their planktonic larval dispersal and opportunistic colonization of habitats and foraging behavior, large scale eradication of lionfish will not be feasible. Although sustained control measures may mitigate the eventual extent of lionfish populations at the local scale, the cost of large scale removal, especially in remote locations, will have to be evaluated against the minimal ecological benefit to be gained. At this stage, the potential role of predation in decreasing the number of lionfish is unknown, as is the effect of lionfish on populations of native apex predators. Controlled laboratory and correlative field studies investigating this possibility are an important research need and may shed light on natural lionfish control.

Predation by large carnivores such as groupers and sharks may represent one of the best controls for invasive lionfish (Albins and Hixon, 2008), as low densities (approximately 2.2 individuals/ha) of lionfish were observed in their native range on Palauan reefs with robust grouper populations (Grubich et al., 2009). Reduced numbers of large predators in many invaded locations means that predation on lionfish may not provide effective control. Increased densities of exploited predators in marine reserves are often the first signs of positive responses to protection from fishing (e.g., Mosquera et al., 2000). If predation on lionfish is a controlling mechanism, marine reserves may act as refugia where community assemblages are maintained with low densities of invaders by healthy populations of large predators. Predator abundance within the sanctuary appears to be relatively high (approximately 35 sites supported predators [groupers, barracuda, jacks, sharks] ≥ 100 cm TL), possibly due to its remote location combined with long-term management of fishing effort. The levels measured in this study are probably far below natural levels before harvesting began. We hypothesize that this may lead to predation on lionfish, allowing large predators to act as a natural control in this protected area. Future studies of natural control by predation should examine the correlation between the density of predators and lionfish, and reduced sizes and numbers of lionfish at sites where large predators are found. The lionfish invasion at FGBNMS is in its early stages, so it is timely that managers begin planning for continued monitoring of large predator abundance, as well as removal of lionfish from the sanctuary by researchers, fishers and divers. An ongoing study from a comparable location (remote, protected management status, abundance of large predators) in the Tortugas South Ecological Reserve in the FKNMS is examining similar questions and will make a valuable reference site for predator-lionfish comparisons in FGBNMS. Such studies could have important implications for the management and natural control of lionfish by increasing our understanding of how marine reserves, biodiversity, and community structure facilitate resilience to invasion.

7.2. RECOMMENDATIONS

While there are many pressures and threats facing the sanctuary ecosystem, perhaps the most important is the impacts associated with fishing. As of 2012, the United Nations Food and Agriculture Organization report that over 25% of the world's fish stocks are either overexploited or depleted. Another 52% is fully exploited and in imminent danger of overexploitation (harvesting beyond maximum sustainable production level) and collapse. Thus a total of almost 80% of the world's fisheries are fully- to over-exploited, depleted, or in a state of collapse. Not only does fishing pressure reduce the target population, it also reduces non-targeted species. The removal of key species can cause ecological shifts in the fish community that may transfer to the associated benthic invertebrate community (Hay, 1984; McClanahan et al., 1996; Pauly et al., 1998).

The objective of this work was to develop a comprehensive survey design that develops baseline information for fish and benthic communities throughout the sanctuary, and forms the biological foundation for establishing a no-take research area. The research-only area will help provide perspective on the magnitude of fishing impacts in the sanctuary as well as provide the setting for important ecological studies, such as Interactions between invasive lionfish and resident fish communities.

The baseline data provided here can be used to detect community level responses to various levels of fishing effort or other significant impacts. The robust data collected across the entirety of habitats and depth strata, using complementary optical and acoustic methods, within the reserve boundaries allows for simulation studies that can help guide the design of a research-only area in the sanctuary. NOAA's National Center for Coastal Ocean Science (NCCOS) has previously worked with Gray's Reef National Marine Sanctuary in developing a simulation approach to implement a research-only area (Kendall and Eschelbach, 2006). The suggestion that a working group, with public input, be formed to establish the priorities and guidelines for developing a research only area is fully supported. The findings presented indicate that the scleractinian coral reef that extends into the upper mesophotic zone (to approximately 46 m) may be a likely candidate as a focal point for a research only area with select, contiguous areas extending into the deeper habitat (Figure 7.2). Spatial analysis tools and fish community metrics (e.g., biomass of large predators, species diversity, size spectra of key species) based on this study will be extremely useful in evaluating design options to meet management criteria and establishing performance measures and monitoring metrics.

Conclusion

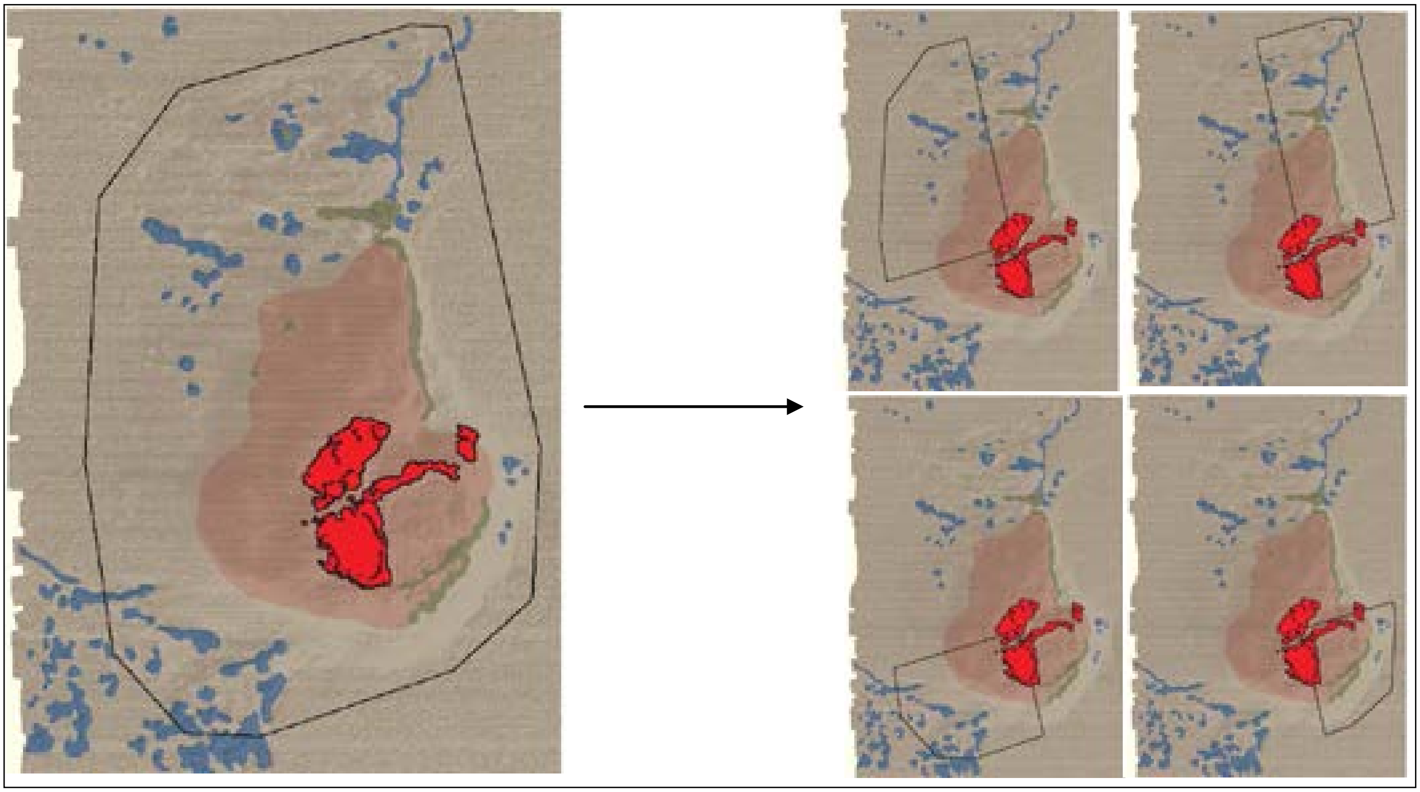

Figure 7.2. Example of hypothetical boundaries for a research-only area for East Bank.

A data gap identified here is the lack of information on fishing pressure and effort throughout the sanctuary. Fishing effort at the Flower Garden Banks and other regions of the northwest Gulf of Mexico is poorly understood. We concur with the recommendation of the sanctuary advisory council that dedicated research should be conducted to better understand the type, timing and intensity of fishing effort. Finally, we propose continued monitoring of the entire sanctuary and utilizing the proposed research-only area to detect changes and trends in fish community metrics. This monitoring effort should be conducted in the context of population dynamics of key species in the northwest Gulf of Mexico. Appropriate control sites should be established and monitored with the same level of effort and methodology used here. We recommend repeating this study as a comprehensive sampling program, surveying fish and benthic invertebrates across the full depth and broad habitats found in the sanctuary.

7.3. REFERENCES

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Appendices

Photo courtesy of G. McFall (NOAA NOS/NOS/ONMS/GRNMS)

APPENDIX A: SCUBA Visual Observation Benthic Survey Methods

Once in the field, the boat captain navigates to previously selected sites using a handheld GPS unit. On-site, divers are deployed and maintain visual contact with each other throughout the entire census. One diver is responsible for collecting data on benthic composition. This diver follows the belt-transect diver and records data on small-scale benthic habitat composition and structure within a 1 m^2 quadrat divided into 100 (10 x 10 cm squares) at four separate positions along the transect. Each position is randomly chosen before entering the water such that there is one random point within every 6 m interval along the transect. Percent cover is obtained as if looking at the quadrat in a two dimensional plane (i.e., a photograph) versus three dimensions where percent cover could add up to greater than 100%. To estimate percent cover, the diver first positions the quadrat at the chosen meter mark along a randomly chosen side of the transect tape. The remaining quadrats are placed on alternating sides of the transect at the subsequent three locations.

Data are collected on the following:

1. Logistic information - diver name, dive buddy, date, time of survey, site code and meter numbers at which the quadrat is placed.
2. Habitat structure - to characterize the benthic habitats of the dive site, the habitat diver first categorizes the habitat structure of the site (high or low relief). This is done by quantification to the nearest 5% of the dominant coral forms within a 25 m radius of the transect starting point. High relief habitat is characterized by the dominance of coral colonies in the genus *Montastraea/Orbicella* and *Psuedodiploria* while the low relief habitat is characterized by the dominance of *Madracis auretenra*. The habitat category to which a site is assigned should be made independently of the map so that in situ data can be used for map validation.
3. Transect depth profile - the depth at each quadrat position. Depth is measured with a digital depth gauge to the nearest 0.3 m (1 ft).
4. Abiotic footprint - defined as the percent cover (to the nearest 1%) of hard bottom, sand, rubble and fine sediments within a 1 m^2 quadrat. Rubble refers to rocks and coral fragments that are moveable; immovable rocks are considered hard bottom. The percent cover given as a part of the abiotic footprint should total 100%. In a hard coral area for example, despite the fact that hard corals may provide 50% cover the underlying substrate is 100% hard substrate so this is what is recorded. The diver then estimates the height (in cm) of the hardbottom within each quadrat from the substrate.
5. Biotic footprint - defined as the percent cover (to the nearest 0.1%) of live corals, algae, sponges, gorgonians and other biota (tunicates, anemones, zooanthids and hydroids) within a 1 m^2 quadrat. The remaining cover is recorded as bare substrate to bring the total to 100%. Again, the diver must use a planar view to estimate percent cover of the biota. Species covering less than 0.1% of the area are not recorded. Taxa are identified to the following levels: stony coral to species, algae to morphological group (macro, turf, crustose), and sponge to morphological group (barrel/tube/vase or encrusting). Macroalgae is defined as algae equal to or greater than 1 cm in height whereas turf is identified as a mix of short algae less than 1 cm high. For stony corals, the approximate area covered by living coral tissue is recorded. Coral skeleton (without living tissue) is usually categorized as turf algae or uncolonized substrate. Data on the condition of coral colonies are also recorded. When coral is noticeably bleached, the entire colony is considered affected and is recorded as bleached to the nearest 0.1%. Diseased/dead coral refers to coral skeleton that has recently lost living tissue because of disease or damage, and has not yet been colonized by turf algae.

6. Maximum canopy height - for each soft biota type (e.g., gorgonians, sponges-except encrusting form, algae) the maximum height is recorded to the nearest 1 cm.
7. Abundance and maturity of queen conchs (*Strombus gigas*) - conch encountered within the 25 x 4 m belt transect are enumerated. The maturity of each conch is determined by the presence or absence of a flared lip and labeled mature or immature respectively.
8. Abundance of spiny lobsters (*Panulirus argus*) - a count of the total number of lobsters encountered within the 25 x 4 m belt transect.
9. Abundance of long-spined urchin (*Diadema antillarum*) - a count of the total number of urchins encountered within the 25 x 4 m belt transect.
10. Photos – Two photos are taken in opposite directions at each transects starting position to document the surrounding habitat. Additional photos may be taken to document disease, bleaching or other events of note.
11. Marine debris – type of marine debris within the 25 x 4 m belt transect is noted. The size of the marine debris and area of habitat that it is affecting is also recorded along with a note identifying any flora or fauna that has colonized it.

APPENDIX B: SCUBA Visual Observation Fish Survey Methods

Once in the field, the boat captain navigates to previously selected sites using a handheld GPS unit. On-site, divers are deployed and maintain visual contact with each other throughout the entire census. One diver is responsible for collecting data on the fish communities utilizing the belt-transect visual census technique over an area of 100 m^2 (25 m length x 4 m width). The belt-transect diver obtains a random compass heading for the transect prior to entering the water and records the compass bearing (0-360°) on the data sheet. Visibility at each site must be sufficient to allow for identification of fish at a minimum of 2 m away. Once reasonable visibility is ascertained, the diver attaches a tape measure to the substrate and allows it to roll out for 25 m while they are collecting data.

Although the habitat should not be altered in any manner by lifting or moving structure, the observer should record fish seen in holes, under ledges and in the water column. To identify, enumerate or locate new individuals, divers may move off the centerline of the transect as long as they stay within the 4 m transect width and do not look back along area already covered. The diver is allowed to look forward toward the end of the transect for the distance remaining (i.e., if the diver is at meter 15, he can look 10 m distant, but if he is at meter 23, he can only look 2 m ahead).

On-site, no attempt to avoid structural features within a habitat such as a sand patch or an anchor should be made as these features affect fish communities and are "real" features of the habitats. The only instance where the transect should deviate from the designated path is to stay above 33.5 m (110 ft). The transect should take 15 minutes regardless of habitat type or number of animals present. This allows more mobile animals the opportunity to swim through the transect, thus standardizing the samples collected to allow for comparisons.

Data are collected on the following:

1. Identification - as the tape roles out at a relatively constant speed, the diver records all fish species to the lowest taxonomic level possible that come within 2 m of either side of the transect and towards the end of the transect. To decrease the total time spent writing, four letter codes are used that consist of the first two letters of the genus name followed by the first two letters of the species name. In the rare case that two species have the same four-letter code, letters are added to the species name until a difference occurs. If the fish can only be identified to the family or genus level then this is all that is recorded. If the fish cannot be identified to the family level then no entry is necessary. Individuals too difficult to identify or unique in some manner may be photographed for later clarification.
2. Abundance and size - the number of individuals per species is tallied in 5 cm size class increments up to 35 cm using visual estimation of fork length. If an individual is greater than 35 cm, then an estimate of the actual fork length is recorded.
3. Logistic information - diver name, dive buddy, date, time of survey, site code, transect bearing.

Appendices

APPENDIX C: Diver Survey Species List

Table C.1. Mean (SE) fish density and biomass by depth strata (shallow and upper mesophotic [UM]) from diver surveys (2010 - 2012). Primary trophic group for each species is also provided as herbivore H), invertivore (I), piscivore (P), planktivore (PL), and zooplanktivore (Z).

Genus species	Common name	Trophic Group	Density		Biomass	
			Shallow (±SE)	UM (±SE)	Shallow (±SE)	UM (±SE)
Acanthuridae						
Acanthurus bahianus	ocean surgeonfish	H	0.31 (0.07)	1.06 (0.25)	0.03 (<0.01)	0.07 (0.02)
Acanthurus chirurgus	doctorfish	H	0.61 (0.10)	0.17 (0.08)	0.03 (<0.01)	0.03 (0.02)
Acanthurus coeruleus	blue tang	H	2.84 (0.19)	3.68 (0.49)	0.25 (0.02)	0.71 (0.14)
Apogonidae						
Apogon pseudomaculatus	twospot cardinalfish	PL	<0.01 (<0.01)		<0.01 (<0.01)	
Apogon species	cardinalfish species	I		0.03 (0.03)		<0.01 (<0.01)
Aulostomidae						
Aulostomus maculatus	trumpetfish	P	<0.01 (<0.01)		<0.01 (<0.01)	
Balistidae						
Balistes capriscus	gray triggerfish	I	<0.01 (<0.01)		<0.01 (<0.01)	
Balistes vetula	queen triggerfish	I	0.06 (0.02)	0.41 (0.09)	0.10 (0.03)	0.30 (0.08)
Canthidermis sufflamen	ocean triggerfish	I	0.42 (0.07)	0.11 (0.09)	0.43 (0.10)	0.26 (0.25)
Melichthys niger	black durgon	H	1.43 (0.17)	1.30 (0.32)	0.49 (0.09)	0.16 (0.04)
Xanthichthys ringens	sargassum triggerfish	Z		0.12 (0.11)		0.02 (0.02)
Blennidae						
Ophioblennius macclurei	redlip blenny	H	0.22 (0.04)	0.02 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Parablennius marmoreus	seaweed blenny	I	0.07 (0.02)	1.53 (0.55)	<0.01 (<0.01)	<0.01 (<0.01)
Carangidae						
Carangoides bartholomaei	yellow jack	P	0.02 (0.02)	0.09 (0.05)	<0.01 (<0.01)	0.20 (0.13)
Carangoides ruber	bar jack	P	4.32 (1.52)	4.74 (2.20)	0.06 (0.02)	0.73 (0.28)
Caranx crysos	blue runner	P	0.05 (0.05)	0.21 (0.21)	0.01 (0.01)	0.09 (0.09)
Caranx hippos	crevalle jack	P	0.14 (0.11)	0.11 (0.09)	0.36 (0.27)	0.38 (0.31)
Caranx latus	horse-eye jack	P	0.67 (0.16)	1.05 (0.56)	1.84 (0.47)	2.60 (1.43)
Caranx lugubris	black jack	P	0.26 (0.05)	0.12 (0.08)	0.23 (0.06)	0.12 (0.09)
Seriola dumerili	greater amberjack	P	<0.01 (<0.01)	0.17 (0.08)	0.01 (0.01)	0.30 (0.17)
Carcharhinidae						
Carcharhinus perezi	Caribbean Reef Shark	P	<0.01 (<0.01)		0.28 (0.28)	
Carcharhinus plumbeus	sandbar Shark	P	<0.01 (<0.01)		0.09 (0.09)	
Carcharhinus species	requiem Sharks	P		0.03 (0.03)		0.07 (0.07)
Galeocerdo cuvier	tiger shark	P	<0.01 (<0.01)	0.03 (0.02)	0.43 (0.43)	1.42 (1.06)
Chaetodontidae						
Chaetodon ocellatus	spotfin butterflyfish	I	0.46 (0.06)	0.18 (0.07)	0.03 (<0.01)	<0.01 (<0.01)
Chaetodon sedentarius	reef butterflyfish	I	1.96 (0.11)	2.29 (0.24)	0.05 (<0.01)	0.07 (0.02)
Chaetodon striatus	banded butterflyfish	I	0.07 (0.03)	0.03 (0.03)	<0.01 (<0.01)	<0.01 (<0.01)
Prognathodes aculeatus	longsnout butterflyfish	I	0.52 (0.06)	1.18 (0.17)	0.02 (0.01)	0.01 (<0.01)
Cirrhitidae						
Amblycirrhitus pinos	redspotted hawkfish	I	0.16 (0.04)	0.03 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Diodontidae						
Diodon holocanthus	balloonfish	I	<0.01 (<0.01)	0.05 (0.03)	<0.01 (<0.01)	<0.01 (<0.01)
Diodon hystrix	porcupinefish	I	0.01 (<0.01)	0.05 (0.03)	0.02 (0.01)	0.02 (0.02)
Echeneidae						
Echeneis naucrates	sharksucker	PL	<0.01 (<0.01)		<0.01 (<0.01)	
Gobiidae						
Coryphopterus dicrus	colon goby	I		0.03 (0.03)		<0.01 (<0.01)
Coryphopterus glaucofraenum	bridled goby	I	0.03 (0.01)		<0.01 (<0.01)	
Coryphopterus personatus/ hyalinus	masked/glass goby	I	0.09 (0.09)		<0.01 (<0.01)	
Elacatinus oceanops	neon goby	I	1.36 (0.14)	1.65 (0.72)	<0.01 (<0.01)	<0.01 (<0.01)
Gnatholepis thompsoni	goldspot goby	H	0.40 (0.09)	0.68 (0.50)	<0.01 (<0.01)	<0.01 (<0.01)

Genus species	Common name	Trophic Group	Density		Biomass	
			Shallow (±SE)	UM (±SE)	Shallow (±SE)	UM (±SE)
Haemulidae						
Haemulon melanurum	cottonwick	I	0.01 (0.01)	1.65 (1.59)	<0.01 (<0.01)	0.16 (0.15)
Haemulon plumierii	white grunt	I		0.02 (0.02)		<0.01 (<0.01)
Holocentridae						
Holocentrus adscensionis	squirrelfish	I	0.09 (0.03)	1.49 (0.27)	0.03 (0.02)	0.12 (0.03)
Holocentrus rufus	longspine squirrelfish	I	0.14 (0.03)	0.59 (0.17)	<0.01 (<0.01)	0.04 (0.01)
Myripristis jacobus	blackbar soldierfish	I	0.02 (0.01)	0.26 (0.12)	<0.01 (<0.01)	0.03 (0.01)
Neoniphon marianus	longjaw squirrelfish	I		0.06 (0.04)		<0.01 (<0.01)
Sargocentron bullisi	deepwater squirrelfish	I		0.05 (0.03)		<0.01 (<0.01)
Inermiidae						
Emmelichthyops atlanticus	bonnetmouth	P	11.64 (4.54)	1.52 (1.52)	0.02 (<0.01)	0.01 (0.01)
Inermia vittata	boga	PL	4.74 (2.10)		<0.01 (<0.01)	
Kyphosidae						
Kyphosus sectator	chub (Bermuda/yellow)	H	12.84 (2.45)	4.06 (1.65)	4.82 (1.04)	2.43 (0.97)
Labridae						
Bodianus pulchellus	spotfin hogfish	I	1.11 (0.18)	6.20 (0.57)	0.02 (<0.01)	0.04 (<0.01)
Bodianus rufus	Spanish hogfish	I	7.55 (0.33)	4.50 (0.66)	0.12 (0.02)	0.16 (0.03)
Clepticus parrae	creole wrasse	PL	59.20 (11.94)	97.12 (27.76)	1.87 (0.54)	10.95 (4.61)
Halichoeres bivittatus	slippery dick	I	0.48 (0.08)	0.03 (0.03)	<0.01 (<0.01)	<0.01 (<0.01)
Halichoeres burekae	mardi gras wrasse	I	0.02 (0.01)		<0.01 (<0.01)	
Halichoeres garnoti	yellowhead wrasse	I	0.80 (0.17)	1.96 (0.66)	<0.01 (<0.01)	<0.01 (<0.01)
Halichoeres maculipinna	clown wrasse	I	0.43 (0.08)	0.59 (0.24)	<0.01 (<0.01)	<0.01 (<0.01)
Halichoeres radiatus	puddingwife	I	0.13 (0.03)	0.15 (0.07)	<0.01 (<0.01)	0.06 (0.04)
Thalassoma bifasciatum	bluehead wrasse	I	38.04 (2.31)	15.94 (2.24)	0.04 (<0.01)	0.03 (<0.01)
Lutjanidae						
Lutjanus cyanopterus	cubera snapper	P		0.03 (0.03)		0.02 (0.02)
Lutjanus griseus	gray snapper	I	1.56 (0.26)	4.94 (1.83)	0.56 (0.09)	3.22 (1.98)
Lutjanus jocu	dog snapper	P	0.41 (0.06)	0.44 (0.09)	0.89 (0.16)	1.45 (0.40)
Lutjanus mahogoni	mahogany snapper	P	0.06 (0.04)		0.05 (0.03)	
Ocyurus chrysurus	yellowtail snapper	PL	<0.01 (<0.01)		<0.01 (<0.01)	
Malacanthidae						
Malacanthus plumieri	sand tilefish	I		0.09 (0.07)		0.01 (0.01)
Microdesmidae						
Ptereleotris calliura	blue Goby	Z		0.05 (0.05)		<0.01 (<0.01)
Monacanthidae						
Cantherhines macrocerus	American whitespotted filefish	I	<0.01 (<0.01)		<0.01 (<0.01)	
Cantherhines pullus	orangespotted filefish	I	0.05 (0.02)	0.03 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Mullidae						
Mulloidichthys martinicus	yellow goatfish	I	0.34 (0.08)	13.82 (4.38)	0.06 (0.03)	2.13 (0.91)
Pseudupeneus maculatus	spotted goatfish	I	0.22 (0.09)	1.70 (0.29)	0.02 (<0.01)	0.03 (<0.01)
Muraenidae						
Gymnothorax miliaris	goldentail moray	I	<0.01 (<0.01)		<0.01 (<0.01)	
Gymnothorax moringa	spotted moray	P	0.04 (0.02)	0.02 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Myliodatidae						
Manta birostris	giant manta	PL	<0.01 (<0.01)		0.73 (0.60)	
Opistognathidae						
Opistognathus aurifrons	yellowhead jawfish	PL	0.08 (0.05)	0.15 (0.11)	<0.01 (<0.01)	<0.01 (<0.01)
Ostraciidae						
Acanthostracion polygonius	honeycomb cowfish	I	0.03 (0.01)	0.05 (0.03)	0.01 (<0.01)	<0.01 (<0.01)
Lactophrys bicaudalis	spotted trunkfish	I	<0.01 (<0.01)	0.03 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Lactophrys triqueter	smooth trunkfish	I	0.50 (0.05)	0.24 (0.06)	0.03 (<0.01)	0.02 (<0.01)

Appendices

Genus species	Common name	Trophic Group	Density		Biomass	
			Shallow (±SE)	UM (±SE)	Shallow (±SE)	UM (±SE)
Pomacanthidae						
Centropyge argi	cherubfish	H	0.03 (0.01)		<0.01 (<0.01)	
Holacanthus bermudensis	blue angelfish	I	0.04 (0.02)	0.17 (0.06)	0.01 (<0.01)	0.11 (0.04)
Holacanthus ciliaris	queen angelfish	I	0.16 (0.03)	0.29 (0.08)	0.07 (0.02)	0.18 (0.05)
Holacanthus tricolor	rock beauty	I	0.48 (0.08)	1.29 (0.16)	0.04 (<0.01)	0.17 (0.04)
Pomacanthus paru	French angelfish	I	0.34 (0.05)	0.67 (0.12)	0.31 (0.05)	0.59 (0.11)
Pomacentridae						
Abudefduf saxatilis	sergeant major	I	0.28 (0.19)		<0.01 (<0.01)	
Chromis cyanea	blue chromis	PL	3.30 (0.34)	9.55 (2.18)	0.01 (<0.01)	0.06 (0.02)
Chromis enchrysurus	yellowtail Reeffish	PL		0.11 (0.11)		<0.01 (<0.01)
Chromis insolata	sunshinefish	PL	13.33 (1.87)	97.14 (19.13)	0.03 (<0.01)	0.19 (0.06)
Chromis multilineata	brown chromis	I	72.16 (5.80)	38.67 (9.88)	0.42 (0.09)	0.17 (0.06)
Chromis scotti	purple reeffish	PL	2.73 (0.31)	14.41 (2.44)	<0.01 (<0.01)	0.02 (<0.01)
Microspathodon chrysurus	yellowtail damselfish	H	0.26 (0.04)		0.02 (<0.01)	
Stegastes adustus	dusky damselfish	H	0.56 (0.12)	7.23 (5.30)	<0.01 (<0.01)	<0.01 (<0.01)
Stegastes diencaeus	longfin damselfish	H	0.54 (0.14)	0.12 (0.09)	<0.01 (<0.01)	<0.01 (<0.01)
Stegastes leucostictus	beaugregory	I	0.29 (0.09)	0.32 (0.21)	<0.01 (<0.01)	<0.01 (<0.01)
Stegastes partitus	bicolor damselfish	H	10.99 (1.31)	6.17 (1.89)	0.02 (<0.01)	<0.01 (<0.01)
Stegastes planifrons	threespot damselfish	I	10.63 (0.66)	4.80 (0.78)	0.06 (<0.01)	0.02 (<0.01)
Stegastes variabilis	cocoa damselfish	H	2.00 (0.20)	12.71 (6.71)	<0.01 (<0.01)	0.01 (<0.01)
Scarinae						
Scarus iseri	striped parrotfish	H	1.84 (0.27)	0.88 (0.30)	<0.01 (<0.01)	<0.01 (<0.01)
Scarus taeniopterus	princess parrotfish	H	1.83 (0.45)	3.46 (0.80)	0.06 (0.01)	0.09 (0.02)
Scarus vetula	queen parrotfish	H	1.28 (0.11)	0.59 (0.20)	0.28 (0.03)	0.05 (0.02)
Sparisoma atomarium	greenblotch parrotfish	H	2.08 (0.35)	4.99 (1.34)	<0.01 (<0.01)	0.02 (<0.01)
Sparisoma aurofrenatum	redband parrotfish	H	3.23 (0.32)	3.35 (0.33)	0.08 (0.01)	0.07 (0.02)
Sparisoma radians	bucktooth parrotfish	H	0.10 (0.07)	0.09 (0.09)	<0.01 (<0.01)	<0.01 (<0.01)
Sparisoma viride	stoplight parrotfish	H	1.72 (0.14)	1.33 (0.25)	0.51 (0.06)	0.62 (0.12)
Sciaenidae						
Equetus lanceolatus	jackknife fish	I		0.02 (0.02)		<0.01 (<0.01)
Equetus punctatus	spotted drum	I	<0.01 (<0.01)		<0.01 (<0.01)	
Pareques acuminatus	highhat	I		0.02 (0.02)		<0.01 (<0.01)
Scorpaenidae						
Pterois volitans	red Lionfish	P	<0.01 (<0.01)	0.68 (0.18)	<0.01 (<0.01)	0.03 (0.01)
Serranidae						
Cephalopholis cruentata	graysby	P	0.68 (0.07)	0.80 (0.11)	0.06 (0.01)	0.09 (0.02)
Cephalopholis fulva	coney	I	0.08 (0.02)	0.06 (0.05)	0.01 (<0.01)	<0.01 (<0.01)
Dermatolepis inermis	marbled grouper	P	0.08 (0.02)	0.14 (0.04)	0.01 (<0.01)	0.05 (0.02)
Epinephelus adscensionis	rock hind	I	0.08 (0.03)	0.46 (0.10)	<0.01 (<0.01)	0.15 (0.04)
Epinephelus guttatus	red hind	I	0.08 (0.02)	0.39 (0.11)	0.05 (0.02)	0.39 (0.12)
Hypoplectrus species	hamlet species	I	<0.01 (<0.01)		<0.01 (<0.01)	
Liopropoma rubre	peppermint basslet	P	0.04 (0.01)	0.06 (0.04)	<0.01 (<0.01)	<0.01 (<0.01)
Mycteroperca bonaci	black grouper	P	0.07 (0.02)	0.46 (0.08)	0.26 (0.11)	6.14 (1.78)
Mycteroperca interstitialis	yellowmouth grouper	P	0.97 (0.08)	1.91 (0.21)	0.47 (0.06)	1.79 (0.36)
Mycteroperca phenax	scamp	P	0.04 (0.01)	0.08 (0.05)	0.02 (<0.01)	0.04 (0.03)
Mycteroperca tigris	tiger grouper	P	0.44 (0.05)	0.70 (0.12)	0.50 (0.10)	1.51 (0.35)
Mycteroperca venenosa	yellowfin grouper	P	0.02 (<0.01)	0.46 (0.10)	0.01 (<0.01)	1.13 (0.28)
Paranthias furcifer	Atlantic creolefish	PL	75.46 (11.96)	192.47 (43.95)	7.21 (1.20)	19.28 (6.10)
Serranus annularis	orangeback bass	P	<0.01 (<0.01)	0.02 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Serranus baldwini	lantern bass	P		0.03 (0.02)		<0.01 (<0.01)
Serranus species	seabass species	P		0.06 (0.05)	<0.01 (<0.01)	
Serranus tabacarius	tobaccofish	P	0.01 (<0.01)			<0.01 (<0.01)

Genus species	Common name	Trophic Group	Density		Biomass	
			Shallow (±SE)	UM (±SE)	Shallow (±SE)	UM (±SE)
Sparidae						
Calamus calamus	saucereye porgy	I	0.01 (<0.01)	0.02 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Calamus nodosus	knobbed porgy	I	0.03 (0.01)	0.38 (0.09)	0.01 (<0.01)	0.11 (0.03)
Calamus species	porgy species	I		0.05 (0.05)		0.04 (0.04)
Sphyraenidae						
Sphyraena barracuda	great barracuda	P	1.24 (0.14)	0.53 (0.10)	2.04 (0.28)	1.87 (0.57)
Synodontidae						
Synodus intermedius	sand diver	P	0.02 (0.01)	0.02 (0.02)	<0.01 (<0.01)	<0.01 (<0.01)
Synodus saurus	bluestriped lizardfish	P	<0.01 (<0.01)		<0.01 (<0.01)	
Tetraodontidae						
Canthigaster jamestyleri	goldface toby	I	<0.01 (<0.01)		<0.01 (<0.01)	
Canthigaster rostrata	sharpnose puffer	I	6.27 (0.29)	6.21 (1.14)	0.02 (<0.01)	0.01 (<0.01)
Sphoeroides spengleri	bandtail puffer	I	<0.01 (<0.01)		<0.01 (<0.01)	

APPENDIX D: Specifications for the Remotely Operated Vehicle (ROV) Used to Survey Fish and Benthic Communities in the Flower Garden Banks National Marine Sanctuary

VEHICLE DESCRIPTION

The Super Phantom S2 is a powerful, versatile remotely operated vehicle (ROV) with high reliability and mobility. This light weight system can be deployed by two operators and is designed as an underwater platform which provides support services including color video, digital still photography, navigation instruments, lights and a powered tilt platform. A wide array of specialty tools and sampling devices are available. The UVP/UNCW Phantom S2 remotely operated vehicle (ROV) was built and purchased in 1987 from Deep Ocean Engineering in San Leandro, CA. The vehicle was upgraded to the high voltage DC power Spectrum series in 1995. The basic configuration of the ROV provides color video, digital still photos, laser scaling device, and position information of the ROV and support ship, vehicle heading, and vehicle depth. Spare wires with access in a topside junction box are available for the integration of scientist provided instrumentation or the additional equipment described below. The number of spare wires and payload available on the vehicle will limit how many pieces of equipment can be mounted on the ROV at any one time; therefore, it is critical that the scientist contact the ROV operators well before the mission if other equipment is to be added to the ROV. Basic characteristics of the vehicle are provided in Table D.1.

Table D.1. ROV specifications.

Specification	Measurement	
Height	24 in	61 cm
Length	55 in	140 cm
Width	33 in	84 cm
Weight	170 lb	77 kg
Payload	~33 lb	14.8 kg
Depth rating	1,500 ft	475 m
Cable length	1100 ft	335 m
Power requirement	4.5 KVA (requires dedicated 110 VAC/30 amp circuit breaker)	
Horizontal speed	2 kts	
Vertical speed	~1 ft/s	
Thrusters	Two 1/2 HP horizontal motors providing 75 lb. of forward thrust. Two 1/4 HP "vertran" motors providing 30 lb of vertical thrust	

CAMERAS AND LIGHTS

Color Video – A Sony high resolution, single-chip color camera with auto iris, 12:1 zoom, and auto/manual focus provides video documentation during ROV operations. The video signal is routed from the ROV console to the On-Screen Display (OSD) video overlay device to a Panasonic mini-DV cassette recorder (one hour record time), then to a Panasonic DVD recorder (two hours record time), and finally to a JVC 13" color monitor. This ensures that if you are seeing the video and overlay information on the monitor that it is being correctly routed through the video recording equipment. This does not ensure that the video recording equipment is recording. The OSD provides a title page with information such as mission number, P.I. name, support vessel name, general location, ROV Dive number, mini-DV tape number, DVD disk number, and any other pertinent information that the scientist requires. This information is recorded as a header at the beginning of each mini-DV tape and DVD disk. The OSD also provides data overlay on the video including time, date, ROV heading ribbon, ROV numeric heading in degrees, ROV depth in feet, and ROV umbilical cable turns counter for the pilot. All video documentation can be geo-referenced to ROV position by matching the time and date on

Table D.2. Sony NTSC video format specifications.

Specification	Measurement
Image Sensor	1/3" IT CCD
Pixels/effective pixels	410 x 380
Picture Elements	768 (H) x 484 (V)
Horizontal Resolution (Center)	>460 TV Lines
Vertical Resolution (Center)	>350 TV Lines
Lens	12X zoom, auto focus, f1.8 to f2.7
Diagonal Angle of view in air	Wide angle: 117º, Tele: 10º
Diagonal Angle of view in water	Wide angle: 79º, Tele: 7.4º
Minimum Illumination	2 LUX (F1.8)
Signal to Noise Ratio	>48 dB
White Balance	TTL Auto tracking

the video to the navigation files (Excel spreadsheet) provided to the scientists by the ROV operations staff. An omni-directional microphone is available for audio annotation onto the recording media by the scientists. The Sony NTSC video format specifications shown in Table D.2.

Digital Still Photographs – Geo-referenced digital still pictures are acquired with an Insite-Tritech Scorpio Plus digital still color TV camera and strobe. The camera provides "Through the lens" color video to the ROV video monitor that not only allows the operator to accurately frame still images, but it also can be used for video documentation. Note that only one camera (either the color video camera or the digital still camera) on the ROV can be monitored at one time. Switching between these two cameras is done at the ROV control console by the ROV pilot at the request of the scientific observer. This camera features a 4X zoom lens and corrected optics that virtually eliminates geometric and chromatic distortion. The internal electronics and imaging device is basically a Nikon Coolpix 995 digital still camera which is controlled by a laptop PC via RS-485 communications using a shielded-twisted pair of wires in the ROV umbilical cable. In fine resolution setting, the 1 gigabyte, onboard compact flash card can store 664 images in JPG format (approximately 1.0 Mb each), which are provided to the scientist on CD or DVD media. Images are downloaded from the camera at the end of each day via a USB/ Ethernet cable into the laptop, and are stored in two locations at all times. The strobe unit, also built by Insite-Tritech, is a 62 Watt Seconds flash that is TTL auto controlled by the camera. The strobe unit is powered by four rechargeable AA batteries, which allows the strobe to be fired approximately 280 times per ROV dive. A running tally of the number of images taken during each dive is monitored, and the batteries will be replaced between dives if there is any possibility that the next dive will require additional strobe power. The digital still camera can be mounted either on the tilt platform or in a vertical configuration (straight down) on the crash frame. Additional digital still camera specifications are shown in Table D.3.

Table D.3. Digital still camera specifications.

Specification	Measurement
Image Sensor	1 1/8" High density CCD
Image size	3.34 megapixel ultrahigh definition (2048 x 1536 pixel)
Lens	4X Zoom-Nikkor, f8, 32 mm
Focus range	30 cm to ∞

Dual Red Laser Scaling Device – The two lasers used for scaling objects underwater were made by Harbor Branch Oceanographic Institution. They are mounted in a precision machined aluminum block to maintain the lasers in a parallel orientation at exactly 10 cm. The output power is 8.6 mW. The lasers are mounted on the ROV tilt platform directly under the video camera and are usually in the video and digital still frame unless the ROV is very close to the object or the cameras are zoomed in. The laser scaling device can be moved to the digital still camera when it is mounted in the vertical position on the ROV. This provides accurate scaling over the entire image in either axis.

Lights – Two 250-watt tungsten-halogen lights made by Deep Sea Power and Light are mounted on top of the ROV tilt platform and provide illumination for the color video camera. The light output can be controlled by a three position dimmer switch, which is useful for viewing light colored material such as sand. Red colored lenses can be mounted onto the light housings to provide a light frequency that is less distracting to fish behavior.

Navigation and Tracking – The ROV uses an integrated navigation system consisting of Hypack Max software on a Dell 1.6 GHz computer, ORE Offshore 4410C Trackpoint II Underwater Acoustic Tracking System with an ORE Offshore 4377A transponder with depth telemetry, Northstar 951XD differential GPS, and Azimuth 1000 digital compass. This system provides real time tracking of the ROV and ship to the ROV operator and the support vessel's bridge for navigation. ROV personnel install a Northstar DGPS antenna and an ORE hydrophone on the vessel (See Figure 3) and survey their positions with respect to a reference point at the center of the vessel. The hydrophone mounting alignment is checked at the dock using submerged transponders. DGPS antenna and hydrophone offsets, as well as ship dimensions, are entered into the Trackpoint II. The Trackpoint II interrogates the ORE 4377A-SL transponder on the ROV. Using the ORE three-element hydrophone, Trackpoint II determines slant range, bearing, and depth. The real-time Hypack navigation screen (See Figure 4) accurately displays the ship (to scale) with proper position and heading, and the position of the ROV. Ship and ROV positions are logged and processed for each dive and provided to the scientist in an Excel file. Geo-referenced .tif files obtained with multibeam or side scan sonar can be entered into Hypack as background files to display target sites and features of interest to aid in ROV and support vessel navigation. The Trackpoint II acoustic tracking system can track up to 6 targets at one time, and additional Benthos UAT-376 transponders (transmit 25 or 27 kHz) and Helle pingers (27 kHz) are available to relocate instrumentation packages deployed in the ocean. ORE Offshore 4410C Trackpoint II acoustic tracking system specifications are shown in Table D.4. ORE Offshore 4377A transponder specifications are shown in Table D5.

Table D.4. ORE Offshore 4410C Trackpoint II acoustic tracking system specifications.

Specification	Measurement
Horizontal Pinger Position Accuracy	± 0.75% RMS of Slant Range (depression angle > 45º from horizontal)
Horizontal Transponder Position Accuracy	Absolute Accuracy: ± 0.5% RMS of Slant Range Repeatability Accuracy: ± 0.5% RMS of Slant Range
Slant Range Accuracy	± 1 meter (assuming correct speed of sound input)
Slant Range Resolution	0.3 meters
Receive Signal Frequency	22-30 kHz in 500 Hz increments
Receive Signal Pulse Duration	1.33 ms minimum
Receive Signal/Noise Ratio	>40 dB at wideband filter
Transmitter Output Frequency	4.5-30 kHz in 500 Hz increments
Transmitter Output Pulse Width	1 to 15 ms in 0.1 ms increments
Transmitter Output Repetition Rate	1 to 20 seconds
Transmitter Output Power	100 or 500 watts into 300Ω, user selectable

Table D.5. ORE Offshore 4377A transponder specifications.

Specification	Measurement
Receive Frequency	19 kHz
Receive Pulse Width	5 ms minimum
Transmit Frequency for Navigation	24 kHz
Transmit Frequency for Depth Telemetry	23 kHz
Transponder Turn Around Time	15 ms
Transponder Lock-Out Time	1.9 seconds (Minimum interrogation rate is 2 sec.)
Transponder Depth Rating	1000 meters

APPENDIX E: ROV Species List

Table E.1. Mean (SE) fish density and biomass by habitat type from ROV surveys (2010 - 2012). AN=algal nodule, CA=coralline algal reef, DR=deep reef, SB=soft bottom. Primary trophic group for each species is labled as herbivore H), invertivore (I), piscivore (P), and planktivore (PL).

Genus species	Common name	Trophic Group	Density				Biomass			
			AN (±SE)	CA (±SE)	DR (±SE)	SB (±SE)	AN (±SE)	CA (±SE)	DR (±SE)	SB **(±SE)**
Acanthuridae										
Acanthurus coeruleus	blue tang	H	0.07 0.06	0.03 0.04			13.94 10.63	13.71 13.50		
Apogonidae										
Apogon UNK	cardinalfish species	I			0.01 0.01				0.05 0.05	
Balisitidae										
Balistes vetula	queen triggerfish	I	0.06 0.03	0.02 0.02			40.62 20.47	8.31 8.31		
Canthidermis sufflamen	ocean triggerfish	I		<0.01 <0.01				1.04 1.04		
Bothidae										
Bothus species	flounder species	I			0.01 0.01	0.01 0.01			0.69 0.42	0.52 0.52
Carangidae										
Caranx lugubris	black jack	P		0.01 0.01	0.02 0.02			18.20 18.20	32.28 32.28	
Caranx species	jack species	P	0.07 0.07				41.57 41.57			
Seriola dumerili	greater amberjack	P	0.04 0.02	0.04 0.04	<0.01 <0.01		55.01 31.10	81.91 81.91	14.23 14.23	
Seriola rivoliana	almaco jack	P	0.01 0.01	0.17 0.34	0.57 0.50		8.54 8.54	998.16 636.78	985.18 839.03	
Chaetodontidae										
Chaetodon ocellatus	spotfin butterflyfish	I			<0.01 <0.01				0.29 0.29	
Chaetodon sedentarius	reef butterflyfish	I	0.32 0.10	0.05 0.33	0.39 0.20	0.02 0.02	6.14 2.46	5.08 1.04	16.05 11.97	0.29 0.29
Prognathodes aculeatus	longsnout butterflyfish	I	0.06 0.03	0.02 0.05	0.10 0.10		0.33 0.21	0.28 0.15	0.08 0.08	
Prognathodes aya	bank butterflyfish	I		0.03 0.07	0.11 0.04			0.81 0.33	2.48 0.80	
Diodontidae										
Diodon holocanthus	balloonfish	I	0.02 0.02				6.72 6.72			
Chilomycterus antillarum	web burrfish	I	0.01 0.01	0.01 0.01			0.18 0.18	1.46 1.46		
Gobiidae										
Gobiidae species	goby species	I		0.01 0.01	<0.01 <0.01			0.01 0.01	<0.01 <0.01	
Ptereleotris helenae	hovering goby	PL		0.02 0.03				0.10 0.06		
Haemulidae										
Haemulon melanurum	cottonwick	I		0.02 0.03	0.01 0.01			4.58 2.98	2.00 2.00	
Holocentridae										
Sargocentron bullisi	deepwater squirrelfish	I	0.06 0.03	0.05 0.19	0.16 0.10		1.60 1.01	4.19 1.12	1.88 0.88	
Corniger spinosus	spinycheek soldierfish	I		0.01 0.01	0.05 0.01			1.40 1.01	5.75 1.78	
Holocentrus adscensionis	squirrelfish	I	0.13 0.08	0.07 0.21	0.20 0.20		12.15 8.63	22.65 7.60	8.06 8.06	
Holocentrus species	squirrelfish species	I		0.01 0.01				0.36 0.36		
Labridae										
Bodianus rufus	Spanish hogfish	I	0.05 0.03				0.80 0.73			
Bodianus pulchellus	spotfin hogfish	I	0.24 0.07	0.12 0.64	0.16 0.10		2.37 1.34	7.50 2.44	2.12 1.06	
Decodon puellaris	red hogfish	I		0.06 0.14	0.29 0.06	0.12 0.04		0.65 0.31	2.70 0.68	0.65 0.21
Halichoeres bathyphilus	greenband wrasse	I	0.04 0.03	0.04 0.10	0.06 0.03	0.01 0.01	0.23 0.17	0.67 0.29	0.32 0.16	0.07 0.07
Halichoeres species	wrasse species	I	0.01 0.01				<0.01 <0.01			

Genus species	Common name	Trophic Group	Density				Biomass			
			AN (±SE)	CA (±SE)	DR (±SE)	SB (±SE)	AN (±SE)	CA (±SE)	DR (±SE)	SB (±SE)
Lutjanidae										
Lutjanus campechanus	red snapper	I	0.08 0.06	0.11 0.36	1.01 0.30	0.59 0.43	170.83 132.25	404.05 148.47	1600.70 416.67	972.72 760.12
Rhomboplites aurorubens	vermillion snapper	I		0.70 0.80	0.05 0.03			30.00 23.52	24.07 19.23	
Malacanthidae										
Malacanthus plumieri	sand tilefish	I	0.03 0.02	0.02 0.04		0.04 0.02	6.58 5.01	5.90 3.55		1.21 0.93
Mullidae										
Mullus auratus	red goatfish	I				0.01 0.01				0.19 0.19
Pseudupeneus maculatus	spotted goatfish	I	0.01 0.01				0.39 0.39			
Muraenidae										
Gymnothorax species	moray eel species	P			<0.01 <0.01				1.11 1.11	
Ophidiidae										
Brotula barbata	bearded brotula	I			<0.01 <0.01				0.03 0.03	
Opistognathidae										
Opistognathus aurifrons	yellowhead jawfish	PL	0.15 0.06	0.02 0.02			0.56 0.22	0.09 0.09		
Opistognathus UNK	jawfish species	PL	0.18 0.14				0.70 0.52			
Ostraciidae										
Acanthostracion polygonius	honeycomb cowfish	I		0.01 0.01				2.09 2.09		
Acanthostracion quadricornis	scrawled cowfish	I	0.01 0.01				0.73 0.73			
Pomacanthidae										
Centropyge argi	cherubfish	H	0.85 0.29	0.09 0.17			0.58 0.20	0.23 0.13		
Holacanthus bermudensis	blue angelfish	I	0.02 0.02	0.01 0.01			10.82 8.28	5.98 5.98		
Holacanthus tricolor	rock beauty	I	0.04 0.03	0.03 0.05			1.12 0.99	7.86 7.14		
Pomacanthus paru	French angelfish	I	0.03 0.03				28.98 25.94			
Pomacentridae										
Chromis cyanea	blue chromis	PL	0.02 0.01				0.05 0.05			
Chromis enchrysura	yellowtail reeffish	PL	0.67 0.32	1.81 4.06	0.32 0.11		0.91 0.68	39.98 9.98	9.35 4.34	
Chromis insolata	sunshinefish	PL	4.03 1.42	0.78 2.09	0.12 0.10	0.01 0.01	8.91 3.41	8.68 4.98	0.82 0.78	<0.01 <0.01
Chromis scotti	purple reeffish	PL	0.15 0.13				0.08 0.07			
Stegastes partitus	bicolor damselfish	H	0.14 0.07	0.01 0.01	0.01 0.01		0.31 0.24	0.09 0.09	0.01 <0.01	
Stegastes species	damselfish species	H		0.03 0.03				0.38 0.33		
Priacanthidae										
Priacanthus arenatus	bigeye	PL	0.04 0.03	0.02 0.04	0.17 0.07	0.02 0.02	4.32 3.10	3.20 1.35	12.32 7.50	1.80 1.80
Pristigenys alta	short bigeye	PL		0.02 0.03	0.04 0.01	0.03 0.03		2.03 1.54	2.52 1.07	1.80 1.80
Scarinae										
Sparisoma atomarium	greenblotch parrotfish	H	0.40 0.12	0.02 0.04	<0.01 <0.01		0.72 0.30	0.15 0.11	<0.01 <0.01	
Sparisoma viride	stoplight parrotfish	H	0.01 0.01				10.88 10.88			
Sparisoma species	parrotfish species	H		0.01 0.01				0.61 0.56		
Sciaenidae										
Equetus lanceolatus	jackknife fish	I		0.02 0.03	0.02 0.02			0.39 0.28	0.40 0.31	
Pareques umbrosus	cubbyu	I			0.01 0.01				0.27 0.19	

Genus species	Common name	Trophic Group	Density				Biomass			
			AN (±SE)	CA (±SE)	DR (±SE)	SB (±SE)	AN (±SE)	CA (±SE)	DR (±SE)	SB **(±SE)**
Scorpaenidae										
Pterois volitans	red lionfish	P	0.01 0.01				1.17 1.17			
Scorpaena dispar	hunchback scorpion-fish	I			0.04 0.02				8.16 4.63	
Serranidae										
Anthias tenuis	threadnose bass	I	3.82 2.75	17.17 63.28	68.89 13.85	0.93 0.71	2.53 1.56	80.64 21.42	44.31 10.83	0.40 0.30
Cephalopholis cruentata	graysby	P	0.01 0.01	0.01 0.02	0.01 0.01		1.03 1.03	9.69 7.96	2.14 2.14	
Dermatolepis inermis	marbeled grouper	P	0.01 0.01	0.01 0.01			1.33 1.33	1.58 1.17		
Epinephelus adscensionis	rock hind	I		0.01 0.01				0.25 0.25		
Epinephelus guttatus	red hind	I		0.01 0.02				16.66 10.39		
Gonioplectrus hispanus	spanish flag	P		0.01 0.02	0.32 0.19			0.73 0.45	26.80 14.17	
Hemanthias leptus	longfin bass	P	0.02 0.01	0.16 0.09	0.32 0.15		0.05 0.05	1.81 1.08	4.14 1.55	
Hemanthias vivanus	red barbier	O		0.01 0.01	0.03 0.02	0.05 0.05		0.07 0.07	0.46 0.32	0.39 0.39
Hyporthodus flavolimbatus	yellowedge grouper	I		<0.01 <0.01	0.02 0.01			10.17 10.17	40.35 28.48	
Liopropoma eukrines	wrasse bass	I	0.01 0.01	0.02 0.08	0.13 0.03		0.33 0.33	0.82 0.34	2.45 0.78	
Mycteroperca bonaci	black grouper	P		0.01 0.02	0.01 0.01			35.49 23.39	9.82 9.82	
Mycteroperca interstitialis	yellowmouth grouper	P	0.02 0.02	0.01 0.03	0.03 0.02		45.58 31.80	53.33 23.59	74.93 39.81	
Mycteroperca microlepis	gag grouper	P			0.01 0.01				9.23 9.23	
Mycteroperca phenax	scamp	P	0.10 0.05	0.05 0.22	0.27 0.10		152.88 99.05	268.33 70.50	356.68 110.15	
Mycteroperca venenosa	yellowfin grouper	P	0.01 0.01	0.01 0.01	<0.01 <0.01		18.20 18.20	18.58 13.40	6.40 6.40	
Mycteroperca species	grouper species	P	0.01 0.01	0.01 0.03	0.01 0.01		8.62 8.62	38.57 28.98	15.95 14.34	
Paranthias furcifer	Atlantic creolefish	PL	2.75 1.77	0.87 1.96	1.30 0.70		162.92 111.69	116.85 66.49	157.23 104.18	
Pronotogrammus martinicensis	roughtongue bass	PL	4.89 3.01	7.43 51.94	56.51 6.48	0.02 0.02	43.02 34.91	302.31 57.58	270.77 33.23	0.26 0.26
Serranus annularis	orangeback bass	P	0.38 0.12	0.02 0.04			1.21 0.71	0.10 0.05		
Serranus notospilus	saddle bass	P		0.01 0.01	<0.01 <0.01	0.09 0.07		0.18 0.18	0.08 0.05	0.44 0.35
Serranus phoebe	tattler	I	0.05 0.02	0.04 0.13	0.26 0.10	0.07 0.04	0.29 0.14	1.60 0.62	3.57 0.97	0.40 0.26
Serranus species	seabass species	P	0.02 0.01			0.01 0.01	0.06 0.05			0.25 0.25
Sphyraenidae										
Sphyraena barracuda	great barracuda	P	0.01 0.01	0.01 0.01			10.73 10.73	11.40 11.40		
Synodontidae										
Synodus species	lizardfish species	P	0.01 0.01		0.09 0.04	0.04 0.03	0.05 0.05		1.27 0.65	0.32 0.23
Tetraodontidae										
Canthigaster jamestyleri	goldface toby	I		0.01 0.01	<0.01 <0.01			0.01 0.01	0.03 0.03	
Canthigaster rostrata	sharpnose puffer	I	0.03 0.02	0.01 0.01			0.02 0.01	<0.01 <0.01		
Triglidae										
Prionotus species	sea robin species	I				0.01 0.01				0.05 0.05

Appendices

APPENDIX F: SPECIFICATIONS FOR THE SPLITBEAM ECHOSOUNDERS USED TO MAP FISH ABUNDANCE IN THE FLOWER GARDEN BANKS NATIONAL MARINE SANCTUARY

SYSTEM DESCRIPTION AND OPERATING PARAMETERS

The Simrad EK60 scientific splitbeam echosounder is a versatile survey tool in fisheries and ecosystem research. ER60 is the software that controls the EK60 system and allows for the specification of system parameters to optimize the vertical resolution for detecting individual fish targets while minimizing the influence of external noise and interference caused by electrical signals on the ship, air bubbles and other interference in the water column (Simmonds and MacLennan, 2005). A table of system parameters used during survey missions on the NOAA Ship R/V NANCY FOSTER and MANTA is provided (Table F.1). Sound velocity and sound absorption are approximate with exact values specific to the environmental conditions during each survey.

Table F.1. System acquisition parameters used in ER60 software.

Parameter	NOAA Ship	
	R/V NANCY FOSTER	*R/V MANTA*
Frequencies (kHz)	38, 120, 200	120
Transducer depth (m below surface)	3.49	1.4
Sound velocity (m/s, nominal)	1535	1535
Sound absorption (dB/km, nominal)	0.04	0.04
Receiver gain (dB)	26.8	26.8
Transmit power (W ref: dB 1m)	600, 500, 120	500
Pulse duration/width (ms)	0.256, 0.128, 0.128	0.128
Pule repetition rate (Hz)	3-10	5-10

SYSTEM CALIBRATION

During each survey mission, the splibeam echosounder was calibrated using methods described in Foote et al. (1987). Briefly, a standard 38.1 mm diameter tungsten-carbide calibration sphere is hung below the transducer. This standard target has a known acoustic reflectivity and intensity (target strength) that is a function of size, materials and environmental conditions. Prior to lowering the sphere, the target strength is calculated using theoretical models that consider temperature, salinity, depth and operating frequency. The LOBE program in ER60 software is used to acquire position and target strength for the sphere. The calibration sphere is systematically moved through the beam, covering forward, aft and each side of the beam quadrants. Upon completion, the LOBE program calculates the system receiver gain to bring the observed target strength in concordance with the theoretical target strength of the sphere.

ESTIMATING FISH SIZE FROM SPLITBEAM ACOUSTIC BACKSCATTER AND TARGET STRENGTH

Acoustic echoes from individual fish insonified by the acoustic beam are positioned using phase detection across four quadrants in the acoustic beam. Multiple echoes from individual fish are consolidated and the average target strength (in dB) is calculated. We adopted a generalized target strength-fish length relationship based on Love (1977) as adapted in Kracker et al. (2011). The total length (TL, in cm) of a fish is an exponential function of the observed target strength (TS, Figure F.1).

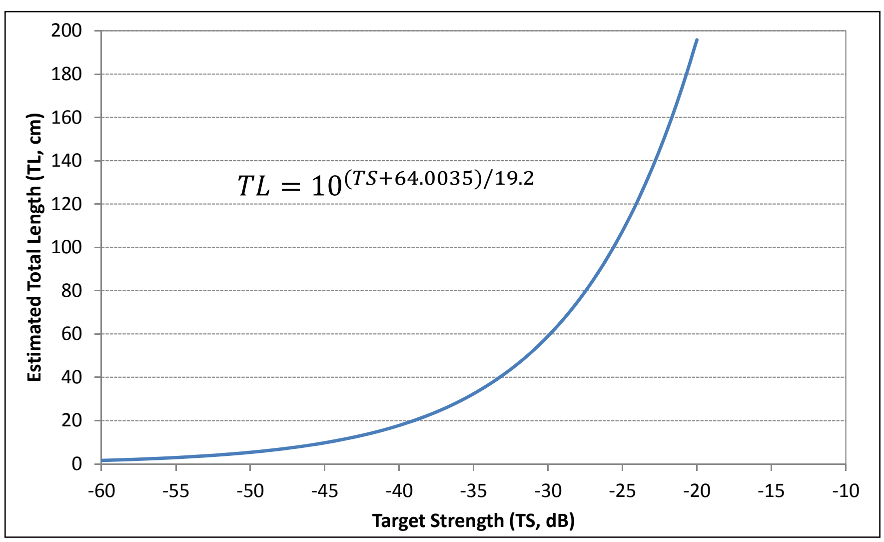

Figure F.1. Relationship between target strength and estimated total length.

ACCOUNTING FOR DETECTION VARIATION OVER DEPTH AND CALCULATING FISH DENSITIES

The transducer beam has a swath angle of approximately 7°, which results in increased detection of fishes with range from the transducer. To correct for varying detections with range, we applied a weighting function to standardize the counts of fish throughout the depth range sampled to a constant swath of 1 m:

$$C_w = 2 \times range \times \tan(0.5BA)^{-1}$$

where C_w is a weighted value that is dependent on range from the transducer and the tangent of half beam angle (*BA*; e.g., 7°). Summing the weighted counts along a 100 m segment of a transect produces densities that have the units of fish 100 m^{-2}. The transect segment is arbitrarily set prior to the beginning of this study.

REFERENCES FOR APPENDICES F AND G

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