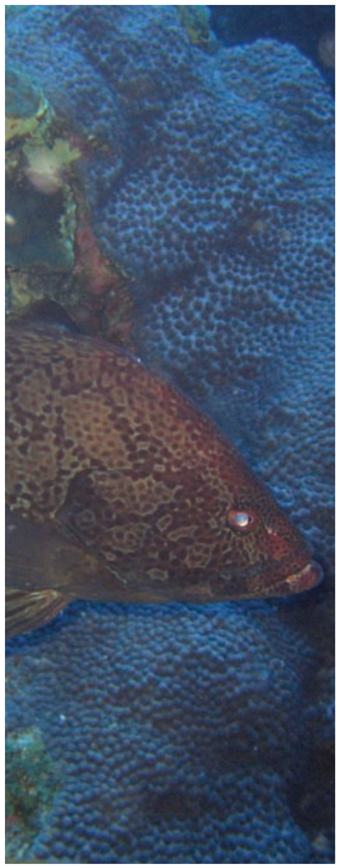
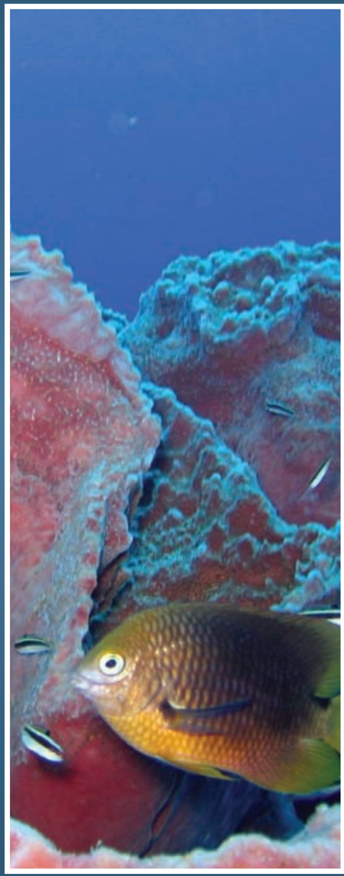
Biogeographic Characterization of Fish Communities and Associated Benthic Habitats within the Flower Garden Banks National Marine Sanctuary

Sampling Design and Implementation of Scuba Surveys on the Coral Caps





June 2009

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National Marine Sanctuaries
Flower Garden Banks

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Cover photos (left to right): Image of Dermatolepis inermis *(CCMA);* Madracis mirabilis *and sponges (E. Hickerson); diver and* Caranx latus *(CCMA); and* Stegastes planifrons, *juvenile* Thalassoma bifasciatum *and large barrel sponge (E. Hickerson).*

Biogeographic Characterization of Fish Communities and Associated Benthic Habitats within the Flower Garden Banks National Marine Sanctuary:

Sampling Design and Implementation of Scuba Surveys on the Coral Caps

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About this Document

This biogeographic characterization is a project formed from the Office of National Marine Sanctuaries - National Centers for Coastal Ocean Science Long-term Agreement. This agreement originates from the common belief of these two programs within NOAA's National Ocean Service that it is critical to have incorporated the best available science when making management decisions regarding our nation's coastal waters. This statement is echoed as each sanctuary undergoes a revision process to their management plans. The revision process evaluates the degree to which each sanctuary meets its goals and allows an opportunity for the public to determine if there are new directions or issues they feel the sanctuary should address. The need for ecosystem based management informed by an adequate understanding of sanctuary living marine resources is consistently raised as a pressing issue in this process. The current document is one of a series of such projects aimed at providing sanctuary managers critical information on the distribution of those resources relevant to the regions they manage. This NOAA Technical Memorandum focuses on providing a spatial and quantitative characterization of the fish communities associated with the coral cap regions of the Flower Garden Banks National Marine Sanctuary. Also included, is a characterization of associated benthic habitats and sections detailing sampling design, methods and the creation of a habitat map essential in selecting sampling strata.

Related projects funded through the Office of National Marine Sanctuaries - National Centers for Coastal Ocean Science Long-term Agreement:

Biogeographic Assessment off North/Central California for the Office of National Marine Sanctuaries: Phases 1 & 2 - Marine Birds, Fishes and Mammals - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/ca_nms2.html

Biogeographic Assessment off North/Central California in support of the National Marine Sanctuaries of Cordell Bank, Gulf of the Farallones and Monterey Bay. Phase II Environmental setting and update to marine birds and mammals http://ccma.nos.noaa.gov/ecosystems/sanctuaries/ca_nms2.html

Biogeographic Assessment of the Channel Islands National Marine Sanctuary to Support Boundary Alternative Assessments - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/chanisl_nms.html

Boundary Options for a Research Area within Grays Reef National Marine Sanctuary - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/grays_boundary.html

Characterization of the Fish, Benthos and Marine Debris at the Grays Reef National Marine Sanctuary - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/grays_nms.html

Biogeographic Assessment of Stellwagen Bank National Marine Sanctuary - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/stellwagen_nms.html

Biogeographic Assessment of the Northwestern Hawaiian Islands to Support the Papahanoumokuakea Marine National Monument - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/nwhi.html

Oceanographic Assessment of Olympic Coast National Marine Sanctuary - http://ccma.nos.noaa.gov/ecosystems/sanctuaries/olympic_nms.html

For more information on this effort please visit the NCCOS CCMA Biogeography Branch web page dedicated to this project at http://ccma.nos.noaa.gov/ecosystems/sanctuaries/fgb_nms.html or direct questions and comments to:

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Executive Summary

OVERVIEW

The Flower Garden Banks National Marine Sanctuary (FGBNMS) is located in the northwestern Gulf of Mexico approximately 180 km south of Galveston, Texas. The sanctuary's distance from shore combined with its depth (the coral caps reach to within approximately 17 m of the surface) result in limited exposure of this coral reef ecosystem to natural and human-induced impacts compared to other coral reefs of the western Atlantic. In spite of this, the sanctuary still confronts serious impacts including hurricanes events, recent outbreaks of coral disease, an increase in the frequency of coral bleaching and the massive *Diadema antillarum* die-off during the mid-1980s. Anthropogenic impacts include large vessel anchoring, commercial and recreational fishing, recreational scuba diving, and oil and gas related activities. The FGBNMS was designated in 1992 to help protect against some of these impacts.

Basic monitoring and research efforts have been conducted on the banks since the 1970s. Early on, these efforts focused primarily on describing the benthic communities (corals, sponges) and providing qualitative characterizations of the fish community. Subsequently, more quantitative work has been conducted; however, it has been limited in spatial scope. To complement these efforts, the current study addresses the following two goals put forth by sanctuary management: 1) to develop a sampling design for monitoring benthic fish communities across the coral caps; and 2) to obtain a spatial and quantitative characterization of those communities and their associated habitats.

SAMPLING DESIGN

In order to meet the first goal, a sampling design was produced to ensure surveys were conducted in an efficient manner. Sampling occurred in 2006 and 2007 allowing an analysis of the data collected from the first year to improve the design subsequently implemented in year two. Candidate sampling designs ranged from simple random sampling to a variety of stratified random sampling scenarios. The designs were selected based on how well they addressed the following four sanctuary objectives: 1) to determine long-term changes in fish community structure using metrics of diversity, density and trophic ratios; 2) to determine long-term changes in density and mean-size of selected economically important taxa; 3) to determine the relationship between physical measures such as habitat type, depth, slope and geographic location with the associated fish community using metrics of diversity, density and biomass; and 4) to find better ways to collect information such that the probability of detecting change over time or space is increased.

This report focuses on sampling the Shallow Coral Cap (SCC) region on East Bank (EB) and West Bank (WB), as this region is readily sampled via basic scuba diving techniques (using standard air or Nitrox). The SCC is the region of the coral caps shallower than 33.5 m and composes 90% of the total coral cap area. Using scuba researchers are able to survey three to four sites in this region per day. The sampling design recommended here can also be used to survey the remaining 10% of the coral caps; however, more technical scuba diving techniques would need to be employed. These necessitate additional equipment and training for the divers and allow fewer stations to be surveyed.

During 2006, a total of 73 sites were surveyed, 49 sites on EB and 24 sites on WB randomly positioned within a strata. In 2007, 32 sites primarily located along the southern portion of EB were surveyed before the mission was aborted due to inclement weather. Fish data were collected at each site along a 25 x 4 m transect and benthic data were collected within four randomly positioned 1 m^2 quadrats located along the transect. At each survey location fish abundance and size frequency data were collected to the lowest possible taxonomic resolution. These data were complemented by data describing the abiotic composition (substrate type, habitat type) and biotic composition of the banks (corals, sponges, algae), and anthropogenic impacts observed (marine debris).

Analysis on the fish data revealed the stratification design incorporating bank, habitat and depth to be the optimal selection. The design optimized sample allocation by incorporating both strata area and variance components. While the resultant six strata: EB and WB high relief shallow, high relief deep, and low relief were the most efficient of those tested, a large sample size was still required to adequately meet the objectives. Potential management options include relaxing the precision requirements (a CV of 10% was used), using a different, less variable proxy (e.g., presence/absence versus density), or continuing to look for more cost effective sampling designs. An evaluation of the selected sampling design using 2007 data revealed a clear separation between habitat type; however, only a moderate division by depth was apparent. As new data are collected this may be reassessed and depth may no longer be a necessary component

BENTHIC COMPOSITION

Initial data analysis was aimed at addressing the second goal set forth by sanctuary management: to provide a spatial and quantitative characterization of the benthic composition. The benthic data were analyzed to provide coverage estimates for the entire SCC region and then differences between selected strata were evaluated. To explore potential relationships among various cover types, correlation analyses were also conducted. Finally the data were interpolated to form mapped surfaces that could be investigated for spatial trends in the different biota types. Additional analyses examined percentage and distribution of coral bleaching and provided a basic characterization of marine debris. The relationship between the SCC benthic community and others with similar biota was explored through comparisons with three locations in the U.S. Caribbean where identical data collection methods had been employed.

Overall the SCC region was comprised primarily of hardbottom (89%) with limited amounts of rubble (9%) and sand (3%). Rubble was more dominant in the low relief strata (46%) where dead *Madracis mirabilis* had broken apart, compared with the high relief strata (2%). Estimates of coral cover were high for the coral caps as a whole (48%) when compared with algae (13%) or sponges (1%). This value is comparable to historical values reported for live coral cover at the banks of nearly 50% and is between 6 and 11 times higher than values estimated for the U.S. Caribbean locations.

High relief habitats were generally coincident with the upper coral caps and were dominated by colonies of *Montastraea* and *Diploria* while low relief habitats were found typically in deeper waters and were dominated by live *Ma. mirabilis* and rubble. Coral cover tended to be higher on the high relief habitats and lower on the deeper low relief areas, while algae showed the opposite trend. Of the dominant taxa, *Montastraea franksi* and *Mo. faveolata* were more prevalent in the high relief habitats, *Diploria strigosa*, *Montastraea cavernosa*, *Porites astreoides* and *Colpophyllia natans* were distributed throughout the banks; *Ma. mirabilis* dominated the low relief habitat. While coral coverage was estimated to be high, 18% of it was estimated to be affected by coral bleaching. Highest incidences of bleaching were reported in *Millepora alcicornis, Siderastrea siderea* and *Mo. cavernosa.* The high values reported for coral bleaching suggest that the sanctuary may be more susceptible to environmental impacts than previously known.

Algae were more prevalent in the deeper low relief habitat, however it was found throughout the banks. A positive relationship was observed between macroalgal cover and sponge cover as well as depth, while macroalgal cover was negatively related to coral cover. No significant differences were found in sponge cover between strata. However, similarly to algae, sponges were found to be negatively correlated with coral cover.

Marine debris has been demonstrated to negatively impact coral reef environments through entanglement or habitat degradation. Few instances of marine debris were reported during the course of this baseline assessment. Debris observed included anchor, fishing line and rope. The anchors and associated anchor line observed were colonized by sizeable coral heads suggesting a lengthy period of time since their appearance on the reef. More research is required to determine the ecological impact of the other debris items encountered.

FISH CHARACTERIZATION

Analysis of the fish data focused on providing a spatially-explicit characterization and baseline assessment of fish community structure at depths shallower than 33.5 m. This work is a complement to earlier studies which have provided both a more general overall characterization and quantitative information for a comparatively spatially constrained portion of the SCC.

Similar to the benthic data, fish data were analyzed to provide population estimates for the SCC and then differences between strata (those selected for benthic analyses) were evaluated. Correlation analyses were conducted to explore potential relationships between the various fish assemblage metrics and benthic habitat measures such as coverage of coral, algae or depth. Finally, the data were interpolated to form mapped surfaces that could be investigated for spatial trends in the different metrics. Analyses were performed at the community level, family level, species level and by trophic groupings.

A total of 117 species from 37 families were observed during the course of the surveys. With the exception of species richness, which was significantly lower in the low relief habitat than the high relief at either bank, the other community level metrics, biomass, density and diversity were not significantly different among strata. The lower number of species in the low relief habitat is likely a function of habitat complexity.

Two species known only from relatively recent surveys of the coral caps were also observed during the course of this study, *Abudefduf saxatilis* and *Halichoeres burekae*. *H. burekae* is cryptogenic in origin, while *A. saxatilis* is believed to have arrived at the banks from neighboring oil platforms. Also of note are the first sighting of the Nassau grouper (*Epinephelus* striatus) and the second of the goliath grouper (*Epinephelus itajara*).

The three most abundant families observed at the banks were Labridae (35%) dominated by *Thalassoma bifasciatum* and *Clepticus parrae*; Pomacentridae (30%) dominated by species from the genera *Chromis* and *Stegastes*; and Serranidae (14%) primarily composed of *Paranthias furcifer*. Biomass was dominated by species in the family Serranidae (42%) followed by Kyphosidae (15%), Lutjanidae (7%), Carangidae (6%) and Scaridae (6%). The invertivore and zooplanktivore trophic groupings dominate numerically while the piscivores dominate by biomass.

Within the Serranidae, *P. furcifer* was the most abundant species while *Mycteroperca bonaci* and *M. tigris* dominated by biomass. The larger individuals were typically observed near the intersection of the high and low relief habitats on the edges of the coral caps. Of the large bodied groupers *Mycteroperca interstitialis* was the most abundant followed by *Mycteroperca tigris, M. bonaci*, *Dermatolepis inermis*, *Epinephelus adscensionis*, *E. guttatus*, *M. venenosa* and *M. phenax*.

Lutjanidae composed less than one percent of the total abundance of fish observed during the surveys while they composed 7% of the biomass. In order of abundance, *Lutjanus jocu* was the most frequently observed species in the family followed by *L. griseus, L. analis* and *L. cyanopterus*. No discernible spatial patterns were observed at the family level.

Six species of the family Scaridae were observed, all with relatively high sighting frequencies: *Sparisoma aurofrenatum, Sp. viride, Scarus vetula, Sc. taeniopterus, Sp. atomarium* and *Sc. iseri*. The greatest density was observed on low relief habitat which was influenced by the high abundance of *Sp. atomarium* observed there. *Sc. iseri* and *Sc. taeniopterus* densities were significantly greater on both EB habitats than WB. Both *Sp. viride* and *Sc. vetula* were significantly more abundant on high relief habitat.

Three species of Carangidae were observed during the study and are listed in order of sighting frequency: *Carangoides ruber*, *Caranx lugubris* and *Cx. latus*. Collectively, they composed approximately 2% of the total abundance and 6% of the biomass. Spatial patterns were difficult to discern in large part due to the aggregating nature of these species.

Within the family Pomacentridae, the territorial damselfish *Stegastes planifrons* was one of the most abundant species found on the banks. It is typically associated with healthy ecosystems characterized by high live coral estimates. On the banks high numbers of juveniles were found associated with *Madracis*; however, in general, the highest concentrations of the species were found associated with the high relief habitats dominated by *Montastraea*.

A cluster analysis of the density data revealed three distinct fish assemblages on the banks. The first was a deep water (32 m) assemblage typically associated with the low relief habitat. This assemblage was dominated by *Sp. atomarium*, *Stegastes variabilis*, *S. planifrons*, *Gnatholepis thompsoni* and *Opistognathus aurifrons*. A shallow water (24 m) assemblage associated primarily with high relief habitat included most notably *Sp. viride, Sc. iseri* and *Sc. vetula*. The third assemblage contained nearly all the piscivores as well as *P. furcifer, Bodianus rufus* and *Acanthurus* spp. This assemblage was observed spatially where the high and low relief habitats came together (29 m).

Comparisons made with data collected using identical sampling methods at three locations in the U.S. Caribbean revealed significantly higher density and biomass on the FGBNMS coral caps. Biomass on the corals caps was dominated by apex predators, which comprised 36% of the total observed biomass. Apex predators such as *Mycteroperca* are virtually absent from surveys in the U.S. Caribbean as are large sized snappers and jacks. Zooplanktivores are also significantly more abundant and have higher biomass on the coral caps.

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LIST OF ACRONYMS

BB - Biogeography Branch
CCA - crustose coralline algae
CCMA - Center for Coastal Monitoring and Assessment
EB - East Bank
EBH - East Bank high-relief
EBHD - East Bank high-relief, deep
EBHS - East Bank high-relief, shallow
EBL - East Bank low-relief
DCC - deep coral cap
DCS - decompression sickness
DNCC - deep non-coral cap
FGBNMS - Flower Garden Banks National Marine Sanctuary
GOMFMC - Gulf of Mexico Fishery Management Council
LTM - Long-term monitoring
MMS - Minerals Management Service
MMU - minimum mapping unit
MPA - marine protected area
NOAA - National Oceanic and Atmospheric Administration
ONMS - Office of National Marine Sanctuaries
REEF - Reef Environmental Education Foundation
SAFMC - South Atlantic Fishery Management Council
SCC - shallow coral cap
scuba - self contained underwater breathing apparatus
SRS - simple random sampling
StRS - stratified random sampling
USVI - U.S. Virgin Islands
WB - West Bank
WBH - West Bank high-relief
WBHD - West Bank high-relief, deep
WBHS - West Bank high-relief, shallow
WBL - West Bank low-relief

Chapter 1: Introduction and Background Information

1.1 GOALS

This project is a result of a collaboration between the National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment's Biogeography Branch (CCMA), and the Office of National Marine Sanctuaries' (ONMS) Flower Garden Banks National Marine Sanctuary (FGBNMS). The goal of this project was to provide FGBNMS staff with information on biogeographic patterns within the sanctuary critical to decision making. Specifically, this project focused on two explicit management goals of the sanctuary:

1) To develop a sampling design for monitoring benthic fish communities on the coral caps;
2) To obtain a spatial and quantitative characterization of those fish communities and their associated benthic habitats.

1.2 DESCRIPTION OF SANCTUARY

The FGBNMS is located in the northwestern Gulf of Mexico approximately 180 km south of Galveston, Texas (Figure 1.1). The distance of the banks from the coastline reduces direct coastal impacts on the resident coral population. In addition, their depth combined with local oceanographic processes may provide some shelter for resident species from thermal stresses and hurricanes. The healthy corals resident there support a high abundance of coral reef fishes and other associated marine organisms. Studies of the coral assemblages on the banks report live coral cover at nearly 50% (Gittings, 1998; Aronson et al., 2005), a value among the highest reported for coral reefs in the Caribbean (Pattengill-Semmens and Gittings, 2003). Despite the demise of other reefs in the region, this statistic has not changed significantly in the FGBNMS in over 30 years of monitoring (Bright and Pequegnat, 1974; Rezak, 1977; Gittings et al., 1992; Gittings, 1998; Precht et al., 2006).

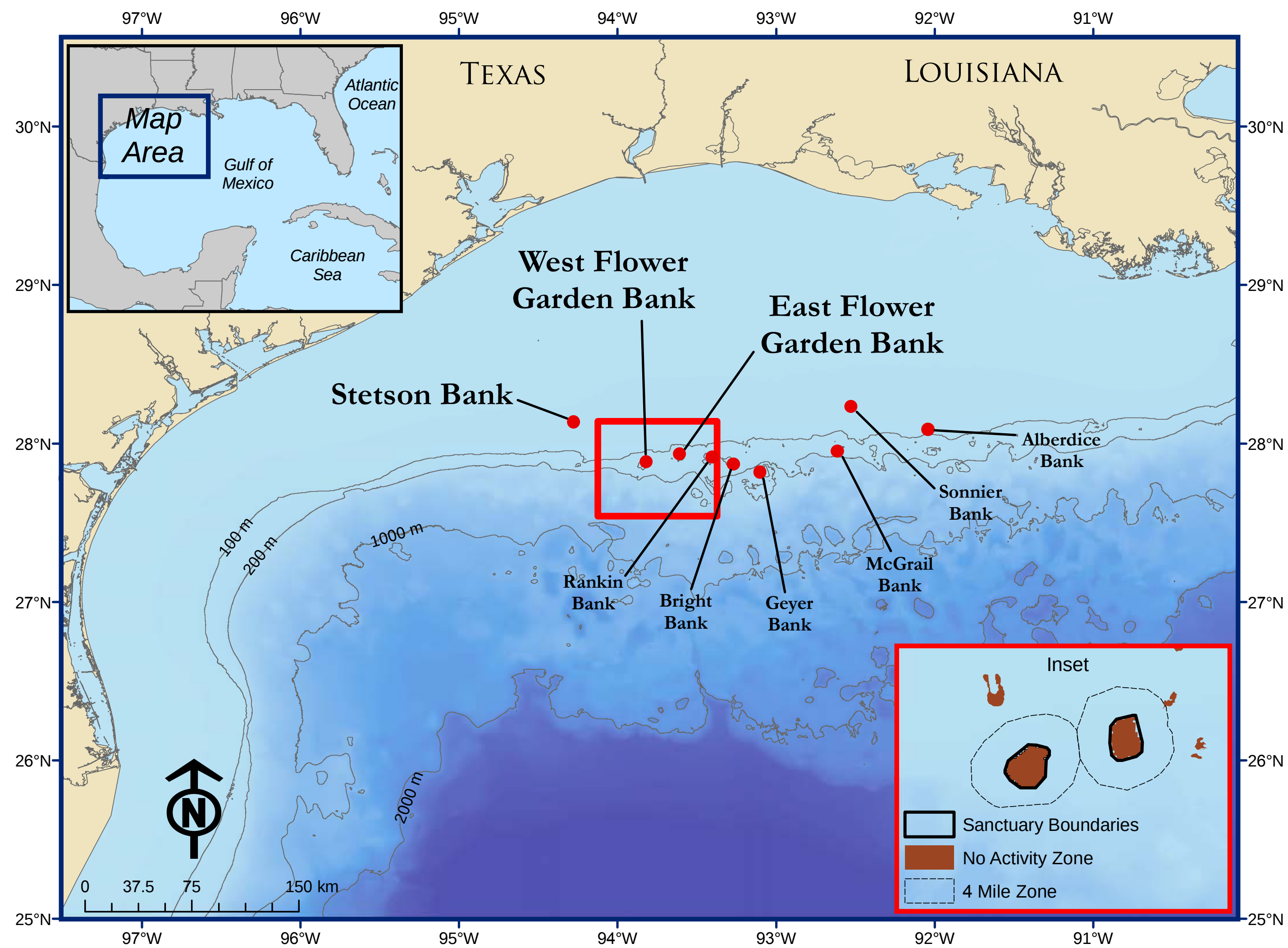


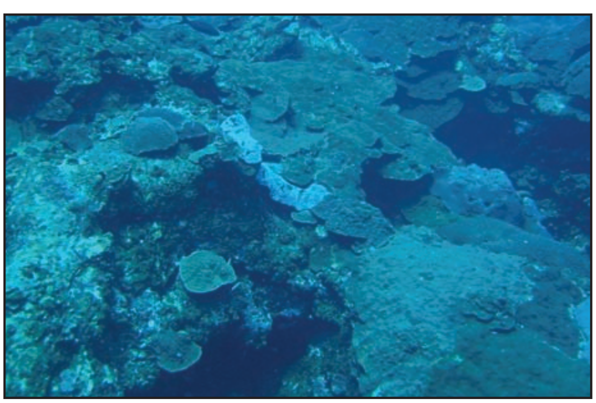
Figure 1.1 Map showing the location of the Flower Garden Banks National Marine Sanctuary in the northwestern Gulf of Mexico.

The FGBNMS is comprised of three underwater banks: East Flower Garden Bank (EB), West Flower Garden Bank (WB) and Stetson Bank that rise above a primarily soft bottom shelf habitat, (Parker and Curray, 1956). Located along the steep edge of the Sigsbee Escarpment, the banks originate from relic salt domes created by seafloor uplift and range in depth from 17-150 m (Shepard, 1937; Rezak et al., 1985). The EB is a younger, rejuvenated salt dome, while the WB has characteristics of a mature salt dome (Kennedy et al., 1983). Stetson Bank is a deeper bank that does not support the biological assemblages or diversity seen on either the EB or WB (Rezak et al., 1985).

Figure 1.2. Image of Madracis mirabilis *rubble covered with coralline algae (top) and polyps extended (bottom). (CCMA, E. Hickerson).*

McGrail (1982) classified the banks' benthic habitats into five major zones: 1) the upper coral caps ranging from 17-40 m depth (Rezak et al. [1985] define the upper limit as 15 m); 2) the lower algal-sponge dominated habitat between 40-90 m; 3) other shallow reef structures above 90 m; 4) the deep reef structures below 90 m; and 5) the soft bottom habitat below 90 m depth. The reef building corals that dominate the caps represent the northernmost coral reefs on the North American continental shelf. Coral assemblages on the caps can be divided into a shallower community (15-36 m) dominated by *Diploria, Montastraea* and *Porites* species; and a deeper community (down to 52 m) dominated by *Stephanocoenia* and *Millepora* species (Rezak et al., 1985). Along portions of the banks an area dominated by the ahermatypic yellow pencil coral (*Madracis mirabilis*) forms in between these two assemblages (Figure 1.2). Despite the similarities in species composition to Caribbean reefs, there is a noticeable lack of shallow water corals (e.g., *Acropora palmata, Porites astreoides*) and octocorals (Rezak et al., 1985; Aronson et al., 2005), although two small Acroporid colonies have been reported in recent years (Zimmer et al., 2006).

The region below the coral caps has been the subject of recent exploration efforts led by the FGBNMS staff, however, more comprehensive exploration and research is warranted. The deepwater benthic habitat is known to include: "algal sponge zones, honeycomb reefs, coralline algae reefs, highly eroded outcroppings, mud flats, mounds and mud volcanoes, and at least one brine seep" (FGBNMS, 2006). Researchers are currently investigating these deepwater environments using submersibles and remotely operated vehicles (ROVs) in an effort to understand connectivity between these habitats and the shallower water portions of the banks as well as to investigate linkages with the greater Gulf of Mexico region.



Coral assemblage (CCMA)

Dramatic oceanographic and climatological processes including storm fronts from the north, tropical cyclones as well as cyclonic and anticylconic gyres shape these habitats and their associated communities (Rezak et al., 1985; Lugo-Fernandez et al., 2001; Deslarzes and Lugo-Fernandez, 2007). On the surface, the major current affecting the banks is the Loop Current which originates in the Yucatan Channel and moves clockwise around the Gulf before exiting through the Florida Straits (McGrail et al., 1982). At deeper depths, McGrail (1982) discovered that bottom waters move around the individual banks rather than over them as originally hypothesized thus dispelling theories of regional upwelling. As a result, corals thrive in the warmer surface waters while cold bottom waters remain below 100 m (Deslarzes and Lugo-Fernandez, 2007). The dynamic oceanographic patterns connect the otherwise isolated banks to the Gulf of Mexico basin and further link them biologically to the nearest tropical reefs in the Bay of Campeche, Mexico (Villalobos, 1971).

While the health of most of the sanctuary resources has been rated as either "good" or "good/fair" in a recent condition report, the sanctuary is still subject to both natural and anthropogenic impacts (ONMS, 2008). Biological impacts within the FGBNMS include pathogens, which virtually eliminated the *Diadema antillarum* population from the Atlantic basin (including the FGBNMS) in the early 1980s (Rezak et al., 1985). Fortunately, reports of coral disease in the FGBNMS have been minimal, despite the increased occurrence throughout neighboring Caribbean waters (Borneman and Wellington, 2005). In the past, limited bleaching events were reported within the FGBNMS and the coral demonstrated excellent recovery capabilities (Hagman and Gittings, 1992; Dokken et al., 2003; Pattengill-Semmens and Gittings, 2003). With increased sea surface temperatures throughout tropical waters, there has been an increase in the severity of coral bleaching events reported at the FGBNMS. While historically the corals have demonstrated high resilience, additional monitoring is necessary to investigate whether they are able to maintain this resistance to temperature stress (ONMS, 2008). Hurricanes also are capable of impacting the banks both directly through physical damage from water motion and from substantial land-based runoff of contaminated water which has been shown to reach the banks (ONMS, 2008).

Known anthropogenic impacts in the FGBNMS include anchoring, which has resulted in significant damage to coral structure, fishing activity, shipping and recreational diving (Gittings and Bright, 1986; ONMS, 2008). In addition to direct impacts, these activities collectively have resulted in the deposition of remnant marine debris, such as anchors, abandoned fishing nets, longlines and diving gear. Until they degrade, these anthropogenic remnants can be deleterious to benthic habitats as well as their associated communities for many years (Chiappone et al., 2005). With an increase in the number of annual visitors to the FGBNMS, there is concern regarding the cumulative impacts of these activities on the marine habitat and organisms. Oil and gas related activities, which began near the FGBNMS in the late 1970s, have also been identified as potential stressors to the biological communities (Hickerson et al., 2008; Figure 1.3). Studies examining the effects of drilling practices on resident coral communities have found no negative impacts to-date (Bright and Rezak, 1976; Hudson and Robbin, 1980; Shinn et al., 1980); however, it has been suggested that oil and gas structures may act as vectors in the spread of invasive and exotic species (Pattengill, 1998). Currently, there are 14 production platforms and more than 180 km of pipeline within a four mile radius of the FGBNMS boundaries (Hickerson et al., 2008). Proposed aquaculture facilities on oil and gas platforms as well as existing artificial reefs may also impact the sanctuary.

Recognizing the intrinsic value of the banks as well as the need for protection from the various activities occurring there, the FGBNMS was created in 1992. It currently encompasses the EB (27°54.5'N, 93°36.0'W; 65.86 km^2); WB (27°52.5 N, 93°49.0' W; 77.54 km^2); and Stetson Bank (28'09.8'N, 94'17.9'W; 2.18 km^2). Stetson Bank was added to the sanctuary in 1996 and is located 48 km northwest of the WB.

Figure 1.3. Image of oil rig topside (top) and underwater (bottom) in East Bank. (CCMA)

A variety of regulations were put into place enabling the sanctuary to provide additional protections to the natural resources present (NOAA, 2001a). While fishing is permitted within the sanctuary it is restricted to conventional hook and line and take of other resources is generally prohibited. A "No Activity Zone" was established within the sanctuary by Minerals Management Service (MMS) where oil exploration and production are prohibited (MMS, 2004; Figure 1.1). In addition, within the sanctuary, the discharging of any pollutants or waste is prohibited and there is no permissible altering of the seabed, including from the placement of structure (e.g., oil transportation lines, fish traps, nets, and/or anchors; NOAA, 2001a). Within a four mile buffer zone surrounding sanctuary, regulations require that drilling waste be discharged no more than 10 m from the seabed, thereby reducing the impact of sediment plumes on the coral community (MMS, 2004; Figure 1.1). In 1990, mooring buoys were installed on EB and WB to prevent anchor damage, but are restricted to boats <30.5 m (100 ft) long because boats of any larger capacity are considered a threat to the existing coral (NOAA, 2001a). In 2001 the International Maritime Organization designated the FGBNMS the first international no-anchoring zone and requires it be depicted as such on international charts (NOAA, 2001b).

1.3 DOCUMENT STRUCTURE

This document is organized around the two management goals provided by the sanctuary (see Section 1.1). Chapter 2 discusses sampling designs for characterizing and monitoring the reef fish community. It begins with a statement of objectives and identification of the data to be collected and population to be sampled. Following this, survey methods are discussed along with the potential benefits and difficulties associated with implementing them. This section is followed by an evaluation of candidate sampling designs including an analysis of covariance between the fish data collected and spatially explicit physical variables such as: habitat type, depth, gradient and bank location. Finally, data storage and distribution are discussed. As part of this project, a GIS tool was developed to assist with the implementation of the sampling design, including site selection. This tool and its user manual are both available from http://ccma.nos.noaa.gov/ecosystems/sanctuaries/fgb_nms.html.

Chapter 3 includes a comprehensive characterization of the biotic and abiotic benthic data collected at each survey station. Habitat structure including measures of percent cover of corals, sponges and algae are evaluated for the entire survey area, as well as compared among strata. Additionally, the level of coral bleaching is noted and spatial patterns are analyzed. A description of marine debris associated with the banks is also provided. Lastly, comparisons are made between the habitat data collected on the FGBNMS coral caps and study sites in the U.S. Caribbean.

Red hind (*Epinephelus guttatus*) (CCMA)

Chapter 4 provides baseline data describing the fish communities along with a spatial and quantitative characterization of those communities. Analyses are conducted at the assemblage, trophic, family and individual species level. Estimates of abundance, biomass and size are provided sanctuary wide and comparisons are made between strata. As in Chapter 3, comparisons are made between FGBNMS study sites and those in the U.S. Caribbean.

Field methods detailing the techniques utilized to collect both fish and associated habitat information are contained in Appendix A and B, respectively (Figure 1.4). Appendix C discusses the methods used to construct the benthic habitat map necessary for the creation of sampling strata. Finally, Appendix D presents a table with an example sample allocation and site selection.

Figure 1.4. Diver collecting benthic habitat data (left) and a diver collecting fish data (right). (CCMA)

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Chapter 2: Sampling Design for a Fish Community Survey

2.1 OBJECTIVES

Surveys of the fish community provide characterization and monitoring information relevant to sanctuary management, such as species diversity, relative abundance, size-distribution and spatial patterns. Surveys can take many forms ranging from complete censuses to samples of opportunity. A comprehensive census of the entire FGBNMS fish community cannot be accomplished because the fish community is too large. The next best option is to sample the fish community. If completed properly, samples provide necessary estimates of population parameters and uncertainty.

Sampling requires a significant investment of planning to ensure that limited fiscal and personnel resources are used effectively. Planning should resolve the location, timing, number of samples and measurements to be taken. Collectively, this information is known as the sampling design. The first and most important step of generating a sampling design is an explicit statement of objectives. Establishing sampling design objectives is an iterative process and should be revisited in concordance with updates to the sanctuary management plan where management needs are outlined. As this work was undertaken prior to the development of this plan, the following objectives were established in consultation with sanctuary management staff during a meeting in May 2006:

1) Determine long-term changes in fish community structure using metrics of diversity, density and trophic ratios;
2) Determine long-term changes in density and mean-size of selected economically important taxa;
3) Determine the relationship between physical measures such as habitat type, depth, slope and geographic location with the associated fish community using metrics of fish diversity, density and biomass; and
4) Find better ways to collect information such that the probability of detecting change over time or space is increased.

2.2 BASIC STRATEGY

There are many useful sampling designs, each with distinct advantages and disadvantages. Only probabilistic sampling designs are considered herein, because they are the only type to allow reliable inferences to be made from sample units to the sampled population and quantify uncertainty. Simple random sampling (SRS) is the simplest and most fundamental probability-based survey design. The SRS design considers all sample units equal (i.e., all sample units have the same probability of being selected) and thus is appropriate for situations where there is no spatial structure in the variance of investigated metrics or in situations where no prior knowledge exists regarding this structure.

The assumption of homogenous spatial structure in fish communities is rarely met. More often fish species show a strong association with benthic habitats, depths, salinity and other environmental covariates (Ault et al., 1999; Kendall et al., 2003) and thus are heterogeneous. Communities with a heterogeneous spatial distribution can be sampled more effectively if the population can be divided into internally homogeneous groups. This is the goal of stratified random sampling (StRS). A StRS design may divide the survey domain (study area to be sampled) into regions of relatively homogeneous variance called strata and by sampling more intensively in highly-variable strata, a StRS design can achieve more precise results than a SRS design using the same sample size. The ultimate effect is that the likelihood of detecting spatial and temporal changes in observed metrics is increased.

One method to divide the fish community into strata is to parse the population by environmental covariates of the fish community (see Section 2.6.2). This process requires maps of environmental variables at appropriate spatial scales. Benthic habitat maps are ideal because they can integrate multiple environmental variables and have a proven track record in fish sampling designs in both Florida (Ault et al., 2001) and the U.S. Virgin Islands (USVI; Menza et al., 2006).

Incorporating an understanding of costs associated with sampling, such as transportation, equipment rental and time can also improve the efficiency of sampling. For instance, sampling more intensively in areas with lower sampling costs can increase sample size while keeping costs the same. In the marine environment, sampling costs are strongly linked to depth, because sampling in deeper environments often requires special equipment and/or training, and more time (see Section 2.4). Consequently, one of the principal methods to decrease costs is to stratify by depth.

In this report, we recommend stratifying by both sampling cost and environmental covariates to produce high-precision estimates of population and community metrics at a minimum of cost. At the broadest spatial scale, dividing the sanctuary into areas representing the coral caps and areas representing the remaining deep habitats will drastically reduce costs. In addition, this division would allow the entire coral cap community to be sampled with scuba using the same underwater census methods (i.e., diver belt-transect), while a more expensive, alternative method can be used to survey the deeper habitats.

A further division of the coral caps into areas which can be sampled using conventional scuba and those that must use technical scuba is beneficial as well. This would ensure greater than 90% of the coral cap community can be sampled with relatively cost-effective conventional scuba equipment. In addition to reducing sampling cost, these divisions allow each area (strata) to be monitored independently, in case they cannot be sampled together. For instance, we recommend monitoring the shallow portions of the coral caps even if the deeper regions cannot be monitored due to monetary or time constraints. This would provide at least some data for long-term monitoring.

The division of the aforementioned three strata was completed using a fine-scale, half-meter resolution bathymetric model (source: sanctuary staff) for the East Flower Garden Bank (EB) and West Flower Garden Bank (WB). The EB and WB coral caps were divided using the 33.5 m (110 ft) isobath into the Shallow Coral Cap (SCC) and Deep Coral Cap (DCC) strata and the remaining area within sanctuary boundaries was designated as the Deep Non-Coral Cap (DNCC) stratum (Figure 2.1). These three mutually-exclusive areas exhaustively cover the entire fish community in the FGBNMS.

In this report we discuss the iterative process of developing a sampling design for the SCC (see Section 2.6). A separate report should discuss sampling the DCC and DNCC. If technical diving is an option, the process of selecting a sampling design recommended in this report may be useful for the DCC as well. The DNCC will require different technologies and methods and will likely be much more costly.

2.3 TARGET POPULATION

The target population is limited to the SCC, which includes the areas of East and West Banks readily surveyed using conventional scuba diving techniques (<33.5 m). It is also limited to those species of visible, diurnally-active fish typically associated with the reef and less adequately describes the more pelagic, small, cryptic and/or nocturnal species.

The SCC encompasses a total area of 1.09 km^2 and is approximately 0.5% of the entire FGBNMS area. Approximately two thirds of the SCC is part of the EB and the remaining third is part of the WB. The SCC component of the EB is further divided into two areas, with the majority of area in a contiguous southern section (Figure 2.1, insets 2 and 3). Although the SCC represents a small fraction of the FGBNMS, it is of great importance to sanctuary managers because of its distinct fish and coral communities and use by sanctuary visitors.

2.4 DATA TO BE COLLECTED (portions excerpted from Menza et al., 2006)

In order to meet the objectives stated in Section 2.1, collection of the following information relating to the fish community is essential: identification to the lowest possible taxonomic classification of each individual, abundance and size-frequency.

Concurrently collected physical data can be assimilated in a survey design to improve survey performance. It is important to collect information on parameters used in maps such as habitat type, depth, slope, and bank location in order that these factors can be analyzed to determine their respective roles in structuring the fish communities. As demonstrated in the Florida Keys and USVI, an accurate benthic habitat map can be effectively utilized to meet this need (Ault et al., 2001; Menza et al., 2006).

In addition, information describing the associated benthic habitat should be collected to further assist in the interpretation of the fish data. This complementary data should include: abiotic information (substrate type, habitat type), biotic information (corals, sponges, algae) and anthropogenic information (marine debris).

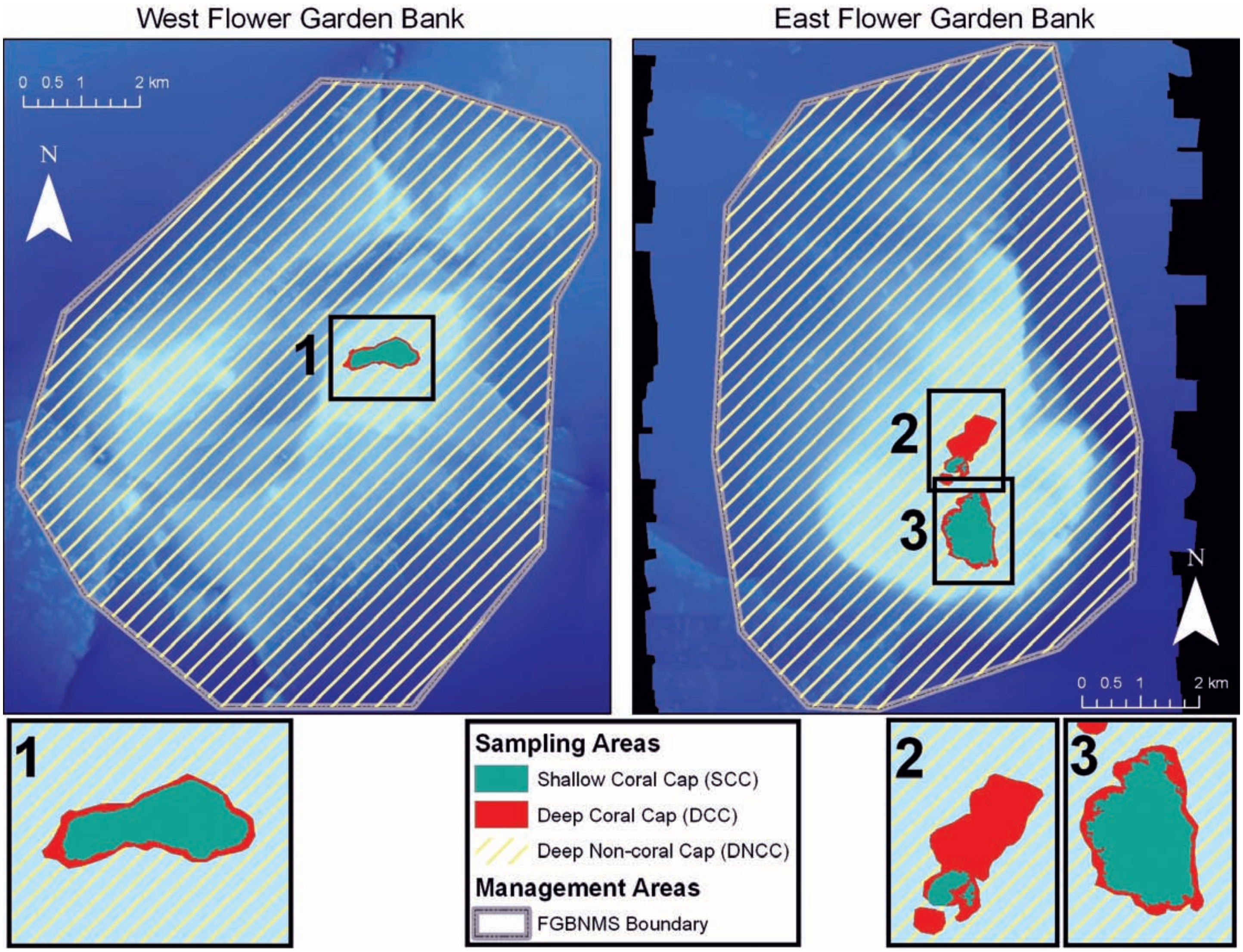

Figure 2.1. The East Flower Garden Bank (EB) and West Flower Garden Bank (WB) divided into three sampling areas. The insets, numbered 1 through 3, represent three geographic regions, not strata.

Detailed methods for collection of both fish and benthic composition data are available in Appendices A and B, respectively

2.5 SURVEY TECHNOLOGIES

As interest in ocean exploration, characterization, and monitoring advances, so must the methods and technologies employed for data collection. These technologies provide a range of various methods and approaches to data collection and monitoring. This section discusses applicable technologies used to sample and monitor coral reef communities along a transect using visual underwater census methods. Four factors should be considered when approaching each technology: 1) the cost of equipment and operation; 2) the quality and quantity of data collected; 3) the practicality of the method with regards to the overall goals of the mission or project; and 4) safety. The two technologies discussed below have been chosen because they satisfy the objectives listed in Section 2.1, are simple and cost effective, and have an extensive history of use. Additionally, they meet management concerns requiring the use of non-destructive assessment methods in the sanctuary.

2.5.1 Basic Scuba Diving

Scuba diving is one of the most common methods used for data collection, sampling, monitoring, and studying coral reef communities. Two common gas mixtures used for basic diving are a standard air mixture and an oxygen enriched air mixture. Standard air (normal atmospheric air) is comprised of approximately 21% oxygen and 78% nitrogen and 1% other; while Enriched Air (or “nitrox”) generally contains between 32-36% oxygen.

Advantages

Using scuba to conduct survey work has many operational advantages. *In situ* observations cannot be replaced by video or still photographic images, because a diver has the ability to visualize the survey area multi-dimensionally, identify distance and lengths, and investigate holes or crevices. Also, the preponderance of studies which have used basic scuba allows standardized comparisons of results. Another advantage is the relatively simple nature of basic scuba operations which requires minimal technical or operational support, easy gear assembly and breakdown, and widely available training.

The technical advantages of diving with nitrox over standard air are numerous, particularly when conducting multiple dives over more than one day (Mastro and Dinsmore, 1989). Compared to diving with air, using nitrox increases bottom-time and shortens surface intervals (PADI, 2003). Nitrox also has an advantage over air by reducing decompression requirements and the occurrence of decompression sickness (DCS, or the “bends”). By replacing nitrogen with oxygen in the gas mixture, nitrogen build-up in the body is reduced due to the increased oxygen in the breathing mixture (Wells, 1989).

Disadvantages

There are both technical and operational disadvantages to using scuba as well. Diving to depths of 35 m and greater or for extended times using air or Nitrox is achievable, but not practical or recommended. Most organizations and educational programs have a diving limit of 33-37 m without advanced or technical certifications. Many of these restrictions are based on health risks that can be potentially serious, such as DCS, arterial gas embolism and oxygen toxicity.

Operationally, the disadvantages of conducting visual surveys include inter-diver variability in data collection and the effect of human activity on the organisms surveyed. For fish estimates, data variability can include: under or over estimation of size, quantity and distance (Edgar et al., 2004). Much of this variability can be explained by diver experience (Harvey et al., 2004). Also, the presence of human activity can create audible and physical disturbances that can serve to attract or deter potential species of interest (Lobel, 2001; Harvey et al., 2004).

2.5.2 Technical Scuba Diving

Technical diving exceeds the scope and limits of recreational diving requiring additional training, equipment and extensive experience. Many organizations and programs provide training for technical diving. Technical diving can include the use of common gas mixtures such as “trimix” and “heliox” gases, as well as closed-circuit systems (“rebreathers”). The operational advantages and disadvantages of survey data collection using these technical diving methods are the same as those mentioned in Section 2.5.1; however, there are additional considerations which are outline below.

Advantages

The advantages of diving on mixed gases are extended bottom-times and greater maximum operating depths. This is made possible by the multiple gases utilized and the gas ratio combinations both of which can be adjusted based on depth and dive time to meet the requirements of the sampling design. The advantages of rebreathers include a longer bottom-time, shorter topside operations, cheaper costs and fewer imposed disturbances on habitats and organisms (the latter results in a more accurate representation of species composition and abundance). A comparison study of open versus closed-circuit diving conducted by Parrish and Pyle (2002) demonstrated that divers using open-circuit equipment required more gas, more preparation time and spent more on consumable gases (i.e., oxygen, air, heliox premix).

Disadvantages

One of the disadvantages for all three of these types of technical diving is the additional equipment and training required for the use of the equipment. Rebreathers and trimix/heliox require extra equipment maintenance and calibration, and can be costly in the short-term. The same medical risks involved with standard and enriched air diving pertain to these three technical diving operations too. However, working with technical equipment and diving deeper than recreational depths also increases decompression times and the potential for accidents to occur. Additionally, since divers must attain additional levels of certification and diving platforms must have the equipment onboard to support closed-circuit system diving, these requirements may impose further restrictions on the proposed sampling design.

2.6 SAMPLING DESIGN

A common first step for characterizing and monitoring any natural resource, including a fish community, is to initially select an uncomplicated sampling design (e.g., simple random) given the current understanding of the community. Then as data are gathered, more efficient sampling designs can be compared to the original sampling design, and adopted if proven effective. This iterative process ensures collected data are used to produce efficient sampling designs as data properties are better understood.

To ensure fish community data are collected efficiently, six distinct sampling designs were identified and then compared based on estimated precision of population and community metrics. Analyses were undertaken using fish data collected in 2006. These data were collected by CCMA and ONMS personnel, using a stratified random design (measurement methods are described in Appendices A and B). Four strata were used to parse sample sites (see Appendix C). Strata were composed of areas with differing bathymetric slope (i.e., steep versus flat) and geographic location (i.e., EB versus WB; Figure 2.2). A total of 73 samples were collected using randomly positioned belt-transects within each stratum. Each transect was allocated among strata proportional to area, with a minimum of five samples allocated to each to ensure an adequate sample size for computing precise strata metrics (i.e., reducing standard error).

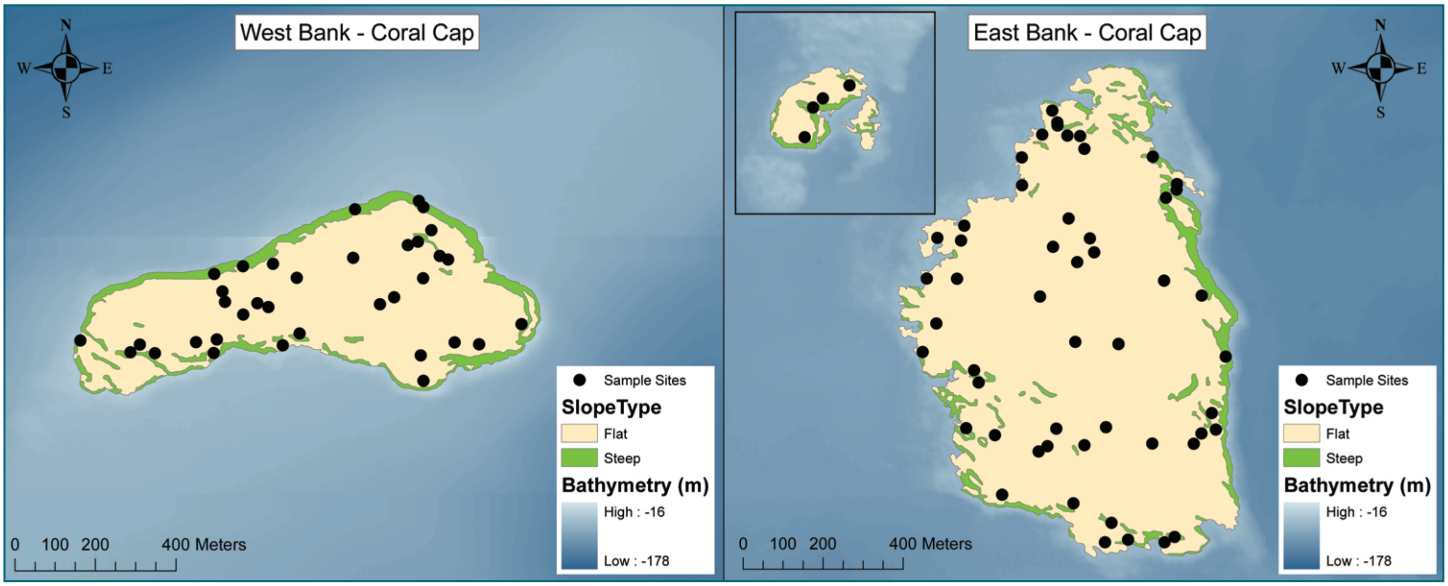

Figure 2.2. Location of samples and strata from the 2006 field mission. At each sample location fish and benthic habitat data were collected.

Results from candidate sampling design analysis (Section 2.6.2) were used to choose a sampling design for the 2007 field mission. It was anticipated that the data gathered in 2007 would be used in a second round of candidate design analysis, but field operations were cancelled prematurely and provided insufficient data to complete this task.

The following sections describe the process and outcome of design analysis using 2006 data.

2.6.1 Candidate Design Identification

Six candidate sampling designs were identified to compare design performance. Four distinct stratified sampling designs were produced using different combinations of geographic location, depth and benthic habitat to parse the survey domain. The fifth design was the sampling design used in 2006 (StRS-2006) and the sixth was a simple random sampling design (SRS) used to assess design efficiency without stratification.

To ensure the designs would satisfy all sanctuary objectives (Section 2.1), an assortment of population and community metrics (e.g., species density, species composition) and fish assemblages (e.g., all fish species, herbivores and specific species) were used to evaluate design performance (Table 2.1.). The combinations of metrics and fish assemblages used in the evaluation process are hereafter referred to as indices.

Number of species, herbivore:piscivore (H:P) ratios, density and biomass of all species combined are community-level metrics and were chosen to reflect community regime shifts associated with major environmental perturbations or fishing pressure. Number of species is simply the number of distinct species observed at a site. The H:P ratios are determined from the densities and biomass of each trophic group. Assignment to a particular trophic group was determined using FishBase (Froese and Pauley, 2007). Total density is simply the sum of all individuals of each species observed within a sample unit. Biomass is the sum of biomass from all fish species. Details of the computations are provided by Menza et al. (2006). Density, biomass and average size of groupers are assemblage-level metrics and were chosen to reflect changes in fishing pressure. The grouper assemblage was a subset of species in the Serranidae family, consisting of all species in the *Mycteroperca, Cephalopholis, Epinephelus* and *Dermatolepis* genera. Average size was computed from the midpoint of size-class data collected in the field. Density was computed as defined previously, but constrained to the subset of grouper species defined above. The remaining indices (density of grey snapper [*Lutjanus griseus*], yellowmouth grouper [*Mycteroperca interstitialis*], tiger grouper [*Mycteroperca tigris*] and marbled grouper [*Dermatolepis inermis*]) are species-level metrics and were chosen to provide data on key taxa of interest to sanctuary managers. Again, density was computed as defined previously, but only for individuals of each corresponding species.

Table 2.1. Indices used in the sampling design evaluation process.

Index
Number of species
Density (No. / 100m^2)
All species
Groupers
Herbivore:Piscivore Ratio
Grey snapper (*Lutjanus griseus*)
Yellowmouth grouper (*Mycteroperca interstitialis*)
Tiger grouper (*Mycteroperca tigris*)
Marbled grouper (*Dermatolepis inermis*)
Average Size (cm)
Groupers
Biomass (g / 100m^2)
All species
Groupers
Herbivore:Piscivore Ratio

Two of the potential environmental covariates investigated for stratified designs, slope and depth, were continuous variables. To include these variables in analyses they were first grouped into discrete categories. Since stratified sampling designs are most effective when the measurements of interest (i.e., indices) are divided into internally homogenous groups, an analysis of variance was undertaken to identify suitable breakpoints by which to categorize these two continuous variables. The analysis of variance was accomplished using recursive partitioning (function Partition: JMP© by SAS Institute Inc., 2000). This process recursively divided each index into groups such that the ratio of variance within groups to among groups was minimized. A non-parametric analysis of variance test (Kruskal-Wallis ANOVA) was used to indicate which groupings were significantly different (Kruskal-Wallis; $p<0.10$; Table 2.2). A Type I error probability of 0.10 was used instead of 0.05, because this was an exploratory analysis and Type I error was not a critical concern. The results suggest a division by slope at 37° and by depth at 30 or 32 m provide suitable breakpoints for indices. To ensure a parsimonious stratification scheme the depth breakpoints were averaged to 31 m, thus one stratum was composed of all areas deeper than 31 m and another for areas shallower than 31 m.

Table 2.2. Breakpoints in slope and depth determined from analysis of variance of nine fish indices. Breakpoints with an asterisk denote a significant difference using ANOVA.

Index	Slope (°)	Depth (m)
Number of species	42	32
Density (No. / 100m^2)		
All species	39	32
Groupers	17	32
Herbivore:Piscivore Ratio	18	32*
Grey snapper (*L. griseus*)	37*	32
Yellowmouth grouper (*M. interstitialis*)	42	27
Tiger grouper (*M. tigris*)	16	32
Marbled grouper (*D. inermis*)	4	32
Average Size (cm)		
Groupers	42	30*
Biomass (g / 100m^2)		
All species	14	22
Groupers	14	22
Herbivore:Piscivore Ratio	31	32*

Benthic habitat strata were taken from a benthic habitat map of the coral caps (Appendix C, Map 2). Two distinct benthic habitats were identified: low-relief coral (mixture of *Madracis* and rubble) and high-relief coral (mixture of boulder and plate corals). Geographic location was simply used to divide the shallow coral cap community into an EB and a WB category.

Kruskal-Wallis ANOVA was also used to indicate if covariate groupings were significantly different (Kruskal-Wallis; $p<0.10$; Table 2.3). The decision of which variables to use in generating strata for the candidate sampling designs balanced potential increases in efficiency (as shown by ANOVA results), logistical value in the field, likely consistency over time and parsimony. For these reasons, geographic location, depth and benthic habitat were selected for stratification schemes and slope was not.

Candidate sampling designs and variables used to delineate strata are listed in Table 2.4. All stratification schemes for candidate sampling designs, except for the SRS used geographic location to divide banks, because geographic location simplifies field logistics. The composite design (STRS-Composite) divided high-relief coral habitats into two depth categories (shallow and deep) and a single low-relief coral habitat category. Low-relief coral habitat was not divided by depth, because of the small sampling area involved in shallow areas. A simple t-test was used to show deep and shallow low-relief coral habitat strata were not different (t-test; $p>0.10$).

Table 2.3. Results from non-parametric analysis of variance of nine fish indices among groups (ANOVA levels) derived from geographic location, depth, slope and benthic habitat type. Values represent the probability that data from each group come from the same population (p-value). Asterisks denote probabilities lower than 0.10.

Index	Geographic Location	Benthic Habitat	Slope	Depth
Number of species	0.12	0.04*	0.60	0.25
Density (No. / 100m^2)				
All species	0.47	0.54	0.36	0.38
Groupers	0.13	0.07*	0.32	0.60
Herbivore:Piscivore Ratio	0.49	0.31	0.24	0.04*
Grey snapper	0.03*	0.24	0.07*	0.47
Yellowmouth grouper	0.48	0.10*	0.77	0.49
Tiger grouper	0.86	0.57	0.16	0.86
Marbled grouper	0.81	0.80	0.84	0.39
Average Size (cm)				
Groupers	0.25	0.03*	0.12	0.01*
Biomass (g / 100m^2)				
All species	0.51	0.66	0.11	0.81
Groupers	0.08*	0.06*	0.52	0.64
Herbivore:Piscivore Ratio	0.84	0.18	0.26	0.09*

Table 2.4. List and description of identified sampling designs.

Sampling Design	Variables used to delineate strata
SRS	None
StRS-Bank	Geographic location
StRS-2006	Geographic location, slope
StRs-Depth	Geographic location, depth
StRS-Benthic Habitat	Geographic location, benthic habitat
StRS-Composite (used in 2007)	Geographic location, benthic habitat, depth

2.6.2 Candidate Sampling Design Comparison

Design performance was evaluated using the sample size required to obtain a coefficient of variation (CV) of 10% for each index. For these comparisons the precise value of the CV was not important; rather it was important to keep the CV constant to apply a standardized statistic in comparisons. Cochran (1977) describes the process of post-stratification and corresponding computations for both SRS and StRS designs. Post-stratification analysis is required because the domains used to parse the sampling frame are not the same as the strata used for obtaining data (except for StRS-2006). A limitation of post-stratification analysis is that estimates of variance for any given fish metric are not technically valid for data collected under a different stratification scheme (except for SRS). Thus the results of this section are for comparative analysis only and are not a re-estimation of variance.

Sample size requirements were computed for a specified CV, because unlike other performance measures such as a confidence interval, the computation of a CV does not make any assumptions concerning data structure (e.g., Normal distribution). If needed, CVs of a given size can be explained in terms of confidence intervals of a given size because they are different by a common factor. Sample size requirements were determined using the methods described in Cochran (1977) for a StRS design. Under a presumed optimal allocation scheme sample size is given by

$$n^* = \frac{\left(\sum_{j=1}^{J} W_j s_j\right)^2}{\left(CV\left[\overline{X}_{st}\right]\overline{X}_{st}\right)^2 + \frac{1}{\sum_{j=1}^{J} N_j}\sum_{j=1}^{J} W_j s_j^2}$$

where s_j is the stratum standard deviation, $CV[\bar{X}_{st}]$ is the desired coefficient of variation (i.e., 0.10) and $\bar{X}_{st}$ is the survey-wide mean. A fundamental requirement for the computation is accurate stratum weighting factors. These are computed using

$$W_j = \frac{N_j M_j}{\sum_{j=1}^{J} N_j M_j} \approx \frac{A_j}{\sum_{j=1}^{J} A_j}$$

where N_j is the number of sample units in stratum j, M_j is the number of transects which can fit in each sample unit, A_j is the area of a transect in stratum j, and J is the number of strata. Estimates for the mean and variance within a particular stratum were computed from transects within a given stratum. Stratum designations were determined using the intersection of the sample design map and transect coordinates in a GIS (ESRI, 2006).

Sample size requirements of each tested design are provided in Table 2.5. Results suggest StRS-Composite is the optimal choice for sampling the SCC. Not only was StRS-Composite the best design for the most number of indices, but StRS-Composite possessed the lowest sum of sample size requirements from all indices and had the lowest maximum.

Table 2.5. The sample size requirements needed to obtain a CV of 10% for nine reef fish indices using six distinct sampling designs. Numbers in bold represent the minimum sample size requirement for a given index.

Community Index	SRS	StRS 2006	StRS Bank	StRS Depth	StRS Benthic Habitat	StRS Composite
Number of species	2	2	2	2	2	**2**
Density (No. / 100m^2)						
All species	37	32	**30**	37	36	34
Groupers	74	76	74	72	76	**70**
Herbivore:Piscivore Ratio	121	123	119	**90**	127	109
Grey snapper	730	710	764	727	825	**574**
Yellowmouth grouper	158	167	157	167	155	**143**
Tiger grouper	234	239	242	252	239	**216**
Marbled grouper	676	741	704	861	**703**	731
Average Size (cm)						
Groupers	36	36	**35**	54	48	45
Biomass (g / 100m^2)						
All species	186	456	146	130	446	**129**
Groupers	842	**145**	455	455	1465	425
Herbivore:Piscivore Ratio	817	550	588	**237**	913	530
Totals	3913	3277	3316	3084	5035	3008
Maximums	842	741	764	861	1465	731
Averages	356	298	301	280	458	273

StRS-Composite proved superior to alternative candidate designs, because it ultimately could satisfy the multiple and diverse objectives selected by sanctuary managers (Section 2.1) at a minimum of cost. Although StRS-Composite was optimal, the large sample sizes required for some metrics (e.g., marbled grouper, grey snapper) are greater than what typically can be afforded during most sampling missions. Consequently, managers and researchers have three options. They can relax precision requirements (i.e., CV>10%), or use a different, less variable proxy (e.g., presence-absence) or continue to look for more cost effective sampling designs. These options are discussed further in Section 2.6.4.

A principal reason why StRS-Composite was optimal is that most groupers and snappers were sighted along the margins of East and West Banks (see Section 3). These areas are dissimilar from the remaining SCC, because they are characterized by benthic habitat transition zones.

Data gathered in 2007 was to be used in a similar analysis of the spatial relationships among fish indices and environmental covariates, but due to low sample size and lack of data for most strata, only a cursory analysis was possible. A total of 70 surveys were scheduled, but the field mission was cut short by severe weather associated with Hurricane Humberto and only 32 surveys were taken (Figure 2.3). No samples were taken from the WB and samples collected on the EB were biased towards the southern areas of the coral cap.

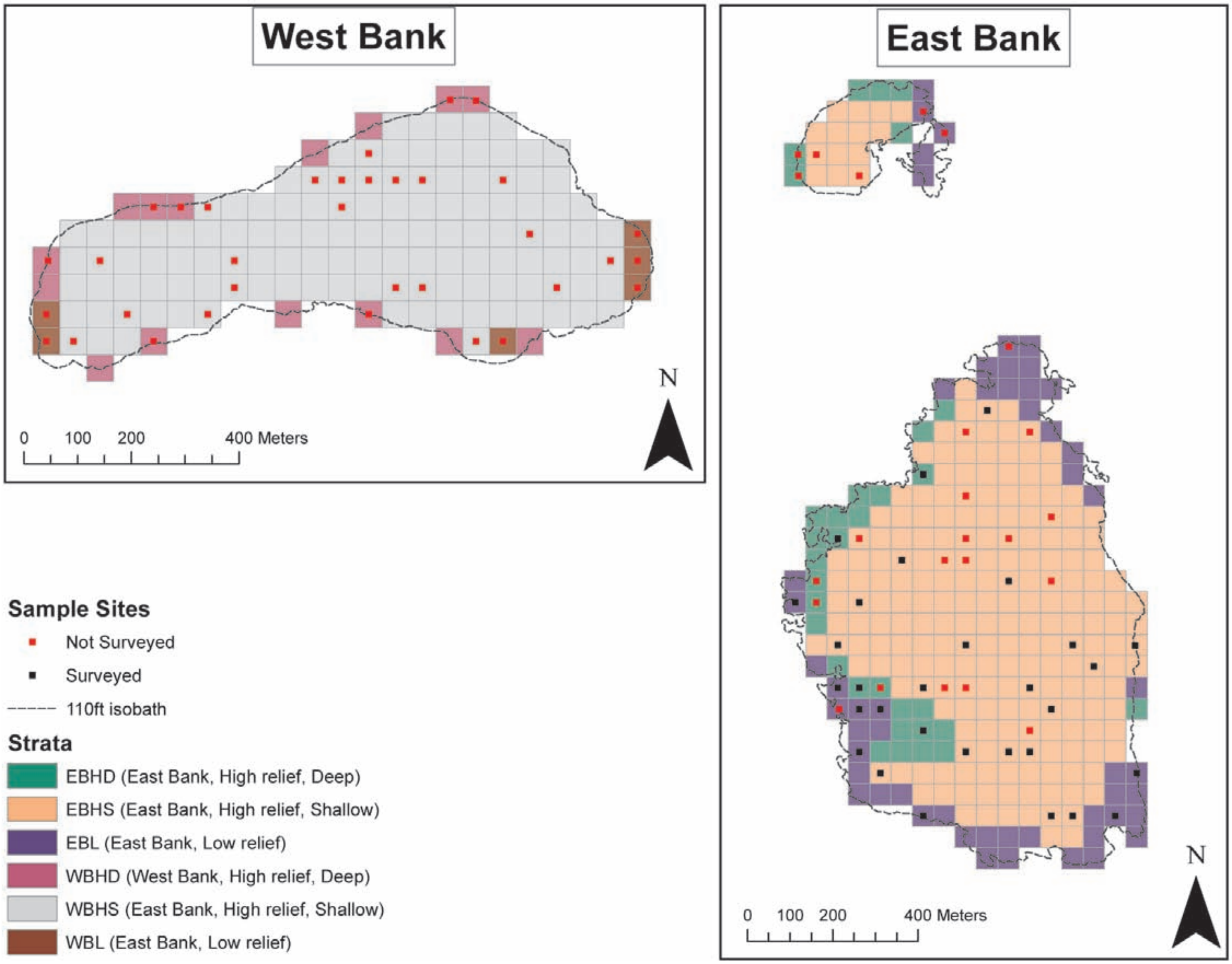

Figure 2.3. Location of samples and strata from the 2007 field mission.

The sampling design used in 2007 (see Appendix C) was different than the one used in 2006 in two respects. First, a different set of strata were used. The strata used in 2007 were based on the optimal candidate design from analysis of 2006 data. Second, the method used to position sample sites was different. In 2006, sites were selected by randomly placing geographic coordinates within a polygon representing a given strata. In 2007 a uniform distribution of points separated by 50 m was overlaid on the coral caps and a random selection of these points was taken. A separation of 50 m was used to ensure 25 m transects in neighboring sample units would not overlap. A detailed comparison of these methods and the rationale behind using the latter are described in Menza et al. (2007). The approach used in 2007 has the advantage of incorporating a habitat's sphere of influence, ensuring sample units are exhaustive and mutually-exclusive, and providing a means to update the design more easily. This sample frame is easily adapted to new stratification schemes. For example, should a new marine protected area (MPA) be established on one of the banks, a new objective can be devised to assess its effectiveness. All sample units (i.e., points) within the spatial limits of the MPA could be incorporated into new strata (or a new stratum), while all the remaining sampling units retain their original stratum designations. Then community and population metrics computed from strata with and without the MPA can be compared in order to assess the new management regime's effectiveness.

Multiple discriminant analysis (MDA) was used to assess index differences among strata using data collected in 2007. Only strata on the east bank were investigated due to the absence of data from the west bank. The results show a moderate separation between low-relief habitat and high-relief habitats, but only a marginal division between shallow and deep high-relief habitats (Figure 2.4). Based on these results the current strata should be maintained, but as new data are gathered the strata should be reassessed, especially with respect to differences among shallow and deep high-relief habitats. If future data do not show clear differences among shallow and deep habitats, aggregation is warranted.

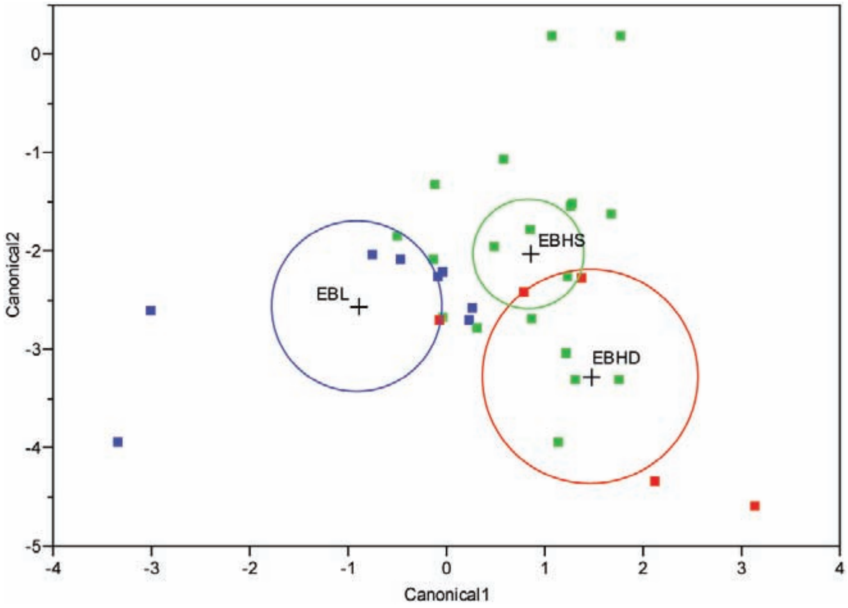

Figure 2.4. A canonical plot of multivariate discriminant analysis. Strata designations are EBL (Blue)– East Bank low-relief, EBHS (Green)– East Bank high-relief shallow, and EBHD– East Bank high-relief deep. Circles correspond to 95% confidence limits of multivariate means.

2.6.3 Example of Implementation

In this section we provide an example of implementing StRS-Composite to obtain a sample of survey sites. The generated sample coordinates can be used for a future mission. It is important to rerun the randomization process once the sample coordinates presented here are used to eliminate sampling bias.

In this example the sample frame constructed for the 2007 field mission was used for sample site selection. Ideally desired levels of sampling precision and statistical power are used to set the most efficient sample size for a sampling design. Unfortunately, this process is not feasible in many circumstances, because many important metrics require samples size which cannot be afforded. Consequently, sample size is set by fiscal and logistical constraints. Total sample size is generally between 50 and 100 and is strongly related to the number of divers and time available for diving. In 2006 CCMA and sanctuary staff were able to obtain 73 samples in six days using five buddy pairs. On average each diver conducted three dives per day. These numbers can be used to infer an estimate of sample size given personnel, equipment and logistical constraints for future missions. In this example, we used a sample size of 70.

Once a total sample size is determined it must be allocated among strata. The simplest allocation scheme and the one recommended here is sample size proportional to area. In this method, samples are allocated among strata proportional to their size (Table 2.6). Thus, large strata (e.g., East Bank high relief-Shallow [EBHS]) receive more samples than small strata (e.g., East Bank high relief- Deep [EBHD]). The Neyman allocation scheme is an alternative which can increase the precision of population and community estimates if accurate strata variance estimates are known. The latter is not recommended here, because the lack of data increases uncertainty when identifying highly variable strata for multiple indices.

Table 2.6. Sample allocation of 70 sample sites.

Strata	Area (m^2)	Weights	Proportional Allocation	Adjusted Allocation
EBHD	15,303	0.01	1	6
EBHS	579,115	0.12	37	26
EBL	128,826	0.53	8	10
WBHD	38,155	0.03	2	6
WBHS	322,074	0.30	21	17
WBL	7,503	0.01	0	5
Totals	1,090,976	1.00	70	70

One problem with the allocation scheme is that not all strata garner adequate samples to obtain an estimate for variance (e.g., WBL in Table 2.6). An adjusted form of the allocation scheme, which reduces the total n by $H*X$, where H is the number of strata, and then adds X samples to each strata after allocation, is used to ensure an adequate sample size within each stratum (Table 2.6). At a minimum X must be 2 to obtain an estimate of variance, but an X of 5 will reduce the standard error in small strata. The adjusted stratum sample size in Table 2.6 uses an X of 5.

The final step before field operations can take place is the random selection of samples among strata based on the allocation identified in the previous step. A tool was developed by CCMA to help in this capacity (see http://ccma.nos.noaa.gov/ecosystems/sanctuaries/fgb_nms.html). An alternative method is to use a program with a random selection procedure such as SAS or an equivalent statistical package. It is important to include all sample units within a given stratum when a random selection is being made. This ensures a selection bias is not incorporated into the sampling design. Figure 2.5 shows the location of survey sites of a stratified random sample which can be used in a future survey (see Appendix D for sample coordinates).

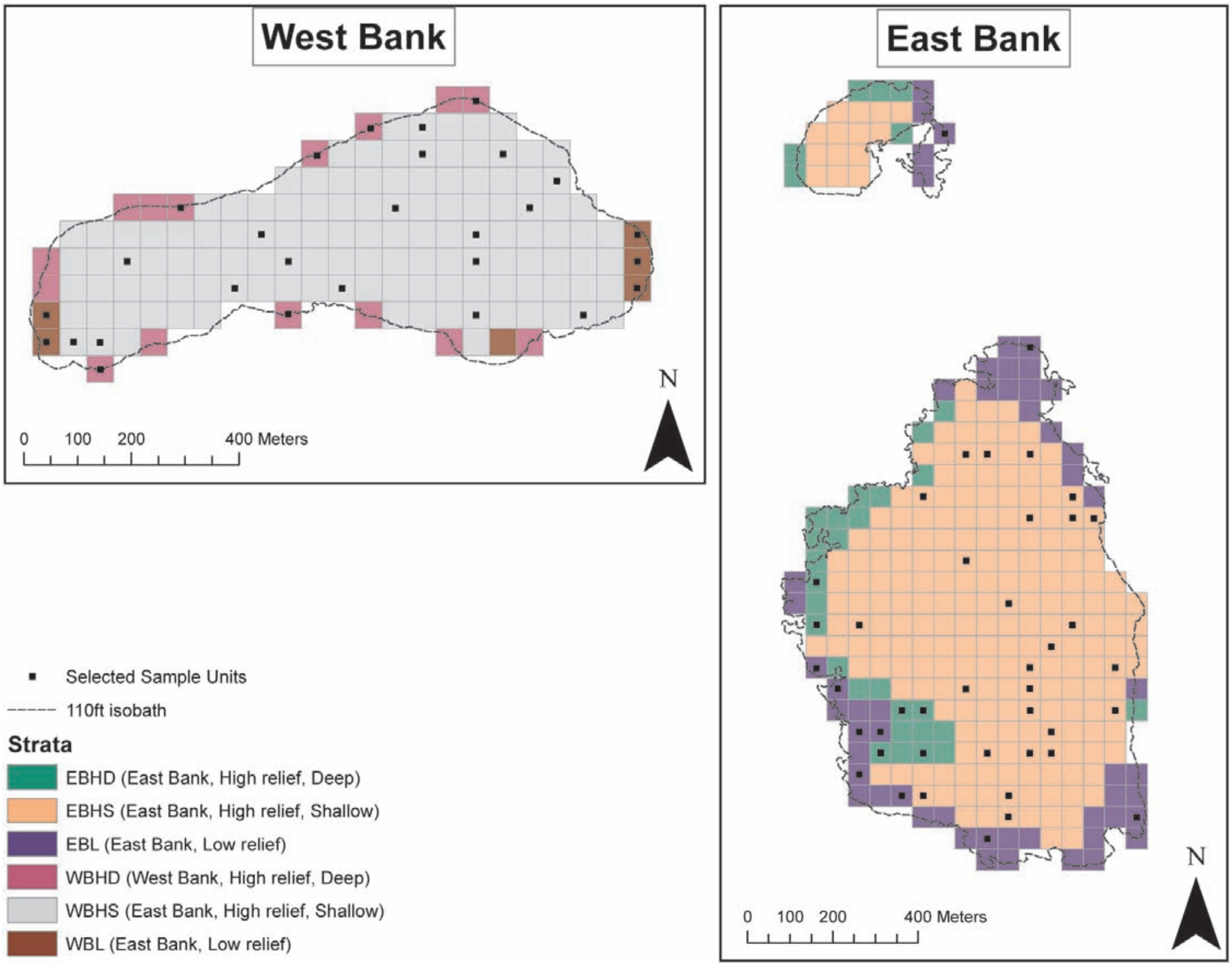

Figure 2.5. Sample example selected from strata of the shallow coral cap.

Once survey data is collected it must be stored and analyzed. Information on data storage and distribution is detailed in Menza et al (2006). Data analysis typically focuses on a set of simple descriptive statistics computed to assess populations and communities and identify temporal changes. It is important to keep in mind that when statistics are computed using multiple strata they must be weighted appropriately. Several basic sampling references (e.g., Cochran, 1977; Lohr, 1999) provide information on how to calculate sampling weights and use them to define descriptive statistics. Menza et al. (2006) define a set of descriptive statistics commonly used to assess reef fish communities and populations, and identify appropriate computations.

2.6.4 Future Direction

Although StRS-Composite was chosen as the optimal design, sample size requirements varied over two orders of magnitude depending on the reef fish index in question. Design selection must be tempered with realistic projections of maximum sample size. Given likely logistical constraints a maximum sample size will lie between 50 and 100. All community-level and assemblage-level indices will likely be sampled with a CV<10% when n≤100, but all species-level indices require n>150 and some require n>600 to ensure CV<10%. These large sample sizes are effectively impossible given likely logistical and monetary constraints of sampling on the SCC. To provide suitable data, managers may relax precision requirements (i.e., CV>10%), use a different less variable metric (e.g., presence-absence), or continue to look for more cost effective sampling designs.

The reason for high CVs and large sample size requirements is the heavily skewed distribution of data with many zeros. Zeros occur where a species was not observed within a sample and are common for rare species. More abundant species typically have fewer zeros, lower CVs and consequently lower sample size requirements (e.g., Table 2.7). The probability of detecting an item can be increased by using either a different sampling design or survey method.

Table 2.7. Coefficient of variation (CV) and sample size requirements (n) to obtain a CV=10% for density of six species of reef fish. Data from 2006 sampling mission.

Fish Index	% Occurrence	CV	n
Density – Bluehead wrasse (*Thalassoma bifasciatum*)	100	0.08	44
Density – Spanish hogfish (*Bodianus rufus*)	97	0.08	47
Density – Redband parrotfish (*Sparisoma aurofrenatum*)	89	0.09	64
Density – Yellowhead wrasse (*Halichoeres garnoti*)	85	0.12	94
Density – Queen parrotfish (*Scarus vetula*)	64	0.16	169
Density – Marbled grouper (*Dermatolepis inermis*)	14	0.31	607

Most researchers agree that a stratified design is necessary for sampling rare species (Thompson, 2002), but an alternative to the candidate designs investigated in this first stage of analysis may help ensure rare species are sampled better. One alternative is to use adaptive sampling, but the approach has acknowledged difficulties in large areas like the SCC and modifying field logistics on the fly can be difficult (Thompson and Seber, 1996). Alternatively, as more data becomes available better relationships among fish indices and environmental variables may emerge. Incorporating these relationships into a stratified design will increase design performance.

Alternate survey methods may also be employed to generate more precise survey estimates with less zeros. The most practical modification is a survey targeting rare species. This may increase sample size, allow larger areas to be surveyed, and increase the probability of detection of groupers and snappers. A two-part survey, one targeting the entire community (such as described in this study) and another survey targeting large species, is an option. These two phases could be completed simultaneously or one immediately after the other during a single dive. In addition, a larger sample plot may increase the probability of detection of rare species and decrease CVs; however an increase in plot size must be balanced with the time needed to survey the plot. If the increase in bottom time and required surface intervals decrease total sample size, the positive effect may be negated. Finally, a survey method which does not use divers (e.g., ROV, submersible, bait camera) could be used, but the sampling costs associated with these methods will likely be higher.

2.7 SUMMARY AND RECOMMENDATIONS

- Use a stratified-random sampling design. Results indicate depth and benthic habitat are covariates of several fish indices and can be used to effectively sample the coral caps.
- The development of a sampling design is an iterative process. As new data are gathered the strata should be reassessed, especially with respect to differences among shallow and deep high-relief habitats. If future data do not show clear differences among shallow and deep habitats, aggregation is warranted.
- A sample size between 60 sites and 100 sites is adequate to survey community-level and assemblage-level indices, but not species-level indices (e.g., marbled grouper [*Dermatolepis inermis*] density).
- To adequately survey species level metrics consider a different approach: both sampling design and survey method. Potential changes are suggested in Section 2.6.4
- Continue to acquire samples throughout the SCC and among all habitats. These data will allow a more refined analysis of fish-habitat relationships; improve the stratification scheme; and quantify changes associated with natural population variability, changes in management strategy, or environmental and anthropogenic impacts.
- Survey the remainder of the sanctuary not included in the SCC to provide comprehensive population estimates for the sanctuary and to identify linkages.

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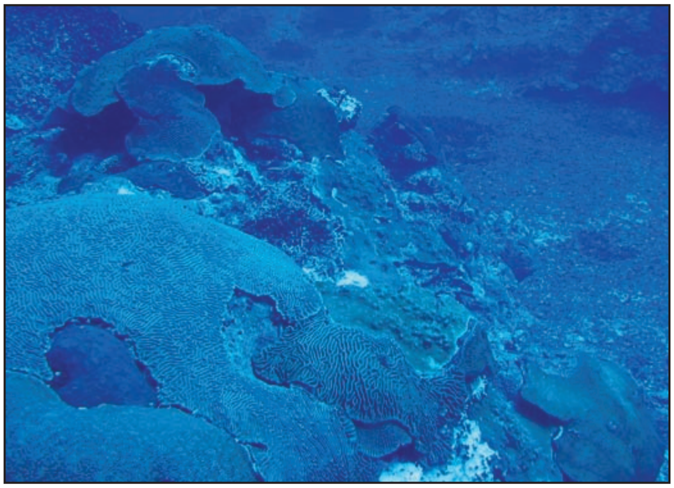
Chapter 3: Benthic Composition

3.1 INTRODUCTION

The coral caps are dominated by reef building corals, which have been estimated to cover nearly 50% of the shallower portions of the reef (Gittings, 1998; Aronson et al., 2005) and up to 82% in deeper waters (32-40 m; Precht et al., 2008a). Coral colonies over 2 m in height are commonly encountered and provide shelter for a thriving reef fish community (see Chapter 4: Fish Communities).



Montastraea, Diploria and *Madracis* corals (CCMA)

Scientific research has been conducted on the banks for more than 30 years (Gittings and Hickerson, 1998). Past studies evaluating potential impacts from the oil and gas industry have provided valuable information on the benthic communities (Bright and Pequegnat, 1974; Viada, 1980; Gittings, 1998). In the late 1980s permanent stations for long-term monitoring were established on the banks and continue to be surveyed annually (Precht et al., 2008a). This monitoring effort has focused on relatively small portions of the coral cap environments and thus provides a spatially limited scope of inference.

The research presented here complements these prior studies with the development of a spatial framework and sampling design that includes 90% of the coral cap community. The data, analysis and results presented here provide a spatially-explicit characterization of the benthic community at depths shallower than 33.5 m (100 ft). For comparative purposes, analyses are presented contrasting the benthic communities of the Shallow Coral Cap (SCC) to coral reef systems in St. Croix and St. John in the U.S. Virgin Islands (USVI) as well as southwestern Puerto Rico.

3.2 METHODS

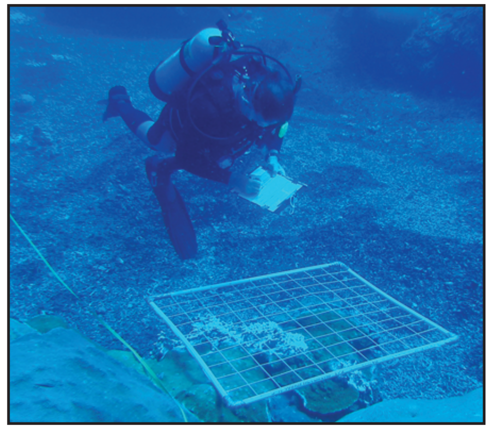
3.2.1 Survey Data

Benthic data were collected using underwater visual surveys of the East Bank (EB) and West Bank (WB) coral caps in 2006 (73 sites) and 2007 (32 sites). As detailed in Chapter 2, the 2007 sampling mission was concluded early due to Hurricane Humberto. Each sampling mission consisted of a randomly selected set of sample sites from which benthic and fish community data were gathered simultaneously. During each survey, data on benthic composition were collected from four 1 m^2 quadrats which were randomly placed along a 25 m belt transect used to census fish. In each quadrat abiotic and biotic components of the benthic habitat were measured (Table 3.1) and the mean for that survey location calculated. Abiotic data included: percent cover of hardbottom, sand and rubble components, and marine debris. Cover estimates included the bottom directly beneath living organisms such that an area recorded as 100% live coral would also receive a value of 100% hardbottom. Biotic data included: percent cover of coral species, algae classes, sponge morphotypes and coral bleaching. Data on each abiotic and biotic variable were averaged from the four quadrats to obtain synoptic representative

Table 3.1. Benthic variables measured to characterize the benthic community of the Flower Garden Banks National Marine Sanctuary (FGBNMS).

Benthic Biota	Measurements		
	Percent cover	**Height (cm)**	**Abundance (#)**
Abiotic			
Hardbottom	X	X	
Sand	X		
Rubble	X		
Fine Sediment	X		
Biotic			
Corals (by species)	X		
Macroalgae	X	X	
Seagrasses (by species)			
Sponges			
Barrel, tube, vase morphology	X	X	X
Encrusting morphology	X		
Other benthic macrofauna			
Anemones and hydroids	X		X
Tunicates and zoanthids	X		
Macroinvertebrates			
Queen conch			X
Spiny lobster			X
Long-spined urchin			X
Marine debris (type, area of debris, area affected, colonized by)			

estimates of the variables at each site. Coral colonies were reported as entirely bleached if they contained any portion of white, blotchy, mottled or pale tissue. This protocol assumes stress throughout the colony and estimates maximum bleaching impact. To determine susceptibility of the banks as a whole or individual species to coral bleaching, percent bleaching was standardized by the total coral cover or total percent cover of the particular species respectively. The average from the four quadrats represents a spatial average and was used as an independent replicate to derive domain-wide and strata specific means and standard errors. In addition to the data obtained from quadrat surveys key macroinvertebrates (queen conch [*Strombus gigas*], Caribbean spiny lobster [*Panulirus argus*], long-spined sea urchin [*Diadema antillarum*]) and marine debris observed within the 25 x 4 m transect were recorded. See Chapter 2 for more information on sampling designs and Appendix B for measurement protocols.



Diver collecting benthic data. (CCMA)

3.2.2 Data analysis

A series of analyses were conducted to provide an overall characterization of the benthic composition for the banks. Unbiased, domain-wide estimates of percent cover are provided for the SCC. Following this, comparative and correlative techniques were utilized to identify differences among strata and explore relationships between variables. Through the development of interpolated surfaces and through the technique of clustering, spatial patterning of key variables was observed. A quantification of marine debris is provided as one measure of anthropogenic stress. Finally, the data collected were compared with data from the U.S. Caribbean to determine how this system compares with other systems similar in species composition.

Domain-wide Population Estimates

Domain-wide estimates were computed employing methods described by Cochran (1977) for a stratified sampling design using 2006 data, strata and corresponding sampling weights. Measurements collected in 2007 were not included because the incomplete field mission imposed spatial bias (see Chapter 2 for details regarding sampling design). Mean percent cover and standard error were calculated for each major taxonomic group (coral, macroalgae and sponges) as well as for abiotic data categories (hardbottom, rubble and sand).

Strata Comparisons

For comparative analyses, the 73 sample sites surveyed in 2006 were classified into three different strata: East Bank High relief (EBH), East Bank Low relief (EBL) and West Bank High relief (WBH) depending on benthic relief and geographic location. There were no low relief habitat sites surveyed on the WB, and therefore no West Bank Low relief stratum was used in the analyses or presented in results. Benthic relief was derived from a benthic habitat map developed for this study (see Appendix C).

These strata are different from those actually used for the 2006 sampling design. It must be noted therefore, that estimates of means and variances are not technically valid; however since a random proportional-to-area design was used, the difference between the valid and computed estimates are assumed to be negligible. As with the domain-wide calculations, data from the 2007 mission were not included because the incomplete field mission imposed spatial bias.

Data analysis included mean percent cover of each major taxonomic group (coral, macroalgae and sponges), individual coral species data and abiotic data (hardbottom, rubble and sand). Shannon's diversity (H', log_{10}) and Pielou's evenness (J') indices were calculated on the coral data using MVSP® (Kovach Computing Systems, 1985). Shannon's diversity index (H') is represented as:

$$H' = - \Sigma_i \, p_i \, (\ln p_i)$$

where H' is a weighted combination of: total number of species (richness) and the extent to which the total abundance is spread equally amongst the observed species (evenness) p_i is the proportion of the total count arising from the i th species.

Pielou's evenness (*J'*) index is represented as:

$$J' = H' / \ln S$$

where *S* is the total number of species.

As data did not conform to assumptions of normality due to high frequency of zero values, a non-parametric Kruskal-Wallis test was run on ranked data in JMP© (SAS Institute Inc., 2000) to identify differences among strata (EBH, EBL and WBH). While sample sizes among strata were not equal, impacts on the tests were assumed negligible because data dispersion was similar. Pairwise comparisons were performed using the Nemenyi test (Zar, 1999).

Correlative Analyses

Correlations were investigated among percent cover of corals, macroalgae, and sponges as well as depth using non-parameteric Spearman's analysis on percent cover. Data collected during both the 2006 and 2007 field missions were utilized for these analyses. Results are reported as Spearman's ρ (Rho).

Interpolations

Percent cover data were interpolated and mapped using inverse distance weighting (IDW) to guide interpretation of spatial patterns. These interpolations were created without separating data by strata; therefore, where observed patterns cross strata they must be interpreted with consideration given to strata differences (e.g., a site containing high percent cover in the EBH strata near the border of EBL may result in the adjacent area in EBL appearing high as well which may not reflect reality). Interpolated surfaces were generated for each bank with ArcGIS spatial analyst (ESRI, 2006) using all data from 2006 and 2007. By combining the years we increase our coverage of sample points on EB; however, this assumes no differences between years. EB was further segregated into two portions: the main portion of the bank and a smaller mound that is approximately 470 m northeast of the main portion. Only three surveys were conducted on this smaller portion of EB, thus an interpolated surface for this region was not generated.

Clustering

Cluster analysis was conducted to further understand the spatial distributions of coral species within the SCC. Cluster analysis is an analytical method used to summarize information into groups based on similarities among variables (Sokal and Rohlf, 1995). This was accomplished using the Bray-Curtis proximity coefficient, hierarchical cluster analyses of the species and sites, and a nodal analysis of the species and site cluster analyses. The nodal analysis, the intersection of the species clusters and the site clusters, was used to identify the species assemblages defining the site groupings. Analysis included data from both 2006 and 2007. *Montastraea annularis* complex (composed of *Mo. franksi*, *Mo. faveolata* and *Mo. annularis*) was removed from this analysis; however the unique *Montastraea* species were kept. Additionally, rare coral species (species that occurred in three or fewer samples) were omitted from the analysis, resulting in 18 species for all sample sites.

Prior to calculating the Bray-Curtis coefficient, the coral species percentages were converted to integers by multiplying the coral species cover percentages by 100. A matrix of Bray-Curtis dissimilarity coefficients were calculated for the species and site data. These matrices were processed by the Cluster Procedure in the SAS/STAT© software (SAS Institute Inc., 2006). Scree plots of cluster distances were examined to determine where breaks in the dissimilarity level among the clusters occurred. Nodal analysis was then used to relate coral assemblages with site groups.

Marine Debris

A map depicting the distribution of marine debris was created allowing spatial patterns to be interpreted.

Comparison with U.S. Caribbean

Spatial patterns in benthic composition and diversity were examined across three U.S. Caribbean reef ecosystems that have been monitored using the same methods since 2001 (Menza et al., 2006; Pittman et al., 2007). Methods are further detailed in Section 3.4.1.

3.3 RESULTS AND DISCUSSION

3.3.1 Abiotic Cover

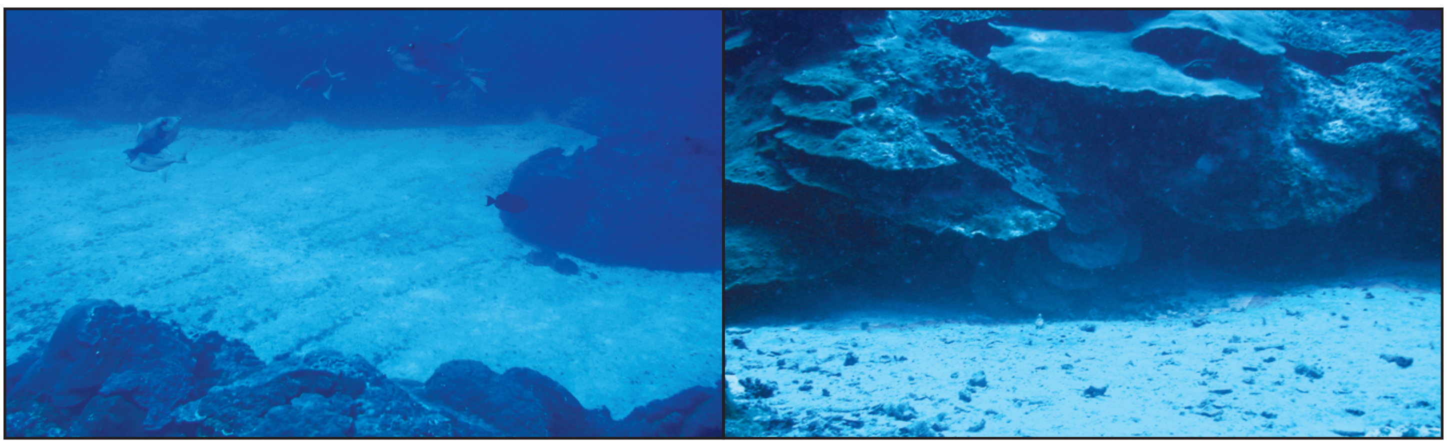
Total hardbottom cover of the SCC was estimated at 89% (Table 3.2). This estimate is consistent with observations by Rezak et al. (1985) who used photo transect surveys to derive a value of 85% hardbottom on the coral caps. Significant differences were found between strata (p=0.0009) with nearly double the coverage in high relief areas (EBH at 92%, WBH at 94%) compared to low relief (EBL at 53.8%).

Table 3.2. Mean percent cover (± SE) for abiotic categories among strata and for the sanctuary.

Strata	N Rows	Hard (± SE)	Rubble (± SE)	Sand (± SE)
EBH	39	93 (2.3)	2 (0.9)	4 (1.6)
EBL	10	54 (12.6)	46 (12.6)	0 (0)
WBH	24	95 (3.0)	2 (1.9)	2 (1.9)
FGB	73	89 (1.6)	8 (1.4)	3 (0.7)

The difference in hardbottom coverage between strata is in part due to a relatively recent large-scale mortality event in the EBL which converted substantial portions of the *Madracis mirabilis* fields to rubble. It is hypothesized that the observed mortality is the result of damage from Hurricane Rita which came within 83 km of the sanctuary in 2005 (Hickerson et al., 2008; Precht et al., 2008b). On average 8% of the SCC was rubble, with significantly more observed in the low relief strata (46%; p<0.0001; Table 3.2). Rubble made up significantly less of the benthic habitat than hardbottom and was completely absent in nearly 75% of transects.

Sand comprised a relatively small proportion of benthic cover (3%). While there are a few large sand patches and sand channels present on the coral caps the majority of sand habitat is restricted to areas under coral plate edges or in between coral colonies. It should be noted that substantial volumes of sand and sediment (up to 1 m deep) were displaced as a result of Hurricane Rita in 2005 and values obtained during the current study may be lower than they were even one year prior as a result (Hickerson et al., 2008; Precht et al., 2008b). No significant differences were detected between the two high relief habitats and no sand was observed on the low relief stratum.



Sand channels between coral colonies (left) and sand patch under coral plate edges (right). (E. Hickerson, CCMA)

3.3.2 Coral Cover

Stony corals dominated the benthic community structure comprising 48% of the benthos (Table 3.3). During 2006 and 2007, a total of 20 corals were identified to the species level among 14 genera. The most dominant in terms of cover were *Montastraea*, *Diploria* and *Madracis*. Gittings (1998) analyzed research studies spanning 20 years and obtained a value of nearly 50% comparing favorably with the current study and emphasizing the stability of the coral community.

Table 3.3. Mean percent cover (± SE) for biotic categories surveyed by strata and for the sanctuary.

Strata	N Rows	Corals (± SE)	Macroalgae (± SE)	Sponges (± SE)
EBH	39	58 (3.3)	14 (2.2)	0.7 (0.2)
EBL	10	32 (6.4)	23 (6.6)	0.7 (0.2)
WBH	24	59 (4.3)	8 (1.6)	0.7 (0.2)
FGB	73	48 (2.0)	13 (1.0)	0.7 (0.1)

The highest coral cover was observed in the high relief strata (EBH=58% and WBH=59%; Figure 3.1) and was almost twice as high as the coral cover found in the low relief stratum (EBL=32%; p=0.0001). Coral cover within each stratum was generally homogenous, showing no consistent spatial pattern. *Montastraea franksi* was the most dominant species on the high relief strata, whereas the most abundant coral in the low relief stratum was *Ma. mirabilis* (Table 3.4). Among these strata both evenness and diversity were lowest on low relief habitats

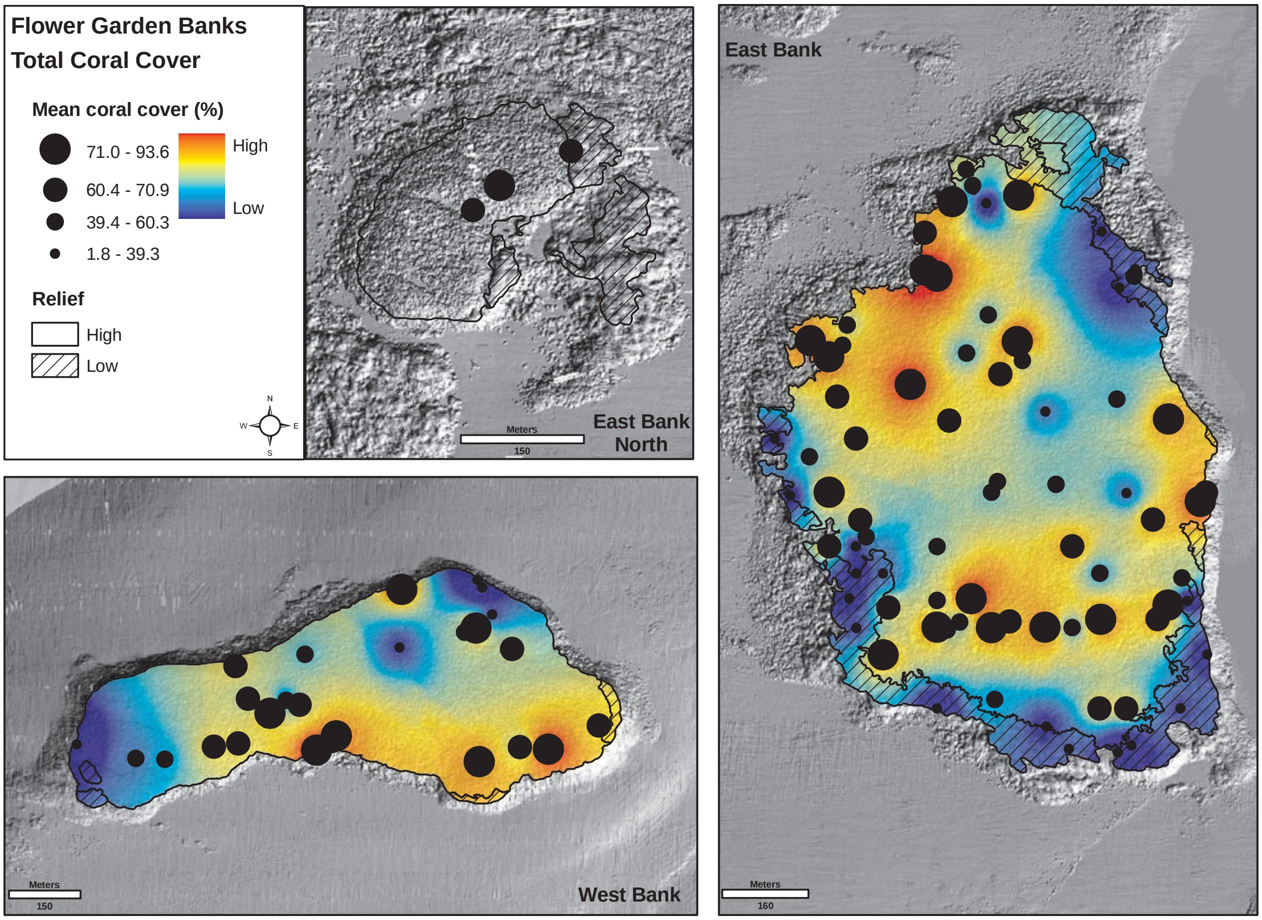


Figure 3.1. Spatial interpolation of mean coral cover among sample sites within FGBNMS.

Table 3.4. Mean percent cover (± SE) of coral species sampled in 2006 among strata.

Coral species	EBH (± SE)	EBL (± SE)	WBH (± SE)
Agaricia agaricites	0.1 (0.03)	<0.1 (<0.1)	0.1 (0.04)
Agaricia fragilis	<0.1 (<0.1)	0	<0.1 (<0.1)
Agaricia spp	<0.1 (<0.1)	<0.1 (<0.1)	<0.1 (<0.1)
Colpophyllia natans	3.3 (1.0)	2.8 (2.6)	1.6 (0.7)
Diploria spp	0.2 (0.2)	0	<0.1 (<0.1)
Diploria strigosa	4.5 (0.8)	3.6 (2.4)	5.6 (2.0)
Madracis decactis	0.4 (0.1)	<0.1 (<0.1)	0.3 (0.1)
Madracis mirabilis	0.5 (0.5)	13.0 (5.0)	<0.1 (<0.1)
Madracis spp	0	0	0.1 (0.1)
Millepora alcicornis	1.0 (0.5)	0.9 (0.5)	0.7 (0.2)
Millepora spp	<0.1 (<0.1)	0	0.4 (0.2)
Montastraea annularis	1.6 (1.0)	0.9 (0.9)	4.6 (2.7)
Montastraea annularis complex	0.8 (0.8)	0	7.9 (4.6)
Montastraea cavernosa	3.7 (1.0)	2.8 (2.4)	3.7 (1.3)
Montastraea faveolata	5.7 (1.7)	0	2.1 (1.0)
Montastraea franksi	32.2 (3.2)	4.1 (2.6)	28.3 (3.9)
Mussa angulosa	0.2 (0.05)	<0.1 (<0.1)	<0.1 (<0.1)
Porites astreoides	3.2 (0.5)	3.8 (1.9)	2.0 (0.3)
Porites spp	0	0	<0.1 (<0.1)
Scolymia cubensis	<0.1 (<0.1)	<0.1 (<0.1)	<0.1 (<0.1)
Scolymia spp	<0.1 (<0.1)	0	<0.1 (<0.1)
Siderastrea siderea	0.4 (0.2)	0	0
Stephanocoenia intercepta	0.4 (0.1)	0.2 (0.1)	1.4 (0.5)

however these differences were not significant (Figure 3.2). A strong, negative relationship was found between total coral cover and both macroalgae (ρ = -0.71) and sponges (ρ = -0.56) and a weak correlation was observed with depth (ρ = -0.16).

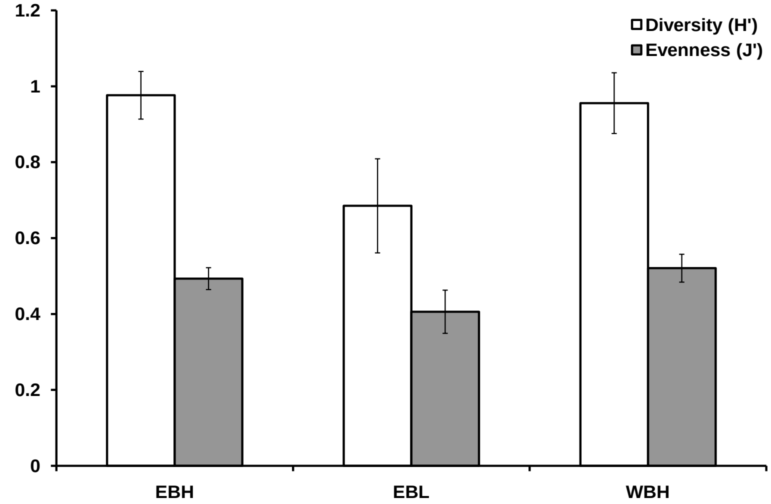
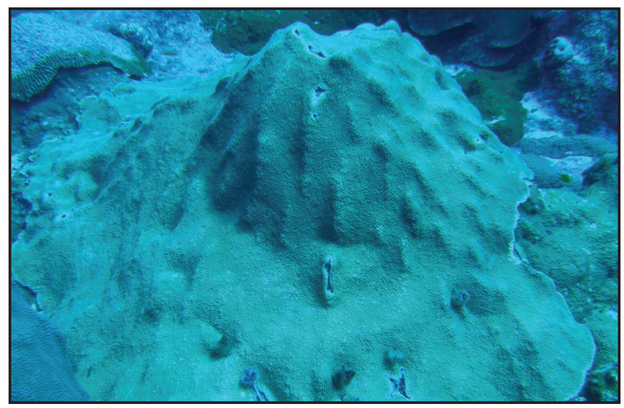


Figure 3.2. Shannon-Weiner diversity indices (H') and evenness (J') for corals surveyed by strata. Error bars represent standard error.

3.3.2.1 Montastraea *species*

The genus *Montastraea* contains four species in the Caribbean, all of which occur within the FGBNMS. They account for over three quarters of the total coral coverage on high relief strata (Table 3.4). These observations concur with those of Rezak et al. (1985) who found *Montastraea* to be the most dominant hermatypic coral on the banks. On the top of the coral caps these species form mounding boulder colonies often taller in size than a diver; however on the edges of the caps in deeper water, plating morphotypes become more prevalent. *Montastraea* colonies typically contain numerous cracks and crevices that provide shelter for smaller fish species while also providing large overhangs and holes suitable for the many large species of grouper residing on the banks.



Montastraea faveolata (CCMA)

It must be noted that the following species specific analyses may be confounded by the occasional recording of *Mo. annularis* complex *in lieu* of other species of *Montastraea* with the exception of *Montastraea cavernosa*. This classification was utilized six times in 2006 and once in 2007.

3.3.2.1.1 *Montastraea franksi*

Mo. franksi made up the largest proportion of cover among all corals in high relief strata (EBH=32.2% and WBH=28.3%; Table 3.4). Although it was significantly less common in the low relief stratum (p=0.0006), it was the second most dominant coral species (4.1%) in that area and its distribution was very similar to the distribution of total coral cover (Figure 3.3). *Mo. franksi* also displayed strong moderate relationships with both macroalgae (ρ = -0.40) and sponges (ρ = -0.35) and a weak relationship with depth (ρ = -0.15).

3.3.2.1.2 *Montastraea faveolata*

Mo. faveolata was present in fewer than half of the sites surveyed and completely absent from surveys on the EBL stratum. Overall cover in the high relief strata was not significantly different with 5.7% in EBH and 2.1% in WBH. *Mo. faveolata* was generally more common along the eastern half of the EB, with a relatively low representation within the WB (Figure 3.4). *Mo. faveolata* showed a strong negative relationship with depth (ρ = -0.32) but no relationship with either macroalgae or sponges.

3.3.2.1.3 *Montastraea cavernosa*

Mo. cavernosa cover was relatively low (approximately 4%) and patchily distributed (Figure 3.5). Unlike the other *Montastraea* species, *Mo. cavernosa* cover estimates were similar among the varying relief strata, exhibiting no relationship with depth or presence of either macroalgae or sponges.

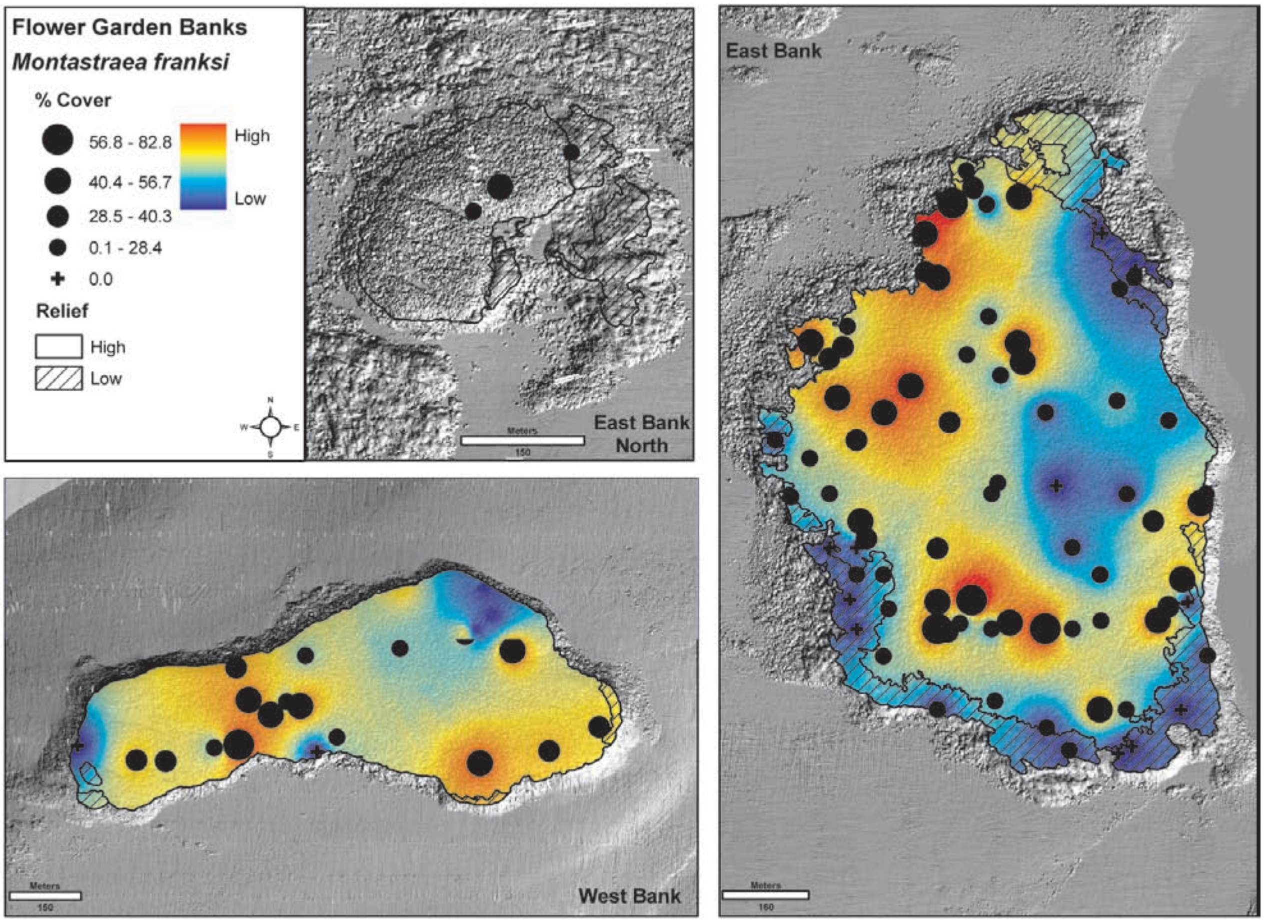


Figure 3.3. Spatial interpolation of Montastraea franksi *among sample sites within FGBNMS.*

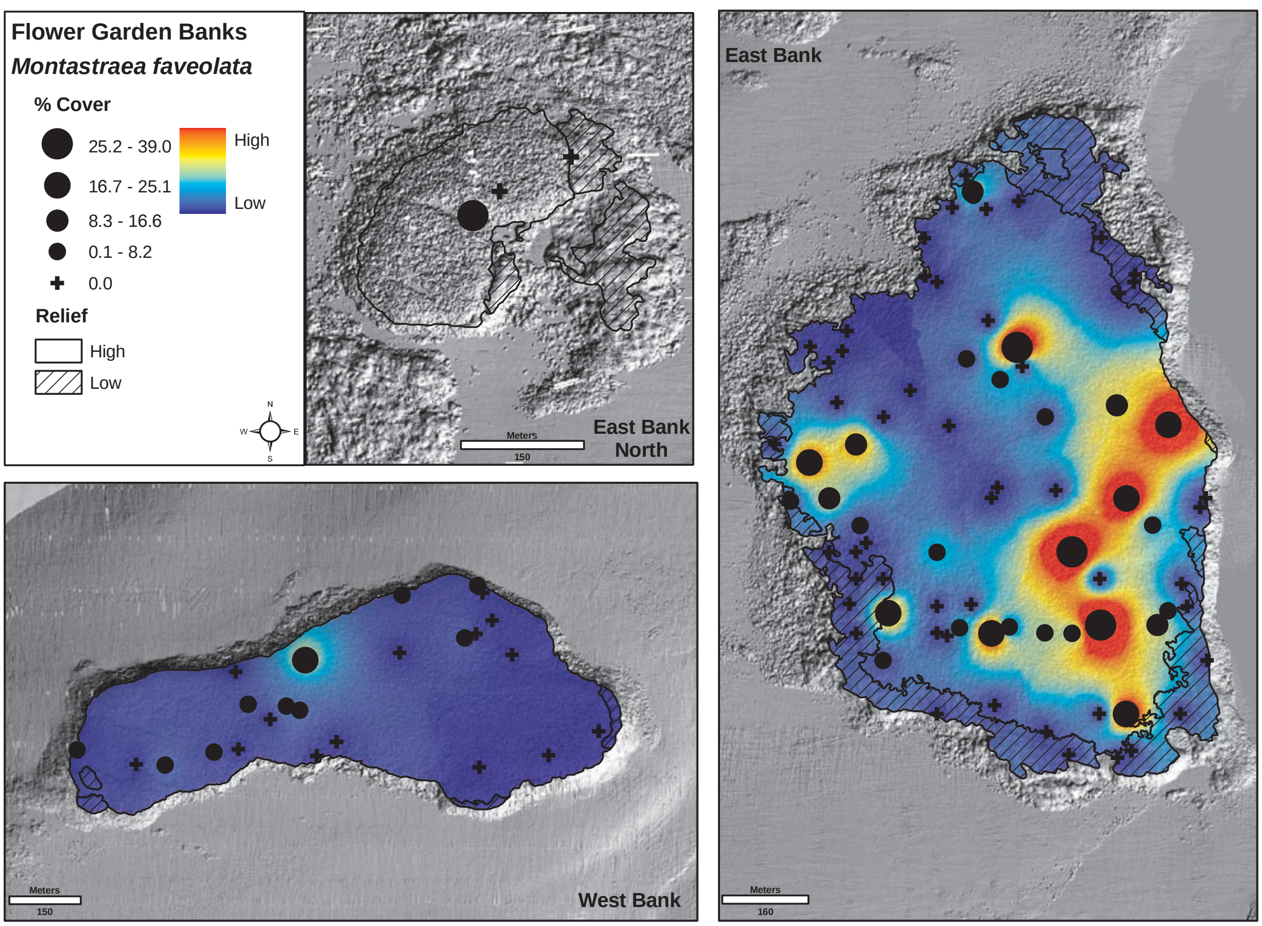


Figure 3.4. Spatial interpolation of Montastraea faveolata *among sample sites within FGBNMS.*

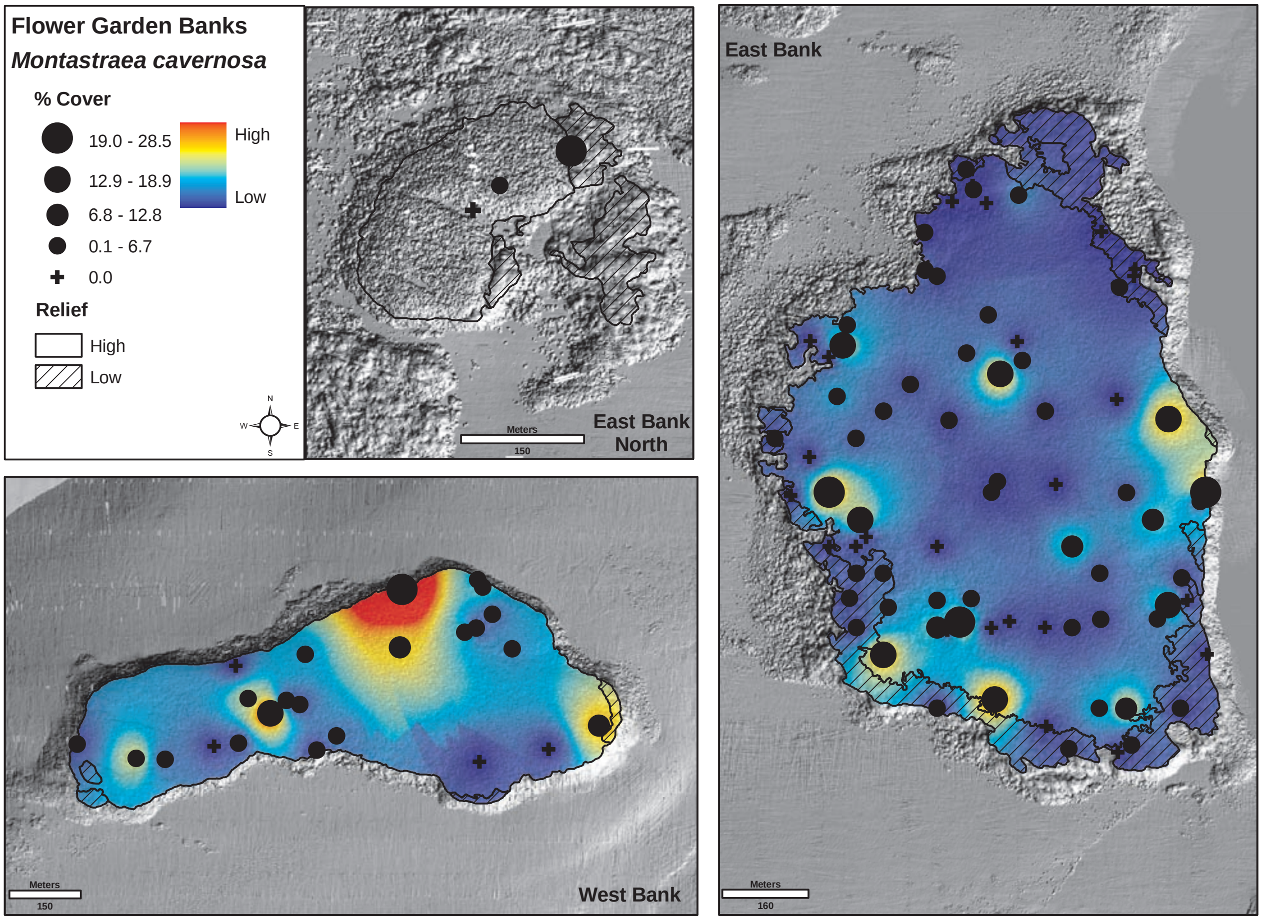


Figure 3.5. Spatial interpolation of Montastraea cavernosa *among sample sites within FGBNMS.*

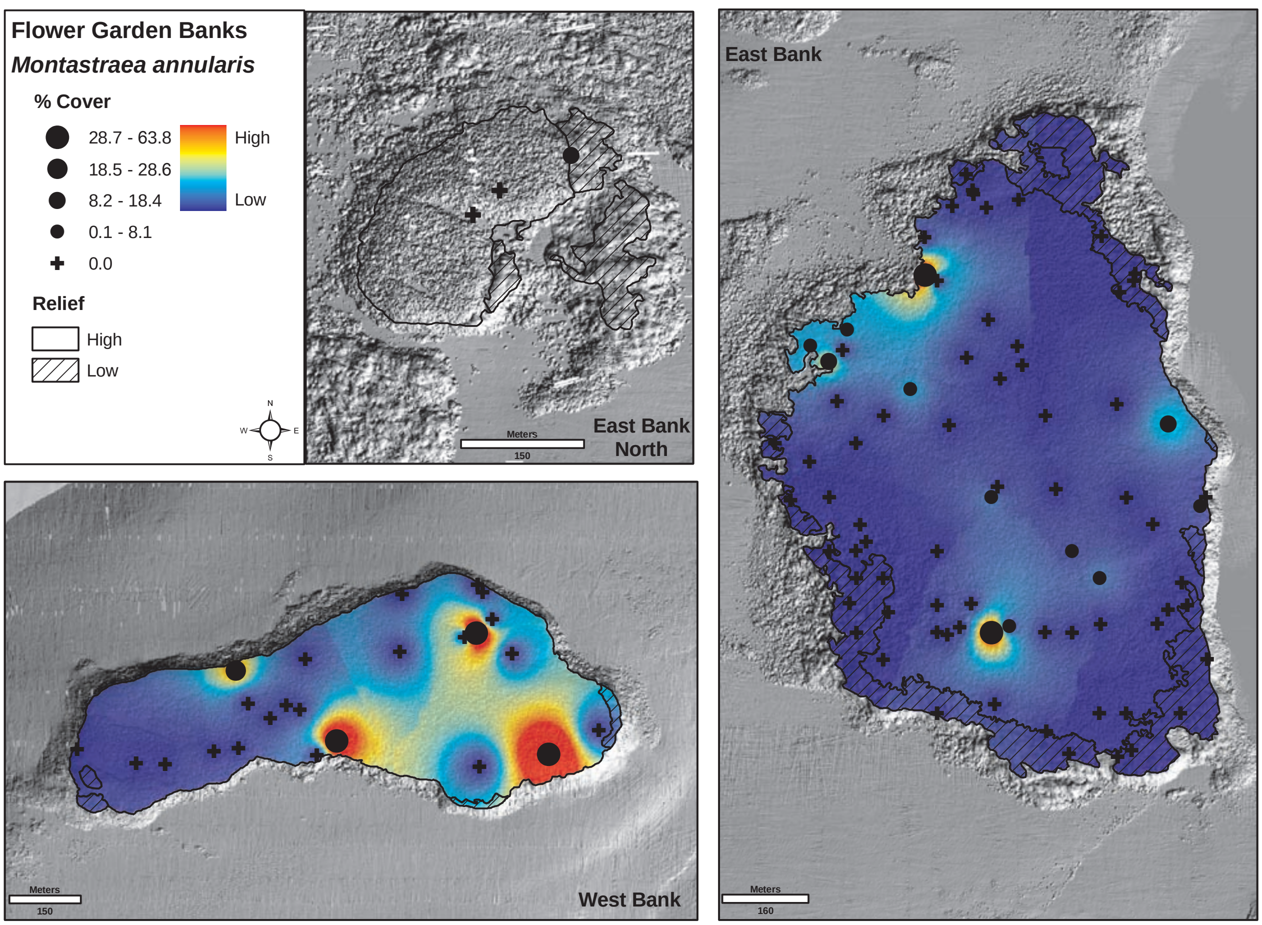


Figure 3.6. Spatial interpolation of Montastraea annularis *among sample sites within FGBNMS.*

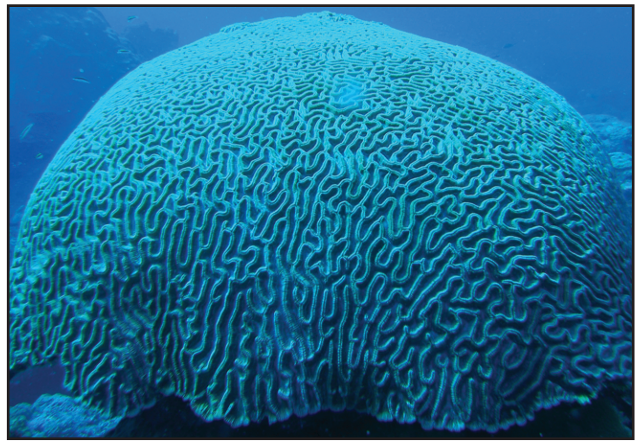
3.3.2.1.4 *Montastraea annularis*

Only ten surveys in 2006 contained recorded instances of *Mo. annularis* and only one of those was from the EBL. No significant differences were detected among strata. *Mo. annularis* cover among strata ranged from <1% in EBL to 2% in EBH and 5% in WBH (Table 3.4). Similar to *Mo. cavernosa*, *Mo. annularis* showed a patchy distribution with no discernible spatial patterning (Figure 3.6). In spite of the low frequency of occurrence, moderately negative relationships were observed with depth (ρ = -0.21), macroalgae (ρ = -0.36) and sponges (ρ = -0.26).

3.3.2.2 Diploria *species*

The genus *Diploria* contains members of the larger group commonly referred to as the brain corals. Similar to members of the genus *Montastraea*, colonies of *Diploria* take boulder form in shallower waters and tend to plate out at deeper depths. Only one species, *Diploria strigosa*, was observed during the course of the surveys on the coral caps. As a result of its size (up to 2 m) and its relative abundance, *D. strigosa* contributes substantially to the habitat complexity of the coral cap floor.

It should be noted that in 2006 there is one instance of a *Diploria* colony that was not identified to the species level.



Diploria strigosa colony (CCMA)

3.3.2.2.1 *Diploria strigosa*

D. strigosa was the second most dominant coral species in terms of coral cover on the SCC and was relatively evenly distributed among strata (Table 3.4). While no significant differences were detected, the highest coverage was observed within the interior portions of the EB and along the edges of the WB (Figure 3.7). This contrast in distribution between the banks merits further investigation. There was a moderately negative correlation between this species and depth (ρ = -0.31) as well as macroalgae (ρ = -0.27); however no relationship was observed between *D. strigosa* and sponges.

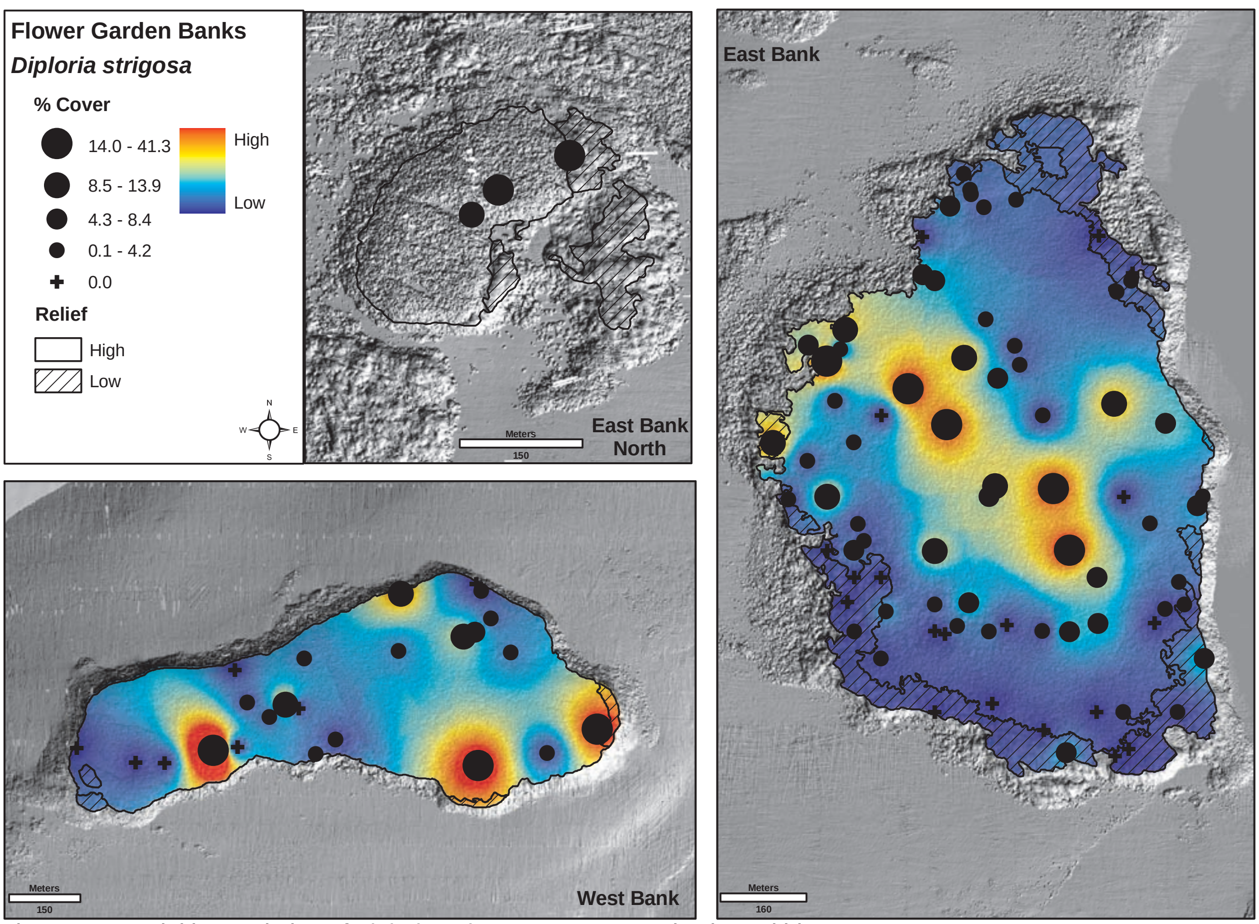
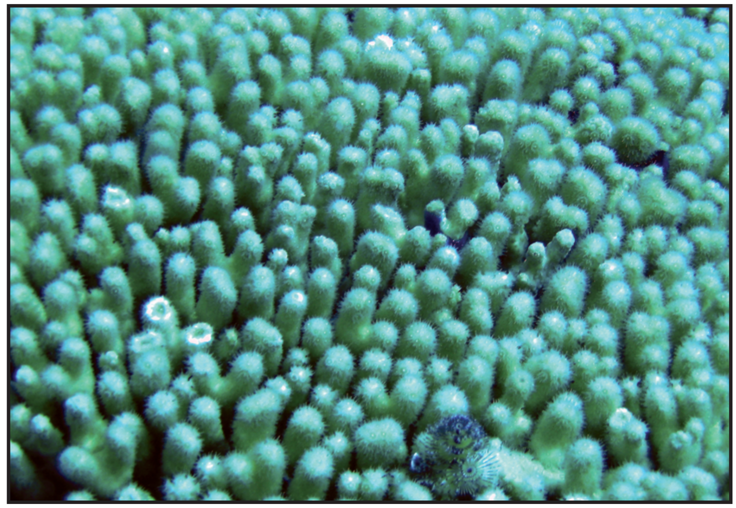


Figure 3.7.Spatial interpolation of Diploria strigosa *among sample sites within FGBNMS.*

3.3.2.3 Madracis *species*

Corals of the genus *Madracis* are most commonly encountered in deeper waters around the Caribbean. In morphology *Madracis* species can be knobbed, lobed, encrusting, branching or a combination adding additional complexity to the benthos resident on the banks and providing additional structure for a variety of reef fish species. At the FGBNMS, seven species have been recorded, the majority during submersible surveys in waters greater than 45 m. Two of the shallower water species in the genus were observed during the course of this study, *Ma. mirabilis* and *Madracis decactis*; however, the latter contributed less than 0.4% coverage in any of the strata.



Madracis mirabilis (CCMA)

3.3.2.3.1 <u>*Madracis mirabilis*</u>

Ma. mirabilis was the most dominant coral within the EBL stratum (13%) where it blanketed large areas at depths below the more complex *Montastraea* and *Diploria* dominated portion of the caps. Large portions of this stratum contained *Ma. mirabilis* rubble from colonies thought to be impacted by a recent hurricane, which suggests that coverage historically was higher. *Ma. mirabilis* coverage was significantly less in high relief strata ($p<0.0001$) than in the EBL (Table 3.4; Figure 3.8) and showed a moderate positive relationship with depth (ρ = 0.29) and macroalgae (ρ = 0.22) and a weak relationship with sponges (ρ = -0.13). Although *Ma. mirabilis* is not as abundant overall as the other two genera analyzed, it may prove critical to the sanctuary's fish population as its branching morphology provides substantial shelter for juvenile fish species. This could be particularly important on the banks where typical nursery habitats such as seagrass beds and mangroves are absent. Additionally, there is evidence from blowholes at Bright Bank (east of the FGBNMS) to suggest that *Madracis* may be a principal framework builder for the banks on the outer shelf, probably by virtue of its relative fast growth and ability to recover after fragmentation during storms (Gittings, pers. comm.).

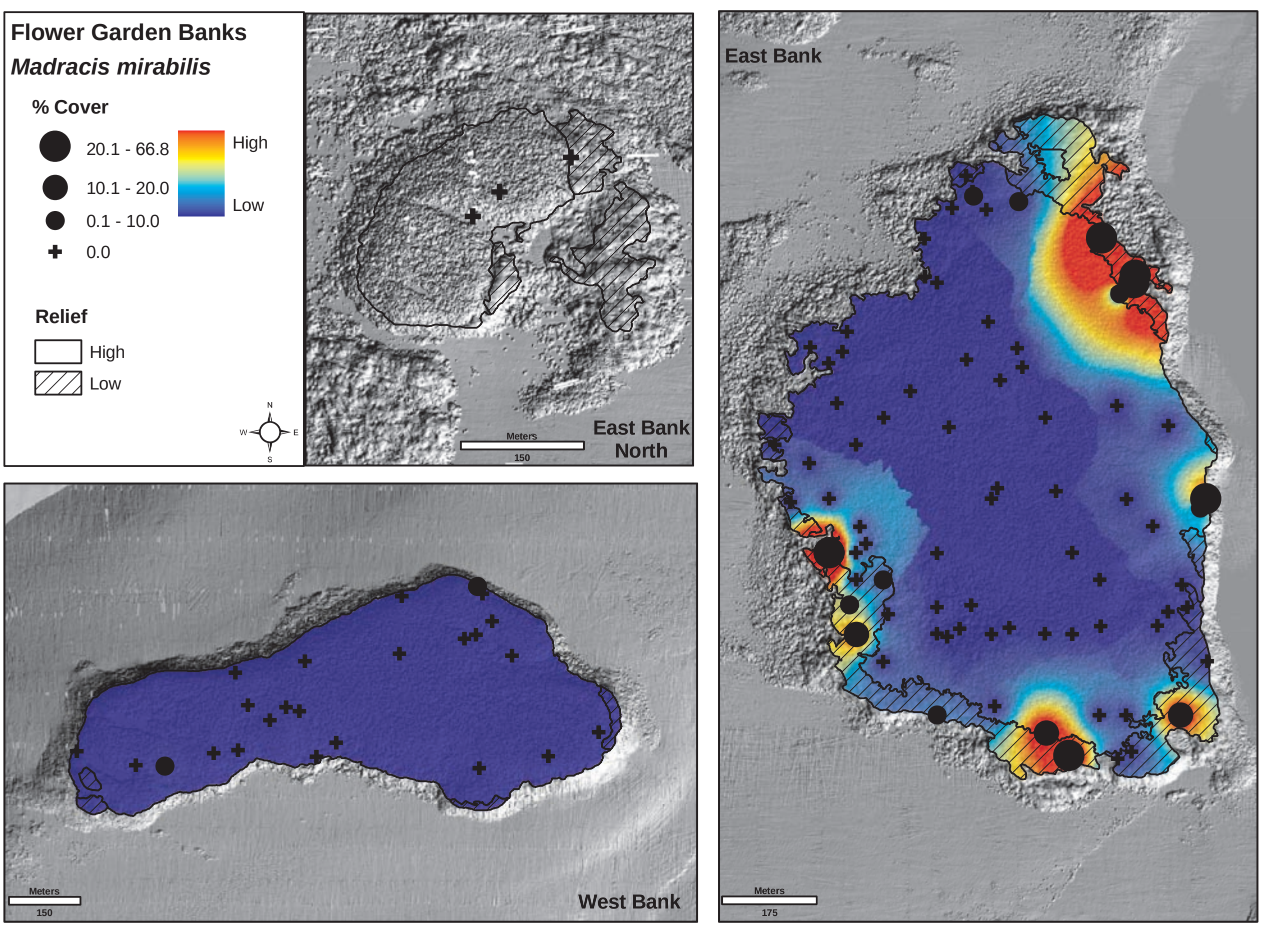


Figure 3.8. Spatial interpolation of Madracis mirabilis *among sample sites within FGBNMS.*

3.3.2.4 *Distribution of Coral Assemblages*

The nodal analysis of species and site groups utilizing the Bray-Curtis proximity coefficient revealed additional information about spatial patterning among corals (Figure 3.9). Group 1 was defined by the dominance of *Ma. mirabilis* and relatively low species richness. Groups 2 and 3 were dominated by *Stephanocoenia intercepta* and *Porites astreoides,* respectively; however other species such as *Mo. cavernosa*, *Mo. franksi* and *D. strigosa* were also present. Both groupings also contained colonies of *Ma. mirabilis*. These first three groupings were observed almost exclusively in the deeper water at the edges of the survey domain and comprised the majority of surveys conducted on the EBL. Group 4 was the largest of the groupings and contained the highest occurrences of the species most dominant on the high relief strata including *Mo. franksi*, *Mo. faveolata* and *D. strigosa*.

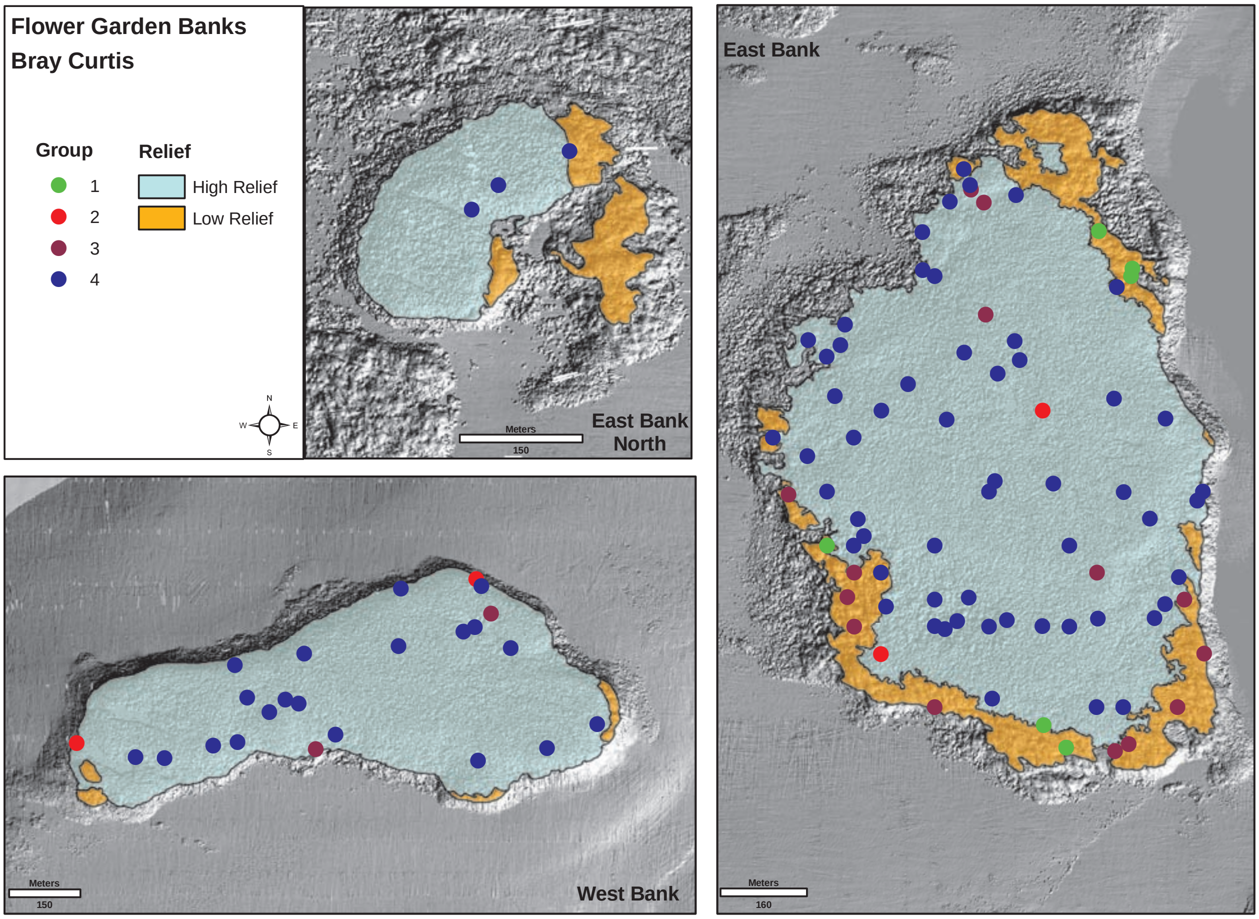
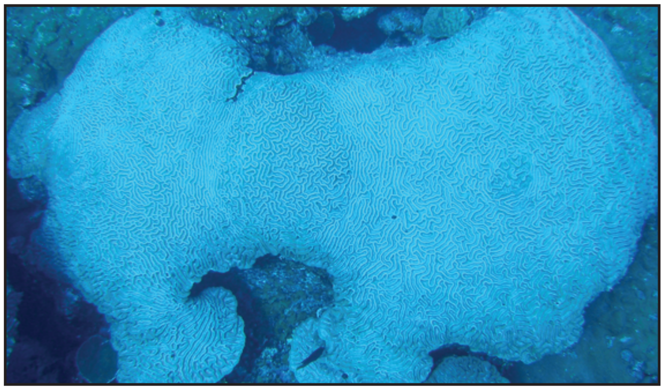


Figure 3.9. Spatial array of the site groups utilizing the Bray-Curtis proximity coefficient and color coded for reference.

3.3.3 Coral Bleaching

Throughout the Caribbean bleaching events have become more prevalent primarily due to increased sea surface temperatures (Wilkinson and Souter, 2008). An increase in temperature of even 1-1.5°C above summer maximums causes severe stress on coral colonies and results in the expulsion of zooxanthellae (Kleypas and Hoegh-Guldberg, 2008). Within the FGBNMS, there has been an increase in recent years of bleaching accounts; however, resident corals have demonstrated excellent recovery capabilities and re-establishment of symbiotic algae after bleaching (Gittings, 1998). According to reports by Hickerson et al (2008) bleaching estimates following two months of elevated temperatures in 2005 reached an estimated maximum of 46% of individual colonies. By March of 2006, bleaching had declined to 4%.



Bleached *Diploria strigosa* colony (CCMA)

In the current study, bleached corals comprised 18% of total coral cover (this includes *Millepora* spp.; 17% without). It must be noted that this estimate is different than proportion of colonies impacted as reported by Hickerson et al. (2008). Species specific bleaching estimates ranged widely from highly impacted species such as, *Millepora alcicornis* (92%), *Siderastrea siderea* (53%) and *Mo. cavernosa* (40%) to species minimally impacted, including *S. intercepta* (8%), *Colpophyllia natans* (7%) and *Ma. decactis* (3%). Several species reported no signs of bleaching. While bleaching showed no relationship with depth (Figure 3.10), the deeper water *Ma. mirabilis* was one of few coral species not impacted by the bleaching event (Table 3.5). Data presented are concordant with observations from Hickerson et al. (2008), who also observed high levels of susceptibility to bleaching in both *Mi. alcicornis* and *Mo. cavernosa*.

Table 3.5. Percent bleaching among coral species at FGBNMS in 2006.

Coral species	% bleach	Coral species	% bleach
Agaricia agaricites	19.8	*Montastraea faveolata*	27.1
Agaricia fragilis	38.9	*Montastraea franksi*	18.7
Colpophyllia natans	6.5	*Mussa angulosa*	0
Diploria strigosa	20.2	*Porites astreoides*	11.1
Madracis decactis	2.6	*Scolymia cubensis*	0
Madracis mirabilis	0	*Scolymia spp*	0
Millepora alcicornis	91.6	*Siderastrea siderea*	53.1
Montastraea annularis	18.2	*Stephanocoenia intercepta*	7.5
Montastraea cavernosa	39.9		

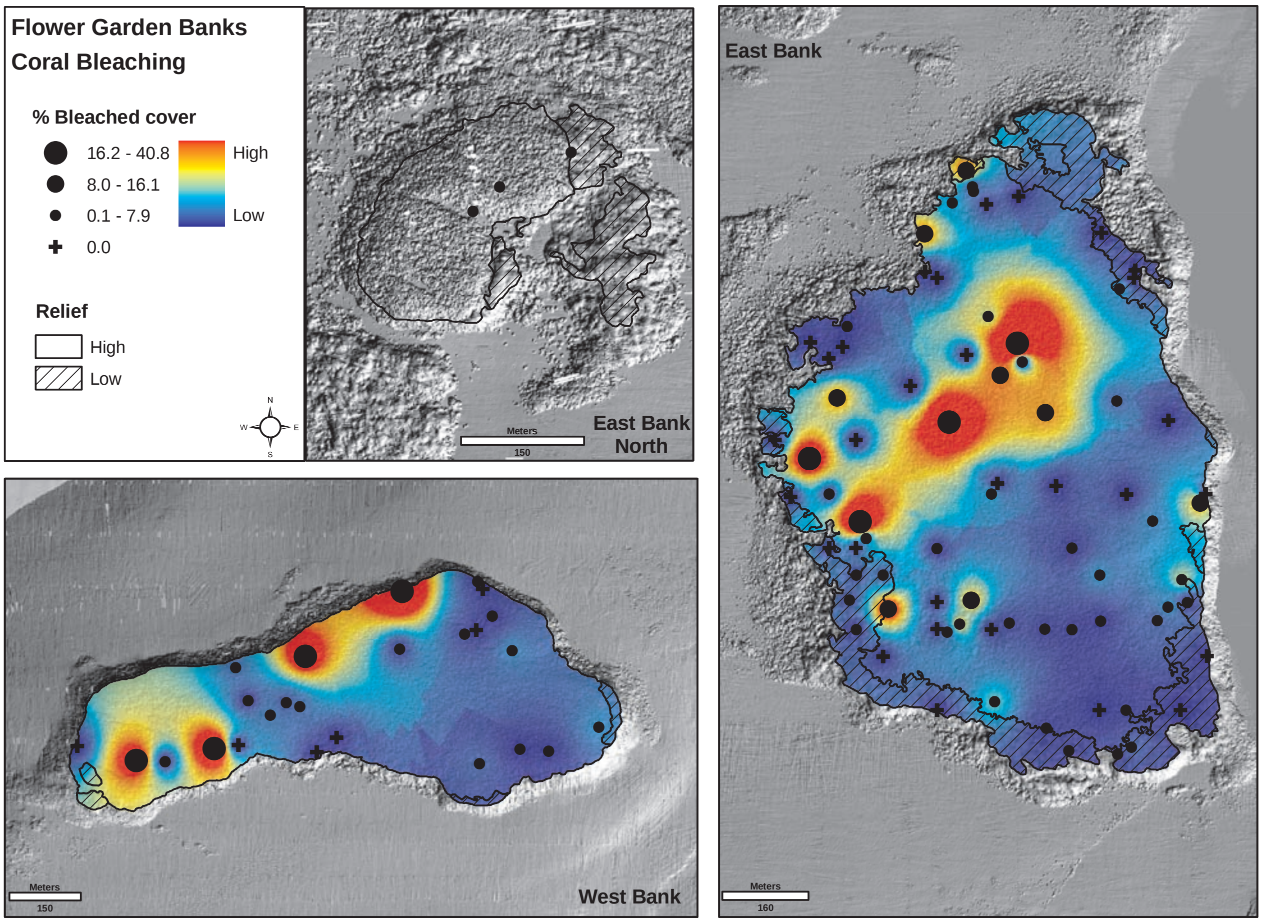
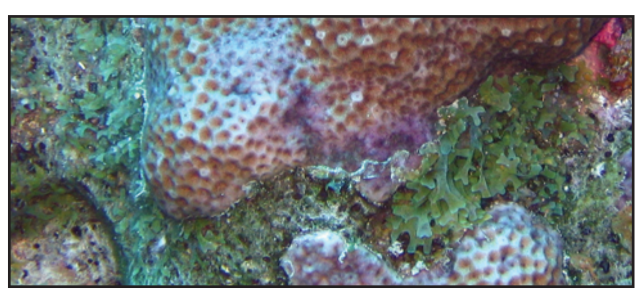


Figure 3.10. Spatial interpolation of bleached coral among sample sites within the FGBNMS.

3.3.4 Algal Cover

Algae covered approximately one quarter (28%) of the coral caps. This estimate is less than similar habitats in most Caribbean systems, which have experienced dramatic changes in benthic composition associated with declining coral populations. Macroalgae was the predominant functional algae group on the banks (13%), followed closely by turf algae (11%) and then crustose coralline algae (CCA; 4%; Table 3.6).



Dictyota spp. around *Siderastrea* spp. (CCMA)

Macroalgal cover, dominated by *Lobophora* and *Dictyota* spp., varied between 0% and 59% across sites. Significant differences were detected among strata (p=0.0332); however, the Nemenyi test was unable to parse out these differences possibly due to unequal sample sizes. In general, macroalgae was most common on EBL followed by EBH and finally WBH (Table 3.6; Figure 3.11). While macroalgae distribution was inversely related to live coral cover (ρ = -0.71) it had a positive relationship with both depth (ρ = 0.23) and sponges (ρ = 0.51). The strong relationship with live coral cover may reflect competition among these two benthic colonizers for space.

Table 3.6. Mean percent cover (± SE) of algal classes by strata and for the sanctuary. CCA=Crustose coralline algae.

Strata	N Rows	Macroalgae (± SE)	CCA (± SE)	Turf algae (± SE)
EBH	39	14 (2.2)	3 (0.6)	9 (1.9)
EBL	10	23 (6.6)	2 (0.7)	10 (3.7)
WBH	24	8 (1.6)	4 (1.0)	17 (3.2)
FGB	73	13 (1.6)	4 (0.5)	11 (1.6)

Turf algae was dominant within the WBH (17%) stratum at nearly double the mean percent cover of either habitat type on EB (Table 3.6); although, only differences between WBH and EBH were significant (p=0.0247). As with macroalgae, turf algal cover was moderately related to sponge cover (ρ = 0.30) and negatively related to coral cover (ρ = -0.21); however, no relationship was detected with depth.

Rezak et al. (1985) found CCA to be the dominant algal cover on the banks as a whole; however, on the coral caps CCA cover was relatively low (<4%) among the benthic habitat community compared to other functional groups. While no significant relations were observed among strata, similar to the other two functional groups, CCA was negatively related to coral cover (ρ = -0.39) and positively related to sponges (ρ = 0.23). No relationship was detected with depth.

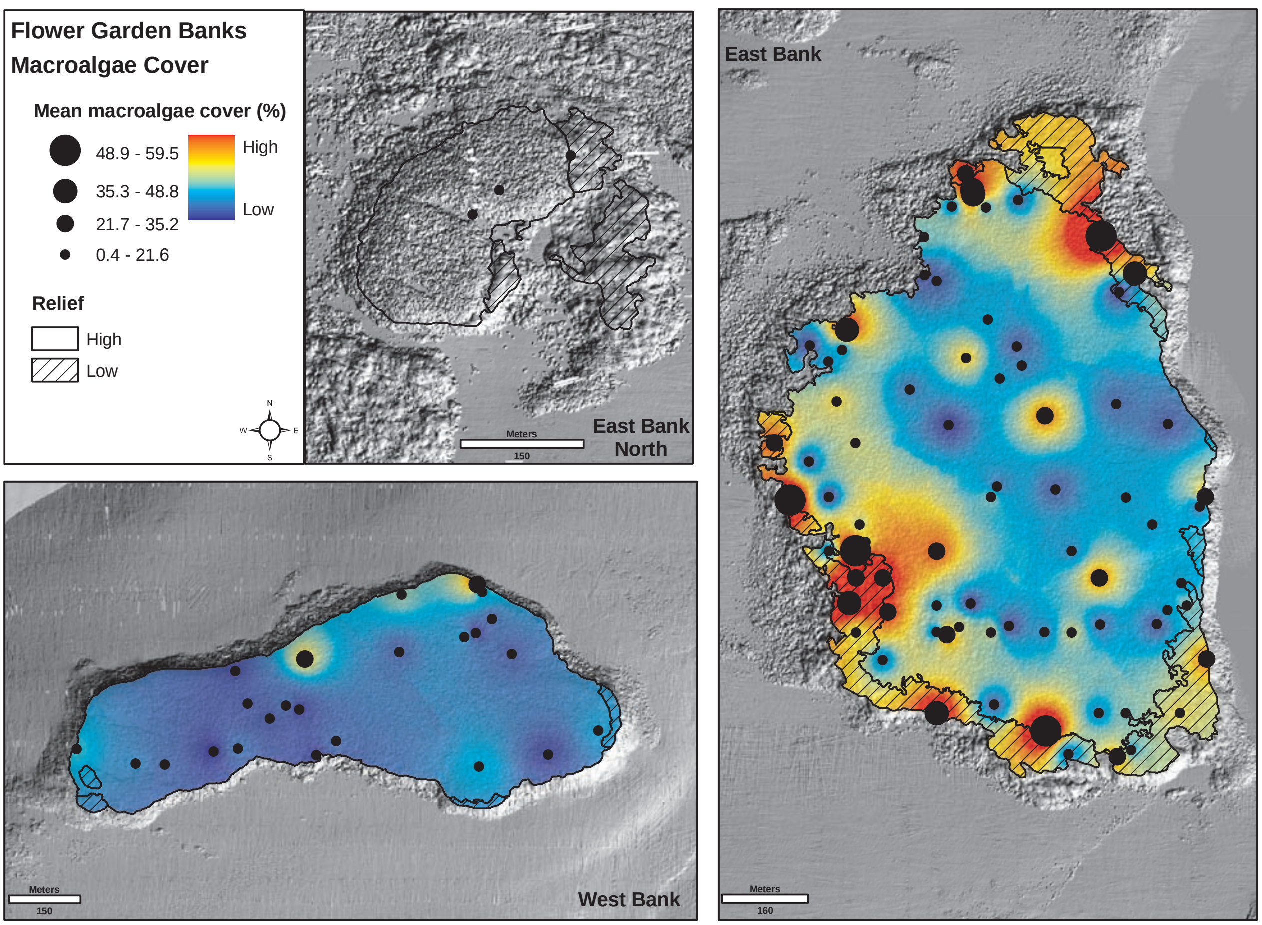
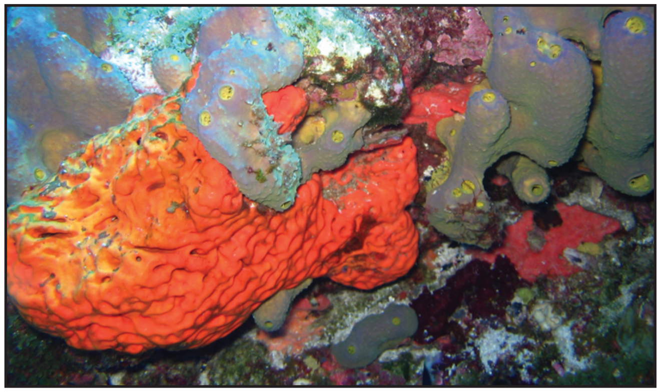


Figure 3.11. Spatial interpolation of macroalgae among sample sites within the FGBNMS.

3.3.5 Sponge Cover

Sponges made up a small proportion of the coral cap's benthic habitat. Mean sponge cover was <1% among all strata (Table 3.7) with only a third of the sites (25 out of 73) having more than 0.5% cover. Those sites with more than 1% cover were generally deeper and many were in low relief habitat, although differences among strata were not observed.



Sponge species *Agela clathrodes* and *Aiolochroia crassa.* (E. Hickerson)

The sponge community of the SCC was classified into two groupings based on morphology: the more commonly encountered group including barrel, tube and vase sponges and a group of encrusting sponges. The multitude of morphologies represented by the two groups provides a number of different habitats for fish species. Neither grouping showed clear spatial patterns by bank or depth although as previously mentioned sponges as a whole showed strong relationships with both corals (ρ = -0.56) and macroalgae (ρ = 0.51).

Table 3.7. Mean percent cover (± SE) of compiled sponges, barrel/tube/vase sponges (Porifera btv) and encrusting sponges (Porifera enc) by strata and for the sanctuary.

Strata	N Rows	Sponges (± SE)	Porifera btv (± SE)	Porifera enc (± SE)
EBH	39	0.7 (0.2)	0.4 (0.2)	0.3 (0.1)
EBL	10	0.7 (0.2)	0.5 (0.2)	0.2 (0.1)
WBH	24	0.7 (0.2)	0.4 (0.2)	0.3 (0.1)
FGB	73	0.7 (0.1)	0.4 (0.1)	0.3 (0.1)

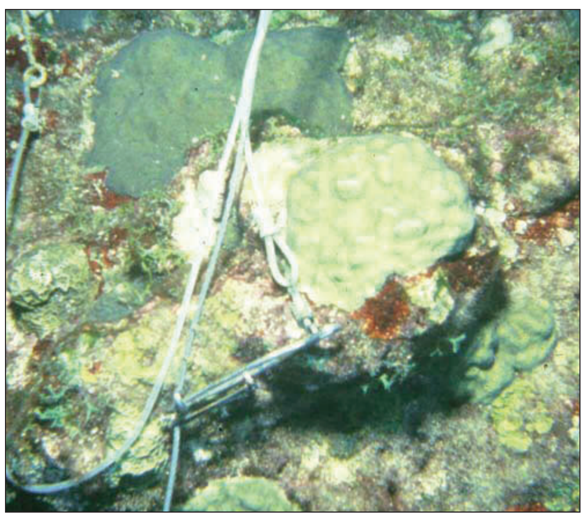
The sponge community of the banks is not prevalent above the 33.5 m (110 ft) isobath of the SCC, but it thrives in waters beyond the limits of conventional scuba diving. According to Rezak et al. (1985) the sponges, particularly *Neofibularia nolitangere*, are most dominant between depths of 46-82 m on the EB and 46-88 m on the WB.

3.3.6 Macroinvertebrates

Macroinvertebrates were included in the surveys to capture other key invertebrate organisms within the SCC. During the course of the 2006 and 2007 surveys only one Caribbean spiny lobster (*P. argus*) and one long-spined urchin (*D. antillarum*) were observed. *P. argus* have been reported as rare previously (Pequegnat and Ray, 1974) and a total of only four were recorded at the banks between 1998 and 2005 during the FGBNMS long-term monitoring (FGBNMS LTM) study (Dokken et al., 2003; Precht et al., 2006; Precht et al., 2008a). The density of *D. antillarum*, however, was historically reported as high as 2 individuals/m^2 (Burke, 1974). This was prior to the massive die-off that occurred region wide in the early 1980s. The most recent estimates provided from the FGNMS LTM study show *D. antillarum* densities between 0.005-0.11 individuals/m^2 (Precht et al. 2008a). Limited information is available on queen conch (*S. gigas*) on the banks; however initial population estimates are being researched (http://flowergarden.noaa.gov/science/conch_burnside.html).

3.3.7 Marine Debris

Marine debris at FGBNMS consists primarily of materials from scientific experiments destroyed in storms; oil exploration (seismic cables); vessels (lines, cables and anchors); and fishing gear (longlines and hook-and-line gear; Gittings, pers. comm.). During 2006, marine debris was reported at 10 sites (7.3% of all sites); four sites on the EB and six sites on the WB (Figure 3.12). Consistent with Gittings' observations, reported debris included anchors, fishing line and rope (Table 3.8). The anchors and associated anchor line observed were colonized by sizeable coral heads suggesting a lengthy period of time since their appearance on the reef. During 2007, marine debris was reported at only one site before the mission was terminated early. There may be a correlation between reports of marine debris and distance from the mooring buoys but that is speculative and more data are needed to target this research question.



Marine debris (Burek)

Marine debris is a potential threat to sessile benthic fauna as well as more mobile resident and transient fauna (i.e., sea turtles, sharks and reef fishes) within the sanctuary, either directly through entanglement (e.g., fishing line) or indirectly though habitat degradation (e.g., anchor damage). Our data indicate marine debris is present but is encountered relatively infrequently. Continued marine debris monitoring is needed to identify areas more prone to accumulation and confirm the apparent low frequency of debris introduction.

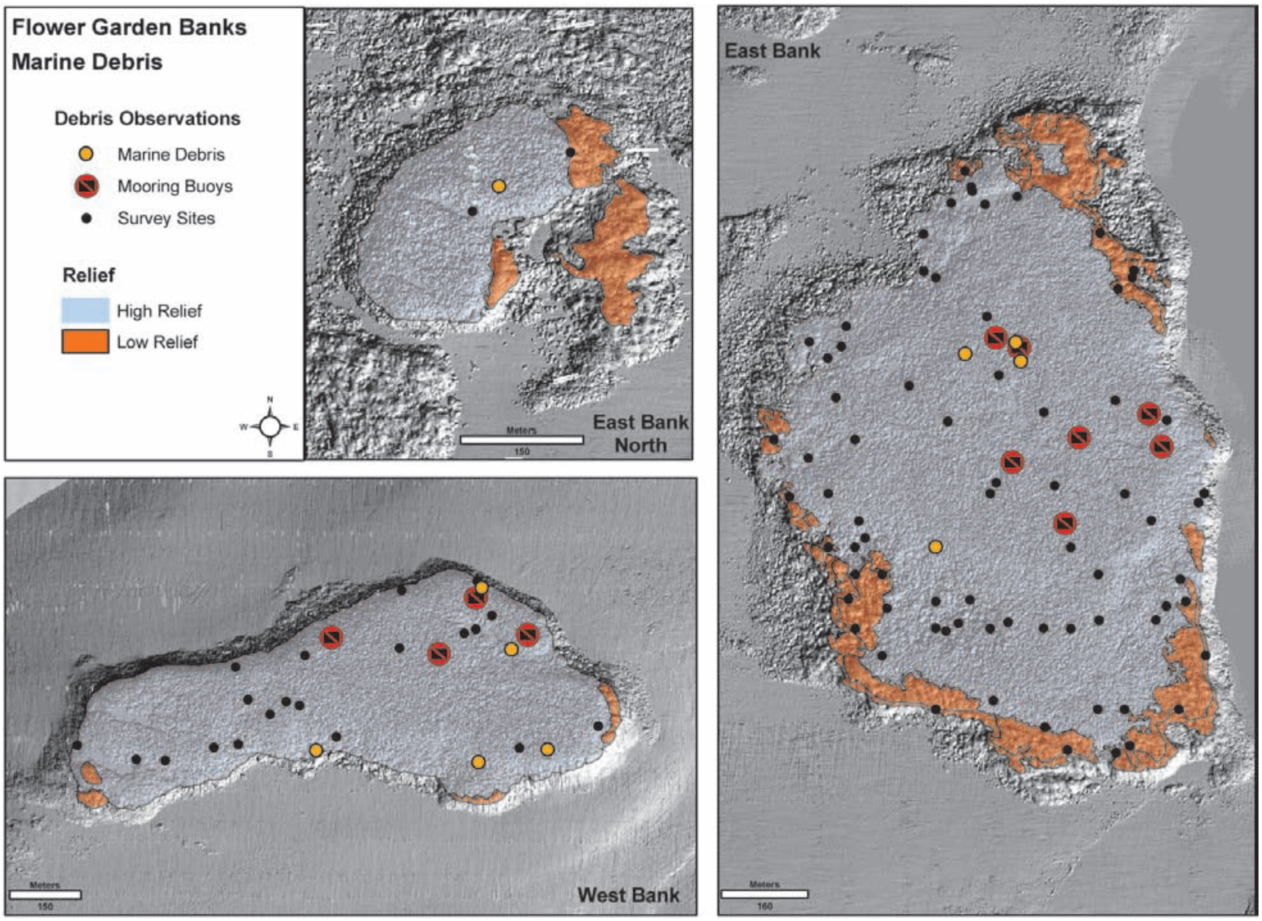


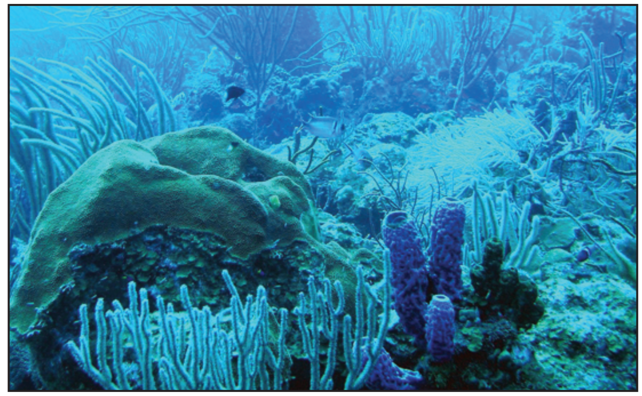
Figure 3.12. Locations of marine debris reports during 2006-2007 surveys in the FGBNMS. The locations of mooring buoys are noted for reference.

Table 3.8. Marine debris documented in the FGBNMS during 2006 and 2007 sampling.

Site #	Debris Type	Debris area (cm^2)	Colonized By	Area affected (cm^2)
2006				
EBF79	1” line - grown over	366	hard corals, calcareous algae, macroalgae	366
WBF8	anchor	75	macroalgae	75
WBF8	anchor line	50	algae and corals	50
WBF19	anchor	700	*Millepora alcicornis*	700
EBF43	anchor (old)	18	coral encrusted	18
EBF75	anchor rope	1000	reef/macroalgae	1000
WBS4	fishing line	10	*Millepora* spp.	10
EBF62	fishing line	150	crustose coralline algae	150
WBF30	fishing lure w/ monofilament	40	none	40
WBF7	plastic coated cable	40	turf algae	50
WBF16	rope	400	turf algae	1000
2007				
E981	fishing line	100	encrusting sponge	100

3.4 CARIBBEAN COMPARISON

Impacts such as chronic over-fishing, pollution, climate change and disease have deteriorated reefs globally. Resulting losses observed in coral cover and large predators have serious ramifications to supporting ecological function and diversity in reef ecosystems (Gardner et al., 2003; Sandin et al., 2008). As a mechanism for estimating the measure of these impacts, scientists have provided examples of comparatively pristine reefs in the Pacific Ocean (Friedlander and DeMartini, 2002). Few examples exist (e.g., Bonaire) or have yet to be described in the tropical western Atlantic Ocean. While we recognize the physical and geomorphological differences between the FGBNMS coral reefs and those in the Caribbean, the similarities in marine fauna, provide us an excellent opportunity to make similar comparisons between impacted and relatively non-impacted systems.



Caribbean shallow-water habitat type in St. John, USVI. (CCMA).

Benthic habitat data collected in 2006 was compared to other sites in the Caribbean (La Parguera, Puerto Rico; St. Croix and St. John, USVI; Figure 3.13) that have been monitored by CCMA using the same methods. To obtain sufficient sample size for analysis 2003-2006 data from the Caribbean locations was pooled. This assumes that there were no major changes in cover between the years and may be an invalid assumption given the 2005 coral bleaching event.

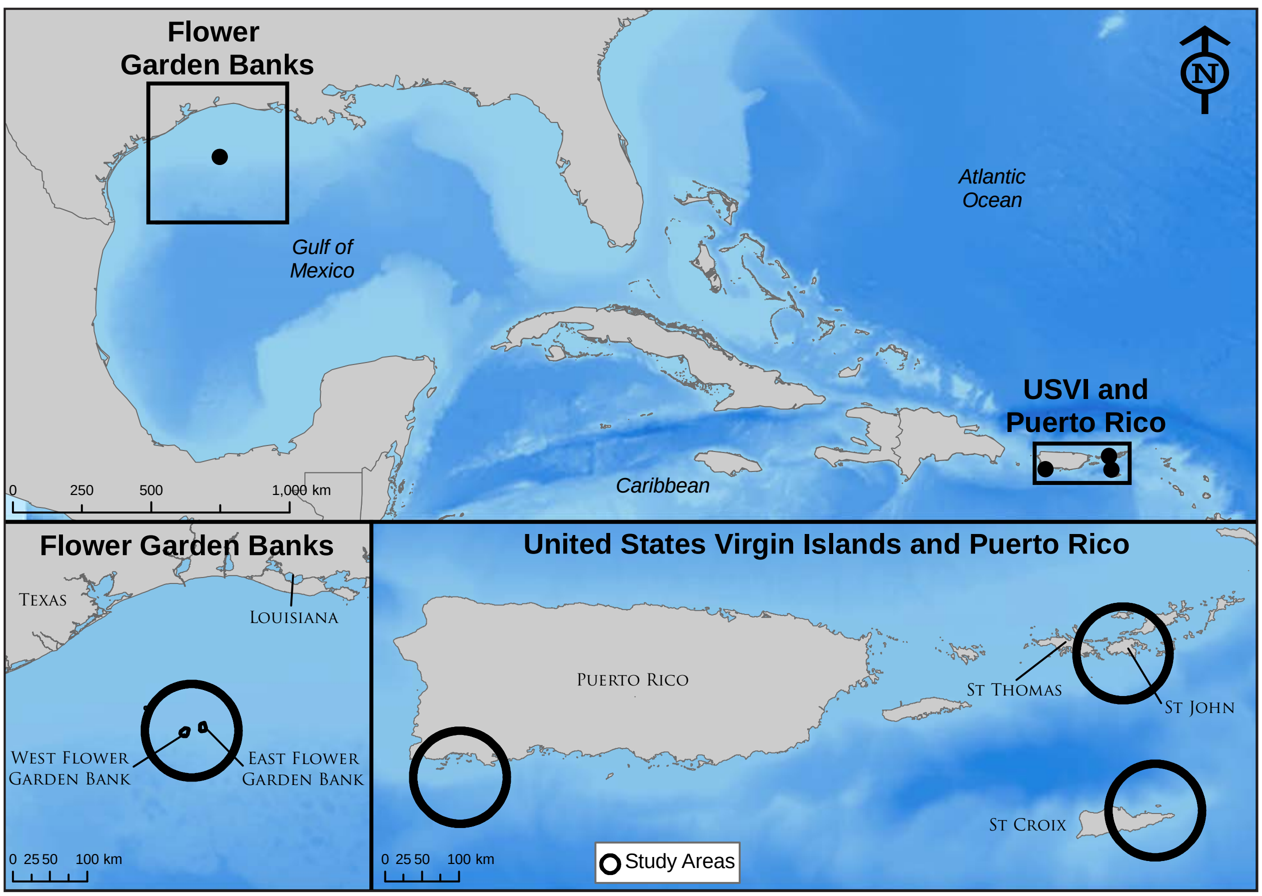


Figure 3.13. Locations of areas sampled in the US Caribbean.

3.4.1 Methods

3.4.1.1 *Study Areas*

The FGBNMS study area has been previously described in Chapter 2.

Puerto Rico

The La Parguera study area is located along the southwest corner of the island within the La Parguera National Wildlife Reserve. The broad shelf area contains a variety of habitat types including coral reefs, seagrass and sand patches, as well as an extensive system of mangroves along the shoreline and on offshore islands (Kendall et al., 2001).

Mangroves in Puerto Rico. (CCMA)

St. John, USVI

The St. John study area encompasses the Virgin Islands Coral Reef National Monument (VICRNM) and Virgin Islands National Park (VINP) managed by the U.S. National Park Service (NPS), as well as territorial waters. It includes the same habitat types as are found in Puerto Rico. The VICRNM was designated a “no-take” area (with limited exceptions for certain species of jacks and baitfish) in 2001; however, these regulations were not enforced until recently (see Monaco et al., 2007 for baseline assessment). The VINP permits resource harvest by artisanal fishers as allowed in its enabling legislation as well as hook and line fishing.

Fish assemblage in St. John, USVI. (CCMA)

St. Croix, USVI

In St. Croix, the study area is located on the northeastern shelf of the island and encompasses portions of the Buck Island Reef National Monument (BIRNM) also managed by the NPS and the East End Marine Park (EEMP) managed by the USVI territory. This area includes a lagoon environment as well as a shallow shelf community with coral reefs, seagrass and sand (Kendall et al., 2001). Mangroves are very limited within close proximity to the study area. Portions of BIRNM have been designated “no-take” since the 1960s, however the majority of the current boundaries including all the deeper waters were only recently designated no-take areas in 2001. These regulations, like VICRNM, were not enforced until recently.

Reef in St. Croix, USVI. (CCMA)

3.4.1.2 *Survey Data*

Section 3.2.1 describes sampling methods and Appendix B further details specifics of data collection. See Chapter 2 for information on site selection. While identical methods were utilized in the U.S. Caribbean, an additional quadrat was surveyed per station. Additionally, *Mo. annularis*, *Mo. faveolata*, and *Mo. franksi* were identified as *Mo. annularis* complex. For the purposes of comparison, data from FGBNMS were combined into this same grouping. Only data collected from hardbottom habitat types were used. Additionally, Caribbean surveys were further subset to include only those below 18 m (60) ft to match conditions at FGBNMS. As such, total sites for comparison from each location were: FGBNMS (n=73), La Parguera (n=61), St. Croix (n=66) and St. John (n=38). It must be noted that dives conducted in La Parguera and St. Croix typically do not exceed 27 m (90 ft) and so differences in depth profiles may contribute to observed differences at these locations. Other factors differing between study locations that may impact observed differences between communities include: oceanography, local geology, and availability and configuration of habitat types.

3.4.1.3 *Data Analysis*

Abiotic and biotic percent cover data were compared among the four study regions. Non-parametric Kruskall-Wallis tests were used to examine potential differences between locations and, where differences were statistically significant, Nemenyi tests (Zar, 1999) were used to compare pairwise differences.

3.4.2 Results and Discussion

3.4.2.1 *Abiotic Cover*

Abiotic substrate at all sites consisted of mixed proportions of hardbottom, rubble and sand, with considerable variability between locations (Figure 3.14). Hard-bottom at FGBNMS comprised nearly 88% of the total abiotic substrate and was significantly greater ($p<0.0001$) than St. Croix (78%), Puerto Rico (50%) and St. John (40%). Rubble was the smallest abiotic component at most locations but was a significant component at St. John (21%) where it was significantly greater ($p<0.0001$) than all other locations. Rubble was proportionally similar at St. Croix, Puerto Rico and FGBNMS ranging from 8-10%. Approximately 40% of substrate at sites in Puerto Rico and St. John was sand, which were significantly greater ($p<0.0001$) than St. Croix (13%) and FGBNMS (3%).

3.4.2.2 *Biotic Cover*

Five classes of biotic cover (coral, macroalgal, turf algal, CCA and sponge) were compared among locations. Like the abiotic cover, considerable variability was observed among the classes between locations (Figure 3.15). The most notable difference was observed in coral cover. As previously mentioned, FGBNMS coral cover accounted for 48% of the benthos and was 8-23 times greater than that observed at Caribbean locations ($p<0.0001$). Puerto Rico exhibited the second highest coral cover (7%) followed by St. John (5%) and St. Croix (2%).

Across all regions, *Mo. annularis* complex, *Mo. cavernosa* and *P. astreoides* were the dominant coral types (Table 3.9). The brain corals, *D. strigosa* and *C. natans*, which were also comparatively dominant at the FGBNMS were less prevalent within the U.S. Caribbean locations and *Ma. mirabilis* which was found to be the dominant coral in the EBL strata was not observed.

Macroalgal cover in the SCC (13%) was comparable to that observed at St. Croix (14%) and Puerto Rico (15%). Macroalgal cover at St. John (25%) was significantly greater ($p=0.0025$) than all three locations.

Turf algal cover ranged from 12% in FGBNMS to 45% in St. Croix. In both Puerto Rico and St. Croix turf algal cover was greater than macroalgal cover. Turf algae was significantly greater at St. Croix ($p<0.0001$) with values 2-3 times greater than the other locations and values in Puerto Rico were significantly greater than St. John and FGBNMS.

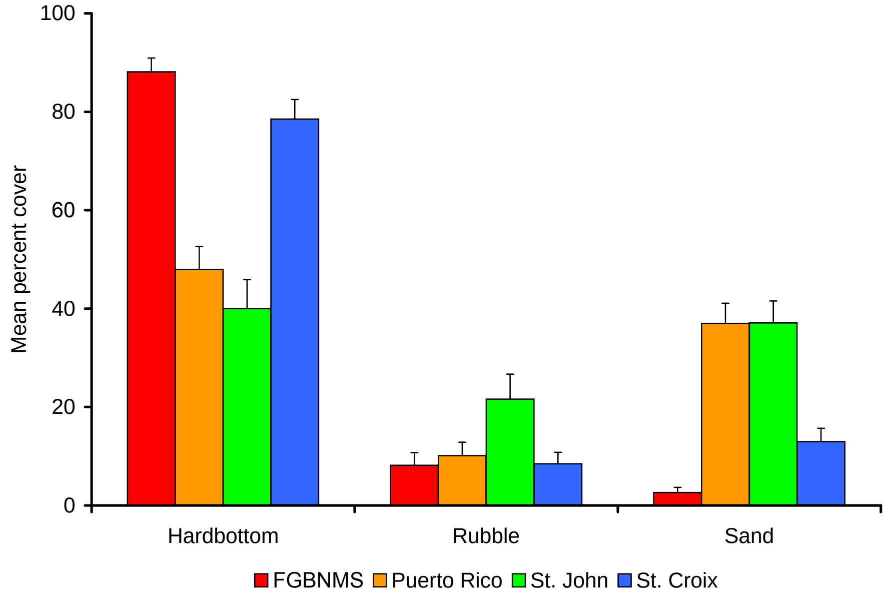


Figure 3.14. Estimated mean percent cover (± SE) of abiotic habitat groups among Caribbean (2003-2006; >18 m in depth) and FGBNMS (2006) sampling locations.

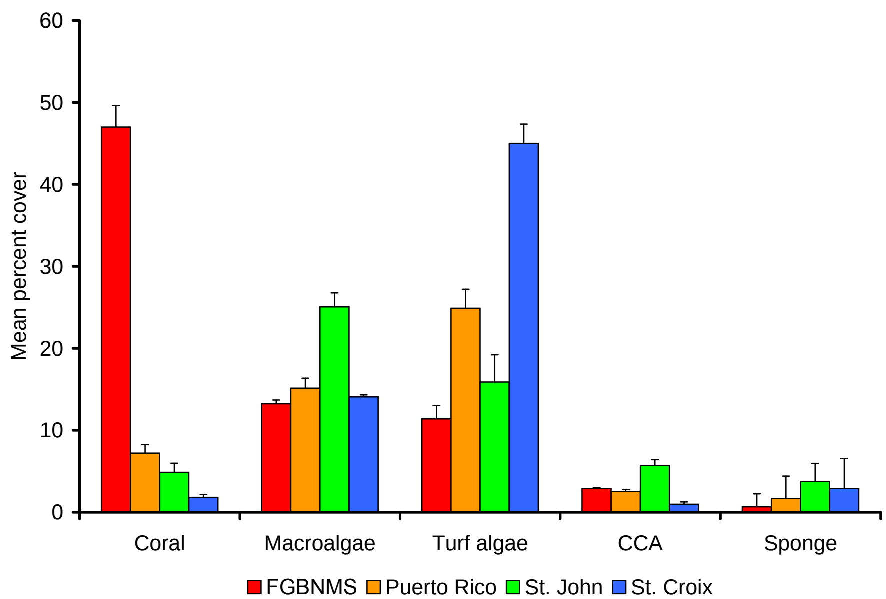


Figure 3.15. Estimated mean percent cover (± SE) of species groups at CCMA Caribbean (2003-2006; >18 m in depth) and FGBNMS (2006) sampling locations.

CCA was less than 10% at all locations. Cover at FGBNMS and St. John was significantly greater ($p<0.0001$) than in St. Croix and Puerto Rico.

Sponge cover was also a small component of the benthos at all sites ranging from 3.7% at St. John to 0.7% at FGBNMS. Sponge cover was comparable at St. John and St. Croix, but was significantly greater than Puerto Rico and FGBNMS ($p<0.0001$). Prior benthic investigations at FGBNMS have noted limited sponge abundance within the sanctuary.

Table 3.9. Mean percent coral cover (± SE) for individual species among Caribbean(2003-2006; >18 m in depth) and FGBNMS (2006) locations. FGBNMS values are corrected for sample area for comparison.

Coral species	**FGBNMS (± SE)**	**Puerto Rico (± SE)**	**St. John (± SE)**	**St. Croix (± SE)**
Montastraea annularis complex	36.34 (2.80)	3.84 (0.76)	2.72 (0.81)	0.39 (0.13)
Diploria strigosa	4.74 (0.84)	0.16 (0.06)	0.03 (0.02)	0.14 (0.03)
Montastraea cavernosa	3.59 (0.73)	0.73 (0.13)	0.48 (0.13)	0.22 (0.08)
Porites astreoides	2.89 (0.38)	0.65 (0.13)	0.35 (0.12)	0.19 (0.03)
Colpophyllia natans	2.68 (0.68)	0.10 (0.04)	0.23 (0.11)	0.02 (0.02)
Madracis mirabilis	2.06 (0.87)			
Millepora alcicornis	0.88 (0.29)			
Stephanocoenia intercepta	0.71 (0.20)	0.06 (0.02)	0.03 (0.01)	0.03 (0.01)
Madracis decactis	0.30 (0.09)	<0.01 (<0.01)	0.04 (0.03)	0.01 (0.01)
Siderastrea siderea	0.20 (0.12)	0.19 (0.06)	0.49 (0.18)	0.13 (0.04)
Millepora spp	0.16 (0.08)			
Mussa angulosa	0.12 (0.03)			<0.01 (<0.01)
Agaricia agaricites	0.12 (0.02)	0.03 (0.03)	0.15 (0.05)	0.04 (0.02)
Diploria spp	0.10 (0.09)			0.03 (0.01)
Acropora cervicornis	0.05 (0.05)	0.05 (0.05)		
Agaricia spp	0.04 (0.01)	0.64 (0.13)	0.23 (0.11)	0.10 (0.04)
Madracis spp	0.04 (0.04)	0.02 (0.01)		0.01 (0.01)
Agaricia fragilis	0.03 (0.01)			
Scolymia cubensis	0.01 (<0.01)			
Agaricia grahamae	<0.01 (<0.01)		0.01 (0.01)	0.01 (0.01)
Agaricia lamarcki	<0.01 (<0.01)		<0.01 (<0.01)	
Porites porites	<0.01 (<0.01)	0.20 (0.09)	0.20 (0.09)	0.06 (0.02)
Porites spp	<0.01 (<0.01)			0.02 (0.01)
Scolymia spp	<0.01 (<0.01)	0.01 (0.01)	<0.01 (<0.01)	0.01 (<0.01)
Dichocoenia stokesii		0.03 (0.01)	0.01 (<0.01)	0.02 (0.01)
Diploria clivosa		0.08 (0.04)	0.01 (0.01)	0.03 (0.02)
Diploria labyrinthiformis		0.06 (0.02)	0.02 (0.02)	0.02 (0.01)
Eusmilia fastigiata		<0.01 (<0.01)	0.01 (0.01)	<0.01 (<0.01)
Favia fragum		0.01 (0.01)		<0.01 (<0.01)
Helioceris cucullata		0.02 (0.02)	0.01 (0.01)	
Isophyllastrea rigida				<0.01 (<0.01)
Isophyllia sinuosa		<0.01 (<0.01)		0.01 (<0.01)
Manicina areolata		0.01 (0.01)	0.01 (0.01)	0.02 (0.01)
Meandrina meandrites		0.13 (0.03)	0.07 (0.03)	0.11 (0.02)
Mycetophyllia danaana			<0.01 (<0.01)	
Mycetophyllia ferox				0.01 (0.01)
Mycetophyllia lamarckiana		0.01 (0.01)	<0.01 (<0.01)	
Mycetophyllia reesi		<0.01 (<0.01)		0.15 (0.15)
Mycetophyllia spp		<0.01 (<0.01)	0.01 (<0.01)	<0.01 (<0.01)
Oculina diffusa			0.01 (0.01)	
Siderastrea radians		0.09 (0.03)	0.03 (0.02)	0.04 (0.01)
Siderastrea spp		0.07 (0.03)	0.01 (0.01)	0.01 (0.01)
Solenastrea spp		<0.01 (<0.01)		<0.01 (<0.01)
Tubastraea coccinea		0.01 (0.01)		

3.5 SUMMARY AND RECOMMENDATIONS

- Estimates of coral cover were high for the FGBNMS coral caps (48%) when compared with algae (13%) or sponges (1%). This value is comparable to historical values reported for live coral cover at the banks and is between 6 and 11 times higher than values estimated for the U.S. Caribbean locations.
- Coral cover tended to be higher on the high relief habitats and lower on the low relief areas while algae exhibited the opposite trend.
- Of the dominant taxa, *Mo. franksi* and *Mo. faveolata* were more prevalent in high relief habitats; *D. strigosa*, *Mo. cavernosa*, *P. astreoides* and *C. natans* were distributed throughout the banks; and *Ma. mirabilis* dominated low relief habitat.
- While coral coverage was estimated to be high, 18% was estimated to be affected by coral bleaching. The high values reported for coral bleaching suggest that the sanctuary may be more susceptible to environmental impacts than previously known.
- Reports of marine debris from this baseline assessment included anchors, fishing line and rope. While many of the items encountered were overgrown by corals and limited in their ecological impact, continued marine debris monitoring is needed to identify areas more prone to accumulation and confirm the apparent low frequency of debris introduction.
- Further monitoring and characterization of the benthic community will enable linkages to be made with the fish community (e.g., the role of *Ma. mirabilis*), a better understanding of impact and recovery from bleaching events or coral disease, and an evaluation of marine debris impacts.
- A better understanding of the deep water habitats surrounding the banks will provide the sanctuary with a better understanding of ecological linkages between these areas and the SCC.

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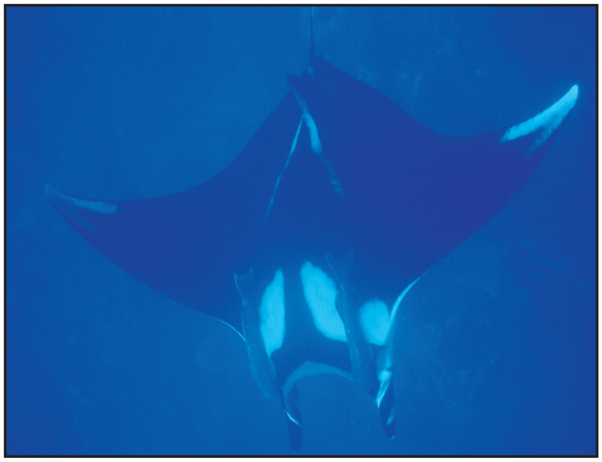
Chapter 4: Fish Communities

4.1 INTRODUCTION

The fish community on the FGBNMS coral caps is very similar to that of the Caribbean and represents the northernmost coral reef fish community in the US. Unlike many Caribbean reefs, the sanctuary is comparatively isolated from land based impacts as well as other coral reef ecosystems. This isolation has resulted in the fish fauna being recognized as more characteristic of historical coral reef communities prior to declines caused by deteriorating habitats, overfishing and other factors (Caldow et al., 2008). Anthropogenic and environmental stresses do occur; however, a better understanding of the sanctuary's natural resources is needed for the sanctuary to accomplish its goals and mission (NOAA, 1991).

Fish population data necessary for guiding management decisions on the banks is relatively sparse. While monitoring efforts at FGBNMS began in the 1970s (Table 4.1) early work focused primarily on monitoring the benthos with video transects and photostations documenting changes in coral, algae and sponge communities over time. Until relatively recently, little was done to monitor the associated fish community. Initial efforts centered on providing species lists and examining habitat associations with depth. Researchers utilized a variety of techniques including scuba diving, hook and line, trawls, and submersibles to determine assemblage composition on the banks. In 1996 the Reef Environmental Education Foundation (REEF) began surveys of the sanctuary and utilized a combination of REEF personnel, volunteers, and sanctuary staff to visually census reef fish populations via roving diver surveys. These surveys have been invaluable in terms of species list development and understanding the ranges of these species.


Manta ray (*Manta birostris*) (CCMA)

Monitoring of the fish communities began with video transects conducted in the late 1980s and early 1990s and was replicated in 1996 and 1997; however, this work was limited to large bodied fishes identifiable in the footage. A more quantitative approach was taken by Pattengill-Semmens et al. (1997) who utilized a stationary point-count technique to quantify community metrics such as species abundance and trophic structure at selected locations. This work was followed up in 2002 by PBS&J who employed the same technique in their current monitoring efforts. Both PBS&J surveys and those conducted by Pattengill-Semmens et al. (1997) focused on a relatively small portion of the East Bank (EB) and West Bank's (WB) coral cap environments. Many species which live on the coral caps are likely to be underrepresented in these spatially-constrained surveys. Therefore, their scope of inference is limited to these portions of the banks making them difficult to utilize in developing population estimates at the scale of the sanctuary; however, both data sets provide important starting points for characterizing the fish community.

The current effort complements these prior studies with the development of a spatial framework and sampling design that can be used to cover the entirety of the shallow coral caps (SCC; <33.5 m). The data, analysis and results presented here provide a spatially-explicit characterization and baseline of fish community structure for this extent that will support FGBNMS management strategies. Additional analyses were performed comparing the resident fish communities of on the SCC with those in other US coral reef ecosystems in Puerto Rico as well as St. Croix and St. John in the U.S. Virgin Islands (USVI) to explore the community structure of relatively undisturbed locations versus those more heavily impacted.

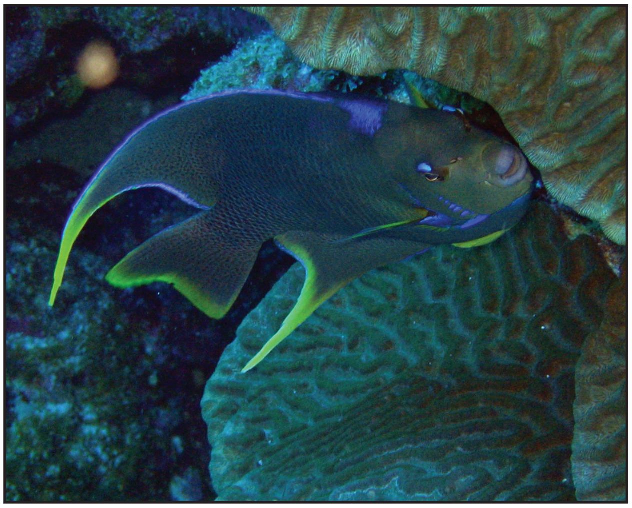

Blue angelfish (*Holacanthus bermudensis*) (CCMA)

Table 4.1. Prior research activities at Flower Garden Banks National Marine Sanctuary (FGBNMS).

Citation	Technique	Time Period	Location	Sampling Frequency	Data
Bright and Cashman (1974)	hook and line; traps; divers; dipnets, trawls; towed camera; rotenone; submersible	1970 - 1972	WB	17 cruises	size data and area recorded from video transects
Dennis and Bright (1988)	submersible transects from crest to depth; diving	1970 - 1972	EB and WB	unknown	abundance (this is standardized by time at a given depth)
Dennis (1985)	submersible transects from crest to depth; scuba roving diver; accounts of records of reef fish taken by hook and line, ichthyocides, spear, and trawl	70s and 80s; Scuba (1978 – 1981)	EB and WB	unknown	size, abundance (this is standardized by time at a given depth)
Boland et al. (1983)	Hook and line; traps; divers; trawls; towed camera	1980-1982	EB and WB	8 cruises	size data and area recorded from video transects
Gittings et al. (1992)	video transects	1989-1991	EB and WB (100 x 100 m plots)	6 cruises (2 transects/bank/ cruise)	large bodied (approximately 15 cm in length) fish were enumerated
Pattengill (1998)	modified stationary point count (6.5m radius); roving diver technique	1994-1997	within 100 m of mooring buoy 2 EB and buoy 5 WB	6 cruises (approximately 24 surveys/bank/trip)	abundance on all species observed for stationary diver; log scale abundance for roving diver; no size
Dokken et al. (1999)	video transects	1996-1997	EB and WB (100 x 100 m plots)	2 cruises (two transects/bank/ cruise)	large bodied (approx 15 cm in length) fish were enumerated
Pattengill-Semmens and Semmens (1998)	roving diver	1996 - present	EB and WB	biannually to annually	log scale abundance
Pattengill-Semmens et al. (2000)	belt transects and roving diver	1999	EB and WB	1 cruise (12 transects/bank)	transect: selected taxa enumerated; size recorded in 5–10 cm bins; roving diver:all species encountered identified and assigned to log scale abundance estimate
Precht et al. (2006)	point counts	2003 - present	EB and WB (100 x 100 m plots)	annually (16 counts/bank)	size data (mean, max, min)

4.2 METHODS

4.2.1 Survey

Visual fish surveys and accompanying benthic habitat information were collected along 100 m^2 transects at randomly selected sites as detailed in Chapter 2. All fish were identified to species or the lowest possible taxon and enumerated. All fish were sized using fork length (FL) in 5 cm categories up to 35 cm and actual values were used for fish greater than 35 cm (refer to Appendix A for detailed methods).

4.2.2 Data analysis

Domain-wide Population Estimates

Domain-wide estimates were computed employing methods described by Cochran (1977) for a stratified sampling design using 2006 data, strata and corresponding sampling weights. Measurements collected in 2007 were not included because the incomplete field mission imposed spatial bias (see Chapter 2 for details regarding sampling design).

Summary statistics including: total species occurrence, percent occurrence, total abundance, mean abundance (± standard error [SE]), total biomass and mean biomass (± SE), were generated for all species observed for each bank. Biomass was calculated using published length-weight relationships using the formula,

$$W = \alpha L^{\beta}$$

where L is length is in centimeters and weight is in grams. The midpoint of each size class was used for L values, or actual length was used for fish >35 cm (for fish at 0-5 cm, 3 cm was used as we don't typically observed fish <1 cm). Values for the α and β coefficients were obtained from FishBase (Froese and Pauly, 2007). Biomass for species with no published length-weight relationships was calculated using terms for the closest congener based on morphology.

Community metrics were compared with historical surveys that used different sampling methods to provide insights into the benefits of each method and to examine patterns between the surveys.

Strata Comparisons

For comparative analyses, the 73 sample sites surveyed in 2006 were classified into three different strata: East Bank High relief (EBH), East Bank Low relief (EBL) and West Bank High relief (WBH) depending on benthic relief and geographic location. There were no low relief habitat sites surveyed on West Bank, and therefore no West Bank Low relief stratum was used in the analyses or presented in results. Benthic relief was taken from a benthic habitat map developed for this study (see Appendix C).

These strata are different from those actually used for the 2006 sampling design. It must be noted therefore, that estimates of means and variances are not technically valid; however since a random proportional-to-area design was used, the difference between the valid and computed estimates are assumed to be negligible. As with the domain-wide calculations, data from the 2007 mission were not included because the incomplete field mission imposed spatial bias.

Differences in fish communities among strata were evaluated by comparing overall abundance, biomass, species richness (number of species), Shannon's index of diversity (H') and Peilou's evenness (J') index for strata. Shannon's index of diversity is defined as,

$$H' = - S_i \, p_i \, (\ln p_i)$$

where H' is a weighted combination of species richness and the extent to which the total abundance is spread equally among the observed species and p_i is the proportion of the total count arising from the i th species.

Pielou's evenness is represented as:

$$J' = H' / \ln S$$

where S is the total number of species.

Data were log transformed to meet normality and homogeneity of variance assumptions, with the exception of species richness which exhibited a normal distribution. Analysis of Variance (ANOVA) was used to compare groups and if appropriate multiple means comparisons were completed using Tukey-Kramer tests. Species diversity, evenness, and density data from FGBNMS and the Caribbean locations did not meet assumptions for homogeneity of variances using Bartlett's test; therefore, non-parameteric Kruskal-Wallis tests were performed on the raw (non-transformed) data to explore potential differences. Pairwise comparisons were performed using the Nemenyi test (Zar, 1999). Biomass data among the locations were log transformed to meet assumptions of normality and an ANOVA test was performed. Pairwise comparisons were evaluated using Tukey-Kramer. All analyses were performed using JMP© statistical software (SAS Institute Inc., 2000).

Additionally, fish species were grouped by trophic guild and abundance and biomass were compared among bank and relief types. These groups include: herbivores, piscivores, invertivores and zooplanktivores (Randall, 1967).

Comparison with REEF and FGBNMS Long-term Monitoring (FGBNMS LTM) Surveys

Prior survey data, using a variety of survey techniques, were compared. REEF surveys from 1995-2005 were examined. Fish data were collected using the Roving Diver Technique (RDT) in close proximity to mooring buoys on both banks (Figure 4.1). The RDT is a non-point survey method where divers move freely about a site. Only data collected by divers classified in the REEF database as being experts were used. Mean frequency of occurrence estimates were calculated and compared. Where appropriate, point count data collected during 1994-1995 were used to examine fish trophic structure (Pattengill-Semmens et al., 1997). This modified Bohnsack and Bannerot (1986) stationary visual census technique samples fish in a cylinder with a radius of 6.5 m and height of 4 m. Fishes were identified and enumerated within a 5 minute duration. Fish survey data collected by the Minerals Management Service (MMS) as part of the FGBNMS LTM surveys was also available for comparison (Figure 4.1). These data were collected during survey missions conducted in October 2002 and August 2003. The surveys also employed a modified Bohnsack and Bannerot (1986) stationary visual census technique with a 7.5 m cylinder radius and height (Precht et al., 2006). Survey time was between 10 and 15 minute duration. Only total abundance values were available, thus presence/absence data for fish species were used for comparison.

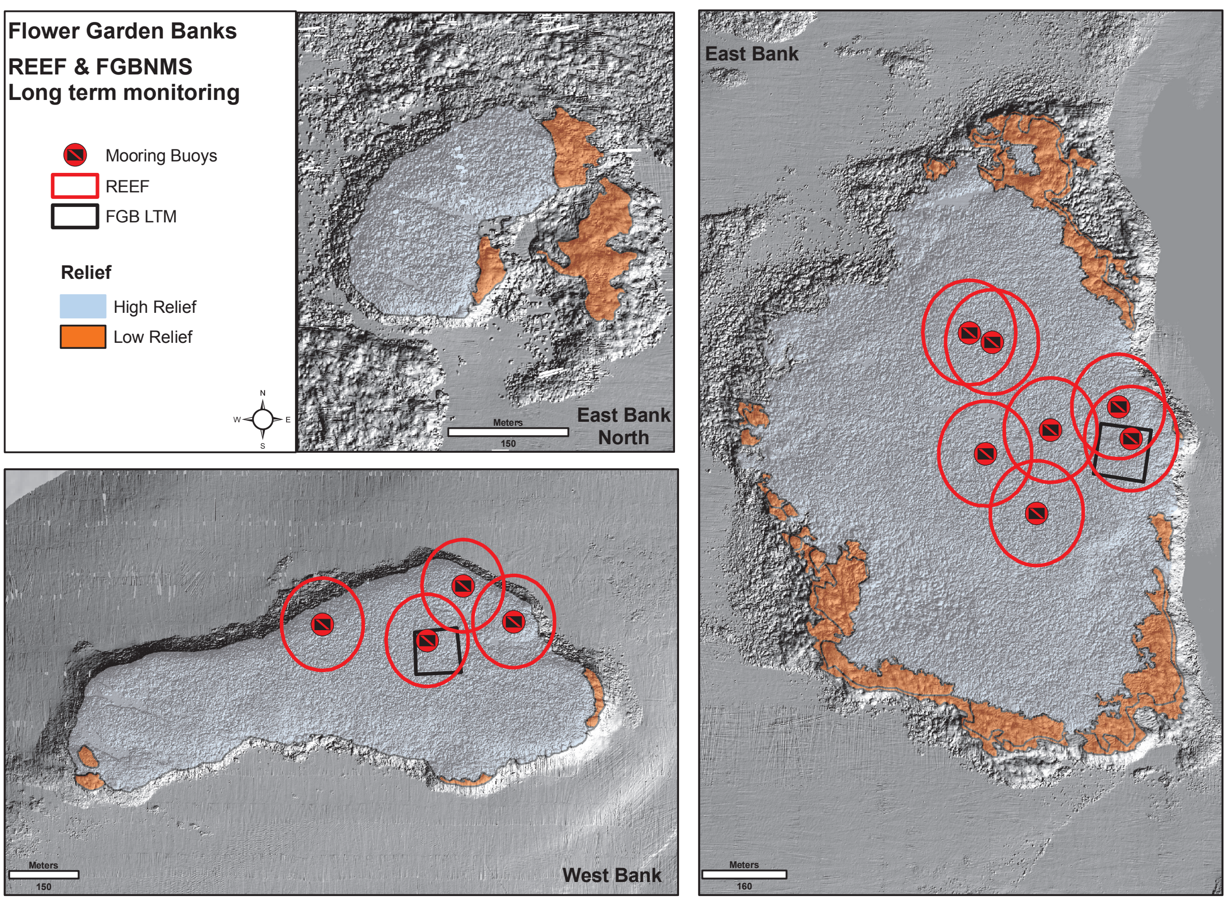


Figure 4.1. Locations of REEF and FGBNMS long-term monitoring stations.

Correlative Analyses

Correlations between community metrics, trophic groups and taxonomic groups with benthic habitat parameters such as percent coral cover, macroalgae cover and depth were examined using non-parameteric Spearman's analysis on percent cover. Data collected during both the 2006 and 2007 field missions were utilized for these analyses. Results are reported as Spearman's ρ (Rho).

Interpolations

Fish community metrics were interpolated and mapped using inverse distance weighting (IDW) to guide interpretation of spatial patterns. These interpolations were created without separating data by strata; therefore, where observed patterns cross strata they must be interpreted with consideration given to strata differences (e.g., a site containing high biomass in the EBH strata near the border of EBL may result in the adjacent area in EBL appearing high as well which may not reflect reality). Interpolated surfaces were generated for each bank with ArcGIS spatial analyst (ESRI, 2006) using all data from 2006 and 2007. By combining the years we increase our coverage of sample points on EB; however, this assumes no differences between years. EB was further segregated into two portions: the main portion of the bank and a smaller mound that is approximately 470 m northeast of the main portion. Only three surveys were conducted on this smaller portion of EB, thus an interpolated surface for this region was not generated.

Select families were separately analyzed and mapped based on ecological or economical importance. Fishes from the families: Serranidae (groupers), Lutjanidae (snappers), Scaridae (parrotfishes), Carangidae (jacks) and Pomacentridae (damselfish) were examined to quantify spatial patterns of abundance, biomass and evaluate ontogenetic preferences or shifts in habitat use.

Cluster Analysis

A hierarchical cluster analysis was used to identify fish species assemblages. A second hierarchical cluster analysis was used to identify spatial patterns of the sample site groups within the sanctuary. A nodal analysis, the intersection of the species clusters and the site clusters, was used to identify the species assemblages defining the site groupings.

Prior to the cluster analysis of the data, rare species (species that occurred in four or less samples) were omitted from the analysis, resulting in 74 species for all sample sites (n=105). The remaining species density data were transformed with the natural log transformation [log (density + 1)]. A matrix of Pearson product moment correlation coefficients were calculated for transformed species data. This matrix was converted to a matrix of distances by subtracting each Pearson coefficient from one. This matrix was processed by the SAS/STAT© software (SAS Institute Inc., 2006). Scree plots of cluster distances were examined to determine where breaks in the similarity level among the species clusters occurred. A similar process was used to identify site groupings. Nodal analysis was used to relate fish assemblages with site groups. Principal component analysis (PCA) was also used to identify assemblages and to compare with the hierarchical technique. Assemblages are further investigated to identify the major species components and possible relationships with coral assemblages (described in Chapter 3) and other habitat parameters.

Comparison with U.S. Caribbean

Spatial patterns of abundance, biomass, and species richness were examined across three Caribbean reef ecosystems that have been monitored using the same methods since 2001 (Christensen et al., 2003; Pittman et al., 2007; Pittman et al., 2008). Transect data from La Parguera in Puerto Rico, and St. John and northeastern St. Croix, USVI were subset using only data from sites located in waters deeper than 18 m (60 ft), to match the bathymetric conditions observed within the SCC. Transect data from the Caribbean locations were collected using the same methods.

4.3 RESULTS AND DISCUSSION

During 2006, 39 sites were surveyed on EBH, 10 on EBL and 24 on WBH. Only 32 stations (nine on EBL and 23 on EBH) were surveyed during 2007 as the mission was interrupted and canceled due to Hurricane Humberto. During this period, 89 of the 105 surveys were conducted on high relief while 19 surveys were conducted on low relief habitats. Overall, 37,517 individuals representing 117 species and 37 families were observed and total biomass exceeded 21,000 kg. On EB, a total of 30,109 individuals from 103 species and 33 families were observed with biomass totaling 17,188 kg (Table 4.2). On WB, a total of 7,408 individuals represented by 85 species and 30 families were observed and biomass amounted to 11,830 kg (Table 4.2). High relief habitats (both banks combined) yielded 30,661 individuals comprised by 114 species from 39 families. Total abundance on low

Table 4.2. Mean (±SE) density and biomass for fishes observed at East and West Flower Garden Banks in 2006.

Species	Common name	Trophic Group	Density			Biomass		
			EBH (± SE)	EBL (± SE)	WBH (± SE)	EBH (± SE)	EBL (± SE)	WBH (± SE)
Abudefduf saxatilis	sargeant major	I			0.25 (0.14)			0.01 (0.01)
Acanthemblemaria spp.	blenny	I	0.03 (0.03)			<0.01 (<0.01)		
Acanthostracion polygonia	honeycomb cowfish	I	0.05 (0.04)			0.03 (0.02)		
Acanthurus bahianus	ocean surgeonfish	H	0.64 (0.18)	0.40 (0.31)	0.29 (0.15)	0.05 (0.02)	0.01 (0.01)	0.01 (0.01)
Acanthurus chirurgus	doctorfish	H	0.46 (0.22)	0.70 (0.40)	0.92 (0.38)	0.02 (0.01)	0.06 (0.05)	0.05 (0.03)
Acanthurus coeruleus	blue tang	H	4.79 (1.28)	0.70 (0.40)	3.38 (0.72)	0.40 (0.21)	0.09 (0.09)	0.38 (0.10)
Amblycirrhitus pinos	redspotted hawkfish	Z	0.05 (0.04)	0.10 (0.10)	0.08 (0.06)	<0.01 (<0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Balistes vetula	queen triggerfish	I	0.03 (0.03)	0.40 (0.40)	0.08 (0.06)	0.01 (0.01)	0.60 (0.60)	0.16 (0.12)
Bodianus pulchellus	spotfin hogfish	I	0.51 (0.15)	0.80 (0.51)	0.79 (0.29)	0.01 (<0.01)	0.02 (0.02)	0.02 (0.01)
Bodianus rufus	Spanish hogfish	I	4.59 (0.53)	6.40 (1.69)	8.50 (1.14)	0.06 (0.01)	0.09 (0.05)	0.23 (0.09)
Calamus calamus	saucereye porgy	I	0.05 (0.04)	0.10 (0.10)		0.02 (0.01)	0.13 (0.13)	
Calamus nodosus	knobbed porgy	I	0.05 (0.04)	0.20 (0.13)		0.04 (0.03)	0.08 (0.06)	
Calamus spp.	porgy species	I	0.03 (0.03)			0.02 (0.02)		
Cantherhines macrocerus	whitespotted filefish	I			0.08 (0.06)			0.09 (0.08)
Cantherhines pullus	orangespotted filefish	H	0.08 (0.04)		0.08 (0.08)	<0.01 (<0.01)		<0.01 (<0.01)
Canthidermis sufflamen	ocean triggerfish	I	0.21 (0.10)	0.20 (0.20)	0.04 (0.04)	0.20 (0.10)	0.15 (0.15)	0.01 (0.01)
Canthigaster jamestyleri	goldface toby	I	0.03 (0.03)			<0.01 (<0.01)		
Canthigaster rostrata	sharpnose puffer	I	5.85 (0.56)	2.70 (0.52)	5.75 (0.61)	0.05 (0.03)	0.01 (<0.01)	0.02 (<0.01)
Carangoides ruber	bar jack	P	13.21 (10.40)	1.00 (0.89)	3.92 (2.95)	0.25 (0.18)	0.03 (0.03)	0.07 (0.03)
Caranx crysos	blue runner	P			2.29 (2.29)			1.40 (1.40)
Caranx latus	horse-eye jack	P	0.05 (0.04)	0.10 (0.10)	0.58 (0.50)	0.05 (0.04)	0.23 (0.23)	1.04 (0.82)
Caranx lugubris	black jack	P	0.21 (0.07)	0.80 (0.42)	0.58 (0.18)	0.26 (0.10)	0.53 (0.24)	0.31 (0.10)
Centropyge aurantonotus	flameback angelfish	H	0.05 (0.04)			<0.01 (<0.01)		
Cephalopholis cruentata	graysby	P	0.77 (0.18)	0.40 (0.22)	1.17 (0.27)	0.07 (0.02)	0.06 (0.06)	0.15 (0.04)
Cephalopholis fulvus	coney	P	0.03 (0.03)			<0.01 (<0.01)		
Chaetodon ocellatus	spotfin butterflyfish	I	0.56 (0.15)		0.42 (0.21)	0.03 (0.01)		0.03 (0.01)
Chaetodon sedentarius	reef butterflyfish	I	2.33 (0.29)	1.70 (0.37)	2.46 (0.32)	0.20 (0.14)	0.04 (0.02)	0.08 (0.02)
Chaetodon striatus	banded butterflyfish	I	0.03 (0.03)			<0.01 (<0.01)		
Chromis cyanea	blue chromis	Z	2.31 (0.46)	3.10 (1.12)	6.04 (1.14)	0.02 (<0.01)	0.02 (0.01)	0.04 (0.01)
Chromis insolata	sunshinefish	Z	7.62 (2.15)	36.80 (14.07)	13.67 (4.38)	0.01 (<0.01)	0.07 (0.04)	0.03 (0.01)
Chromis multilineata	brown chromis	Z	43.51 (8.16)	19.00 (11.81)	39.08 (12.32)	0.29 (0.06)	0.15 (0.09)	0.62 (0.37)
Chromis scotti	purple reeffish	Z	4.46 (1.16)	27.40 (23.44)	5.67 (1.45)	0.02 (0.01)	0.26 (0.26)	0.02 (<0.01)
Clepticus parrae	creole wrasse	Z	31.62 (9.81)	0.10 (0.10)	32.13 (18.83)	1.21 (0.48)	<0.01 (<0.01)	0.82 (0.47)
Coryphopterus eidolon	pallid goby	I			0.08 (0.06)			<0.01 (<0.01)
Coryphopterus glaucofraenum	bridled goby	I	0.05 (0.04)		0.04 (0.04)	<0.01 (<0.01)		<0.01 (<0.01)
Coryphopterus personatus/ hyalinus	masked goby	I	0.49 (0.28)		0.25 (0.25)	<0.01 (<0.01)		<0.01 (<0.01)
Dasyatis americana	southern stingray	I			0.04 (0.04)			<0.01 (<0.01)
Dermatolepis inermis	marbled grouper	P	0.21 (0.11)	0.20 (0.13)	0.13 (0.07)	0.07 (0.05)	0.09 (0.06)	0.04 (0.02)
Diodon holocanthus	balloonfish	I	0.03 (0.03)		0.08 (0.06)	0.01 (0.01)		0.09 (0.08)
Echeneis naucrates	sharksucker	Z			0.04 (0.04)			<0.01 (<0.01)
Elacatinus chancei	shortstripe goby	I			0.04 (0.04)			<0.01 (<0.01)
Elacatinus oceanops	neon goby	I	0.38 (0.15)	0.50 (0.34)	0.33 (0.16)	<0.01 (<0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Emmelichthyops atlanticus	bonnetmouth	P	0.51 (0.51)			<0.01 (<0.01)		
Epinephelus adscensionis	rock hind	I	0.13 (0.05)	0.10 (0.10)		0.06 (0.03)	0.01 (0.01)	
Epinephelus guttatus	red hind	P	0.10 (0.06)		0.25 (0.17)	0.07 (0.07)		0.11 (0.08)
Ginglymostoma cirratum	nurse shark	P			0.04 (0.04)			0.72 (0.72)
Gnatholepis thompsoni	goldspot goby	H	1.85 (0.46)	2.40 (0.75)	0.75 (0.36)	0.01 (0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Gymnothorax miliaris	goldentail moray	P	0.03 (0.03)		0.08 (0.06)	<0.01 (<0.01)		<0.01 (<0.01)
Gymnothorax moringa	spotted moray	P	0.05 (0.04)		0.04 (0.04)	0.04 (0.04)		0.03 (0.03)
Haemulon parra	sailors choice	I	0.03 (0.03)			0.01 (0.01)		
Halichoeres bivittatus	slippery dick	I	0.18 (0.13)			<0.01 (<0.01)		
Halichoeres burekae	mardi gras wrasse	I	0.21 (0.18)		0.38 (0.26)	<0.01 (<0.01)		<0.01 (<0.01)
Halichoeres garnoti	yellowhead wrasse	I	2.77 (0.42)	3.50 (1.38)	3.00 (0.50)	0.03 (0.01)	0.03 (0.01)	0.05 (0.01)

Table 4.2 Continued...

Species	Common name	Trophic Group	Density			Biomass		
			EBH (± SE)	EBL (± SE)	WBH (± SE)	EBH (± SE)	EBL (± SE)	WBH (± SE)
Halichoeres maculipinna	clown wrasse	I	1.77 (0.42)	0.70 (0.50)	1.67 (0.33)	0.01 (0.00)	<0.01 (<0.01)	0.01 (<0.01)
Halichoeres radiatus	puddingwife	I	0.18 (0.08)	0.50 (0.27)	0.25 (0.11)	<0.01 (<0.01)	0.06 (0.05)	0.02 (0.01)
Heteropriacanthus cruentatus	glasseye	Z			0.04 (0.04)			0.01 (0.01)
Holacanthus ciliaris	queen angelfish	I	0.23 (0.10)	0.10 (0.10)	0.17 (0.10)	0.08 (0.04)	0.05 (0.05)	0.06 (0.04)
Holacanthus tricolor	rock beauty	I	0.10 (0.06)	0.60 (0.27)	0.46 (0.15)	<0.01 (<0.01)	0.01 (0.01)	0.03 (0.01)
Holacanthus bermudensis	blue angelfish	I	0.08 (0.04)	0.20 (0.20)		0.04 (0.03)	0.05 (0.05)	
Holocentrus adscensionis	squirrelfish	I	0.05 (0.04)			0.01 (0.01)		
Holocentrus rufus	longspine squirrelfish	I	0.38 (0.14)		0.29 (0.13)	0.05 (0.02)		0.02 (0.01)
Kyphosus sectatrix/incisor	bermuda/yellow chub	H	1.49 (0.54)	1.00 (1.00)	4.67 (1.99)	0.76 (0.32)	0.86 (0.86)	3.42 (1.58)
Lactophrys triqueter	smooth trunkfish	I	0.44 (0.10)	0.20 (0.13)	0.67 (0.21)	0.05 (0.02)	0.02 (0.02)	0.07 (0.02)
Lutjanus analis	mutton snapper	P	0.03 (0.03)		0.04 (0.04)	0.02 (0.02)		0.38 (0.38)
Lutjanus cyanopterus	cubera snapper	P			0.04 (0.04)			0.12 (0.12)
Lutjanus griseus	gray snapper	P	1.23 (0.77)	0.70 (0.60)	1.04 (0.35)	0.74 (0.47)	0.69 (0.56)	0.69 (0.25)
Lutjanus jocu	dog snapper	P	0.46 (0.21)	0.10 (0.10)	0.29 (0.11)	0.92 (0.40)	0.32 (0.32)	1.20 (0.57)
Lutjanus spp.	snapper species	P	0.03 (0.03)			0.01 (0.01)		
Malacanthus plumieri	sand tilefish	I			0.04 (0.04)			<0.01 (<0.01)
Manta birostris	manta ray	Z			0.04 (0.04)			18.45 (18.45)
Melichthys niger	black durgon	H	0.82 (0.25)	0.40 (0.22)	1.00 (0.25)	0.52 (0.17)	0.36 (0.27)	0.32 (0.07)
Microspathodon chrysurus	yellowtail damselfish	H	0.03 (0.03)		0.29 (0.13)	<0.01 (<0.01)		0.01 (0.01)
Mulloidichthys martinicus	yellow goatfish	I			0.75 (0.59)			0.09 (0.07)
Mycteroperca bonaci	black grouper	P	0.18 (0.06)	1.20 (1.20)	0.29 (0.11)	1.34 (0.90)	0.64 (0.64)	9.61 (5.96)
Mycteroperca interstitialis	yellowmouth grouper	P	1.56 (0.38)	0.70 (0.33)	1.42 (0.42)	0.96 (0.26)	0.55 (0.42)	1.06 (0.32)
Mycteroperca phenax	scamp	P	0.03 (0.03)		0.04 (0.04)	<0.01 (<0.01)		<0.01 (<0.01)
Mycteroperca spp.	grouper species	P			0.04 (0.04)			<0.01 (<0.01)
Mycteroperca tigris	tiger grouper	P	0.79 (0.26)	0.60 (0.43)	0.42 (0.12)	1.04 (0.35)	0.58 (0.40)	0.87 (0.31)
Mycteroperca venenosa	yellowfin grouper	P	0.03 (0.03)		0.04 (0.04)	0.07 (0.07)		0.47 (0.47)
Myripristis jacobus	blackbar soldierfish	I			0.08 (0.08)			0.01 (0.01)
Neoniphon marianus	longjaw squirrelfish	I			0.04 (0.04)			<0.01 (<0.01)
Ophioblennius macclurei	redlip blenny	H	0.03 (0.03)		0.17 (0.10)	<0.01 (<0.01)		<0.01 (<0.01)
Opistognathus aurifrons	yellowhead jawfish	Z		6.40 (2.82)			0.02 (0.01)	
Paranthias furcifer	Atlantic creolefish	Z	45.64 (11.38)	62.40 (27.74)	33.29 (12.44)	6.63 (2.08)	6.22 (1.92)	2.51 (1.00)
Pomacanthus paru	French angelfish	I	0.72 (0.37)	1.40 (0.54)	0.04 (0.04)	0.40 (0.15)	1.30 (0.55)	0.05 (0.05)
Prognathodes aculeatus	longsnout butterflyfish	I	0.51 (0.11)		0.96 (0.20)	0.01 (<0.01)		0.01 (<0.01)
Pseudupeneus maculatus	spotted goatfish	I	0.15 (0.09)	0.30 (0.21)	0.13 (0.07)	0.01 (<0.01)	0.01 (0.01)	0.01 (0.01)
Remora remora	remora	Z			0.08 (0.08)			0.02 (0.02)
Sargocentron bullisi	deepwater squirrelfish	I	0.05 (0.05)			<0.01 (<0.01)		
Sargocentron vexillarium	dusky squirrelfish	I			0.04 (0.04)			<0.01 (<0.01)
Scarus iseri	striped parrotfish	H	0.23 (0.11)	0.10 (0.10)		<0.01 (<0.01)	<0.01 (<0.01)	
Scarus taeniopterus	princess parrotfish	H	2.15 (0.46)	2.80 (0.95)	0.79 (0.25)	0.09 (0.05)	0.07 (0.04)	0.04 (0.02)
Scarus vetula	queen parrotfish	H	2.03 (0.48)	0.50 (0.22)	1.92 (0.35)	0.51 (0.13)	0.01 (<0.01)	0.60 (0.17)
Serranus tigrinus	harlequin bass	I	0.03 (0.03)	0.50 (0.17)		<0.01 (<0.01)	<0.01 (<0.01)	
Sparisoma atomarium	greenblotch parrotfish	H	1.49 (0.36)	9.50 (2.99)	0.33 (0.29)	<0.01 (<0.01)	0.01 (0.00)	<0.01 (<0.01)
Sparisoma aurofrenatum	redband parrotfish	H	4.10 (0.49)	4.60 (1.70)	4.75 (0.67)	0.10 (0.04)	0.04 (0.03)	0.09 (0.02)
Sparisoma viride	stoplight parrotfish	H	3.28 (0.49)	2.20 (0.93)	1.83 (0.36)	0.61 (0.14)	1.20 (0.74)	0.61 (0.22)
Sphyraena barracuda	great barracuda	P	0.36 (0.12)		0.83 (0.27)	0.76 (0.27)		1.44 (0.60)
Stegastes adustus	dusky damselfish	H	0.33 (0.12)	0.10 (0.10)	0.17 (0.08)	<0.01 (<0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Stegastes diencaeus	longfin damselfish	H	0.05 (0.04)		0.04 (0.04)	<0.01 (<0.01)		<0.01 (<0.01)
Stegastes leucostictus	beaugregory	H	0.13 (0.07)	0.80 (0.80)	0.42 (0.17)	<0.01 (<0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Stegastes partitus	bicolor damselfish	H	5.46 (0.71)	16.60 (4.31)	5.83 (1.14)	0.03 (0.01)	0.06 (0.03)	0.05 (0.01)
Stegastes planifrons	threespot damselfish	I	17.59 (1.64)	23.50 (7.02)	16.58 (2.43)	0.13 (0.01)	0.05 (0.01)	0.15 (0.02)
Stegastes variabilis	cocoa damselfish	H	0.69 (0.19)	1.90 (0.64)	0.08 (0.06)	<0.01 (<0.01)	<0.01 (<0.01)	<0.01 (<0.01)
Synodus intermedius	sand diver	P	0.03 (0.03)			<0.01 (<0.01)		
Synodus saurus	Atlantic lizardfish	P	0.03 (0.03)			<0.01 (<0.01)		
Thalassoma bifasciatum	bluehead wrasse	I	81.85 (7.62)	88.80 (19.14)	93.00 (16.17)	0.16 (0.03)	0.16 (0.05)	0.16 (0.03)

relief habitats was lower, 6,856 individuals represented by 68 species from 25 families. Two of three species recently added to the FGBNMS species list were also recorded during the course of this study: 18 mardi gras wrasses (*Halichoeres burekae*) first described at FGBNMS in 2006 (Weaver and Rocha, 2007) and 11 sergeant majors (*Abudefduf saxatilis*). No observations of the third species, yellowtail snapper (*Ocyurus chrysurus*), were reported during 2006-2007.

Fish density was higher on East Bank low relief (EBL) habitats, although the relationship was not statistically significant (Table 4.3). Mean biomass on West Bank high relief (WBH) was more than twice that observed on either relief type on EB. The estimate in Table 4.3 includes the single observation of a large manta ray (*Manta birostris*) and if excluded biomass was not significantly different between banks. Species richness on both East Bank high relief (EBH) and WBH were significantly greater than EBL habitats (p=0.017). Species diversity, evenness and family representation was similar among all bank/habitat types.

Table 4.3. Summary statistics (mean ±SE) for fish community metrics by bank/habitat type for 2006. H' =Shannon's diversity index; J' = evenness. Asterisks (*) indicate statistical significance.

Strata	N	Density (± SE)	Biomass (kg) (± SE)	Species Richness (± SE)	H' (± SE)	J' (± SE)	# Families (± SE)
EBH	39	308 (24.88)	19.71 (3.28)	25.72 (0.55)	2.14 (0.06)	0.66 (0.02)	11.38 (0.23)
EBL	10	339.2 (73.66)	16.13 (3.19)	22.70* (1.14)	2.13 (0.07)	0.68 (0.03)	10.5 (0.58)
WBH	24	309.21 (46.23)	30.84 (9.55)	26.83 (0.88)	2.21 (0.06)	0.68 (0.02)	12.29 (0.47)

Peak areas of fish density were predominantly observed on the edges of the coral caps (including both high and low relief) while fewer peaks were observed in the shallower, high relief, central area of EB (Figure 4.2). Biomass peaks were also predominant on the edges of the coral caps, most notably WB where *M. birostris* and large black grouper (*Mycteroperca bonaci*) were observed (Figure 4.3). Localized areas of low and high species richness were evident throughout the sanctuary (Figure 4.4).

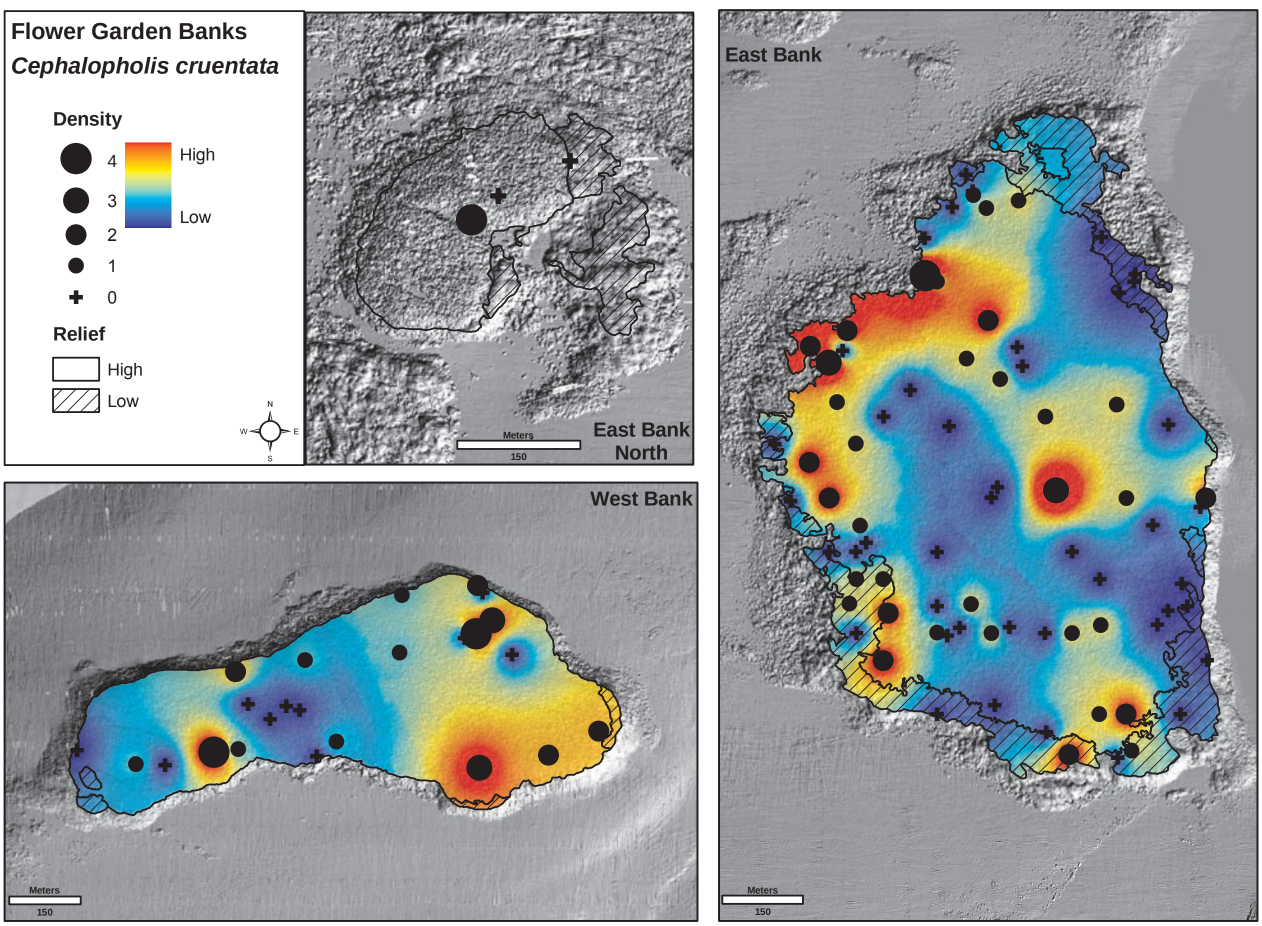


Figure 4.2. Observed (dots) and spatially interpolated density (#/100 m²) for all species observed in CCMA surveys for 2006-2007.

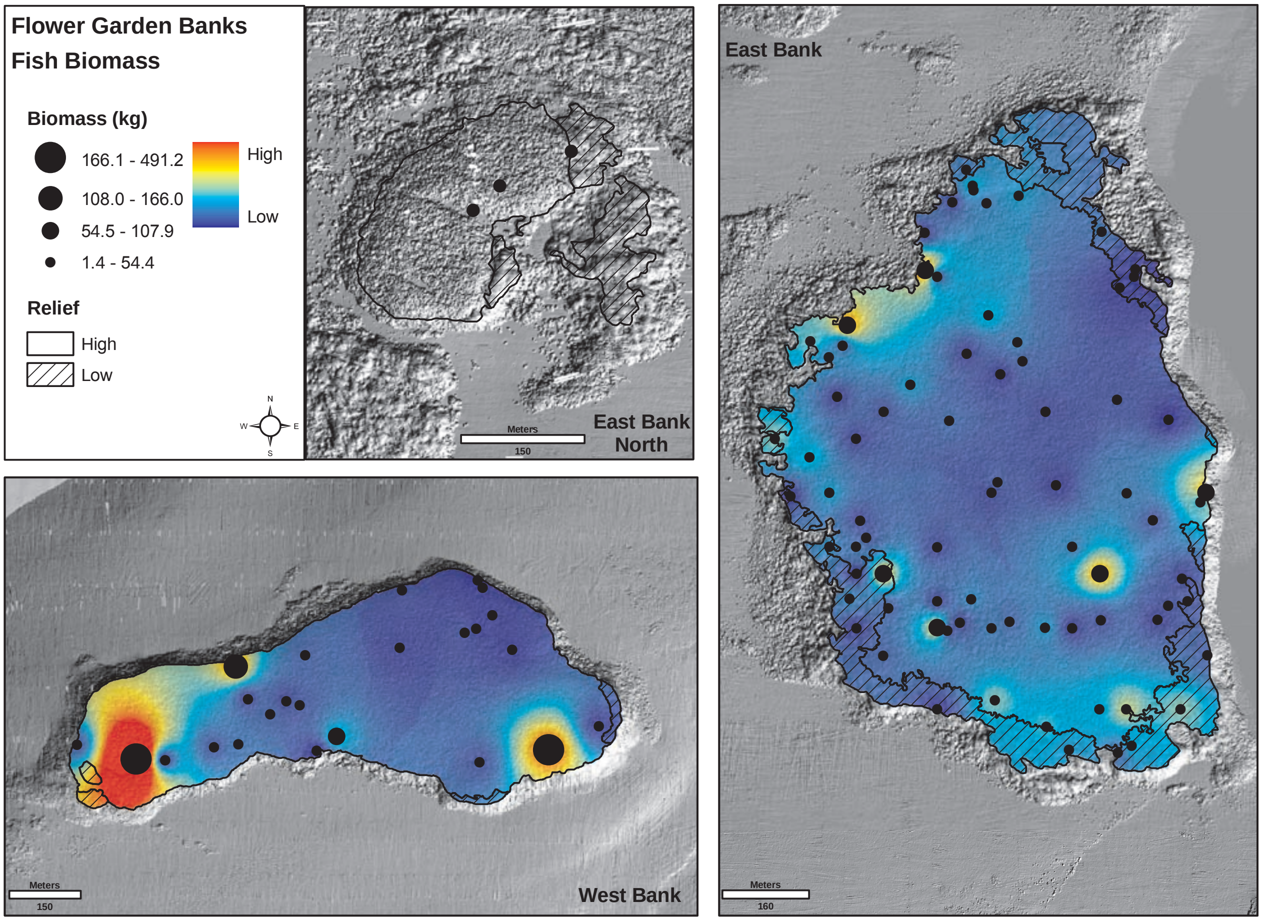


Figure 4.3. Observed (dots) and spatially interpolated biomass (kg/100 m^2) for all species observed in CCMA surveys for 2006-2007.

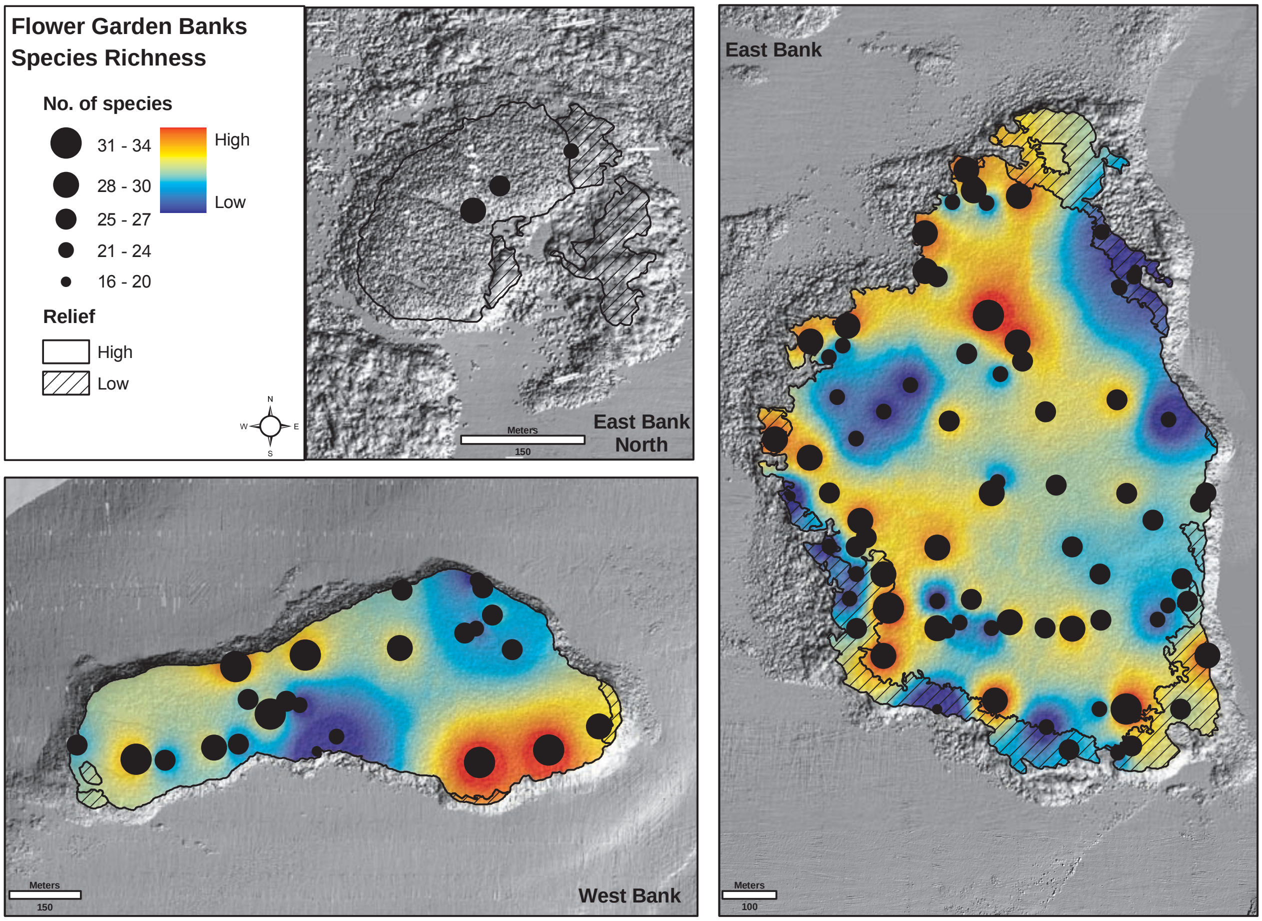


Figure 4.4. Observed (dots) and spatially interpolated species richness observed in CCMA surveys for 2006-2007.

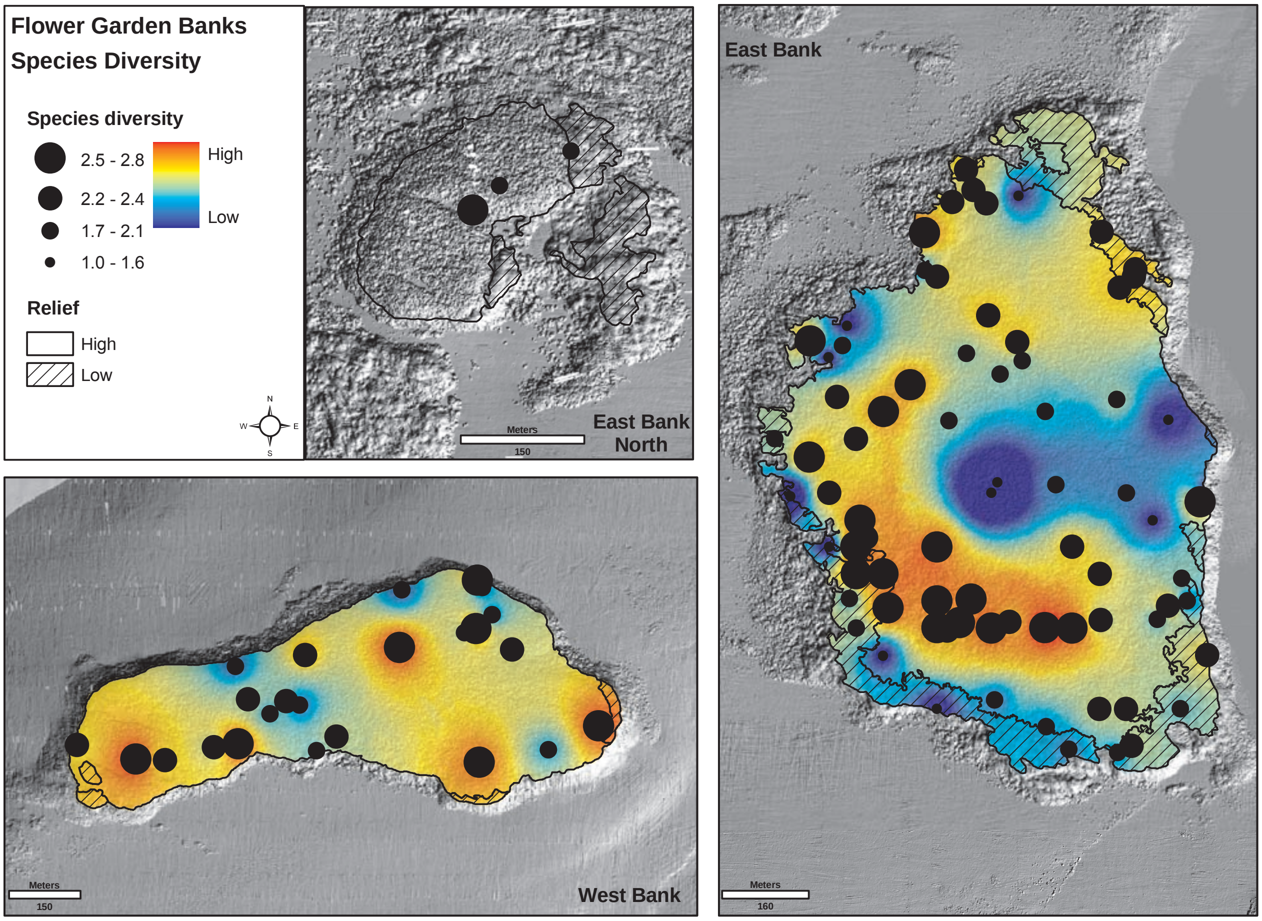


Figure 4.5. Observed (dots) and spatially interpolated species diversity observed in CCMA surveys for 2006-2007.

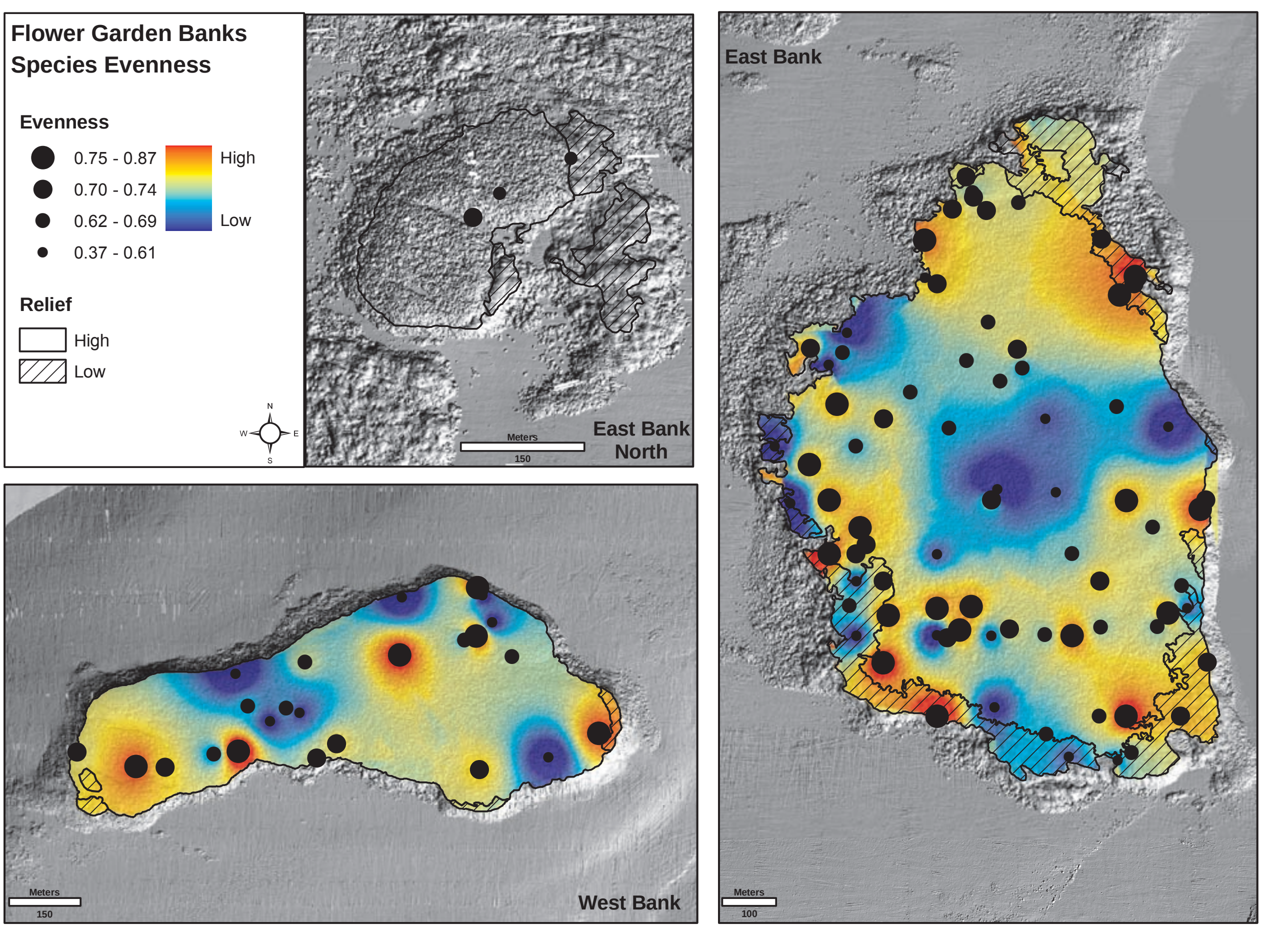


Figure 4.6. Observed (dots) and spatially interpolated species evenness observed in CCMA surveys for 2006-2007.

IDW results highlight the significantly lower species richness on EBL habitats. Species diversity was uniform on WB with small localized areas of lower diversity (Figure 4.5). In contrast, a horseshoe shaped area of high diversity that surrounds the central portion of the EB coral cap was observed. This pattern strongly resembles that of the interpolated distribution for *Montastraea franksi* (see Figure 3.3 in Chapter 3), the dominant coral species observed. Patterns of species evenness were patchy across both banks (Figure 4.6)

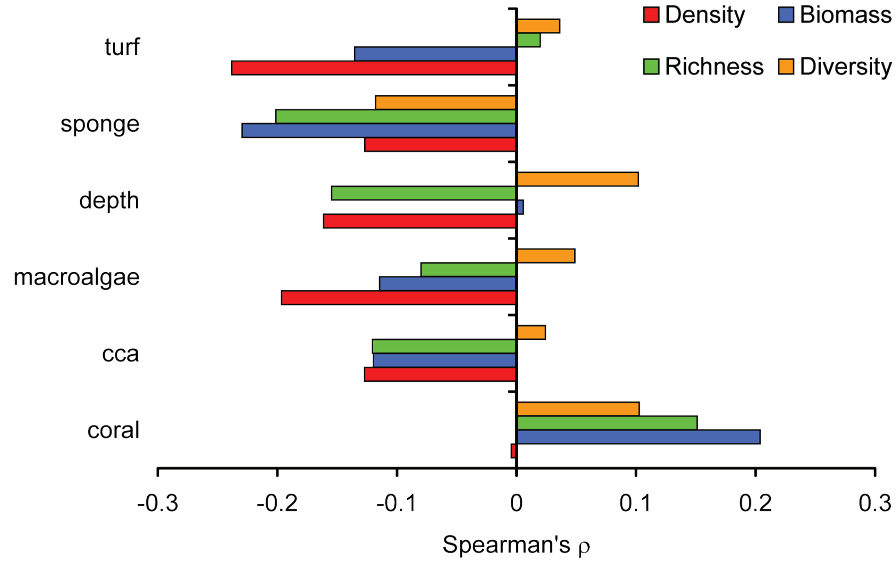


Figure 4.7. Summary results of Spearman's ρ correlation between community metrics and benthic cover and depth from CCMA observations for 2006-2007.

Fish density was not correlated with percent total coral cover (Figure 4.7) and negatively correlated with the remaining benthic parameters. These results indicate higher fish density on the shallower portions of the banks regardless of total coral cover; however, density was positively correlated with specific coral species, such as *Montastraea annularis* ($\rho = 0.26$), one of the key coral species on the banks. Fish biomass was also positively correlated with *Mo. annularis* cover ($\rho = 0.32$) and total coral cover ($\rho = 0.21$). Species richness was positively correlated with coral cover ($\rho = 0.27$) and depth ($\rho = 0.25$), but inversely correlated with *Madracis mirabilis* ($\rho = -0.28$), which was the key coral species on low relief habitats. Both species richness and diversity ($\rho = 0.24$ and $\rho = 0.23$, respectively) were positively correlated with *Mo. franksi*, the most abundant coral species on the banks. These patterns reflect the numerous micro-habitat types on the banks, especially on the deeper portions.

Approximately 65% of the total abundance was comprised of wrasse (Labridae) and damselfish (Pomacentridae) species (Figure 4.8). Most notably, the bluehead wrasse (*Thalassoma bifasciatum*), creole wrasse (*Clepticus parrae*) and brown chromis (*Chromis multilineata*) were among the top four species in total abundance (Figure 4.9). Serranidae (groupers) were the next most abundant family, predominantly represented by the Atlantic creolefish (*Paranthias furcifer*). Patterns of abundance by the most abundant species were variable compared to REEF and FGBNMS LTM surveys (Tables 4.4 and 4.5).

Chromis multilineata and *Diploria* species (CCMA)

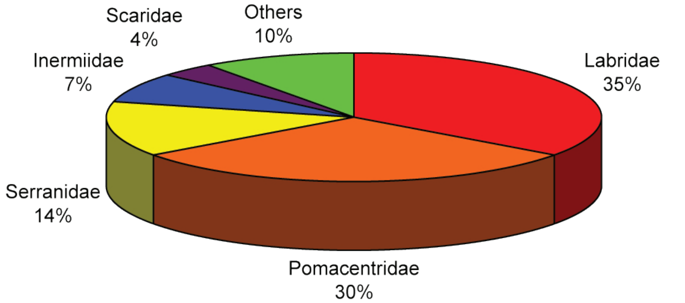


Figure 4.8. Top five most abundant families during CCMA surveys for 2006-2007.

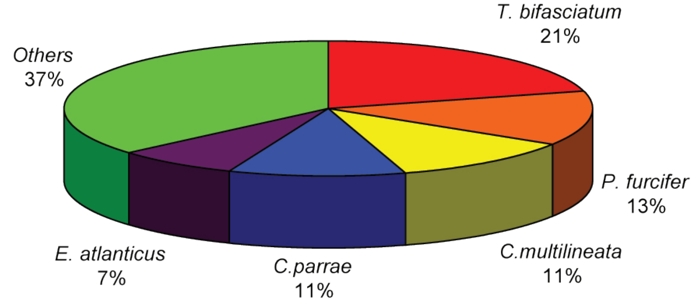


Figure 4.9. Top five most abundant species during CCMA surveys for 2006-2007.

Table 4.4. The 20 most abundant species at East Bank for CCMA, MMS and REEF surveys. Asterisk (*) indicates relative density (Pattengill-Semmens, 2006).

CCMA	Mean Density (#/100 m²)	MMS	Mean Density (#/100 m²)	REEF	Mean Density*
Thalassoma bifasciatum	69.38	*Clepticus parrae*	67.97	*Emmelichthyops atlanticus*	3.86
Paranthias furcifer	49.98	*Chromis multilineata*	33.00	*Chromis multilineata*	3.79
Clepticus parrae	41.40	*Chromis cyanea*	22.03	*Paranthias furcifer*	3.44
Chromis multilineata	40.96	*Thalassoma bifasciatum*	20.93	*Thalassoma bifasciatum*	3.44
Emmelichthyops atlanticus	34.32	*Paranthias furcifer*	18.40	*Kyphosus sectatrix/incisor*	3.21
Chromis insolata	22.42	*Stegastes planifrons*	9.57	*Clepticus parrae*	3.07
Stegastes partitus	19.47	*Kyphosus sectatrix/incisor*	7.60	*Stegastes planifrons*	3.07
Stegastes planifrons	17.86	*Stegastes partitus*	7.37	*Stegastes partitus*	3.04
Chromis scotti	6.94	*Emmelichthyops atlanticus*	6.67	*Canthigaster rostrata*	2.80
Carangoides ruber	6.75	Atherinidae	3.33	*Scarus vetula*	2.73
Canthigaster rostrata	5.58	*Scarus vetula*	2.37	*Chromis cyanea*	2.66
Bodianus rufus	4.74	*Acanthurus chirurgus*	1.50	*Sphyraena barracuda*	2.66
Sparisoma aurofrenatum	4.69	*Melichthys niger*	1.47	*Sparisoma viride*	2.57
Kyphosus sectatrix/incisor	3.94	*Sparisoma viride*	1.40	*Bodianus rufus*	2.53
Chromis cyanea	3.84	*Acanthurus coeruleus*	1.37	*Acanthurus coeruleus*	2.50
Acanthurus coeruleus	3.73	*Chaetodon sedentarius*	1.37	*Lactophrys triqueter*	2.44
Sparisoma viride	3.20	*Bodianus rufus*	1.27	*Melichthys niger*	2.44
Halichoeres garnoti	2.65	*Scarus taeniopterus*	1.03	*Stegastes variabilis*	2.36
Chaetodon sedentarius	2.46	*Carangoides ruber*	0.90	*Halichoeres garnoti*	2.34
Scarus taeniopterus	2.43	*Elacatinus oceanops*	0.87	*Chromis scotti*	2.33

Table 4.5. The 20 most abundant species at West Bank for CCMA, MMS and REEF surveys. Asterisk (*) indicates relative density (Pattengill-Semmens, 2006).

CCMA	Mean Density (#/100 m²)	MMS	Mean Density (#/100 m²)	REEF	Mean Density*
Thalassoma bifasciatum	93.00	*Paranthias furcifer*	26.25	*Emmelichthyops atlanticus*	3.80
Chromis multilineata	39.08	*Thalassoma bifasciatum*	20.94	*Chromis multilineata*	3.68
Paranthias furcifer	33.29	*Chromis multilineata*	20.94	*Paranthias furcifer*	3.53
Clepticus parrae	32.13	*Clepticus parrae*	14.72	*Thalassoma bifasciatum*	3.48
Stegastes planifrons	16.58	*Stegastes planifrons*	9.00	*Clepticus parrae*	3.20
Chromis insolata	13.67	*Chromis cyanea*	8.25	*Stegastes planifrons*	3.05
Bodianus rufus	8.50	*Emmelichthyops atlanticus*	6.88	*Chromis cyanea*	2.98
Chromis cyanea	6.04	*Stegastes partitus*	6.25	*Kyphosus sectatrix/incisor*	2.98
Stegastes partitus	5.83	*Acanthurus coeruleus*	2.41	*Stegastes partitus*	2.88
Canthigaster rostrata	5.75	*Melichthys niger*	2.31	*Chromis scotti*	2.73
Chromis scotti	5.67	*Scarus vetula*	2.31	*Sphyraena barracuda*	2.73
Sparisoma aurofrenatum	4.75	*Sphyraena barracuda*	2.19	*Scarus vetula*	2.68
Kyphosus sectatrix/incisor	4.67	*Halichoeres garnoti*	2.00	*Canthigaster rostrata*	2.65
Carangoides ruber	3.92	*Chaetodon sedentarius*	1.56	*Chromis insolata*	2.60
Acanthurus coeruleus	3.38	*Bodianus rufus*	1.56	*Bodianus rufus*	2.58
Halichoeres garnoti	3.00	*Sparisoma viride*	1.53	*Caranx crysos*	2.50
Chaetodon sedentarius	2.46	*Kyphosus sectatrix/incisor*	1.47	*Melichthys niger*	2.48
Caranx crysos	2.29	*Stegastes variabilis*	1.47	*Lactophrys triqueter*	2.45
Scarus vetula	1.92	*Elacatinus oceanops*	1.38	*Sparisoma viride*	2.43
Sparisoma viride	1.83	*Canthigaster rostrata*	1.28	*Acanthurus coeruleus*	2.38

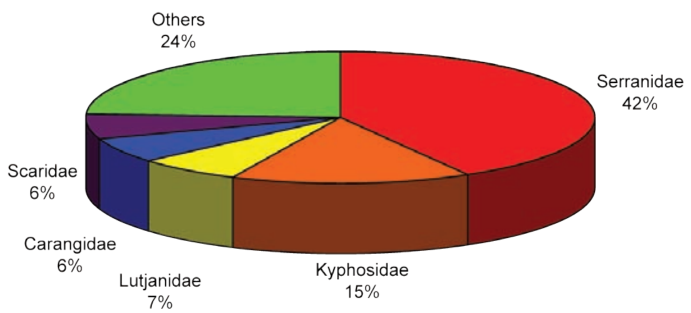


Figure 4.10. Top five families in total biomass (%) during CCMA surveys for 2006-2007.

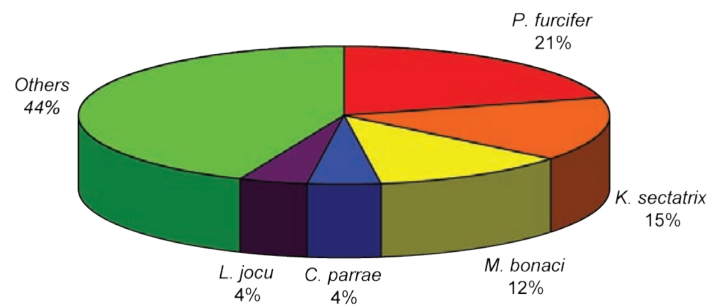


Figure 4.11. Top five species in total biomass (%) during CCMA surveys for 2006-2007.

Biomass was dominated by species from the family Serranidae, accounting for almost half of the total biomass observed in the sanctuary (Figure 4.10). Most notably, the numerically abundant *P. furcifer* and the heavy bodied *M. bonaci* account for the majority of serranid biomass (Figure 4.11). The family Kyphosidae, comprised of the single species bermuda/yellow chub (*Kyphosus sectatrix/incisor*) ranked second, followed by snappers (Lutjanidae), jacks (Carangidae) and parrotfish (Scaridae). The abundant medium sized *C. parrae* (fourth in total abundance) enabled it to rank fourth in total biomass. While not particularly abundant, large dog snappers (*Lutjanus jocu*) amassed considerable biomass ranking fifth overall.

4.3.1 Size Frequency

Figure 4.12 displays mean length frequency for all bank/relief type combinations. As expected, fish density (# individuals/100 m^2) was greatest in the smaller size categories and declines with increasing fish size. Mean density totaled 311/100 m^2 and over 70% were less than 10 cm. Density significantly declined for individuals greater than 10 cm and mean density for fish >30 cm totaled 7/100 m^2. Fish in the size class 0-5 cm were more abundant on low relief habitats, but were not significantly greater than high relief on either bank.

4.3.2 Trophic Groups

Planktivores and invertivores were numerically dominant ($p<0.0001$) regardless of bank or relief type (Figure 4.13). Mean piscivore abundance was significantly lower than all other trophic groups ($p<0.0001$) for all bank/habitat type combinations, while herbivore mean abundance was significantly greater than piscivores and significantly lower than planktivores and invertivores ($p<0.001$). Herbivore abundance was significantly greater on EBL ($p<0.001$) than high relief on either bank. Piscivore abundance was lowest on low relief habitats, but not significantly different than high relief on either bank. Herbivore composition was nearly twice as high as REEF observations (Table 4.6).

Kyphosus sectatrix/incisor (CCMA)

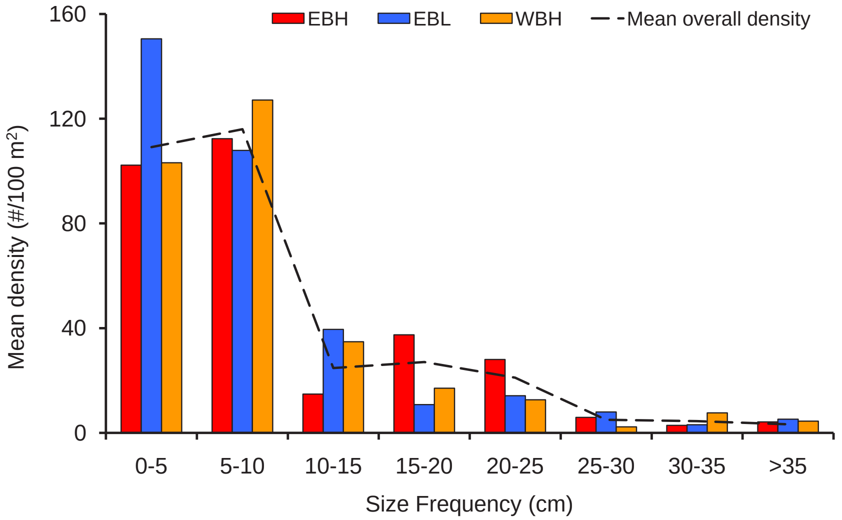


Figure 4.12. Size frequency (± SE) for all fish species observed in CCMA surveys for 2006-2007. Dashed line represents overall mean density per 100 m^2.

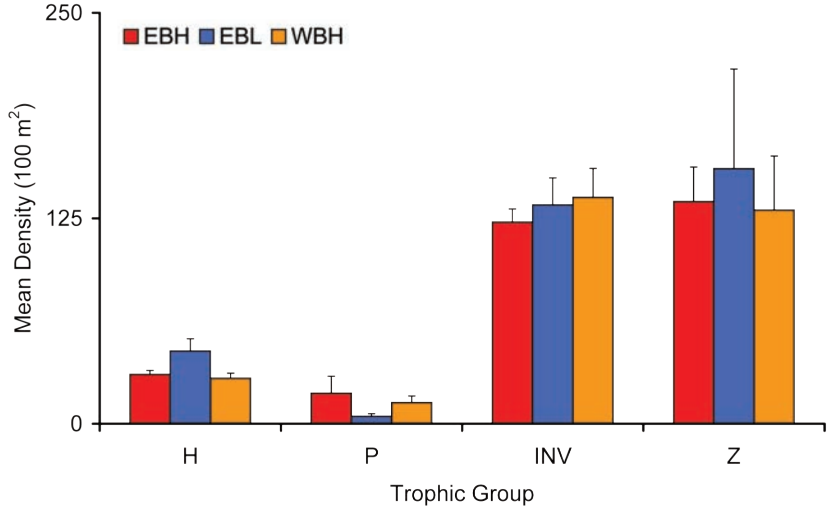


Figure 4.13. Trophic group mean density (± SE) by bank/relief. H= herbivore, P= piscivore, INV= invertivore, Z= zooplanktivore. EBH= East Bank High relief; EBL= East Bank Low relief; WBH= West Bank High relief.

Table 4.6. Percentage of total density by trophic groups observed by CCMA (2006) and REEF (1994-1995).

	CCMA East Bank	CCMA West Bank	REEF East Bank	REEF West Bank
Herbivore	12.36	8.96	23.00	17.00
Invertivore	29.92	44.57	34.30	26.60
Zooplanktivore	44.59	42.09	41.40	53.80
Piscivore	12.36	4.38	1.30	1.90

Overall, piscivore biomass accounted for 46% of the total biomass. This ratio is comparable to coral reef ecosystems with limited anthropogenic impacts in the Pacific where piscivores and apex predators dominated (Friedlander and DeMartini, 2002). Several large (150 cm) *M. bonaci* were observed on WB resulting in significantly greater piscivore biomass on high relief habitats (Figure 4.14) than all other trophic groups for any bank/habitat combination (p<0.0001). Biomass for all other groups was not significantly different for all bank/habitat type combinations.

In general, herbivore abundance was three times greater than piscivores and only seven surveys yielded a ratio of 1:1 or greater for piscivore abundance. Due to significantly lower piscivore abundance on low relief habitats (Figure 4.15), piscivore/herbivore abundance ratio was also significantly lower there (p=0.03) compared to high relief habitats. Overall, piscivore biomass was six times greater than herbivores, but no significant differences were observed between the bank/relief type combinations.

Piscivore density (Figure 4.16) was positively correlated with percent coral cover (ρ = 0.12) and negatively correlated with the other benthic parameters. Invertivore density was negatively correlated with depth (ρ = -0.28). Abundance and biomass for each trophic group were not significantly correlated with coral or macroalgae cover; invertivore abundance was negative correlated with depth (ρ = -0.30) while depth was not a significant factor for other trophic groups.

Zooplanktivores were the dominant trophic group on both banks during 2006 and 2007 (Table 4.6). This pattern was similar for surveys using a stationary point-count technique conducted in 1994-1995 (Pattengill-Semmens et al., 1997). Observed herbivore density was lower on East and West Banks while piscivore abundance was 10 times greater on EB and twice as high on WB. Differences in trophic structure are presumed to reflect the sampling method used where the stationary point count method which may under-sample fishes, that hide in the crevices of reef structure.

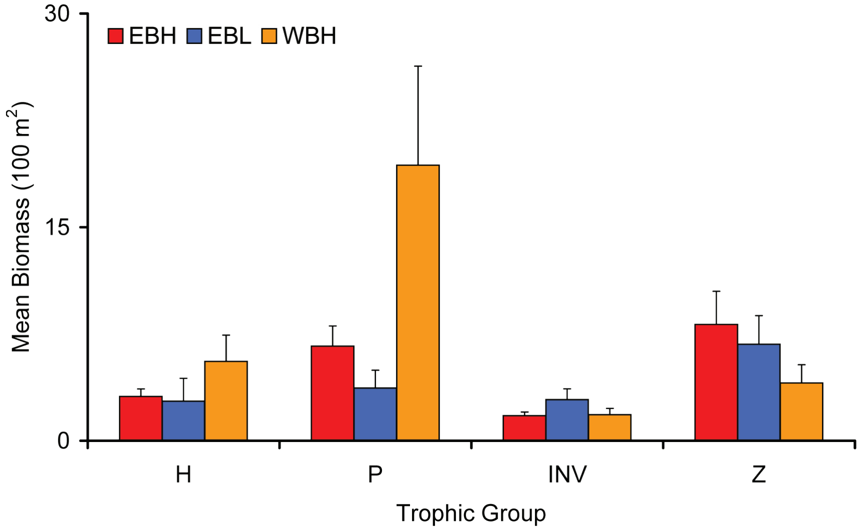


Figure 4.14. Trophic group mean biomass (kg; ± SE) by bank/relief. H= herbivore, P= piscivore, INV= invertivore, Z= zooplanktivore. EBH= East Bank High relief; EBL= East Bank Low relief; WBH= West Bank High relief.

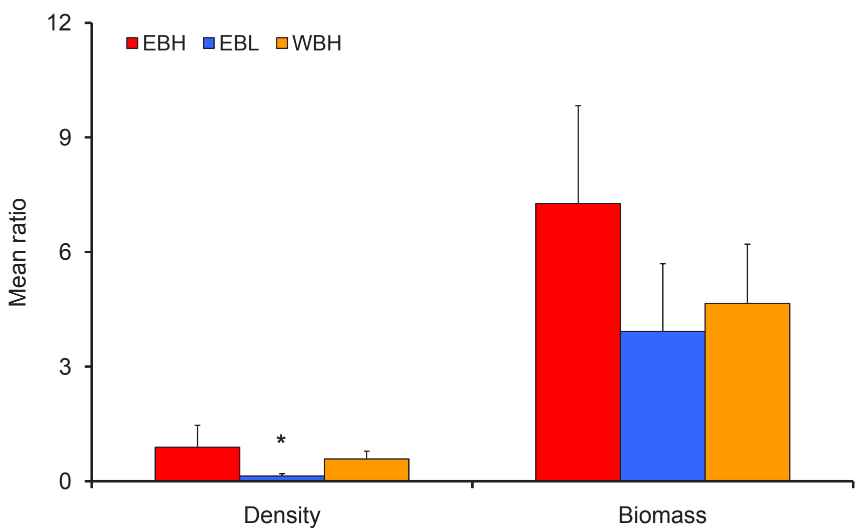


Figure 4.15. Piscivore/herbivore abundance and biomass mean ratio (± SE) for each bank/relief type combination. Asterisk (*) indicates statistical significance. EBH= East Bank High relief; EBL= East Bank Low relief; WBH= West Bank High relief.

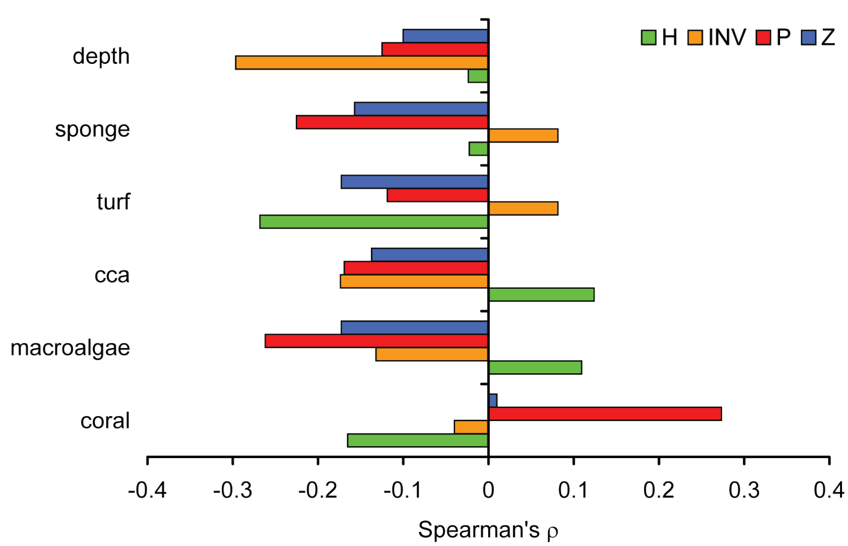


Figure 4.16. Spearman's ρ correlations between trophic groups and percent cover of benthic habitat parameters. H= herbivore, INV= invertivore, P= piscivore, Z= zooplanktivore.

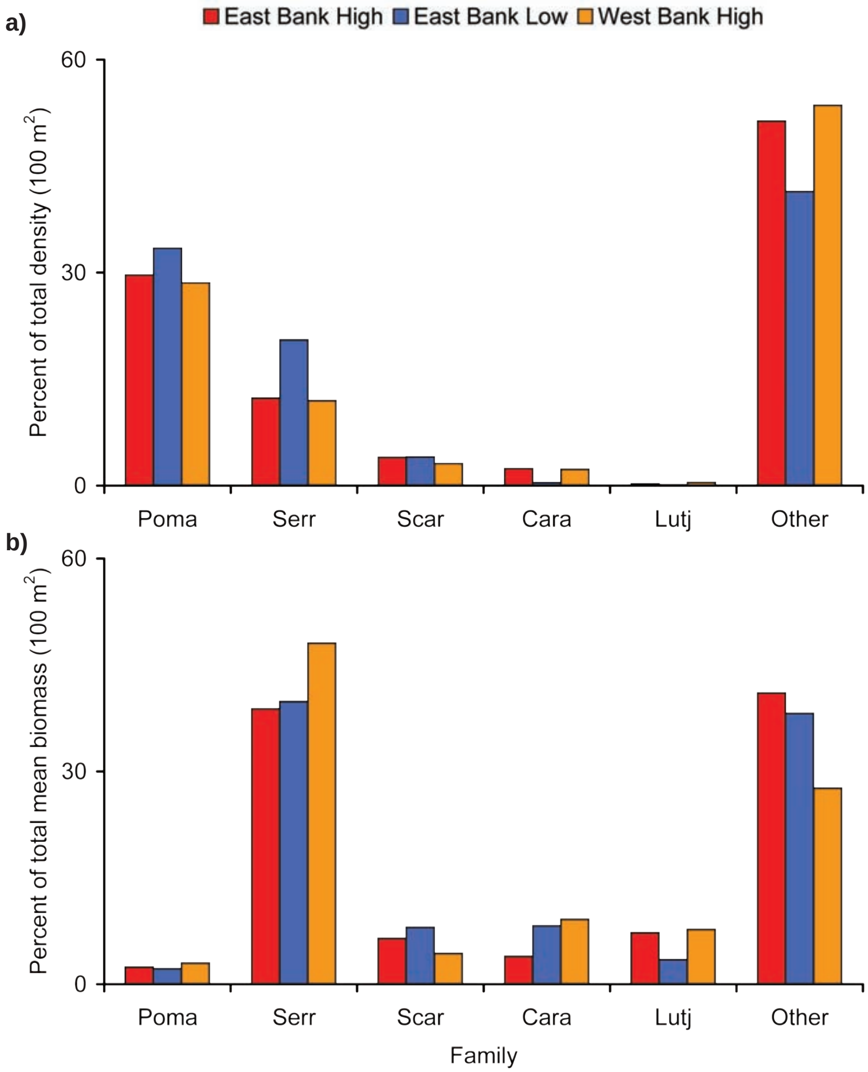


Figure 4.17. Percent of total mean (a) density and (b) biomass for five ecologically and commercial/recreationally important families observed during CCMA surveys for 2006-2007. Poma= Pomacentridae; Serr= Serranidae; Scar= Scaridae; Cara= Carangidae; Lutj= Lutjanidae.

4.3.3 Taxonomic Groups

Five families of fish within the SCC were selected for additional analysis because of their ecological or commercial/recreational importance (Serranidae, Lutjanidae, Scaridae, Carangidae and Pomacentridae). Combined, these families comprised nearly 50% of the total fish density and up to 70% of the biomass on each bank/habitat type combination (Figure 4.17). A more detailed description of the spatial patterns of abundance and biomass for the families as a whole and select species within each family follows.

Fish assemblage (CCMA)

4.3.3.1 *Serranidae (Groupers)*

With the exception of a few species that have been assessed in U.S. waters, there is little data on many grouper species in the Gulf of Mexico and Caribbean, making it difficult to adequately assess the status of many species. Estimates of total abundance are difficult to obtain for species such as groupers that are strongly associated with physical structures, like reefs, where they typically hide during the day.

Mycteroperca tigris (CCMA)

Many grouper species, including those evaluated here, are protogynous hermaphrodites. They begin their lives as females and become males as they grow larger (Heemstra and Randall, 1993). The larger males are often targeted by commercial and sport fishing, thus altering natural gender ratios. The tendency of groupers to exhibit site-specificity combined with slow growth rates create an enhanced susceptibility to overfishing (Heemstra and Randall, 1993).

P. furcifer, a planktivore, and 10 species of commercially and/or recreationally important species of serranids belonging to the genera *Cephalopholis*, *Epinephelus* and *Mycteroperca*, here-after referred to as groupers, (*Cephalopholis cruentata*, *Cephalopholis fulva*, **Dermatolepis inermis*, **Epinephelus adscensionis*, **Epinephelus guttatus*, **Mycteroperca bonaci*, **Mycteroperca interstitialis*, **Mycteroperca phenax*, *Mycteroperca tigris* and **Mycteroperca venenosa*) were observed within the SCC during the study period. Seven of these species (indicated previously with an asterisk) are managed as the reef fish complex in the Gulf of Mexico Fishery Management Council (GOMFMC) Plan (GOMFMC, 2004). Approximately 70 million pounds of grouper from the shallow water complex have been commercially harvested within the Gulf of Mexico during 2000-2006 (GOMFMC, 2004) and nearly 99% of this harvest was landed in Florida. Approximately 115,000 pounds of grouper have been landed in Texas since 2000 including three species (*M. bonaci*, *M. phenax* and *M. venenosa*) observed within the SCC; however, these constitute a small proportion of Gulf-wide landings. Recreational landings of groupers within the Gulf are infrequent and are not described herein.

Sighting frequencies were comparable between the two sets of surveys with some exceptions (Table 4.7). *C. cruentata* and *M. tigris* were sighted considerably more frequently by REEF on both East and West Banks. Surveys during 2006 and 2007 yielded greater sighting frequency for *M. bonaci*, *D. inermis* and *E. guttatus* on both banks. The first sighting of Nassau grouper (*Epinephelus striatus*) and second sighting of goliath grouper (*Epinephelus itajara*) within the sanctuary occurred during the study period (Foley et al., 2007).

Serranids as a whole were the third most abundant family during the time period and were observed on all banks and relief types. Grouper density patterns were not significantly different between bank and habitat type combinations (Figure 4.19). In general, density was greatest on the margins of the coral caps with fewer individuals observed in the central, shallow region. Density was dominated by *P. furcifer* comprising 93% of serranid abundance. Similarly, grouper biomass (Figure 4.19) was not significantly different between bank and habitat type combinations and was highest on the margins of the coral caps. Biomass was dominated by the larger groupers, such as *M. bonaci* and *M. tigris*. The majority of groupers observed were in the 20-40 cm size class, with the larger individuals typically found at the edges of the coral caps (Figure 4.20). Grouper abundance and biomass were not correlated with coral, macroalgae cover or depth.

Table 4.7. Sighting frequency of select Serranidae species from CCMA, REEF and MMS surveys. REEF estimates (means) are from expert surveys only. MMS data only reflect presence/absence indicated by +/- .

	East Bank			West Bank		
Species	**CCMA**	**REEF**	**MMS**	**CCMA**	**REEF**	**MMS**
Paranthias furcifer	96.30	93.76	+	83.33	95.43	+
Cephalopholis cruentata	44.44	88.13	-	58.33	79.40	+
Mycteroperca interstitialis	48.15	69.76	+	66.67	45.83	-
Mycteroperca tigris	28.40	60.89	+	37.50	45.78	-
Epinephelus adscensionis	12.35	23.39	+	0	5.60	+
Mycteroperca bonaci	13.58	7.09	-	25.00	9.93	-
Mycteroperca venenosa	3.70	5.25	+	4.17	2.38	-
Cephalopholis fulva	2.47	2.93	+	0	8.85	+
Dermatolepis inermis	13.58	2.28	+	12.50	0	+
Mycteroperca phenax	1.23	0.93	-	4.17	0	-
Epinephelus guttatus	8.64	0.64	-	12.50	2.90	+

The following contains more detailed information regarding spatial patterns of density and biomass for each grouper species and for *P. furcifer*.

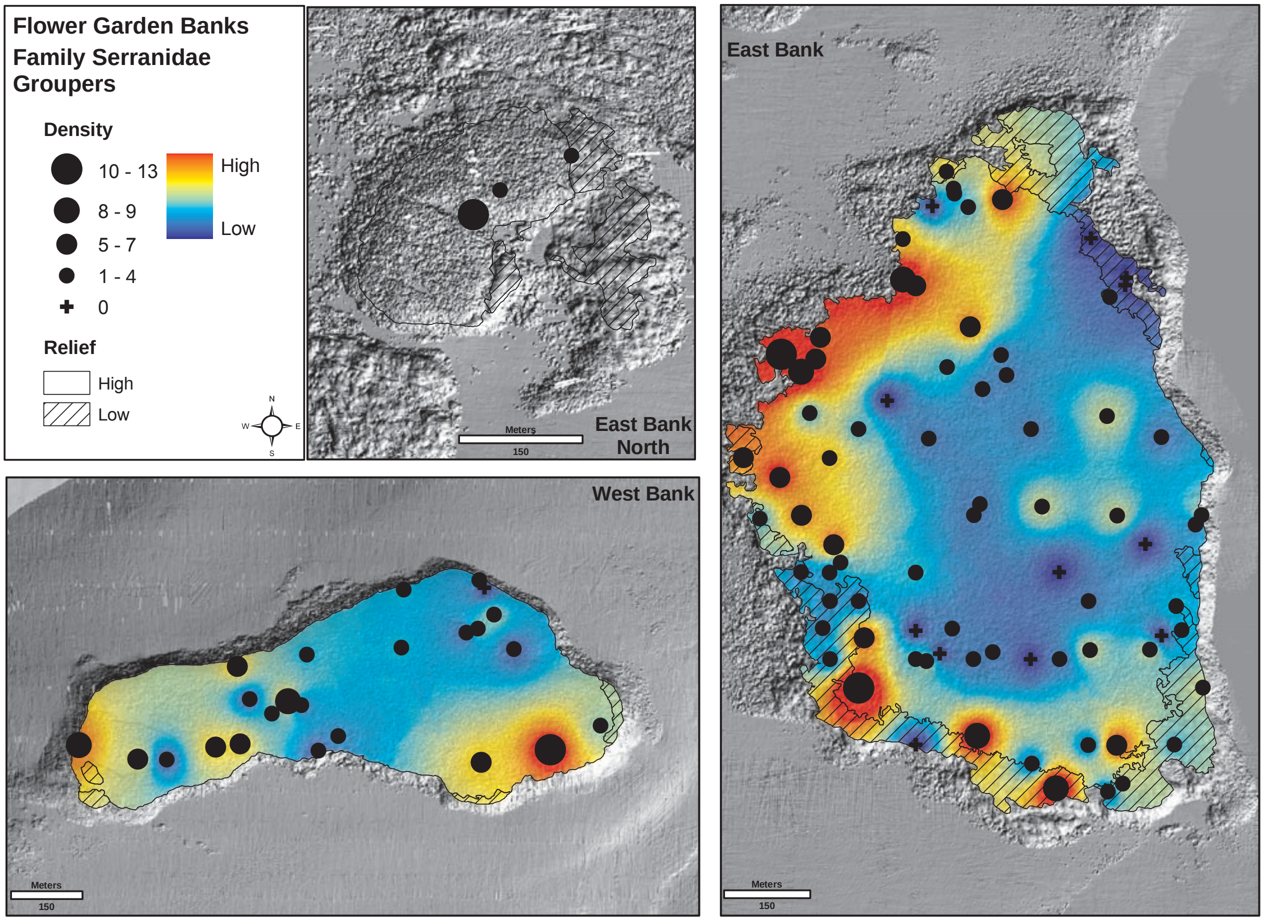


Figure 4.18. Observed (dots) and spatially interpolated density (#/100 m^2) for selected grouper species observed in CCMA surveys for 2006-2007.

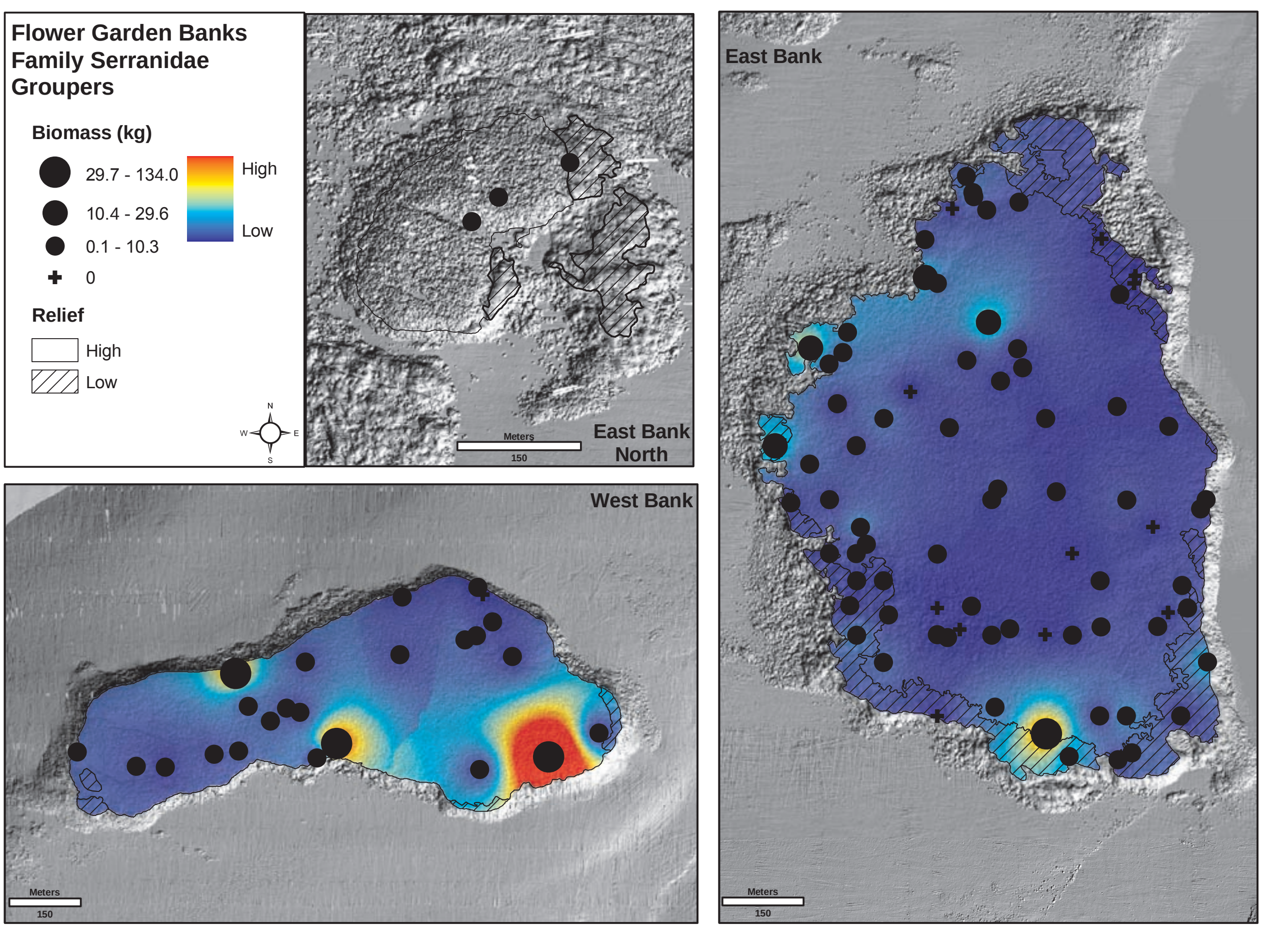


Figure 4.19. Observed (dots) and spatially interpolated biomass (kg) for selected grouper species observed in CCMA surveys for 2006-2007.

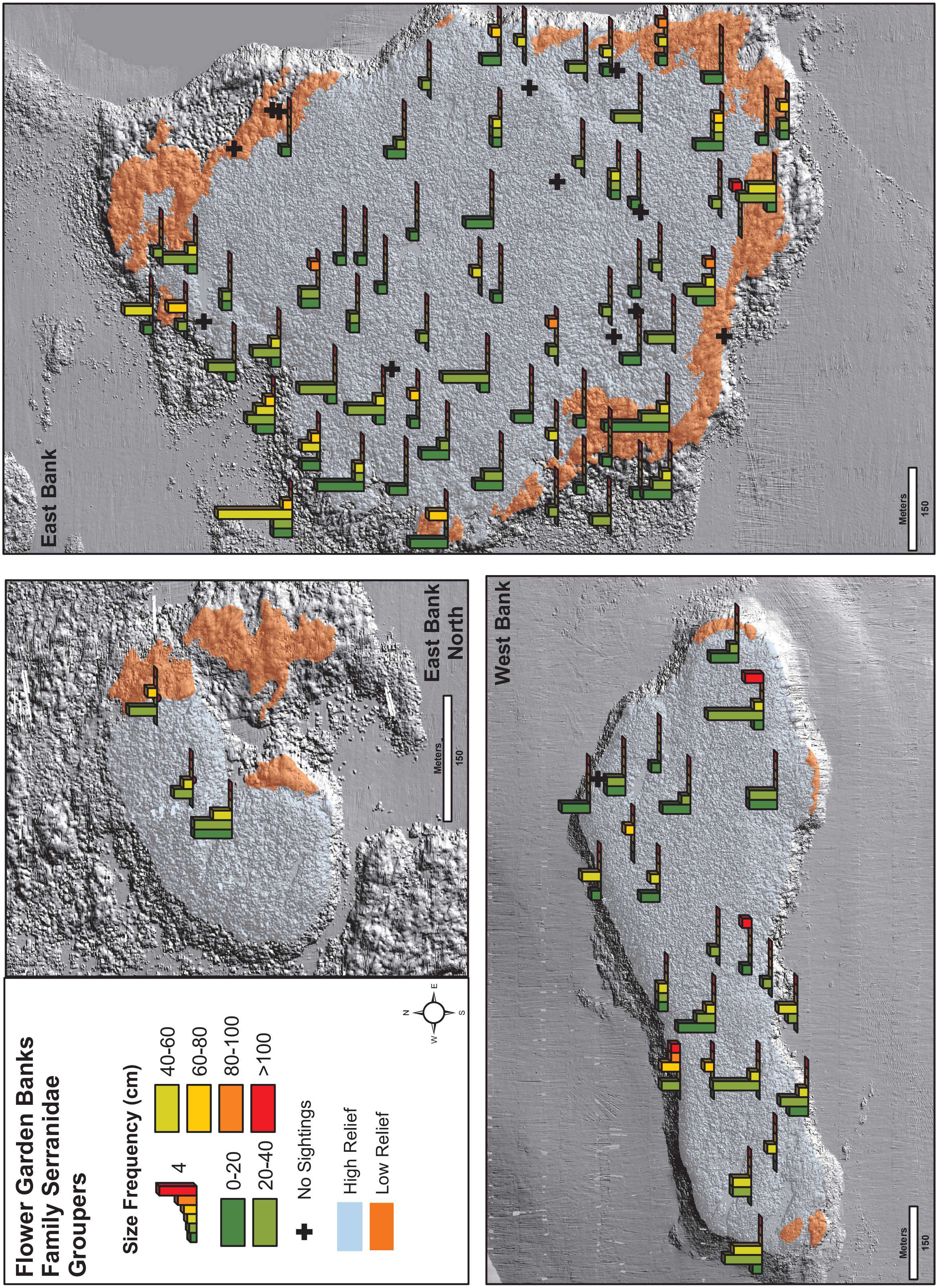
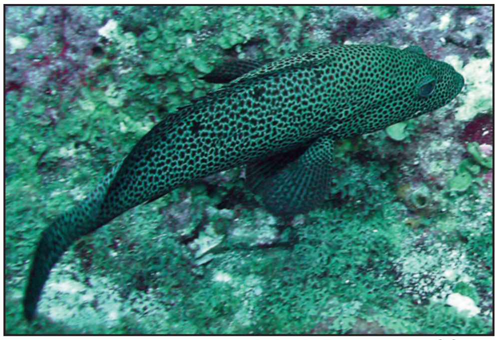


Figure 4.20. Spatial distribution of selected grouper size frequency observed in CCMA surveys for 2006-2007. The tallest bar in the legend histogram represents four individuals.

4.3.3.1.1 Graysby (*Cephalopholis cruentata*)

C. cruentata are common coral reef or rocky ledge inhabitants found at depths from 5-170 m and ranging from North Carolina to the Gulf of Mexico and Caribbean (SAFMC, 2005). *C. cruentata* is typically sedentary, hiding in the reef during the day and feeding nocturnally. Adults are primarily piscivores (Randall, 1967) and attain maximum size of 42.6 cm and approximately 1.1 kg (Erdman, 1976). Adults attain sexual maturity at 14 cm total length (TL)/FL in the Caribbean (Nagelkerken, 1979). Due to its small size, there is no commercial value for this species; however it has significant value to subsistence fisheries in the Caribbean (Heemstra and Randall, 1993).



Cephalopholis cruentata (CCMA)

Frequency of occurrence for *C. cruentata* was considerably lower than that observed by REEF (Table 4.7). This pattern is most notable on EB, but frequency of occurrence was reduced on WB as well. REEF's roving diver method is not restrained to predefined observation areas and may have an influence on this pattern.

The majority of *C. cruentata* observed were 10-25 cm FL with few less than five and greater than 30 cm (Figure 4.21). *C. cruentata* density (Figure 4.22) was generally higher on WBH, but was not significantly different from EBL or WBH. Mean density (individuals/100 m^2), by bank, was slightly higher on WB (1.14/100 m^2) than EB (0.71/100 m^2). Density appears to be higher on the western edge of EB and eastern portion of WB. Little difference was observed between mean biomass on EB (0.07 kg/100 m^2) and WB (0.11 kg/100 m^2). Biomass was higher on WBH (Figure 4.23), but this pattern was not significantly different compared to habitats on EB. Modal size frequency was smaller on EBH compared to WBH (Figure 4.24).

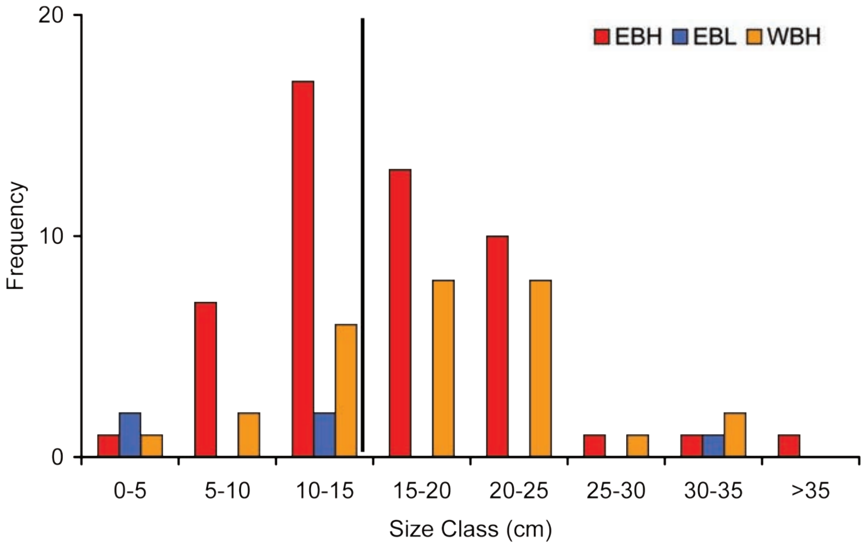


*Figure 4.21. Length frequency of graysby (*C. cruentata*) from CCMA surveys for 2006-2007. Vertical black line represents size at maturity (Nagelkerken, 1979). EBH=East Bank High relief; EBL=East Bank Low relief; WBH=West Bank High relief.*

C. cruentata density was positively correlated with coral cover (ρ = 0.20) but not correlated with macroalgae cover or depth. Biomass and fish size were not correlated with any of the benthic parameters.

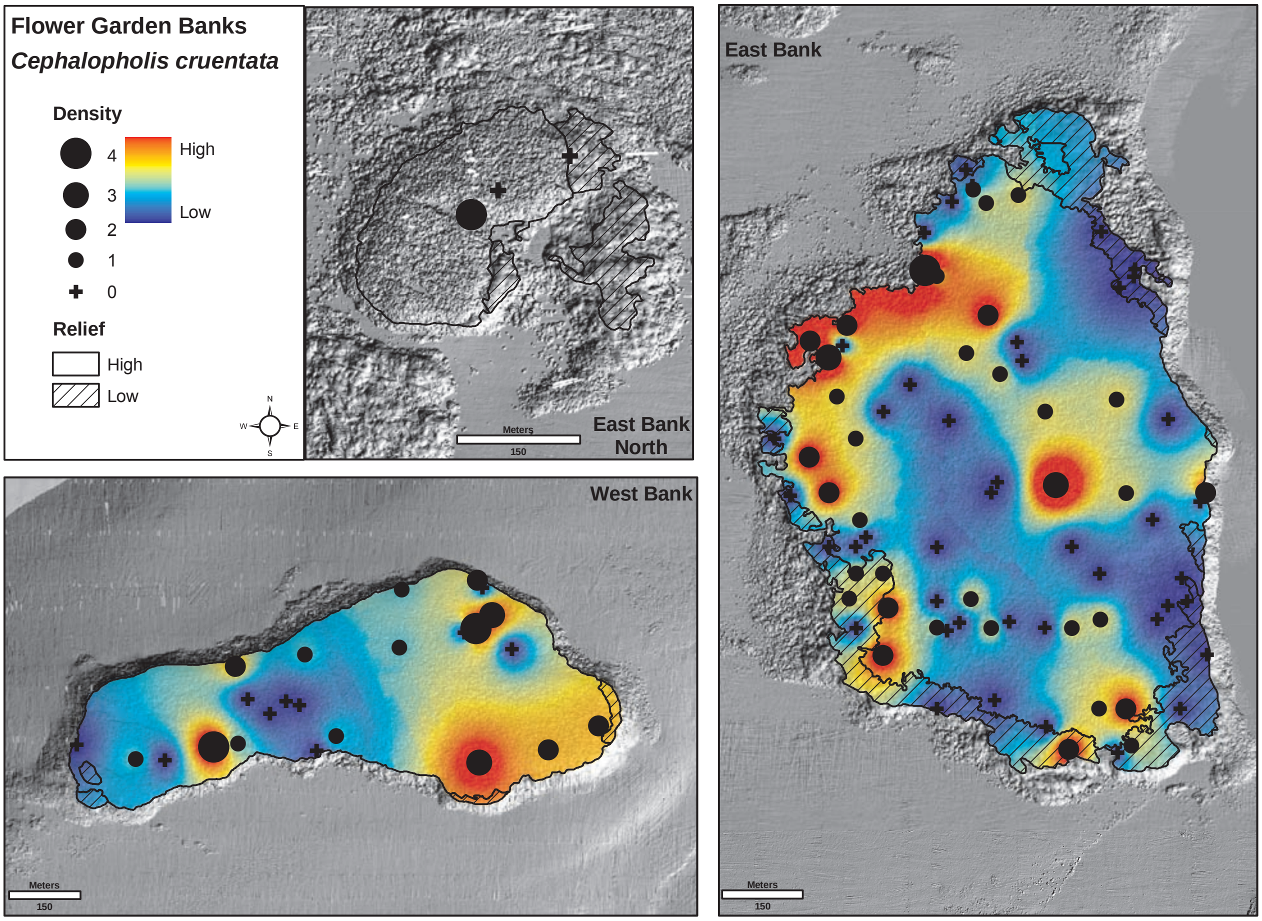


*Figure 4.22. Observed (dots) and spatially interpolated density (#/100 m^2) for graysby (*C. cruentata*) observed in CCMA surveys for 2006-2007.*

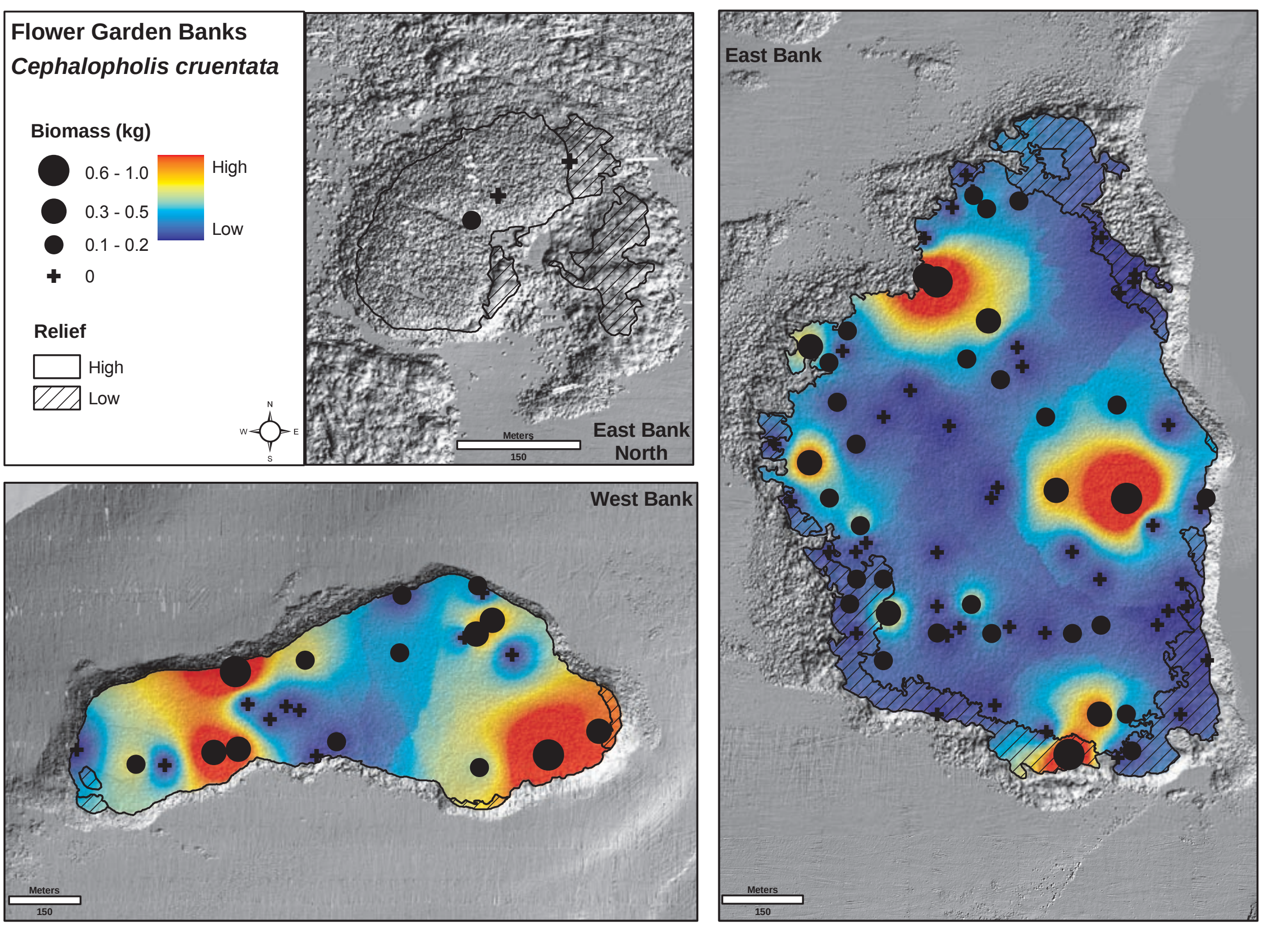


*Figure 4.23. Observed (dots) and spatially interpolated biomass (kg) for graysby (*C. cruentata*) observed in CCMA surveys for 2006-2007.*

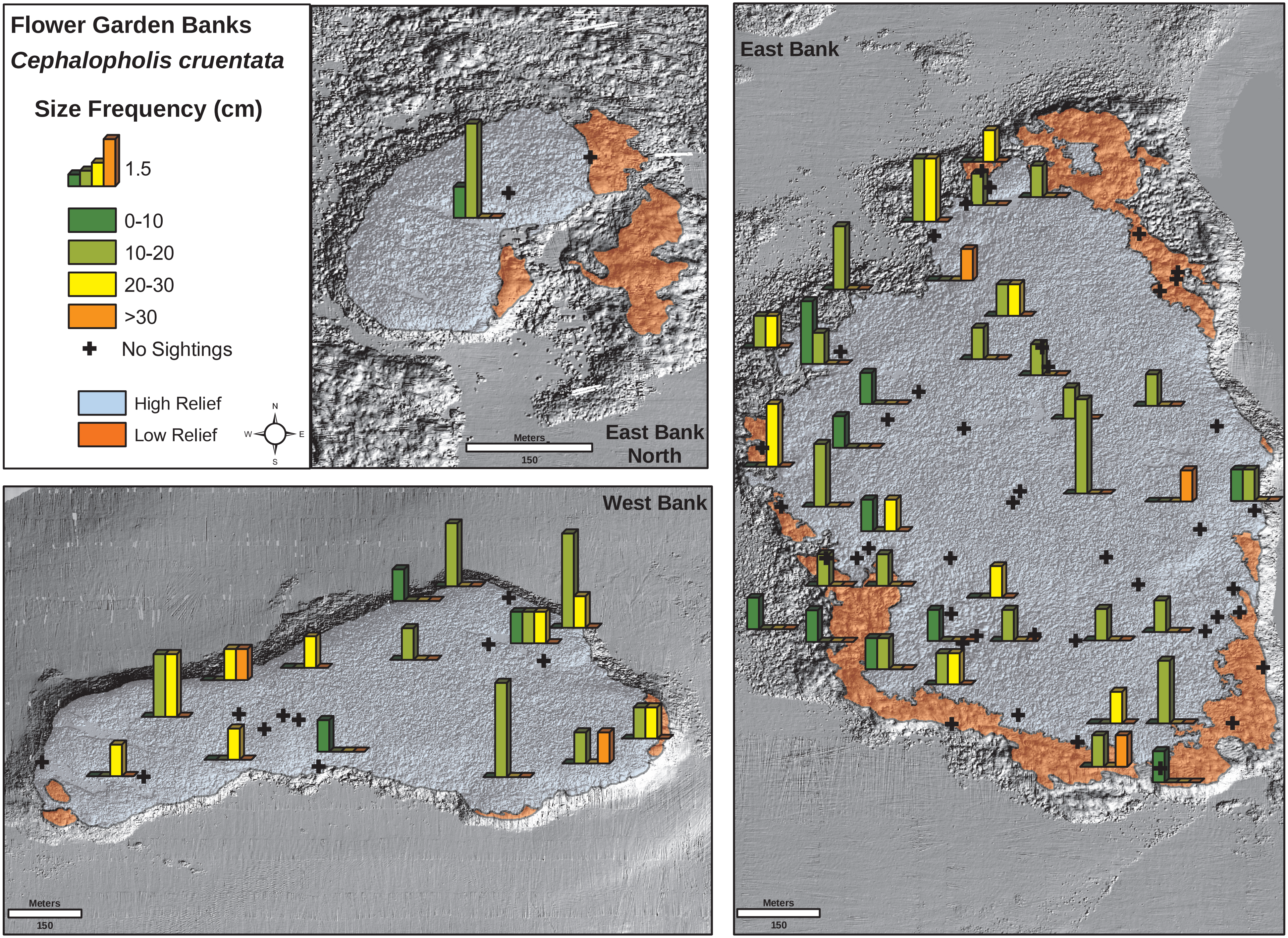
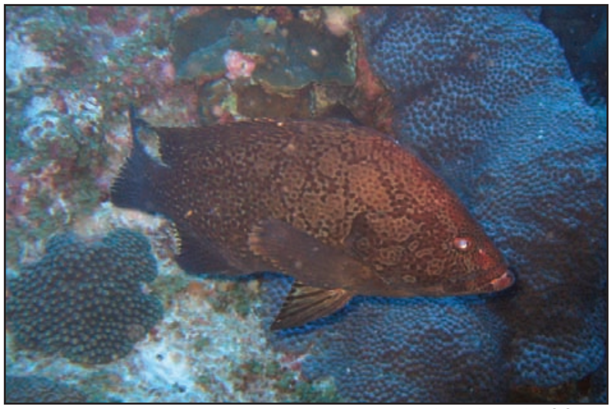


Figure 4.24. Spatial distribution for graysby (C. cruentata) size frequency observed in CCMA surveys for 2006-2007. The histogram bar in the legend represents 1.5 individuals.

4.3.3.1.2 Marbled grouper (*Dermatolepis inermis*)

D. inermis is a medium sized serranid capable of reaching 91 cm TL and 10 kg (Heemstra and Randall, 1993). It is found on reefs, especially caves and crevices, at depths between 3-213 m from North Carolina to Brazil, including the Gulf of Mexico and the Caribbean. Maximum age and size at maturity are not currently known. Population status throughout its range is uncertain; however it is listed on the International Union for Conservation of Nature (IUCN) list of threatened species (Huntsman, 1996).



Dermatolepis inermis (CCMA)

D. inermis sighting frequency ranged from 12-13% on West and East Banks, respectively (Table 4.7). In comparison, *D. inermis* frequency estimated by REEF was only 2.2% on EB and none were sighted on WB. Differences in sampling methods could explain these contrasting patterns. CCMA transect method is more efficient for observing secretive fish that hide in reef crevices. Additionally, REEF surveys were conducted near the mooring buoys in the central, shallow portion of the coral caps while CCMA surveys were more spatially comprehensive and sightings were not in close proximity to the mooring buoys.

Only 15 individuals were observed during 2006-2007 and, with the exception of one transect where two fish were sighted, single fish observations were recorded. Individuals were all greater than 20 cm FL and the majority were in the 40-60 cm size class (Figure 4.25). Density was concentrated at the edges of the coral caps (Figure 4.26) with limited sightings in the center or shallow portion of the coral caps. Mean density was similar on both banks (EB=0.16/100 m^2, WB=0.12/100 m^2). Consequently, biomass was also centered around the coral cap edges (Figure 4.27). Although observations were limited, *D. inermis* exhibited greater density and size on EB (Figure 4.28).

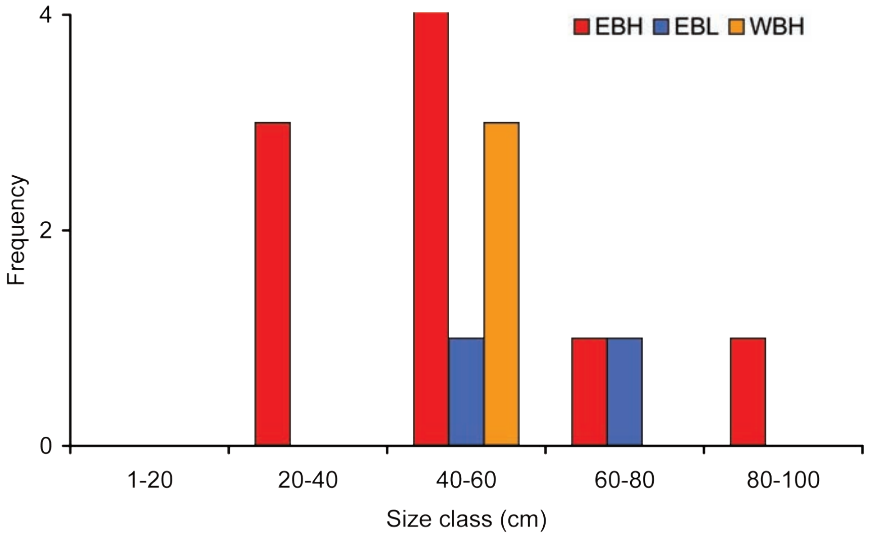


*Figure 4.25. Length frequency of marbled grouper (*D. inermis*) observed in CCMA surveys for 2006-2007.*

Due to low sighting frequency, statistical correlations with benthic features were not possible; however, *D. inermis* were generally found on habitats with >60% coral cover, <40% macroalgae cover and at depths >24 m around bank edges.

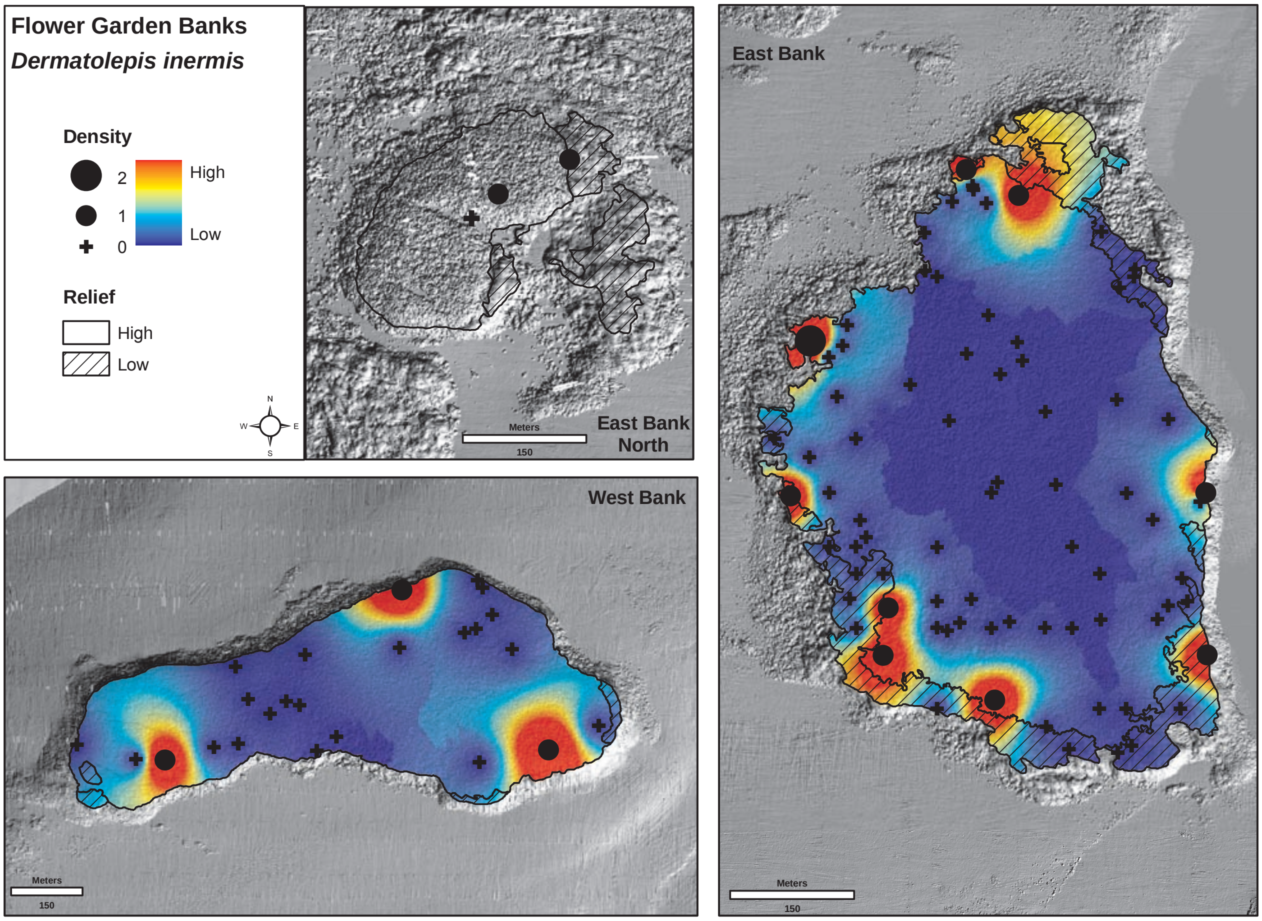


Figure 4.26. Observed (dots) and spatially interpolated density (#/100 m^2) for marbled grouper (D. inermis) observed in CCMA surveys for 2006-2007.

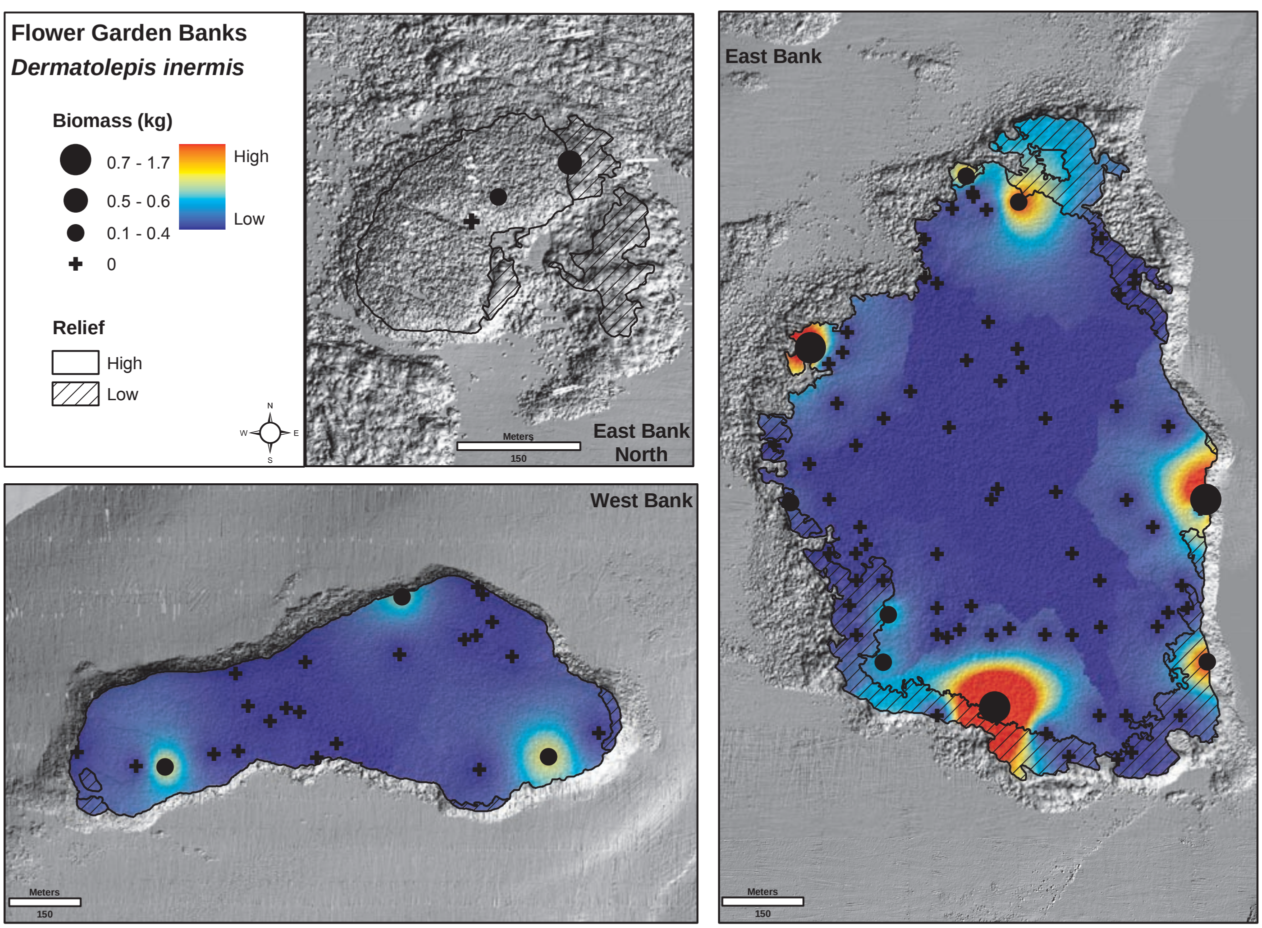


Figure 4.27. Observed (dots) and spatially interpolated biomass (kg) for marbled grouper (D. inermis) observed in CCMA surveys for 2006-2007.

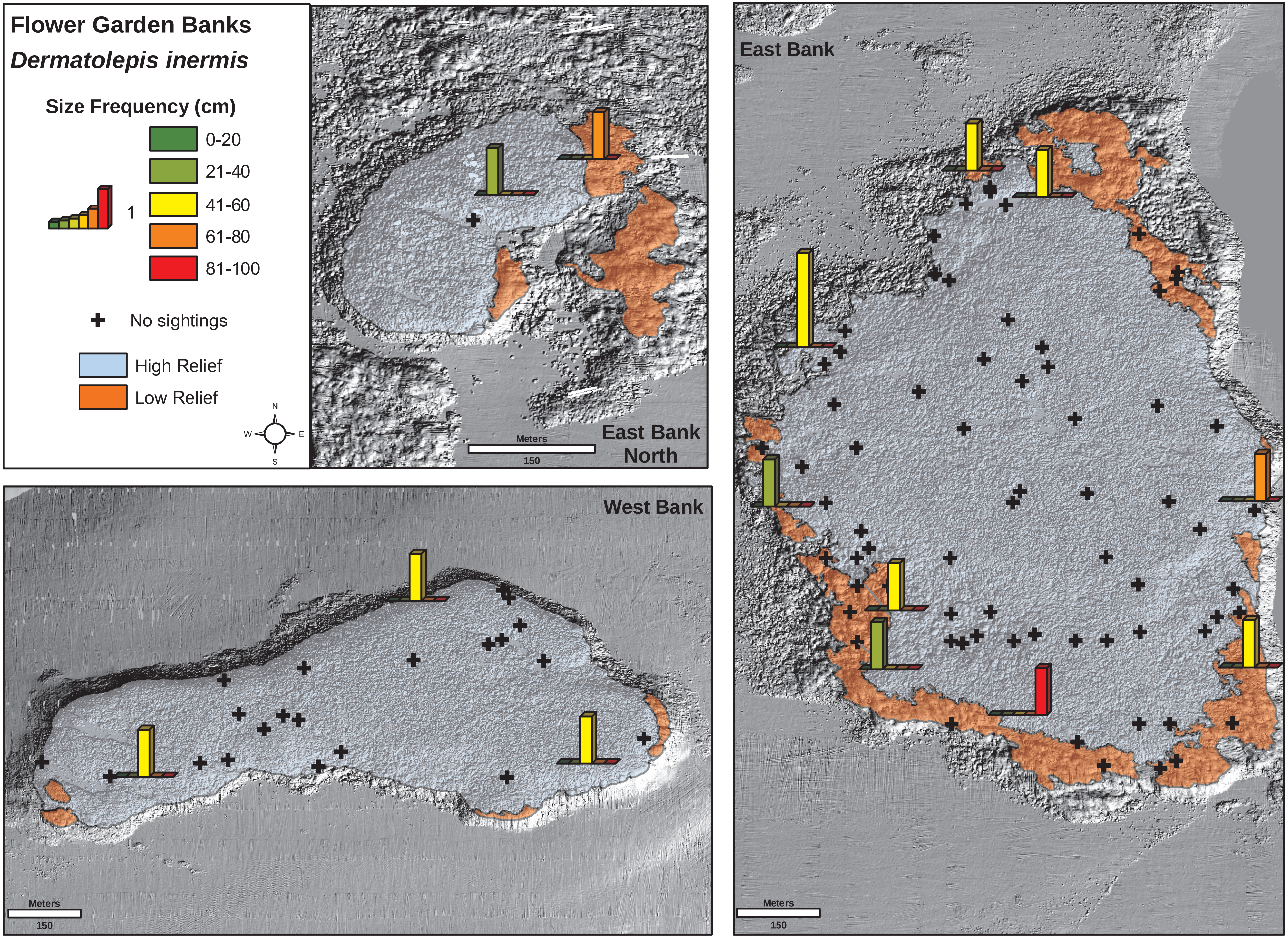
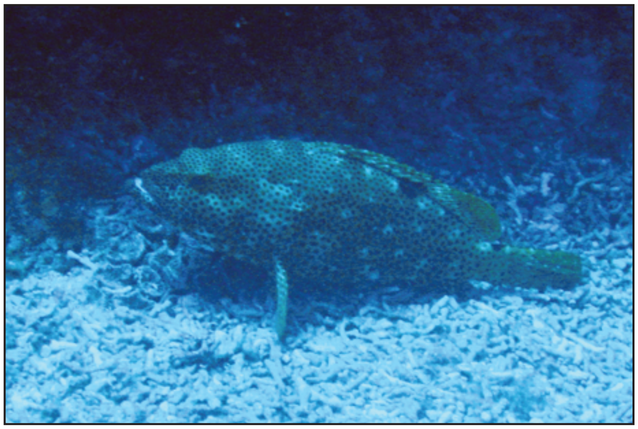


Figure 4.28. Spatial distribution for marbled grouper (D. inermis) size frequency observed in CCMA surveys 2006-2007. The tallest histogram bar in the legend represents one individual.

4.3.3.1.3 Rock hind (*Epinephelus adscensionis*)

E. adscensionis is a common serranid on rocky reefs at depths from 5-120 m, ranging from Massachusetts to Brazil, including the Gulf of Mexico and Caribbean (Smith, 1997). *E. adscensioni*s attain maximum size at approximately 61 cm and weigh up to 4 kg (Heemstra and Randall, 1993). Similar to other serranids, growth is slow where individuals off Florida (Bullock and Smith, 1991) reach sexual maturity at 25 cm FL with an approximate age of 6.1 years (Potts and Manooch, 1995). *E. adscensionis* is of minor importance to commercial and sport fisheries in the western Atlantic and Caribbean, as it seems to be less common than most other groupers (Heemstra and Randall, 1993); however, they are susceptible to overfishing due to their size and age at maturity (Cheung et al., 2005).



Epinephelus adscensionis (CCMA)

E. adscensionis sighting frequency was generally lower than that reported by REEF (Table 4.7). REEF sighting frequency averaged 23% on EB, approximately twice the frequency observed during 2006-2007. No *E. adscensionis* were sighted on WB; however, REEF and MMS observed them but at considerably lower frequency than EB.

During 2006-2007, observations of *E. adscensionis* were low (n=12) and the majority were juveniles (Figure 4.29). All sightings occurred on EB (Figure 4.30) with a mean density of 0.12/100 m^2. Density was patchy with most observations occurring on the edge of EB. Density was greatest on high relief habitat, as only one individual was observed on low relief. Biomass was also concentrated at the edge of the coral cap (Figure 4.31). Size frequency is displayed in Figure 4.32, however, no spatial patterns emerged with limited observations. Both juveniles and adults were observed on high relief habitats in equal proportions, while only a single juvenile was observed on low relief.

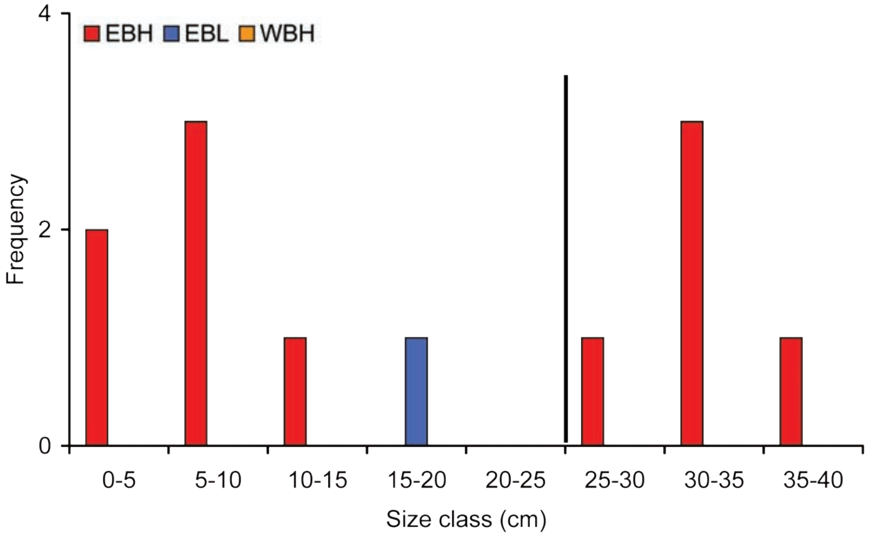


*Figure 4.29. Length frequency of rock hind (*E. adscensionis*) observed in CCMA surveys for 2006-2007. Vertical black line represents estimated size at maturity (Bullock and Smith, 1991).*

Due to low numbers of individuals observed, correlations with benthic features were not conducted.

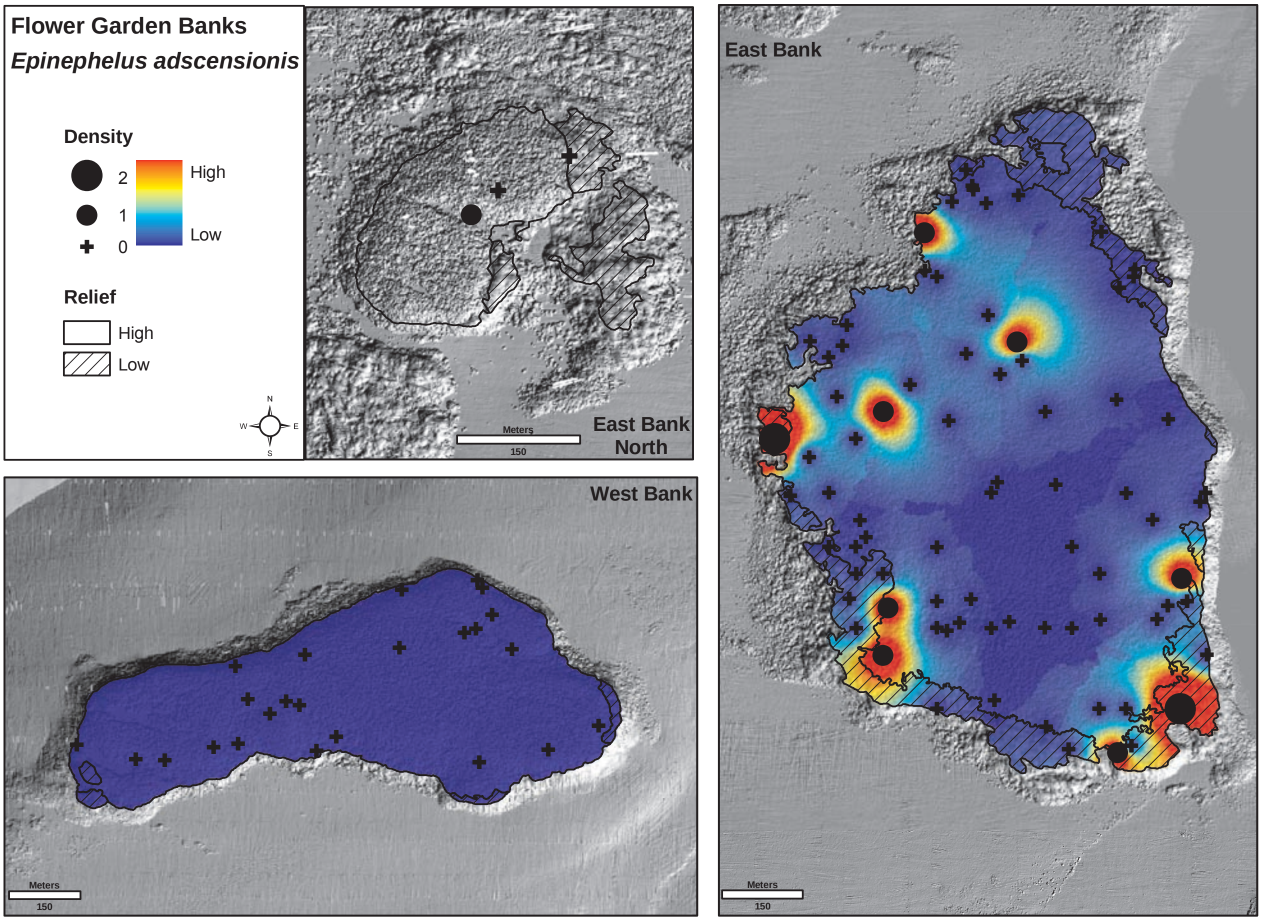


Figure 4.30. Observed (dots) and spatially interpolated density (#/100 m^2) for rock hind (E. adscensionis) observed in CCMA surveys for 2006-2007.

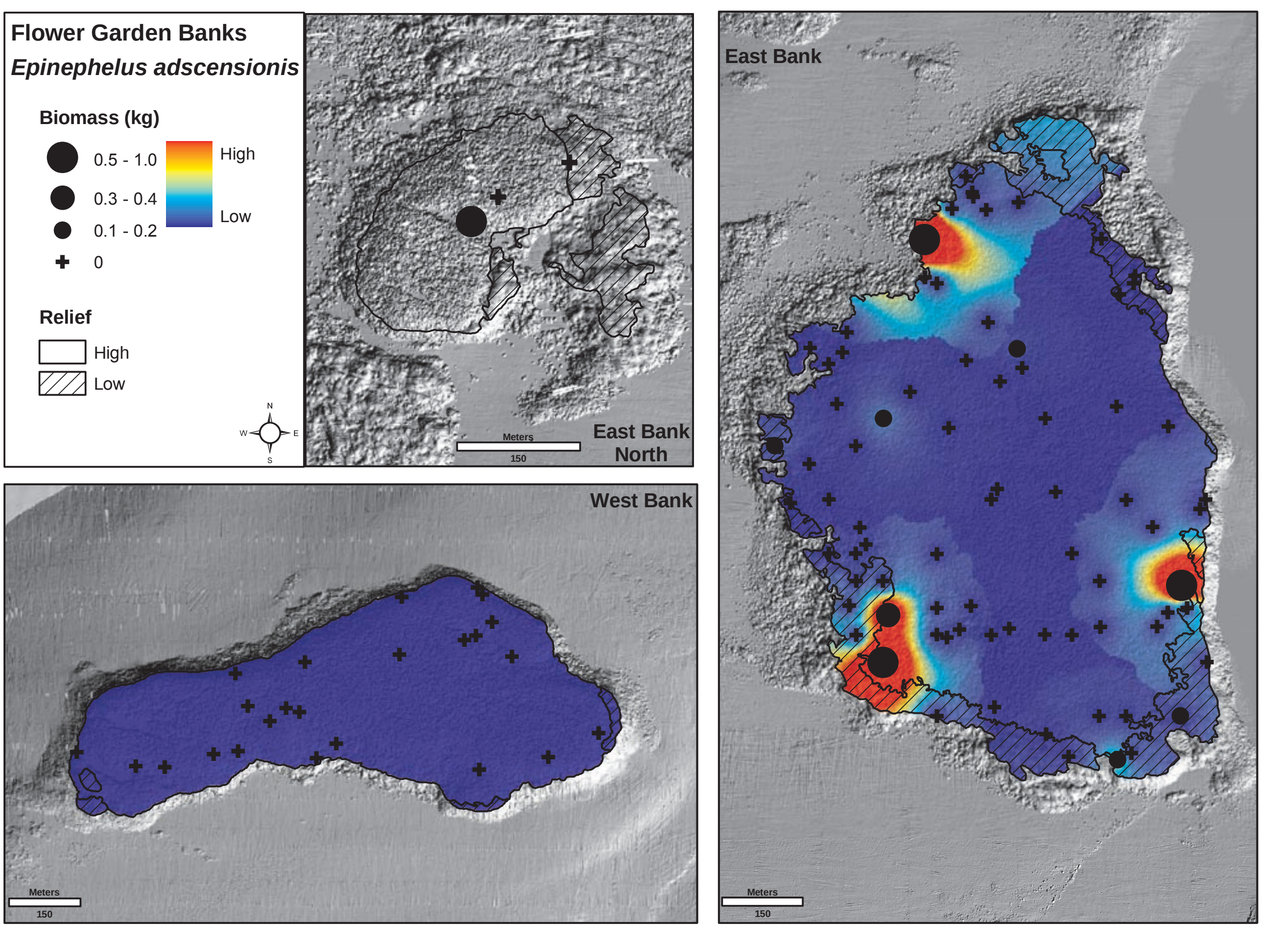


Figure 4.31. Observed (dots) and spatially interpolated biomass (kg) for rock hind (E. adscensionis) observed in CCMA surveys for 2006-2007.

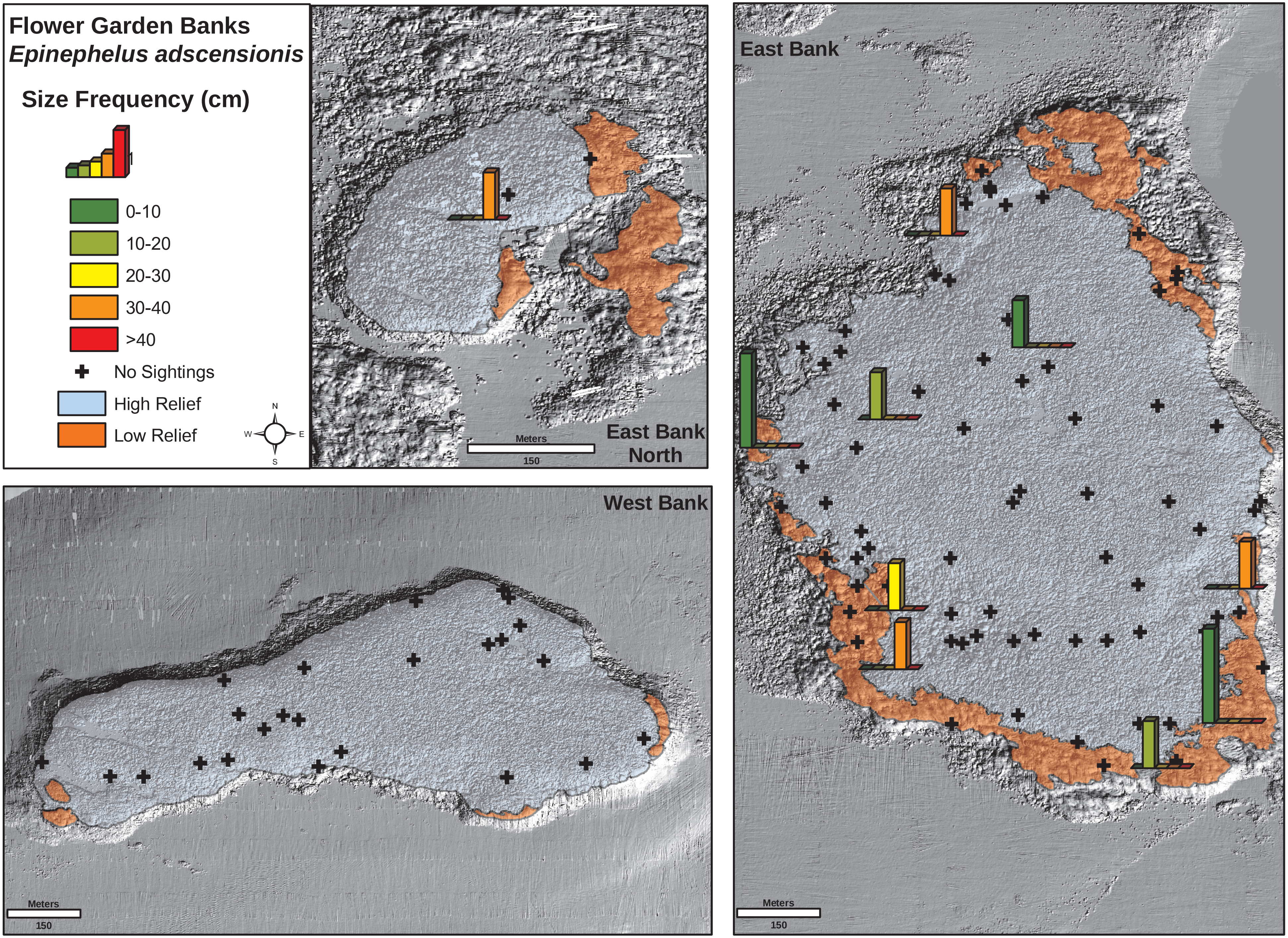


*Figure 4.32. Spatial distribution for rock hind (*E. adscensionis*) size frequency observed in CCMA surveys 2006-2007. The tallest histogram bar in the legend represents one individual.*

4.3.3.1.4 Red hind (*Epinephelus guttatus*)

E. guttatus exhibit similar habitat preferences as the rock hind. *E. guttatus* distribution extends from North Carolina to Venezuela, including the Gulf of Mexico and the Caribbean (Smith, 1997). *E. guttatus* are typically found over shallow reefs and rocky habitats at depths from 2-100 m (Froese and Pauly, 2007). *E. guttatus* exhibit faster growth than rock hind attaining maximum size of 76 cm TL and can weigh up to 25 kg (Heemstra and Randall, 1993). *E. guttatus* are considered moderately vulnerable to fishing pressure (Cheung et al., 2005). *E. guttatus* collected in Puerto Rico attain sexual maturity at approximately 21.5 cm FL (Sadovy et al., 1994) and the size at which 50% of individuals were mature for fish captured in Jamaica was 25 cm FL (Thompson and Munro, 1978). Although not as large as some other groupers, it is the most important species in the Caribbean grouper fishery (Heemstra and Randall, 1993) and contributes a minor component of grouper landings in the Gulf of Mexico (GOMFMC, 2004)

Epinephelus guttatus (CCMA)

Sighting frequency for *E. guttatus* was greater than that documented by REEF (Table 4.7). Again, this discrepancy is likely explained by the nature of the two methods where the transect method is more efficient to observe fish that tend to hide in reef crevices.

E. guttatus abundance was low during the study period (n=15) where adults and juveniles were found on high relief habitats (Figure 4.33). Mean density was higher on WB (0.24/100 m^2) compared to EB (0.08/100 m^2). High *E. guttatus* density and biomass on EB (Figures 4.34 and 4.35) was primarily observed on the south and eastern edge of the bank, while no specific pattern was observed on WB. While density was generally lower on EB, larger individuals predominated (Figure 4.36).

Due to the limited sighting frequency, habitat correlations were not conducted, but in general *E. guttatus* were only observed on high relief habitats associated with relatively high coral cover (>40%) and low macroalgae cover (<30%).

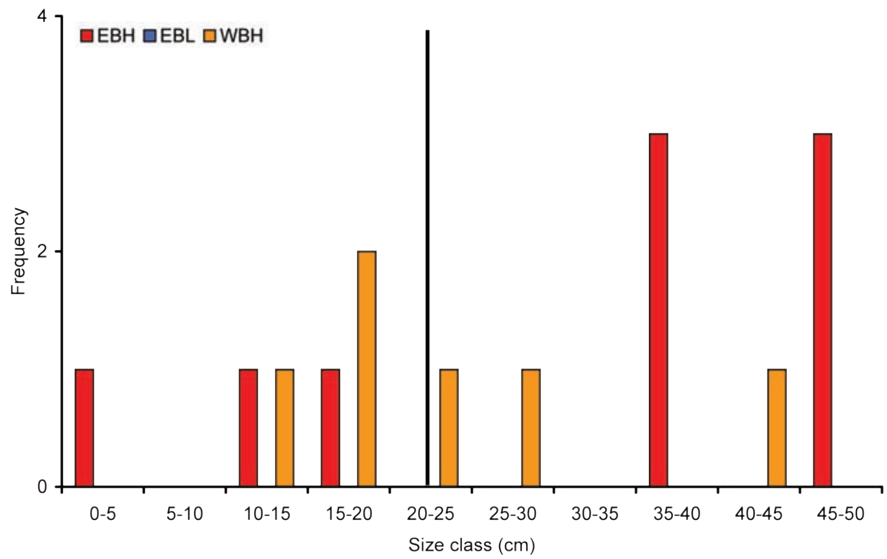


*Figure 4.33. Length frequency for red hind (*E. guttatus*) observed in CCMA surveys for 2006-2007. Vertical black line represents estimated size at maturity (Sadovy et al., 1994).*

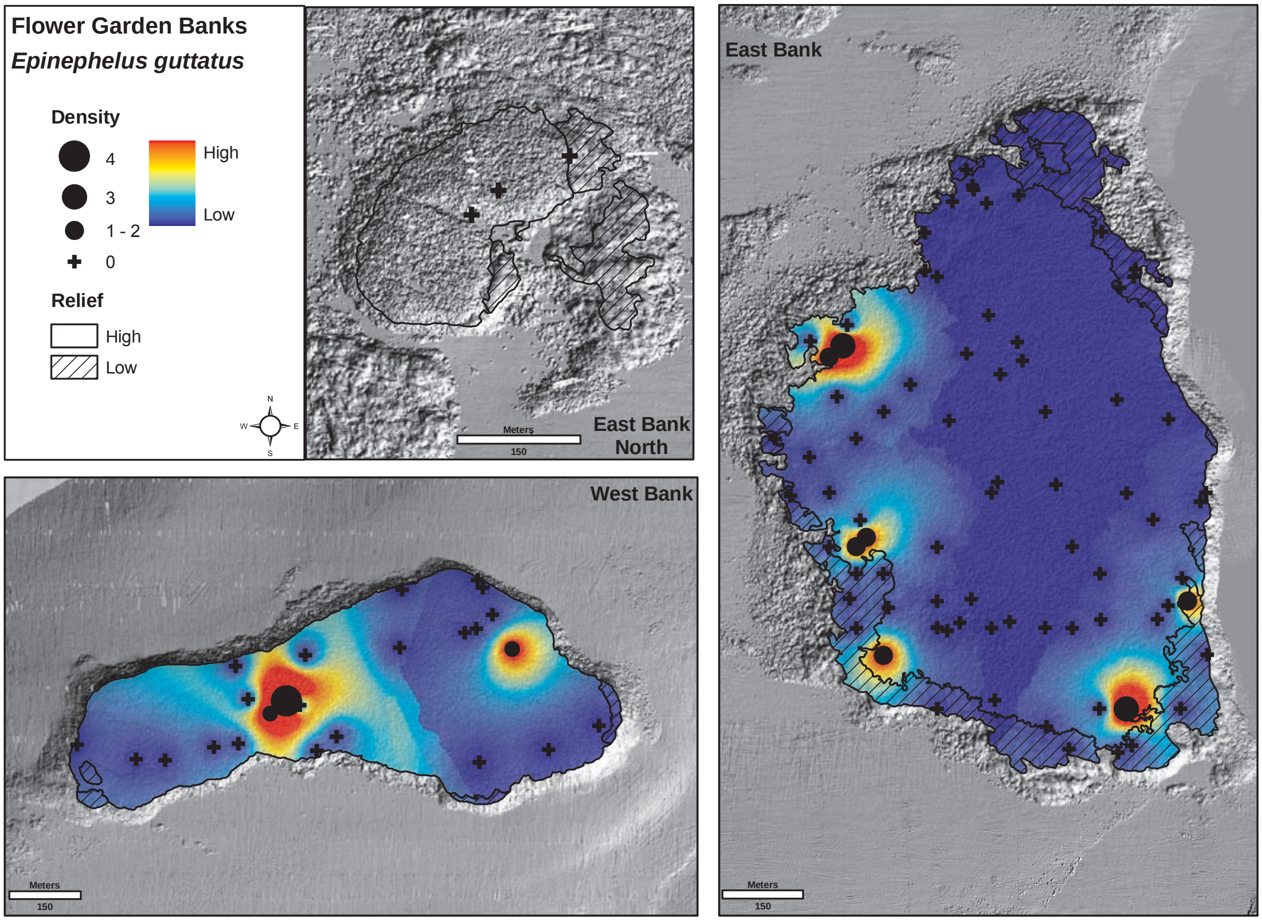


*Figure 4.34. Observed (dots) and spatially interpolated density (#/100 m²) for red hind (*E.guttatus*) observed in CCMA surveys for 2006-2007.*

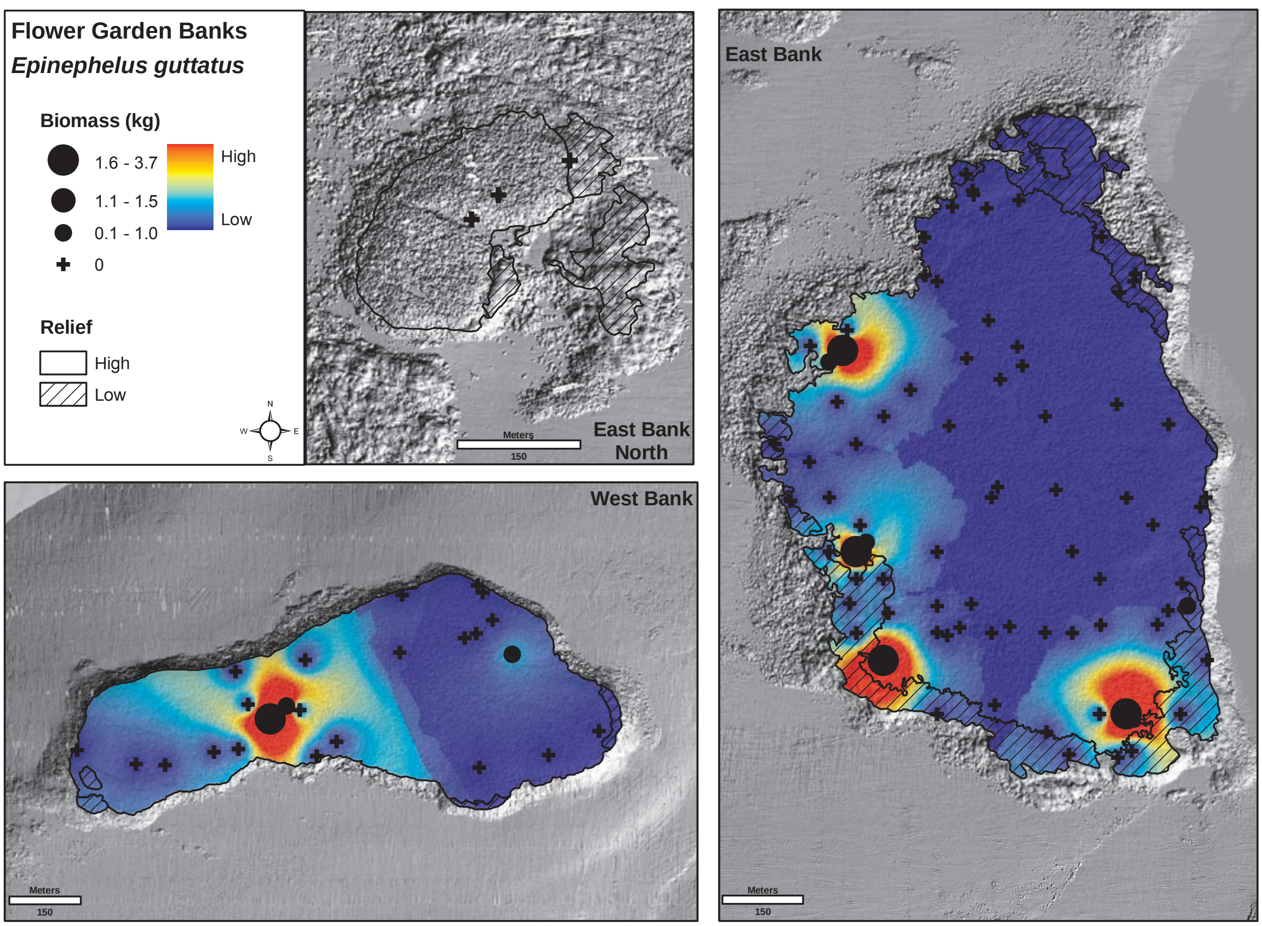


*Figure 4.35. Observed (dots) and spatially interpolated biomass (kg) for red hind (*E. guttatus*) observed in CCMA surveys for 2006-2007.*

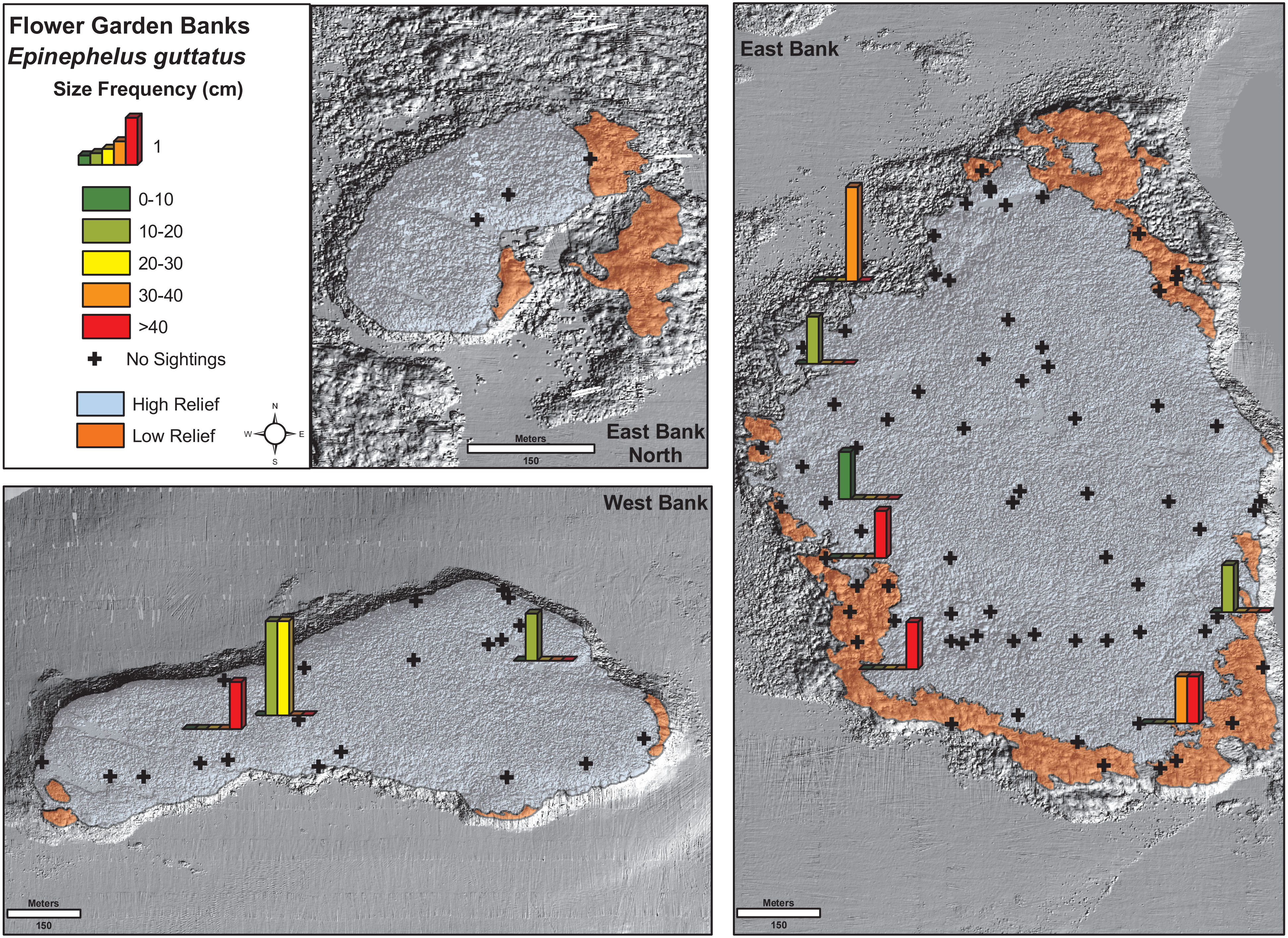
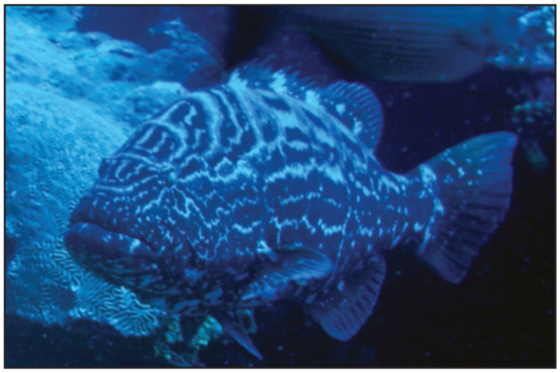


Figure 4.36. Spatial distribution for red hind (E. guttatus) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents one individual.

4.3.3.1.5 Black grouper (*Mycteroperca bonaci*)



Mycteroperca bonaci (CCMA)

M. bonaci range from New England to southeastern Brazil, including Bermuda, Florida, the Gulf of Mexico, the Bahamas and the Caribbean (Fischer, 1978; Böhlke and Chaplin, 1993). They are abundant in south Florida, the Florida Keys, Cuba and the Bahamas, but less common in the eastern Gulf of Mexico (Randall, 1968; Smith et al., 1975; Jory and Iversen, 1989). *M. bonaci* attain a maximum size of approximately 150 cm TL and 81 kg (Mowbray, 1950); however, most are caught at less than 70 cm and weigh less than 26 kg. They may live 33 years or longer (Crabtree and Bullock, 1998). Adults are found over hard bottoms such as coral reefs and rocky ledges and occur at depths of 9 to 30 m; maximum depth is approximately 100 m (Heemstra and Randall, 1993). Juveniles are typically found at shallower depths than adults (Bullock and Smith, 1991). *M. bonaci* exhibit fast growth throughout the first 10 years (Crabtree and Bullock, 1998) and slowing thereafter. Size of 50% maturity for females caught off the Yucatan, Mexico was 72.1 cm and 82.6 cm FL in Florida (Brule et al., 2003).

M. bonaci are the dominant commercial grouper species in the Florida Keys (SAFMC, 2005) and the second most commercial species (by pounds) in Texas waters during 2000-2006 (NMFS, unpublished data; http://www.st.nmfs.noaa.gov/st1/commercial/index.html). *M. bonaci* are important in hook and line and trap fisheries in the southern Gulf of Mexico, West Indies and the eastern coast of Venezuela (Heemstra and Randall, 1993). *M. bonaci* is federally managed under the GOMFMC's shallow water grouper complex and size limits for the commercial and recreational fisheries are 60.9 cm and 55.8 cm, respectively (GOMFMC, 2008a,b).

CCMA sighting frequency was nearly three times higher on EB and twice as high on WB than that reported by REEF during 1995-2005 (Table 4.7). *M. bonaci* sightings by CCMA were predominately located on the edges of banks, where REEF surveys were not conducted, which in combination with differing sampling methods, might account for the variability.

Most *M. bonaci* were singly observed throughout the sanctuary (Figure 4.37). All adults were observed on high relief habitats, while only juveniles were found across all relief types. Most *M. bonaci* were greater than commercial and recreational size limits. Overall, 21 *M. bonaci* were observed during the study period with a total biomass of 3,116 kg that accounts for 12% of the total biomass among all species observed during the surveys. Mean density was comparable between the two banks (0.22/100 m^2 on EB and 0.29/100 m^2 on WB). The majority of sightings occurred on high relief habitats near the edge of the coral caps (Figure 4.38).

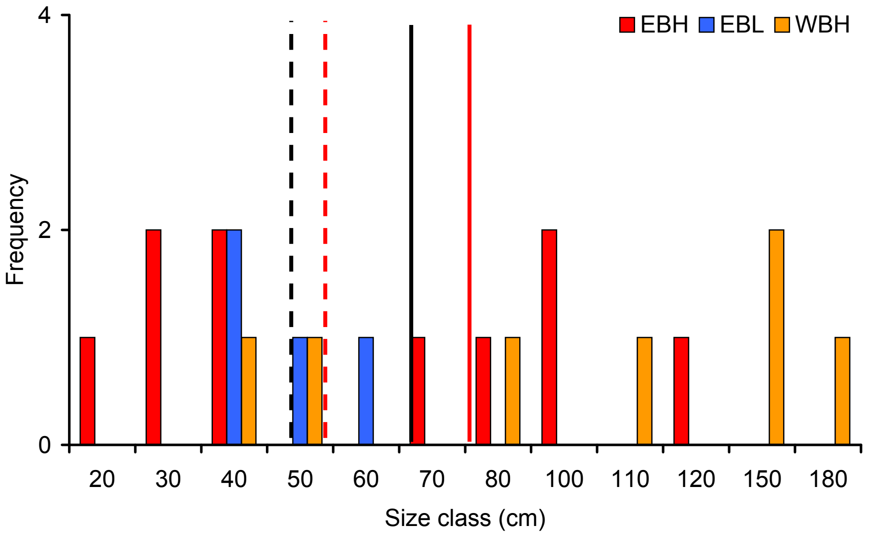


*Figure 4.37. Length frequency of black grouper (*M. bonaci*) observed in CCMA surveys for 2006-2007. Vertical solid black and red lines represents size of 50% maturity for females observed in the southern Gulf of Mexico and Florida, respectively (Brule et al., 2003). Dashed black and red lines represent the size limits for the recreational and commercial fisheries, respectively (GOMFMC, 2008a,b).*

While density was comparable among the banks, biomass was nearly three times higher on WB (Figure 4.39) due to three sightings of individuals 150 cm or greater (Figure 4.40). One fish was recorded at 175 cm FL, which is greater than the maximum size reported (IGFA, 2001). Density and biomass distribution were predominately located near the bank edge of EB, while no pattern was discernible on WB. Due to low sighting frequency, correlations with benthic features and depth were not conducted. Sites where *M. bonaci* were found were variable in coral (20-80%) and macroalgal (0-70%) cover.

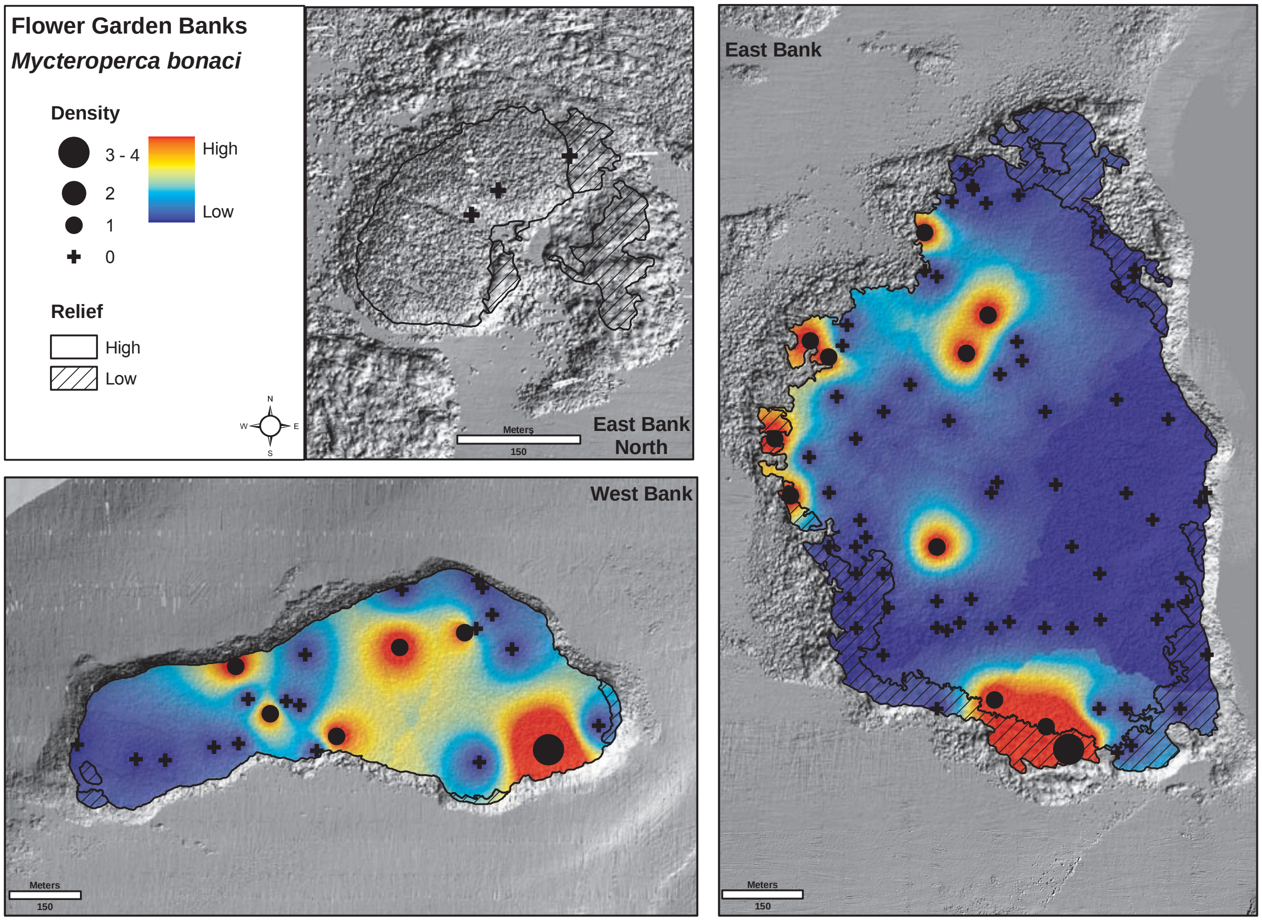


Figure 4.38. Observed (dots) and spatially interpolated density (#/100 m^2) for black grouper (M. bonaci) observed in CCMA surveys for 2006-2007.

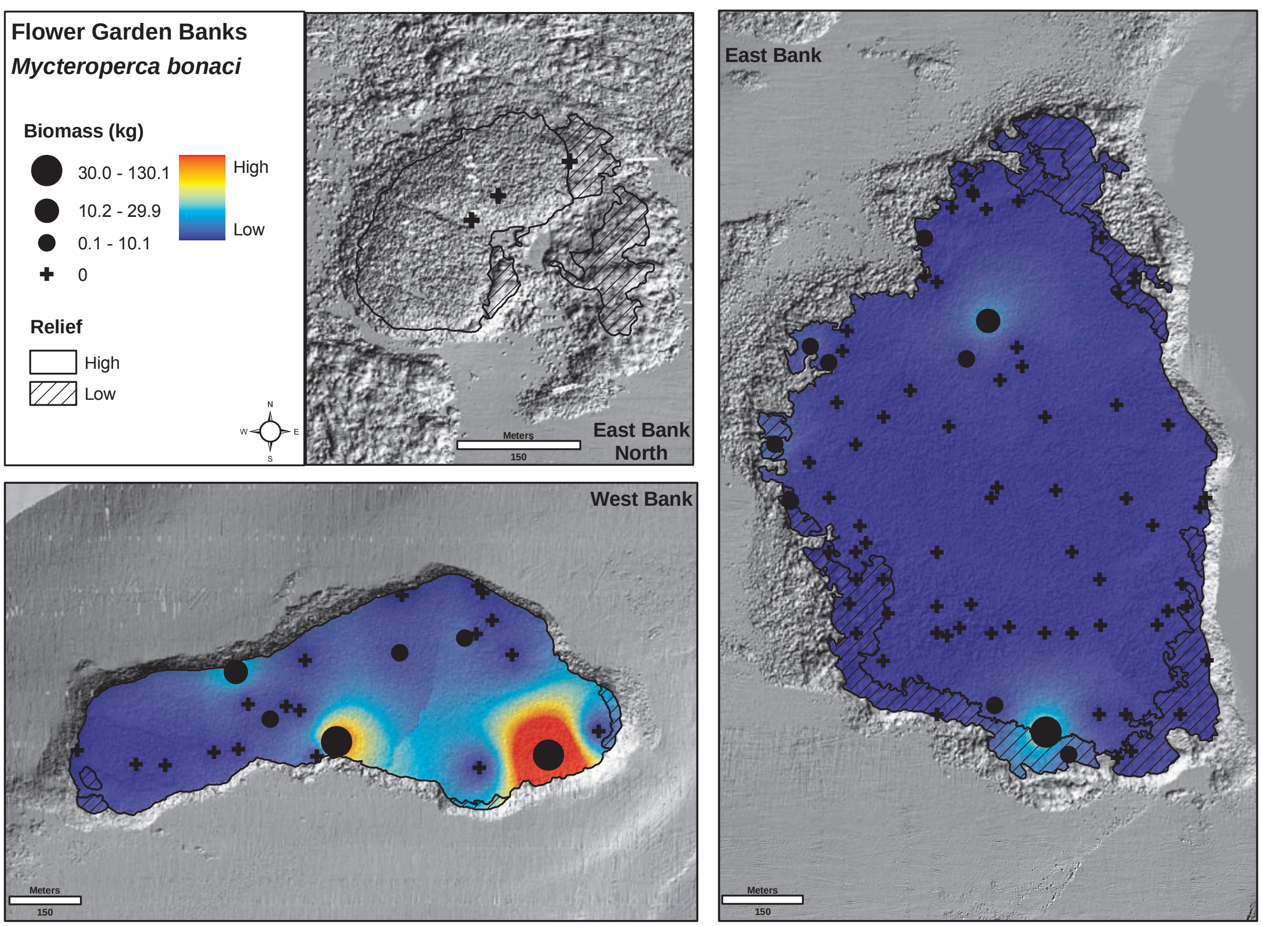


Figure 4.39. Observed (dots) and spatially interpolated biomass (kg) for black grouper (M. bonaci) observed in CCMA surveys for 2006-2007.

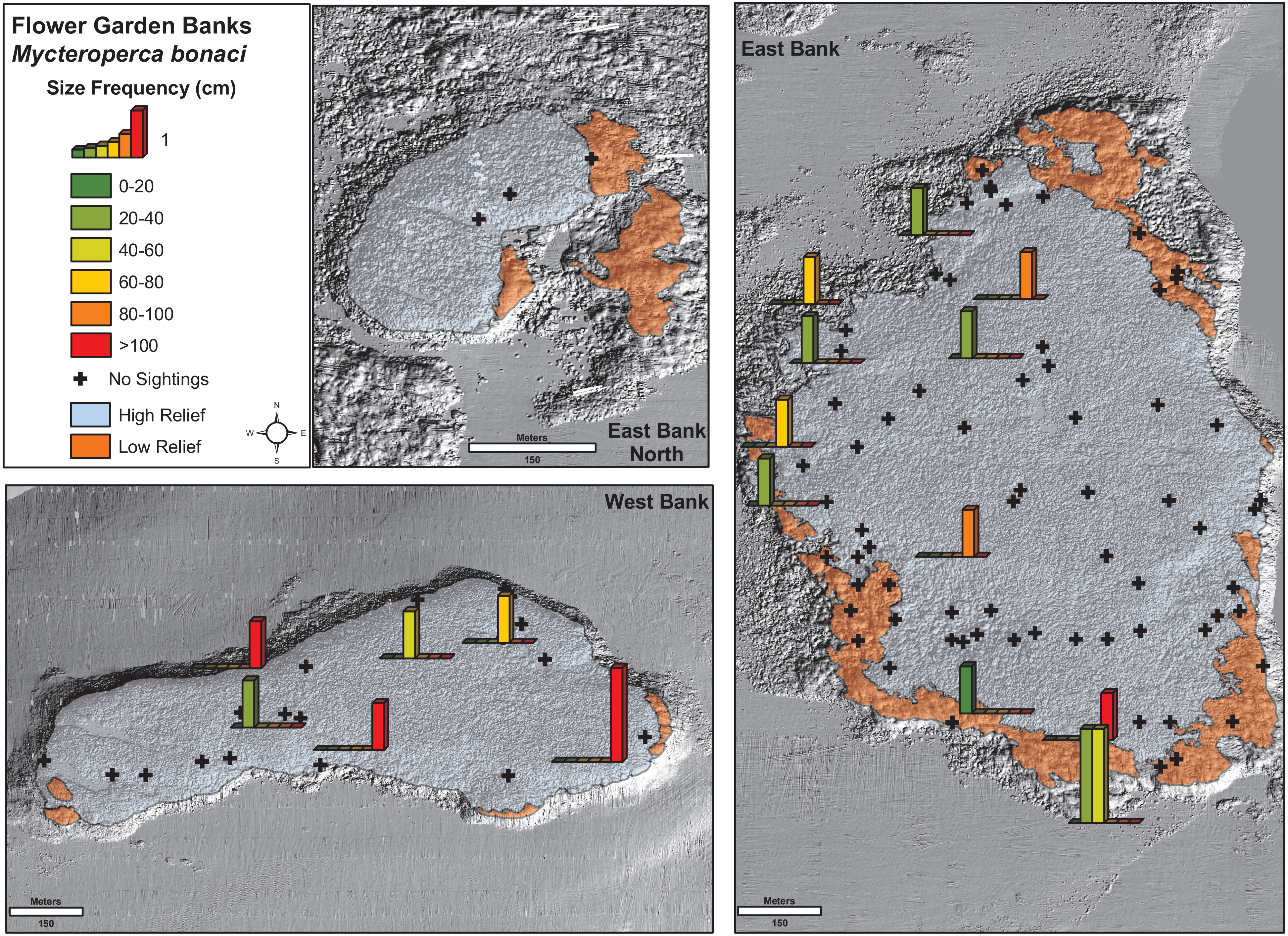
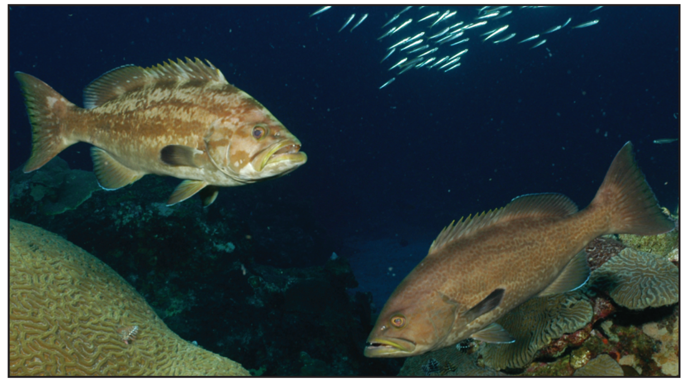


Figure 4.40. Spatial distribution for black grouper (M. bonaci) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents one individual.

4.3.3.1.6 Yellowmouth grouper (*Mycteroperca interstitialis*)

M. interstitialis range from the southeast U.S. through the Gulf of Mexico, Caribbean and West Indies to southern Brazil (Heemstra and Randall, 1993). Adults are commonly found over rocky hard bottom and coral reefs near the shoreline to depths of 55 m. Individuals have been found as deep as 150 m (SAFMC, 2005). *M. interstitialis* attain maximum size at approximately 84 cm and can weigh as much as 10.2 kg (Froese and Pauly, 2007). Females become mature at 40-45 cm TL and sexual transition occurs from 50.3 to 64.3 cm and (SAFMC, 2005). Heemstra and Randall (1993) state that the yellowmouth is an important component of the Gulf and Caribbean grouper fishery; however, detailed landings data in the Gulf of Mexico is limited. Yellowmouth are federally managed in the GOMFMC shallow-water grouper complex; however, no size limit exists for either commercial or recreational fisheries (GOMFMC, 2008a,b).



Color variations of *Mycteroperca interstitialis* (G.P. Schmahl)

Juvenile *Mycteroperca interstitialis* (CCMA)

Sighting frequencies of *M. interstitialis* were variable between surveys (Table 4.7). REEF frequencies were higher on EB, while higher frequency was observed on WB during 2006-2007. *M. interstitialis* were the most frequently sighted grouper at FGBNMS for both surveys. FGBNMS LTM surveys only documented *M. interstitialis* on EB.

Most of the individuals were not sexually mature, based on reported size at maturity data (Figure 4.41). During the study period, 108 *M. interstitialis* were observed and no significant differences were observed between density and bank/habitat type. Density was also comparable by bank comparison: 1.03/100 m^2 on EB, 1.30/100 m^2 on WB. Overall, *M. interstitialis* exhibited the highest density among the grouper species observed on the SCC (1.12/100 m^2). Yellowmouth density (Figure 4.42) and biomass (Figure 4.43) were greater on the edges of both banks, while fewer individuals were observed in the central portions of the banks. Biomass was significantly lower on low relief habitats (p=0.04), as larger individuals and greater density was observed on high relief habitats on both banks (Figure 4.44). Neither density nor biomass was correlated with coral or algal cover or depth. Density was inversely correlated with the spatial cover of crustose coralline algae (ρ = -0.20) and *Ma. mirabilis* (ρ = -0.28) both of which exhibited significantly higher cover on low relief habitats than high relief habitats.

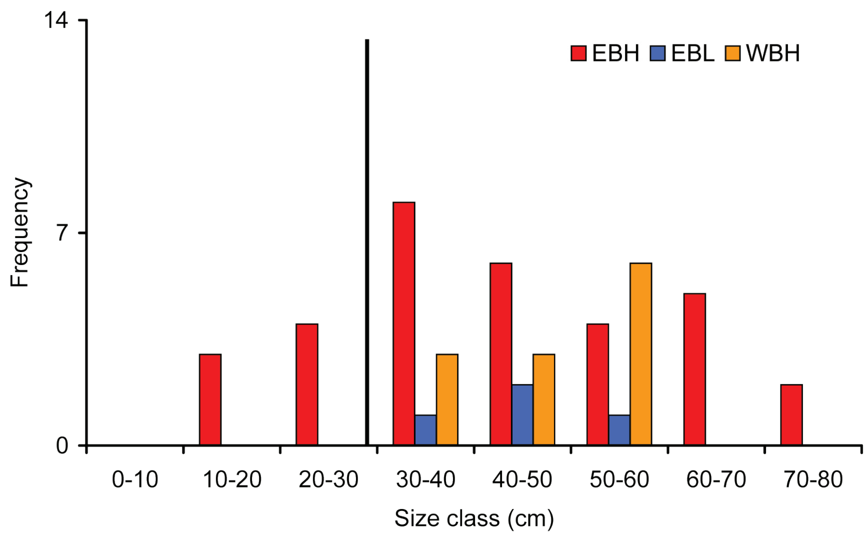


Figure 4.41. Length frequency for yellowmouth grouper (M. interstitialis*) observed in CCMA surveys for 2006-2007. Vertical solid black represents estimated size at maturity (SAFMC, 2005).*

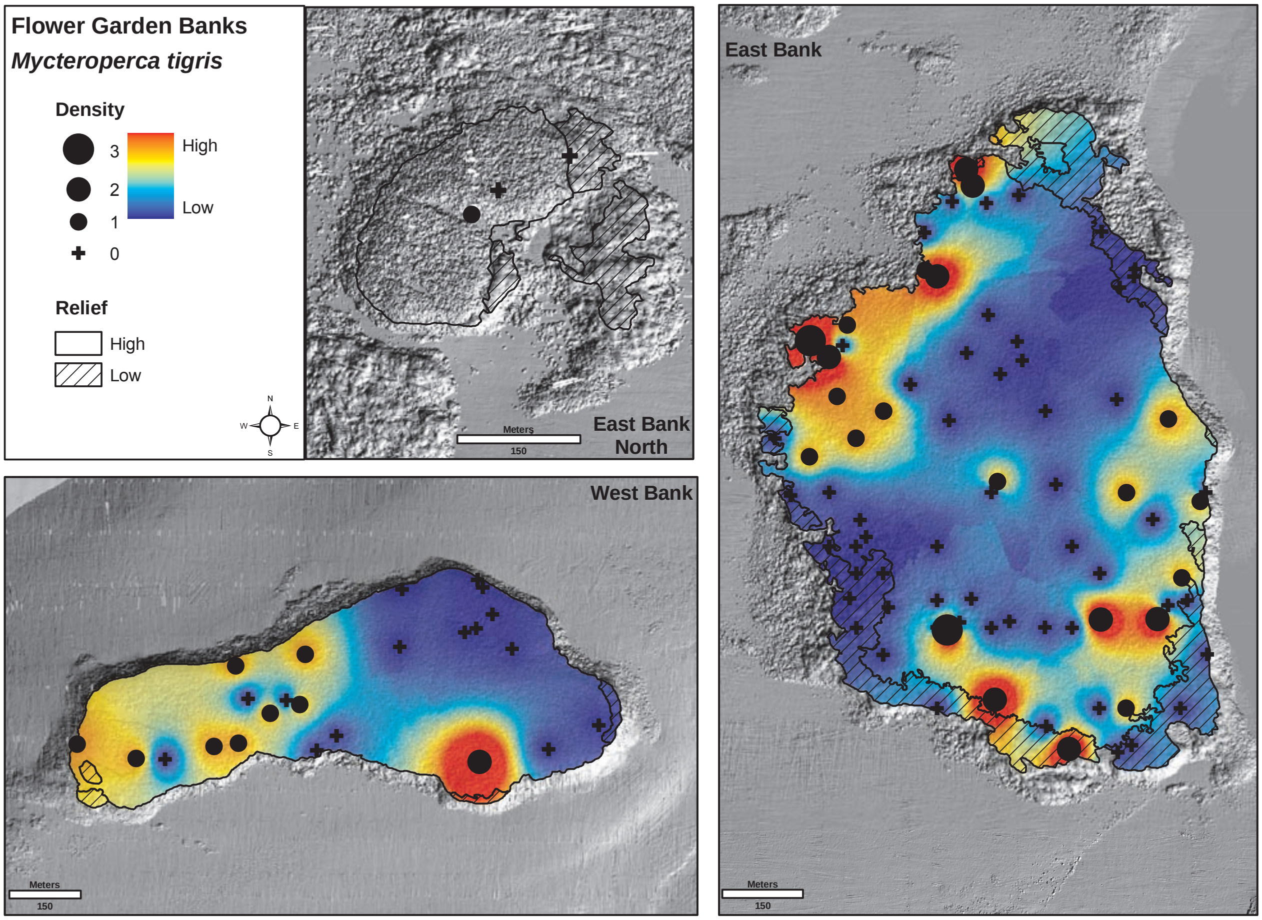


*Figure 4.42. Observed (dots) and spatially interpolated density (#/100 m^2) for yellowmouth grouper (*M. interstitialis*) observed in CCMA surveys for 2006-2007.*

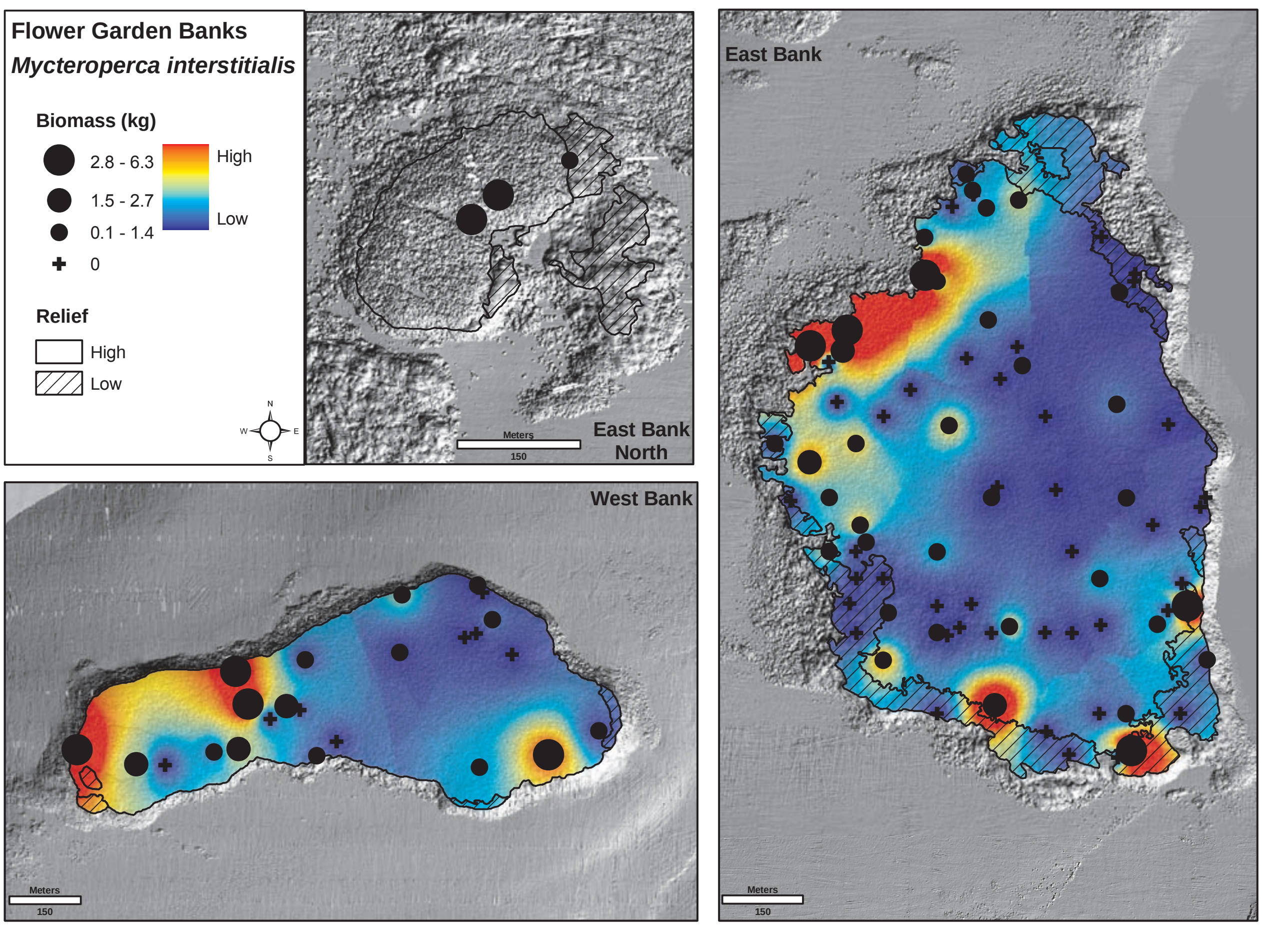


*Figure 4.43. Observed (dots) and spatially interpolated biomass (kg) for yellowmouth grouper (*M. interstitialis*) observed in CCMA surveys for 2006-2007.*

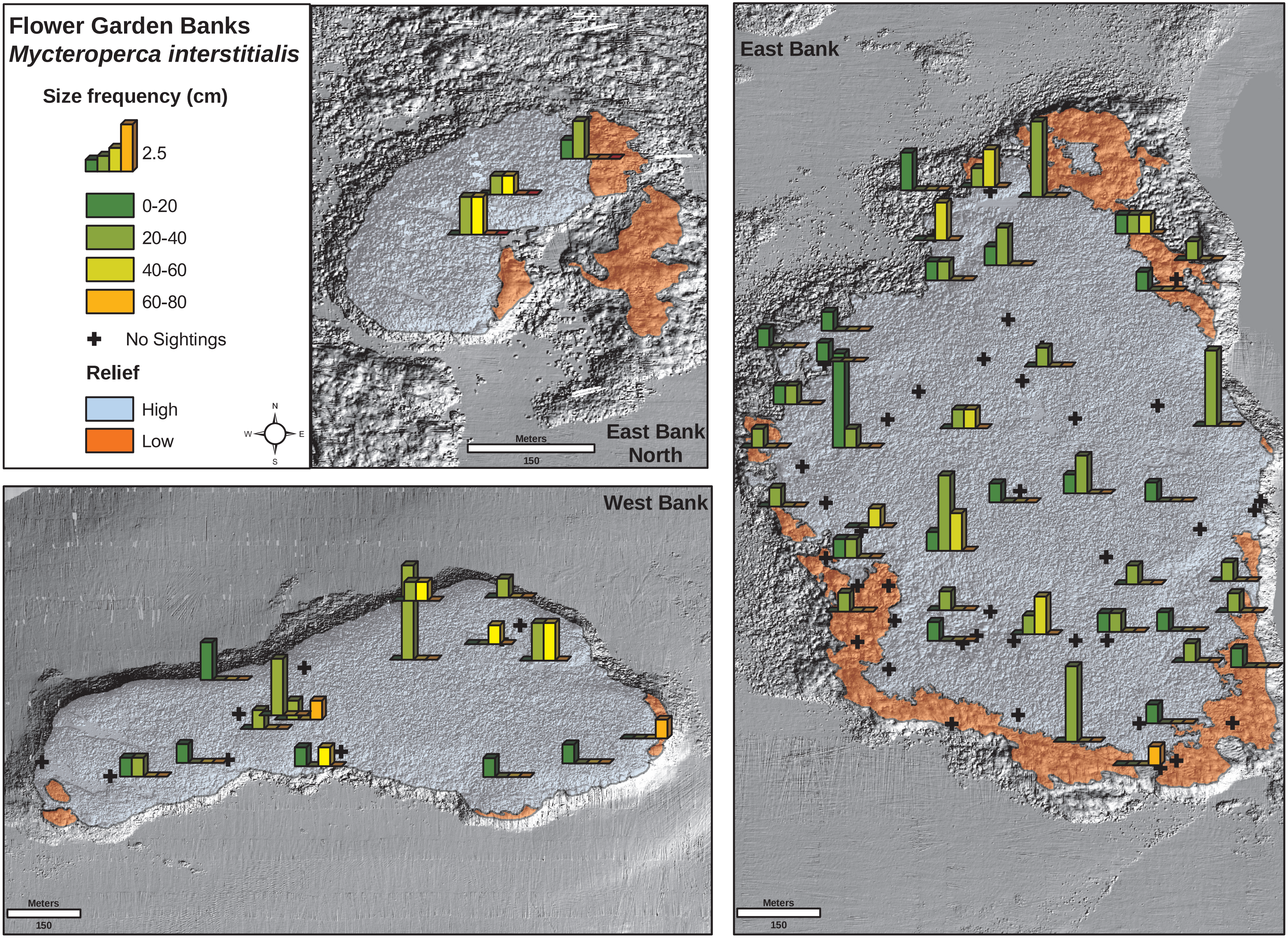
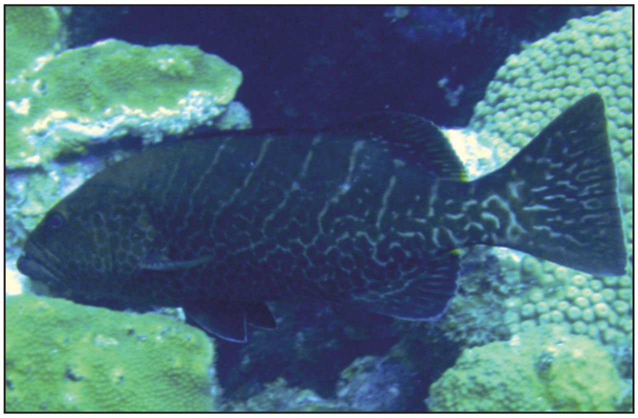


*Figure 4.44. Spatial distribution for yellowmouth grouper (*M. interstitialis*) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 2.5 individuals.*

4.3.3.1.7 Tiger grouper (*Mycteroperca tigris*)



Mycteroperca tigris (CCMA)

M. tigris are large-bodied serranids attaining a maximum size of 100 cm and capable of weighing 10 kg (Heemstra and Randall, 1993). *M. tigris* are found on coral reefs and rocky substrates at depths between 10-40 m from Bermuda, south Florida, Gulf of Mexico, Caribbean/West Indies, and oceanic islands off Venezuela and Brazil (Heemstra and Randall, 1993). Fish collected from Bermuda and Puerto Rico indicate that maturity is attained at 25 cm standard length (SL) while sex transition occurs between 37-45 cm and (28.5 cm FL; Sadovy et al., 1994). *M. tigris* abundance has been documented to be common throughout its range but commercial and recreational catches are high at sights of spawning aggregations (Matos and Posada, 1998). As most individuals caught during such aggregations are mature (Matos and Padilla, 1995), their large-scale removal is severely detrimental to the reproductive potential of the species. *M. tigris* are commercially important in Bermuda and the Caribbean, although population status and commercial landings data are lacking. In U.S. waters, *M. tigris* are managed in the shallow-water grouper complex by the South Atlantic Fishery Management Council (SAFMC) and GOMFMC (SAFMC, 2005; GOMFMC, 2005) and in the Reef Fish Fishery Management Plan in the U.S. Caribbean (CFMC, 1985).

M. tigris sighting frequency was considerably lower than REEF estimates for EB and slightly lower on WB (Table 4.7). Despite this, *M. tigris* were the second most abundant grouper species among both surveys.

Most individuals observed were adults based on estimated size at maturity information (Figure 4.45). Few juveniles were observed and only occurred on EBH habitats. *M. tigris* density (Figure 4.46) was comparable between the two banks, EB (0.56/100 m^2) and WB (0.41/100 m^2). Both density and biomass were significantly greater (p=0.0036, p=0.0067, respectively) on EBH habitats than WBH habitats (Figures 4.46 and 4.47). This is largely reflected in the size structure of fish observed on each bank. Significantly greater numbers of fish (p<0.0001) were observed in all size classes on EBH (Figure 4.48), in particular those greater than 60 cm.

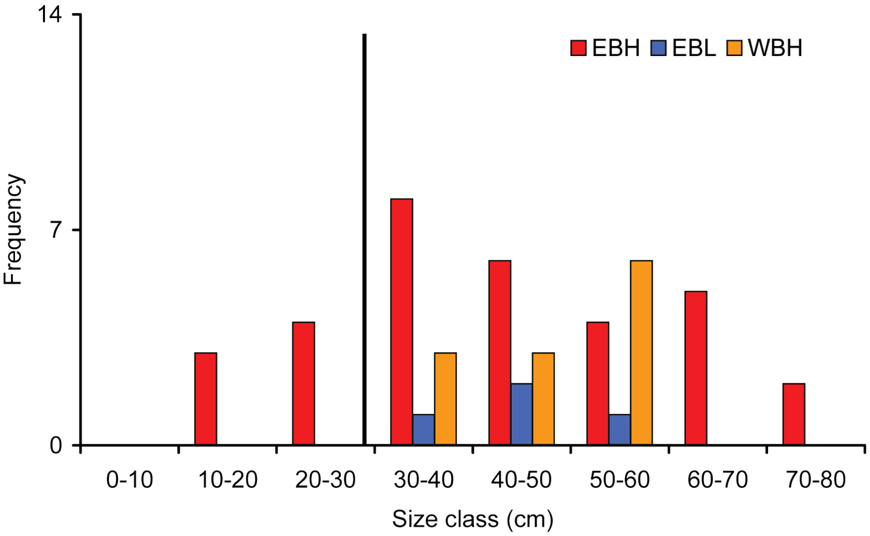


*Figure 4.45. Length frequency of tiger grouper (*M. tigris*) observed in CCMA surveys for 2006-2007. Vertical solid black represents estimated size at maturity (Heemstra and Randall, 1993).*

Density and biomass (Figures 4.46 and 4.47) were positively correlated with total coral cover (ρ = 0.19, ρ = 0.20, respectively) but not correlated with macroalgal cover or depth. More specifically, density and biomass were positively correlated with *Mo. franksi* (ρ = 0.26 and ρ = 0.27, respectively) the dominant coral on high relief habitats.

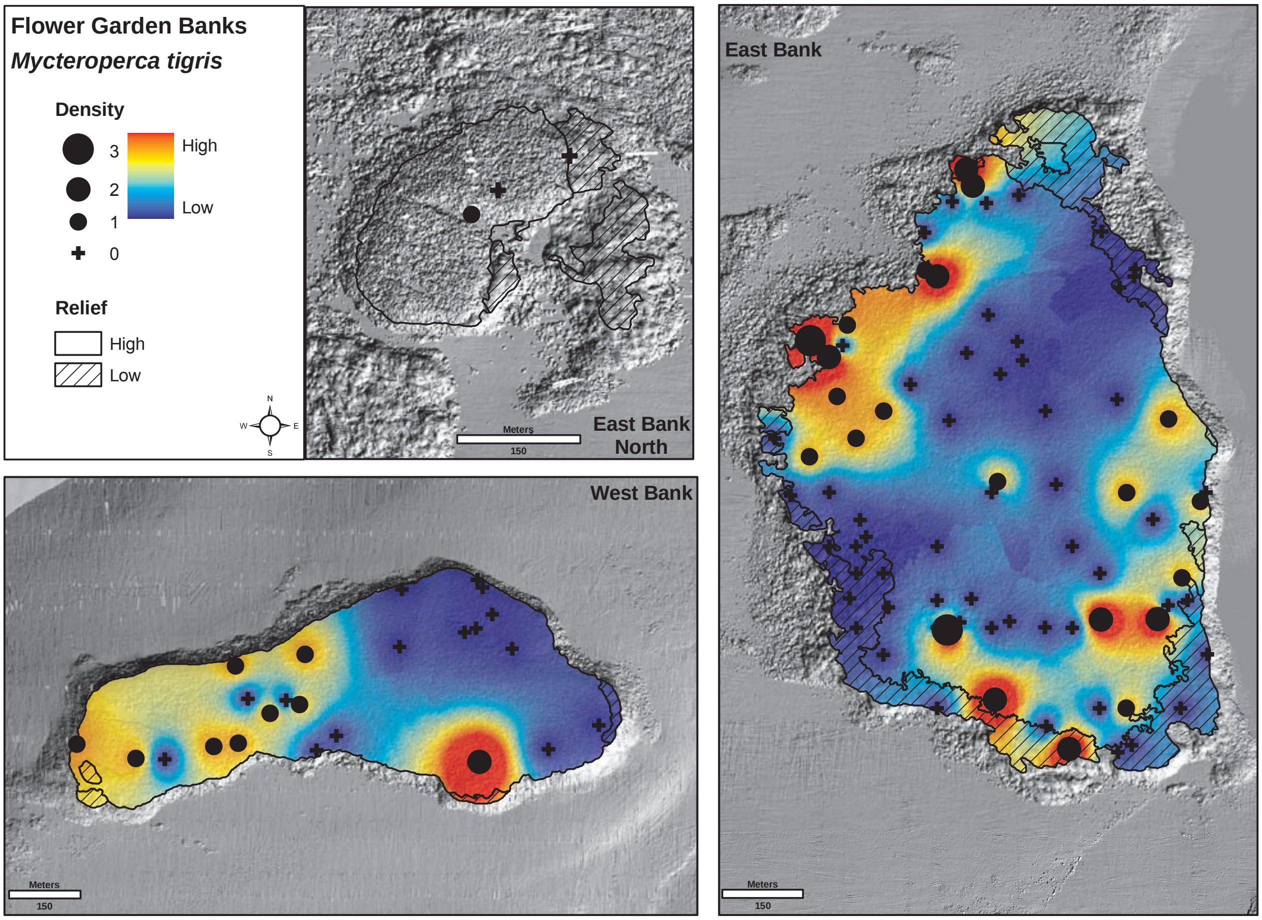


Figure 4.46. Observed (dots) and spatially interpolated density (#/100 m^2) for tiger grouper (M. tigris) observed in CCMA surveys for 2006-2007.

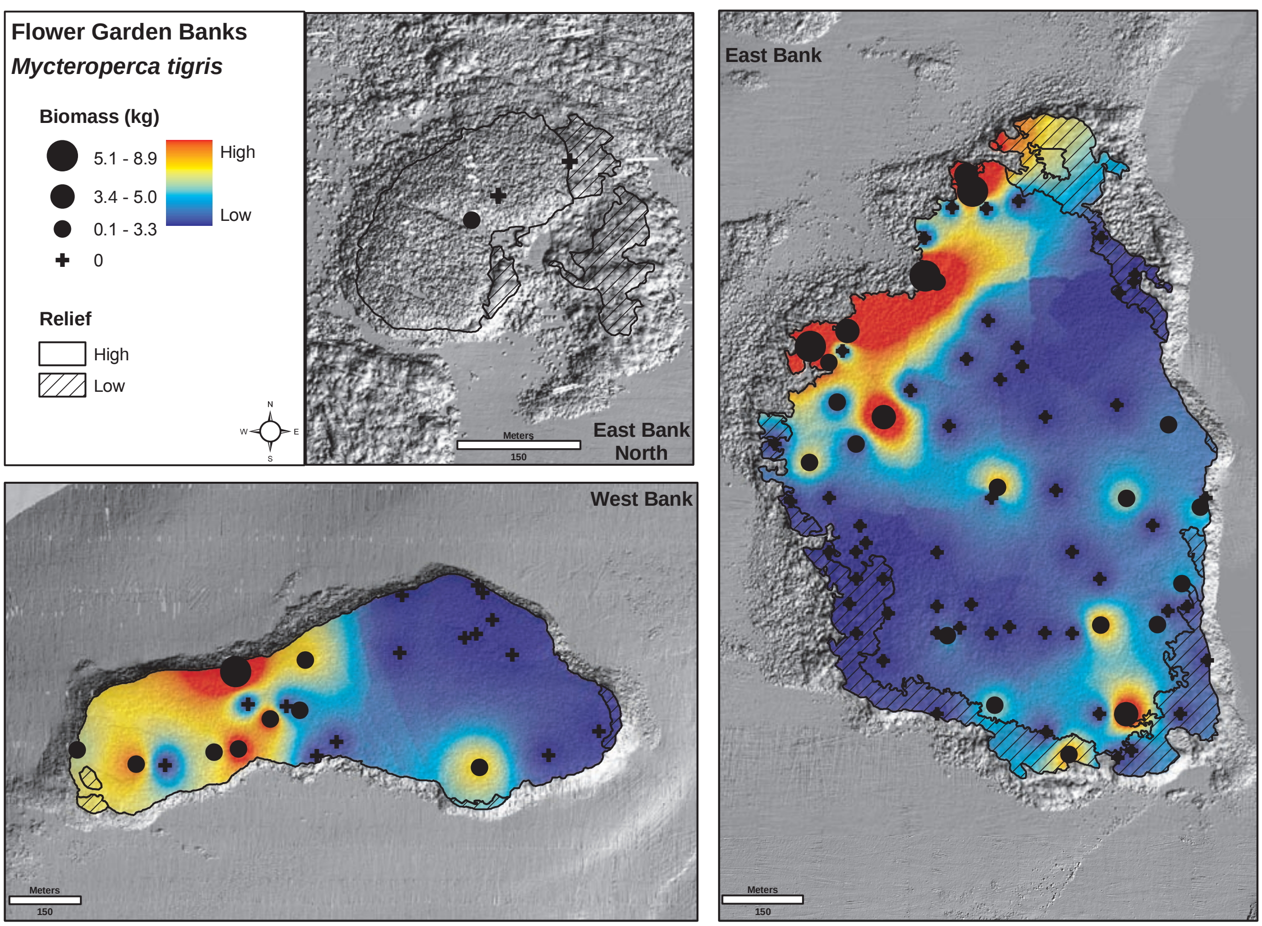


Figure 4.47. Observed (dots) and spatially interpolated biomass (kg) for tiger grouper (M. tigris) observed in CCMA surveys for 2006-2007.

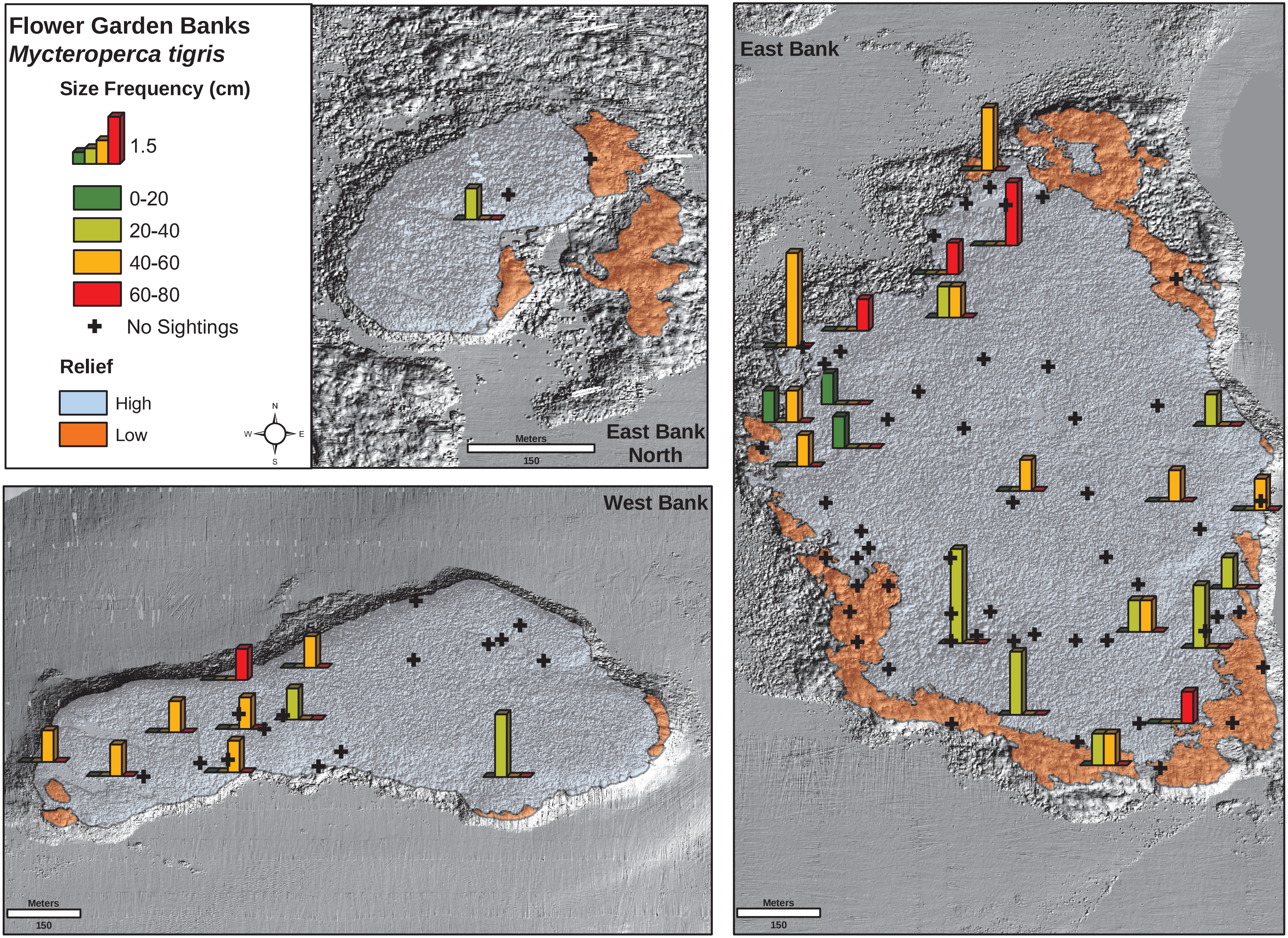


Figure 4.48. Spatial distribution for tiger grouper (M. tigris) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 1.5 individuals.

4.3.3.1.8 Atlantic creolefish (*Paranthias furcifer*)

The *P. furcifer* is a subtropical fish whose distribution is found at depths between 10-64 m (FAO) throughout the Western Atlantic from Bermuda to Brazil (Heemstra and Randall, 1993). It is an abundant zooplanktivore whose numerical dominance has been documented on FGBNMS coral caps by REEF and MMS (Table 4.4). Because of its small size, there is not much of a commercial or recreational fishery for this species; however, they are a preferred baitfish for other fisheries (Heemstra and Randall, 1993). Information regarding size at sexual maturity is scarce; however, Posada-Lopez and Appeldoorn (1996) noted minimum size of sexual maturity at approximately 14 cm in southwestern Puerto Rico.

Adult *Paranthias furcifer* (larger fish) and initial phase *Bodianus rufus* (Burek)

P. furcifer sighting frequency was comparable to that of REEF with slightly lower values on WB (Table 4.7). Density values from both surveys ranked in the top three for all species observed on both banks (Tables 4.4 and 4.5).

Adults were observed predominantly on EBH, while juveniles were found on all bank/habitat type combinations (Figure 4.49). Density ranked second among all fish observed on EB (48.53/100 m^2) and third on WB (32.57/100 m^2); however, spatial patterns were patchy (Figure 4.50). Distribution patterns show that *P. furcifer* were observed throughout the sanctuary, but with areas of peak density near the coral cap margins on both banks. Density was not significantly different among bank/habitat types, but the general pattern displayed higher density on EB.

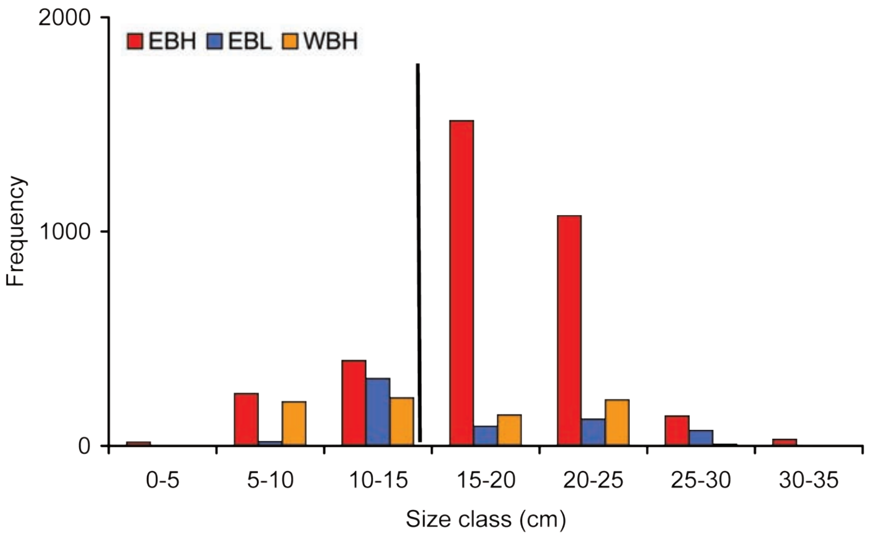


*Figure 4.49. Length frequency of Atlantic creolefish (*P. furcifer*) observed in CCMA surveys for 2006-2007. Vertical black line represents size at maturity (Posada-Lopez and Appeldoorn, 1996). EBH=East Bank High relief; EBL=East Bank Low relief; WBH=West Bank High relief.*

P. furcifer ranked first in mean biomass (3.31 kg/100 m^2) on EBH which was significantly greater ($p=0.019$) on high relief habitats than WBH (0.87 kg/100 m^2). Biomass on low relief was not significantly different from high relief on either bank. Biomass followed the same spatial pattern as density where higher values were observed on the margins of the coral caps (Figure 4.51). Examination of the spatial distribution of length frequency indicate that larger fish were found on EB compared to WB (Figure 4.52).

Both density and biomass were not correlated with coral cover, macroalgal cover or depth.

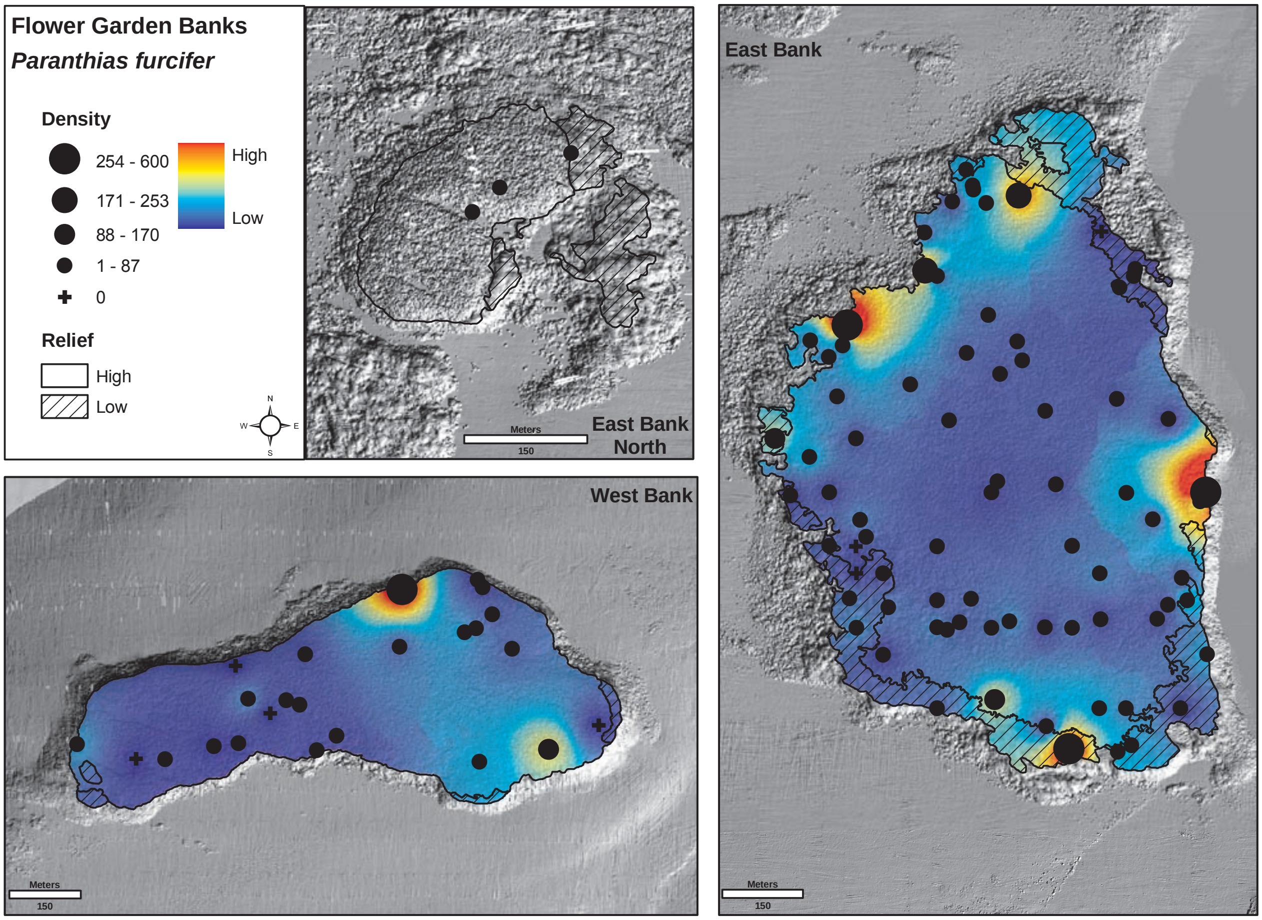


*Figure 4.50. Observed (dots) and spatially interpolated density (#/100 m²) for Atlantic creolefish (*P. furcifer*) observed in CCMA surveys for 2006-2007.*

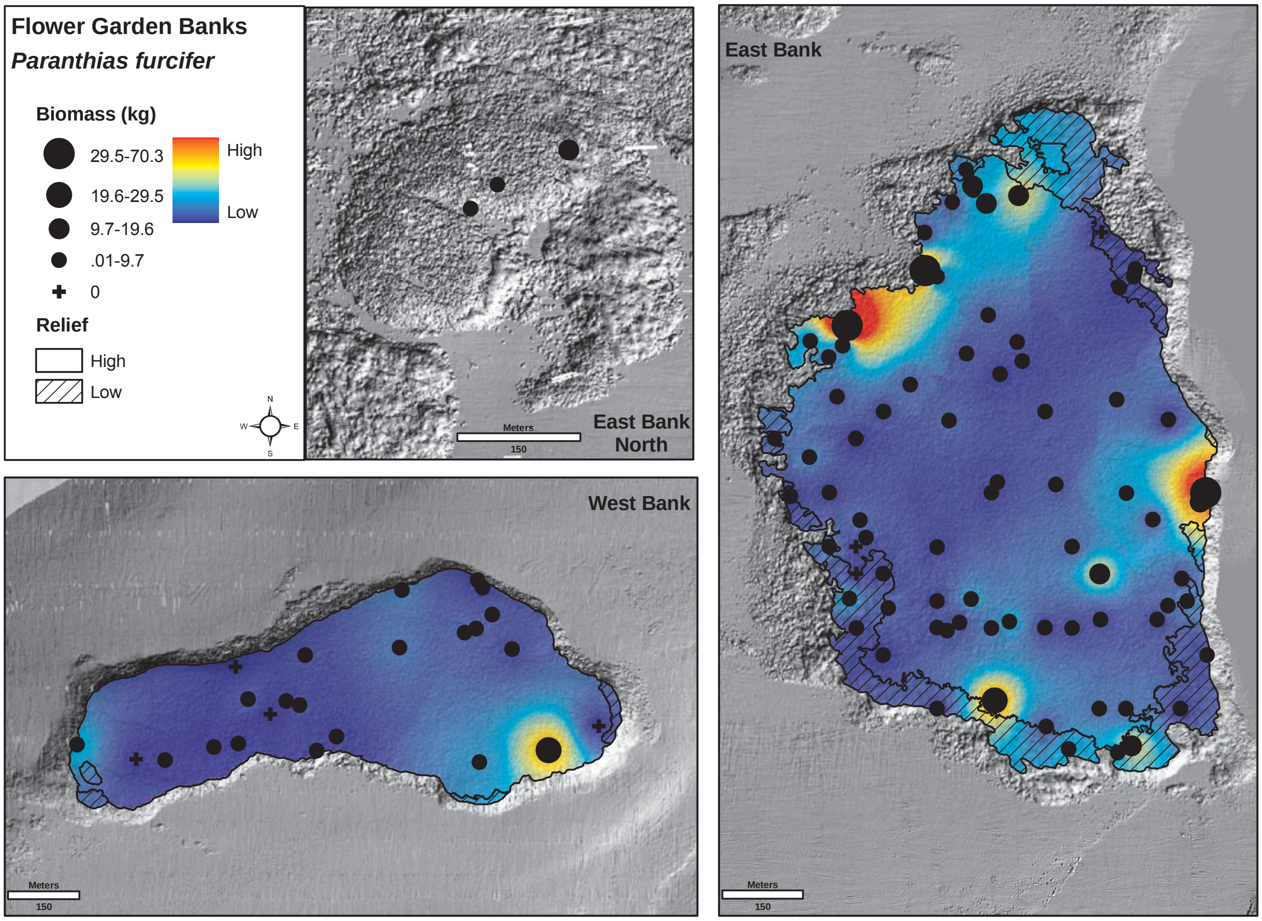


*Figure 4.51. Observed (dots) and spatially interpolated biomass (kg) for Atlantic creolefish (*P. furcifer*) observed in CCMA surveys for 2006-2007.*

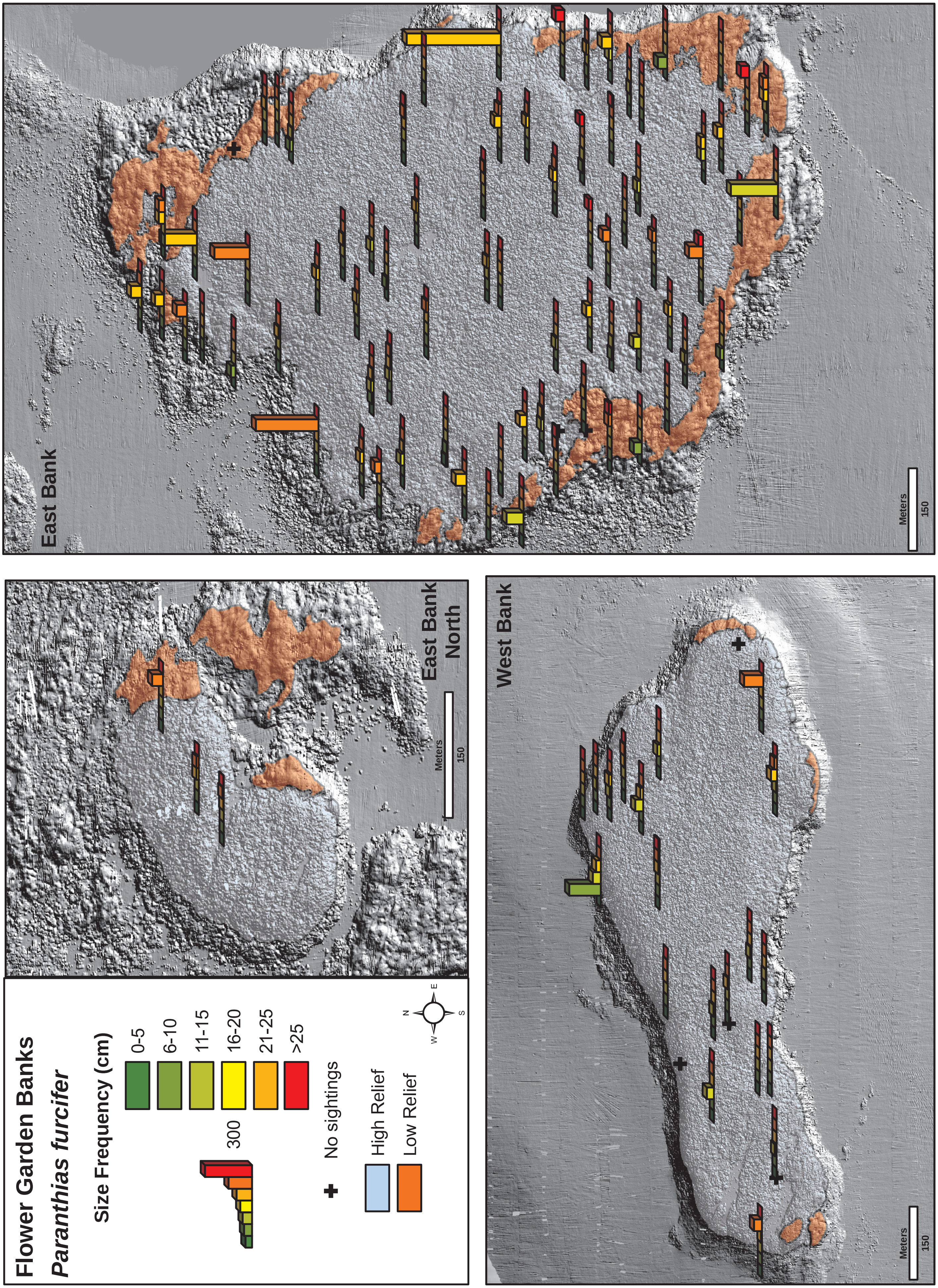
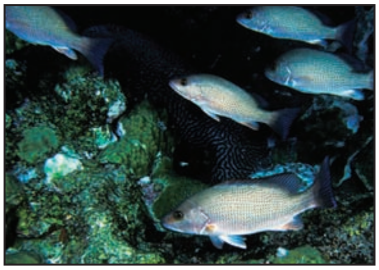


Figure 4.52. Spatial distribution of Atlantic creolefish (P. furcifer) size frequency observed in CCMA surveys for 2006-2007. The tallest bar in the legend histogram represents 300 individuals.

4.3.3.2 Lutjanidae (*Snappers*)



Lutjanus grisus (Burek)

Snappers of the genus Lutjanus are common inhabitants of coral reefs and rocky substrates in the Gulf of Mexico and Caribbean and are keystone species in coral reef ecosystems (Parrish, 1987). Snappers are generally slow-growing and moderately long-lived (CFMC, 1985) thus populations can be sensitive to fishing and habitat disturbances.

All species have complex life histories, with most dependent on different habitats during the egg, larval, juvenile and adult phases of their life cycle. Eggs and early larvae are typically pelagic. No long-lived oceanic larval or post-larval phases have been reported for snappers, as have been reported for many other reef fish families. Thus, they probably have a relatively short planktonic larval or post-larval life (Thompson and Munro, 1974a). Larvae settle into various nearshore nursery habitats such as seagrass beds, mangroves, oyster reefs and marshes (Coleman et al., 2000). These habitats are noticeably absent from FGBNMS and snapper recruitment into the sanctuary is unknown. Adults are generally sedentary and residential. Movement is generally localized and exhibits an offshore-inshore pattern, usually associated with spawning events. Many species have been reported to form mass spawning aggregations, where hundreds or even thousands of fish convene to reproduce (Rielinger, 1999). Snapper movement at FGBNMS is currently unknown.

Snappers are important to artisanal fisheries, but seldom the prime interest of major commercial fishing activities; many are fine foodfishes, frequently found in markets. The species that reach large sizes are important recreational fishes in some areas (Coleman et al., 2000).

Table 4.8. Sighting frequency of select snapper (Lutjanidae) species from CCMA, REEF and MMS surveys. REEF estimates (means) are from expert surveys only. MMS data only reflect presence/absence indicated by +/-.

Species	East Bank CCMA	East Bank REEF	East Bank MMS	West Bank CCMA	West Bank REEF	West Bank MMS
Lutjanus analis	1.23	0	-	4.16	0.14	-
Lutjanus buccanella	0	0.8	-	0	0.8	-
Lutjanus cyanopterus	1.23	0	-	4.16	0	-
Lutjanus griseus	16.1	33.95	-	41.67	33.95	+
Lutjanus jocu	22.22	21.20	+	25	21.2	+
Lutjanus mahogoni	0	1.3	-	0	1.3	-
Lutjanus synagris	0	0.8	-	0	0.8	-

In the Flower Gardens, shallow water species, such as dog (*L. jocu*), gray (*Lutjanus griseus*), lane (*Lutjanus synagris*), mahogany (*Lutjanus mahogoni*) and mutton (*Lutjanus analis*) snappers; and deep-water species, such as blackfin (*Lutjanus buccanella*) and red (*Lutjanus campechanus*) snappers, have been observed in surveys by REEF (Pattengill-Semmens and Semmens, 1998) and MMS (Precht et al., 2006). Overall, lutjanids were not a dominant species in CCMA surveys (Table 4.8) with only two species (*L. griseus* and *L. jocu*) exhibiting sighting frequencies greater than 1.5% in REEF surveys (Pattengill-Semmens, 2006). These species were also the only species observed in FGBNMS LTM surveys where abundance was low (Precht et al., 2006). Sighting frequency for *L. jocu* was similar between CCMA and REEF surveys, while *L. griseus* were sighted more frequently on EB by REEF. CCMA recorded slightly higher sighting frequency on WB. *L. buccanella*, *L. mahogoni* and *L. synagris* were not observed by CCMA and cubera snapper (*Lutjanus cyanopterus)* was not documented in REEF or FGBNMS LTM surveys. *O. chrysurus* a common continental/insular reef species, has been observed at FGBNMS in REEF observations.

Less than 1% of the total abundance of fish observed were from the family Lutjanidae. Overall, 104 individuals were observed from four species (*L. analis*, *L. cyanopterus*, *L. griseus*, *L. jocu*). Similar to the results observed by REEF, sighting frequencies were highest for *L. griseus* and *L. jocu*. Snappers were observed on both East and West Banks and density (Figure 4.53) was not significantly different among the bank/habitat type combinations. No discernible spatial pattern was evident for density on either bank.

Total snapper biomass amounted to 1,686 kg which accounted for 7% of the total biomass observed during the study period. No apparent spatial pattern was observed for biomass, although one area of peak biomass is noticeable on the western end of WB (Figure 4.54). Snapper biomass was not significantly different among banks or relief types. Snapper density and biomass were not correlated with any of the benthic cover parameters or depth.

Only two individuals were less than 20 cm, while nearly half of the snappers observed were greater than 35 cm (Figure 4.55). As previously mentioned, typical snapper nursery/recruitment habitats (mangroves, seagrass) are not present at FGBNMS. At present, it is uncertain as to how the sanctuary maintains its snapper population. Currently it is uncertain if larval or juvenile recruitment occurs in the sanctuary and from where they come from. These questions are critical to understanding the snapper population structure and ecological function.

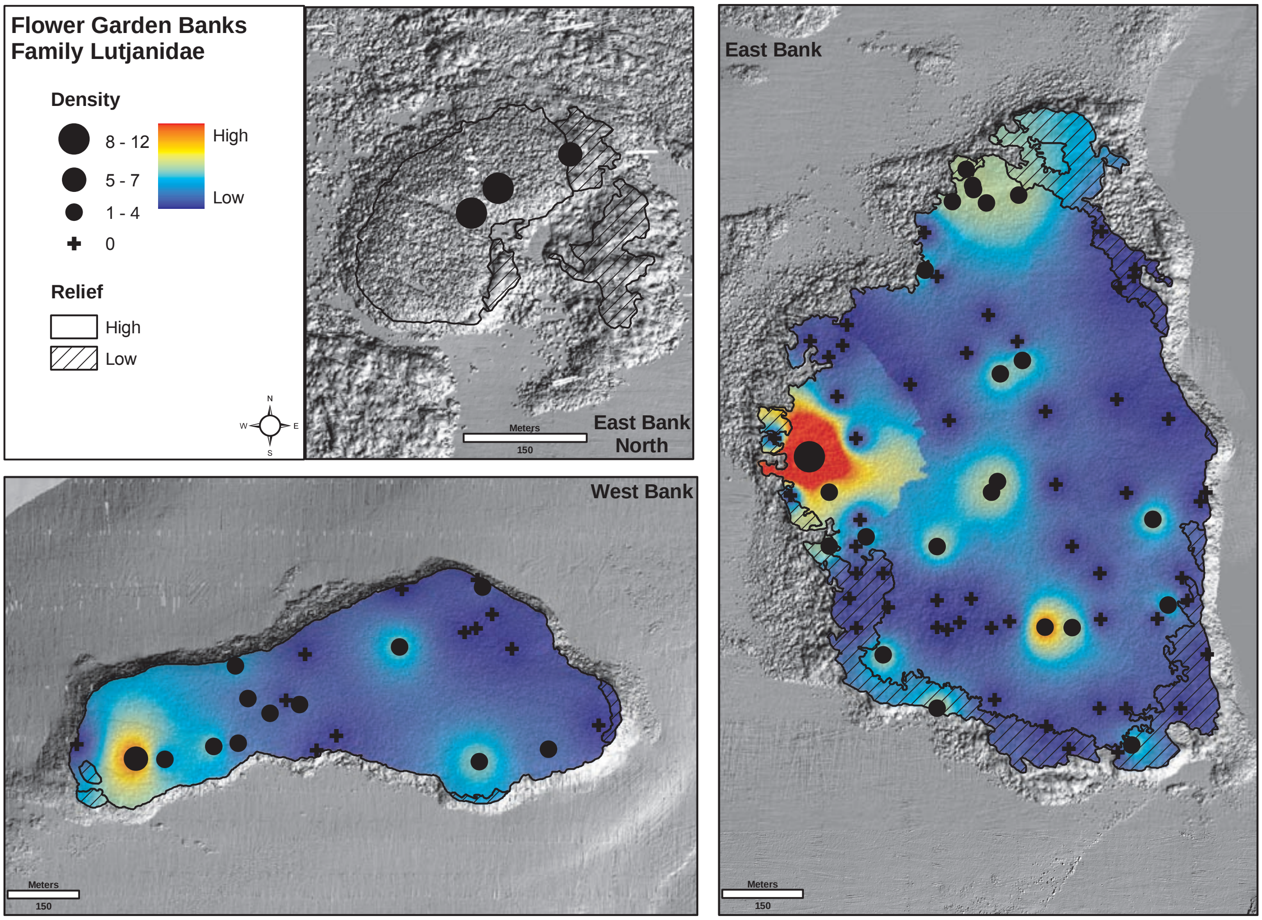


Figure 4.53. Observed (dots) and spatially interpolated density (#/100 m^2) for all snappers (Lutjanidae) observed in CCMA surveys for 2006-2007.

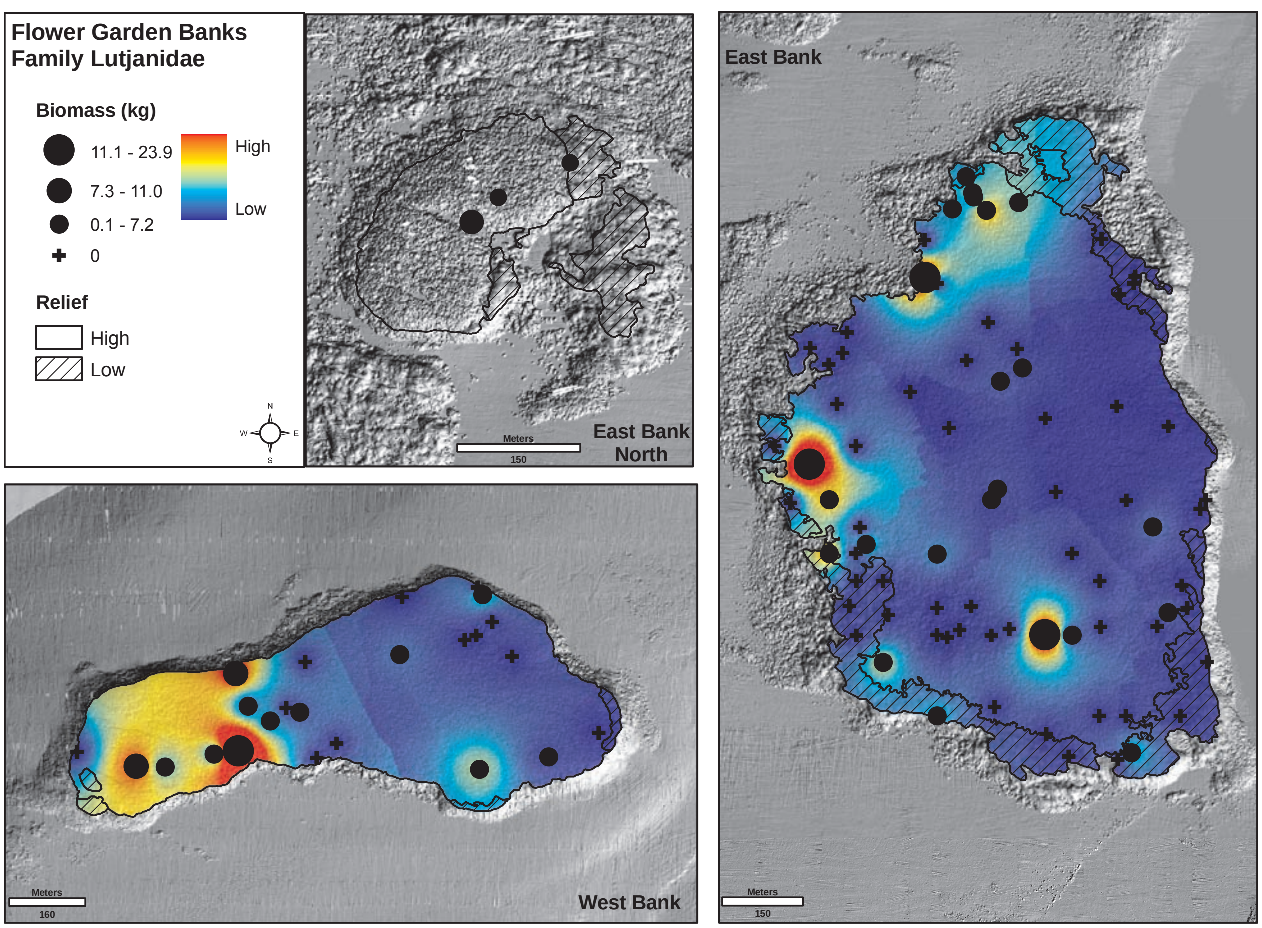


Figure 4.54. Observed (dots) and spatially interpolated biomass (kg) for all snappers (Lutjanidae) observed in CCMA surveys for 2006-2007.

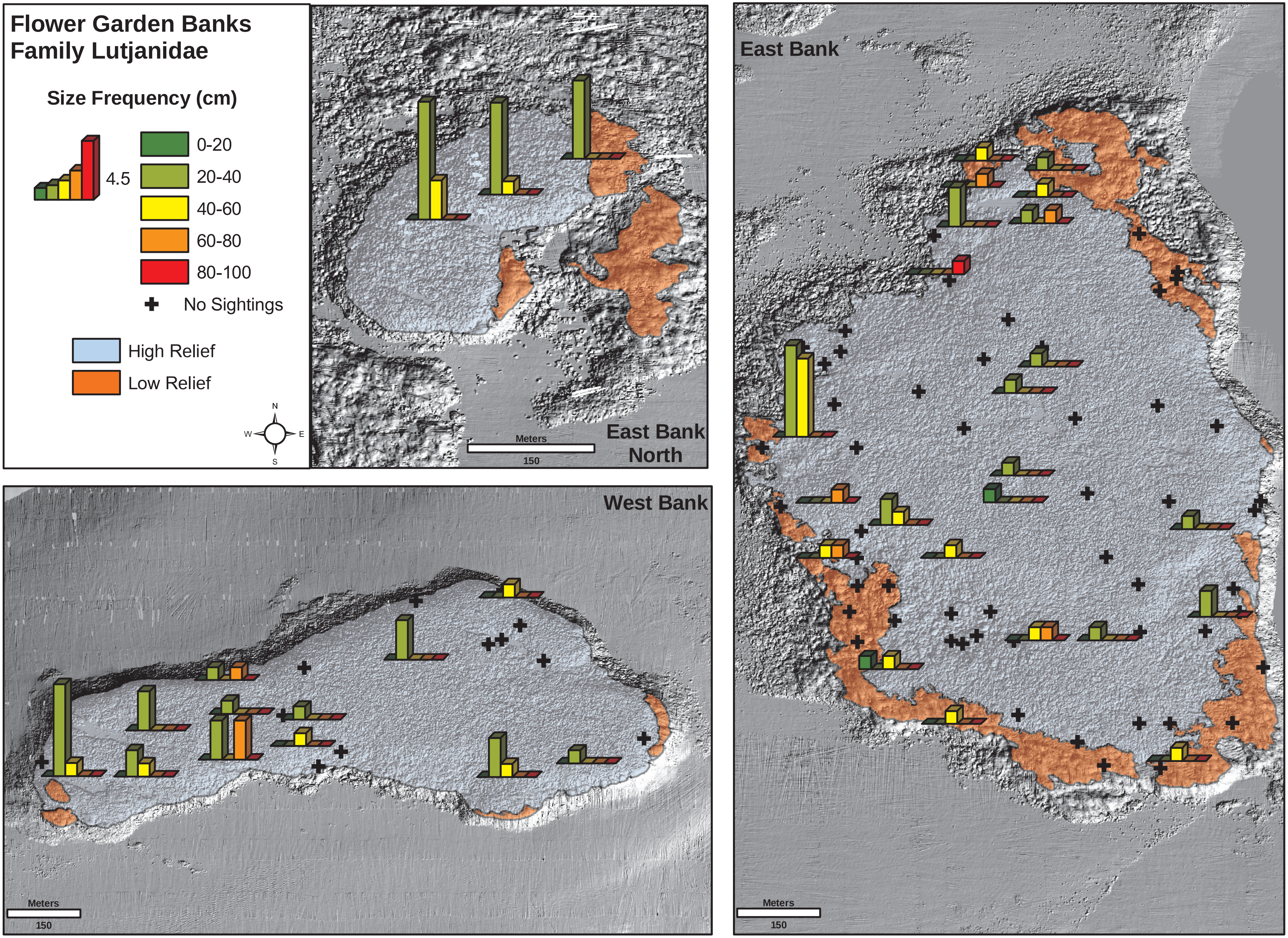


Figure 4.55. Spatial distribution for all snapper (Lutjanidae) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 4.5 individuals.

4.3.3.2.1 Gray snapper (*Lutjanus griseus*)

L. griseus are tropical/sub-tropical species that occur from the U.S. mid-Atlantic south to Rio de Janeiro, Brazil, including the Gulf of Mexico and Caribbean. Aside from FGBNMS, *L. griseus* are found in a variety of habitats, including coral reefs, rocky areas, mangrove sloughs, estuaries, tidal creeks, lower reaches of rivers, and on occasion fresh waters (Carpenter, 2002). *L. griseus* can grow to about 76 cm TL (Manooch and Matheson, 1981) but are more commonly observed at 55 cm (Carpenter, 2002). Size at maturity varies throughout its range: 23 cm FL in Florida and 28 cm FL in Cuba (Garcia-Cagide et al., 1994).

Lutjanus griseus (Burek)

There is a considerable commercial fishery for *L. griseus*, primarily in Florida and Louisiana, exceeding 922,000 kg and 112,000 kg, respectively, during 2000-2006. The fishery appears to be minimal in the western Gulf of Mexico. Recreational landings throughout the Gulf have averaged over four million individuals weighing approximately 753,000 kg (NMFS, unpublished data; http://www.st.nmfs.noaa.gov/st1/commercial/index.html) where landings from Florida comprise approximately 67% of the total landings by biomass. *L. griseus* in federal waters are managed by the GOMFMC under the Reef Fish Fishery Management Plan. Size limits for both the recreational and commercial fisheries are 32 cm FL.

Sighting frequency on EB (16.05%) was similar to that of REEF (19%) and nearly twice as high on WB (41.67% and 22%, respectively). Only one *L. griseus* was observed by MMS in 64 surveys from 2002-2003.

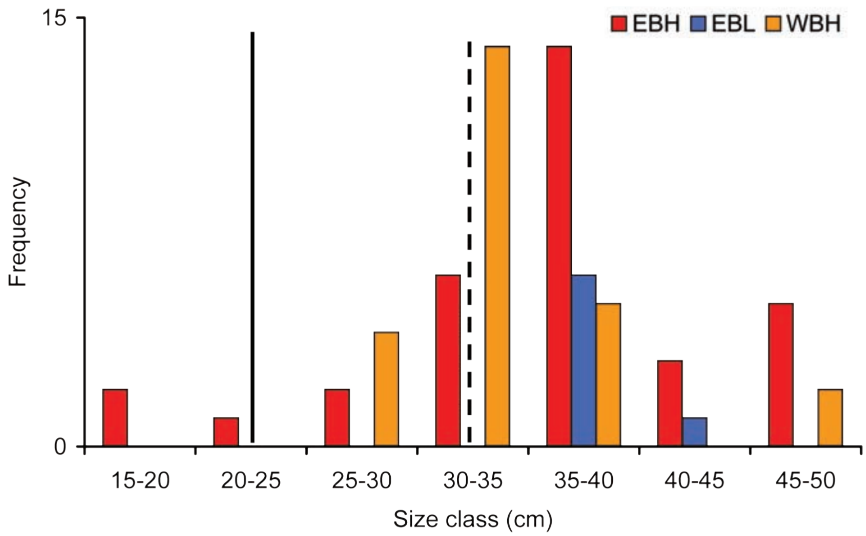


*Figure 4.56. Length frequency of gray snapper (*L. griseus*) observed in CCMA surveys for 2006-2007. Vertical solid black line represents size of maturity for females observed in Florida (Garcia-Cagide et al., 1994). Dashed black line represents the size limit for the recreational and commercial fisheries (GOMFMC, 2008a,b).*

The majority of *L. griseus* observed were adults and were found on all bank/habitat types (Figure 4.56). Nearly all individuals were larger than the minimum take size for the commercial and recreational fisheries. *L. griseus* was the most abundant snapper species observed (n=65) exhibiting a total biomass of 535 kg. Although sighting frequency was higher on WB, density (individuals/100 m^2) was not significantly greater (WB=1.02/100 m^2, EB=0.74/100 m^2; Figure 4.57). Density was lower on low relief habitats than high relief, but this difference was not statistically significant. Similarly, biomass (Figure 4.58) was not significantly different on either bank, but slightly higher on EB (1.31 kg/100 m^2) compared to WB (0.85 kg/100 m^2). Density and biomass were both generally greater on the edge of the EB coral cap, while no apparent patterns were obvious on WB.

No size specific spatial patterns were observed throughout the sanctuary (Figure 4.59).

Coral cover, macroalgal cover and depth were not significantly correlated with *L. griseus* abundance or biomass; however, in general, density was highest at sites with greater than 40% coral cover and low macroalgae cover.

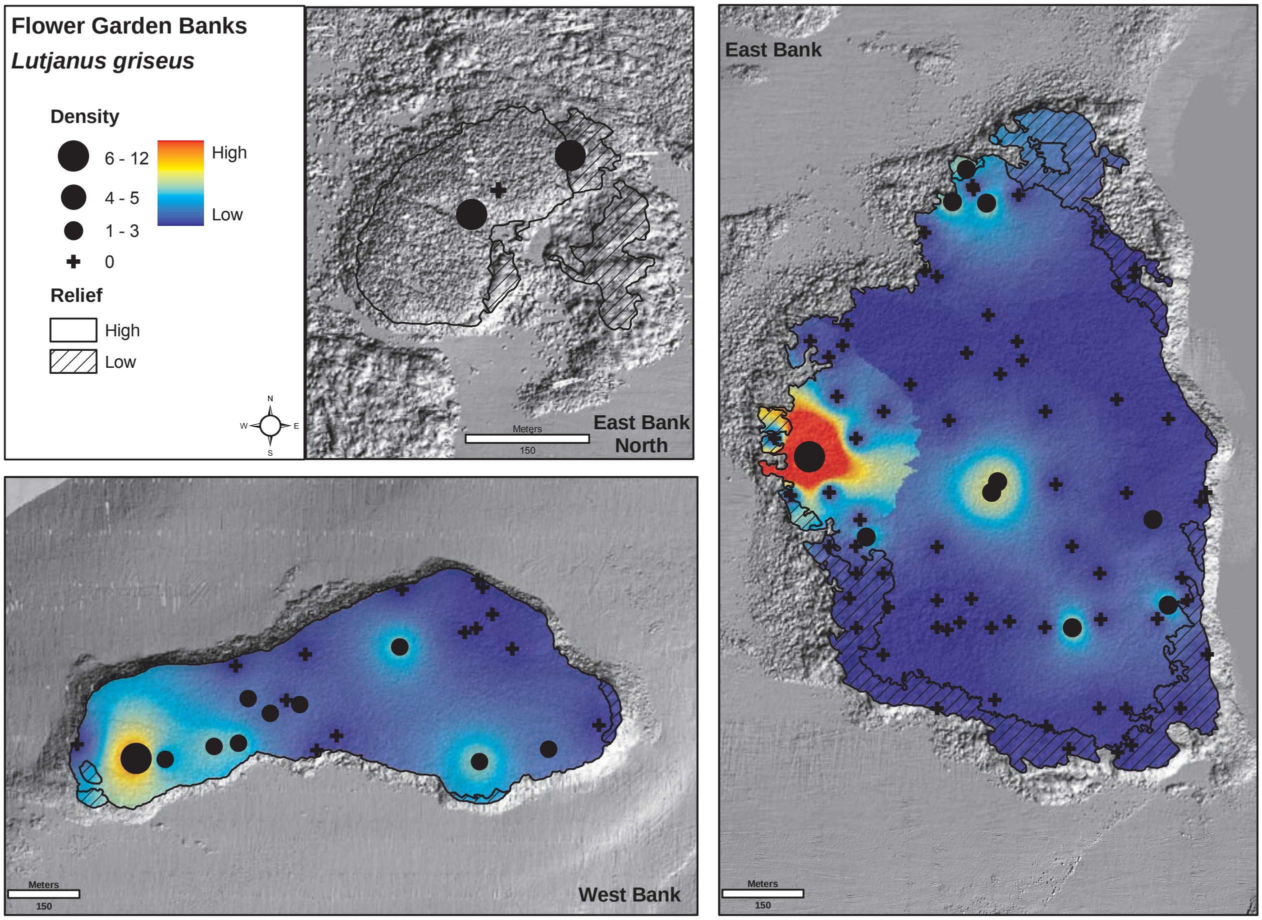


*Figure 4.57. Observed (dots) and spatially interpolated density (#/100 m^2) for gray snapper (*L. griseus*) observed in CCMA surveys for 2006-2007.*

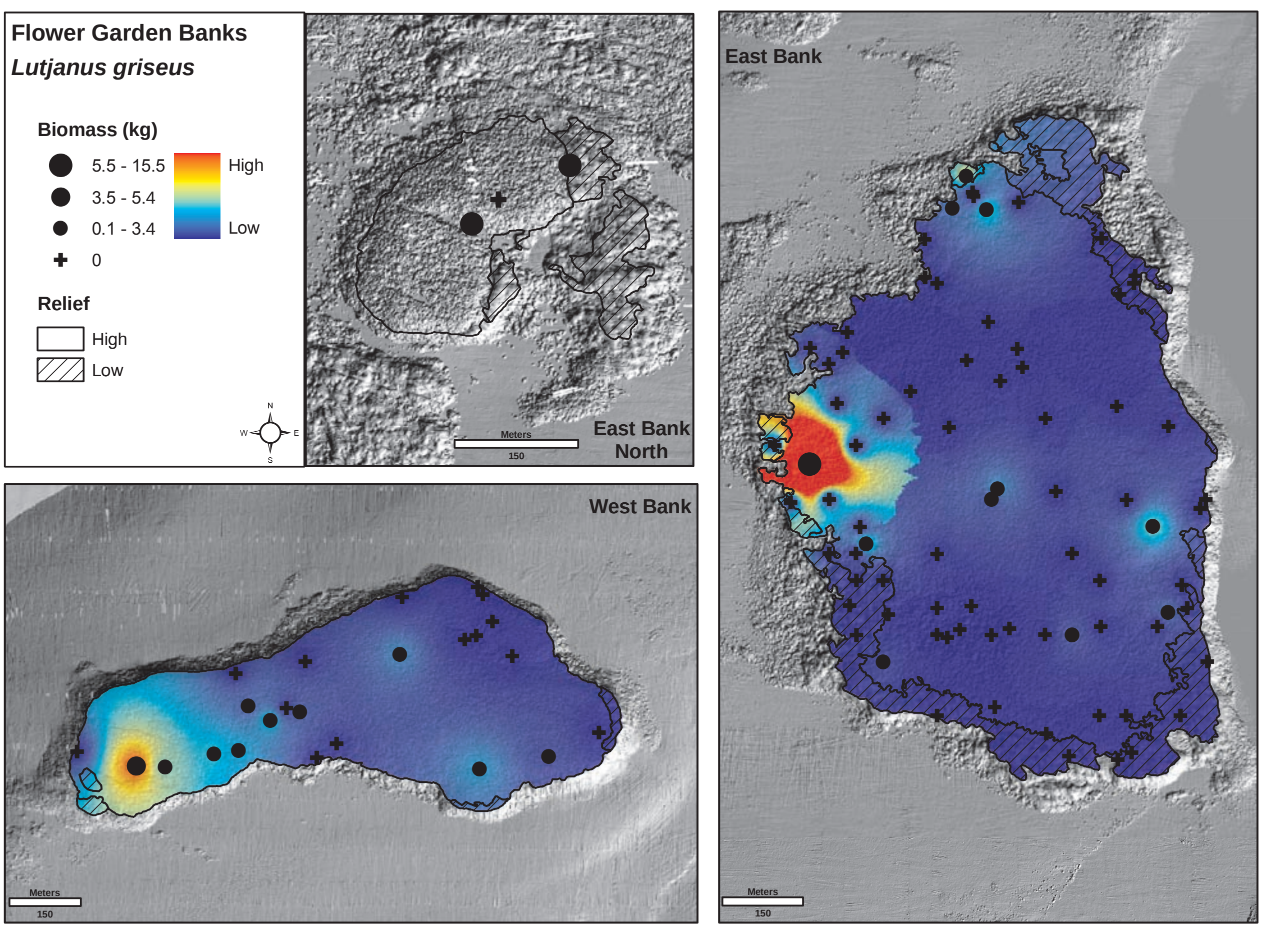


*Figure 4.58. Observed (dots) and spatially interpolated biomass (kg) for gray snapper (*L. griseus*) observed in CCMA surveys for 2006-2007.*

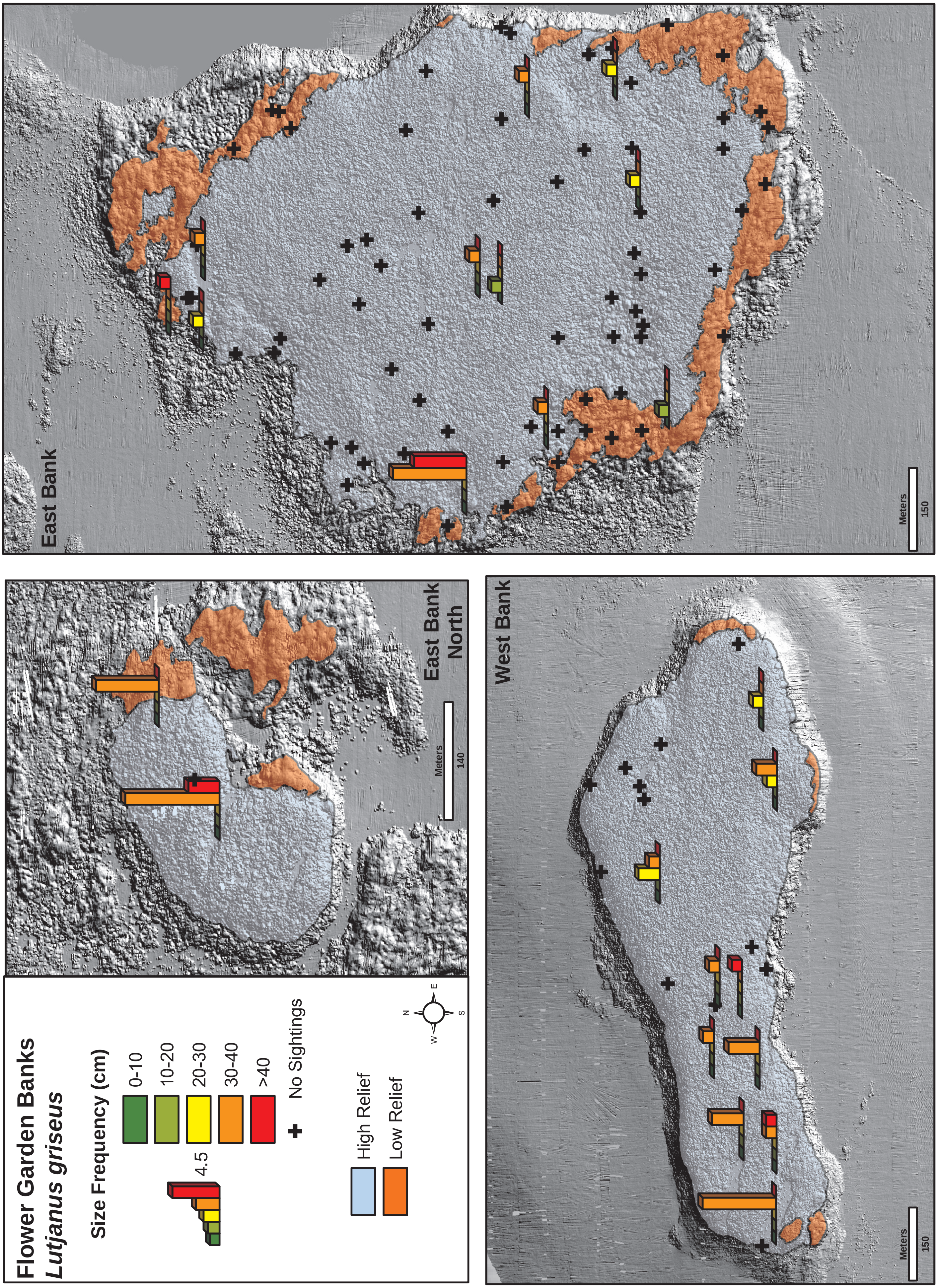


Figure 4.58. Spatial distribution for gray snapper (L. griseus) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 4.5 individuals.

4.3.3.2.2 Dog snapper (*Lutjanus jocu*)

L. jocu are distributed in the western Atlantic from Massachusetts to northern Brazil, including the Gulf of Mexico and Caribbean (SAFMC, 2005). They are common on rocky or coral reefs at depths from 5-30 m. Maximum reported size is 128 cm TL with a biomass of 28.6 kg (Allen, 1985). Mean length of sexually mature males (48 cm) and females (43 cm) have been determined from fish collected in Cuba (Garcia-Cagide et al., 1994).

Lutjanus jocu (CCMA)

Commercial catches primarily occur with handlines, gill nets, and traps and recreational captures typically are harvested with hook and line and spearfishing (SAFMC, 2005). *L. jocu* are infrequently recorded in Gulf of Mexico commercial landings (NMFS, unpublished data; http://www.st.nmfs.noaa.gov/st1/commercial/index.html).

L. jocu sighting frequency was equivalent to that observed by REEF: 22% of surveys on EB and 25% on WB (Table 4.8). Much lower values were documented by MMS (6%) and exhibited density ranging from 0.03-0.13/100 m^2 on East and West Banks, respectively (Precht et al., 2006).

Mean size of individuals was smaller on WB than EB and approximately half were considered adults (Figure 4.60). Nearly all individuals observed were greater than the minimum take size in the commercial and recreational fisheries.

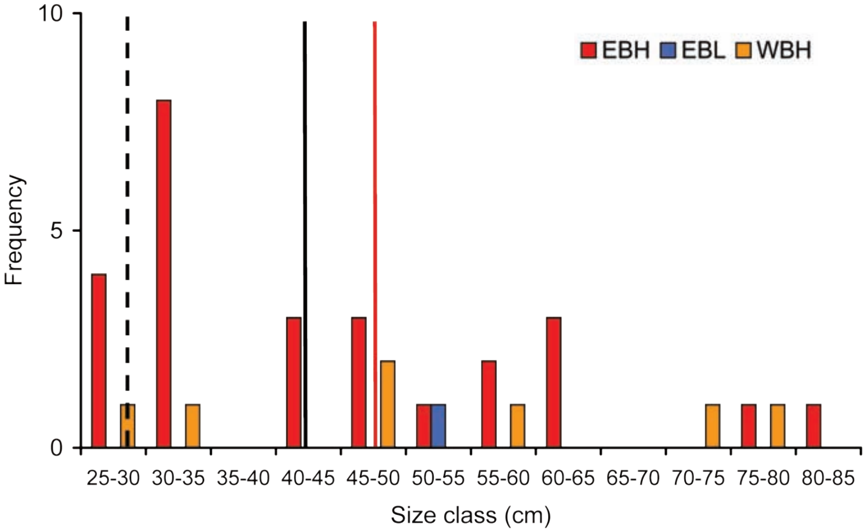


*Figure 4.60. Length frequency of dog snapper (*L. jocu*) observed in CCMA surveys for 2006-2007. Vertical solid black and red lines represent mean size of maturity for females and males, respectively, observed in Cuba (Garcia-Cagide et al., 1994). Dashed black line represents the size limit for the recreational and commercial fisheries (GOMFMC, 2008a,b).*

L. jocu density (EB=0.40/100 m^2, WB=0.29/100 m^2) were lower than that observed for *L. griseus*; however, biomass was nearly two times greater (1,006 kg). Density was distributed primarily on the edges of the coral caps (Figure 4.61) and was significantly greater on EBH ($p<0.0001$). Density was generally lower on low relief habitats, but was not statistically significant. Biomass was greater on EB, but no significant differences were observed between habitat relief types (Figure 4.62). Density and biomass were predominately distributed on the edges of the coral caps, with no obvious spatial pattern for size frequency (Figure 4.63).

Abundance and biomass were not correlated with depth, coral or macroalgae cover.

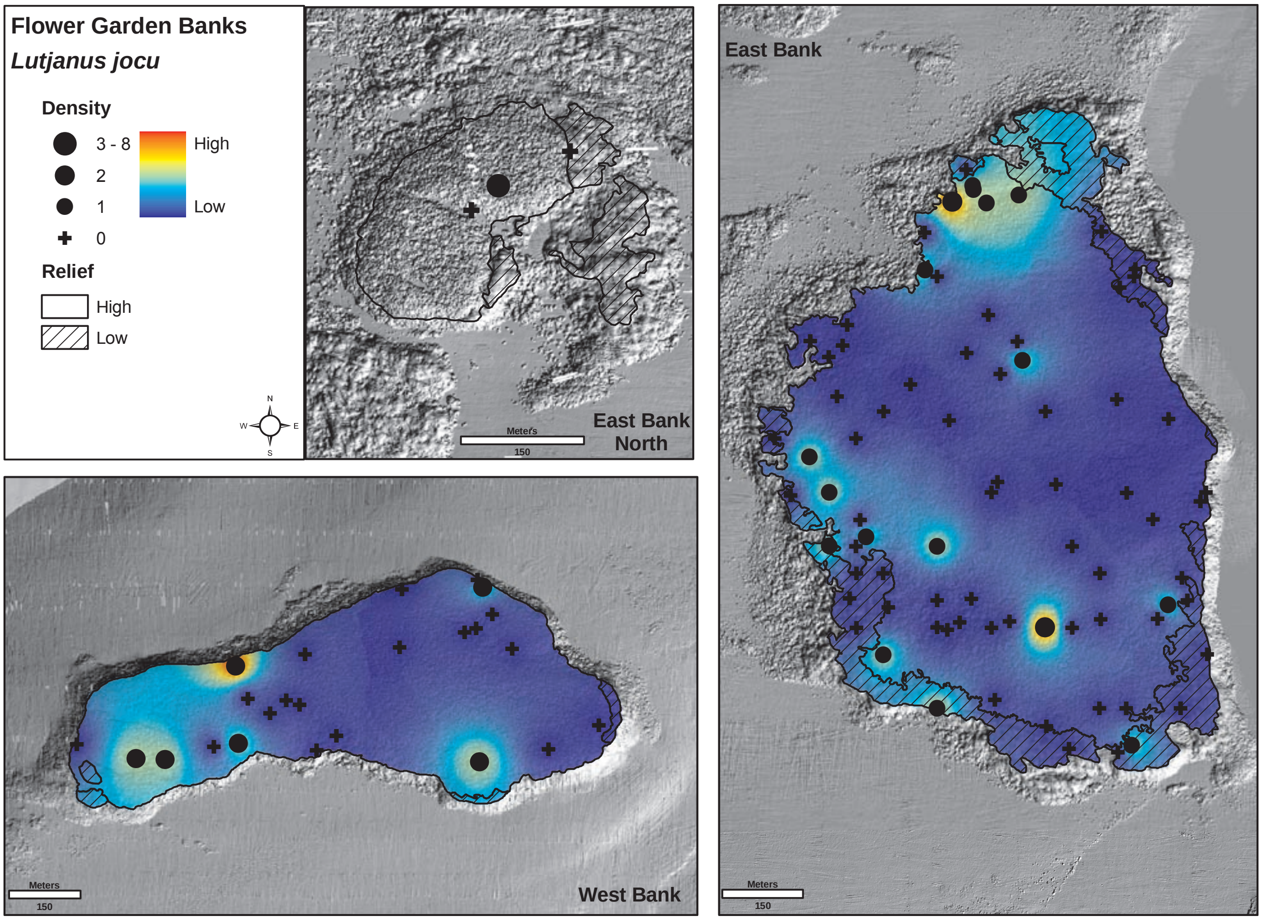


Figure 4.61. Observed (dots) and spatially interpolated density (#/100 m^2) for dog snapper (L. jocu) observed in CCMA surveys for 2006-2007.

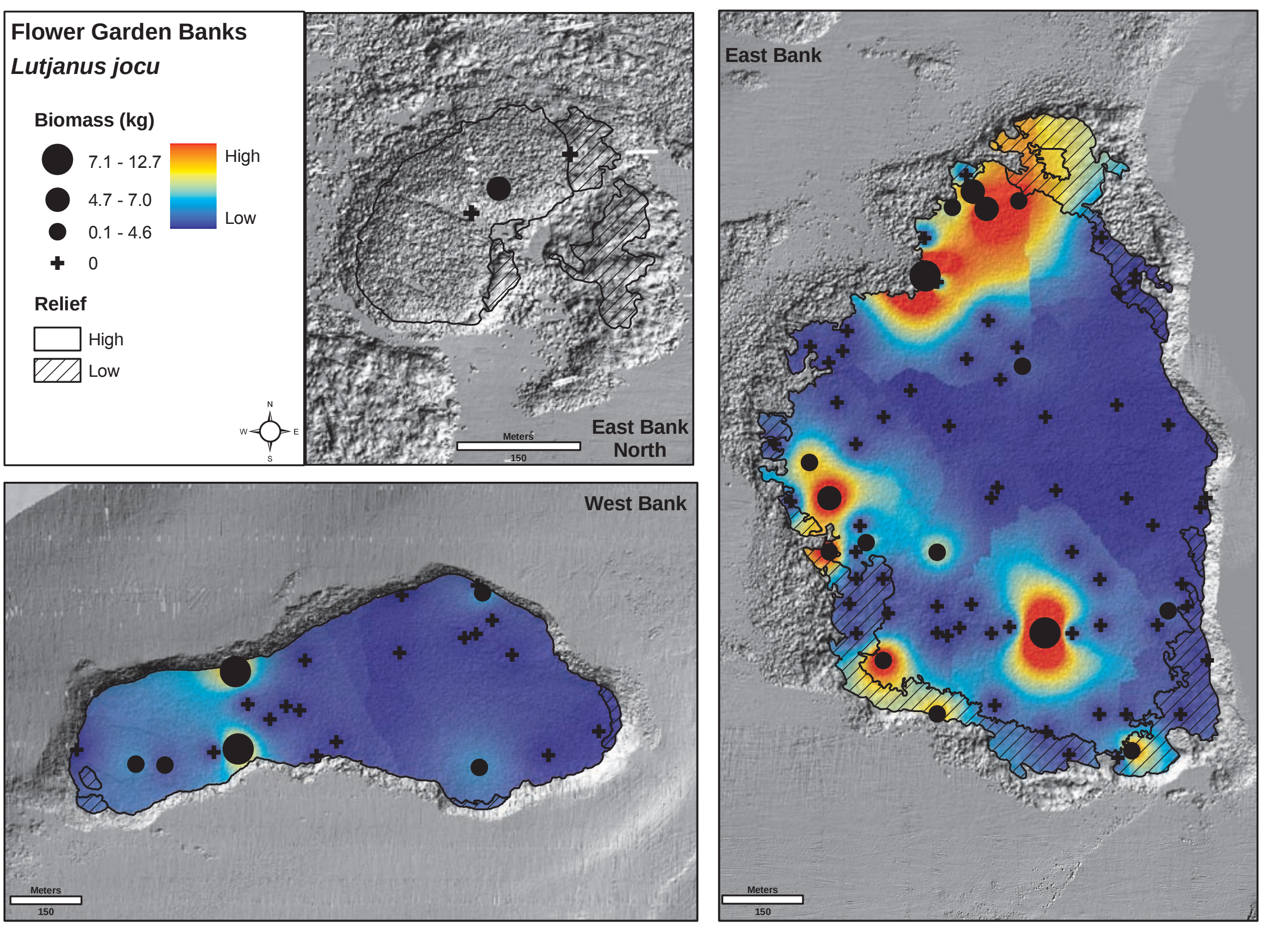


Figure 4.62. Observed (dots) and spatially interpolated biomass (kg) for dog snapper (L. jocu) observed in CCMA surveys for 2006-2007.

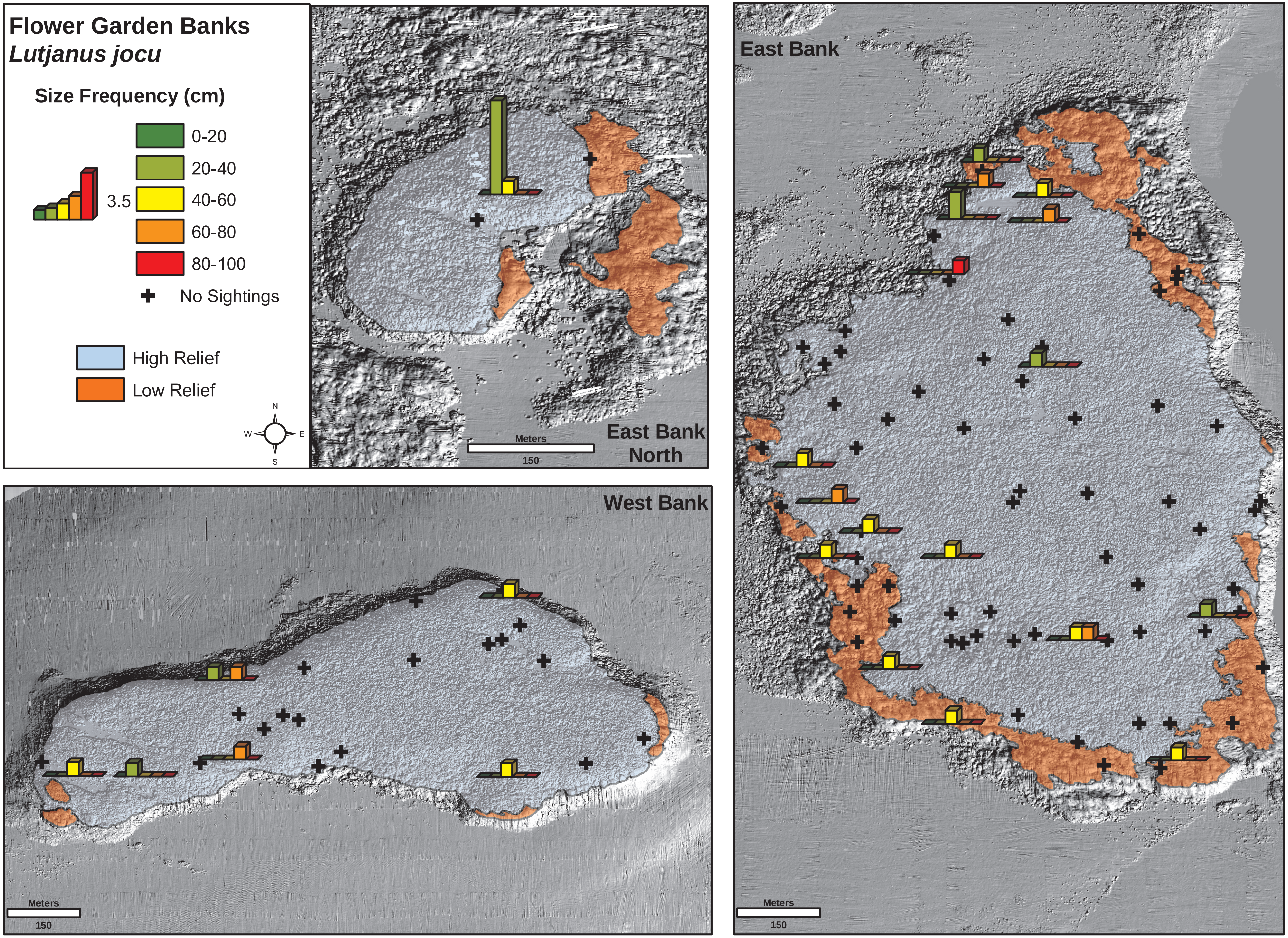


Figure 4.63. Spatial distribution for dog snapper (L. jocu) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 3.5 individuals.

4.3.3.3 *Scaridae (Parrotfishes)*

Parrotfishes are abundant herbivores on tropical coral reefs, where they are often the largest component of the fish biomass. Depth distribution is primarily 1-30 m, with some species occurring down to 80 m. Adult scarids are grazing animals, feeding on the close-cropped algal and bacterial mat covering dead corals and rocks, sea grasses and coral. Juveniles feed on small invertebrates. Parrotfishes feed continuously during the day, often in mixed schools, biting at rocks and corals. In pulverizing the coral and rock fragments and sand they create substantial quantities of sediment. In many areas they are probably the principal producers of sand.

Initial phase *Sparisoma viride* (Burek)

Herbivores are a key functional group on coral reefs by mediating space competition between corals and benthic macroalgae (Mumby, 2006). Experiments have shown that exclusion of herbivores, such as parrotfishes and others, cause a shift from a coral to macroalgal dominated state (Hughes et al., 2007). Some Caribbean parrotfish species, such as rainbow parrotfish (*Scarus guacamaia*), queen parrotfish (*Scarus vetula*), redband parrotfish (*Sparisoma aurofrenatum*) and stoplight parrotfish (*Sparisoma viride*) feed directly on live corals, and thus have the potential to negatively impact coral fitness and survival. Due to the significant role parrotfish contribute to reef functionality, an in-depth examination of abundance and distribution patterns for this family is investigated further.

Table 4.9. Sighting frequency of all parrotfish (Scaridae) species from CCMA, REEF and MMS surveys. REEF estimates (means) are from expert surveys only. MMS data only reflect presence/absence indicated by +/-.

	East Bank			West Bank		
Species	**CCMA**	**REEF**	**MMS**	**CCMA**	**REEF**	**MMS**
Scarus vetula	64.20	88.44	+	79.17	95.50	+
Scarus taeniopterus	56.79	78.31	+	37.50	72.40	+
Scarus iseri	12.35	11.13	+	0	8.78	+
Sparisoma viride	75.31	91.98	+	75	91.82	+
Sparisoma aurofrenatum	85.18	86.68	+	100	82.25	+
Sparisoma atomarium	40.74	15.16	-	8.33	5.80	-

Six species of parrotfish were common during this study period and REEF surveys from 1995-2005 (Table 4.9). Sighting frequencies were considerably lower than REEF estimates for princess parrotfish (*Scarus taeniopterus*), *Sc. vetula* and *Sp. viride* on both banks, while higher estimates were observed for greenblotch parrotfish (*Sparisoma atomarium*) on both banks. *Sp. aurofrenatum* sightings were comparable on EB and higher than that of REEF on WB. *Sc. vetula* was the most abundant parrotfish (72-86% of surveys); densities (individuals/100 m^2) ranged from 0.66-1.06/100 m^2 during FGBNMS LTM surveys (Precht et al., 2006).

Overall, 1,439 individual scarids were observed within the sanctuary with a total biomass of 148 kg. Total mean density for all scarids was 13.7/100 m^2 and was higher on EB than WB. Scarid density by bank and habitat type indicated greater density (18.1/100 m^2) on EBL habitats, followed by 14.2/100 m^2 on EBH habitats and significantly lower density (9.6/100 m^2) on WBH habitats ($p<0.0001$; Figure 4.64). The pattern of high density on low relief was strongly influenced by the high abundance of *Sp. atomarium* which ranked tenth among all species observed on low relief habitat.

Sp. aurofrenatum was the most abundant parrotfish observed, and its abundance was equitable throughout the sanctuary. Striped parrotfish (*Scarus iseri*) and *Sc. taeniopterus* density were significantly greater ($p<0.0001$) on EB (on both high and low relief) than WB, although these species were the least abundant of the parrotfish. *Sp. viride* was the second most abundant and its density was significantly lower ($p=0.04$) on low relief habitat than high relief habitat on either bank. The same pattern was observed for *Sc. vetula* but was not statistically significant. *Sp. atomarium* density was low on high relief habitats on both banks, yet significantly greater on EB ($p<0.0001$). Overall, parrotfish density was highest in the central, shallower portion of the coral caps.

The distribution of parrotfish biomass (Figure 4.65) was patchy, primarily driven by the abundance of larger-bodied species, such as *Sc. vetula* and *Sp. viride* (Figure 4.66). Total parrotfish biomass was not significantly different between banks or between bank/habitat type combinations. Mean biomass was greatest for *Sp. viride* (0.84 kg/100 m^2), followed by *Sc. vetula* (0.43 kg/100 m^2), *Sp. aurofrenatum* (0.07 kg/100 m^2), *Sc. taeniopterus* (0.06 kg/100 m^2), *Sc. iseri* (0.003 kg/100 m^2) and *Sp. atomarium* (0.002 kg/100 m^2).

Overall, parrotfish abundance and biomass were not correlated with depth, coral or macroalgae cover. However, individual species exhibited significant correlations. For example, *Sp. atomarium* density was positively correlated with depth ($\rho = 0.37$) and macroalgae cover ($\rho = 0.32$) and negatively correlated with coral cover ($\rho = -0.32$).

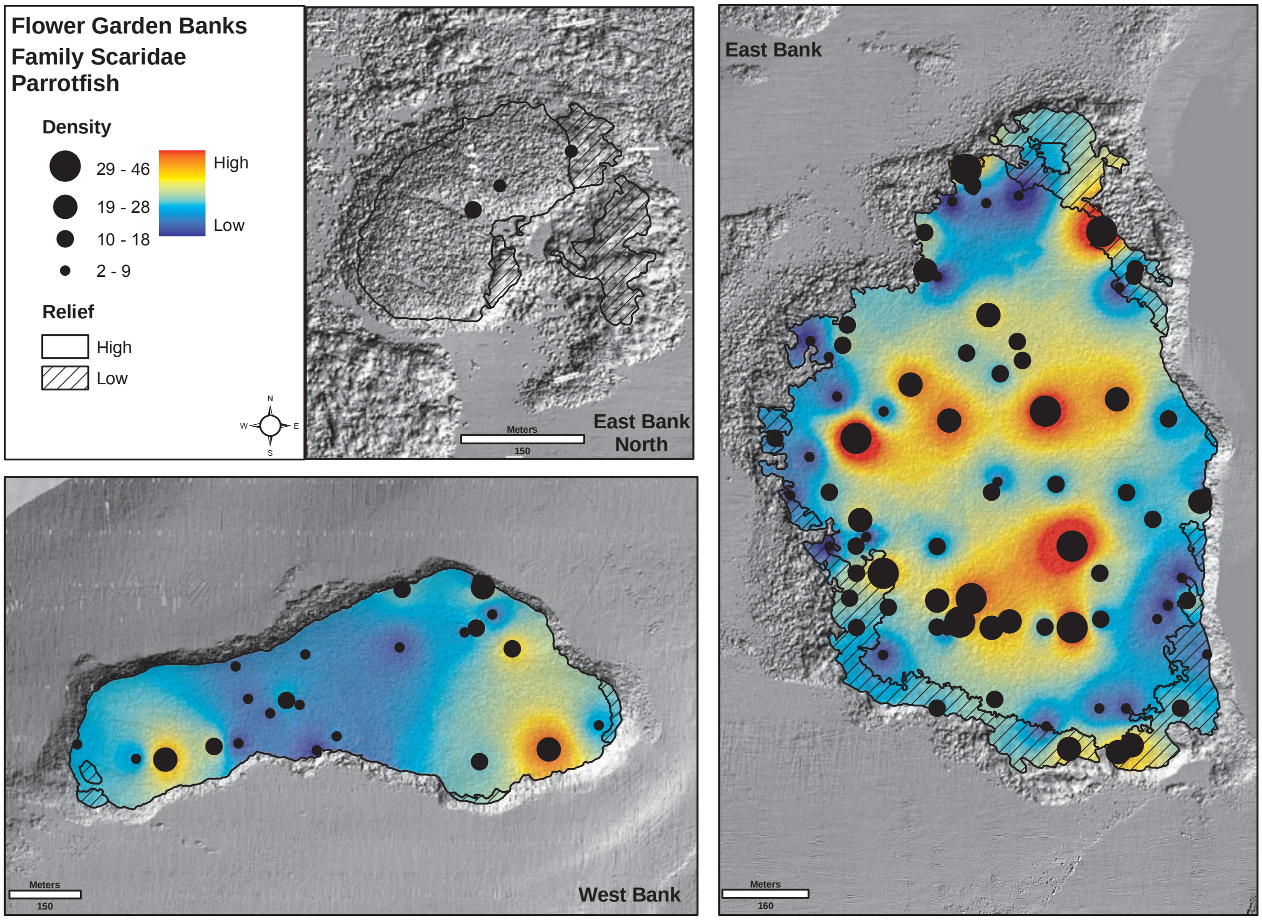


Figure 4.64. Observed (dots) and spatially interpolated density (#/100 m^2) for all parrotfish (Scaridae) observed in CCMA surveys for 2006-2007.

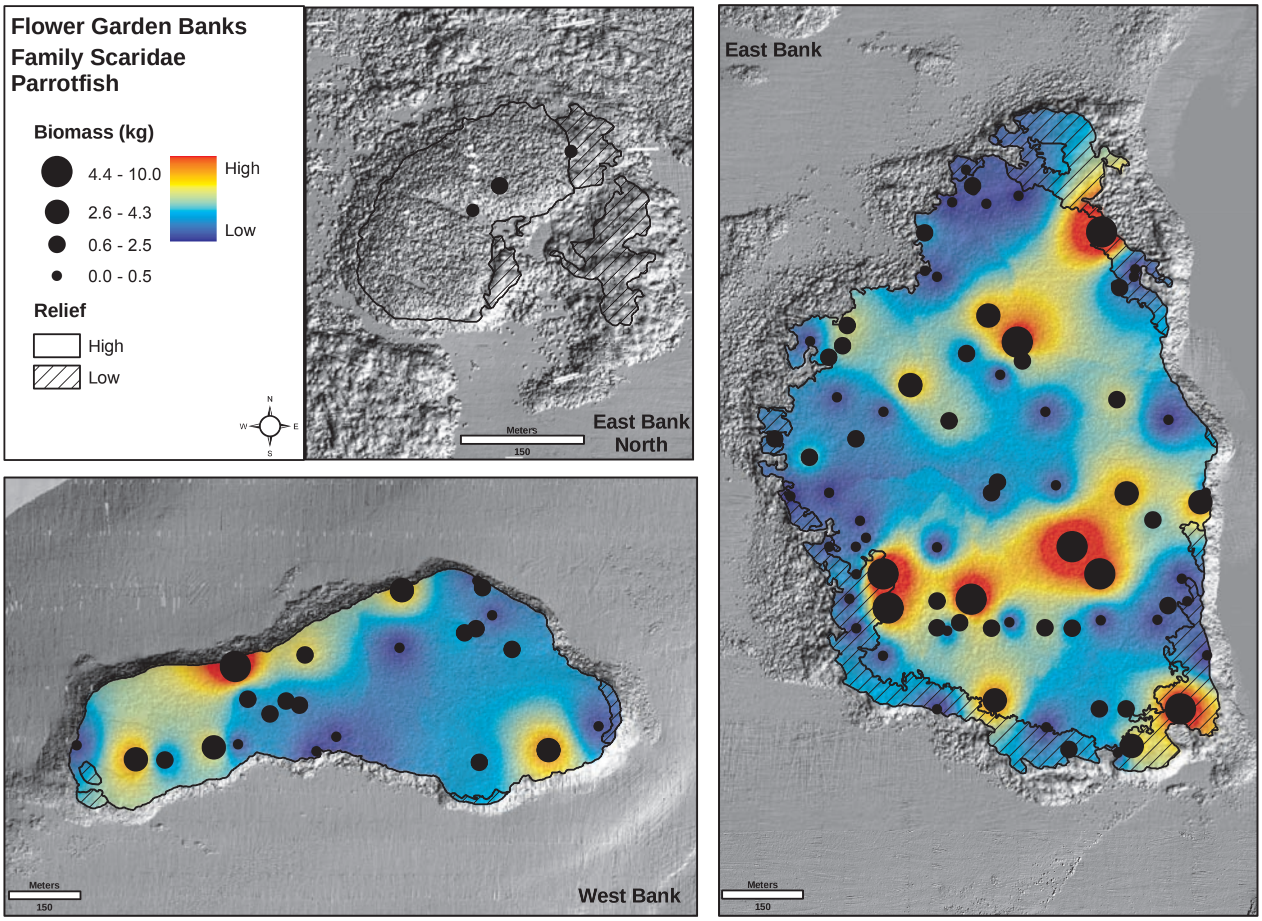


Figure 4.65. Observed (dots) and spatially interpolated biomass (kg) for all parrotfish (Scaridae) observed in CCMA surveys for 2006-2007.

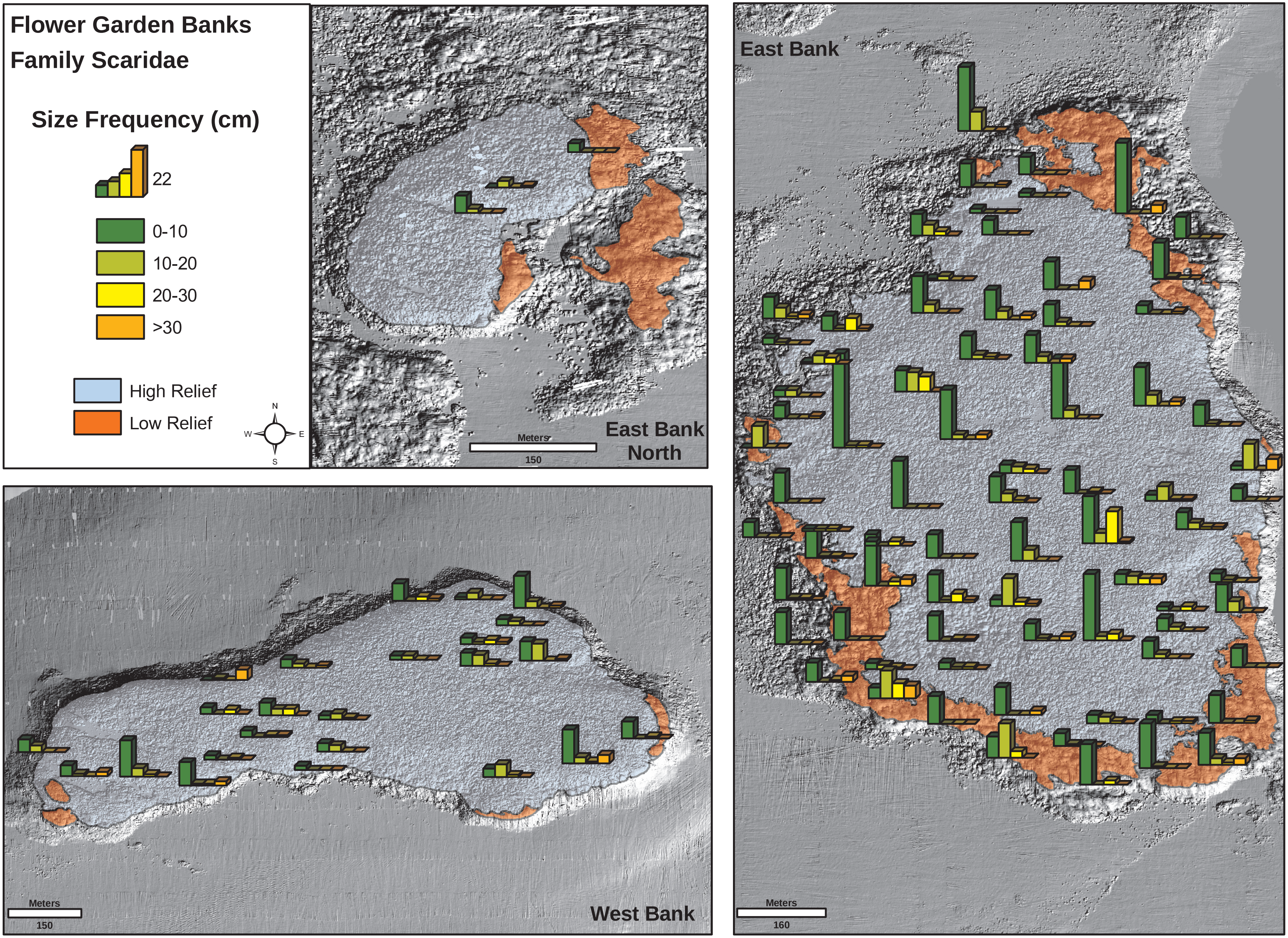
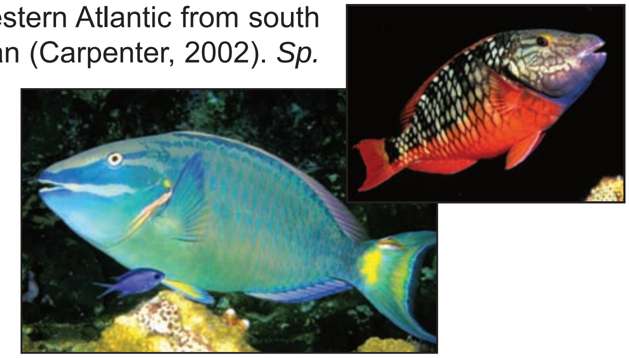


Figure 4.66. Spatial distribution for all parrotfish (Scaridae) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 22 individuals.

Grazers, such as parrotfish, are functionally important in coral reef ecosystems. Algal growth can negatively influence coral recruitment and out-compete corals for space. Mumby et al. (2007) claim that herbivores, such as parrotfish, are necessary for coral health and reef function. As such, the two largest parrotfish observed in CCMA surveys are examined in more detail.

4.3.3.3.1 Stoplight parrotfish (*Sparisoma viride*)

Sp. viride are tropical/sub-tropical reef species found in the western Atlantic from south Florida to Brazil, including the Gulf of Mexico and the Caribbean (Carpenter, 2002). *Sp. viride* are one of the larger members of the scarids reaching a maximum size of 64 cm, biomass of 1.6 kg (Claro, 1994) and maximum age of approximately nine years (Choat et al., 2002). Adults typically are found on reefs, while juveniles may be found on seagrass beds or other heavily vegetated substrates. Food items include mainly macroalgae, but grazing also occurs on live corals, such as *Mo. annularis* (Frydl, 1979). As an excavating substratum feeder (Bellwood, 1994) this species is responsible for a significant component of the grazing and bioerosion of Caribbean reefs (Bruggemann et al., 1996; van Rooij et al., 1998). Size at maturity data from Bermuda indicate that females reach maturity at 16.3 cm SL (19.6 cm FL; Choat et al., 2002).



Terminal and initial phase *Sparisoma viride* (Burek)

Sp. viride was the second most abundant scarid species with 75% frequency of occurrence on each bank. It was also common among REEF surveys during 1995-2005 throughout the sanctuary with sighting frequencies of 88.9% and 84.7% on EB and WB, respectively.

Adults and juveniles were proportionally abundant on both banks, but few individuals were observed on low relief habitats (Figure 4.67). Overall, 303 individuals were observed with a collective biomass of 88.7 kg. Mean density was greater on EB (3.03/100 m^2) compared to WB (1.79/100 m^2). Density was significantly greater (p=0.005) on EBH than EBL but not significantly different than WBH (Figure 4.68). Mean biomass (Figure 4.69) was slightly higher on EB (0.40 kg/100 m^2) than WB (0.26 kg/100 m^2) but not significantly different between bank/habitat type combinations.

No apparent spatial pattern was observed for size frequency (Figure 4.70).

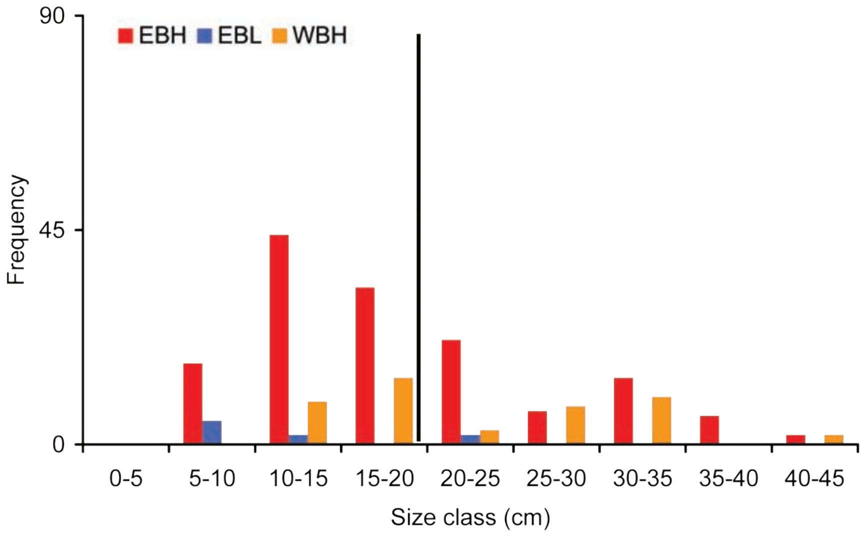


*Figure 4.67. Length frequency for stoplight parrotfish (*Sp. viride*) observed in CCMA surveys for 2006-2007. Vertical solid black line represents mean size of maturity for females (Winn and Bardach, 1960).*

Density and biomass were inversely correlated with depth (ρ = -0.49 and ρ = -0.27, respectively). Neither were correlated with coral or macroalgal cover.

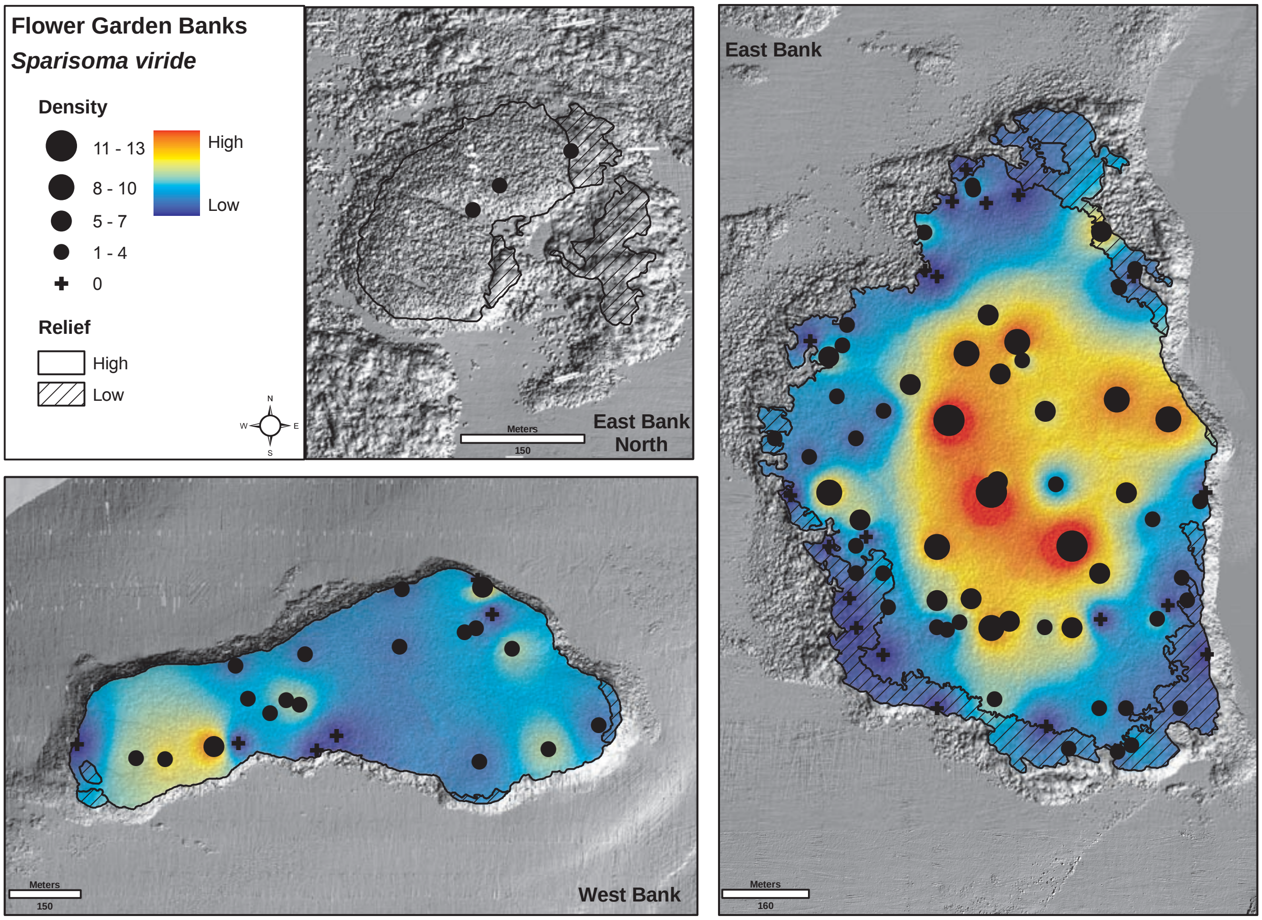


Figure 4.68. Observed (dots) and spatially interpolated density (#/100 m^2) for stoplight parrotfish (Sp. viride) observed in CCMA surveys for 2006-2007.

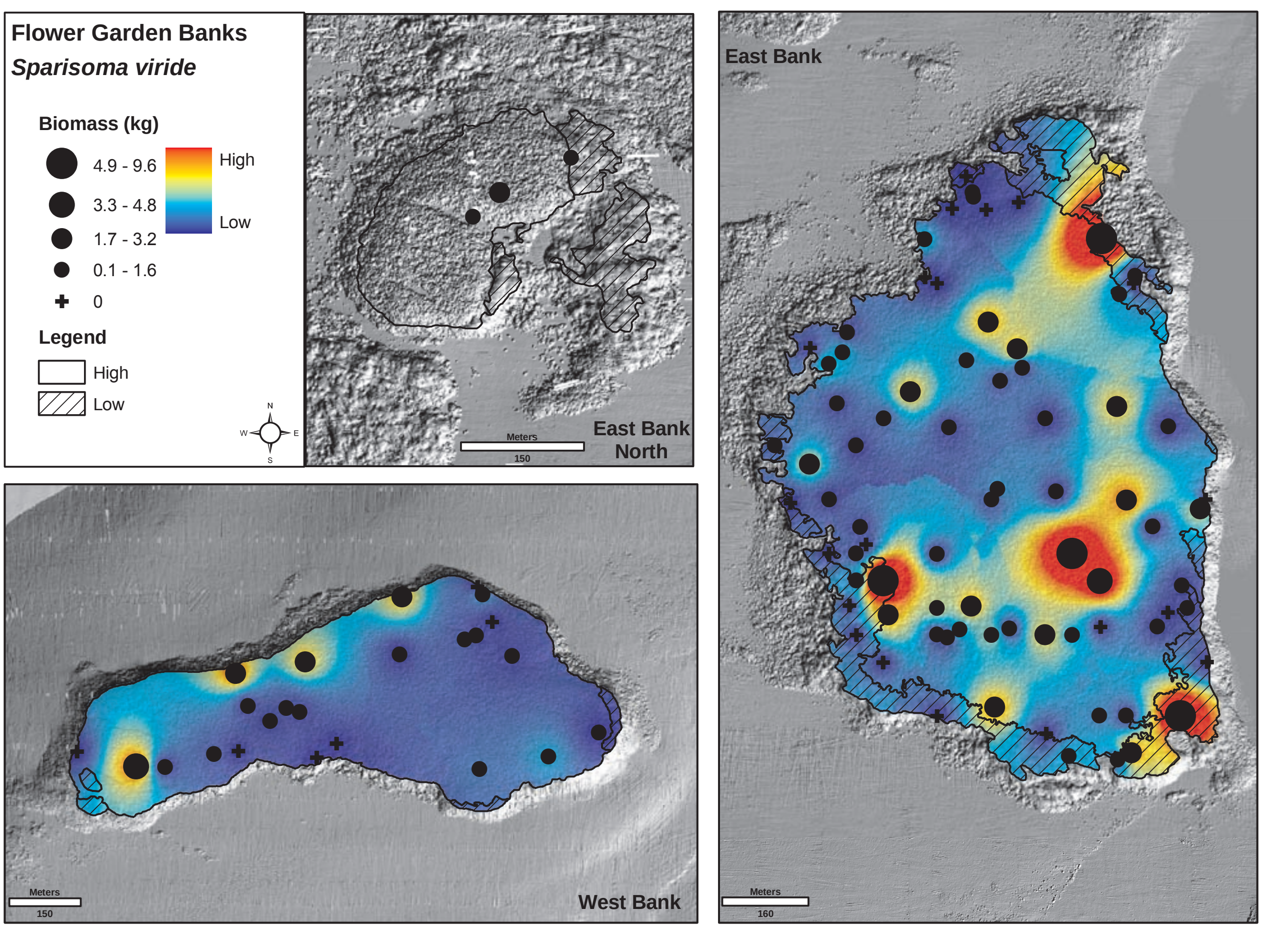


Figure 4.69. Observed (dots) and spatially interpolated biomass (kg) for stoplight parrotfish (Sp. viride) observed in CCMA surveys for 2006-2007.

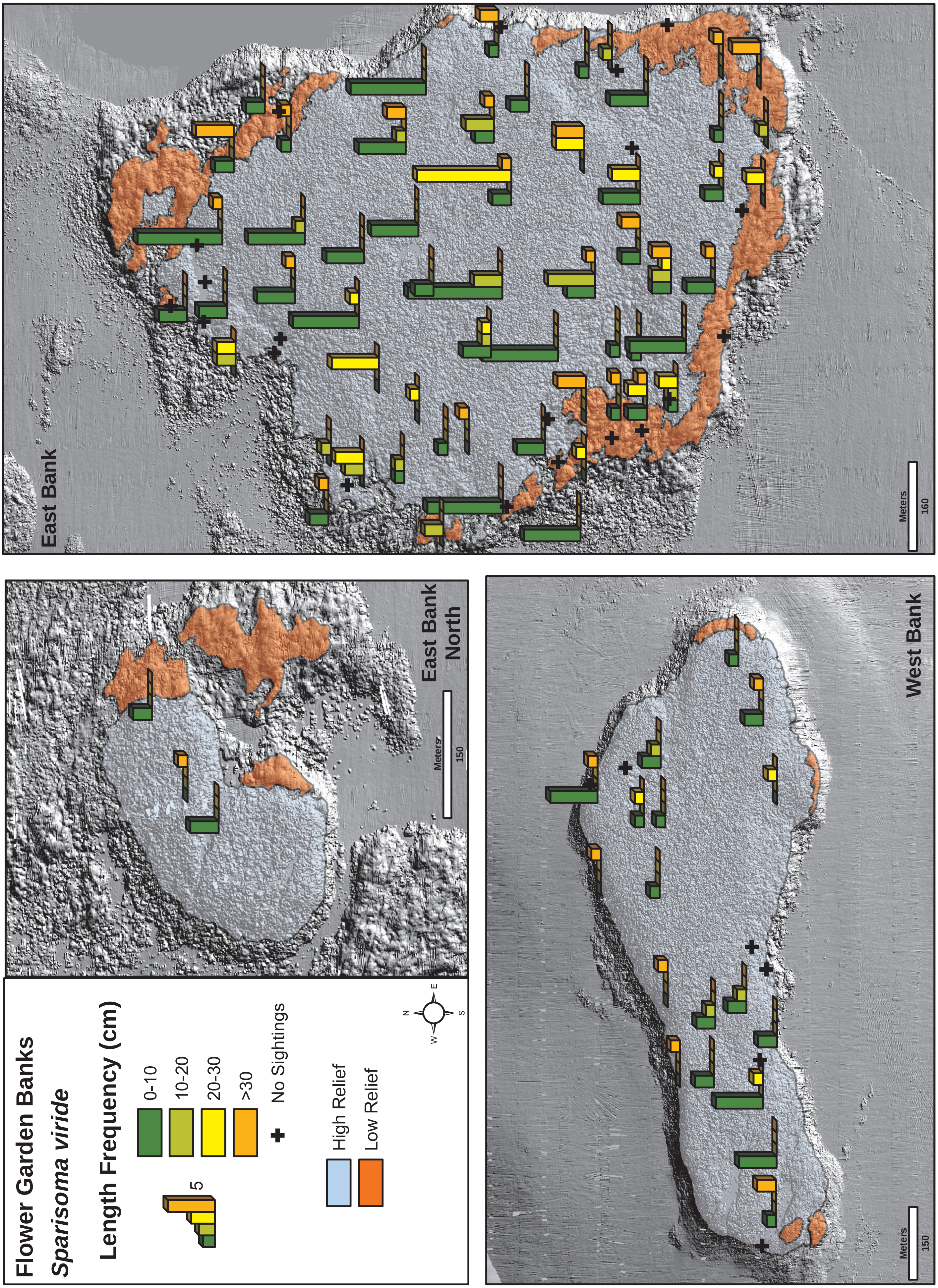


Figure 4.70. Spatial distribution for stoplight parrotfish (Sp. viride) size frequency observed in CCMA surveys 2006-2007. The tallest histogram bar in the legend represents five individuals.

4.3.3.3.2 Queen parrotfish (*Scarus vetula*)

Sc. vetula are a tropical/subtropical reef species ranging from Bermuda, Florida, and the Bahamas to South America, and including the Gulf of Mexico and Caribbean (Carpenter, 2002). *Sc. vetula* inhabit coral reefs and adjacent habitats at depths from 3-25 m. Maximum size has been reported to be approximately 50 cm, but they are more commonly observed to 32 cm. They are largely herbivorous, scraping algal mats from reef surfaces and occasionally bite at corals (Carpenter, 2002). Life history information is lacking for this species, therefore, size at sexual maturity and other characteristics are not available.

Terminal (left) and initial (right) phase *Scarus vetula* (CCMA)

Frequency of occurrence was high on both EB (64%) and WB (79%; Table 4.9). Observations made by REEF exhibited higher sighting frequency and were among the top 10 most frequently sighted fishes on both banks during 1995-2005 (Pattengill-Semmens, 2006). Sighting frequency was not provided by MMS, but documented high abundance with 2.37/100 m^2 on EB and 2.31/100 m^2 on WB (Precht et al., 2006).

Individuals from all size classes were observed on all bank/habitat type combinations; however, larger individuals (>30 cm) were observed on high relief habitats (Figure 4.71). Overall, 200 individuals were observed with a total biomass of 45.3 kg. Similar to stoplight parrotfish, *Sc. vetula* density was greater on EBH habitats, compared to EBL habitats (p=0.005). There was no significant difference between EBL and WBH. Mean density for each bank/habitat type was comparable among high relief habitats, while lower on low relief: EBH (2.2/100 m^2), EBL (0.6/100 m^2); and WBH (1.9/100 m^2). These values were similar to those reported by MMS (by bank comparison). Patterns of density throughout the sanctuary are displayed in Figure 4.72. On EB, density appeared to be greatest in the central/southern portion of the coral cap, while no obvious patterns emerged on WB.

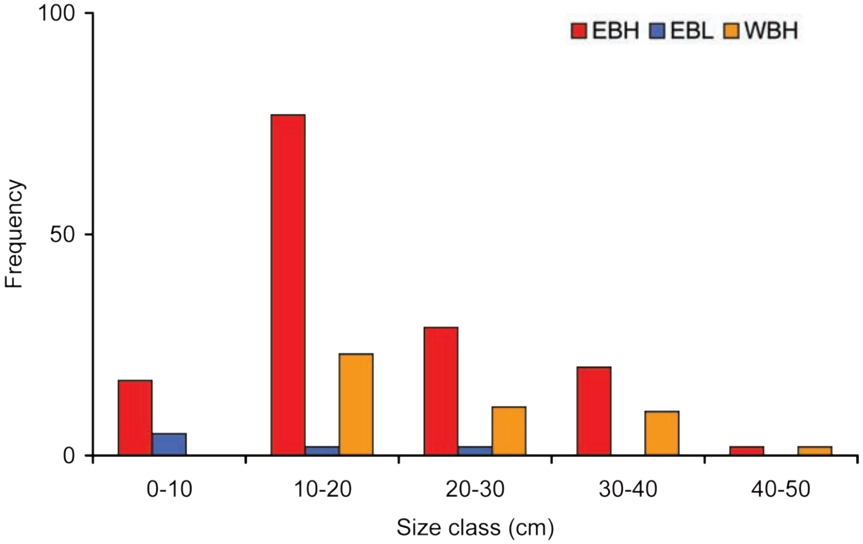


*Figure 4.71. Length frequency for queen parrotfish (*Sc. vetula*) observed in CCMA surveys for 2006-2007.*

Patterns of biomass (Figure 4.73) were similar to density patterns. Due to greater abundance on high relief habitats, biomass was significantly lower on EBL habitats (p=0.002) than on high relief habitats on either bank. This pattern was particularly influenced by larger numbers of smaller individuals (10-25 cm) which enhanced biomass considerably (Figure 4.74). Fish greater than 10 cm were most frequently observed on high relief habitat on both banks, but fish smaller than 10 cm displayed correlation with low relief habits. While the sample size on low relief habitats was small (n=9), these results indicate possible ontogenetic habitat shifts from low to high relief. *Sc. iseri* and *Sc. taeniopterus* were both shown to exhibit ontogenetic shifts in habitat preference in Puerto Rico (Christensen et al., 2003) and juvenile *Sc. vetula* there were also found to be very abundant on low relief habitats (Cerveny, 2006).

Both density and biomass were positively correlated with coral cover (ρ = 0.24 and ρ = 0.32, respectively). Density and biomass were inversely correlated with depth (ρ = -0.35 and ρ = -0.32, respectively).

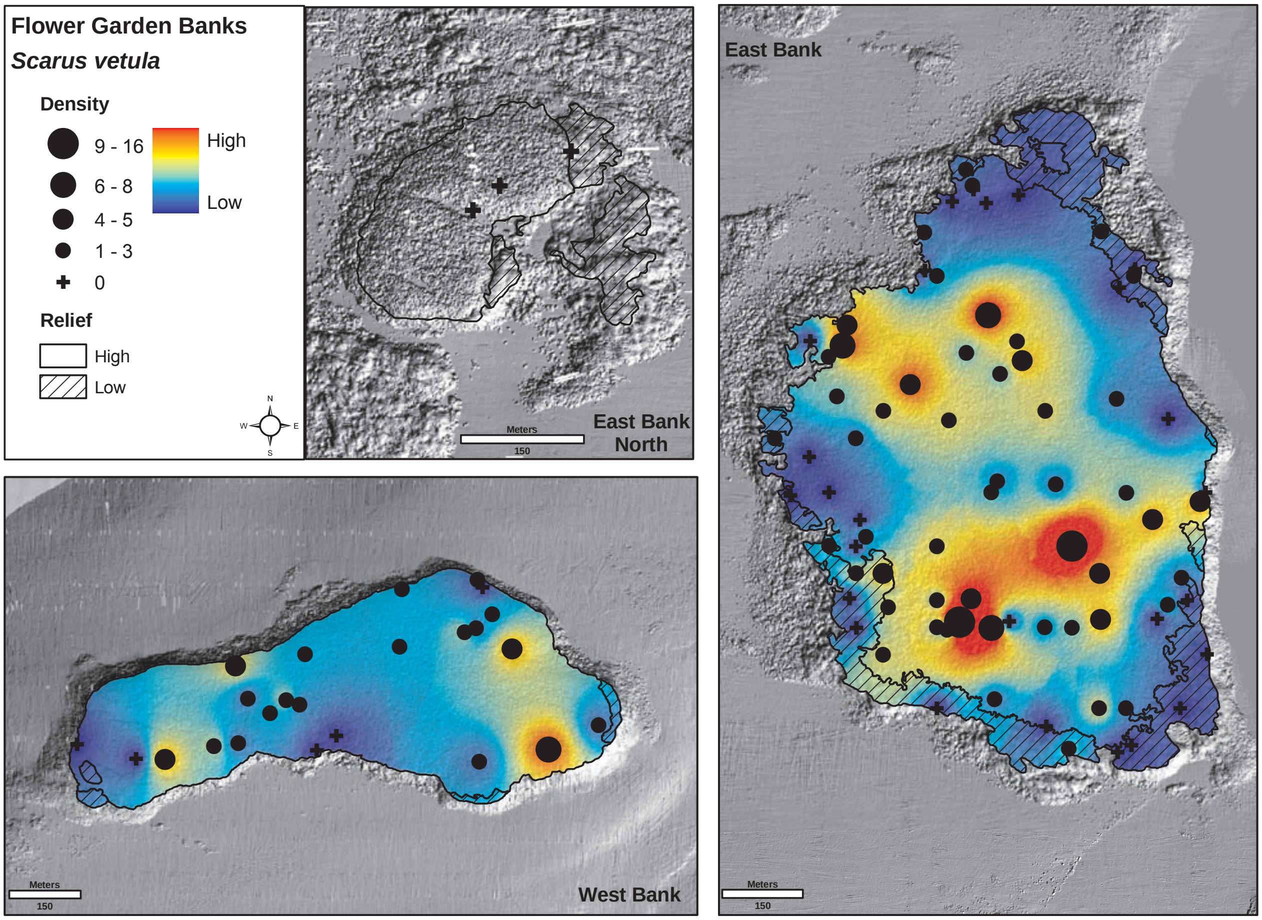


*Figure 4.72. Observed (dots) and spatially interpolated density (#/100 m^2) for queen parrotfish (*Sc. vetula*) observed in CCMA surveys for 2006-2007.*

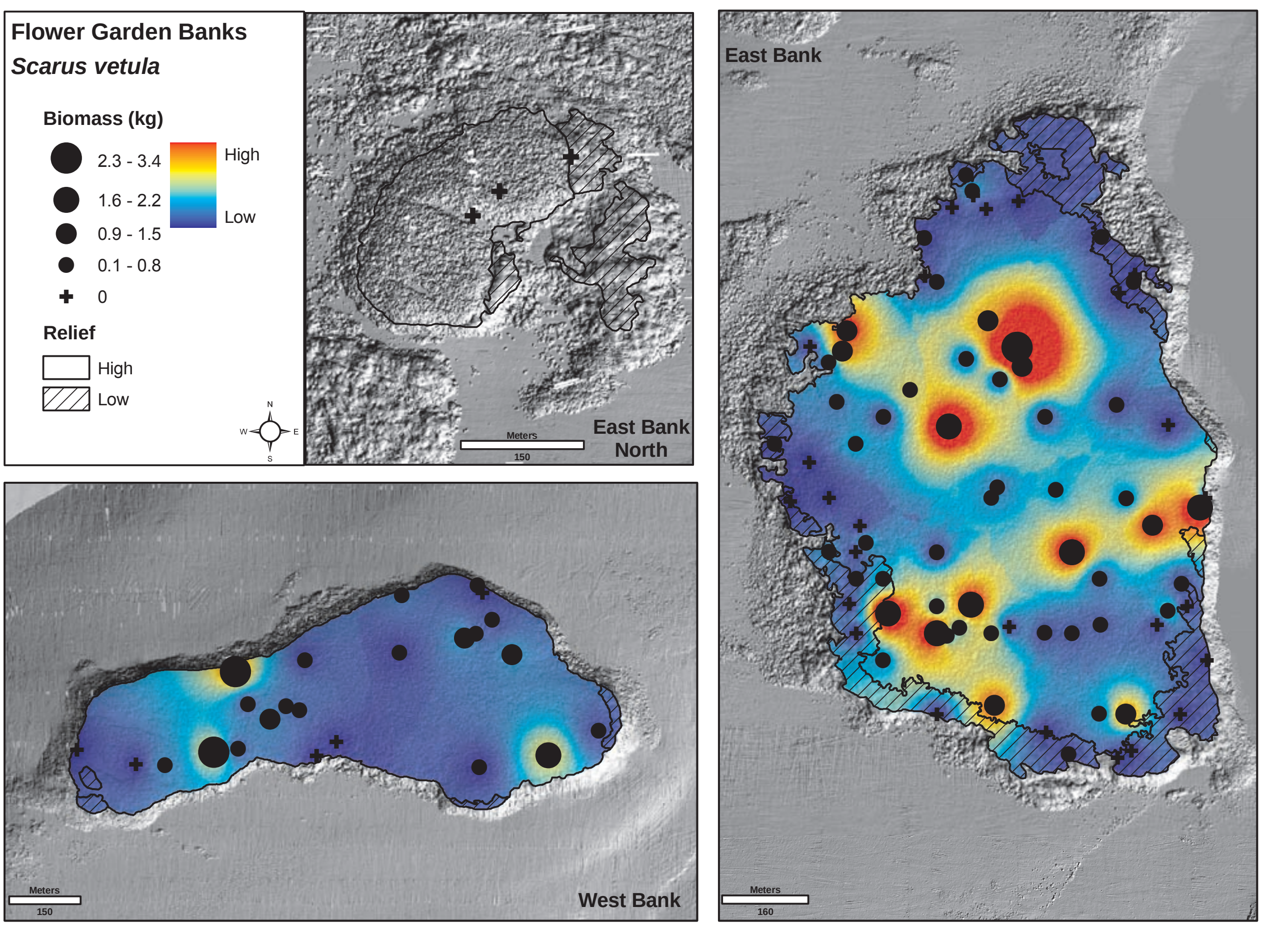


*Figure 4.73. Observed (dots) and spatially interpolated biomass (kg) for queen parrotfish (*Sc. vetula*) observed in CCMA surveys for 2006-2007.*

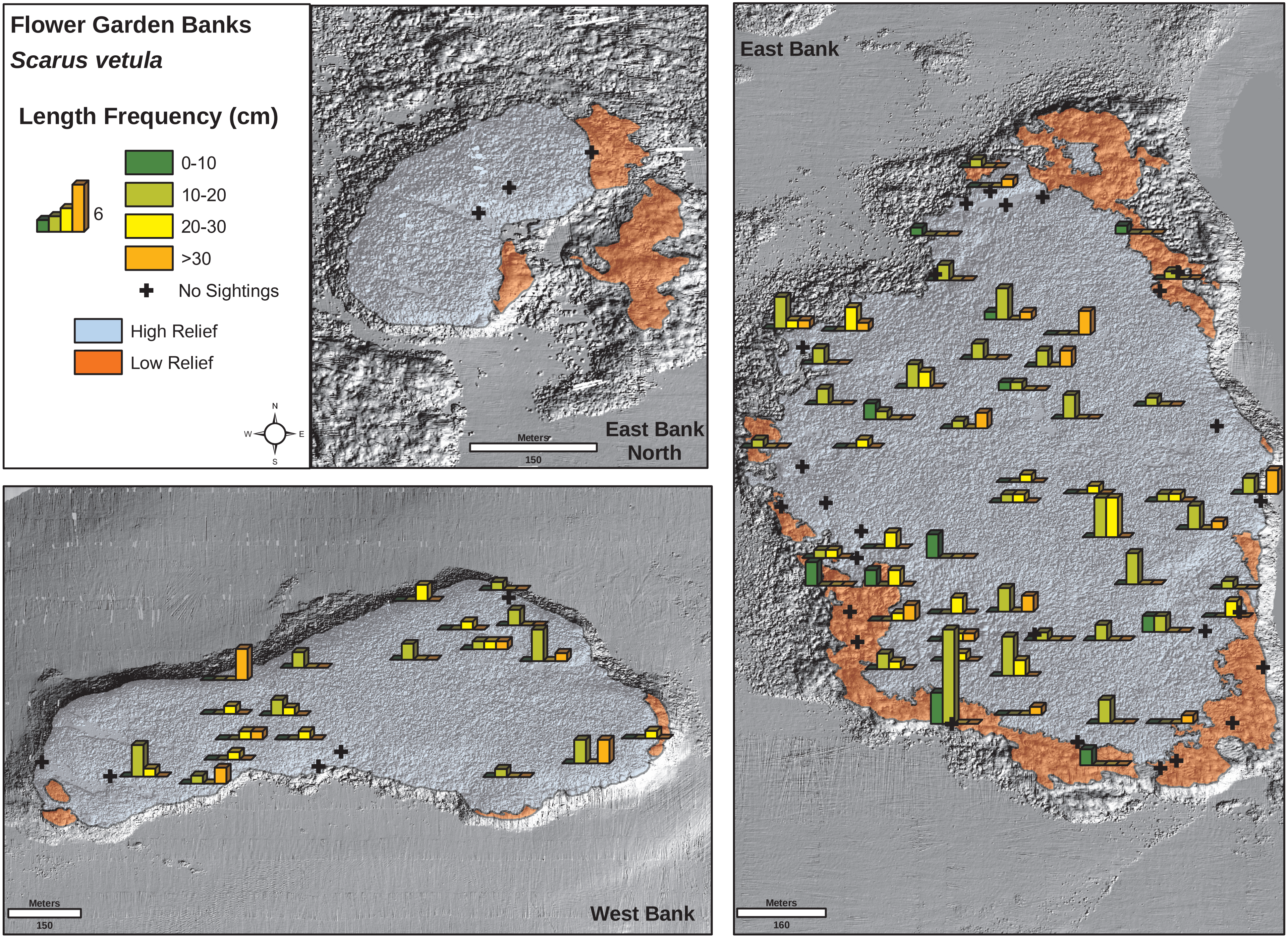


Figure 4.74. Spatial distribution for queen parrotfish (Sc. vetula) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents five individuals.

4.3.3.4 *Carangidae (Jacks)*

Carangids are pelagic over continental and insular shelves that occur globally in tropical to warm temperate seas (McEachran and Fechhelm, 1998). There are approximately 14 species in the Gulf of Mexico. Carangids are generally described as fast-swimming carnivores and pursuit predators; although smaller members of this family are planktivores. Carnivorous carangids are significant apex predators on reef ecosystems capable of consuming 30 tons of biomass annually (Honebrink, 2000). Jacks are highly valued food and gamefish in Hawaii and elsewhere (Thompson and Munro, 1974b); however, little is known about their biology and ecology.

Caranx latus (CCMA)

During 2006 and 2007 four carangid species (*Carangoides ruber*, *Caranx latus*, *Caranx lugubris* and *Caranx crysos*) exhibited moderate to low frequency of occurrence on either bank. These values were similar to those reported by REEF for bar jacks (*Cg. ruber*) and blue runners (*Cx. crysos*); however, horse-eye jacks (*Cx. latus*) were less frequently observed on both banks by CCMA (Table 4.10). In contrast, CCMA observed greater frequency of black jacks (*Cx. lugubris*) on WB. The crevalle jack (*Caranx hippos*) and yellow jack (*Carangoides bartholomaei*) were not observed by CCMA.

Table 4.10. Sighting frequency of all jacks (Carangidae) species from CCMA, REEF and MMS surveys. REEF estimates (means) are from expert surveys only. MMS data only reflect presence/absence indicated by +/-.

	East Bank			West Bank		
Species	**CCMA**	**REEF**	**MMS**	**CCMA**	**REEF**	**MMS**
Carangoides bartholomaei	0	3.95	-	0	3.7	-
Carangoides ruber	32.1	64.13	+	37.5	69.28	+
Caranx crysos	0	1.17	+	4.16	4.3	+
Caranx hippos	0	18.37	+	0	20.08	-
Caranx latus	11.11	49.49	-	12.5	45.28	-
Caranx lugubris	16.05	24.03	+	37.5	34.93	-

Overall, 758 individuals were observed with a total biomass of 151 kg. Carangids comprised approximately 2% of the total abundance and 6% of the total biomass during the study period. *Cg. ruber* was the dominant carangid species (n=641) and *Cx. latus* exhibited the greatest biomass (80 kg).

Spatial patterns of density (individuals/100 m^2; Figure 4.75) were dominated by *Cg. ruber* (11.1/100 m^2 on EB and 3.8/100 m^2 on WB) and no significant correlations were observed among bank/habitat types. In general, density was evenly spread throughout East and West Banks, with the exception of several locations with large schools. Areas of high biomass (Figure 4.76) were reflected by larger species, *Cx. latus* and *Cx. lugubris*, and were not correlated with bank or habitat type. Patterns of density were not correlated with coral and algae percent cover or depth, although there was a general pattern of increased abundance with greater percent coral cover.

Carangoides ruber (CCMA)

Figure 4.77 displays length frequency across the sanctuary for all carangids observed on transects. Peak abundance is noted in the center of EB, reflective of the large school of *Cg. ruber*. No spatial patterns are obvious in relation to carangid size structure within the sanctuary.

As expected for pelagic species, none of the Carangidae species exhibited a significant correlation with coral, macroalgae or depth.

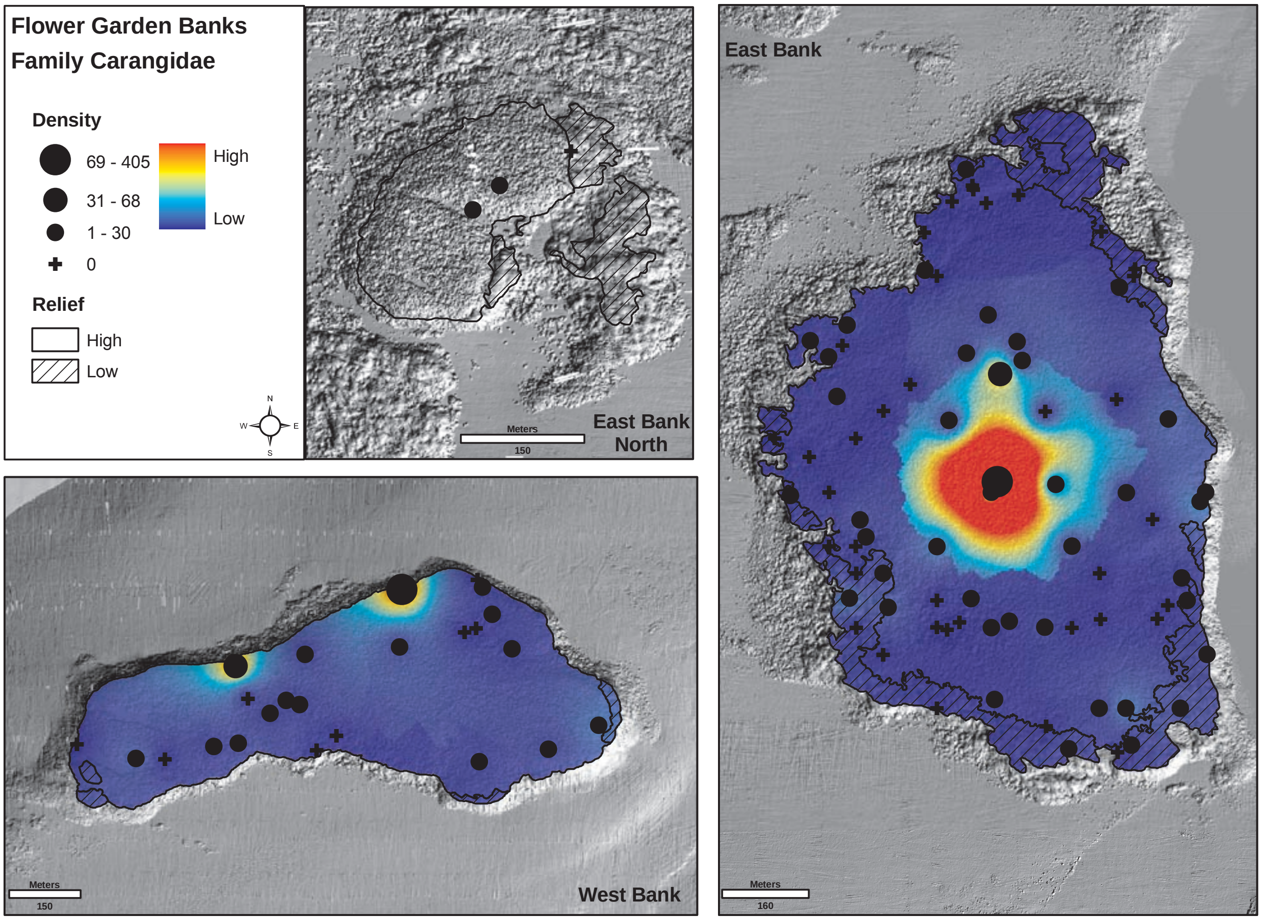


Figure 4.75. Observed (dots) and spatially interpolated density (#/100 m^2) for all jacks (Carangidae) observed in CCMA surveys for 2006-2007.

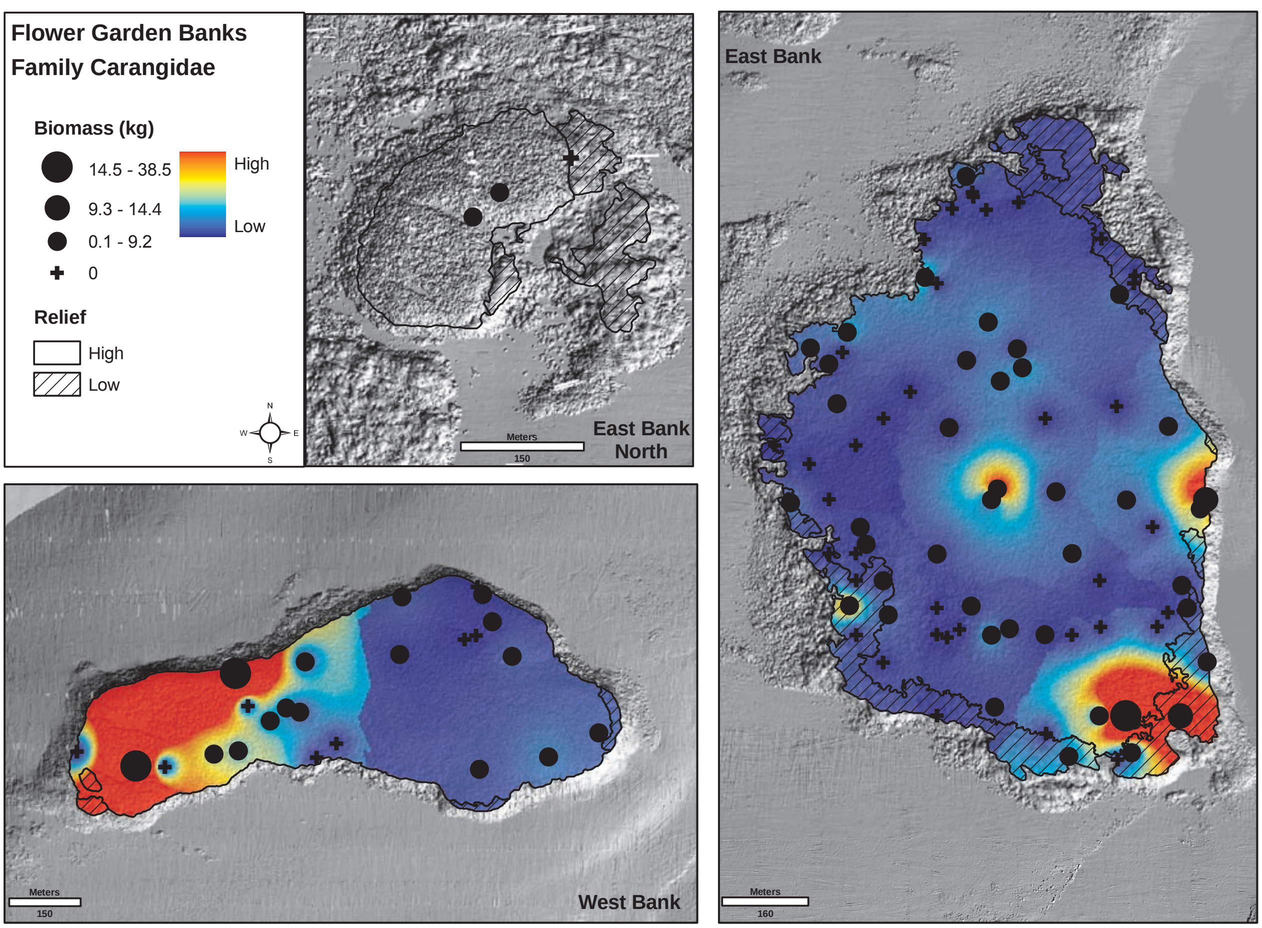


Figure 4.76. Observed (dots) and spatially interpolated biomass (kg) for all jacks (Carangidae) observed in CCMA surveys for 2006-2007.

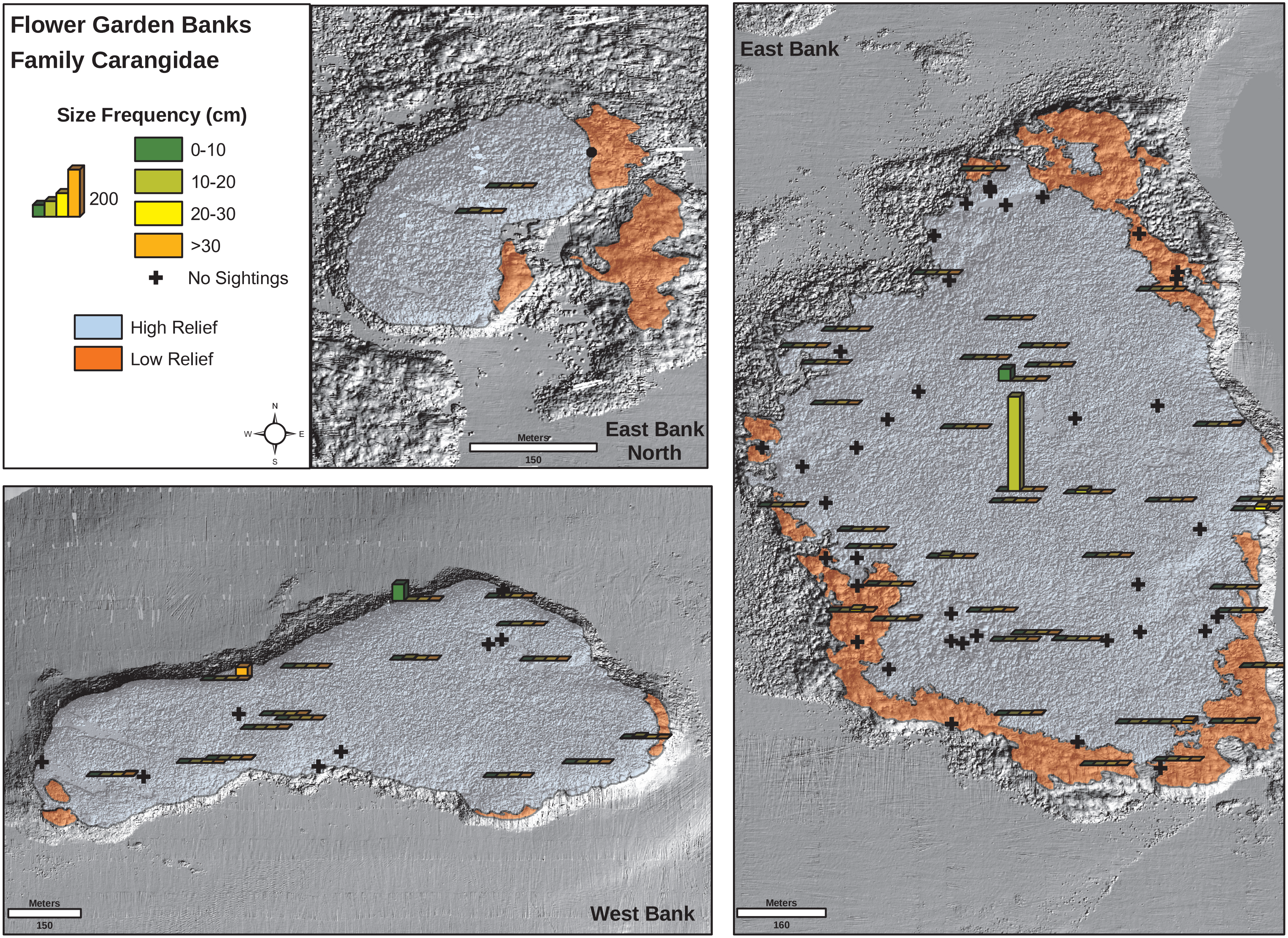
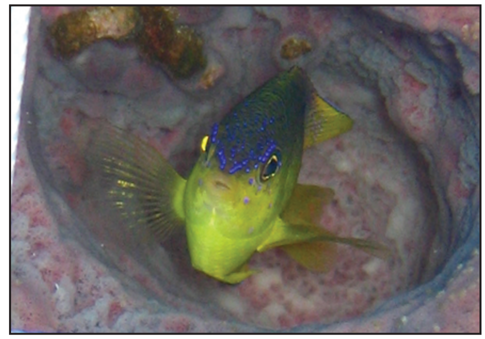


Figure 4.77. Spatial distribution for all jacks (Carangidae) size frequency observed in CCMA surveys for 2006-2007. The tallest histogram bar in the legend represents 200 individuals.

4.3.3.5 *Pomacentridae (Damselfish)*

Damselfish are one of the most diverse reef fish families in tropical seas. Species of the family Pomacentridae are found in coastal waters associated with rocky substrates, usually occurring at moderate depths (20-30 m) and often assembling in large schools (Allen, 1975; Menezes and Figueiredo, 1985). Many of the damselfish species are highly territorial (Randall, 1996). Food habits vary throughout the taxa, but most are herbivorous (Allen, 1991). Damselfish, as well as other herbivorous species, play a significant role in the function of coral reefs where it has been shown that damselfish have suppressed coral recruitment by cultivating algal mats (Birkeland, 1977). In contrast, increases of coral recruitment have also been observed within damselfish territories (Sammarco and Carleton, 1981) and coral survival and zonation has been related to damselfish presence on a reef (Wellington, 1982).



Stegastes variabilis (CCMA)

Approximately 30% of the total abundance of fishes observed on CCMA surveys were comprised of damselfish. Overall, 11,301 individuals comprising 12 species were observed and comprising 630 kg of biomass. Sighting frequency was comparable to most REEF damselfish estimates (Table 4.11) and large deviations may be a result of differences in sampling methods. As such, *A. saxatilis*, yellowtail (*Microspathodon chrysurus*), dusky (*Stegastes adustus*) and cocoa (*Stegastes variabilis*) damselfish sightings were much lower than REEF estimates for both banks. FGBNMS LTM surveys documented 12 species.

Table 4.11. Sighting frequency of all Pomacentridae species from CCMA, REEF and MMS surveys. REEF estimates (means) are from expert surveys only. MMS data only reflect presence/absence indicated by +/-.

	East Bank			West Bank		
Species	**CCMA**	**REEF**	**MMS**	**CCMA**	**REEF**	**MMS**
Abudefduf saxatilis	1.23	33.75	+	12.50	24.28	+
Chromis cyanea	60.49	90.56	+	83.33	90.68	+
Chromis insolata	74.07	36.36	-	75	74.33	+
Chromis multilineata	80.25	97.13	+	66.67	94.98	+
Chromis scotti	59.26	69.53	-	79.17	74.63	+
Microspathodon chrysurus	8.64	85.27	+	20.83	62.10	+
Stegastes adustus	12.35	51.84	-	16.67	32.85	+
Stegastes diencaeus	7.41	12.43	-	4.17	10.70	+
Stegastes leucostictus	8.64	5.58	+	25	7.23	-
Stegastes partitus	93.83	93.16	+	91.67	93.35	+
Stegastes planifrons	98.77	92.21	+	100	92.98	+
Stegastes variabilis	41.98	81.64	+	8.33	66.95	+

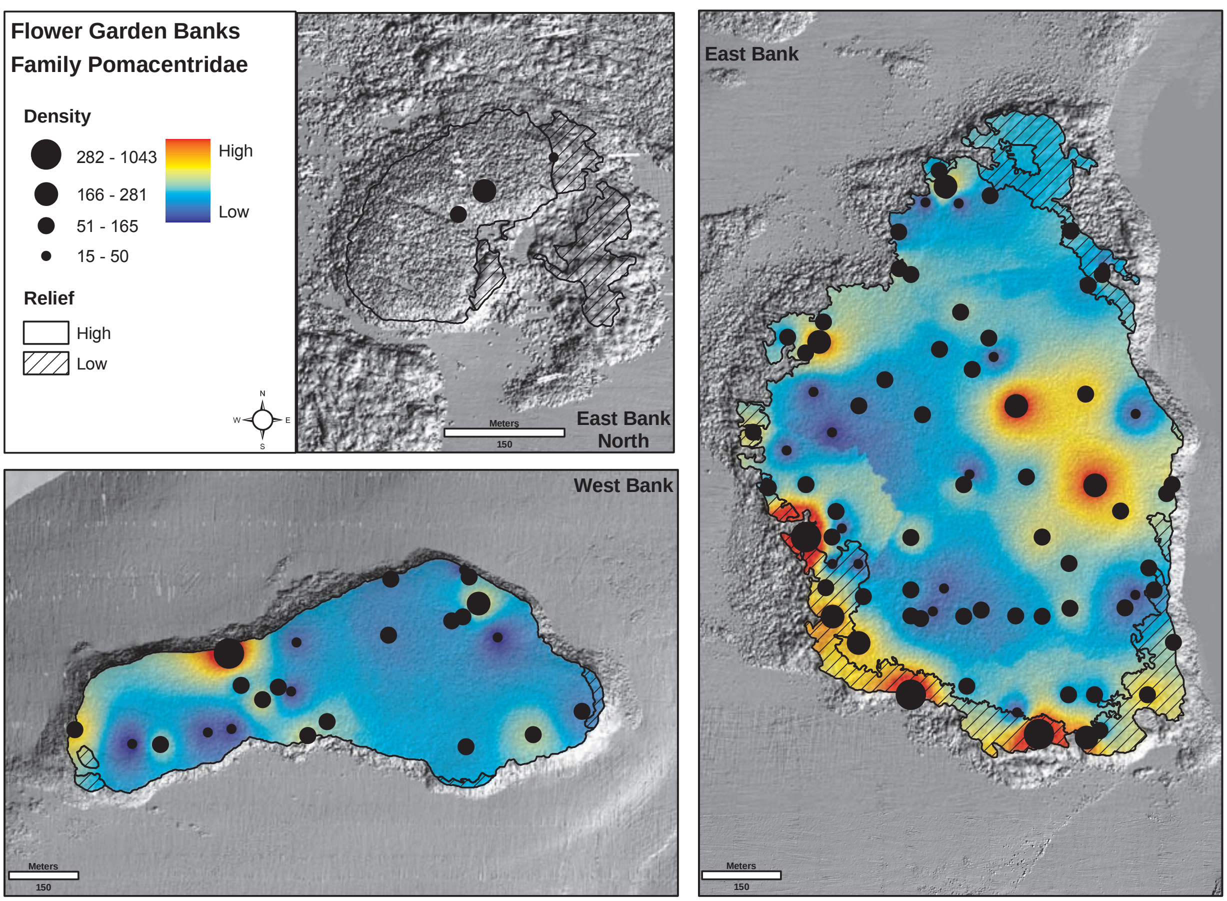


Figure 4.78. Observed (dots) and spatially interpolated density (#/100 m²) for all Pomacentridae observed in CCMA surveys for 2006-2007.

Pomacentrid density (individuals/100 m^2) was comparable on both banks and among all habitat types (Figure 4.78). *C. multilineata* exhibited the highest density (approximately 40/100 m^2 on both banks) among all pomacentrids and was the second most abundant fish species observed on WB and fourth most abundant on EB (Tables 4.4 and 4.5). The sunshine fish (*Chromis insolata*) was also highly abundant on EB (22/100 m^2) but exhibited lower density on WB (13/100 m^2). Bicolor (*Stegastes partitus*) and threespot (*Stegastes planifrons*) damselfish exhibited densities greater than 15/100 m^2 on EB; however, density for *S. partitus* was much lower on WB (5.8/100 m^2). The remaining pomacentrid species exhibited density less than 10/100 m^2 and did not exhibit any considerable differences between banks.

Total pomacentrid abundance did not exhibit a significant relationship with depth or coral and macroalgal cover; however, correlations were evident at the species level that provides significant insight into habitat partitioning within the sanctuary (Figure 4.79). Two species (*C. multilineata* and *S. planifrons*) biomass and density were positively correlated with coral cover, while an inverse correlation was observed for *S. partitus*. *C. multilineata* and *S. planifrons* were also inversely correlated with macroalgae cover (Figure 4.79). Depth was positively correlated with two species biomass and density, the purple reeffish (*Chromis scotti*) and *C. insolata*, while four species were inversely correlated (*A. saxatilis*, *C. multilineata*, *M. chrysurus* and *S. adustus*). Neither biomass nor density were correlated with depth, coral cover and macroalgae cover for *Chromis cyanea*, *Stegastes diencaeus*, *Stegastes leucostictus* and *S. variabilis*.

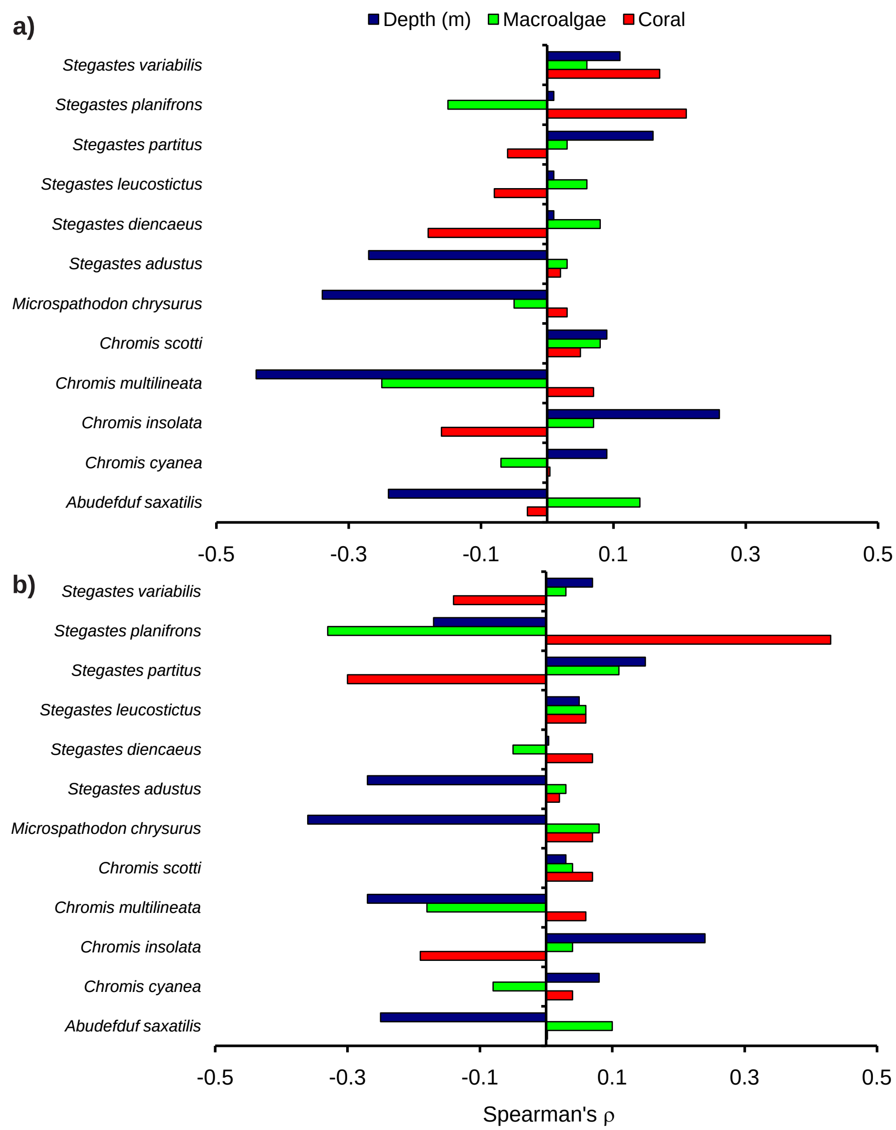


Figure 4.79. Pearson's correlation (ρ) results displaying all Pomacentridae species (a) density and (b) biomass correlations with depth, coral cover and macroalgae cover.

4.3.3.5.1 <u>Threespot damselfish (*Stegastes planifrons*)</u>

Stegastes planifrons (E. Hickerson,FGBNMS)

S. planifrons are a herbivorous and territorial damselfish that is very abundant on Caribbean reefs. They typically occur on shallow reefs from 1-30 m and are often found near staghorn coral (*Acropora cervicornis*; Randall, 1996), star corals (*Montastraea* spp.) and leaf/plate corals (*Agaricia* spp.). *S. planifrons* have been reported to 19 years of age in Florida and 17 years at FGBNMS making them a long-lived resident of the banks (Caldow and Wellington, 2003).

S. planifrons were highly abundant on both banks ranking eighth in mean density (17.8/100 m^2) on EB. This estimate was nearly twice as high (9.57/100 m^2) as that estimated by MMS (Precht et al., 2006). Similarly, mean density on WB (16.5/100 m^2) was nearly double the estimate provided by MMS (9/100 m^2). *S. planifrons* ranked 14th overall in sighting frequency and exhibited high relative density among REEF surveys on both EB and WB from 1995-2005. Overall, 1,845 individuals were observed on 104 of 105 transects. Total biomass amounted to 12.9 kg.

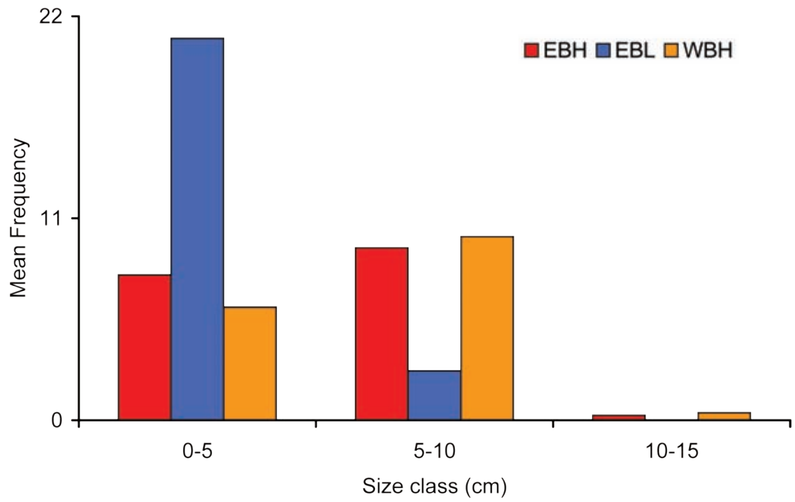


*Figure 4.80. Mean length frequency for threespot damselfish (*S. planifrons*) observed in CCMA surveys for 2006.*

S. planifrons was distributed widely throughout the sanctuary. Individuals <5 cm were dominant on EBL while fish >10 cm were more abundant on high relief habitats on both banks (Figure 4.80). Density was not significantly different among bank/habitat type combinations (Figure 4.81).

Density and biomass were positively correlated with percent coral and macroalgal cover (Figure

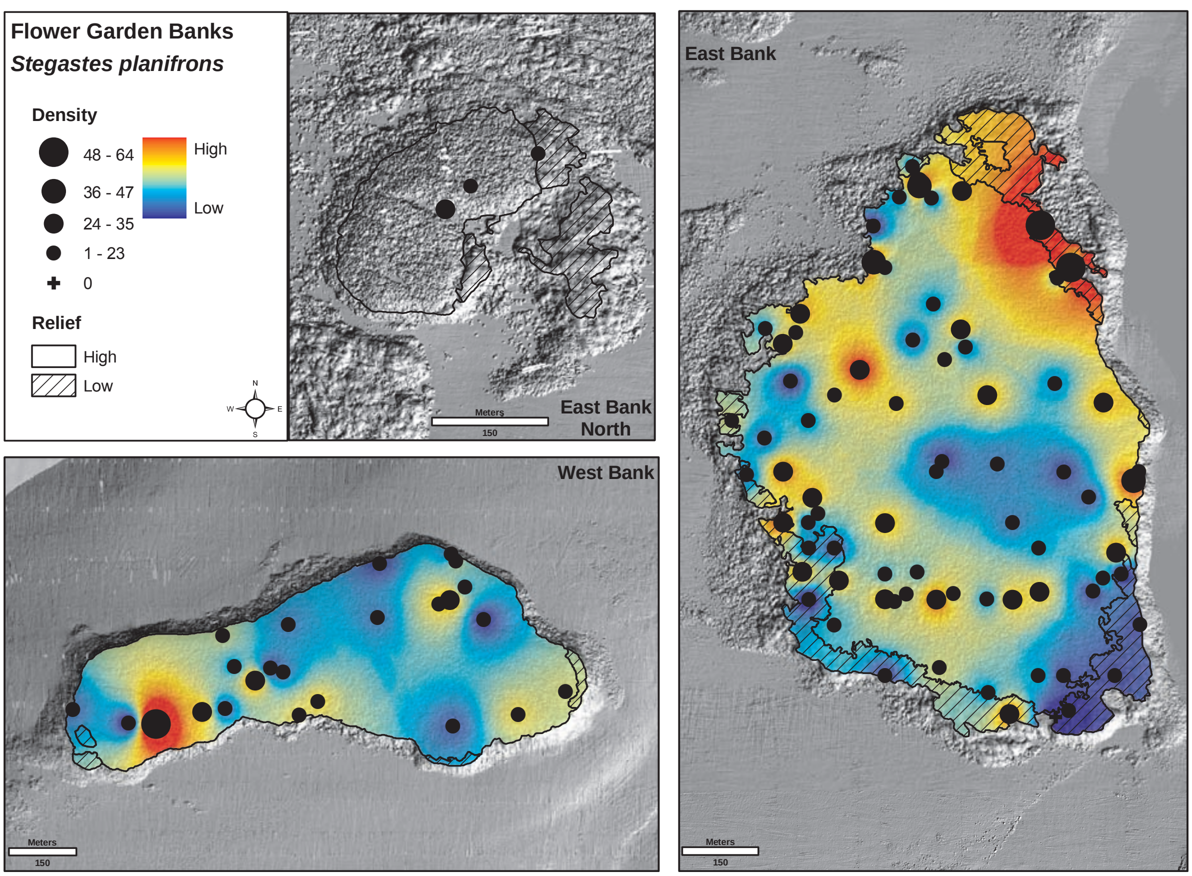


*Figure 4.81. Observed (dots) and spatially interpolated density (#/100 m^2) for threespot damselfish (*S. planifrons*) observed in CCMA surveys for 2006-2007.*

4.79). Density was not correlated with depth; however, biomass was inversely correlated.

Clepticus parrae (CCMA)

4.3.4 Fish assemblages

Cluster analysis identified three distinct fish assemblages within the sanctuary (Figure 4.82). A deep water assemblage contained a small number of species at sites with average depths of 32 m on EB. Typically these sites occurred on low relief habitat and were considerably lower in coral cover (Table 4.12) and differing coral species composition than the other assemblages. These deep, low relief sites were also higher in macroalgae and sponge cover. The deep water assemblage was typically observed on EBL which was characterized by low coral cover. Members of this assemblage including, *Sp. atomarium*, *S. planifrons*, *S. variabilis*, goldspot goby (*Gnatholepis thompsoni*) and yellowhead jawfish (*Opistognathus aurifrons*) were dominant among these habitats and rare elsewhere. Other species, such as *S. partitus* and *C. insolata*, were linked to this assemblage, but were also commonly found among other habitat types.

Table 4.12. Fish assemblages and associated habitat characteristics. MA=macroalgae.

	Mean depth (m)	Mean coral cover (%)	Mean MA cover (%)	Mean sponge cover (%)
Shallow	24	58	25	<1
Mixed	29	61	22	<1
Deep	32	30	45	2

A shallow water assemblage contained the most species and encompasses the central shallow portion of the coral caps. This assemblage inhabits areas of shallow depth, high coral cover consisting primarily of boulder star corals (*Montastraea* spp.) and brain coral (*Diploria strigosa*), low macroalgal and sponge cover (Table 4.12). This diverse assemblage includes most of the parrotfish (notably *Sp. viride*, *Sc. iseri* and *Sc. vetula*), *S. planifrons*, *C. multilineata*, black durgon (*Melichthys niger*), sharpnose puffer (*Canthigaster rostrata*) and *C. parrae*. The third assemblage also contained a high diversity of species and overlapped substantially with many species from the deep and shallow assemblages. Overlapping species, such as *S. partitus*, *C. insolata*, *C. multilineata* and parrotfish species were common throughout the sanctuary but not as dominant numerically as in other assemblages. This assemblage could be termed a transition assemblage that resides between the shallow and deep assemblages, primarily along the transition of low and high relief habitats (Figure 4.82). These sites also exhibited high coral cover (characterized by the plating forms of *Montastraea* spp.), and low macroalgal and sponge cover (Table 4.13). Nearly all the piscivores were contained in this assemblage (groupers, jacks, snappers) as well as Spanish hogfish (*Bodianus rufus*), *P. furcifer* and surgeonfish (*Acanthurus* spp.).

Canthigaster rostrata (CCMA)

PCA results confirmed the species structure for the deep and shallow assemblages. The mixed assemblage was less well defined; however, dominant species listed above were still the primary species comprising the assemblage. PCA indicated that *S. planifrons* was a dominant member of the deep assemblage while cluster analysis identified it as a primary component of the shallow assemblage.

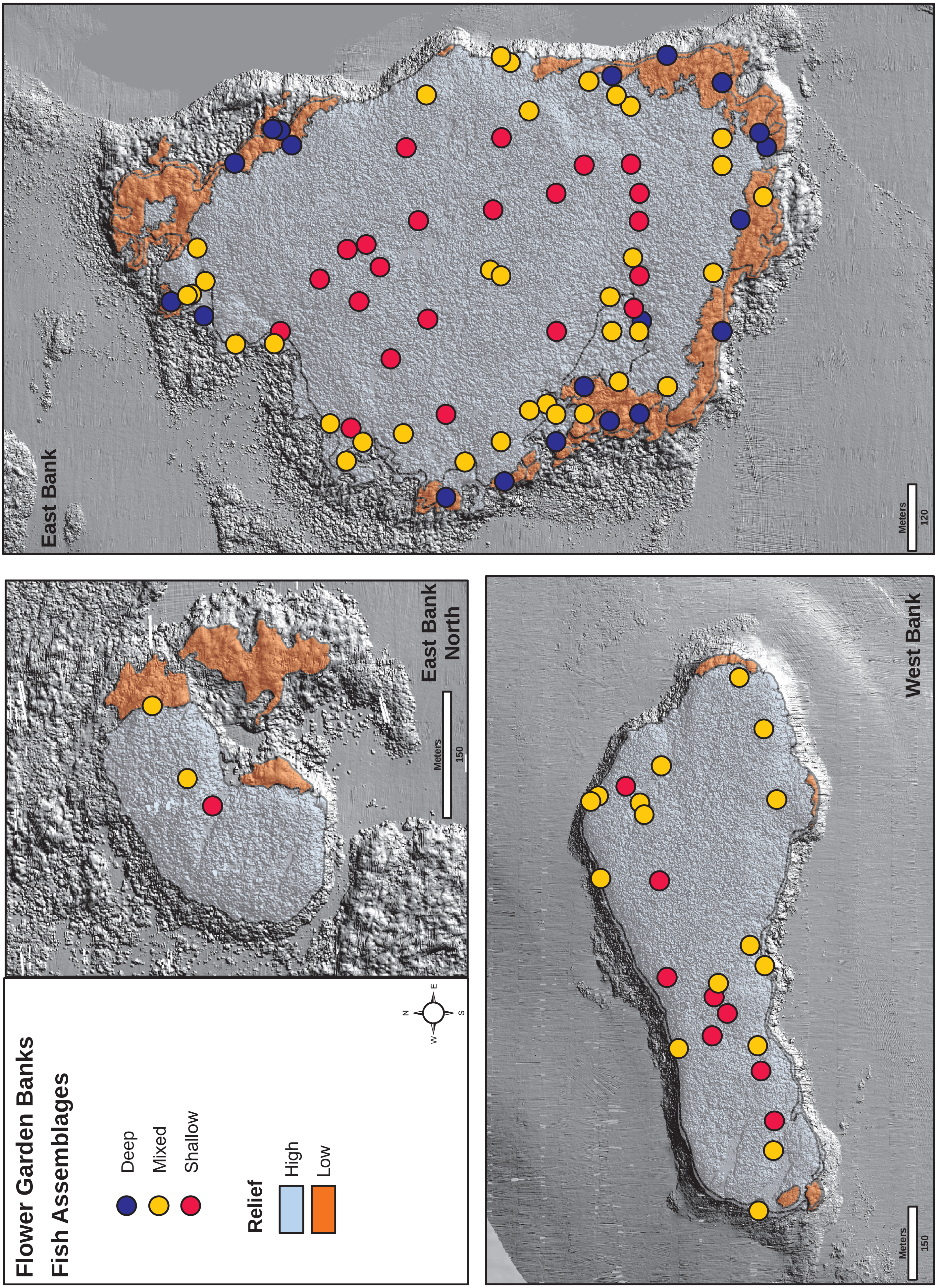


Figure 4.82. Spatial distribution of fish assemblages at CCMA determined by cluster analysis.

4.4 COMPARISON OF FGBNMS COMMUNITY STRUCTURE AND SELECT CARIBBEAN LOCATIONS

Impacts such as chronic over-fishing, pollution, climate change and disease have deteriorated reefs globally. Resulting losses observed in coral cover and large predators have serious ramifications to supporting ecological function and diversity in reef ecosystems (Gardner et al., 2003; Sandin et al., 2008). As a mechanism for estimating the measure of these impacts, scientists have provided examples of comparatively pristine reefs in the Pacific Ocean (Friedlander and DeMartini, 2002). Few examples exist (e.g., Bonaire) or have yet to be described in the tropical western Atlantic Ocean. While we recognize the physical and geomorphological differences between the FGBNMS coral reefs and those in the Caribbean, the similarities in marine fauna, provide us an excellent opportunity to make similar comparisons between impacted and relatively non-impacted systems.

Fish survey information from CCMA monitoring during 2006 are compared to three sites (La Parguera, Puerto Rico; St. Croix and St. John, USVI) that have been extensively monitored by CCMA using the same methods. To obtain sufficient sample size for analysis 2003-2006 data from the Caribbean locations was pooled. Preliminary results have been reported (Caldow et al., 2008) and herein a more detailed comparison is made between these locations.

Epinephelus striatus in St. John. (CCMA)

Mangroves in Puerto Rico (CCMA)

Acropora palmata, St. Croix. (CCMA)

4.4.1 Methods

4.4.1.1 *Study Areas*

The FGBNMS study area has been previously described in Chapter 2 and the Caribbean study areas in Chapter 3.

4.4.1.2 *Survey Data*

Section 4.2.1 describes sampling methods and Appendix A further details specifics of data collection. See Chapter 2 for information on site selection. Only sites at depths greater than 18 m (60 ft) were included from the Caribbean data to match bathymetric conditions at FGBNMS. As such, total sites for comparison from each location were: FGBNMS (n=73), La Parguera (n=61), St. Croix (n=66) and St. John (n=222). It must be noted that dives conducted in La Parguera and St. Croix typically do not exceed 27 m (90 ft) and so differences in depth profiles may contribute to observed differences at these locations. Other factors differing between study locations that may impact observed differences between communities include: oceanography, local geology, and availability and configuration of habitat types.

4.4.1.3 *Data Analysis*

FGBNMS community metrics (richness, density and biomass) were compared to those from La Parguera, St. Croix and St. John.

Species richness and density data did not meet homogeneity of variance assumptions using Bartlett's test, thus nonparametric Kruskal-Wallis tests were used to examine potential differences in community metrics between locations. Where differences were statistically significant, pairwise comparisons were conducted using the Nemenyi test. Biomass data were log transformed to meet normality assumptions and ANOVA was used to examine differences between locations. Tukey-Kramer HSD test was used to evaluate pairwise statistical significance. All analyses were performed using JMP© statistical software (SAS Institute Inc., 2000). Composition and biomass of trophic groups were also compared between locations. As previously mentioned, four trophic groups were classified (herbivore, invertivore, piscivore, zooplanktivore). For this analysis, piscivores have been further categorized as apex predators and other piscivores. Lastly, abundance, biomass and size frequency of groupers, snappers and parrotfish were compared between locations.

4.4.2 Results and Discussion

All community metrics exhibited statistically significant differences for mean density, biomass and species richness among locations ($p<0.0001$ for each). Mean density of fishes was greatest at St. John, but not significantly greater than FGBNMS. Fish density at both sites was significantly greater than St. Croix and La Parguera; density at St. Croix was significantly greater than La Parguera (Figure 4.83a).

Biomass (*M. birostris* excluded) was significantly greater ($p<0.0001$) at FGBNMS (22.8 kg/100 m^2) than any of the Caribbean locations. In fact, mean biomass was greater at FGBNMS than the combination of all three Caribbean sites (Figure 4.83b). The presence of large groupers (*Mycteroperca*, *Dermatolepis*, *Epinephelus* and *Cephalopholis* spp.) at FGBNMS accounts for a large portion of this discrepancy. Mean biomass at FGBNMS (5.06 kg) was nearly seven times greater than St. Croix (0.78 kg), eight times greater than St. John (0.65 kg) and 46 times greater than Puerto Rico (0.11 kg).

Species richness was greatest at FGBNMS; however, it was not significantly greater than St. John. Richness at FGBNMS and St. John were significantly greater than St. Croix and La Parguera; La Parguera species richness was significantly greater than St. Croix (Figure 4.83c).

Greater density of fish at FGBNMS was driven primarily by piscivores, invertivores, and zooplanktivore dominance. Piscivore density was greater than that observed at St. John (although not statistically significant), but both were significantly greater ($p<0.0001$) than St. Croix and Puerto Rico (Figure 4.84). Invertivore density was most abundant at St. John; however they were not significantly greater than that observed at FGBNMS. Invertivore density at St. John and FGBNMS was significantly greater ($p<0.0001$) than St. Croix and Puerto Rico. Herbivore density at FGBNMS was significantly lower ($p<0.0001$) than all Caribbean locations, while St. John exhibited the highest herbivore density.

Biomass at FGBNMS was dominated by apex predators (Table 4.13), comprising 36% of the total observed biomass (Figure 4.85). This percentage was nearly twice that of observations at St. John (20%) and Puerto Rico (16%), and six times greater than St. Croix (6%).

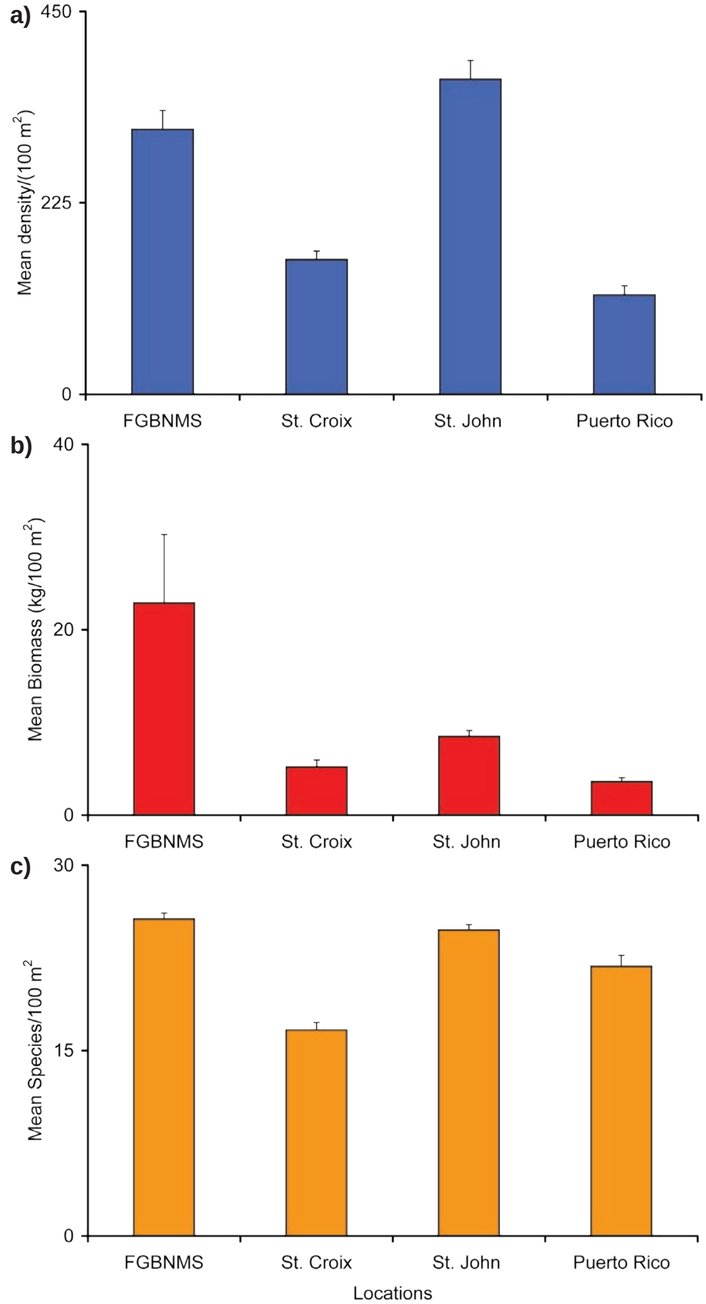


Figure 4.83. Community metric comparisons (± SE) for a) density, b) biomass, c) species richness between FGBNMS, St. Croix, St. John and Puerto Rico.

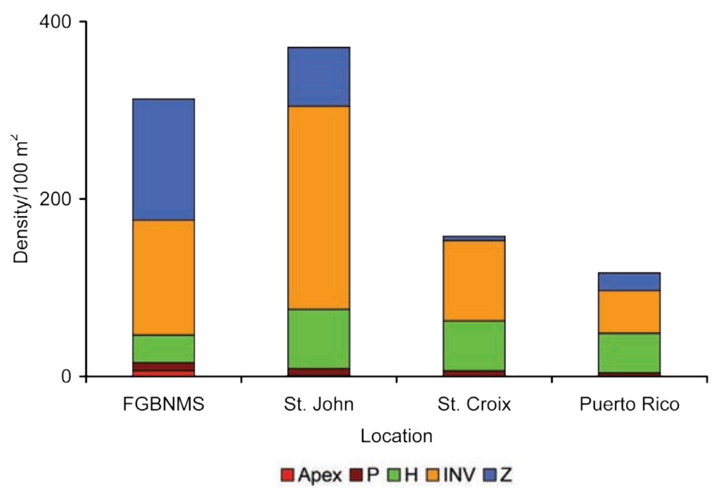


Figure 4.84. Mean density of apex predators (Apex), piscivores (P), herbivores (H), invertivores (INV), and zooplanktivores (Z) at FGBNMS, St. John, St. Croix and Puerto Rico.

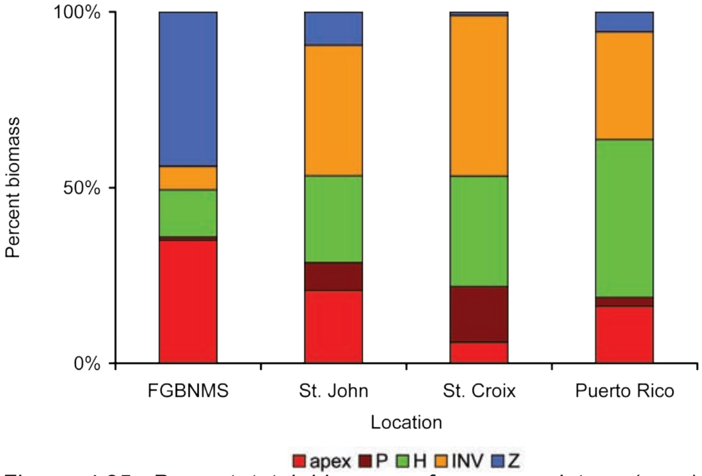


Figure 4.85. Percent total biomass of apex predators (apex), piscivores (P), herbivores (H), invertivores (INV) and zooplanktivores (Z) at FGBNMS, St. John, St. Croix and Puerto Rico.

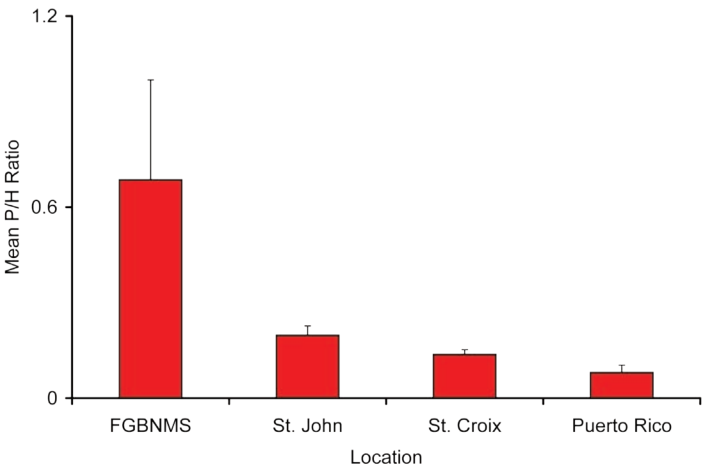


Figure 4.86. Mean piscivore/herbivore density ratio (± SE) at FGBNMS in comparison to locations in St. John, St. Croix and Puerto Rico.

Table 4.13. Apex predators observed at FGBNMS and Caribbean locations.

Species	FGB	PR	STJ	STC
Carangoides bartholomaei			X	
Carangoides ruber	X	X	X	X
Caranx crysos	X		X	X
Caranx latus	X		X	
Caranx lugubris	X		X	
Dermatolepis inermis	X			
Ginglymostoma cirratum	X		X	X
Gymnothorax funebris		X		X
Gymnothorax miliaris	X			
Gymnothorax moringa	X	X	X	X
Lutjanus analis			X	X
Lutjanus apodus		X	X	
Lutjanus cyanopterus	X		X	
Lutjanus griseus	X	X	X	
Lutjanus jocu	X		X	
Lutjanus synagris		X	X	
Mycteroperca bonaci	X		X	
Mycteroperca interstitialis	X		X	
Mycteroperca phenax	X			
Mycteroperca tigris	X		X	
Mycteroperca venenosa	X			
Scomberomorus regalis			X	
Sphyraena barracuda	X	X	X	X
Synodus intermedius	X	X	X	X
Synodus saurus	X			
Trachinotus falcatus			X	

The ratio of piscivore (all piscivores) to herbivore density (Figure 4.86) was significantly greater at FGBNMS ($p<0.0001$) than all Caribbean locations. Values approaching one are representative of equal density while values approaching zero typify communities dominated by herbivores. FGBNMS piscivore/herbivore ratio (0.52) was nearly three times higher than observed at St. John (0.19), over five times higher than St. Croix (0.13), and 8.5 times greater than Puerto Rico (0.08). This comparison demonstrates the strong piscivore community at FGBNMS.

Biomass at FGBNMS was dominated by species from Serranidae, Lutjanidae and Carangidae families. *Mycteroperca* spp. accounted for nearly half of the total apex predator biomass at FGBNMS. *Mycteroperca* spp. were absent in Puerto Rico and St. Croix but present, in low abundance, at St. John (Figure 4.87). Large *Mycteroperca* spp. (>35 cm) density was common throughout the FGBNMS averaging 1.3/100 m^2. Lutjanids were common at three of the four locations (Table 4.13), although species composition was variable. Lutjanids greater than 30 cm (*L. griseus* and *L. jocu*) were considerably more common at FGBNMS than the Caribbean locations (Figure 4.88). Lutjanids were smaller at the Caribbean

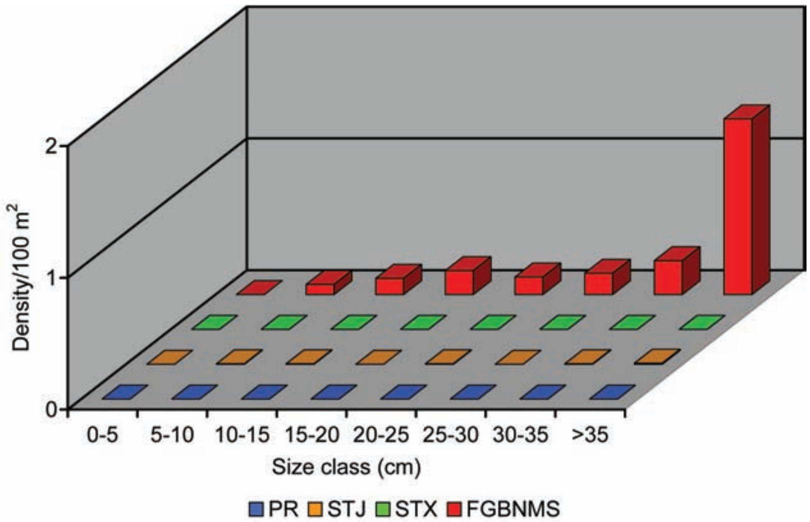


Figure 4.87. Size frequency of Mycteroperca *spp from FGBNMS, Puerto Rico (PR), St. John (STJ) and St. Croix (STX).*

locations and species composition was comprised of smaller-bodied species, such as *L. apodus* and the invertivore *O. chrysurus*. Carangidae species were abundant at both FGBNMS and St. John; however, size frequency, and thus biomass, were considerably different (Figure 4.89). In St. John, mean size for Carangidae species is approximately 14 cm, while at FGBNMS mean size is over 21 cm. Large fish, such as *Cx. lugubris* and *Cx. latus*, exhibited mean sizes of 37 and 51 cm, respectively, at FGBNMS. Only a few jack species were observed in Puerto Rico and St. Croix and these were dominated by the smaller sized *Cg. ruber*.

Herbivores comprised approximately 11% of the total abundance and 25% of total biomass at FGBNMS. Herbivore biomass was significantly greater at FGBNMS than all Caribbean locations (p<0.0001).

Parrotfish were the most common herbivores among all locations; however, species composition, density and biomass varied among the locations (Table 4.14). *Sp. aurofrenatum* was the most abundant parrotfish species at FGBNMS and was also highly abundant at the Caribbean locations. *Sp. viride* and *Sc. vetula* were also highly abundant at FGBNMS and exhibited high biomass. *Sc. iseri* was highly abundant at St. John and Puerto Rico while uncommon at FGBNMS. Overall, density was equivalent or greater for parrotfish less than 25 cm (Figure 4.90). Density for larger parrotfish was greater at FGBNMS than all Caribbean locations.

Fish density at St. John was dominated by invertivores and was significantly greater than FGBNMS, Puerto Rico and St. Croix (p<0.0001). Invertivore biomass was also significantly greater at St. John (p<0.0001) than other locations. Invertivore biomass and density were similar between the four locations with one exception. Grunts (*Haemulon* spp.) are common throughout the Caribbean but are poorly represented (two species, five individuals) at FGBNMS. Grunts are typically found in diverse ecosystems

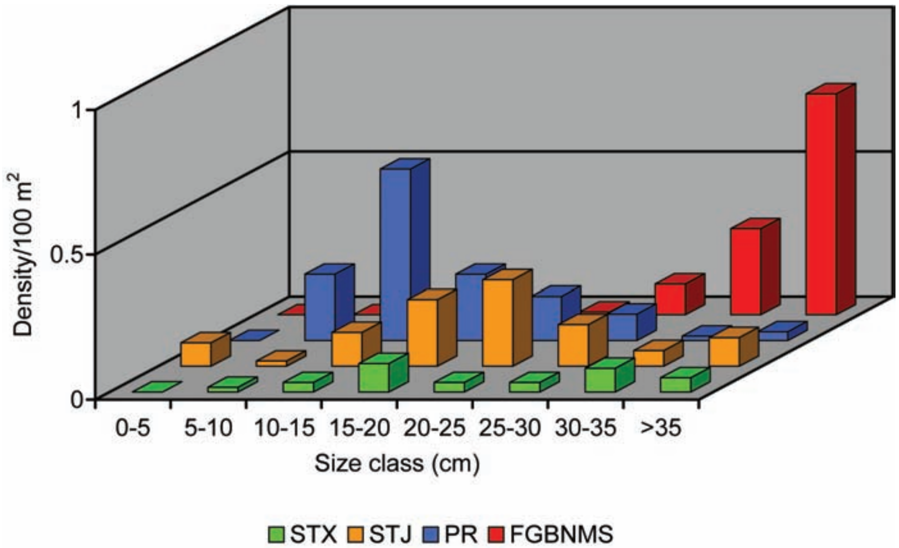


Figure 4.88. Size frequency of select Lutjanidae species from FGBNMS, Puerto Rico (PR), St. John (STJ) and St. Croix (STX).

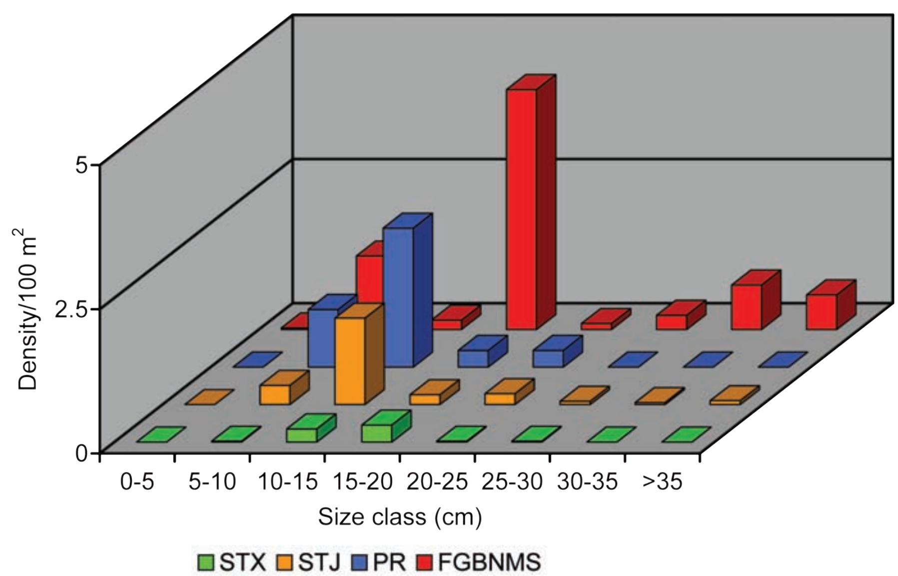


Figure 4.89. Size frequency of select Carangidae species from FGBNMS, Puerto Rico (PR), St. John (STJ) and St. Croix (STX).

Table 4.14. Parrotfish species density (D; # indiv./100 m^2) and biomass (B; kg/100 m^2) at FGBNMS and Caribbean locations.

Species	FGBNMS D	FGBNMS B	Puerto Rico D	Puerto Rico B	St. Croix D	St. Croix B	St. John D	St. John B
Cryptotomus roseus	0	0	0.69	0.001	0.74	0.005	0.59	0.002
Scarus coeruleus	0	0	0	0	0	0	0.005	0.01
Scarus iseri	0.30	0.003	7.26	0.20	0.74	0.04	6.00	0.16
Scarus taeniopterus	2.06	0.06	7.56	0.31	1.92	0.16	13.70	0.57
Scarus vetula	1.92	0.43	0	0	0.11	0.03	0.25	0.06
Sparisoma atomarium	1.86	0.002	0.64	<0.001	1.11	0.001	0.86	0.001
Sparisoma aurofrenatum	4.70	0.07	5.69	0.21	5.73	0.31	9.95	0.25
Sparisoma chrysopterum	0	0	0.03	0.01	0.03	0.003	0.08	0.02
Sparisoma radians	0	0	0.02	<0.001	0.21	0.002	0.09	<0.001
Sparisoma rubripinne	0	0	0.02	<0.001	0.09	0.02	0.01	0.005
Sparisoma viride	2.92	0.85	0.87	0.23	0.26	0.13	1.29	0.29

with large areal extents of sand, seagrass, and mangrove habitats and the lack of these habitats at FGBNMS may suppress the presence of this family.

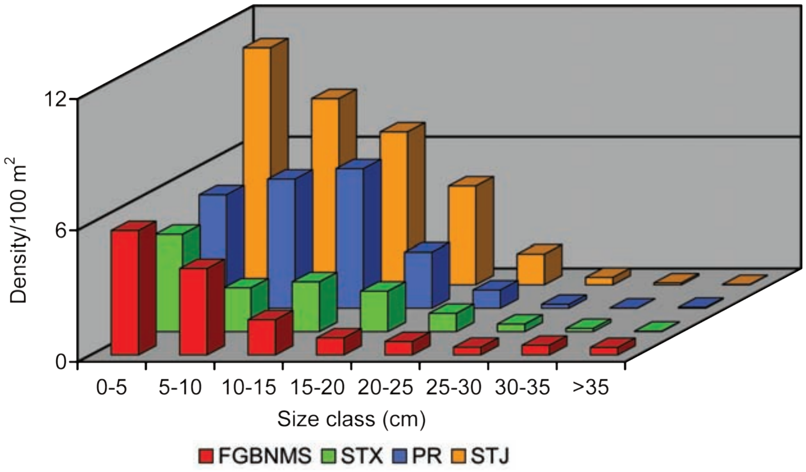


Figure 4.90. Size frequency of select Scaridae species from FGBNMS, Puerto Rico (PR), St. John (STJ) and St. Croix (STX).

Zooplanktivore density at FGBNMS was significantly greater than the Caribbean locations ($p<0.0001$) where three species (*C. multilineata*, *C. parrae*, *P. furcifer*) were numerically dominant. Biomass was also significantly greater at FGBNMS ($p<0.0001$) accounting for approximately 25% of the total biomass observed. Zooplanktivore biomass contributed less than 10% of total biomass at Caribbean locations.

Table 4.15. Trophic structure by density (# individuals/100 m^2) and biomass (kg/100 m^2) at East and West FGB, St. John, St. Croix and Puerto Rico.

	East FGB	West FGB	St. John	St. Croix	Puerto Rico
Density					
Herbivore	49.11	27.71	66.93	56.34	44.77
Invertivore	111.83	137.83	228.79	90.14	48.20
Zooplanktivore	166.67	130.17	66.38	4.95	19.89
Apex predator	4.98	8.54	1.24	0.42	0.67
Piscivore	41.21	5.00	7.83	6.09	3.54
Biomass					
Herbivore	6.01	5.58	2.09	1.60	1.63
Invertivore	2.49	1.84	3.16	2.32	1.11
Zooplanktivore	6.97	22.52	0.80	0.05	0.20
Apex predator	5.60	19.07	1.77	0.31	0.59
Piscivore	0.44	0.28	0.67	0.80	0.09

This section examines community structure among locations with geomorphic and habitat component differences; however, the focus was to compare fish community structure with differing levels of anthropogenic factors. As such, fish trophic structure at FGBNMS is considerably different from coral reef ecosystems in the USVI and Puerto Rico where piscivore and apex predator abundance and biomass were substantially reduced (Table 4.15). Trophic group ratios at FGBNMS closely resemble those with limited or no anthropogenic stressors (see Friedlander and DeMartini, 2002) and are skewed towards herbivores and planktivores in the USVI with greater anthropogenic stressors, most notably fishing pressure. While the ecosystems under comparison are vastly different between geomorphology, latitude and depth structure, the dominance of apex predator biomass at FGBNMS completely distinguishes the sanctuary from the Caribbean locations. Non-apex predator piscivores were generally less abundant at FGBNMS as these populations may be suppressed due to increased apex predator abundance. In contrast, these species may be more abundant in the Caribbean where apex predators are diminished. Additionally, the strong planktivore community at FGBNMS is significantly different both in density and biomass from that observed in the Caribbean and may be reflective of a more “oceanic” reef ecosystem. Herbivores are generally less abundant at FGBNMS than in the Caribbean, but attain larger size and greater biomass. More information throughout the Caribbean and other similar coral cap reefs are necessary to further examine these patterns.

4.5 SUMMARY AND RECOMMENDATIONS

- This work is a complement to earlier studies, which provided both a general overall characterization and quantitative information for a relatively spatially constrained portion of the SCC. This characterization provides the necessary quantitative density, size structure and habitat related information crucial to monitor change across the coral cap community.
- A total of 117 species from 37 families were observed during the course of the surveys including the first sighting of the Nassau grouper (*E. striatus*) and the second of the goliath grouper (*E. itajara*).
- Two of three species recently added to the FGBNMS species list were also recorded during the course of this study: sergeant major (*A. saxatilis*) and mardi gras wrasse (*H. burekae*).
- With the exception of species richness, which was significantly lower in the low relief habitat than the high relief at either bank, the other community level metrics, biomass, density and diversity were not significantly different among strata.
- The three most abundant families observed at the banks were Labridae (35%), Pomacentridae (30%), and Serranidae (14%). Biomass was dominated by species in the family Serranidae (42%) followed by Kyphosidae (15%), Lutjanidae (7%), Carangidae (6%) and Scaridae (6%). The invertivore and zooplanktivore trophic groupings dominate numerically while the piscivores (including apex predators) along with the zooplanktivores dominate by biomass.
- Three distinct fish assemblages were identified on the banks separated by depth: a deep water assemblage typically associated with the low relief habitat; a shallow water assemblage associated primarily with high relief habitat; and an assemblage near the interface of the two habitat types.
- The trophic structure observed on the SCC is comparable to many “pristine” coral reef ecosystems recently described in the Pacific. The dominance of large apex predators and other high trophic level species distinguish the community from coral ecosystems in the U.S. Caribbean.
- Monitoring these assemblages through time will provide information to strengthen the relationships and species memberships observed by CCMA and provide a valuable quantitative indicator for measuring ecosystem change.
- Since these species (groupers, snappers, etc.) are targeted by commercial and recreational fisheries, activities to quantify fishing effort and extraction should be implemented to better understand the level of fishing effort within the sanctuary and its impact to the resource.
- In addition to continued monitoring, emphasis should be placed on identifying potential sources for recruitment into the sanctuary. Currently, only inferences are made as to larval fish origin and limited information exists confirming spawning activity inside the sanctuary.
- Monitoring of the deeper (>30 m) portions of the sanctuary is recommended to understand the connectivity between the deep and shallow habitats. While different sampling methods will have to be implemented, this is an important data gap given that the coral caps comprise only 1% of the sanctuary.

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Appendix A: 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.

A PDF version of the datasheet utilized to collect the above data is available at: http://ccma.nos.noaa.gov/ecosystems/sanctuaries/fgb_nms.html.

Appendix B: Benthic Composition 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* and *Diploria* while the low relief habitat is characterized by the dominance of *Madracis mirabilis*. 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 centimeters) 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 location 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.

A PDF version of the datasheet utilized to collect the above data is available at: http://ccma.nos.noaa.gov/ecosystems/sanctuaries/fgb_nms.html.

Appendix C: Mapping Methods

Fish communities and associated benthic community surveys are greatly aided by benthic habitat maps. Many fish communities are strongly linked to particular benthic habitats and consequently habitats can be used in a stratified sampling design to improve sampling efficiency (Ault et al., 1999, 2005; Menza et al., 2007). Many types of benthic habitat maps exist, ranging from maps devised from a single benthic variable (e.g., depth) to complicated integrators of geomorphology, biological cover and geographical zonation. In this report we produced several different benthic habitat maps for sampling and analysis.

Map # 1

In 2006, a benthic habitat map devised from bathymetric slope was used to stratify sample selection. The sampling domain was also divided into two geographic locations (i.e., east bank, west bank) resulting in four exhaustive and mutually-exclusive strata which covered the shallow coral caps (Figure C1). Bathymetric slope was anticipated to be a covariate of several fish community metrics, because distinct species had been sighted along the steep perimeter of the coral caps (sanctuary staff, pers. comm.) and because it was thought to be a covariate of dominant coral type (e.g., plate coral, head coral, rubble). To divide the survey domain according to slope, areas of high (steep) and relatively low (flat or gently sloping) slopes were programmatically delineated from half-meter resolution bathymetric models (source: sanctuary staff) in a geographic information system (GIS). Models were first smoothed using a 5 x 5 m (10 x 10 cell) nearest neighborhood filter to reduce noise. Then the Slope function in ArcGIS' spatial analyst extension was used to produce a slope surface. This function defined the slope of each raster cell by the maximum rate of change between the cell and its neighbors. Lower slope values indicate flatter habitat and higher values indicate steeper habitat. Areas having a slope greater than 35° were considered steep, while the remainder of area was considered flat or gently sloping. A minimum mapping unit of 100 m^2 was used and thus distinct slope regions smaller than this area were incorporated into surrounding strata.

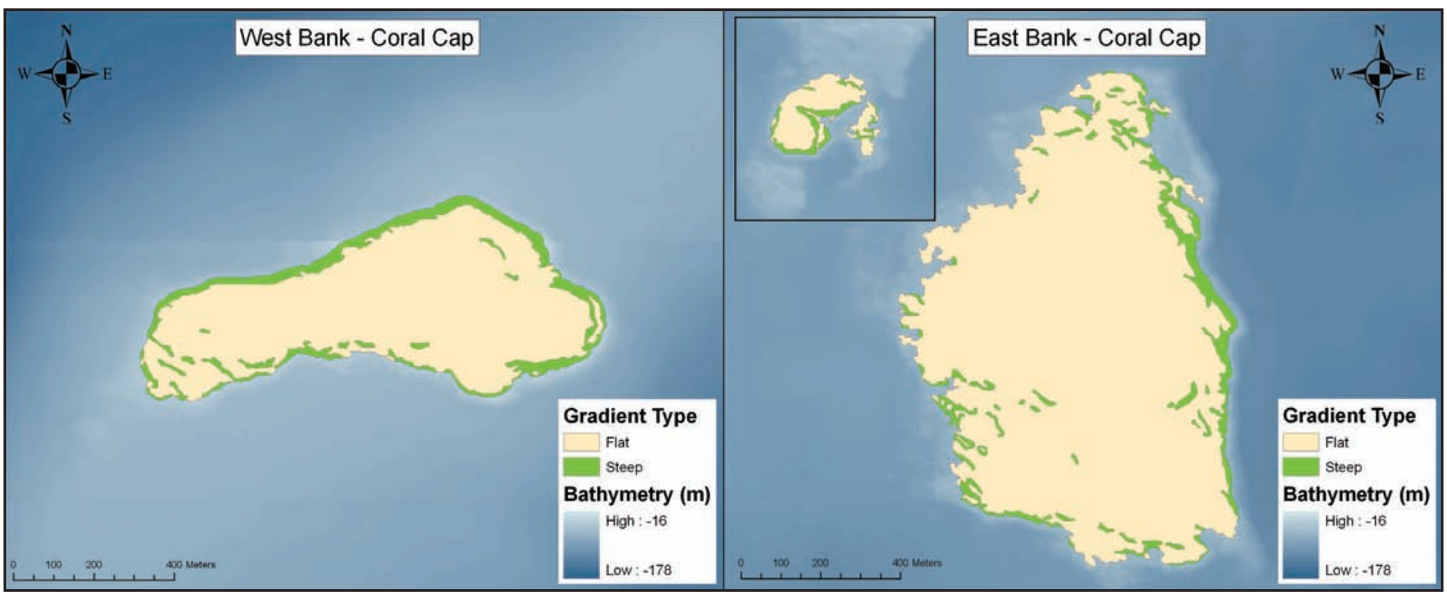

Figure C1. Benthic habitat maps of the West Bank and East Bank coral caps devised from gradient delineations.

Map # 2

After the field mission in 2006, a second benthic habitat map was developed, based on diver observations. Divers noted at least two distinct coral cover types were present on the coral caps and these cover types were each associated with different fish assemblages. The spatial patterns and bathymetric complexity of these cover types suggested that the classification method and spatial scale of analysis used to generate Map #1 were inadequate and required adjustment.

Two distinct cover types, low-relief coral and high-relief coral, were delineated for the new map. Differentiation among these cover types was accomplished using visual interpretation of a half-meter resolution slope surface. The slope surface was identical to the one used to develop the first benthic habitat map, but did not implement a smoothing filter prior to computation of slope, and was not assessed programmatically. Instead, cover was digitized using heads up display on a monitor.

Preliminary visual examination of the slope surface revealed three distinct spatial patterns. High-relief coral habitat was characterized by a relatively large variance of slope per unit area and low-relief coral habitat was characterized by a relatively low variance. A third habitat type consisted of almost no variance and represented areas of flat terrain, probably consisting of sand and little, if any, coral. Diver data revealed low-relief coral habitat consisted of a mixture of *Madracis* coral and rubble (Figure C2a), and high-relief coral habitat was a mixture of boulder and plate corals (Figure C2b).

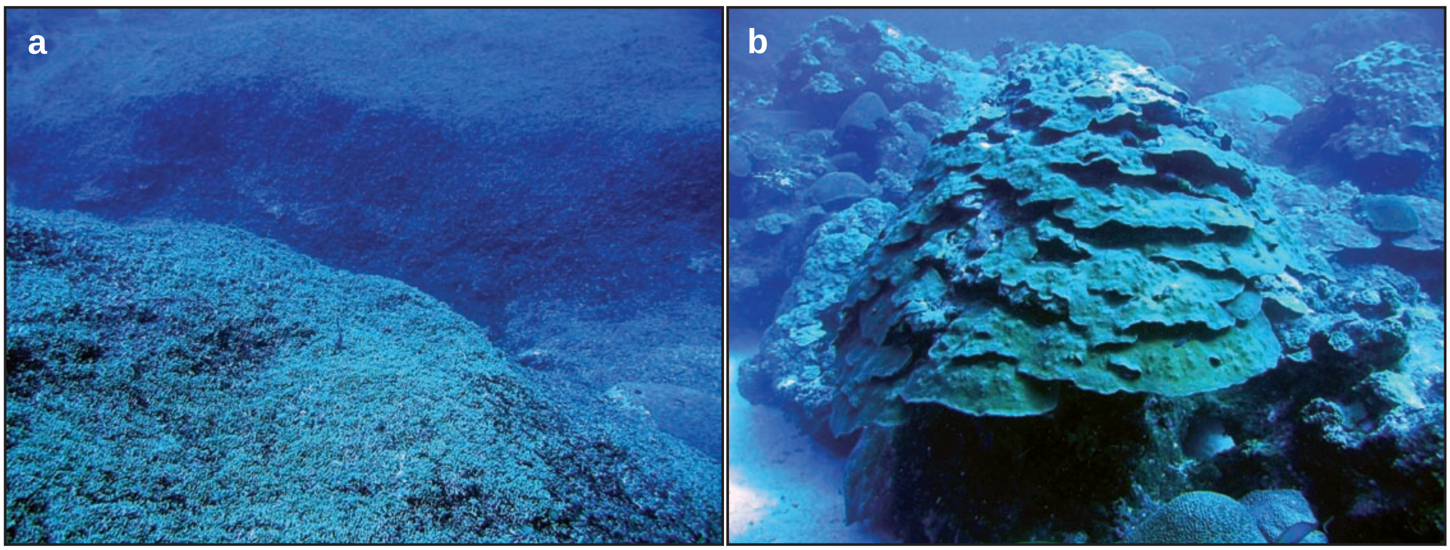

Figure C2. Underwater photos of (a) low-relief coral habitat and (b) high-relief coral habitat.

The benthic habitat map was produced by digitizing contiguous areas of coral cover types which were greater than 2,000 m^2 (i.e., the minimum mapping unit [MMU]) in a GIS (Figure C3). A value of 2000 m^2 was used for the MMU, because this was the spatial scale at which fish and benthic habitat data were collected (i.e., 25 m long transect at random bearing from point; area = π x 252, approximately 2,000 m^2). Areas of habitat with almost no variance in slope were all smaller than the MMU and consequently absorbed within surrounding habitat types.

Approximately 88% of the shallow coral cap area (consisting of EB and WB) was classified as high-relief coral habitat; the remaining 12% was classified as low-relief coral habitat (Figure C4). If the flat sand habitats were incorporated they would have made up less than 1% of the area. The majority of the low-relief coral habitat was at the edges of the corals caps and among deeper depths (mean depth 30 m WB and 31 m EB) compared with high-relief habitats (mean depth 25 m both banks). Strata area was also quite different with substantially smaller regions located in the low-relief strata (0.01 km^2 WB and 0.13 km^2 EB) compared to high-relief (0.34 km^2 WB and 0.62 km^2 EB). This new map was one of the layers used to develop the 2007 sampling design, but should not be confused with the sampling design itself.

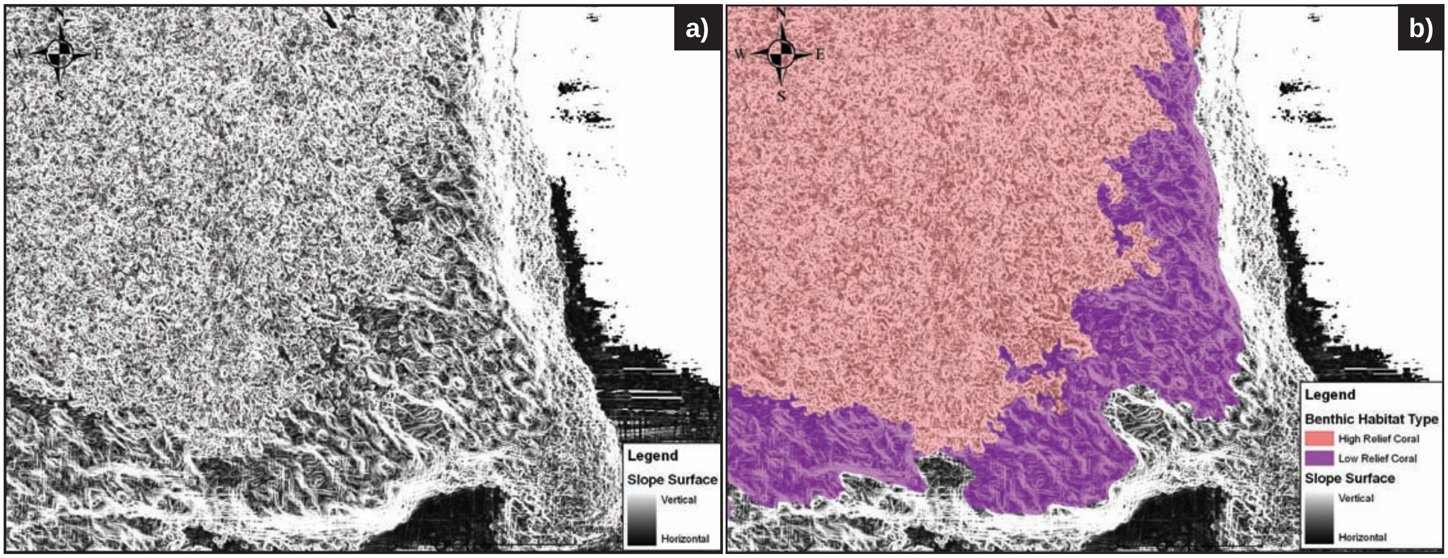

Figure C3. a) Process used to delineate benthic habitat types from slope surface; b) distinct spatial patterns of slope were used to map low-relief and high-relief coral habitats.

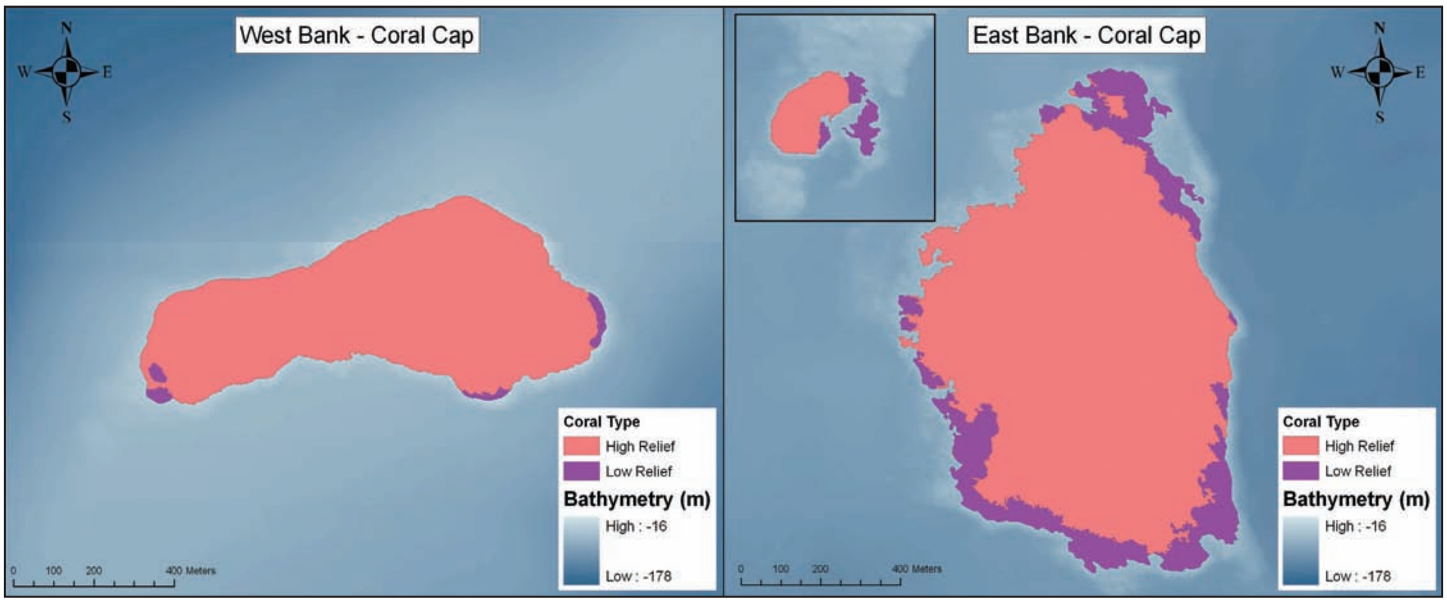

Figure C4. Benthic habitat maps of the West Bank and East Bank coral caps devised from coral habitat delineations.

Map # 3

In 2007, a stratified sampling design which used a regular network of points as sampling units was developed. Each point was the centroid of a square measuring 50 x 50 m and all of which exhaustively covered the coral caps. Several different environmental variables within each square were used to classify the corresponding sampling units, including coral cover type.

To integrate the coral cover type map (Map # 2) into the sampling design, each sampling unit was designated as either high- or low-relief coral habitat (Figure C5; the corresponding squares are used to visualize sampling unit classifications). In essence, the characterized sampling units became another version of the benthic habitat map defined by coral cover. Sampling units were classified by the cover type with the most area within a 25 m radius circle centered on the sampling unit (area approximately 2,000 m^2). The circle was used, because fish and benthic measurements are taken inside the circle (25 m long transect at random bearing from sampling unit). Although this procedure blurs the limits of coral cover types, it also allows sampling units to be updated easily. Updates may be needed if coral cover designations were initially interpreted incorrectly, coral cover type changes over time or new types are needed.

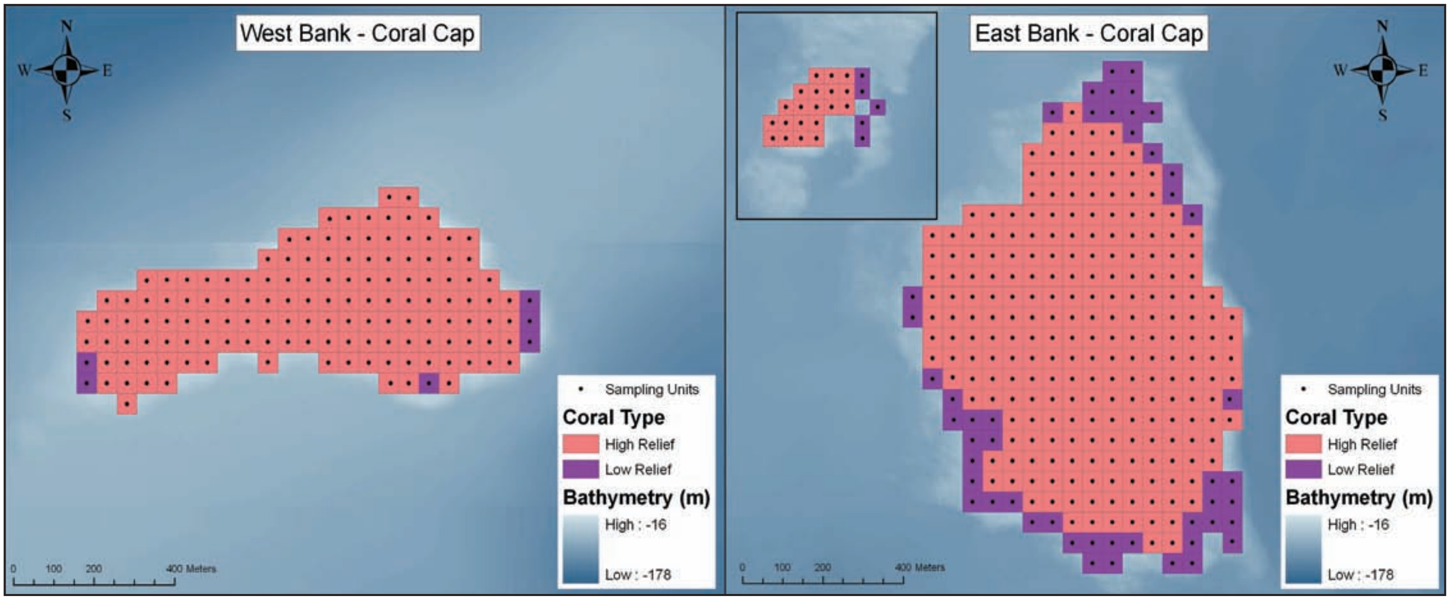

Figure C5. Benthic habitat maps of the West Bank and East Bank coral caps devised from coral habitat delineations within sampling units.

Accuracy of the benthic habitat map was assessed by *in situ* data collection during the 2007 sampling mission. Divers were asked to estimate the percentage of low- and high-relief coral habitat within the 25 m radius circle centered on the sampling unit. Habitat designations were made based on the majority of area within the circle. The designation was made independent of the stratification scheme and of subsequent measurements used in the characterization of fish and benthic habitat.

It was anticipated that over 70 sites would be used to assess the accuracy of the map, but severe weather prematurely canceled the mission and resulted in data collection at only 32 sites. All surveyed sites were on the southern portion of the east bank coral cap. The spatial bias among data means the reference data are not representative of the entire mapped area (east and west banks) and must be interpreted with this in mind. The incorporation of additional data from the northern portion of the east bank and west bank may alter map accuracy.

Overall accuracy of the cover type map was 94% (Table C1; kappa=0.73). Only two errors were identified, one for each cover type. These errors were adjacent to each other in a transition zone and included both cover types at each site (Figure C6). The most plausible explanations for these errors are mistakes in processing reference data for map production (visual interpretation of slope surface) positional error or diver interpretation error.

The benthic habitat map should be reassessed whenever possible. As mentioned previously, the use of a gridded map allows simple updates when new data become available or management objectives change.

Table C1. Error matrix for accuracy assessment.

	Predicted			
Reference	Habitat Type	Low-Relief	High-Relief	User's Accuracy
	Low-Relief	7	1	88%
	High-Relief	1	23	96%
	Producer's Accuracy	88%	96%	

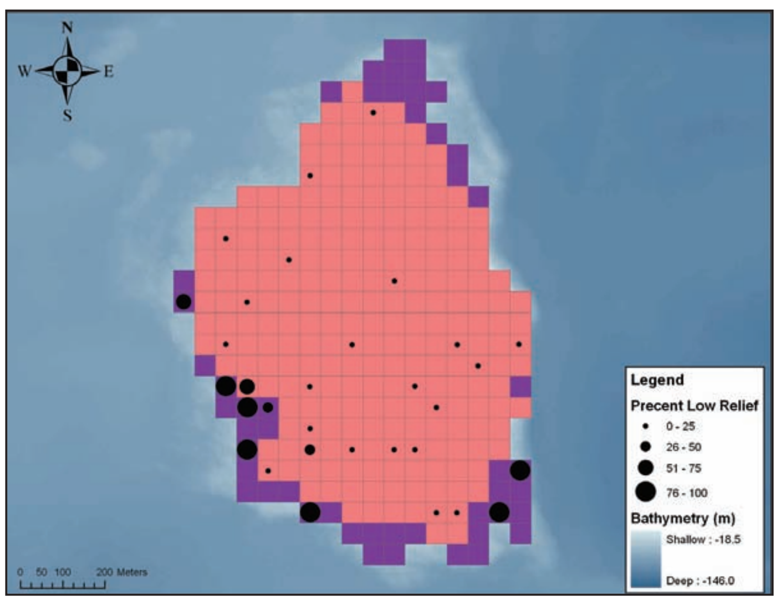

Figure C6. Accuracy assessment of coral cover type map. Graduated symbols represent percent low-relief coral cover recorded by scuba divers within a 25 m radius circle surrounding sampling units.

Map # 4

Three environmental variables were compiled to produce the strata used in the 2007 field mission (StRS; see Section 2.6.1 for details). The map of strata is a benthic habitat map which integrates coral cover type, geographic location and depth (Figure C7). The arrangement of sampling units and the method for coral cover type designation were the same as defined for Map # 3. Sampling units were differentiated according to depth by the maximum depth within a 25 m radius circle centered on the sampling unit. Units with a maximum depth greater than 31 m were considered deep, whereas those units less than 31 m were shallow. Groupings generated by the new environmental variables were not assessed for accuracy because they were not based on interpretation, but rather bathymetric surfaces having assumed negligible positional error.

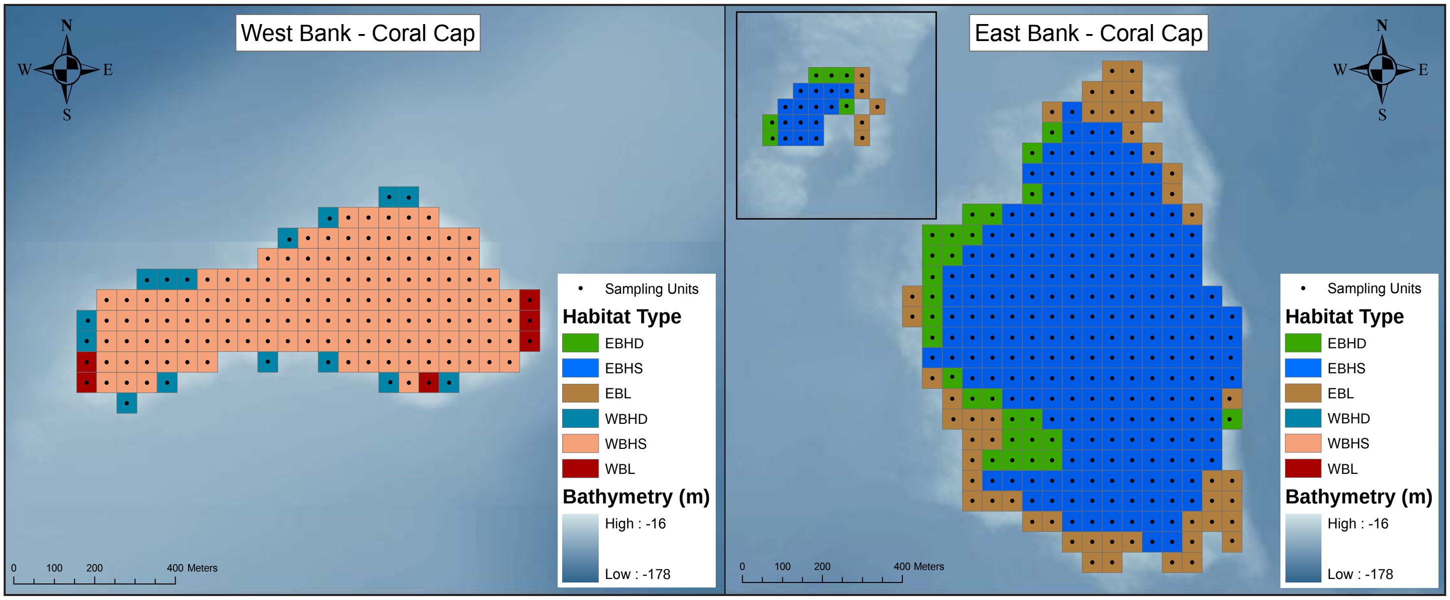

Figure C7. Benthic habitat maps of the West Bank and East Bank coral caps. Maps shows sampling unit stratum designations based on coral habitat type, geographic location and depth. Sampling units are points. Surrounding squares are used to visualize sampling unit classifications.

Appendix D: Sample Selection

Grid_ID	Strata_ID	Latitude	Longitude	Selection Probability	Sampling Weight
E684	EBHRD	27.90861	-93.60387	0.181818	5.5
E686	EBHRD	27.90951	-93.60388	0.181818	5.5
E858	EBHRD	27.90591	-93.60233	0.181818	5.5
E920	EBHRD	27.90681	-93.60183	0.181818	5.5
E978	EBHRD	27.90591	-93.60132	0.181818	5.5
E980	EBHRD	27.90681	-93.60132	0.181818	5.5
E1101	EBHRS	27.90727	-93.60031	0.118721	8.423077
E1107	EBHRS	27.90998	-93.60032	0.118721	8.423077
E1112	EBHRS	27.91223	-93.60034	0.118721	8.423077
E1158	EBHRS	27.90592	-93.59979	0.118721	8.423077
E1172	EBHRS	27.91224	-93.59983	0.118721	8.423077
E1215	EBHRS	27.90457	-93.59928	0.118721	8.423077
E1216	EBHRS	27.90502	-93.59928	0.118721	8.423077
E1225	EBHRS	27.90908	-93.59930	0.118721	8.423077
E1278	EBHRS	27.90592	-93.59878	0.118721	8.423077
E1280	EBHRS	27.90682	-93.59878	0.118721	8.423077
E1281	EBHRS	27.90728	-93.59879	0.118721	8.423077
E1282	EBHRS	27.90773	-93.59879	0.118721	8.423077
E1289	EBHRS	27.91089	-93.59881	0.118721	8.423077
E1292	EBHRS	27.91224	-93.59881	0.118721	8.423077
E1338	EBHRS	27.90592	-93.59827	0.118721	8.423077
E1339	EBHRS	27.90638	-93.59827	0.118721	8.423077
E1343	EBHRS	27.90818	-93.59828	0.118721	8.423077
E1404	EBHRS	27.90863	-93.59778	0.118721	8.423077
E1409	EBHRS	27.91089	-93.59779	0.118721	8.423077
E1410	EBHRS	27.91134	-93.59779	0.118721	8.423077
E1469	EBHRS	27.91089	-93.59728	0.118721	8.423077
E1520	EBHRS	27.90683	-93.59675	0.118721	8.423077
E1522	EBHRS	27.90774	-93.59676	0.118721	8.423077
E804	EBHRS	27.90861	-93.60286	0.118721	8.423077
E976	EBHRS	27.90501	-93.60131	0.118721	8.423077
E990	EBHRS	27.91133	-93.60135	0.118721	8.423077
E1067	EBLR	27.91900	-93.60088	0.185185	5.4
E1154	EBLR	27.90411	-93.59978	0.185185	5.4
E1297	EBLR	27.91450	-93.59882	0.185185	5.4

Grid_ID	Strata_ID	Latitude	Longitude	Selection Probability	Sampling Weight
E1575	EBLR	27.90458	-93.59623	0.185185	5.4
E682	EBLR	27.90770	-93.60387	0.185185	5.4
E741	EBLR	27.90726	-93.60336	0.185185	5.4
E797	EBLR	27.90545	-93.60284	0.185185	5.4
E799	EBLR	27.90636	-93.60284	0.185185	5.4
E859	EBLR	27.90636	-93.60234	0.185185	5.4
E916	EBLR	27.90501	-93.60182	0.185185	5.4
W426	WBHRD	27.87227	-93.82207	0.4	2.5
W492	WBHRD	27.87499	-93.82057	0.4	2.5
W568	WBHRD	27.87321	-93.81853	0.4	2.5
W594	WBHRD	27.87588	-93.81800	0.4	2.5
W635	WBHRD	27.87634	-93.81699	0.4	2.5
W716	WBHRD	27.87682	-93.81500	0.4	2.5
W407	WBHRS	27.87272	-93.82258	0.139344	7.176471
W427	WBHRS	27.87272	-93.82208	0.139344	7.176471
W450	WBHRS	27.87408	-93.82158	0.139344	7.176471
W529	WBHRS	27.87364	-93.81954	0.139344	7.176471
W551	WBHRS	27.87455	-93.81904	0.139344	7.176471
W570	WBHRS	27.87410	-93.81853	0.139344	7.176471
W609	WBHRS	27.87365	-93.81751	0.139344	7.176471
W652	WBHRS	27.87501	-93.81651	0.139344	7.176471
W674	WBHRS	27.87592	-93.81601	0.139344	7.176471
W675	WBHRS	27.87637	-93.81601	0.139344	7.176471
W708	WBHRS	27.87322	-93.81497	0.139344	7.176471
W710	WBHRS	27.87412	-93.81498	0.139344	7.176471
W711	WBHRS	27.87457	-93.81498	0.139344	7.176471
W734	WBHRS	27.87593	-93.81448	0.139344	7.176471
W752	WBHRS	27.87503	-93.81397	0.139344	7.176471
W773	WBHRS	27.87548	-93.81346	0.139344	7.176471
W788	WBHRS	27.87323	-93.81294	0.139344	7.176471
W387	WBLR	27.87272	-93.82309	0.833333	1.2
W388	WBLR	27.87317	-93.82310	0.833333	1.2
W829	WBLR	27.87369	-93.81193	0.833333	1.2
W830	WBLR	27.87414	-93.81193	0.833333	1.2
W831	WBLR	27.87459	-93.81193	0.833333	1.2

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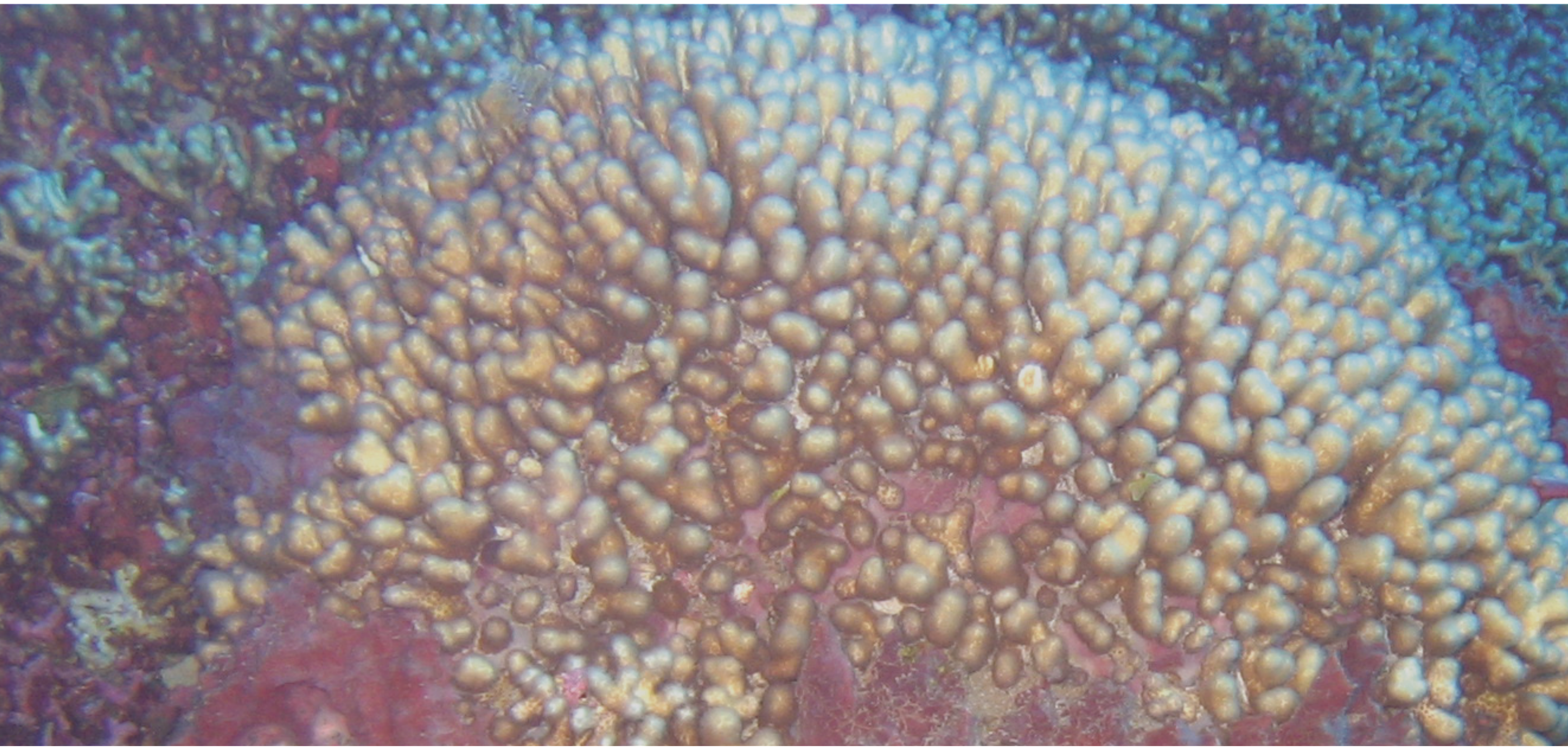

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